FISHERY RESEARCH

POTLATCH RIVER STEELHEAD MONITORING AND EVALUATION PROJECT

2019 AND 2020 BIENNIAL REPORT

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TABLE OF CONTENTS

Page

LIST OF TABLES

Page

LIST OF FIGURES

Page

ABBREVIATIONS AND ACRONYMS

BBC	Big Bear Creek
BPA	Bonneville Power Administration
EFPR	East Fork Potlatch River
EPA	Environmental Protection Agency
GRTS	Generalized Random Tessellation Stratification
ICBTRT	Interior Columbia Basin Technical Recovery Team
IDFG	Idaho Department of Fish and Game
IDL	Idaho Department of Lands
IPTDS	Instream PIT-tag Detection System
ITD	Idaho Department of Transportation
IMW	Intensively Monitored Watershed
LBC	Little Bear Creek
LGR	Lower Granite Dam
LWD	Large Woody Debris
LWHAP	Low Water Habitat Availability Protocol
NOAA	National Oceanic and Atmospheric Administration
NPPC	Northwest Power and Planning Council
PIT	Passive Integrated Transponder
PTAGIS	PIT Tag Information System
PTC	Potlatch Timber Corporation
UILT	Upper Incipient Lethal Temperature
USFS	United States Forest Service
WFLBC	West Fork Little Bear Creek
WFPR	West Fork Potlatch River

FOREWORD

PROJECT OVERVIEW

The Potlatch River basin supports the largest spawning area of wild steelhead (*Oncorhynchus mykiss*) in the Lower Mainstem Clearwater River steelhead population (ICBTRT 2003; Bowersox et al. 2009) and habitat restoration efforts are underway to enhance the production and productivity of wild steelhead within the basin. The Potlatch River Steelhead Monitoring and Evaluation project (hereafter the project) has been designed to measure the success of restoration efforts and is the sole habitat restoration effectiveness monitoring program within the Potlatch River basin. Project data has documented fish response to habitat restoration (Uthe et al. 2017) and is used to prioritize future habitat restoration efforts in the basin (Resource Planning Unlimited 2007; Potlatch Implementation Group 2019). The project was initiated in 2005 with Pacific Coastal Salmon Recovery Funds (PCSRF) and was integrated into the Intensively Monitored Watershed (IMW) program in 2007. In concert with other projects funded by these two programs, the project also contributes information to guide habitat restoration of anadromous salmonids elsewhere in Idaho and across the Pacific Northwest (Bennett et al. 2016; Uthe et al. 2017; Griswold and Phillips 2018; Hillman et al. 2019).

Snake River steelhead were classified as threatened under the Endangered Species Act (ESA) in 1997. Within the Snake River steelhead distinct population segment (DPS), there are six major population groups (MPGs): Lower Snake River, Grande Ronde River, Imnaha River, Clearwater River, Salmon River, and Hells Canyon Tributaries (ICBTRT 2003; NOAA 2017). The Clearwater River MPG supports six independent populations: Lower Mainstem Clearwater, North Fork Clearwater (extirpated), Lolo Creek, Lochsa River, Selway River, and South Fork Clearwater (ICBTRT 2003). The Lower Mainstem Clearwater River steelhead population, which contains the Potlatch River basin, is genetically distinct from other wild Clearwater River steelhead groups (Nielsen et al. 2009; Ackerman et al. 2016). Furthermore, the Lower Mainstem Clearwater River steelhead population comprises the only “large” independent population in the Clearwater River MPG (ICBTRT 2003) and must achieve viability in order for the Clearwater MPG and the Snake River DPS to become viable (NOAA 2017).

The Potlatch River basin is comprised of two distinct areas with notable differences in stream morphology, hydrology, and land use (Johnson 1985; Bowersox and Brindza 2006). In this report, we use the terms lower Potlatch River watershed and upper Potlatch River watershed to characterize each area. The lower Potlatch River watershed is defined as the drainage area downstream of and including Boulder Creek (Figure 1) and is characterized by steep basaltic canyons rimmed by rolling cropland. The predominant stream type in the lower watershed is a canyon stream with relatively high gradient, large substrate size, riffle/pocket water habitat types, and a flashy hydrograph (Bowersox and Brindza 2006). The majority of land in the lower watershed is privately owned and used primarily for agriculture production. In contrast, the upper Potlatch River watershed encompasses the drainage area upstream of Boulder Creek (Figure 1) and is characterized by timbered hills and meadow terrain. The predominant stream type in the upper watershed is a forestland stream with relatively low gradient, neighboring meadow complexes, small substrate composition, and cooler water temperatures (Bowersox and Brindza 2006). The majority of land in the upper watershed is public with large tracts of private timber lands used for timber production.

Land use practices, primarily agriculture and timber harvest, have significantly altered the aquatic habitat and hydrograph in the Potlatch River basin causing limiting factors to differ between the lower and upper watersheds. Primary limiting factors in the lower watershed are low

summer base flows and fish passage barriers (Johnson 1985; Bowersox and Brindza 2006). The Potlatch River basin receives the bulk (95%) of its annual precipitation from December to June (USDA SCS 1994). Thus, there is a natural pattern of high flow periods in the late winter/early spring followed by decreasing flows through the summer. However, conversion of timbered and meadow terrain into cropland in uplands and headwaters of lower watershed tributaries has resulted in higher peak springtime flow and reduced summer base flow. Base flow conditions are significantly limited with most tributaries experiencing flows <0.5 cfs and stream reach de-watering during the summer (Banks and Bowersox 2015; Uthe et al. 2017). Fish passage barriers are the other major factor limiting steelhead rearing habitat in the lower watershed. Barriers exist on nearly every major tributary, most of which are road culverts upstream of canyon reaches.

The primary limiting factor in the upper watershed is a lack of instream complexity resulting in poor juvenile steelhead summer and winter rearing conditions (Johnson 1985; Schriever and Nelson 1999; Bowersox and Brindza 2006). Logging began in the upper Potlatch River watershed in the early 1900s and infrastructure, including rail lines and roads, were built directly in stream channels or floodplains. As a result, streams were often straightened or relocated and riparian vegetation and instream woody debris were removed. Presently, streams in the upper watershed lack large woody debris (LWD) and other complex habitats, and riparian communities are not yet mature enough to actively recruit materials into streams.

Habitat restoration in the basin is guided by the Potlatch River Watershed Management Plan (Resource Planning Unlimited 2007; Potlatch Implementation Group 2019) and is a priority within Idaho Department of Fish and Game (IDFG) Fisheries Management (2019-2024) and Annual Strategic Plans (FY 2020-2023) (IDFG 2019, 2020). The Potlatch River Technical Advisory Group assisted the Latah County Soil and Water Conservation District (LCSWD) in developing the plan using fish, habitat, and water quality information obtained by local, state, and federal investigations in the basin. A prioritization of limiting factors and restoration strategies for key tributaries in the basin were incorporated into the plan.

Restoration strategies have been designed to address key limiting factors unique to each watershed. The primary restoration strategies in the lower Potlatch River watershed are to expand juvenile steelhead rearing habitat by removing barriers and increasing base-flow conditions through summer stream flow supplementation and meadow restoration. The primary restoration strategies in the upper watershed are to increase instream habitat complexity and riparian function by installing log structures, planting and protecting riparian areas, and restoring floodplain access.

PROJECT DESIGN AND OBJECTIVES

The overarching goal of the project is to evaluate fish and habitat responses to habitat restoration in the Potlatch River basin. The project is designed to assess responses in steelhead production and productivity at multiple scales: 1) broad-scale monitoring to document steelhead response within two index watersheds where restoration treatments are being conducted, Big Bear Creek (BBC) and the East Fork Potlatch River (EFPR), each with different limiting factors; 2) a finer-scale effort to assess fish and habitat responses to restoration projects at the tributary level; and 3) reach-scale monitoring to assess whether individual projects produced the intended outcome. The project design allows managers to better understand the relationship between a habitat action and fish response (Bennett et al. 2016) and how localized responses to restoration propagate up to a higher, management-scale level. To implement this design, specific monitoring objectives are:

1. Assess steelhead/resident *O. mykiss* production and productivity within two index watersheds in the Potlatch River basin.
   a. Determine abundance of juvenile steelhead emigrants.
   b. Estimate adult steelhead escapement.
   c. Estimate freshwater productivity (juvenile recruits per spawner) for the index watersheds.

2. Monitor juvenile steelhead density, survival, and growth in habitat treatment areas relative to control areas with no habitat treatments within upper and lower Potlatch River watersheds.

3. Monitor change in habitat variables associated with habitat restoration in control and treatment areas within upper and lower Potlatch River watersheds.

PROJECT TIMELINE

Effectiveness monitoring and habitat restoration implementation are ongoing in the index watersheds (Figures 2 and 3). The first juvenile steelhead surveys were conducted in the Potlatch River during 1995/1996 and 2003/2004, and intensive monitoring in the index watersheds began in 2005 for BBC and in 2008 for the EFPR. The first habitat treatments that directly addressed the primary limiting factors began in 2013 for BBC and in 2009 for the EFPR. In BBC, 10 barriers have been removed or modified, which has increased access to an additional 18 km of spawning and rearing habitat. This includes the Big Meadow Creek culvert modification project (see Chapter 3), which could have significant benefits to juvenile production in the BBC watershed (Uthe et al. 2017). In the EFPR watershed, approximately 8.4 km of stream has been treated and >190 LWD structures have been installed to improve habitat complexity. We anticipate habitat treatment goals will be achieved by 2028 for BBC and by 2029 for the EFPR. Restoration efforts are expected to result in a detectable biological response in juvenile steelhead production within five years (i.e., juvenile distribution, density, survival, growth) and productivity within ten years (i.e., smolt per female productivity) after the treatment goals are achieved in each watershed (Potlatch Implementation Group 2019; Uthe et al. 2017).

REPORT STRUCTURE

Project reporting has evolved over time. Previous years' project data was reported annually from 2006-2012 (Bowersox 2008; Bowersox et al. 2007, 2009, 2011, 2012; Banks and Bowersox 2015). Starting in 2016, Potlatch River steelhead production, diversity, and productivity data was included in statewide annual adult steelhead (Stark et al. 2016; Dobos et al. 2017, 2019, 2020; Knoth et al. 2018; Smith et al. 2021) and anadromous emigrant monitoring (Apperson et al. 2016, 2017; Belnap et al. 2018; Poole et al. 2019; Feeken et al. 2020, McClure et al. 2021) reports. Currently, Potlatch River specific data are reported in a biennial format (Knoth et al. 2021) to focus on trends and annual variation across the Potlatch River basin and to help focus analysis to inform management decisions regarding restoration efforts and monitoring program. In addition, the current format provides an outlet for reporting supplemental data collected by the project.

This report is in three chapters. Chapter 1 presents steelhead abundance and productivity data collected in index watersheds, and Chapter 2 presents data on habitat conditions and juvenile steelhead production in treatment and control tributaries. Spawn years 2019 and 2020 were emphasized in some sections since they were not previously compared to past data in the

current format. Lastly, Chapter 3 presents data on steelhead response to a specific restoration treatment, the Big Meadow Creek culvert enhancement project.

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FIGURES

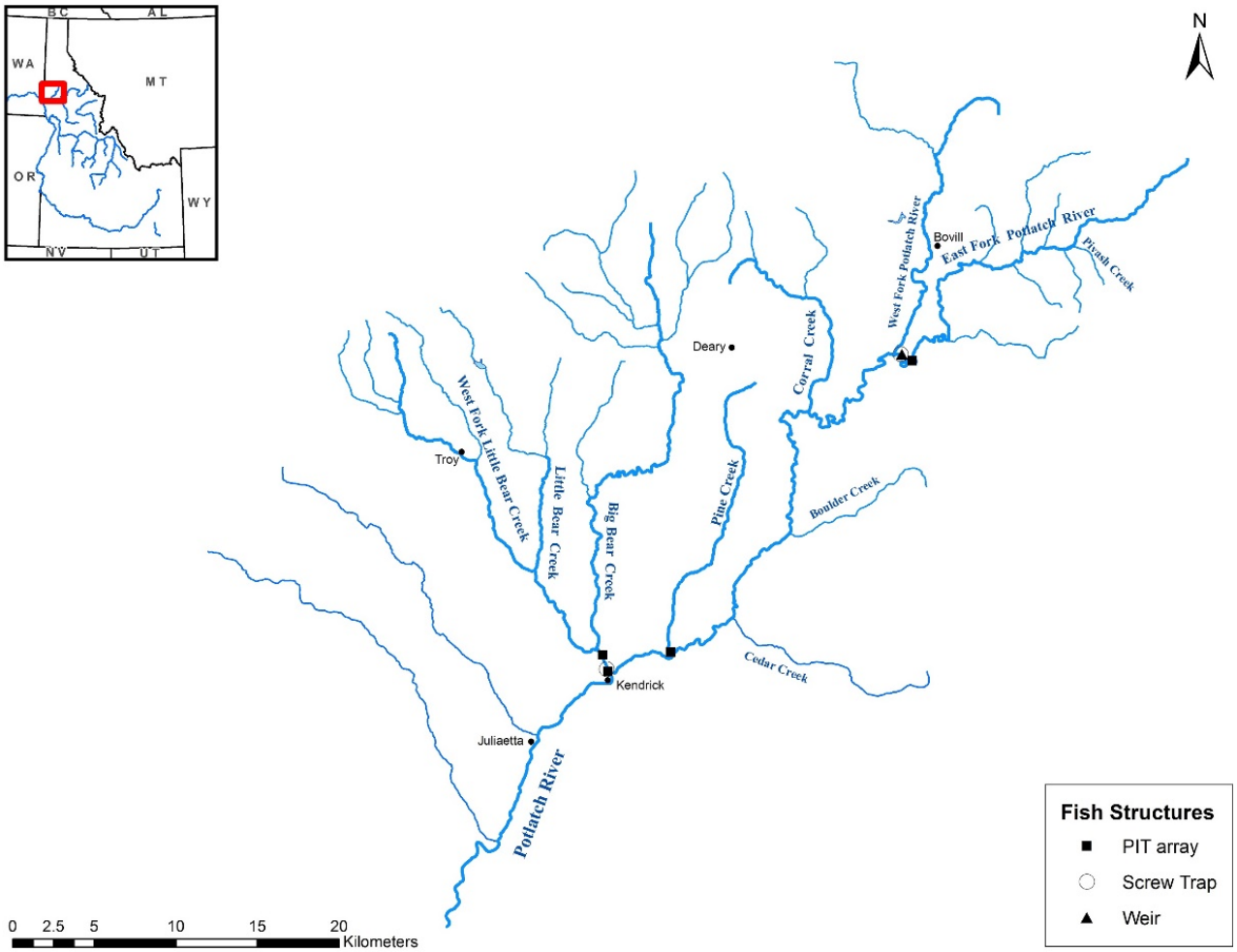

Figure 1. Key features and monitoring infrastructure in the Potlatch River basin in northern Idaho.

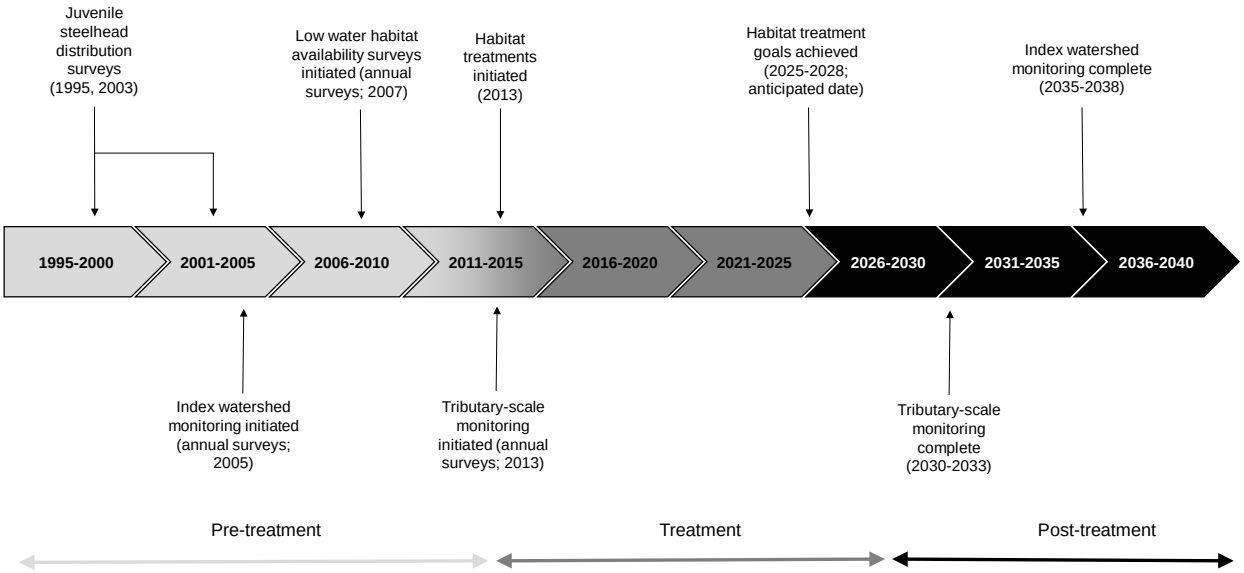

Figure 2. Timeline of project monitoring and restoration implementation in Big Bear Creek watershed in the Potlatch River Basin.

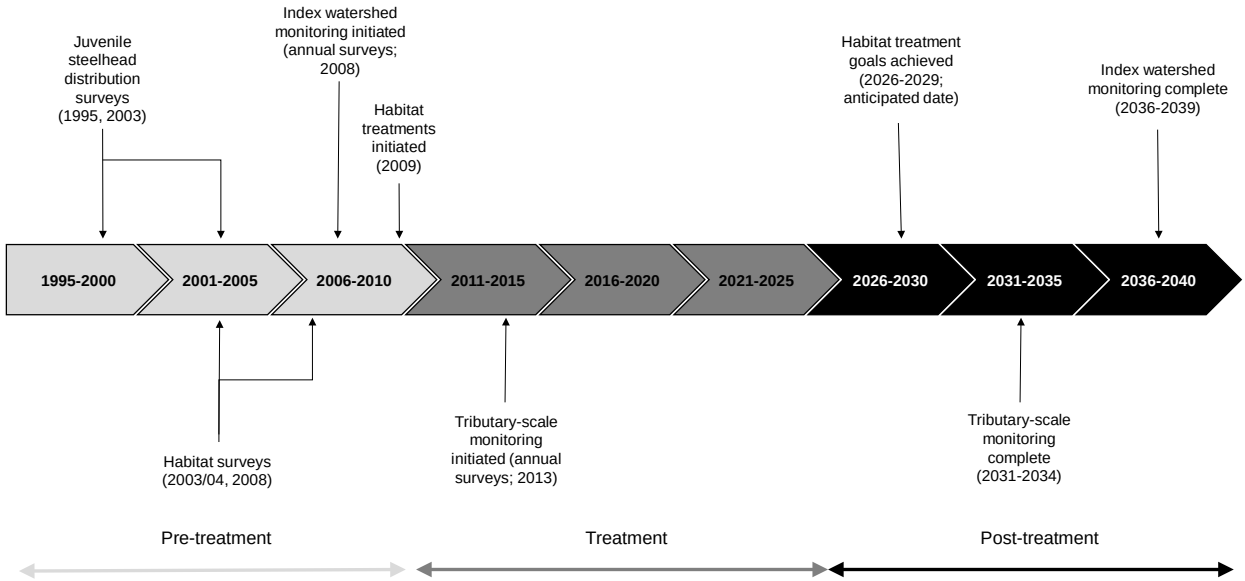

Figure 3. Timeline of project monitoring and restoration implementation in the East Fork Potlatch River watershed in the Potlatch River basin.

CHAPTER 1: ABUNDANCE AND PRODUCTIVITY OF STEELHEAD WITHIN POTLATCH RIVER, IDAHO INDEX WATERSHEDS

ABSTRACT

The Potlatch River Steelhead Monitoring and Evaluation project operates weirs, instream PIT-tag detection systems, and rotary screw traps to monitor the annual abundance and life history characteristics of wild steelhead (*Oncorhynchus mykiss*) in two index watersheds, Big Bear Creek (BBC) and the East Fork Potlatch River (EFPR). In 2019 and 2020, adult steelhead escapement was below long-term averages in both watersheds and ranged from 20-25 fish in BBC and 6-25 fish in EFPR. Big Bear Creek juvenile emigration estimates in 2019 (6,149 fish) and 2020 (8,621 fish) were slightly below the long-term average; whereas, EFPR emigration estimates in 2019 (3,100 fish) and 2020 (2,184 fish) were the lowest on record. Emigrant age structure and length-at-age estimates were relatively consistent across years in BBC, but have shifted to older and larger emigrants in the EFPR. Freshwater productivity estimates averaged 398 juvenile recruits per spawner in BBC and 362 juvenile recruits per spawner in the EFPR. Recruits per spawner decreased as female spawner abundance increased suggesting density-dependent mechanisms occurred, most notably in BBC. The observed shifts in EFPR emigrant age, growth, and survival indicate an initial watershed-scale response in steelhead within the upper Potlatch River watershed.

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INTRODUCTION

Restoring freshwater habitat conditions is a critical component of recovery efforts for Pacific salmon listed under the Endangered Species Act (ESA; Bennett et al. 2016; Griswold and Phillips 2018; Hillman et al. 2019). Habitat restoration efforts are being implemented to improve the production and productivity of wild steelhead (*Oncorhynchus mykiss*) in the Potlatch River basin (Resource Planning Unlimited 2007; Uthe et al. 2017). The Potlatch River Monitoring and Evaluation project (hereafter the project) is designed to assess responses in steelhead production and productivity to restoration activities at the watershed, tributary, and reach scales (Uthe et al. 2017). At the broadest scale, the project is designed to measure the benefits of habitat restoration (sum of all projects) for steelhead in two index watersheds, Big Bear Creek (BBC) and the East Fork Potlatch River (EFPR). These efforts provide the necessary data to assess the steelhead response to restoration actions at the watershed level, which is the scale at which management decisions are focused.

The objective of this chapter is to estimate key demographic metrics for the steelhead in the two index watersheds. We are currently in the treatment phase of habitat restoration in the index watersheds (Figures 2 and 3) and preliminary results presented here include pretreatment data (2005-2013 in BBC; 2008-2009 in EFPR) and partial data from the treatment phase (2014-2020 in BBC; 2010-2020 in EFPR). Ultimately, watershed-scale monitoring data will be assessed as a before-after comparison of steelhead production and productivity in each index watershed. The final assessment will be completed after habitat treatment goals are achieved in each watershed.

METHODS

Adult Steelhead Escapement and Diversity

Adult steelhead escapement into index watersheds was monitored using weirs and instream PIT-tag detection systems (IPTDS; Figure 1). Picket weirs were deployed on BBC and Little Bear Creek to monitor annual adult steelhead escapement into the BBC watershed from 2005-2012. Adult steelhead escapement into the BBC watershed was monitored using an IPTDS beginning in 2013. The area monitored by picket weirs and IPTDS was similar, thus the data are comparable across time. Adult steelhead escapement into the EFPR was monitored using a resistance-board weir since 2008. Weirs were installed as early as possible, typically mid-February (BBC) or March (EFPR), and operated until the kelt (post-spawn fish) outmigration was complete. Detailed information on methods, data analysis, and annual operations for weirs and IPTDS can be found in Potlatch River Steelhead Monitoring and Evaluation Reports (Knoth et al. 2021; additional reports cited in the Foreword) and Idaho Adult Steelhead Monitoring Annual Reports (Smith et al. 2021; additional reports cited in the Foreword). Results are preliminary and unless noted otherwise, we used graphical comparisons for inference.

Juvenile Steelhead Emigration, Diversity, and Survival

Juvenile emigrant production and diversity metrics (i.e., emigrant age composition and size-at-age) from the index watersheds were monitored using rotary screw traps. Emigration estimates have been generated since 2005 in BBC and 2008 in EFPR. Annual operations began as early as possible, typically late January - February (BBC) or March (EFPR), and continued until early June when low flows prevented rotary screw traps from operating. Operations resumed during the fall at both sites when sufficient flows and personnel allowed. Fall emigration could not

be estimated across all years in either BBC or the EFPR because low flow conditions prevented the rotary screw traps from operating or not enough juveniles were captured to generate an estimate (Uthe et al. 2017). Detailed information on methods, data analysis, and annual operations for BBC and EFPR rotary screw traps can be found in Potlatch River Steelhead Monitoring and Evaluation Reports (Knoth et al. 2021; additional reports cited in the Foreword) and Idaho Anadromous Emigrant Monitoring annual reports (McClure et al. 2021; additional reports cited in the Foreword).

We examined survival rates of PIT-tagged juvenile steelhead emigrants from each rotary screw trap to Lower Granite Dam (LGR). Estimating steelhead smolt survival is problematic because steelhead emigrate at different ages and not all fish captured at the rotary screw trap in a given year will emigrate to LGR in the same year (Feeken et al. 2020). We estimated apparent survival as a proxy for actual cohort survival. Apparent survival estimates do not account for delayed emigration and thus are biased low since some individuals will not emigrate until subsequent years. Apparent survival is provided for a migrating cohort not survival of a brood year cohort. We queried the Columbia Basin PIT Tag Information System (PTAGIS; www.ptagis.org) for detections of juvenile steelhead tagged at Potlatch River rotary screw traps. Juvenile detection sites at LGR, Little Goose, Lower Monumental, McNary, John Day, and Bonneville dams and the estuary towed PIT-tag array were selected to search for potential PIT-tagged smolts. Juvenile survival indices to LGR were estimated using PitPro 4.19 (Westhagen and Skalski 2009). The PitPro algorithm combines capture-recapture PIT-tag interrogation data from instream PIT-tag arrays and hydrosystem passage facilities into a Cormack-Jolly-Seber model to estimate survival and capture probabilities to tributary and hydrosystem PIT-tag arrays.

Productivity Estimates

Freshwater productivity estimates, measured as juvenile recruits at the rotary screw trap per female spawner estimated at the weirs or PIT-tag arrays, were computed for each index watershed. Annual abundances of female spawners were calculated by applying the observed sex ratio at the weir or PIT-tag array to the total adult escapement estimate. Juvenile age proportions based on juvenile scale samples were applied to annual emigration estimates at rotary screw traps to determine the total number of juvenile recruits by brood year (BY) for a given trapping year. Juveniles were summed across trapping years for each brood year to determine total juvenile recruits (e.g., BY 2011 females produced age-1 juveniles in 2012, age-2 juveniles in 2013, etc.). Total juvenile recruits for a particular BY was divided by number of female spawners estimated in each BY to estimate juvenile recruits per female spawner. Productivity estimates were examined in relation to female spawner abundance for indication of density-dependence in the two index watersheds.

RESULTS

Adult Steelhead Escapement and Diversity

Big Bear Creek

Adult steelhead escapement into BBC varied 40-fold across years, with a marked decline between 2016 and 2017 and low abundance persisting through 2020 (Figure 4). An estimated 20 adult steelhead returned to BBC in 2019 and 25 adult steelhead returned in 2020. Detection probability could not be calculated in 2019 and 2020 due to the low number of PIT tag detections at the BBC PIT-tag array; therefore, these are considered minimum estimates (Dobos et al. 2020;

Smith et al. 2021). The mean number of adult steelhead (2005-2018) returning to BBC was 150 fish (range = 21-317 fish).

Sex ratio and age composition of BBC adult steelhead were variable over time. The sex ratio of BBC adult steelhead was 75% male and 25% female in both 2019 and 2020 (Figure 5). In 2019, 75% of adults were 1-ocean fish (n = 3) and 25% were 2-ocean fish (n = 1), whereas in 2020, 50% of adults were 1-ocean fish (n = 4) and 50% were 2-ocean fish (n = 4). No 3-ocean fish or repeat spawners were observed in 2019 and 2020 (Figure 6). Mean sex ratio of BBC adult steelhead (2008-2018) was skewed towards females (58%, range = 35-84%). On average, 2-ocean fish comprised 58% of adults, followed by 1-ocean fish at 40%, and 3-ocean fish at 1% from 2008-2018. Single repeat spawners were observed intermittently and comprised between 0-11% of adults by year. Ocean age of adults ranged from one to three years and total ages ranged from three to seven years, with eight different freshwater-saltwater age class combinations (Table 1).

Origin of adult steelhead sampled in BBC showed minimal hatchery influence. No hatchery origin fish were detected at the BBC PIT-tag array in 2019 and one of the 14 adult steelhead detected at the PIT-tag array in 2020 was hatchery origin. Hatchery adult steelhead were observed in eight of 16 years at BBC. The mean number and raw proportion of hatchery origin adults captured at the weir or detected on the PIT-tag array (2005-2018) was 1.5 fish (range = 0-11) and 2.9% (range = 0-20%).

Adult steelhead migration timing at BBC varied across years (Figures 7 and 8). Prespawn and kelt timing were not calculated in 2019 and 2020 due to the low number of PIT-tag detections at the BBC PIT-tag array. The mean date of the first upstream spawner captured or detected (2006-2018) was February 19 (range = February 7-March 16). The mean date that 50% of run passed was March 24 (range = March 6-April 10) and the mean date of the final upstream spawner was April 21 (range = March 30-May 18). The mean date of the first downstream kelt captured or detected (2006-2018) was March 30 (range = February 28-April 22). The mean date that 50% of the kelt run passed the weir or PIT-tag array was April 21 (range = April 5-May 12) and the mean date of the final kelt arrival was May 12 (range = April 25-May 28).

East Fork Potlatch River

Adult escapement into the EFPR varied 13-fold across years and experienced a similar decline as BBC that started in2017 and persisted through 2020 (Figure 4). Six adult steelhead returned to the EFPR weir in 2019 and 25 (95% CI 21-36) adult steelhead returned in 2020. Abundance in 2019 was a minimum estimate and could not be expanded due to the small sample size of kelts to establish the mark rate at the weir (Dobos et al. 2020). The mean number of adult steelhead (2008-2018) returning to the EFPR was 76 fish (range = 11-140 fish).

Sex ratio and age composition of EFPR adult steelhead varied across time. The sex ratio of EFPR adult steelhead was 75% male and 25% female in 2019 and 62% male and 38% female in 2020 (Figure 5). In 2019, 60% of returning adults were 2-ocean fish (n = 5) and 40% were 1-ocean fish (n = 2), whereas in 2020, 70% were 1-ocean fish (n = 23) and 30% were 2-ocean fish (n = 7). No 3-ocean fish or repeat spawners were observed in 2019 and 2020 (Figure 6). Mean sex ratio of EFPR adult steelhead (2008-2018) was skewed towards females (57%, range = 25-76%). On average, 2-ocean fish comprised 50% of adults, followed by 1-ocean fish at 49%, and 3-ocean fish at <1% from 2008-2018. A single repeat spawner was observed in 2010. Ocean age of adults ranged from one to three years and total ages ranged from three to seven years, with

ten different freshwater-saltwater age class combinations (Table 1). No hatchery origin fish have been captured at the EFPR weir.

Adult steelhead migration timing at the EFPR varied across years (Figures 7 and 8). Prespawn and kelt timing in 2019 and 2020 fell within the range of previous estimates though timing was based on relatively low sample sizes, especially in 2019. The mean date of the first upstream spawner (2008-2018) was March 29 (range = March 7-April 18). The mean date that 50% of the run passed the weir was April 17 (range = March 31-April 25) and the mean date of the final upstream spawner was May 9 (range = April 23-May 25). The mean date of the first downstream kelt (2008-2018) was April 29 (range = April 3-May 31). The mean date that 50% of downstream kelts passed the weir was May 11 (range = April 24-June 16) and the mean date of the final kelt arrival was May 30 (range = May 4-June 29).

Juvenile Steelhead Emigration, Diversity, and Survival

Big Bear Creek

Spring emigration from the BBC watershed varied five-fold across years (Figure 9). Juvenile emigration was 6,149 fish (95% CI 5,485-7,168) in 2019 and 8,621 fish (95% CI 7,380-11,193) in 2020, both of which were below average. There were no fall estimates for BBC in 2019 and 2020. The mean number of juvenile emigrants from the spring season (2005-2018) was 9,556 fish (range = 3,837-22,649 fish). Fall emigration estimates were generated in 6 of 14 years and ranged from 91-2,032 fish (Figure 10). Fall emigration averaged 8.5% (range = 1.0-18.6%) of the following years' spring emigration estimate.

Emigrant age composition during the spring season at BBC was typically dominated by age-2 emigrants (Figure 11). Emigrant age composition in 2019 was 15.1% age-1 fish, 76.5% age-2 fish, and 8.3% age-3 fish and in 2020 was 53.4% age-1 fish, 39.3% age-2 fish and 7.3% age-3 fish. Mean emigrant age composition (2008-2018) during the spring was 35.0% (range = 8.8-70.7%) age-1 fish, 60.2% (range = 28.6-86.8%) age-2 fish, and 4.7% (range = 0.6-13.6%) age-3 fish. Mean age composition of fall emigrants was 8.9% (range = 0.0-25.0%) age-0 fish, 86.2% (range = 75.0-100.0%) age-1 fish, and 3.4% (range = 0.0-9.1%) age-2 fish based on four years (2008, 2009, 2010, and 2016) of data (Figure 12).

Emigrant length-at-age varied widely across years and 2019 and 2020 estimates fell within range of previous estimates (Figure 13). In 2019, mean length at age was 135.3 mm (SE = 4.9) for age-1 emigrants, 160.2 mm (SE = 1.6) for age-2 emigrants, and 187.5 mm (SE = 9.2) for age-3 emigrants, and in 2020 it was 142.1 mm (SE = 2.0) for age-1 emigrants, 173.4 mm (SE = 1.8) for age-2 emigrants, and 186.9 mm (SE = 3.9) for age-3 emigrants. Across years (2005-2018), mean FL of age-1 emigrants was 134.8 mm (range = 110.6-148.9 mm), age-2 emigrants was 171.5 mm (range = 160.2-178.6 mm), and age-3 emigrants was 187.8 mm (range = 170.1-210.6 mm).

Apparent survival peaked at three to four year intervals across years (Figure 14). Estimated survival for spring emigrants was 47.6% (SE = 5.2) in 2019 and 37.9% (SE = 4.0) in 2020, and both years fell within the range of previous estimates. Across years (2005-2018), mean apparent survival was 49.0% (range = 26.8-80.9%) and tended to peak in years with older emigrant age structure (i.e., 2011 and 2014; Figures 9 and 12).

East Fork Potlatch River

Spring emigration from the EFPR watershed varied four-fold across years and experienced a marked decline between 2018 and 2019 (Figure 9). Juvenile emigration was 3,100 fish (95% CI 2,138-5166) in 2019 and 2,184 fish (95% CI 1,092-6,552) in 2020, both of which were the lowest estimates on record. There were no fall estimates for EFPR in 2019 and 2020. The mean number of juvenile emigrants for the spring season (2008-2018) was 16,099 fish (range = 7,965-40,224 fish). Fall estimates were generated in 4 of 11 years and ranged from 1,296-1,866 fish (Figure 10). Fall estimates averaged 10.4% (range = 3.2-15.9%) of the following years' spring emigration estimate.

Emigrant age composition during the spring season at the EFPR was typically dominated by age-1 emigrants, though there was a shift towards older emigrants starting in 2014 (Figure 11). Emigrant age composition in 2019 was 45.9% age-1 fish, 44.3% age-2 fish, and 9.8% age-3 fish and in 2020 was 25.0% age-1 fish, 66.7% age-2 fish, and 8.3% age-3 fish. The highest proportions of age-2 and age-3 emigrants on record were observed in 2019 and 2020, though it was based on relatively small sample sizes (n = 61 fish in 2019 and n = 24 fish in 2020). Mean emigrant age composition (2008-2018) during the spring was 70.2% (range = 53.7-86.1%) age-1 fish, 27.4% (range = 12.1-42.7%) age-2 fish, and 2.4% (range = 0.0-6.6%) age-3 fish. Mean age composition of fall emigrants was 21.7% (range = 0.0-41.8%) age-0 fish, 66.2% (range = 50.9-100.0%) age-1 fish, 11.4% (range = 0.0-25.6%) age-2 fish, and 0.6% (range = 0.0-2.6%) age-3 fish based on four years (2008, 2010, 2012, and 2018) of data (Figure 12).

There has been an increase in the emigrant length-at-age in the EFPR across the dataset (Figure 13). In 2019, mean length at age was 97.8 mm (SE = 2.5) for age-1 emigrants, 147.9 mm (SE = 3.1) for age-2 emigrants, and 165.3 mm (SE = 4.8) for age-3 emigrants, and in 2020 it was 122.7 mm (SE = 6.8) for age-1 emigrants, 164.4 mm (SE = 2.0) for age-2 emigrants, and 183.0 mm (SE = 12.0) for age-3 emigrants. The 2020 estimates were the highest on record, though the estimates were based on small sample sizes (n = 24 fish aged in 2020).

Apparent survival for spring emigrants from the EFPR rotary screw trap to LGR varied annually, but increased in recent years (Figure 14). Estimated survival for spring emigrants was 23.2% (SE = 4.0) in 2019 and 33.3% (SE = 10.3) in 2020. Both years were the highest estimates on record, though they were based on relatively small sample sizes (n = 235 in 2019 and n = 78 in 2020). Across years (2008-2018), mean apparent survival was 10.9% (range = 5.0-17.0%).

Population Productivity

Big Bear Creek

Complete BY productivity estimates have been generated for 13 BYs for the BBC watershed. Estimates ranged from 48 juvenile recruits per spawner (BY 2010) to 487 juvenile recruits per spawner (BY 2017) and averaged 147 juvenile recruits per spawner across all complete BYs. Brood year 2017 productivity estimate is biased high because a minimum spawner estimate was used in place of an expanded adult escapement estimate. Productivity estimates for BBC displayed a strong density-dependent relationship (Figure 15).

East Fork Potlatch River

Complete BY productivity estimates have been generated for 10 BYs for the EFPR watershed. Estimates ranged from 127 juvenile recruits per spawner (BY 2015) to 967 juvenile

recruits per spawner (BY 2017) and averaged 456 juvenile recruits per spawner for all complete BYs. Brood year 2011 productivity estimate is biased high because a minimum spawner estimate was used in place of an expanded adult escapement estimate. Productivity estimates for EFPR also displayed a density-dependent relationship (Figure 15).

DISCUSSION

To date, we have completed 15 years of index watershed monitoring activities in BBC and 12 years in the EFPR. In general, adult steelhead escapement was higher in BBC relative to the EFPR across years, but both watersheds experienced sharp declines from 2017-2020. Hatchery-origin steelhead were observed relatively frequently in BBC, but on average the raw proportion of hatchery-origin adults was minimal (≤3.0%). No hatchery-origin fish were captured at the EFPR weir. There was a sharp decline in the EFPR emigrant abundance in 2019-2020, whereas, BBC emigrant abundance remained relatively stable. Emigrant size, age, and survival increased in the EFPR during recent years. Population productivity estimates (juvenile recruits per spawner) displayed a strong density-dependent relationship in BBC, but less so in the EFPR. The limited range of female spawners in the EFPR likely obscures the relationship. The index watershed monitoring framework appears robust, but modifications discussed below will help strengthen the project's ability to monitor watershed-scale responses in steelhead production and productivity to restoration actions.

Environmental conditions in both the marine and freshwater environments can influence the productivity of anadromous salmonid populations. Adult steelhead escapement within Potlatch River index watersheds was highly variable over time, but both watersheds experienced sharp declines starting in 2017. A commensurate decline was observed in steelhead populations throughout Idaho (Knoth et al. 2018; Dobos et al. 2019, 2020; Smith et al. 2021) and largely attributed to poor conditions during outmigration of smolts (Faulkner et al. 2016) and at ocean entry (Peterson et al. 2018). Ocean conditions for juvenile salmonid survival remained poor from 2015 to 2019 (Peterson et al. 2019). Thus, adult steelhead escapement to the Potlatch River and other Idaho drainages have remained low through 2020 (Smith et al. 2021). While recent declines in adult steelhead escapement are discouraging, the plausible mechanisms causing declines likely occurred outside the Potlatch River basin.

We have struggled to accurately estimate spawner abundance in BBC during recent years. Low adult steelhead detections and frequent antenna outages at the PIT-tag array site during the spring resulted in the inability to produce expanded spawner estimates from 2017-2020. We upgraded the BBC PIT-tag array site in the fall of 2019 and 2020 by installing lower profile corded PIT-tag antennas which should reduce outages due to high flow events. Nonetheless, there is a need to explore new analytical techniques to produce expanded estimates during years with low detections at the PIT-tag array. Improvements in the ability to estimate spawner abundance will lead to more accurate adult trend monitoring and productivity analyses (juvenile recruits per spawner) in BBC and enhance our ability to assess the effects of habitat restoration in the watershed. Furthermore, we should consider installing a PIT-tag array on the mainstem Potlatch River near the mouth. This would allow us to monitor total spawners in the basin and evaluate a basin-wide response to restoration actions in the index watersheds.

Freshwater habitat conditions can be a key driver influencing salmon population dynamics (Jones et al. 2020) and improvements in tributary watersheds can improve population resilience even when out of basin conditions are poor (Justice et al. 2017). Restoration efforts in the Potlatch River basin are focused on improving freshwater rearing habitat conditions and the monitoring

framework is focused on freshwater life history in tributary systems to accurately assess effectiveness of restoration efforts. Life cycle monitoring of steelhead inherently includes out of basin factors (i.e. ocean conditions, hydrosystem) which confound results associated with adult returns. The response in adult steelhead to restoration actions will be most evident when improvements to juvenile steelhead growth, survival, and abundance coincide with favorable conditions outside the basin. Continued evaluation of freshwater productivity within the Potlatch River basin will provide meaningful restoration effectiveness monitoring independent of adult returns.

Response of juvenile production and growth can also be associated with density-dependent mechanisms in tributary systems. Monitoring juvenile emigrant production during a variety of adult returns provides a better understanding of recruits per spawner relationships and the ability for the population to persist through years of low adult abundances. We have observed different patterns in emigrant production in the index watersheds during low adult abundance years. Juvenile emigrant production in BBC remained relatively stable, whereas emigrant production in the EFPR declined substantially during the recent period of low adult returns. Population productivity in BBC is highly density dependent as a result of limited juvenile rearing capacity. Density dependent factors such as resource limitation (i.e. food and space) likely regulate emigrant production in years of higher spawner abundance, but are less critical when spawner abundance is low resulting in more consistent emigrant production across years. The extent to which density dependence regulates emigrant production in the EFPR is less clear and is confounded by the fact that not enough adults have returned to indicate what capacity is in the system. Furthermore, a substantial portion of EFPR juveniles emigrate as age-1 fish and rear an additional year outside the drainage before smolting, which makes direct comparisons to BBC challenging. Low emigrant production in the EFPR during recent years is discouraging, but the positive shifts in emigrant age, growth, and survival may provide resilience to the population through periods of low adult abundance.

Restoration efforts that improve the quantity and quality of rearing habitat in the Potlatch River should increase potential capacity of the system, thereby reducing competition for resources (i.e., food or habitat) and improve freshwater productivity. Freshwater productivity improvements may manifest in higher juvenile abundance or juveniles with better survival (Roni 2005), resulting in higher overall population productivity. We hypothesized that the EFPR juvenile emigrant age structure may shift older as habitat improvements allow more juveniles to rear longer in the watershed versus emigrating early and rearing downstream (Uthe et al. 2017). In addition, juvenile emigrant length-at-age may increase as a result of improved growth due to increased resources. To date, nearly 20% of core juvenile steelhead distribution area in the EFPR has been treated (approximately 14 km) and an additional 6 km will be treated by 2029. In recent years we have documented positive trends in emigrant age structure, length-at-age, and apparent survival which is consistent with our hypothesis. These findings are encouraging and continued monitoring of the EFPR emigrant population will further elucidate the relationship between habitat actions and steelhead response in the upper Potlatch River watershed.

Life cycle modeling simulations indicate planned restoration projects have the potential to significantly increase juvenile steelhead production in the index watersheds over the next 10 years (Uthe et al. 2017). In the BBC watershed, three large-scale barrier removal and flow supplementation projects could reconnect nearly 35 km of additional rearing habitat, effectively doubling the linear amount of rearing habitat currently available in the watershed. To date, one of these projects has been completed (Big Meadow Creek culvert modification, Chapter 3) which reconnected approximately 10 km of additional rearing habitat. Given the positive fish and habitat responses to these types of projects (Uthe et al. 2017, Chapter 3) and the extensive size of the

three treatments, we expect to observe a detectable increase in smolt production following implementation. In the EFPR watershed, multiple smaller-scale restoration projects are planned to increase instream habitat complexity. We are encouraged by the positive shifts in the EFPR emigrant age, growth, and survival and expect continued improvements as more projects are implemented. An increase in emigrant abundance and/or shifts to fitter emigrants should result in higher steelhead productivity in the watersheds. Continued implementation of high priority habitat restoration projects within the Potlatch River drainage will move the needle towards those changes.

Steelhead exhibit diverse life histories and delayed ocean migration has been documented by numerous studies (Maher and Larkin 1955; Chapman 1958; Ward and Slaney 1988; Peven et al. 1994). Delayed migration of smolts is common within steelhead populations with the tendency of some to delay migration for an additional year or multiple years after leaving their natal tributary (Dobos et al. 2020). Current methods such as the Cormack-Jolly-Seber model commonly used to estimate smolt survival for Chinook Salmon cannot account for delayed migration and therefore, underestimate steelhead survival to LGR. This issue exists in both index watersheds, although estimating survival in the EFPR is more problematic since a large proportion of emigrants are age-1 fish that rear an additional year outside the EFPR watershed before ocean migration (Bowersox et al. 2011). University of Washington researchers have developed the Basin TribPIT model to estimate brood year survival by accommodating the variation in age at migration of steelhead (Lady et al. 2014; Buchanan et al. 2015). This model was recently evaluated using steelhead emigration data from BBC (Feeken et al. 2020) and provided initial estimates of cohort survival of wild juvenile steelhead from the Potlatch River to LGR. We plan to utilize the model to estimate brood year specific survival of smolts for each index watershed moving forward. Generating brood year smolt survival estimates to LGR will standardize smolt abundance estimates between both index watersheds, as well as with other Idaho steelhead monitoring locations, for large-scale evaluations and improved comparisons among steelhead subpopulations. Estimating age-specific survival of juvenile steelhead will also add to our understanding of the response of fish production to restoration efforts at smaller spatial scales.

Estimating the contribution of all life history characteristics and strategies is imperative when estimating population productivity. The importance of juvenile steelhead emigration during the fall has been documented in other Idaho populations (Dobos et al. 2020). We have experienced considerable challenges monitoring juvenile emigration during the fall because low flow conditions limit the operation of rotary screw traps. The limited data we collected suggests fall juvenile emigration from index watersheds may be considerable in some years (up to 15-18% of following year emigration). In addition, the majority juvenile emigrants leaving the EFPR during the fall are age-0 and age-1 fish, which supports the assumption that juvenile steelhead emigrate early from the drainage as parr, likely due to a lack of suitable overwinter rearing habitat. We plan to explore alternative techniques to monitor juvenile steelhead emigration during low flow periods in the fall. Improving the project's ability to estimate fall emigrant abundance and life history will refine productivity metrics.

In summary, habitat restoration and effectiveness monitoring activities are currently ongoing in the Potlatch River basin. We have established baseline levels of steelhead production and productivity in the index watersheds to compare against future evaluations and developed an understanding of the life histories of the steelhead in the index watersheds (Uthe et al. 2017). We continue to refine our index-watershed monitoring efforts. Recommendations discussed above and outlined below will improve our ability to accurately monitor and evaluate the status and trends of the steelhead populations in the index watersheds. We anticipate habitat treatment goals will be achieved by 2028-2029 in the two index watersheds and expect a 10-year post-treatment

monitoring period will be needed to ensure an adequate data set to detect changes in productivity related to restoration efforts.

RECOMMENDATIONS

1. Develop analytical techniques to generate expanded adult steelhead escapement estimates for BBC during periods of low adult abundance.
2. Generate juvenile steelhead survival estimates from natal reaches to LGR for the EFPR using Basin TribPIT model. Present both BBC and EFPR estimates in upcoming reports.
3. Investigate the feasibility of operating low flow fyke traps to monitor juvenile steelhead emigrants from index watersheds during fall season.
4. Install main-stem PIT-tag array infrastructure to monitor adult escapement in the entire Potlatch River basin and assess the extent of mainstem spawning and rearing.

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TABLES

Table 1. Age frequencies of wild adult steelhead captured at Potlatch River weirs and known destination wild adult steelhead sampled at Lower Granite Dam, 2008-2020. Partially aged fish were indicated by an X and repeat spawners were signified by R. FW is the estimated number of years spent in fresh water and SW is the estimated number of years spent in salt water.

	Adult steelhead age (FW:SW)													
Year	**X:1**	**1:1**	**2:1**	**3:1**	**X:2**	**1:2**	**2:2**	**3:2**	**2:3**	**3:3**	**4:1**	**5:0**	**R**	**Total aged**
Big Bear Creek														
2008	5	1	14	2	3	5	17	4	6	1	0	0	0	58
2009	2	1	26	4	5	1	36	2	0	0	0	0	0	77
2010	11	7	115	7	14	17	79	9	0	0	0	0	1	260
2011	0	4	11	0	3	11	32	0	1	0	0	0	1	63
2012	8	6	42	2	11	14	126	2	0	0	0	0	0	211
2013	0	1	3	0	2	4	9	0	0	0	0	0	0	19
2014	0	2	9	1	0	1	5	1	0	0	0	0	0	19
2015	1	3	11	1	2	4	8	0	0	0	0	0	1	31
2016	0	1	8	1	2	7	15	0	0	0	0	0	0	34
2017	0	0	1	0	0	2	10	2	0	0	0	0	0	15
2018	0	0	7	0	0	0	0	1	0	0	0	0	1	9
2019	0	0	3	0	0	0	1	0	0	0	0	0	0	4
2020	0	1	3	0	0	1	3	0	0	0	0	0	0	8
East Fork Potlatch River														
2008	1	0	33	2	2	1	15	1	2	1	0	0	0	58
2009	8	0	25	12	4	0	20	0	0	0	0	0	0	69
2010	3	0	21	13	3	0	18	10	0	0	0	0	1	69
2011	0	1	8	2	2	0	15	5	0	0	0	0	0	33
2012	3	2	15	11	5	1	24	5	2	0	0	0	0	68
2013	1	0	12	3	4	1	48	8	0	0	0	1	0	78
2014	5	8	38	10	3	4	15	0	0	0	0	0	0	83
2015	3	0	20	12	10	1	29	10	0	0	0	0	0	85
2016	0	1	26	6	1	1	33	7	0	0	1	0	0	76
2017	0	0	1	0	0	0	9	1	0	0	0	0	0	11
2018	0	0	11	5	0	0	3	1	0	0	0	0	0	20
2019	0	0	1	1	0	0	2	1	0	0	0	0	0	5
2020	4	0	8	4	1	1	4	1	0	0	0	0	0	23

FIGURES

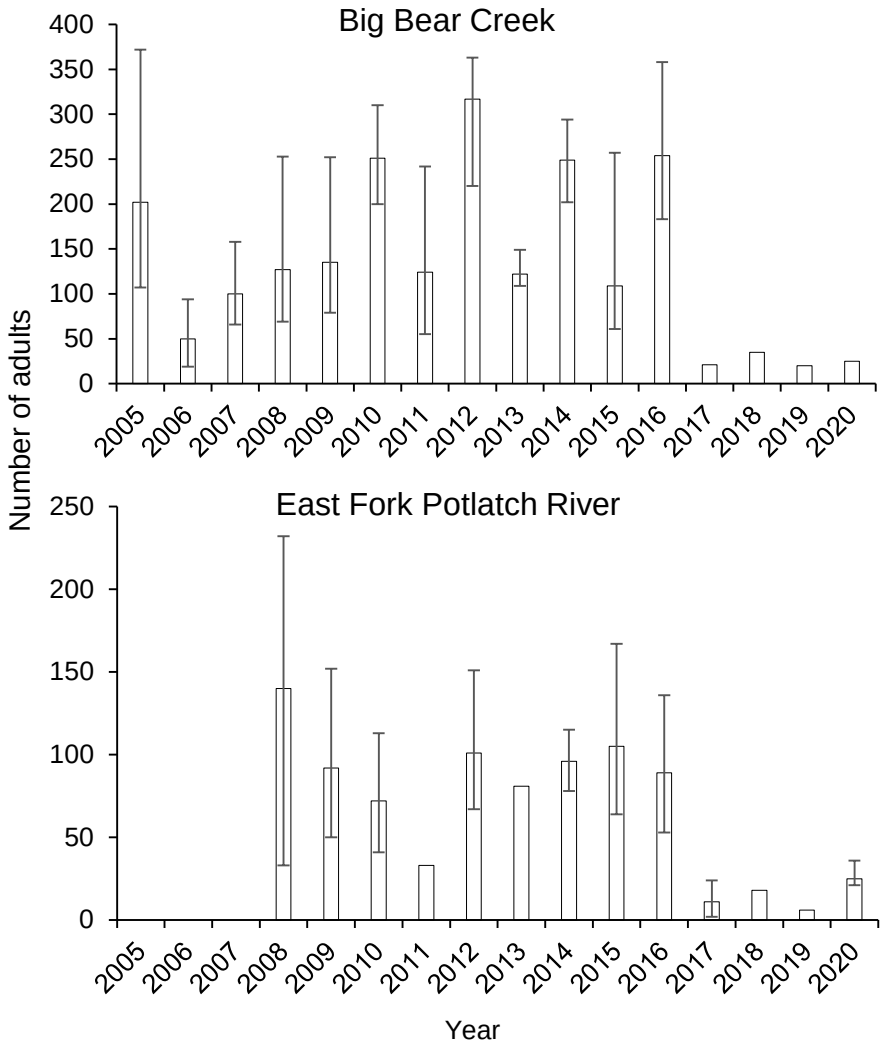

Figure 4. Abundance of wild adult steelhead in Big Bear Creek and the East Fork Potlatch River watersheds, 2005-2020. East Fork Potlatch River estimates begin in 2008. Error bars are at 95% confidence intervals, but could not be calculated in some years due to low detections at PIT-tag arrays.

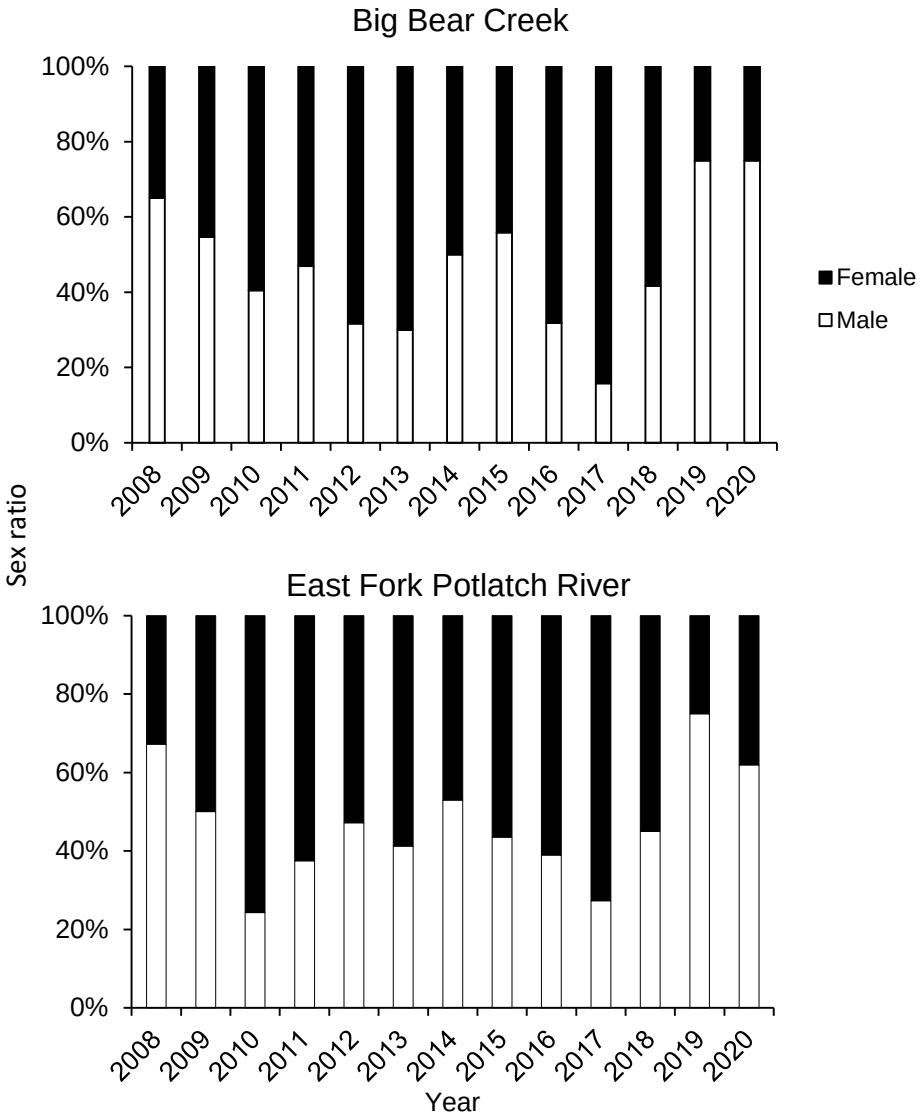

Figure 5. Sex ratio of wild adult steelhead captured at weirs or detected on PIT-tag arrays in the Big Bear Creek and the East Fork Potlatch River watersheds, 2008-2020.

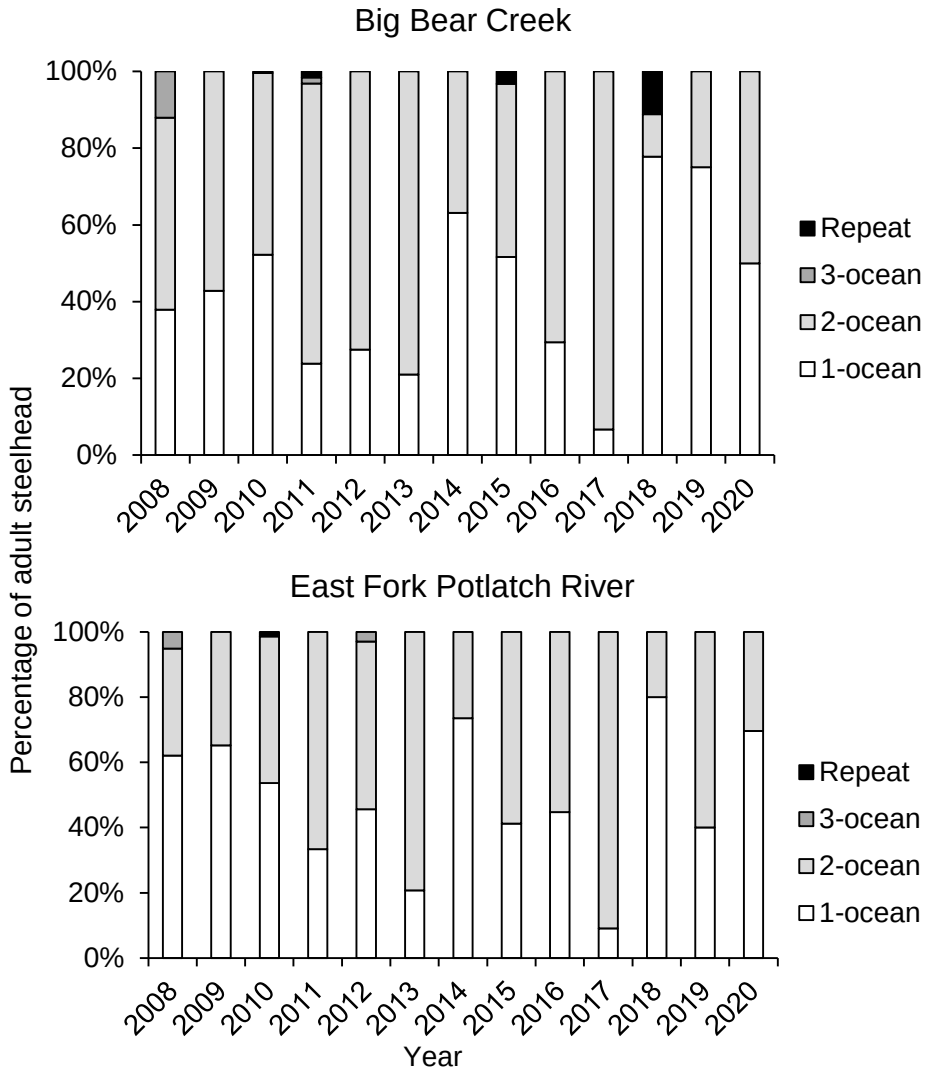

Figure 6. Age composition of wild adult steelhead captured at weirs or detected on PIT-tag arrays in the Big Bear Creek and the East Fork Potlatch River watersheds, 2008-2020.

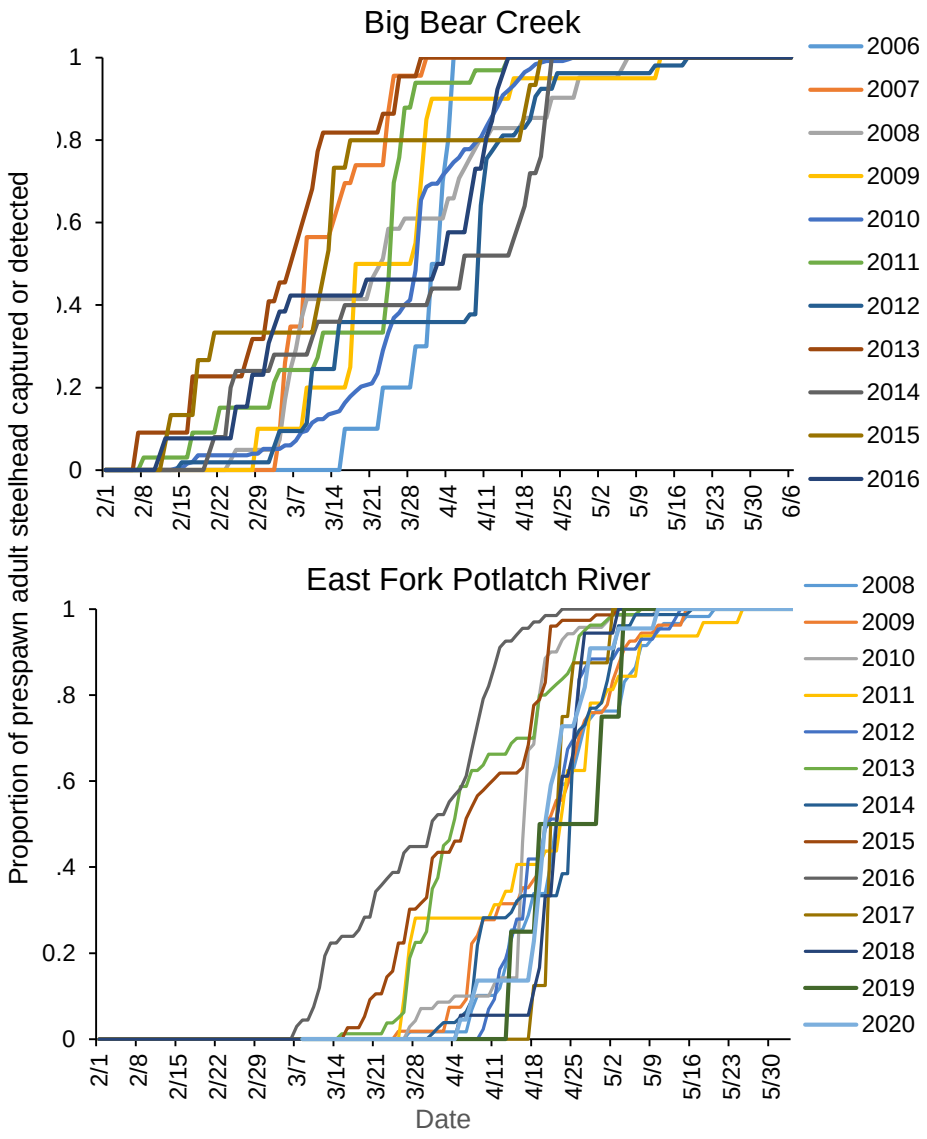

Figure 7. Cumulative run-timing curves for wild adult prespawn steelhead captured at a weir or detected on an PIT-tag array in the Big Bear Creek and the East Fork Potlatch River watersheds, from February through June, 2006-2020. Prespawn timing was not calculated from 2017-2020 in Big Bear Creek due to the low number of PIT-tag detections at the PIT-tag array.

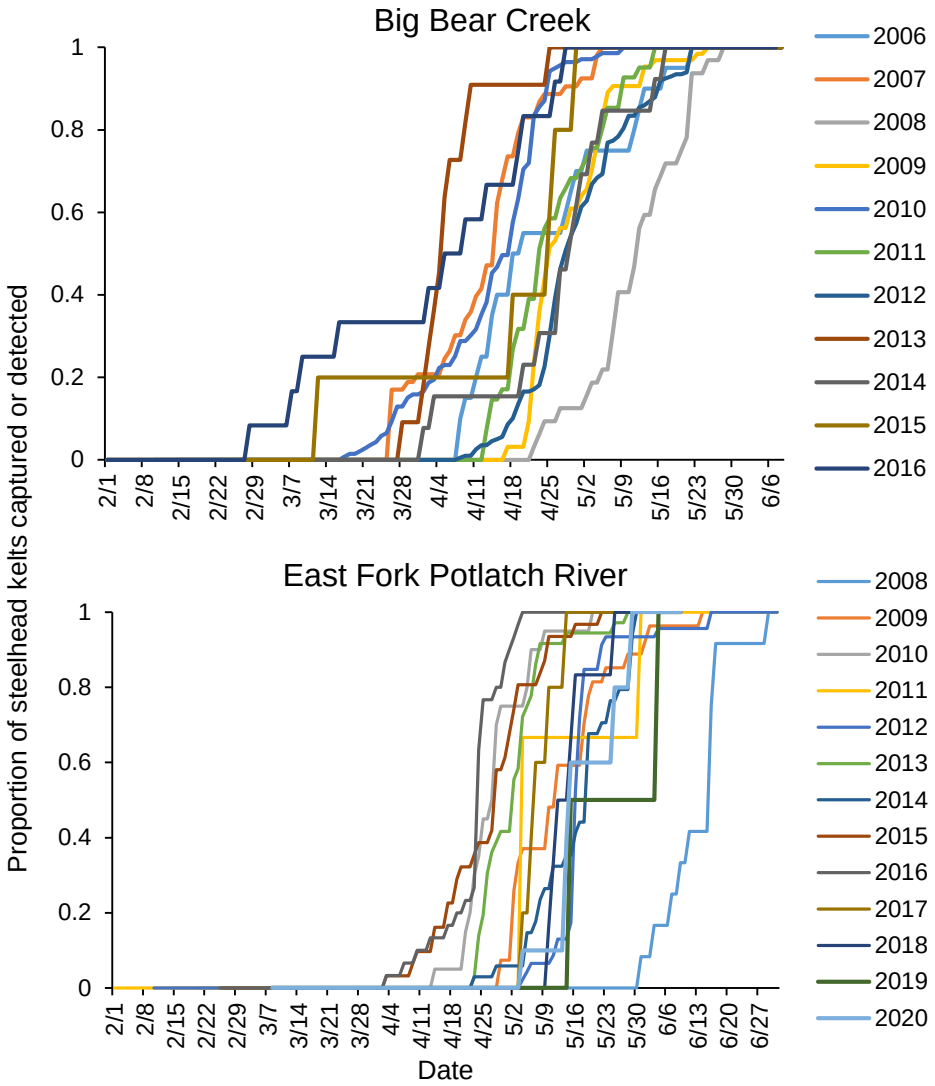

Figure 8. Cumulative run-timing curves for wild adult steelhead kelts captured at a weir or detected on an PIT-tag array at Big Bear Creek and the East Fork Potlatch River watersheds, from February through June, 2006-2020. Kelt timing was not calculated from 2017-2020 in Big Bear Creek due to the low number of PIT-tag detections at the PIT-tag array.

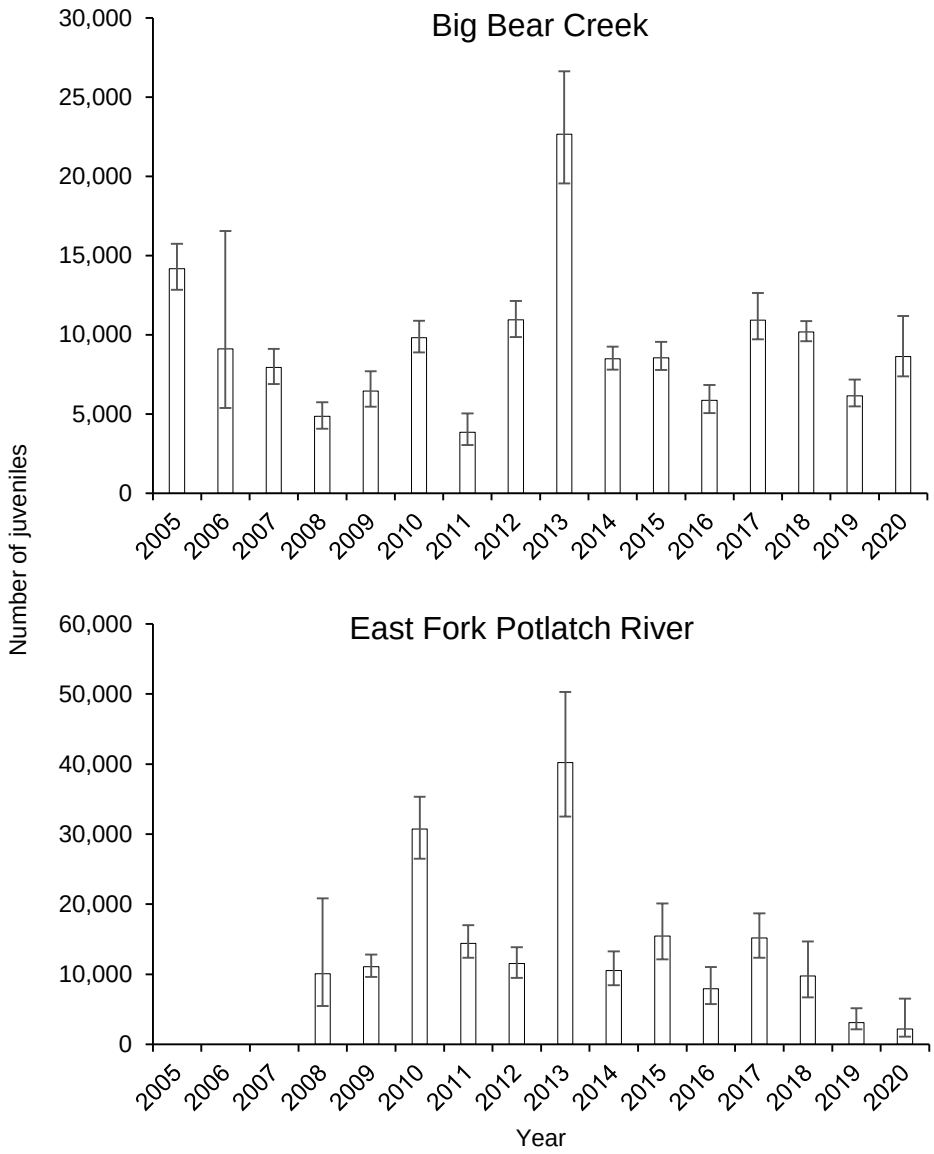

Figure 9. Estimated abundance of wild juvenile steelhead emigrants during the spring season in the Big Bear Creek and East Fork Potlatch River watersheds, 2005-2020. Error bars are 95% confidence intervals. East Fork Potlatch River estimates begin in 2008.

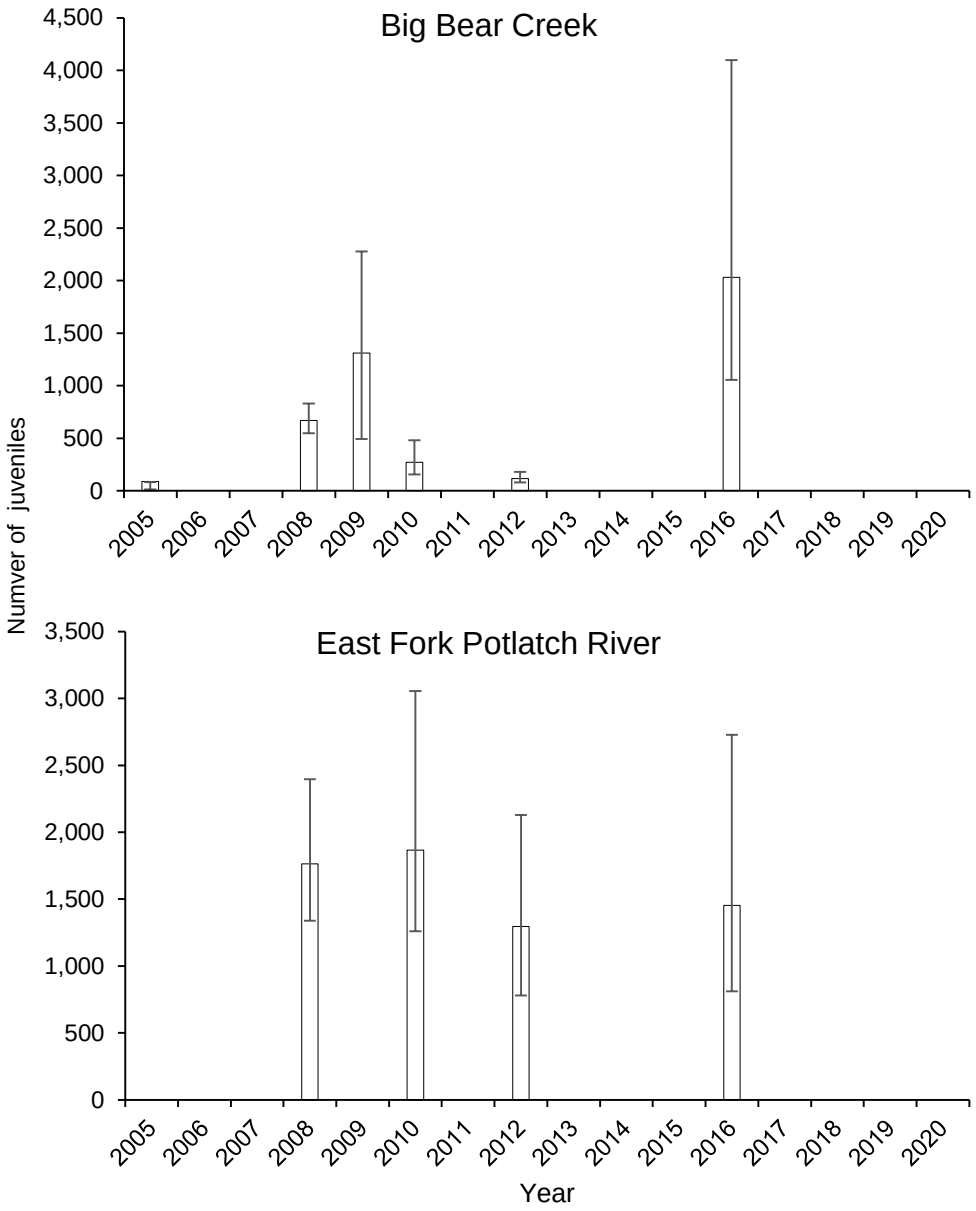

Figure 10. Estimated abundance of wild juvenile steelhead emigrants during the fall season in the Big Bear Creek and the East Fork Potlatch River watersheds, 2005-2020. Error bars are 95% confidence intervals. Rotary screw traps were not operated in years without estimates.

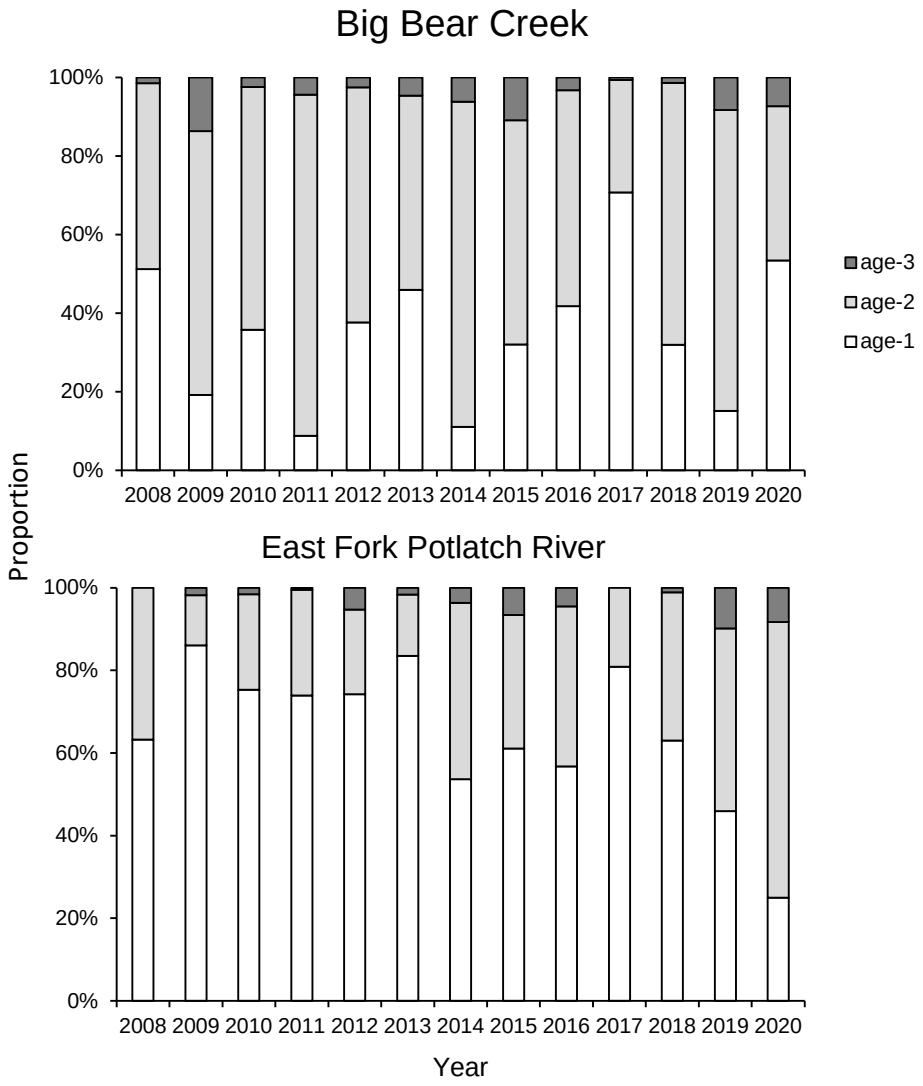

Figure 11. Age composition of wild juvenile steelhead emigrants captured at rotary screw traps during the spring season in the Big Bear Creek and East Fork Potlatch River watersheds, 2008-2020.

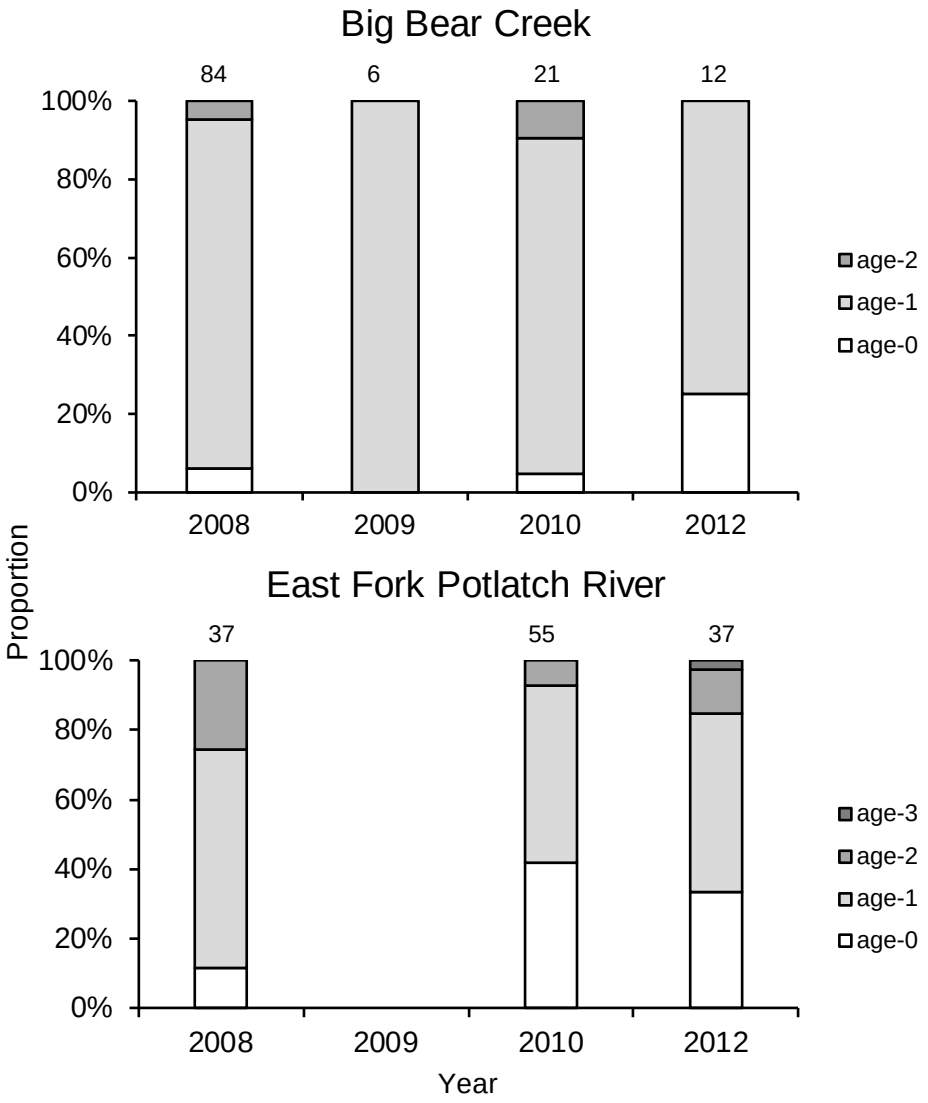

Figure 12. Age composition of wild juvenile steelhead emigrants captured at rotary screw traps during the fall season in the Big Bear Creek and East Fork Potlatch River watersheds, 2008-2012. Fall trapping did not occur in East Fork Potlatch River in 2009. Numbers above bars represent sample size.

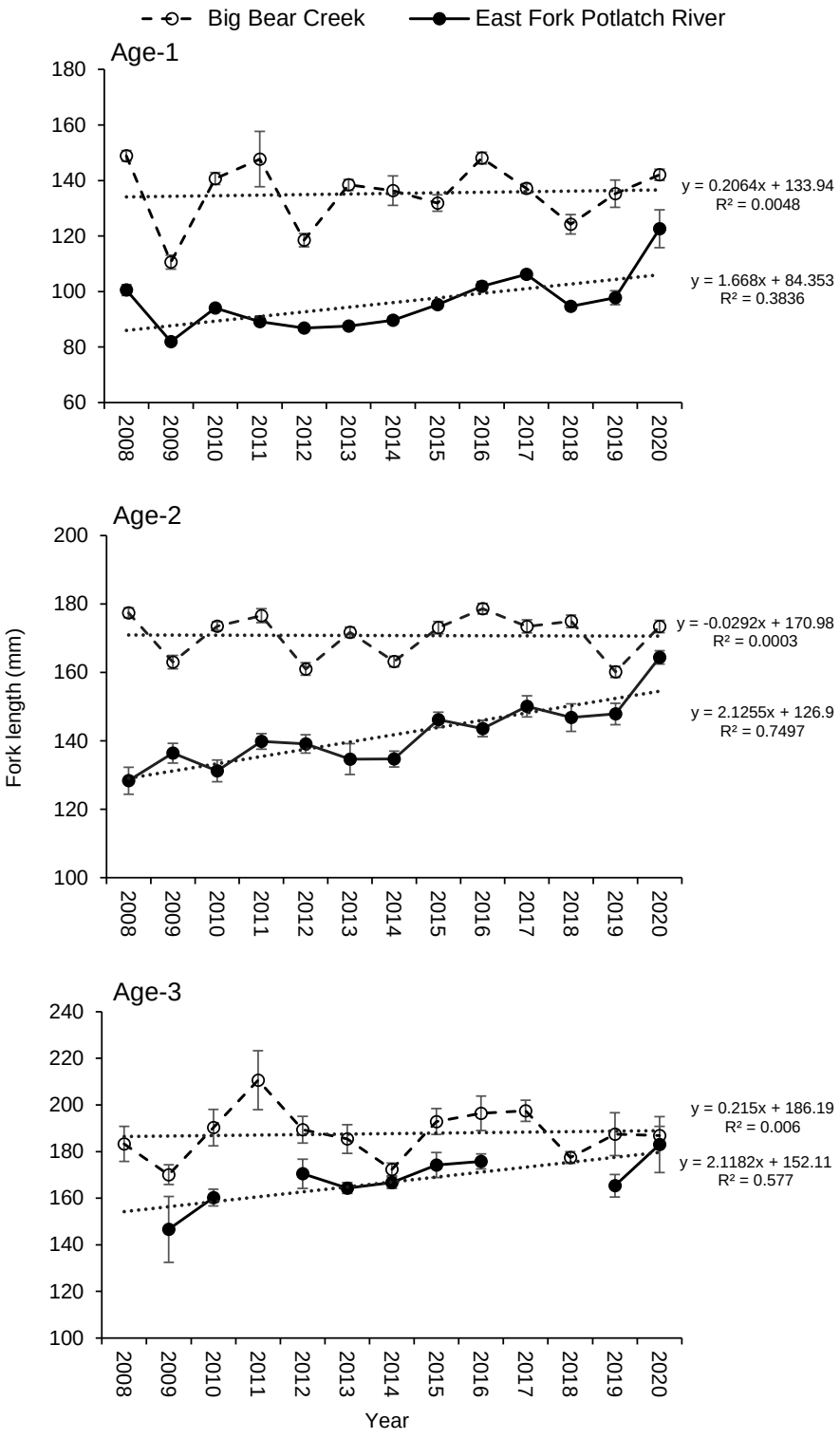

Figure 13. Mean length at age of wild juvenile steelhead emigrants captured at rotary screw traps during the spring season in the Big Bear Creek and East Fork Potlatch River watersheds, 2008-2020. Error bars are S.E.

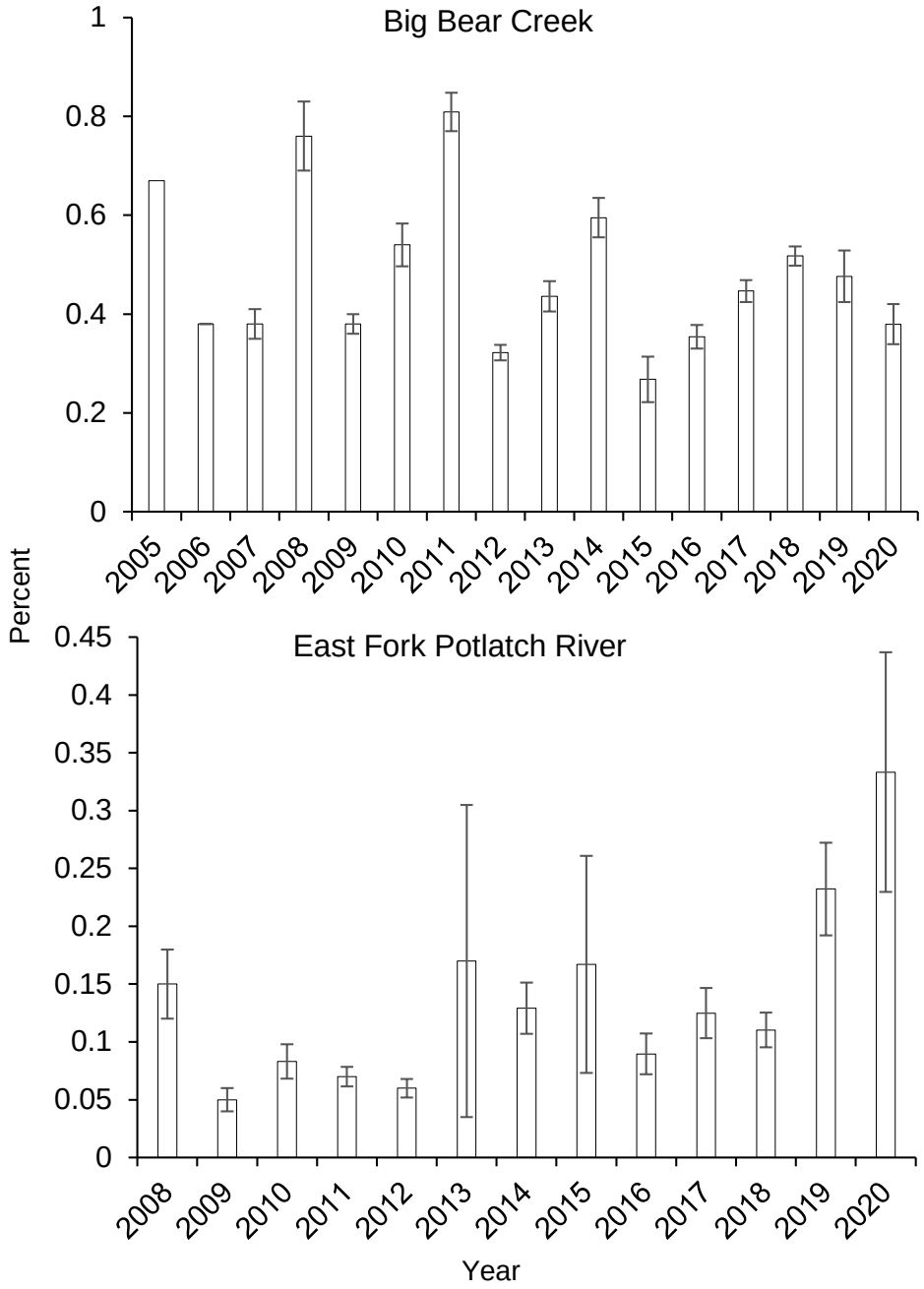

Figure 14. Apparent survival of spring emigrants from Big Bear Creek (top panel) and East Fork Potlatch River (bottom panel) rotary screw traps downstream to Lower Granite Dam, 2005-2020.

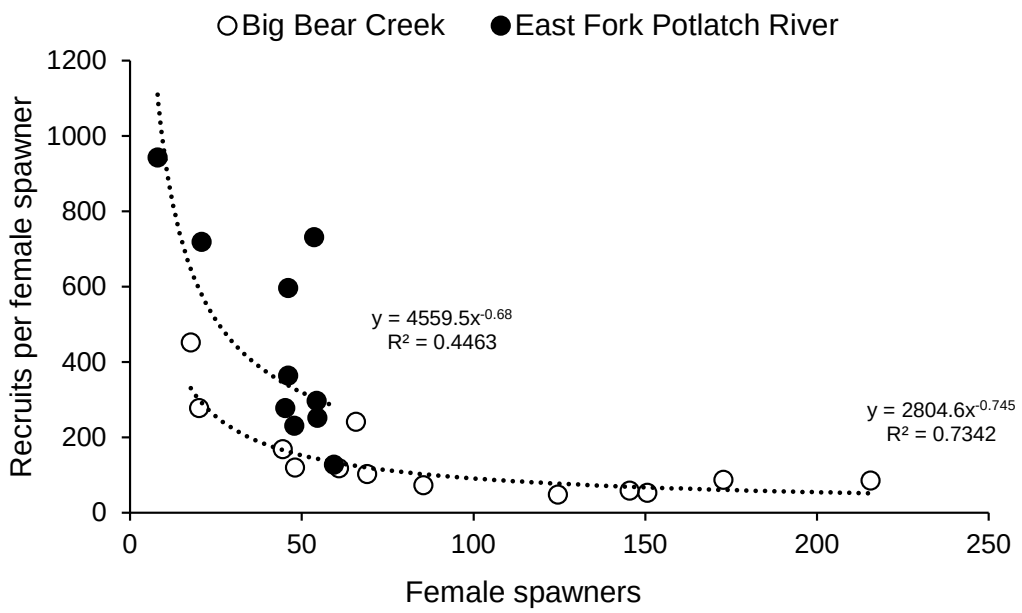

Figure 15. Productivity (juvenile recruits per female spawner) versus number of female spawners for the Big Bear Creek and East Fork Potlatch River watersheds. Big Bear Creek data are BYs 2005-2017 and the East Fork Potlatch River data are BYs 2008-2017.

CHAPTER 2: JUVENILE STEELHEAD PRODUCTIVITY AND HABITAT CONDITION RESPONSE TO HABITAT RESTORATION IN SELECT TRIBUTARIES WITHIN THE POTLATCH RIVER, IDAHO

ABSTRACT

We conducted habitat and electrofishing surveys to monitor rearing habitat conditions and juvenile steelhead (*Oncorhynchus mykiss*) production metrics in treatment and control areas in the Potlatch River basin. Low water habitat surveys documented the extent of de-watering in the lower watershed tributaries and data from 2019 and 2020 fell within the range of previous estimates. In the upper watershed, habitat metrics were highly variable with no discernable improvements in the East Fork Potlatch River treatment area in 2019 and 2020. Juvenile steelhead density estimates in 2019 and 2020 fell within the range of previous estimates for all tributaries except Pine Creek (control tributary) which had the lowest estimates on record. Parr-to-smolt survival estimates decreased in 2019 and 2020, relative to 2018, for all tributaries except the West Fork Little Bear Creek which showed an opposite pattern. The positive correlations between juvenile steelhead growth and survival with the amount of wetted habitat validates the need to improve base flow conditions in the lower Potlatch River watershed. In the upper watershed, a positive correlation between juvenile steelhead density and large wood highlights the need to improve in-stream habitat complexity in the treatment area.

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INTRODUCTION

Tributary-scale monitoring was incorporated into the project's study design to measure the response in juvenile steelhead (*Oncorhynchus mykiss*) production and habitat conditions in treatment areas where habitat treatments occurred relative to control areas where no habitat treatments occurred. These monitoring efforts are complementary to index-watershed monitoring efforts described in Chapter 1 and provide the necessary data to isolate responses to specific restoration techniques which should allow us to better understand mechanisms behind a response. Understanding the causal link between a habitat treatment and fish response is a fundamental step towards replicating restoration efforts elsewhere within and outside the basin (Bennett et al. 2016).

The objective of this chapter is to link habitat restoration efforts to changes in juvenile steelhead production metrics and habitat conditions within monitored tributaries. We are currently in the treatment phase of habitat restoration efforts in the Potlatch River basin (Figures 2 and 3). Preliminary results presented here include pretreatment data and partial data from the treatment phase. Ultimately, tributary-scale monitoring data will be assessed using a before-after-control-impact (BACI) analysis with in-basin and out of basin controls (Roni 2005). The final assessment will be completed after habitat treatment goals are achieved in the basin.

METHODS

Tributary-scale monitoring efforts were conducted concurrently in treatment and control areas within the lower and upper Potlatch River watersheds (Figure 16). In the lower watershed, treatment tributaries included Big Bear Creek (BBC), Little Bear Creek (LBC), and the West Fork Little Bear Creek (WFLBC), and the control tributary was Pine Creek (PNC). In the upper watershed, the treatment area was the main-stem East Fork Potlatch River (EFPR) downstream of Pivash Creek (lower 22 km) and control areas were the main-stem EFPR upstream of Pivash Creek and the West Fork Potlatch River (WFPR). We defined the WFPR as the area upstream of the confluence with the EFPR, which included portions of the main-stem Potlatch River as well as major tributaries Cougar Creek, Feather Creek, and Moose Creek. For habitat surveys in the lower Potlatch River watershed, Corral Creek was monitored as a treatment tributary to evaluate changes in base flow conditions in response to meadow restoration projects in the drainage and Cedar Creek was monitored as an additional control tributary.

Habitat Surveys

We monitored variables associated with the primary limiting factor of low summer base flows in the lower watershed. Low Water Habitat Availability Protocol (LWHAP) surveys (Bowersox et al. 2009) were conducted to evaluate the amount of wetted habitat and pool density within treatment and control tributaries. Sample sites were selected using a Generalized Random Tessellation Stratified (GRTS) design, where tributaries were stratified into upland and canyon reaches to disperse sites throughout each tributary. Two 500m sites were surveyed within each reach (upland and canyon), resulting in four sites per tributary and a total of 26 sites surveyed annually (Figure 17). There were seven years (2007-2013) of pretreatment data and seven years (2014-2020) of treatment data collected across the tributaries in the lower watershed. The LWHAP surveys were designed to be a rapid assessment of base flow conditions and were conducted during the first week of August each year to provide temporal consistency. Length (m) of wetted habitat and number of pools were recorded at each site. Wetted habitat was defined as any area where there was standing water and pools were delineated as any wetted stream with

a maximum depth exceeding 30 cm. We calculated the average proportion of wetted habitat (linear %) and pool density (number per 100 m) for each tributary annually.

Habitat surveys were conducted in the upper Potlatch River watershed to monitor variables associated with the primary limiting factor of low in-stream habitat complexity. Primary response variables included large woody debris (LWD) quantity (number of pieces per km), pool density (number per km), and percent canopy cover. Protocols by Moore et al. (2006) and Bouwes et al. (2011) were modified to focus on these response variables. Briefly, all LWD pieces (≥10 cm in diameter and 1 m in length) were enumerated within the wetted channel. Pools were defined as depressions in the streambed that were concave in profile, laterally and longitudinally, and were bound by a 'head' crest and 'tail' crest. Only main channel pools were enumerated. The analysis was restricted to pools with a modal depth ≥40 cm because they represent typical winter rearing depths of juvenile *Oncorhynchus spp.* (Huusko et al. 2007). Canopy cover was visually estimated within 5-10 m of bankfull during the 2003-2004 and 2008 surveys. From 2013-2020, canopy cover was measured with a densiometer at four points along 10 sub-transects equally distributed throughout each 100 m site, for a total of 40 measurements per 100 m site. Canopy cover was expressed as a percentage of the site surveyed.

Effort placed on habitat surveys in the upper watershed before and during the treatment periods varied. However, the number of sites sampled per area was similar between treatment and control areas within each time frame. Earlier sample sites (2003, 2004, and 2008) were included in the analysis to bolster pretreatment data and were selected using a stratified random sampling technique (Schriever and Nelson 1999). Data from 2003 and 2004 were combined for the analyses because a portion of sites were sampled in 2003 and the rest in 2004. Current sample sites (2013-2020) were selected using a GRTS design and were co-located with electrofishing sites (Figure 18). Total number of sample sites were 40 in 2003/2004, 19 in 2008, and 23-25 in 2013-2020. Data collected in 2003, 2004, and 2008 were pretreatment and data collected in 2013-2020 were treatment.

Trends in habitat variables collected in the treatment and control areas in the upper watershed were compared across study years. Data were averaged annually for each variable across all sites within a given area. In addition, each habitat variable was analyzed as a ratio (treatment:control in a given year) to better illustrate the relative change between treatment and control areas. Within this analysis, a value of 1 would indicate equal quantities/proportions between treatment and control areas. A value >1 indicates the treatment area has a higher value relative to the control area. Over time, an increasing trend in the ratio indicates improvement in the treatment area relative to the control, and vice versa for a negative trend in the value.

Juvenile Steelhead Density, Parr-to-Smolt Survival, and Growth

Juvenile steelhead density was the primary fish response metric in the tributary-scale monitoring. Pretreatment data were limited so surveys conducted prior to the start of the project were incorporated into the analyses. Specifically, surveys conducted in 1996, 2003, and 2004 were included to bolster the pretreatment data set (Figures 18 and 19). The EFPR was not sampled in 1996. There were four years (1996, 2003, 2004, and 2013) of pretreatment data and seven years (2014-2020) of treatment data collected in the lower watershed. There were two years (1996 and 2004) of pretreatment data and eight years (2013-2020) of treatment data collected in the upper watershed.

Single-pass electrofishing was used to estimate trends in juvenile steelhead density within treatment and control tributaries (Kruse et al. 1998). Surveys were conducted during May-July

each year to provide temporal consistency. Site boundaries were established at stream habitat breaks 80-120 m apart. Crews began at the downstream boundary and electrofished in an upstream direction to the upstream boundary. Fish captured were identified to species and enumerated. All juvenile steelhead ≥80 mm were anesthetized using MS-222 solution, weighed (g), measured (FL; mm), and scanned for the presence of PIT tags. Juvenile steelhead (≥80 mm) not previously tagged were tagged in the abdomen with a 12 mm PIT tag. Site length and five widths were measured at each location to estimate area. Mean annual density estimates (fish per 100 m^2) were calculated for each tributary by averaging density from each site within a tributary. Trends in juvenile density in treatment and control areas were compared across the study years. In addition, density data were analyzed as a ratio (treatment:control in a given year) to better illustrate the relative change between treatment and control tributaries (see ratio explanation above).

Apparent survival of PIT-tagged steelhead from Potlatch River tributaries during the previous summer to LGR the following spring was used as an index of parr-to-smolt survival. Methods for estimating apparent survival were similar to methods described in Chapter 1. We conducted roving electrofishing surveys in addition to single-pass surveys during the summer months to increase the number of PIT-tagged juvenile steelhead in treatment and control tributaries. Our goal was to tag 300 juvenile steelhead in each tributary annually for survival and growth analyses. All captured juvenile steelhead ≥80 mm were anesthetized using MS-222 solution, measured (FL; mm), weighed (g), and PIT tagged. We assumed all steelhead PIT tagged during the summer would emigrate the following spring. There were six years (2008-2013) of pretreatment data and seven years (2014-2020) of treatment data collected in the lower watershed. In the upper watershed, parr-to-smolt survival estimates were only generated for the EFPR because too few juvenile steelhead were tagged and subsequently detected in the West Fork Potlatch River to generate an estimate. Trends in apparent survival for tag groups in treatment and control tributaries were compared across years.

Juvenile steelhead growth (summer to fall) was monitored as a response to restoration treatments in the lower watershed. Electrofishing surveys were conducted during late October through early November annually to recapture previously PIT-tagged juvenile steelhead. Surveys were only conducted in the WFLBC and LBC (treatment tributaries) and PNC (control tributary) where the recapture rate was sufficient (n >10). All captured juvenile steelhead ≥80 mm were scanned for PIT tags and measured (FL; mm). Growth (mm per d) was calculated as the change in fork length between time of tagging and time of recapture for each recaptured PIT-tagged fish. Means and standard deviations were calculated for each tributary to compare growth rates between treatment and control tributaries across years.

A Pearson's correlation analysis was used to examine the relationship(s) between juvenile steelhead production metrics (density, growth, and survival) and key habitat metrics in each watershed. In the lower watershed, we examined the relationships between juvenile steelhead density, growth, and survival with the amount of wetted habitat. In the upper watershed, we examined the relationships between juvenile steelhead density and survival with LWD and pool density. In addition, we examined the relationship(s) between juvenile survival and growth with the density of juvenile steelhead (age 1+) in each watershed to assess the influence of density-dependent factors on these metrics. In most instances, data used in the analyses were averages across sites and tributaries for each watershed. One exception was the analyses between juvenile steelhead density and habitat metrics in the upper watershed where site specific values were used because density and habitat sites were co-located.

RESULTS

Habitat Surveys

Lower Potlatch River Watershed

Low water habitat surveys highlighted the extent of low summer base flows in the lower watershed treatment and control tributaries (Figure 20). The amount of wetted habitat ranged from 32-91% in 2019 and 56-96% in 2020, and was highest in the WFLBC in both years. Pool density ranged from 0.7-6.6 pools per 100 m in 2019 and 1.1-7.4 pools per 100 m in 2020, and was highest in Cedar Creek in both years. The WFLBC (treatment tributary) had the highest mean amount of wetted habitat (2007-2018) among tributaries at 92% (range = 53.6-100.0%) and Corral Creek (treatment tributary) had the least amount of wetted habitat at 42% (range = 26.2-68.3%). Cedar Creek (control tributary) had the highest mean pool density (2007-2018) among tributaries at 5.1 pools per 100 m (range = 0.9-8.0 pools per 100 m) among tributaries and Corral Creek (treatment tributary) had the least amount of pools at 1.0 pools per 100 m (range = 0.2-2.3 pools per 100 m). Wetted habitat and pool density estimates tracked relatively closely over time among three treatment tributaries (BBC, WFLBC, and LBC) and one control tributary (Cedar Creek), but not the other tributaries. Pine Creek displayed the least variation in the amount of wetted habitat and pool density across years.

Upper Potlatch River Watershed

Habitat data in upper watershed were highly variable over time and across tributaries (Figure 21). Canopy cover ranged from 40.0-62% in 2019 and 36-71% in 2020. Pool density ranged from 4.5-11.4 pools per km in 2019 and 1.5-3.2 pools per km in 2020. Large wood density ranged from 92-541 pieces per km in 2019 and 70-302 pieces per km in 2020. Canopy cover has increased over time for all tributaries, most notably the EFPR control. Conversely, pool density has decreased for all tributaries since 2014. Large wood density has increased in the EFPR control area over time, but has remained constant in the EFPR treatment and WFPR control areas. Habitat data were not collected in the West Fork Potlatch River in 2017 and LWD data in 2015 were not included in the analysis due to inconsistencies in data collection.

There were few discernable trends in the habitat ratio metrics across years (Figure 22). Canopy cover in the EFPR treatment area has decreased relative to the WFPR control area since 2013, but has remained constant relative to the EFPR control area since 2008. Similarly, pool density in the EFPR treatment area has decreased relative to the WFPR control area since 2016 and has remained constant relative to the EFPR control area since 2013. Large wood density in the EFPR treatment area has remained relatively constant in relation to both control areas across years.

Juvenile Steelhead Density, Parr-to-Smolt Survival, and Growth

Lower Potlatch River Watershed

Juvenile Steelhead Density—Juvenile steelhead density estimates tracked similarly in treatment and control tributaries over time, with peak estimates in 2013 and 2017 (Figure 23). Juvenile steelhead density estimates ranged from 1.1-3.9 fish per 100 m^2 in 2019 and 2.1-9.5 fish per 100 m^2 in 2020. Estimates were the highest in the WFLBC and lowest in PNC in both years. Pine Creek density estimates in 2019 and 2020 were the lowest on record for that drainage. Mean juvenile steelhead density (2013-2018) was 10.7 fish per 100 m^2 in LBC (range = 6.3-20.0 fish

per 100 m^2), 10.1 fish per 100 m^2 in the WFLBC (3.4-18.7 fish per 100 m^2), 7.3 fish per 100 m^2 in PNC (range = 3.4-13.2 fish per 100 m^2), and 5.7 fish per 100 m^2 in BBC (range = 2.4-10.9 fish per 100 m^2).

Juvenile steelhead density in the treatment tributaries (WFLBC, LBC, and BBC) was relatively constant in relation to the control tributary (PNC) from 1996-2018, but increased in 2019 and 2020 (Figure 24). The increases in 2019 and 2020 were driven by the record low density estimates in PNC during these years, rather than high estimates in the treatment tributaries. There was no correlation (r = 0.22) between juvenile steelhead density and the amount of wetted habitat in the lower watershed (Figure 25).

Parr-to-smolt Survival—The number of fish tagged and tags detected in the hydrosystem varied across tributaries and years in the lower watershed (Table 2). The number of fish tagged ranged from 48 to 468 in 2019 and 34 to 339 in 2020 and the least amount of fish were tagged in BBC in both years. Tag distribution in PNC in 2019 and 2020 was below average due to low densities of juvenile steelhead in the drainage. The WFLBC had the highest mean number of fish tagged (2008-2018) among tributaries at 364 fish (range = 113-492 fish). The majority (98%) of the tags detected in the hydrosystem were detected the following spring after tagging.

Apparent survival from tributary to LGR the following spring (parr-to-smolt survival) varied among tributaries and across years (Figure 26). Survival estimates ranged from 4.1-16.5% in 2019 and 5.9-18.1% in 2020 and were highest in the WFLBC in both years. There was a slight increase in survival estimates across all tributaries in 2020 compared to 2019, but within the range of past values. Mean parr-to-smolt survival estimates (2008-2018) were 13.7% in the WFLBC (range = 3.1-25.0%), 13.5% in LBC (range = 3.2-35.0%), 13.0% in PNC (range = 2.5-22.5%), and 10.3% in BBC (range = 4.2-16%). Estimates could not be generated for six years in BBC, one year in the WFLBC and LBC, and four years in PNC because of low detections in the hydrosystem. Parr-to-smolt survival was positively correlated with the amount of wetted habitat (r = 0.36) but not correlated with fish density (r = -0.03) in the lower watershed (Figure 27).

Growth Rates—The number of fish recaptured during fall electrofishing surveys varied across tributaries. The number of steelhead recaptured ranged from 0-30 fish in 2019 and 0-51 fish in 2020 (Table 3). No juvenile steelhead were recaptured in PNC during 2019 and 2020, due to the low number of tags distributed in the drainage. The WFLBC had the highest mean number of recaptures (2014-2018) among tributaries at 31 fish (range = 10-44 fish). Mean time at large ranged from 121 d in the WFLBC to 137 d in PNC (Table 3).

Daily summer growth rates varied among tributaries but was generally higher in the control tributary (PNC) relative to the treatment tributaries (LBC and the WFLBC) (Figure 28, Table 3). Growth rates ranged from 0.018-0.041 mm per d in 2019 and 0.049-0.068 mm per d in 2020, and were highest in the WFLBC in both years. Growth rates could not be calculated for PNC in 2019 and 2020 since no fish were recaptured. Mean summer growth (2014-2018) was the highest in PNC at 0.066 mm per d (range = 0.044-0.086 mm per d). Juvenile steelhead growth showed a weak positive correlation with the amount of wetted habitat (r = 0.33) and a weak negative correlation with fish density (r = -0.28) in the lower watershed (Figure 29).

Upper Potlatch River Watershed

Juvenile Density—Juvenile steelhead density estimates in treatment and control areas displayed relatively similar trends over time, most notably a sharp decrease in 2015 (Figure 30). Juvenile steelhead density estimates ranged from 0.6-4.8 fish per 100 m^2 in 2019 and 1.2-10.5

fish per 100 m^2 in 2020 and were the highest in the EFPR control area in both years. Density estimates in 2019 and 2020 fell within range of previous estimates for each area. Mean juvenile steelhead density (2013-2018) was 2.5 fish per 100 m^2 in the EFPR treatment area (range = 0.5-5.3 fish per 100 m^2), 7.9 fish per 100 m^2 in the EFPR control area (range = 4.4-10.8 fish per 100 m^2), and 0.8 fish per 100 m^2 in the WFPR control area (range = 0.0-2.0 fish per 100 m^2).

Juvenile steelhead density in the EFPR treatment area has declined relative to the West Fork Potlatch River control since 2015, but has remained stable in relation to the EFPR control area across years (Figure 31). Juvenile steelhead density was positively correlated with the density of LWD (r = 0.46) but was not correlated with pool density (r = 0.01) in the upper watershed (Figure 32).

Parr-to-smolt Survival—On average, 281 juvenile steelhead were tagged annually in the EFPR during roving surveys (Table 2). An average of 19 tags were detected in the hydrosystem annually, 87% of which were detected the following spring after tagging. We were able to tag only a few steelhead in the West Fork Potlatch River (<25 fish total) across the years. Of those few tags, only three were detected in the hydrosystem so no results are presented for this group.

Apparent survival from EFPR to LGR the following spring (parr-to-smolt survival) varied three-fold across years (Figure 33). Survival estimates in tag years 2019 and 2020 were the lowest on record at 4.8% and 5.0%, respectively. Mean parr-to-smolt survival (2008-2018) was 8.9% (range = 5.3-14.9%). Estimates could not be generated for tag groups in 2010, 2014, and 2015 because of low detections in the hydrosystem.

DISCUSSION

To date, we have collected over eight years of data characterizing habitat conditions and juvenile steelhead production in the treatment and control areas in the Potlatch River. Low water surveys continue to highlight the severity of low summer base flow conditions in lower watershed tributaries. Habitat metrics in the upper watershed were highly variable across years, with no noticeable improvements in the treatment area relative to the control areas. In the lower watershed, we observed an increase in juvenile steelhead density in the treatment tributaries, relative to the control tributary, during 2019 and 2020. However, the changes were driven by the record low density estimates in the control tributary (PNC), rather than high estimates in the treatment tributaries. Modifications to our tributary-scale monitoring framework discussed below should enhance our ability to detect and isolate responses to particular habitat actions and provide a better understanding of the cause of a response.

De-watering of rearing habitat limits space and food resources for juvenile steelhead and negatively impacts overall productivity. This issue is well documented within the lower Potlatch River watershed, where only 5 of 84 sample sites were 100% wetted in 14 years of monitoring. Multiple studies have documented a strong association between stream discharge and juvenile salmonid abundance (Jager et al. 1997; Mitro et al. 2003; Beecher et al. 2010). For example, Coho Salmon smolt production was positively related to increasing summer flows in Washington streams (Beecher et al. 2010). Likewise, juvenile steelhead survival increased with higher summer flows in California streams (Grantham et al. 2012). Stream flows can mediate the size and suitability of habitat for fish (Dewson et al. 2007). Thus, higher summer flows can benefit rearing conditions by preserving connectivity and pool depth, moderating stream temperatures and dissolved oxygen levels, and maintaining the production and delivery of food resources for juvenile fish (Myrick and Cech 2004; Hayes et al. 2008; Nilsson and Renofalt 2008). Given the

importance of instream flow to juvenile salmonid production, restoration emphasis should continue to focus on increasing baseflow conditions and expanding rearing habitat availability in the lower Potlatch River watershed.

Flow supplementation has been identified as a restoration approach to increase summer base flows within the lower Potlatch River watershed (Resource Planning Unlimited 2007). Life cycle modeling simulations suggest the implementation of a flow supplementation project on Little Bear Creek could increase smolt production in the lower Potlatch River watershed by 27% (Uthe et al. 2017). Idaho Department of Fish and Game evaluated the efficacy of flow supplementation in Little Bear Creek during 2015 and 2016 as part of a pilot study (Uthe et al. 2017; Hand et al. 2020). The project increased the amount of available rearing habitat, improved pool density and connectivity, and moderated stream temperatures and DO levels in the treated reach (Uthe et al. 2017; Hand et al. 2020). In addition, we documented benefits to juvenile steelhead growth, survival, and abundance in response to the flow supplementation (Knoth et al. 2021). However, the pilot project lasted only two years and benefits to juvenile steelhead production were temporary and did not persist past 2017. Flow supplementation projects can rapidly improve juvenile steelhead rearing conditions and productivity and should be prioritized to address low summer base flows in the lower Potlatch River watershed.

Meadow restoration has also been identified as a potential restoration approach to improve low base flow conditions in the Potlatch River basin (Resource Planning Unlimited 2007; Potlatch Implementation Group 2019). The premise behind this technique is that restoring natural prairies and meadow systems will minimize peak storm discharge and increase water infiltration and storage, thereby maintaining adequate summer stream flows. Corral Creek is the primary drainage where this approach is being tested. The first meadow restoration project was implemented in 2013 and to date nearly seven miles of Corral Creek has been treated. Nonetheless, there has not been a detectable increase in the amount of wetted habitat or pool density in Corral Creek during the seven years of restoration treatments. It is likely LWHAP surveys are not at the proper scale to detect fine scale changes in water storage while work within treated meadows has provided some indication of positive response. For example, post-treatment monitoring of the Racetrack Meadow restoration project noted increases in both aquifer and ponded water storage, improvements to channel and riparian conditions, and a decrease in the number of days the project area went dry (Dansart 2018 as cited in Potlatch Implementation Group 2019). Nonetheless, we have not documented an improvement in the quantity of summer flow conditions in Corral Creek on the broad scale that LWHAP surveys monitor. A more detailed analysis is needed if this approach is prioritized in other drainages in the Potlatch River basin.

Restoration treatments in the upper watershed are focused on increasing instream habitat complexity and riparian function through the installation of large wood structures to promote pool formation and floodplain reconnection. These treatments are intended to improve overwinter habitat conditions for juvenile steelhead in the EFPR. Habitat characteristics that provide shelter may positively impact the overwinter survival of juvenile salmonids by mitigating predation risk and decreasing energy costs (Whalen and Parrish 1999; Armstrong and Griffiths 2001; Bradford and Higgins 2001). For example, Mitro and Zale (2002) found that overwinter survival of age-0 rainbow trout was 18–23% greater in reaches with complex bank habitat, high gradient, and large substrate than in reaches with simple bank habitat. Solazzi et al. (2000) found overwinter survival of juvenile Coho Salmon and steelhead increased following habitat modifications that increased the complexity and quantity of winter rearing habitat. Steelhead parr-to-smolt survival rates in the EFPR averaged about 9.0% annually and have not increased during treatment years. However, we have observed positive trends towards older and larger emigrants in the EFPR (see Chapter 2), suggesting improvements in rearing conditions are occurring. Still, the question remains as to

what extent juvenile steelhead are using the LWD treatment reaches as overwinter habitat in the EFPR. To answer this question we are implementing a radio telemetry project in 2021 to document key habitat attributes and areas used by juvenile steelhead in the summer and winter periods in the EFPR. Results from the study will allow us to determine the extent juvenile steelhead are utilizing the LWD treatment areas as overwinter habitat and inform future restoration actions in the drainage.

Juvenile steelhead growth was minimal during the summer in lower Potlatch River tributaries. Multiple studies have investigated seasonal growth patterns of juvenile steelhead in temperate regions and a pattern of rapid growth in winter-spring and minimal growth in summer has been documented in juvenile steelhead populations in California (Hayes et al. 2008; Sogard et al. 2009), Oregon (Tattam et al. 2017), and Idaho (Myrvold and Kennedy 2015). Reduced growth during the dry summer season may be a function of minimum flow rates and low delivery of prey via drift (Sogard et al. 2009). Similarly, Hayes et al. (2008) speculated reduced growth rates during periods of low summer flows were a function of reduced prey availability during a time when warmer temperatures increased metabolic demands of fish. Furthermore, high water temperatures were a strong factor limiting growth in juvenile steelhead during summer in Lapwai Creek, Idaho (Myrvold and Kennedy 2015). Banks and Bowersox (2015) documented summer water temperatures in excess of 20°C in lower Potlatch River tributaries, which approach reported stressful or lethal limits for steelhead (e.g., ≥25°C) (Myrick and Cech 2004). These findings suggest high water temperatures coupled with low flow conditions may regulate juvenile steelhead growth during summer in lower Potlatch River tributaries. Previous temperature monitoring in the Potlatch River was sporadic and generally conducted in conjunction with a specific project (Hand et al. 2020). There is a need to develop a more systematic temperature monitoring plan in the basin to better assess the role of high summer water temperatures in regulating juvenile steelhead growth and how future restoration efforts influence this relationship.

Tributary-scale monitoring data provides valuable information to assess the relationships between juvenile steelhead production and habitat conditions in the basin. This information is critical in understanding the causal link between a habitat treatment and fish response. For example, we documented a positive relationship between juvenile steelhead growth and survival with the amount of wetted habitat in the lower watershed. In addition, we previously documented positive increases in juvenile steelhead growth and survival following a flow supplementation project (Knoth et al. 2021). These findings provide insights into the factors regulating steelhead growth and survival in the lower watershed and validate the need to improve base flow conditions. Similarly, we have documented a positive correlation between juvenile steelhead density and LWD in the upper watershed (Bowersox et al. 2009; Knoth et al. 2021) which highlights the need to improve instream habitat complexity in the drainage. The extent to which fish density mediates juvenile steelhead growth and survival is less clear, likely due to the limited data sets used in the analyses. Our understanding of the relationship between fish density and juvenile steelhead production should improve as we continue to add to the dataset in the future.

We continue to make changes to the project's tributary-scale monitoring framework to strengthen our ability to detect a steelhead response to restoration actions in the drainage. For example, we recently incorporated an additional control area in the upper EFPR to reduce the post-treatment monitoring time needed to detect a response (Uthe et al. 2017). We need to re-run the power analysis to determine the sampling effort needed in each area to reliably detect a significant response. In addition, we are working to establish new PIT-tagging goals for juvenile steelhead in the upper watershed in order to assess changes in juvenile growth and survival in each treatment and control area. As mentioned previously, we will also implement a radio-telemetry project in 2021 to better understand juvenile steelhead habitat requirements in the

EFPR during the summer and winter. These changes should ultimately strengthen our ability to detect a fish response to specific restoration actions in the upper watershed.

RECOMMENDATIONS

1. Work with Potlatch Implementation Group collaborators to develop new analytical methods or monitoring plans to assess the impacts of meadow restoration projects on base flow conditions in the Corral Creek sub-watershed.
2. Develop and implement a systematic water temperature monitoring framework for treatment and control areas in Potlatch River basin.
3. Conduct a power analysis on electrofishing data in the upper watershed to determine the sampling effort needed to detect a statistically significant difference in parr densities in treatment and control tributaries.
4. Implement a radio-telemetry study to better understand the extent to which juvenile steelhead are utilizing the LWD treatment area in the EFPR as critical overwinter habitat.

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TABLES

Table 2. Number of juvenile steelhead (>80 mm) PIT tagged in Potlatch River tributaries for parr-to-smolt survival analyses from 2008-2020. Values in parenthesis indicate number of tagged fish subsequently detected in the hydrosystem.

Tag Year	Big Bear Creek	Little Bear Creek	WFK Little Bear Creek	Pine Creek	East Fork Potlatch River	West Fork Potlatch River
2008	123 (13)	113 (13)	113 (7)	285 (47)	293 (14)	0 (0)
2009	189 (5)	341 (25)	499 (35)	613 (44)	212 (11)	0 (0)
2010	252 (16)	298 (34)	526 (101)	0 (0)	151 (21)	0 (0)
2011	25 (4)	383 (66)	380 (45)	410 (48)	430 (29)	0 (0)
2012	201 (11)	408 (56)	302 (38)	0 (0)	66 (7)	0 (0)
2013	157 (3)	219 (7)	247 (14)	259 (8)	337 (6)	15 (0)
2014	47 (1)	229 (9)	385 (6)	203 (5)	432 (7)	5 (0)
2015	39 (0)	311 (12)	160 (4)	242 (5)	120 (6)	0 (0)
2016	23 (1)	446 (60)	385 (40)	308 (33)	380 (47)	2 (0)
2017	156 (10)	477 (26)	437 (40)	475 (60)	381 (18)	2 (1)
2018	168 (160)	435 (22)	492 (23)	452 (24)	280 (21)	0 (0)
2019	48 (3)	239 (28)	468 (60)	80 (1)	272 (22)	0 (0)
2020	34 (2)	317 (21)	339 (29)	38 (3)	302 (29)	18 (2)

Table 3. Summer-fall growth of juvenile steelhead (≥ 80 mm) in select tributaries in the lower Potlatch River watershed, Idaho from 2014-2020. West Fork Little Bear Creek (WFLBC) and Little Bear Creek (LBC) are treatment tributaries and Pine Creek (PNC) is the control tributary.

Year	Tributary	n	Mean growth (mm)	S.D.	Average time at large (d)	Mean daily growth (mm/d)
2014	PNC	29	8.30	8.00	147	0.057
	WFLBC	26	7.50	7.67	123	0.059
	LBC	0	nd	nd	nd	nd
2015	PNC	17	12.35	16.24	141	0.086
	WFLBC	10	3.70	4.27	132	0.028
	LBC	2	17.00	5.66	138	0.124
2016	PNC	62	5.68	8.36	130	0.044
	WFLBC	44	0.64	4.36	117	0.006
	LBC	77	2.90	5.66	127	0.023
2017	PNC	54	8.26	7.28	133	0.062
	WFLBC	41	1.15	4.77	118	0.009
	LBC	18	1.06	4.18	124	0.008
2018	PNC	29	11.17	6.81	134	0.083
	WFLBC	33	2.91	4.24	116	0.025
	LBC	32	3.75	5.45	141	0.026
2019	PNC	0	nd	nd	nd	nd
	WFLBC	30	4.670	5.020	126	0.041
	LBC	16	2.375	2.705	131	0.018
2020	PNC	0	nd	nd	nd	nd
	WFLBC	29	7.690	4.710	113	0.068
	LBC	51	5.137	4.000	105	0.049

FIGURES

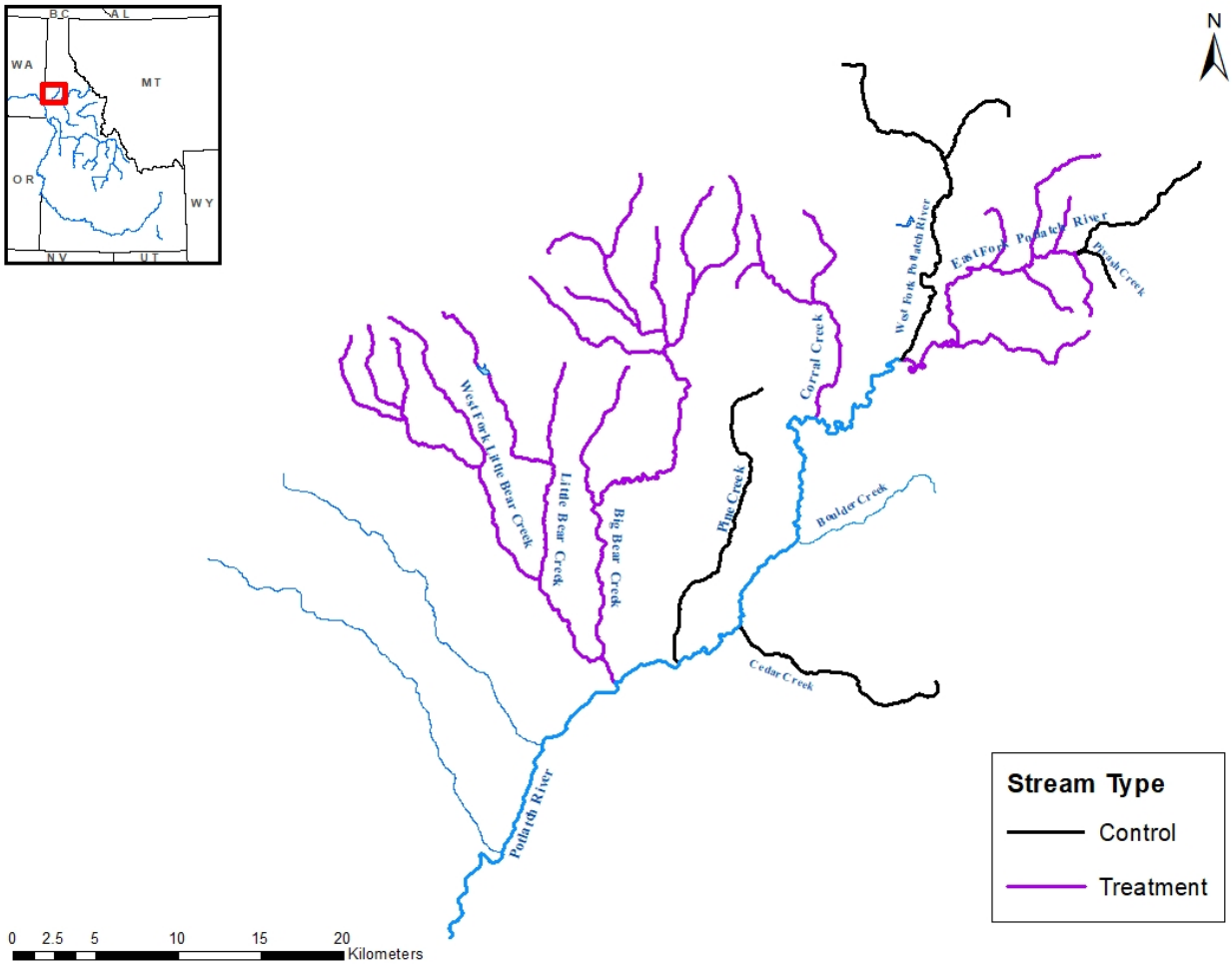

Figure 16. Treatment and control areas within the tributary-scale study design in the Potlatch River basin.

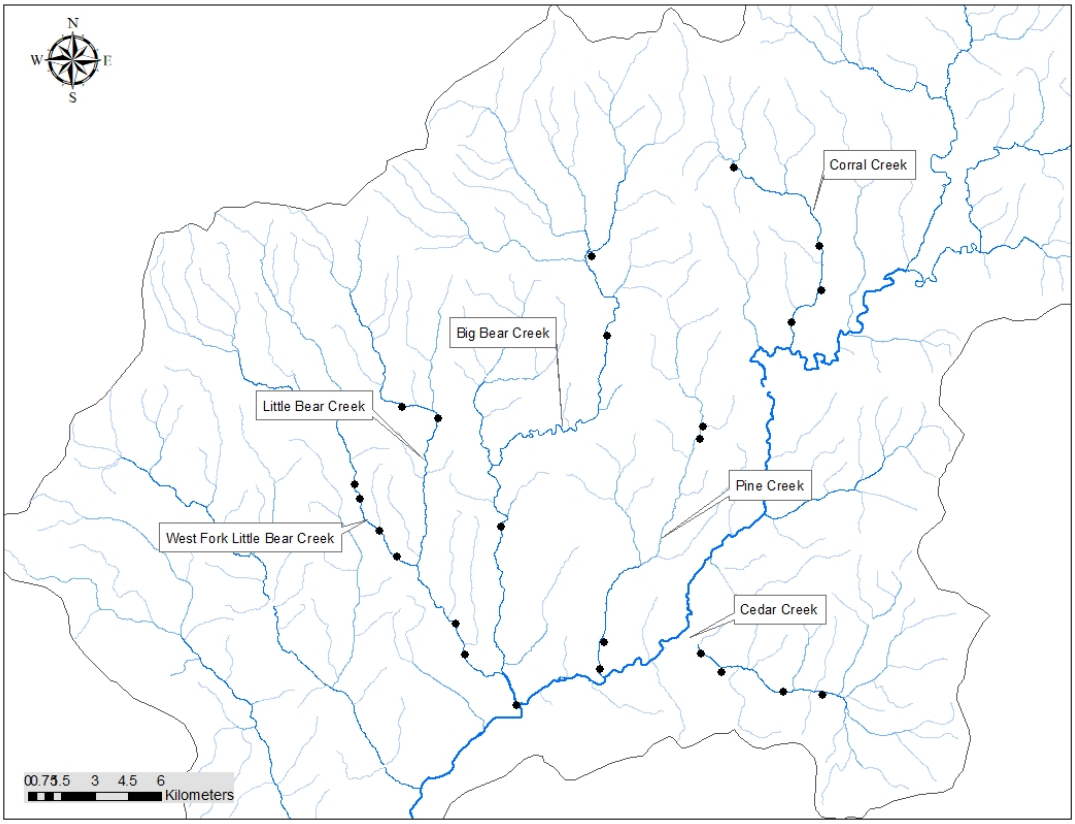

Figure 17. Locations of sites sampled annually 2007-2020 for wetted habitat and pool density in the lower Potlatch River watershed. Treatment tributaries are West Fork Little Bear Creek, Little Bear Creek, Big Bear Creek, and Corral Creek and the control tributaries are Pine Creek and Cedar Creek.

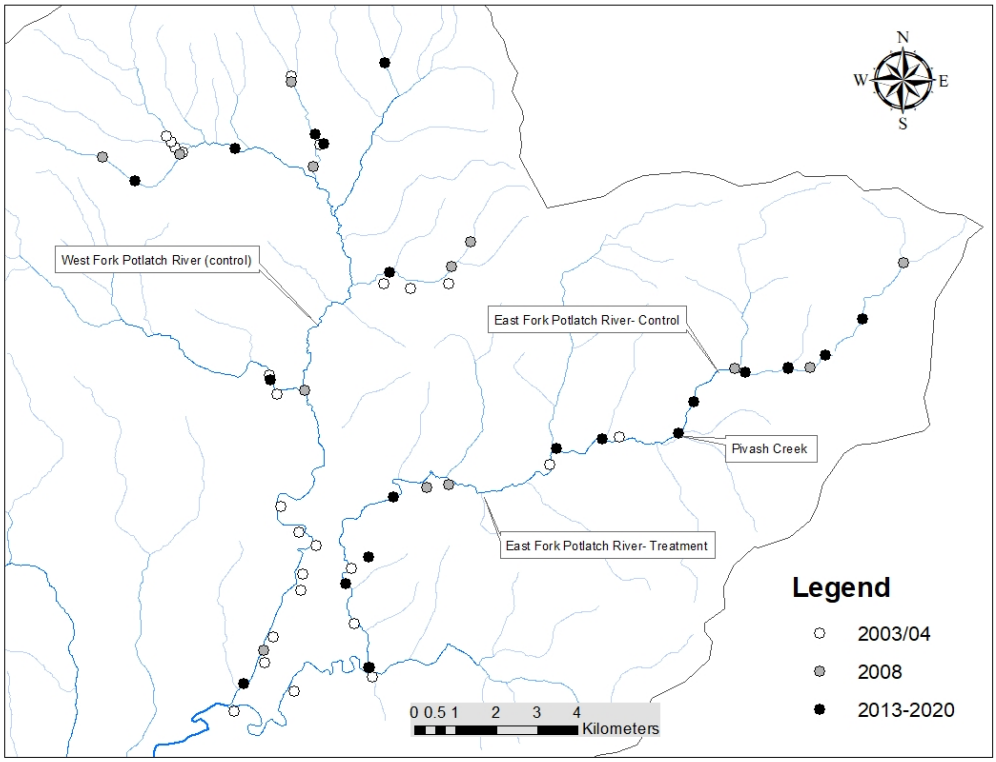

Figure 18. Locations of habitat and electrofishing sites within treatment and control areas in the upper Potlatch River watershed. Sites were surveyed during three time periods (2003/04, 2008, 2013-2020). The treatment area is the East Fork Potlatch River downstream of Pivash Creek and control areas are the West Fork Potlatch River and the East Fork Potlatch River upstream of Pivash Creek.

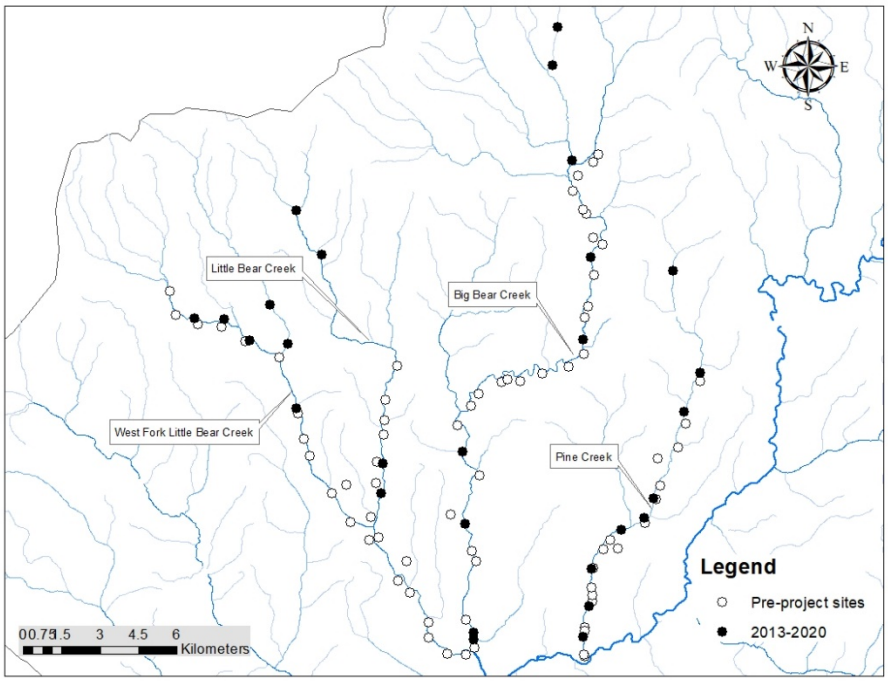

Figure 19. Locations of electrofishing sites within treatment and control tributaries in the lower Potlatch River watershed. Treatment tributaries are West Fork Little Bear Creek, Little Bear Creek, and Big Bear Creek and the control tributary is Pine Creek. Sites surveyed before the current study are distinguished from currently sampled sites.

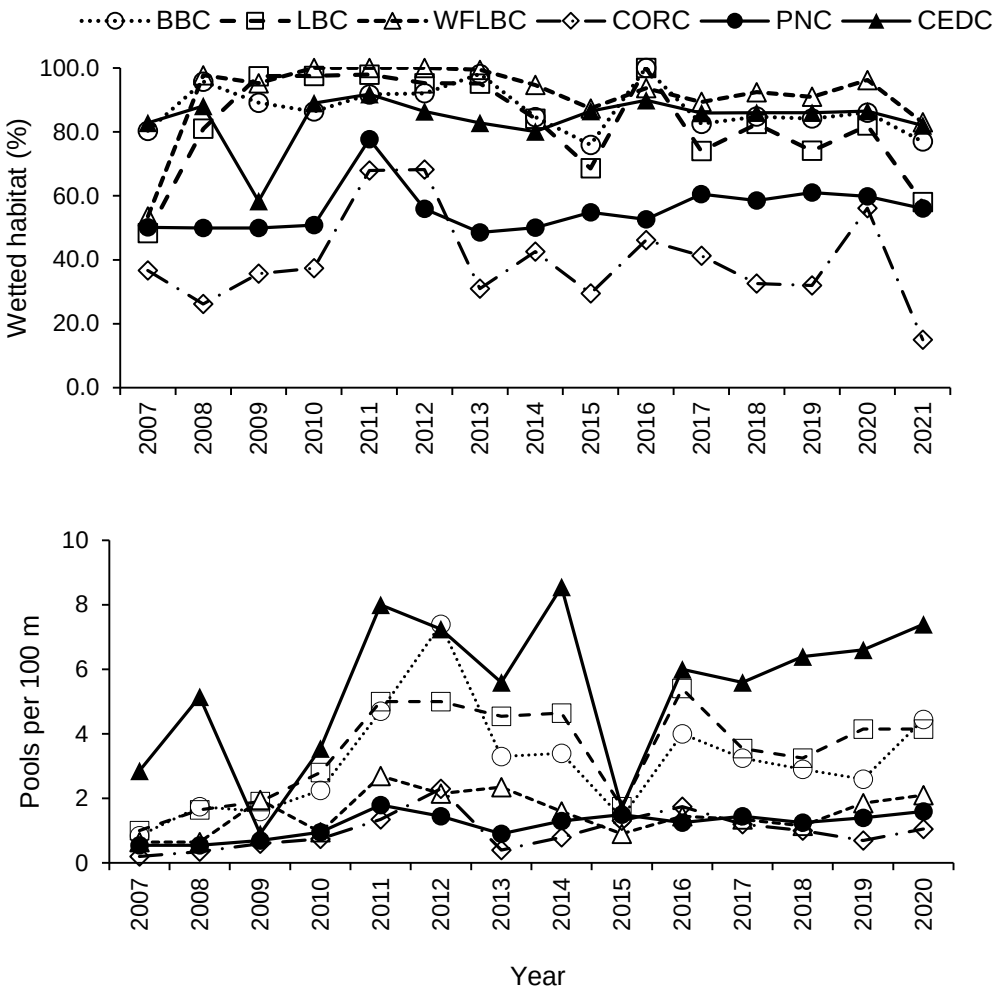

Figure 20. The amount of wetted habitat (panel A) and pool density (panel B) in treatment and control tributaries in lower Potlatch River watershed during 2007-2020. Treatment tributaries include Big Bear Creek (BBC), Little Bear Creek (LBC), West Fork Little Bear Creek (WFLBC), and Corral Creek (CORC) and are indicated by dashed lines and open symbols. Control tributaries include Pine Creek (PNC) and Cedar Creek (CEDC) and are indicated by solid lines and symbols. Restoration treatments began in 2013.

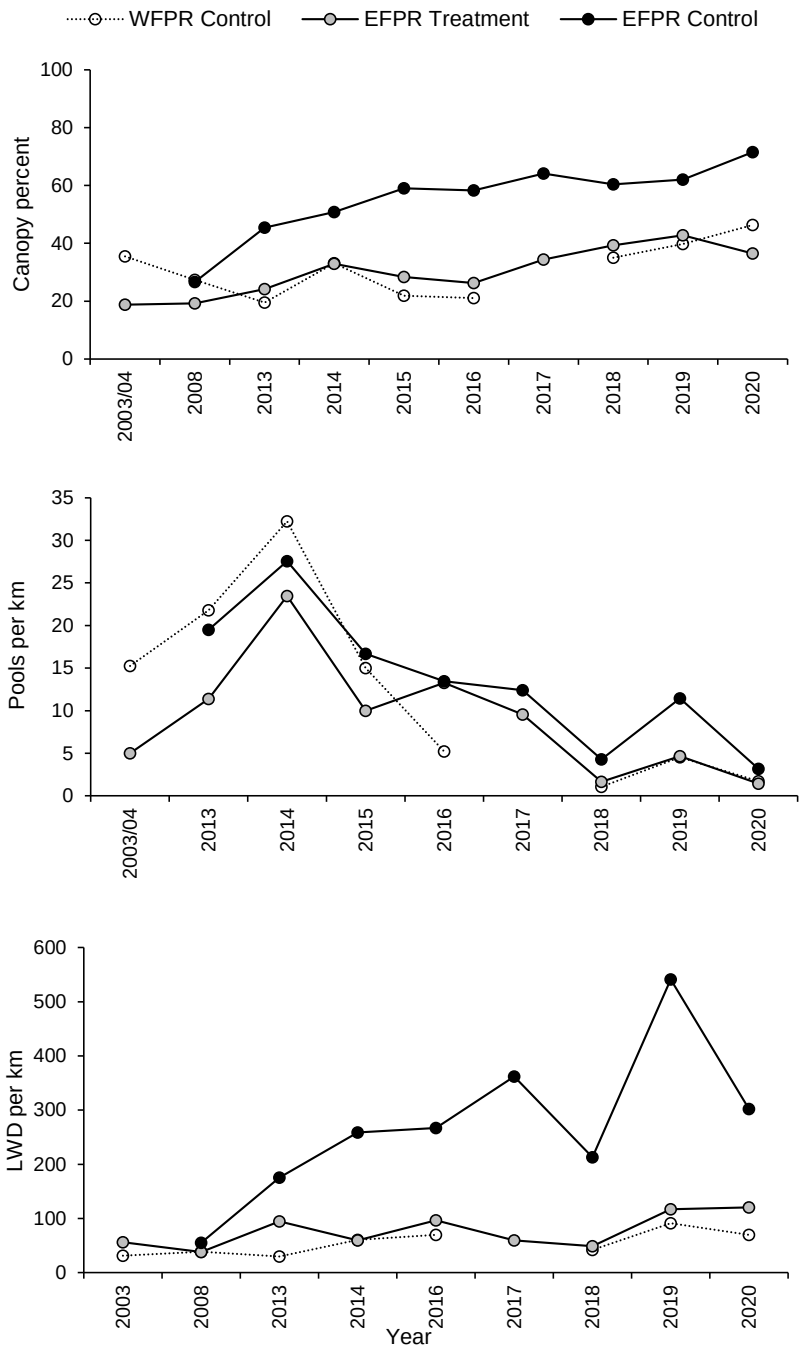

Figure 21. Canopy cover, pool density, and large wood density (LWD) within the treatment area (EFPR Treatment) and control areas (WFPR and EFPR Control) in the upper Potlatch River watershed during 2003–2020. Restoration treatments began in 2009.

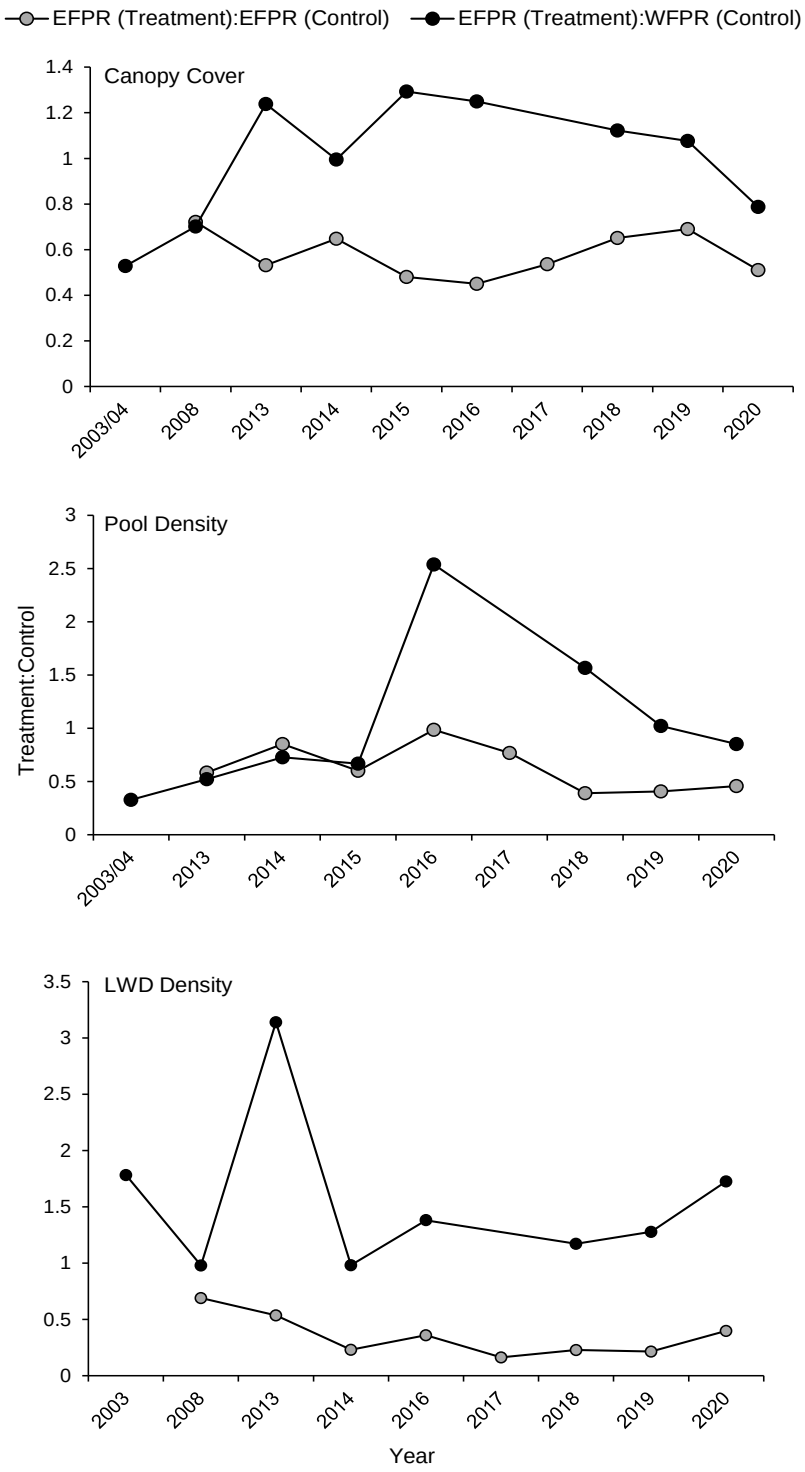

Figure 22. Habitat metric ratio values (treatment:control) for treatment (EFPR Treatment) and control areas (EFPR Control and WFPR Control) in the upper Potlatch River watershed during 2003-2020. The upper, middle, and lower panels indicate canopy cover, pool density, and large woody debris (LWD) density relationships respectively.

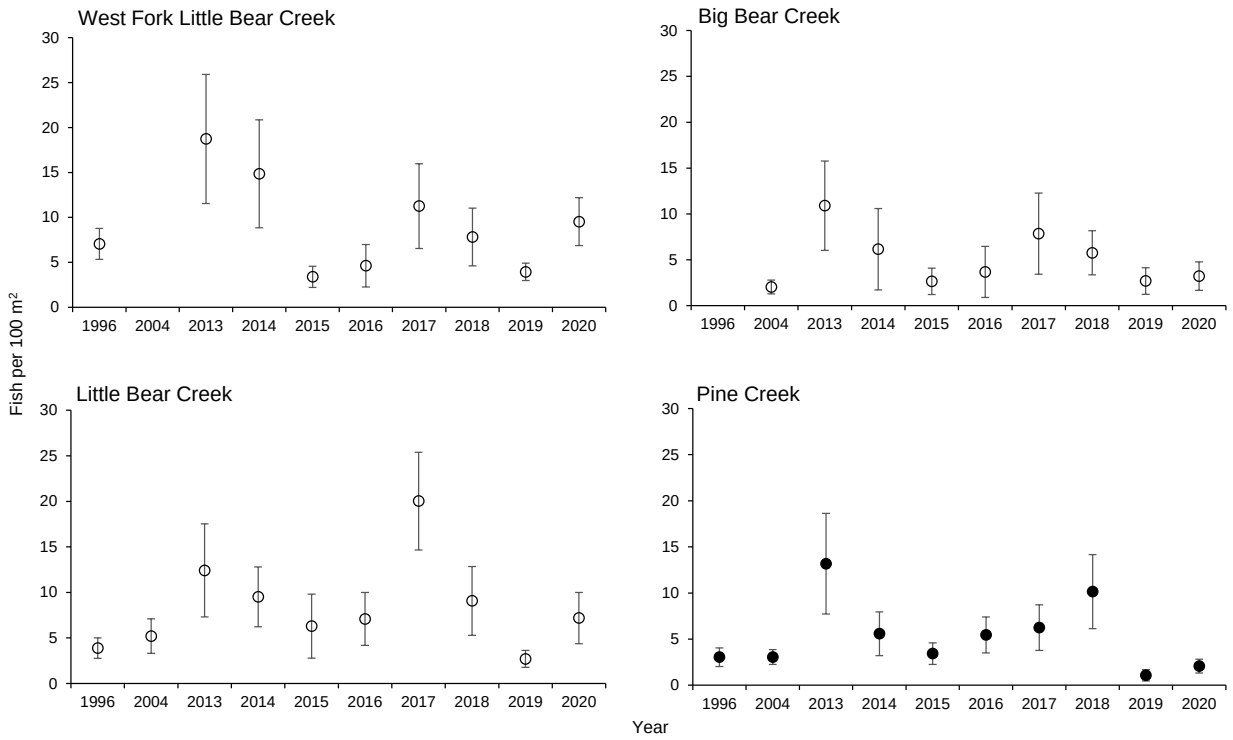

Figure 23. Density of juvenile steelhead ≥80 mm based on single-pass electrofishing surveys in Big Bear Creek, Little Bear Creek, West Fork Little Bear Creek (treatment tributaries) and Pine Creek (control tributary) in the lower Potlatch River watershed during 1996-2020.

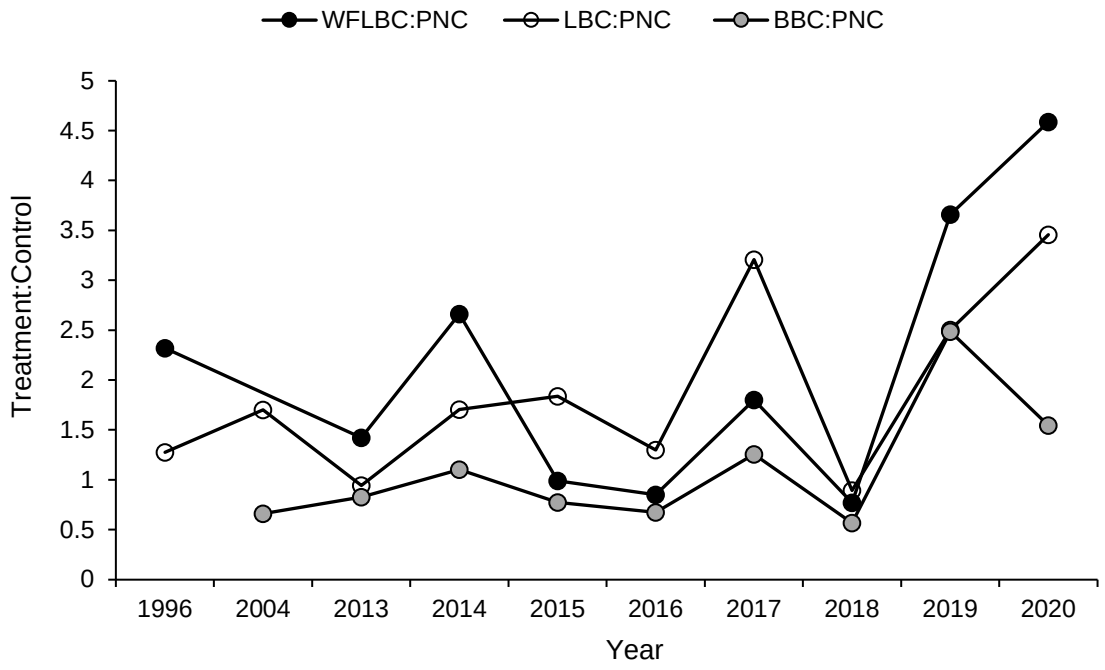

Figure 24. Juvenile steelhead density ratio values (treatment:control) for treatment tributaries West Fork Little Bear Creek (WFLBC), Little Bear Creek (LBC), and Big Bear Creek (BBC) and the control tributary Pine Creek (PNC) in the lower Potlatch River watershed during 1996-2020.

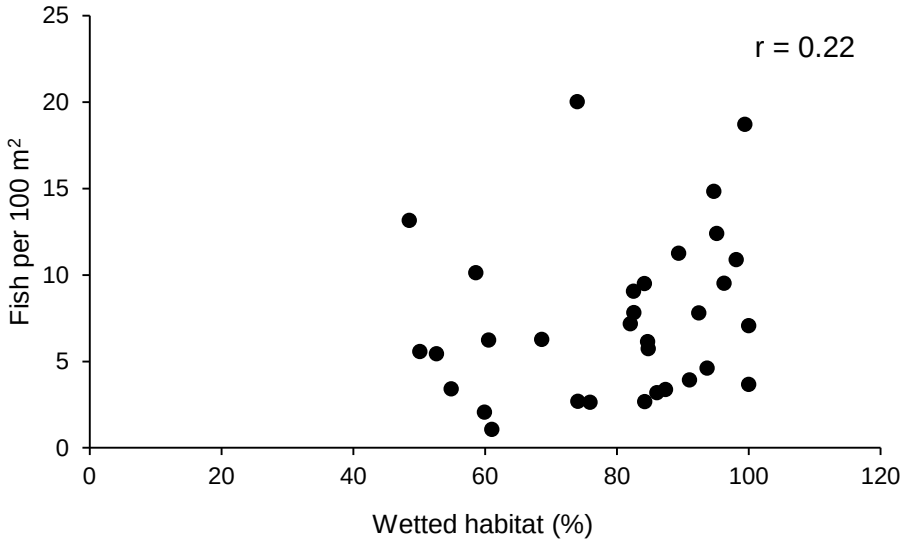

Figure 25. Relationship between juvenile steelhead density and percent wetted habitat in lower Potlatch River watershed.

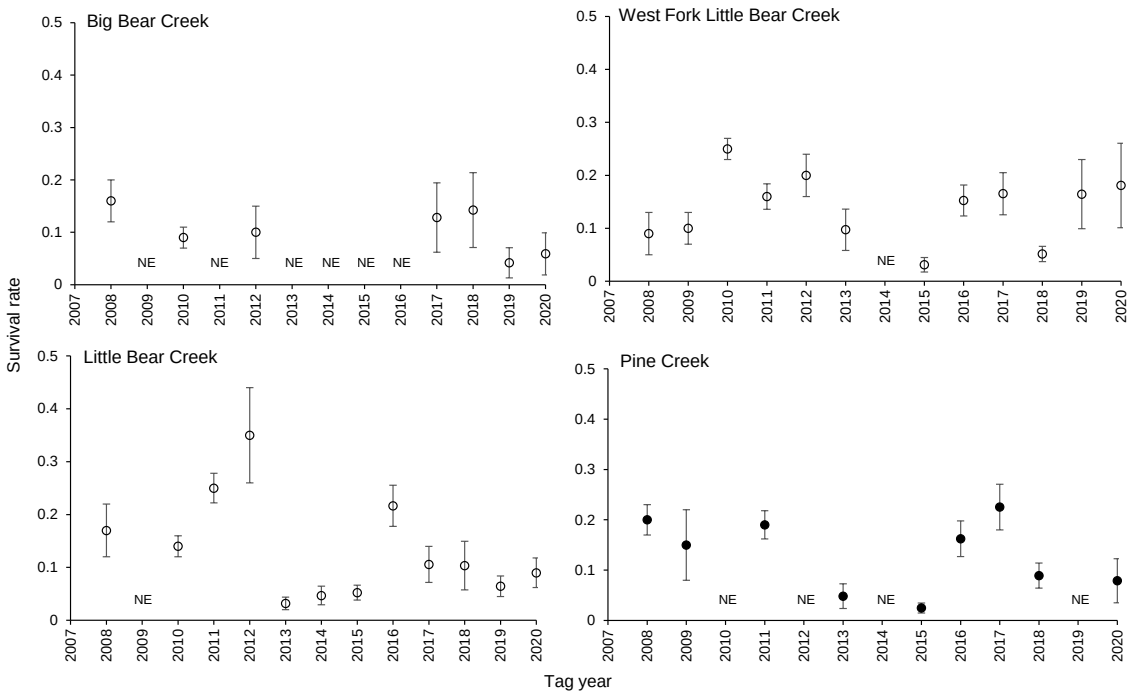

Figure 26. Apparent survival to Lower Granite Dam from Big Bear Creek, Little Bear Creek, and the West Fork Little Bear Creek (treatment tributaries) and Pine Creek (control tributary) in the lower Potlatch River watershed during 2008-2020. No estimate (NE) indicates insufficient detections to generate an estimate.

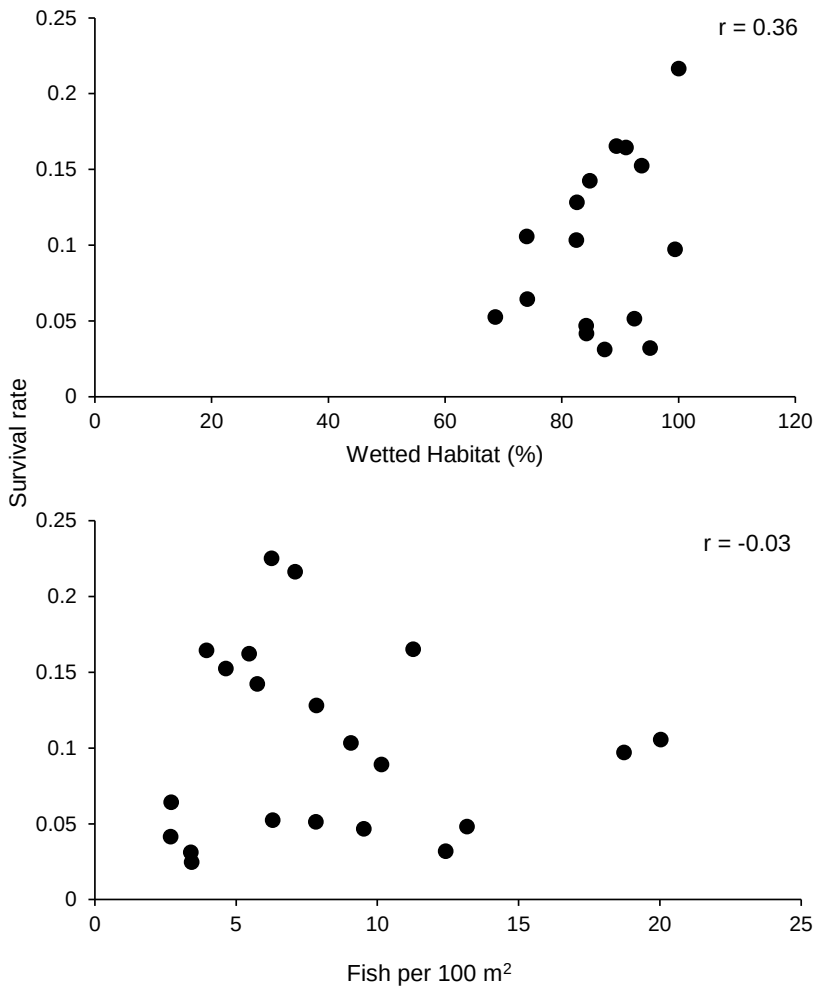

Figure 27. Relationships between juvenile steelhead apparent survival and the amount of wetted habitat (top panel) and juvenile steelhead density (bottom panel) in the lower Potlatch River watershed.

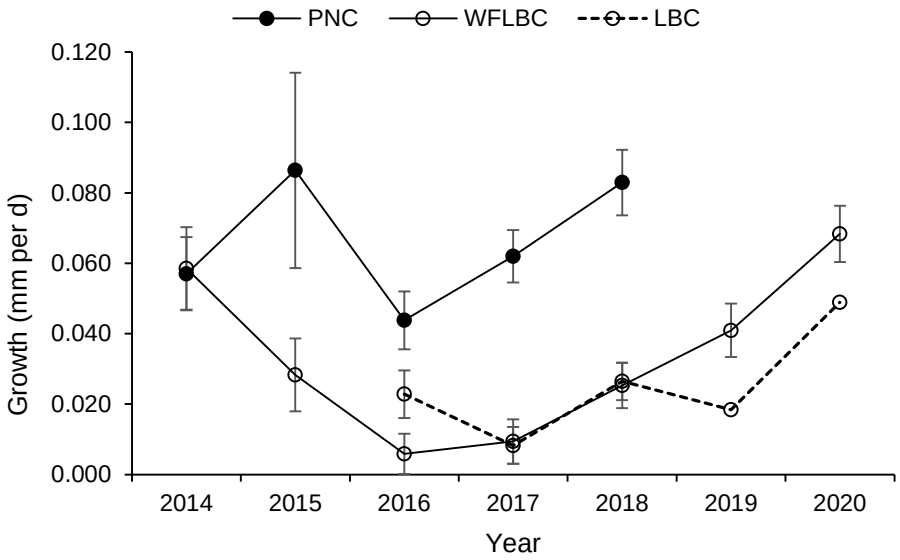

Figure 28. Summer-fall growth rates (mm per d) of juvenile steelhead (≥80 mm) in select treatment tributaries in the lower watershed of the Potlatch River, Idaho from 2014-2020. West Fork Little Bear Creek (WFLBC) and Little Bear Creek (LBC) are treatment tributaries and Pine Creek (PNC) is the control tributary.

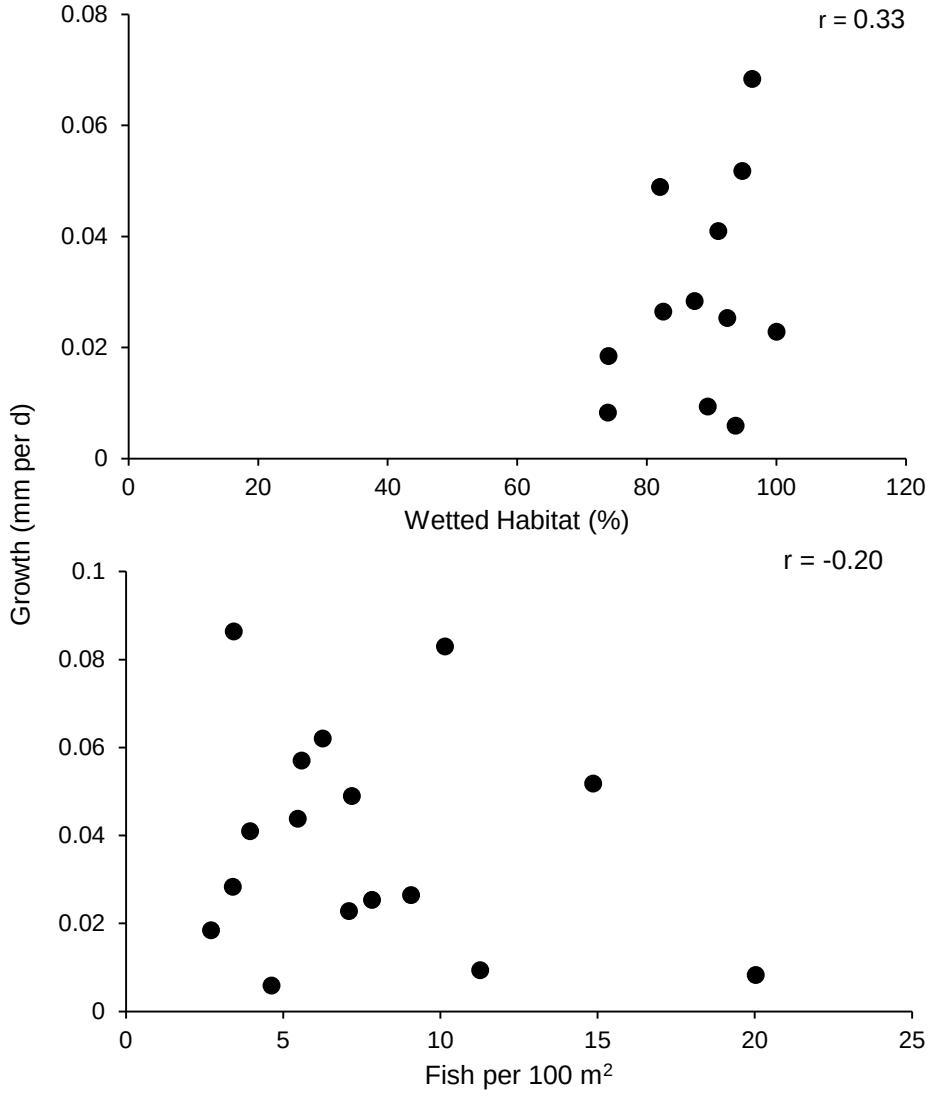

Figure 29. Relationships between juvenile steelhead growth and the amount of wetted habitat (Top panel) and juvenile steelhead density (Bottom panel) in the lower Potlatch River watershed.

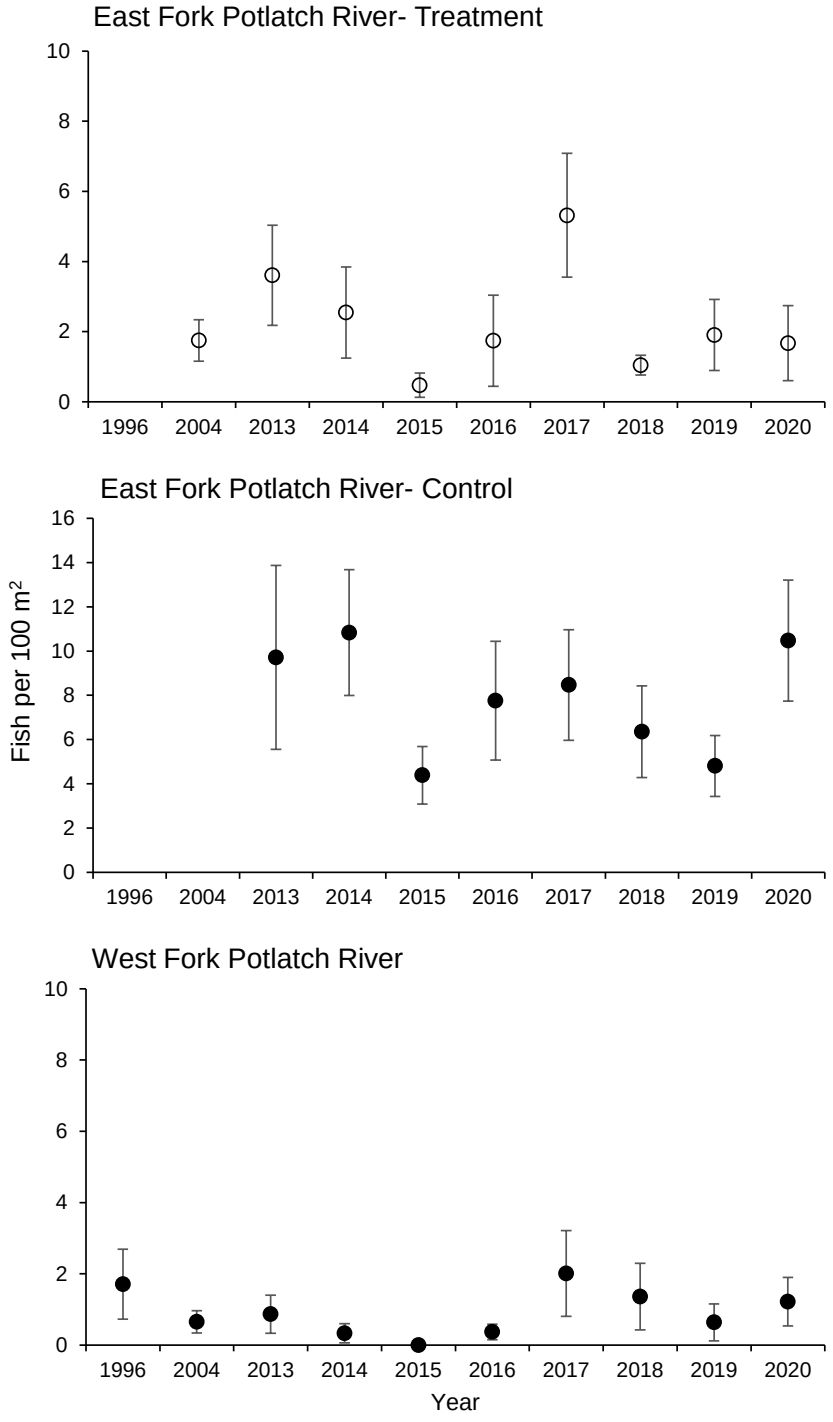

Figure 30. Density of juvenile steelhead ≥80 mm based on single-pass electrofishing surveys in the East Fork Potlatch River treatment area, the East Fork Potlatch River control area, and the West Fork Potlatch River control tributary in the upper Potlatch River watershed during 1996–2020.

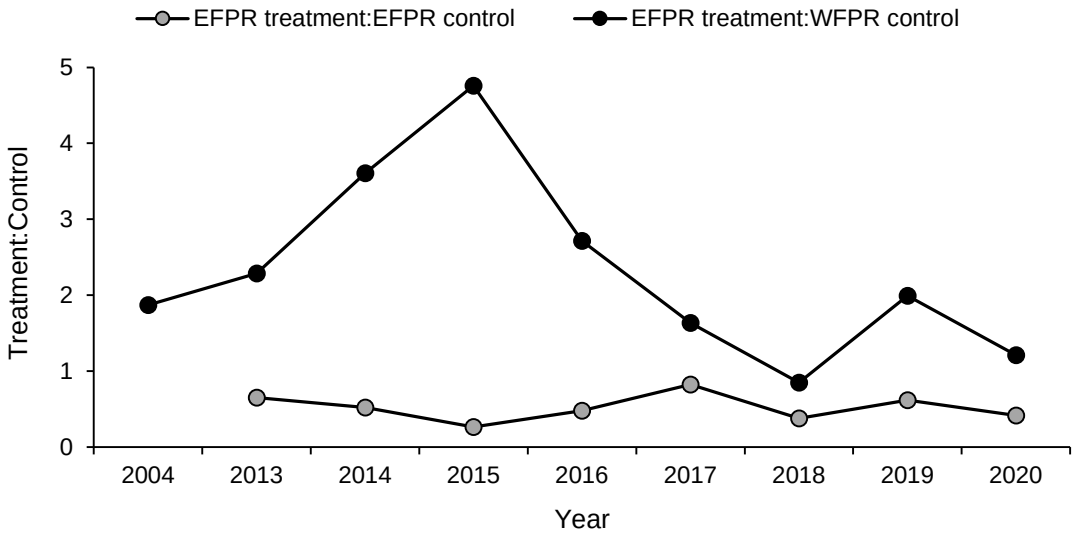

Figure 31. Juvenile steelhead density ratio values (treatment:control) for East Fork Potlatch River treatment area (EFPR treatment), the East Fork Potlatch River control area (EFPR control), and West Fork Potlatch River control area (WFPR control).

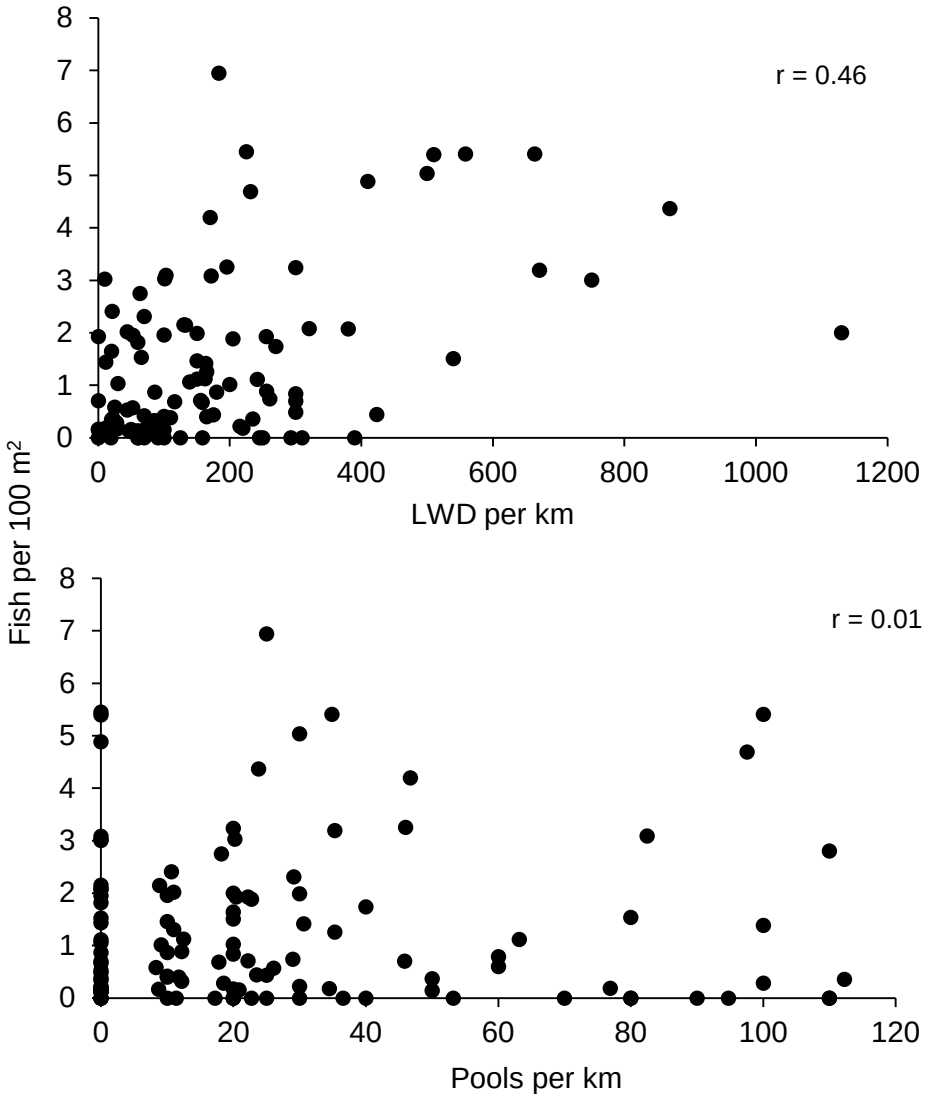

Figure 32. Relationships between juvenile steelhead density and LWD density (Top panel) and pool density (Bottom panel) in the upper Potlatch River watershed.

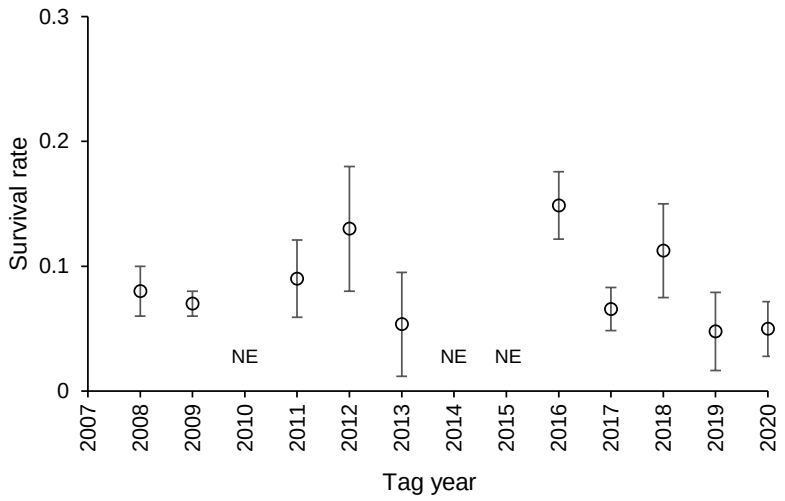

Figure 33. Apparent survival to Lower Granite Dam of juvenile steelhead tagged upstream in the East Fork Potlatch River during 2008-2020. NE = no estimate.

CHAPTER 3: EVALUATION OF THE BIG MEADOW CREEK CULVERT ENHANCEMENT PROJECT

ABSTRACT

A primary habitat restoration strategy in the lower Potlatch River watershed is to improve fish passage at full or partial barriers to expand juvenile steelhead (*Oncorhynchus mykiss*) rearing habitat. Big Meadow Creek (BMC) is the main tributary to the West Fork of Little Bear Creek, which contains the highest rearing densities of juvenile steelhead in the Potlatch River. However, a highway culvert at the mouth of BMC restricted access to an additional 10 km of spawning and rearing habitat. In 2018, the culvert was modified to enhance steelhead passage into BMC. Prior to culvert improvements, the population upstream of the culvert was primarily resident *O. mykiss* while mostly anadromous *O. mykiss* were found downstream of the culvert in the WFLBC. We analyzed the genetic composition of juvenile steelhead captured upstream of the culvert to assess the effectiveness of the BMC culvert modification. With improved fish passage, we hypothesized that the genetic composition of juvenile *O. mykiss* upstream of the culvert would shift from resident to more anadromous characteristics. Genetic samples were collected from juvenile *O. mykiss* in BMC in 2018 (pretreatment), and in 2019 and 2020 (post-treatment) and then compared against steelhead reference samples. We examined potential changes in genetic composition across years using a series of complimentary genetic approaches including analyses of genetic diversity, ancestry, and structure. We observed an increase in genetic diversity and ancestry across years with anadromous signals becoming more dominant upstream of the culvert. With few exceptions, juveniles in 2018 were attributed to the 'resident' *O. mykiss* cluster. In 2019, an increased proportion of juveniles attributed partially or wholly to the anadromous *O. mykiss* cluster and by 2020 the majority of juveniles sampled were placed in the anadromous cluster. Anadromous steelhead that were better able to access BMC due to the culvert improvements are the likely source for the shift in genetic composition of juveniles from resident to anadromous characteristics. The results indicate that modifications to the BMC culvert were successful in enhancing passage of steelhead into BMC.

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INTRODUCTION

Habitat restoration efforts are underway to enhance the production and productivity of wild steelhead (*Oncorhynchus mykiss*) in the Potlatch River. Steelhead production in the lower Potlatch River watershed is highly density dependent (Uthe et al. 2017; Knoth et al. 2021) and the primary factors limiting production are low summer base flows and full or partial barriers that limit fish passage and restrict the amount of available juvenile rearing habitat (Johnson 1985; Bowersox and Brindza 2006). Fish passage barriers exist on nearly every major tributary in the lower watershed, most of which are road culverts upstream of canyon reaches. A primary restoration strategy is to expand juvenile steelhead rearing habitat by removing or modifying barriers for improved fish passage. Previous evaluations of habitat projects in Idaho found that barrier removals have had the largest effect on increasing anadromous fish production (Rich and Petrosky 1994).

Big Meadow Creek (BMC) is the main tributary to the West Fork of Little Bear Creek (WFLBC), both of which are in the lower Potlatch River watershed. Previous electrofishing surveys indicate the highest densities of juvenile steelhead in the lower watershed occur in the WFLBC downstream of the confluence with BMC (Uthe et al. 2017; Knoth et al. 2021). However, a 52 m long cement lined culvert near the mouth of BMC acted as a barrier to adult and juvenile steelhead passage and restricted access to an additional 10 km of spawning and rearing habitat in BMC. Velocities in the culvert were impassable during high flows in the spring, and flows were low enough that the culvert was not passable due to the lack of structure and depth in the summer. In the fall 2018, passage at the BMC culvert was enhanced by installing steel baffles inside the culvert to slow velocities by dissipating energy during high flows and to increase water depth in the culvert at low flows (Figure 34). Life cycle modeling simulations indicated this project could potentially increase smolt production in the Big Bear Creek (BBC) watershed by >2,200 fish (Uthe et al. 2017).

Oncorhynchus mykiss exhibits a diversity of life history types, including a freshwater resident form (Rainbow Trout) and an anadromous form (steelhead) and both forms have been observed in the Potlatch River. Landscape features such as high gradients and barriers can influence the distribution of resident and anadromous *O. mykiss* forms, where higher elevation sites tend to be occupied primarily by the resident form (Narum et al. 2008). Prior to the culvert modifications, juvenile *O. mykiss* densities in BMC were on average four times less then juvenile densities downstream in the WFLBC (unpublished data). Furthermore, trends in juvenile *O. mykiss* densities in BMC upstream of the culvert did not track closely with trends in juvenile *O. mykiss* densities downstream of the culvert in WFLBC, suggesting different factors regulate the abundance of the two groups. Based on these findings we inferred the *O. mykiss* group residing upstream of the BMC culvert were predominantly resident fish and hypothesized we would observe a shift in the genetic composition of BMC *O. mykiss* samples (from resident to anadromous form) following the modification of the passage barrier in 2018. We examined this hypothesis using genotypes of juveniles collected in BMC analyzed in three complementary ways based on different assumptions.

METHODS

Sample Collection

We collected juvenile *O. mykiss* in BMC during late summer through early fall (2018- 2020) using backpack electrofishing techniques. We collected samples at seven sites upstream of the

culvert in BMC that were distributed, on average, 1.38 km apart starting at the culvert and going upstream to ensure spatial distribution of samples collected (Figure 35). Each site was approximately 200 m in length. We measured (FL; mm) and collected a ~1 cm^2 size tissue sample from the caudal fin of each *O. mykiss* sampled. The fin clip was stored on Whatman chromatography paper until analysis (LaHood et al. 2011). Samples collected in 2018 were pretreatment samples (prior to the culver modifications) and samples collected in 2019 and 2020 were post-treatment samples. Reference anadromous *O. mykiss* genetic samples were collected from adult steelhead previously captured at weirs in the Potlatch River basin. Samples from the weirs had previously been genotyped to document the genetic structure of steelhead throughout the Snake River basin for purposes of genetic stock identification (Ackerman et al. 2014; Matala et al. 2014).

Laboratory Analyses

Laboratory protocols for DNA extraction, amplification, and SNP genotyping are detailed in Section 2 of Vu et al. (2015). Samples were processed at Idaho Department of Fish and Game's Eagle Fish Genetics Lab (EFGL) in Eagle, Idaho and individuals were genotyped at a 368 loci panel described in Hargrove et al. (2021). Of the 368 loci, 334 were retained for downstream analysis as they are putatively neutral (i.e., not under selection) and amplified well across all samples. Following SNP amplification, only samples with >90% complete genetic data were retained for downstream analysis.

Summary statistics for BMC collections including the number of alleles (N_A), observed heterozygosity (H_O), and expected heterozygosity (H_E) were generated using GENALEX v6.5 (Peakall and Smouse 2012). Higher values of these metrics are indicative of greater levels of genetic variability within a population, and lower values may indicate decreased genetic diversity attributable to various factors (population bottlenecks, reduced meta-population dynamics). Pairwise F_{ST} (Weir and Cockerham 1984) is an indicator of the level of differentiation between populations and varies from 0 (i.e., both populations share a common set of alleles) to 1 (i.e., no alleles are shared between populations). Estimates of pairwise F_{ST} and allelic differentiation were computed using the *Fst* and *test_diff* functions, respectively, using the algorithms of Genepop (Rousset 2008) as implemented in the R package (R Core Team 2020) 'genepop'. Allelic differentiation was assessed using 10,000 dememorization steps, 100 batches, and 5,000 iterations.

We used principal component analysis (PCA) as a method to visualize patterns in allelic ancestry among collections. Because PCA is a multivariate approach, it is free from population genetic model assumptions (Hardy-Weinberg equilibrium, linkage disequilibrium) and represents a complimentary approach to Bayesian clustering (see STRUCTURE analysis below). The PCA was performed using the 'ade4' and 'adegenet' packages (Thioulouse et al. 1997; Jombart 2008) in R (R Core Team 2020). Missing values were replaced by zero. The graphical output displayed the absolute variance (i.e., eigenvalues) explained by each of the principal components (PCs), with the *x*- and *y*-axes representing the percentage of total variance explained by PC1 and PC2, respectively. Samples that clustered together were interpreted as sharing a common ancestry (e.g., similar genotypes), and the distribution of points provided an indication of diversity levels (e.g., tighter clusters denote lower diversity than do widely dispersed clusters).

We used STRUCTURE 2.3.4 (Pritchard et al. 2000) to infer resident or anadromous heritage using genetic Bayesian clustering methods. Default parameters of admixture and correlated allele frequencies were used to account for recent gene flow among populations and allow some flexibility for linkage disequilibrium within populations. These default settings are most

flexible for dealing with real biological phenomena (Pritchard et al. 2009) and by allowing for admixture, individual fish can be fractionally allocated to one or both clusters (i.e., pure steelhead, pure resident, or some combination thereof). We assumed a total of two potential genetic clusters (*K*) corresponding to resident and anadromous life history strategies likely present in the focal tributaries. Previous work has shown that collections of steelhead in the Potlatch River, its tributaries, and nearby drainages (e.g., Lapwai Cr) belong to a singular genetic stock (Hargrove et al. 2021), justifying the use of $K = 2$. A burn-in length of 100,000 with 200,000 repeats of the Monte Carlo Markov Chain (MCMC) was used to assess genetic structure in the data and 10 repetitions were performed. Cluster membership values were averaged across runs.

RESULTS

We collected a total of 153 juvenile *O. mykiss* samples from BMC (average = 51 samples per year, range = 50 - 53; Table 4). The number of samples collected at each site varied by year, but similar numbers of samples were collected from 3 of 7 sites for all years. Genotyping success rate was 94.8% resulting in 145 samples available for genetic analysis (Table 5).

There was an increase in the number of alleles observed in BMC juveniles through time (Table 5). The number of alleles ($N_a \pm SE$) was 1.73 ± 0.03 in 2018, 1.87 ± 0.03 in 2019, and 1.98 ± 0.03 in 2020. Similar trends were noted for observed heterozygosity and expected heterozygosity (Table 5).

Levels of genetic differentiation (F_{ST}) among BMC collections increased through time (Table 6). In other words, differentiation was lowest for 2018 × 2019 samples (0.006), greater for 2019 × 2020 samples (0.070), and greatest for 2018 × 2020 (0.102) samples. Important to note is that all pairwise comparisons among BMC samples were statistically significant (Table 6). Levels of genetic differentiation between BMC and steelhead collections decreased through time. Specifically, mean values of F_{ST} were lowest when comparing 2020 juveniles from BMC and steelhead collections (0.034). Mean values of F_{ST} were both higher when comparing 2019 juveniles (0.087) and 2018 juveniles (0.115) from BMC against collections of steelhead.

Principal component analysis revealed a pronounced shift in genetic ancestry across sample years (Figure 36). There was minimal overlap between BMC juvenile samples and adult steelhead samples in 2018. There was a higher degree of overlap between the two sample groups in 2019, and the BMC juvenile samples shared an overlapping distribution with adult steelhead samples from nearby tributaries in 2020. The overlapping nature of 2020 juveniles and adult steelhead were indicative of a shared genetic ancestry.

Clustering results indicated that steelhead from reference collections were assigned to cluster 2 (hereafter ‘steelhead genetic cluster’) at high rates (mean assignment = 0.986, SE = 0.002; Figure 37). In contrast, the assignment of BMC juveniles to the steelhead cluster ranged dramatically across years, and was lowest in 2018 (mean assignment across individuals = 0.034, SE = 0.015) and highest in 2020 (mean assignment = 0.832, SE = 0.041). The distribution of steelhead ancestry within BMC juveniles varied considerably by collection year (Figure 37). In 2018, the majority of juveniles (45/48) had steelhead ancestry values <0.1 with only four individuals having low to moderate steelhead influence. In contrast, there were six (of 46) individuals in 2019 that had >0.9 of their genome assigned to the steelhead genetic cluster. Lastly, in 2020, the majority (32/50) of samples from BMC had >0.9 of their genome assigned to the steelhead genetic cluster. Values suggestive of hybridization between resident and anadromous

O. mykiss forms were low throughout the study, but greater in 2020 than in previous years (Figure 38).

DISCUSSION

We used genetic techniques to assess whether steelhead occupied newly available habitat in BMC following the modification of a partial barrier to upstream migration. We showed that prior to barrier modification, genetic signatures of anadromous steelhead were almost entirely absent and levels of genetic diversity were low relative to subsequent sample years. Levels of genetic differentiation between upstream our samples and reference steelhead collections decreased through time, and both multivariate analysis and Bayesian clustering indicated a transition from a genetic signature of resident redband to anadromous *O. mykiss*. Overall, results of our study indicate that modifications to the BMC culvert were successful in enhancing passage of adult steelhead and promoting connectivity to additional spawning and juvenile rearing habitat.

The present study adds to our understanding of *O. mykiss* life history diversity in the Potlatch River. Anadromous and resident *O. mykiss* forms are often sympatric, commonly interbreed, and their offspring may adopt either form (Christie et al. 2011; Courter et al. 2013; Sloat and Reeves 2014). Gene flow readily occurs between sympatric resident and anadromous *O. mykiss* types (Doornik et al. 2013) and the reproductive contribution of resident life history forms may be critical in maintaining genetic diversity in anadromous salmonid populations (Courter et al. 2013). Furthermore, resident *O. mykiss* forms located upstream of a barrier may pass downstream and contribute to the anadromous gene pool (Doornik et al. 2013; Bowersox et al. 2016). Bowersox et al. 2016 documented limited reproductive exchange between resident and anadromous groups in a study in the BBC drainage. We identified low levels of resident x anadromous *O. mykiss* introgression in our study, but introgression levels increased after passage at the culvert improved. Understanding the relationship among the life history types is important for the management of the species.

Evaluating fish response to passage barrier projects typically requires pretreatment and post treatment monitoring of fish movement or abundance to evaluate the effectiveness of a project (Roni 2005). Methods used to quantify fish passage typically involve the use of radio-telemetry and/or PIT-tag technology to monitor movement of individual fish (Koehn 2012). For example, we documented an upstream expansion of adult steelhead spawning distribution using a combination of radio-telemetry and spawning ground surveys following the removal of multiple passage barriers on the West Fork Little Bear Creek (Uthe et al. 2017). However, these techniques were not practical to monitor individual fish passage in the present study due to the low numbers of adult steelhead and subsequent low probability of re-locating tagged fish. The presence of a resident *O. mykiss* group upstream of the culvert provided a unique opportunity to utilize an indirect, genetic approach to assess the effectiveness of the culvert modification in enhancing steelhead passage. Similar techniques have been used to examine the impacts of large-scale dam removals on the population genetics of *O. mykiss* in the Pacific Northwest (Fraik et al. 2021). In addition, a similar genetic approach was used to examine changes in Brook Trout *Salvelinus fontinalis* population structure following culvert removal projects in central Appalachian Mountain headwater streams (Wood et al. 2018). Results from our study demonstrated the genetic approach was an effective technique to evaluate steelhead passage into BMC.

Enhancing steelhead passage at the BMC culvert was a key first step in the restoration strategy of BMC. Re-connecting isolated high quality fish habitat should be the initial restoration

focus in a watershed (Roni et al. 2002). In addition to the BMC culvert modification, three additional road culverts upstream of the highway culvert were also replaced, effectively opening access to the entire 10 km of the BMC drainage. The next step in the BMC restoration plan is to secure and maintain perennial flow throughout the entire drainage. Approximately 4-5 km of BMC becomes intermittent during late summer. Previous studies in BMC and nearby Little Bear Creek have documented the feasibility of maintaining perennial flow in these drainages through water releases from headwater reservoirs (Brooks and Treasure 2014; Hand et al. 2020; Uthe et al. 2017). This approach is currently being pursued to address flow conditions in Little Bear Creek and will likely be used to address flow conditions in BMC. Once perennial flow is secured, the final step will focus on improving riparian function, floodplain access, and in-stream complexity in degraded reaches. Together, these projects will help restore the production potential of BMC and contribute to increased steelhead smolt production in the BBC watershed (Uthe et al. 2017).

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TABLES

Table 4. Number and mean length (fork length, mm) of juvenile *Oncorhynchus mykiss* samples collected from Big Meadow Creek, Idaho from 2018-2020. nd = no data.

		2018		2019		2020	
Site	**Distance from culvert (km)**	**n**	**Mean fork length (mm)**	**n**	**Mean fork length (mm)**	**n**	**Mean fork length (mm)**
Site 0	0	0	nd	4	68	10	107
Site 1	1.9	0	nd	0	nd	10	78
Site 2	3.47	6	144	16	95	10	64
Site 3	4.54	8	188	4	123	10	149
Site 4	5.85	13	165	14	135	10	124
Site 4.5	6.94	15	101	0	nd	0	nd
Site 5	8.29	8	120	14	103	0	nd

Table 5. Details associated with juvenile *Oncorhynchus mykiss* collections from Big Meadow Creek, Idaho including the number of samples collected (n), the proportion of those successfully genotyped, and values of genetic diversity including number of alleles (N_A), observed heterozygosity (H_O) and expected heterozygosity (H_E). Values in parentheses correspond to standard error estimates.

Collection year	**n**	**Genotyped (% successful)**	**N_A**	**H_O**	**H_E**
2018	50	49 (98%)	1.73 (0.03)	0.20 (0.01)	0.22 (0.01)
2019	53	46 (86%)	1.87 (0.03)	0.23 (0.01)	0.24 (0.01)
2020	50	50 (100%)	1.98 (0.03)	0.26 (0.01)	0.27 (0.01)

Table 6. Levels of genetic differentiation (F_{ST}) among collections of *Oncorhynchus mykiss* from Big Meadow Creek and adult steelhead samples collected from tributaries to the Potlatch River. Values of F_{ST} are color coated such that higher levels of differentiation are indicated by red and low values are highlighted green. Values below the dashed lines represent F_{ST} values and p-values are shown above the diagonal. Asterisks (*) indicate that the genotypic differentiation (exact G test) was highly significant and the combined p-value (Fisher's method) could not be calculated. P-values in italics denote values significant at the alpha level of 0.05. Steelhead collections are identified by the abbreviation Sthd.

	Big Meadow Cr. - 2018	**Big Meadow Cr. - 2019**	**Big Meadow Cr. - 2020**	**EF Potlatch (Sthd)**	**Little Bear Cr. (Sthd)**	**WF Potlatch (Sthd)**	**Big Bear Cr (Sthd)**
Big Meadow Cr. - 2018	---	*	*	*	*	*	*
Big Meadow Cr. - 2019	0.0058	---	*	*	*	*	*
Big Meadow Cr. - 2020	0.1018	0.070	---	*	*	*	*
EF Potlatch (Sthd)	0.1107	0.0846	0.0356	---	*	0.610	*0.001*
Little Bear Cr. (Sthd)	0.1053	0.0775	0.0255	0.0147	---	*	0.459
WF Potlatch (Sthd)	0.1087	0.0814	0.0302	0.0017	0.0102	---	*0.035*
Big Bear Cr (Sthd)	0.1365	0.1044	0.0441	0.0271	0.0168	0.0239	---

FIGURES

Figure 34. An interior photo of the Big Meadow Creek culvert before and after the installation of baffles designed to increase water retention during low flows and serve as velocity barriers during periods of high flow. Photo courtesy of: Ryan Banks, OSC.

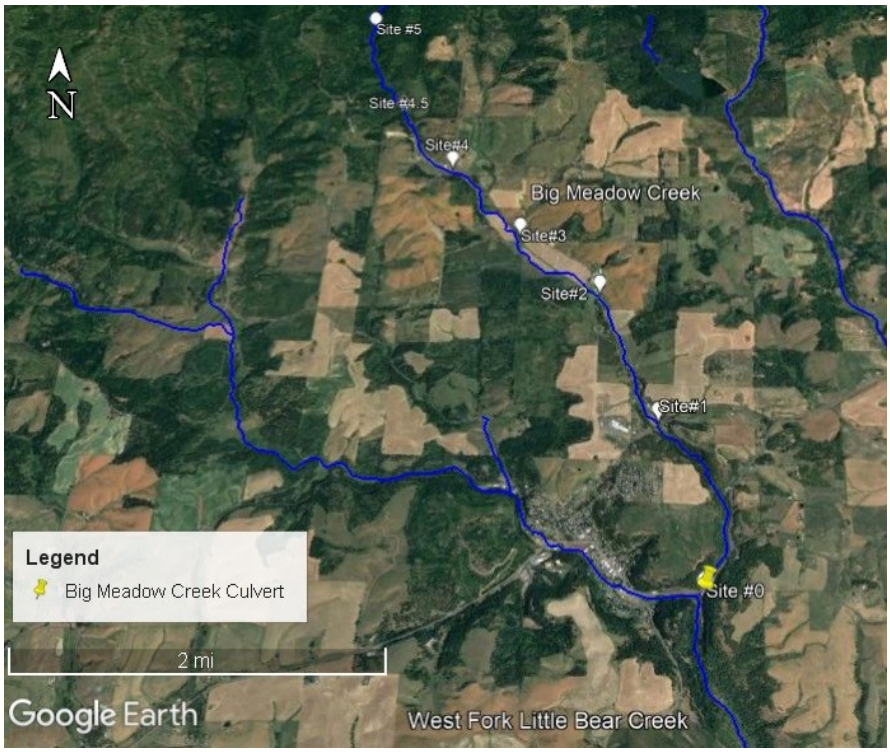

Figure 35. Location of the Big Meadow Creek culvert located at the mouth of Big Meadow Creek, a main tributary to the West Fork Little Bear Creek in the Potlatch River Basin, Idaho. Sites #0-5 show location of juvenile Oncorhynchus mykiss collections in Big Meadow Creek from 2018-2020.

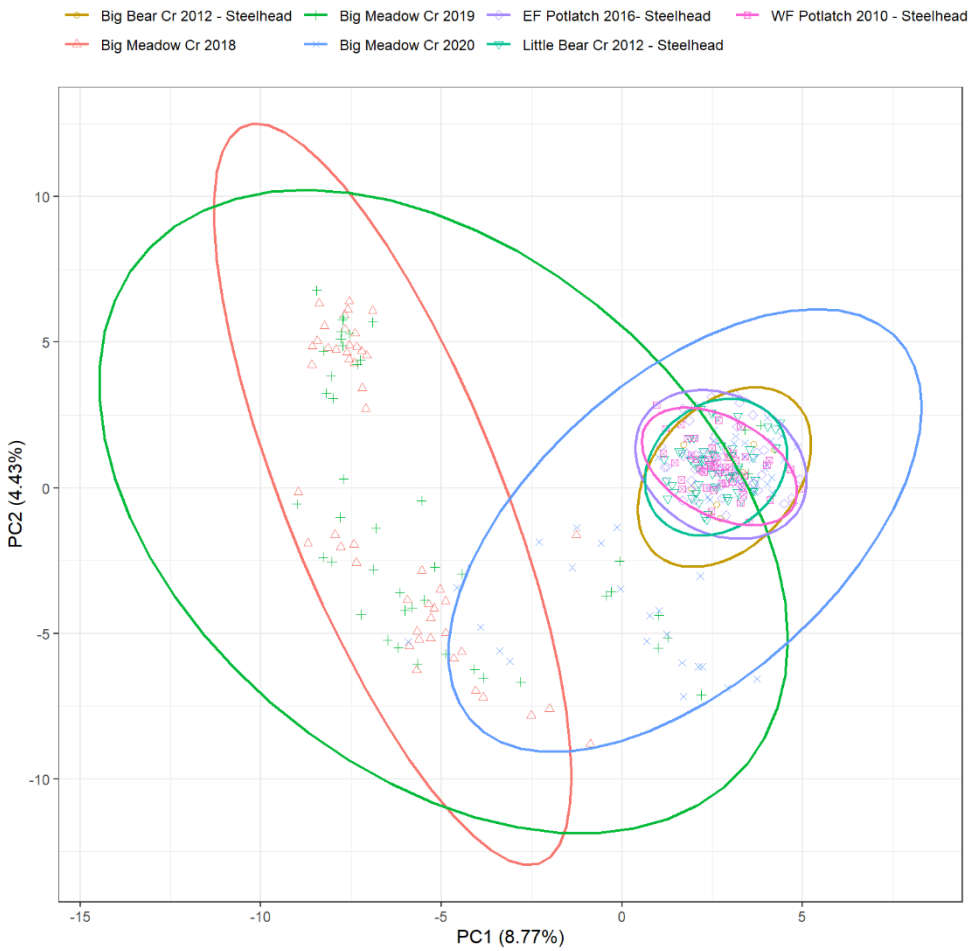

Figure 36. A plot displaying the results from a principal components analysis (PCA) of *Oncorhynchus mykiss* collections from Big Meadow Creek alongside adult steelhead samples collected from tributaries to the Potlatch River. The two axes correspond to the first and second principal components (PC1, PC2) and the percentage of variance explained is presented in parentheses. Points correspond to individual fish, colors correspond to sample collections, and ellipses represent 95% confidence intervals.

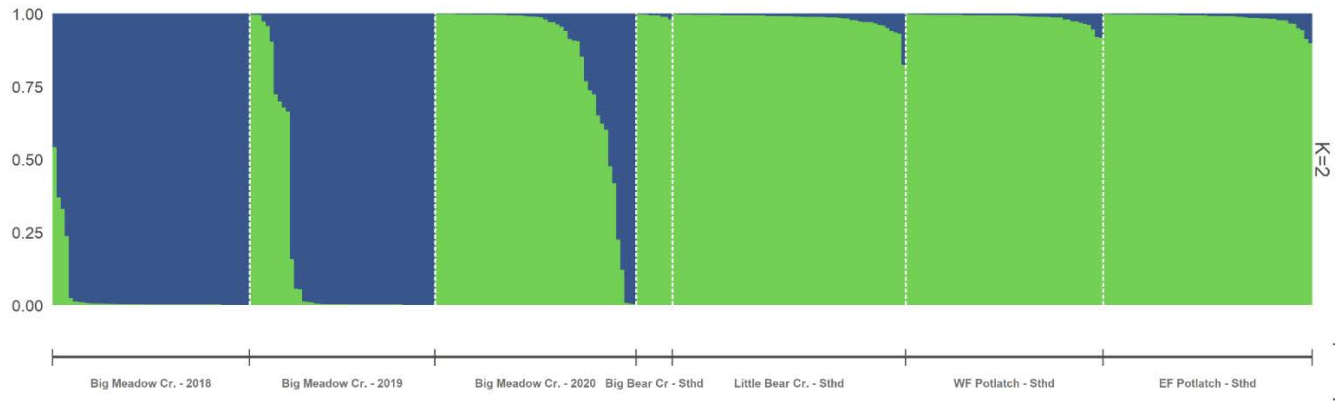

Figure 37. A Structure plot displaying the results of Bayesian clustering analysis of *Oncorhynchus mykiss* samples from Big Meadow Creek along with adult steelhead sampled from neighboring drainages assuming two genetic clusters (K = 2). Each vertical bar denotes an individual fish and colors correspond to different genetic clusters. Individuals that contain multiple different colors correspond to admixed individuals (fish with joint ancestry across multiple genetic groups).

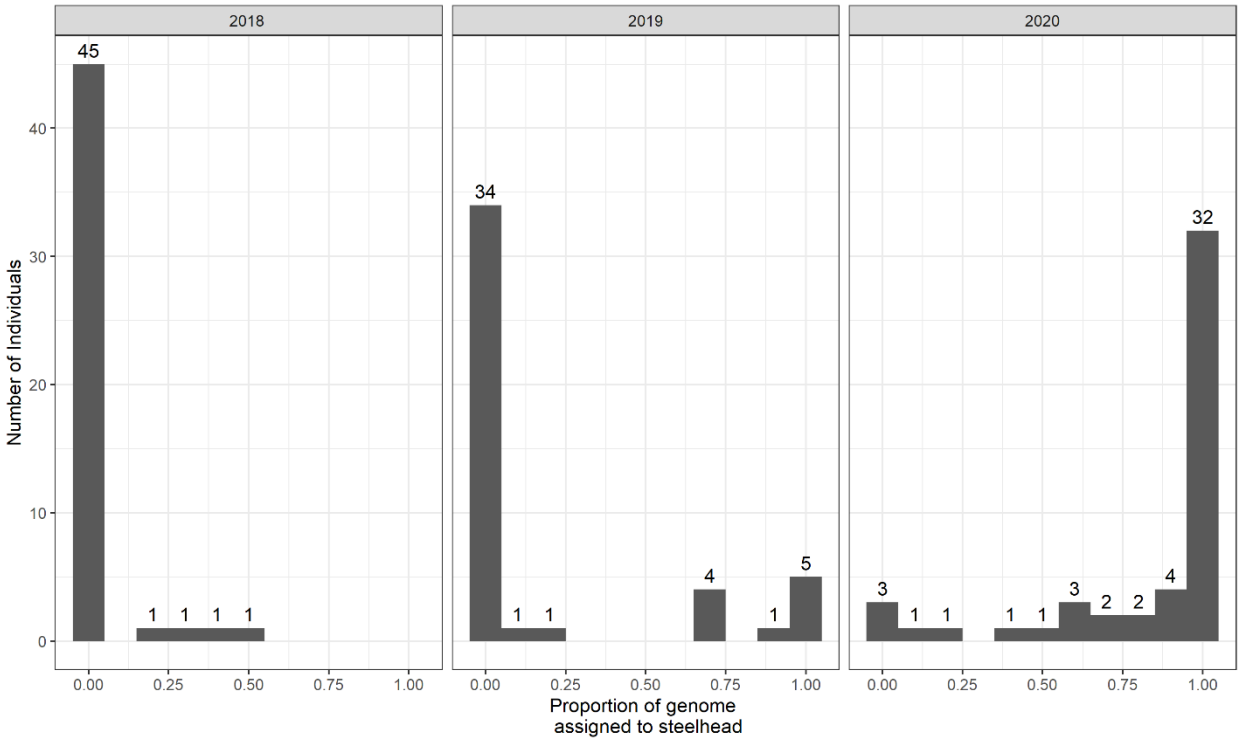

Figure 38. A histogram displaying the proportion of individual genomes (i.e., how much of an individual fish's genome was assigned to the steelhead genetic cluster) across three sampled years of juvenile *Oncorhynchus mykiss* from Big Meadow Creek, Idaho. Genome assignment was based on outputs from the program Structure and values were partitioned into ten bins of equal width (0.1). The numbers above each bar denotes the number of fish belonging to each bin.

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