FISHERY MANAGEMENT INVESTIGATIONS

IDAHO DEPARTMENT OF FISH AND GAME

Intensively Monitored Watersheds and Restoration of Salmon and Steelhead Habitat in Idaho: Fifteen-Year Summary Report

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ABSTRACT

Manipulation of aquatic habitat is a standard tool for fisheries managers. In flowing waters, actions are taken in the stream channel and riparian corridor to improve target aspects of fish population performance. Aquatic habitat projects to benefit salmon are widespread throughout the Pacific Northwest, including Idaho. The Idaho Department of Fish and Game (IDFG) has an active habitat restoration program with the goal to preserve, protect, perpetuate, and manage Idaho's fisheries resources by restoring resiliency and productivity of fish populations through habitat improvements. Monitoring is a necessary part of a habitat restoration program to assess the efficacy and efficiency of actions taken, and to provide guidance for future actions. Difficulties in achieving prescriptive understanding have led to the idea of the Intensively Monitored Watershed (IMW) program, which is a management experiment in one or more watersheds with a well-developed, long-term monitoring program to determine watershed-scale fish and habitat responses to restoration actions via causal mechanisms (e.g., increased survival). In Idaho, there are two IMWs, in the Lemhi River and Potlatch River basins. The goal of this report is to summarize the work conducted by the Idaho IMW projects from 2017 to 2021. The objectives are 1) to document restoration efforts and any continuing or emerging patterns, and 2) to offer conclusions to guide IDFG's habitat restoration program.

The Lemhi River IMW is located in the Salmon River basin in east-central Idaho and is focused on Chinook Salmon, steelhead, and Bull Trout. Restoration in the Lemhi River watershed addresses lack of connectivity between the river and its tributaries, reduction of spawning and rearing habitat, and reduced flow in the mainstem. Restoration actions in the Lemhi River basin have led to an increase in the amount of accessible rearing habitat, particularly during low summer flows and during winter when specific habitat features are crucial for survival. These actions provide opportunities for fish to migrate in and out of new habitats without delay and to increase their distribution. When fry and parr have access to additional habitat, we expect them to use those habitats and not emigrate until the fall parr or smolt life stage, which should translate into an increase in survival rates. Restoration efforts in the Lemhi River basin have been substantial enough to elicit local responses of multiple species and life stages of salmonids. Clearly, restoration actions increased the abundance and distribution of salmonids at varying spatial scales and provided opportunities in higher quality habitats. The indication that survival of age-1 Chinook Salmon smolts may be increasing because of habitat actions in the upper Lemhi River underscores the importance of maintaining the existing IMW monitoring framework in the future. Results from the Lemhi River IMW have been integral in shaping the monitoring framework as well as guiding prioritization and implementation of restoration projects in the basin. To date, the responses to restoration that we have documented are encouraging, but full understanding of fish population and habitat responses in the Lemhi River will require monitoring multiple anadromous fish generations for an additional 10 to 15 years.

The Potlatch River IMW is located in the Clearwater River basin in northern Idaho and is focused on steelhead. Restoration in the Potlatch River addresses tributary blockages and dewatered reaches in the lower watershed (Big Bear Creek) and simplified habitat in the upper watershed (East Fork Potlatch River). Restoration actions in the lower watershed have addressed 14 barriers and restored access to an additional 33 km of potential spawning and rearing habitat. In the upper watershed, restoration actions have treated approximately 12 km of stream habitat with large wood installations to improve habitat complexity. Fish passage projects in the lower watershed have elicited positive responses in steelhead in terms of increased connectivity and distribution. However, benefits in juvenile production have not been realized at this time, likely due to the previously degraded condition of the now restored habitats and/or low abundance of

fish to fully occupy these areas. In the upper watershed, we continued to document a positive shift towards larger and older steelhead smolts emigrating from the East Fork Potlatch River, which suggests improved rearing conditions in the drainage. Ultimately, these changes should lead to an increase in emigrant survival rates. Funding and permitting limitations continue to impact the pace of project implementation, especially regarding large-scale, high impact projects in the lower watershed. The next 10 to 15 years will be critical to the success of the Potlatch IMW project. Plans are in place to address the shortcomings in our monitoring infrastructure and design, especially regarding estimating adult steelhead abundance during low run years. Results from the upcoming reach-scale evaluations (Big Bear Falls trap and haul pilot project and East Fork Potlatch River large wood evaluations) will be vital in determining future restoration efforts in the respective drainages. We have built upon the solid foundation of restoration and monitoring in the IMW project, and these efforts will continue into the future to aid in the recovery of Potlatch River wild steelhead.

The intensive monitoring projects in the Lemhi River and Potlatch River basins have now been operating for more than 15 years. The restoration strategies and specifics of the monitoring are different between the basins, but many experiences are similar. In the last five years, we have seen two trends affecting stream restoration in our study areas: larger scale of restoration projects and declining anadromous adult returns. Both IMWs have seen some successes but those are largely compromised by the latter trend.

REPORT INTRODUCTION

Manipulation of aquatic habitat is a standard tool for fisheries managers. In flowing waters, actions are taken in the stream channel, riparian corridor, and floodplain to improve target aspects of fish population performance. Aquatic habitat projects to benefit salmon are widespread throughout the Pacific Northwest (Barnas et al. 2015), including Idaho. The Idaho Department of Fish and Game (IDFG) has an active habitat restoration program with the goal to preserve, protect, perpetuate, and manage Idaho's fisheries resources by restoring resiliency and productivity of fish populations through habitat improvements (IDFG 2019). This program focuses on native species in priority drainages and on private lands. Habitat restoration projects undertaken by IDFG are strategic, and implementation actions are designed to address limiting factors for fish populations in a given location. Further, pre- and post-restoration monitoring efforts are designed to determine whether focal populations and habitats respond to restoration efforts in a measurable way.

Monitoring is a necessary part of a habitat restoration program. Monitoring has two primary functions: 1) to assess the efficacy and efficiency of actions taken, and 2) to provide guidance for future actions (IDFG 2019). The second function is often complicated because typically multiple actions are undertaken together, thus it may not be clear why (or why not) the target fish population responded in the desired manner (e.g., Richer et al. 2022). Difficulties in achieving prescriptive understanding have led to the idea of the Intensively Monitored Watershed (IMW; Bilby et al. 2004) program. An IMW is a management experiment in one or more watersheds with a well-developed, long-term monitoring program to determine watershed-scale fish and habitat responses to restoration actions via causal mechanisms (Bennett et al. 2016).

In Idaho, IMW efforts are directed towards the Lemhi River and Potlatch River basins. Habitat improvement in these basins is driven by anadromous fisheries management and the availability funding from the Pacific Coast Salmon Recovery Fund and the Bonneville Power Administration. The Lemhi and Potlatch IMWs were first implemented in 2007 and 2008, respectively. The results from the first five years of implementation were summarized by Bowersox and Biggs (2012). Uthe et al. (2017) synthesized results over the first 10 years and offered preliminary conclusions to guide restoration. We have now arrived at the fifteen-year mark for Idaho's IMW program. It is important for monitoring programs to examine, interpret, and present their data (Lovett et al. 2007). Indeed, Uthe et al. (2017) recommended re-visiting the restoration plan on a periodic basis to adaptively manage the program, including the evaluation portion, to help Idaho's habitat restoration program to be more efficient and strategic, building credibility with landowners and cooperating agencies.

The goal of this report is to summarize the work conducted by the Idaho IMW projects from 2017 to 2021 and place it within the full 15-year record. The objectives are 1) to document restoration efforts and any continuing or emerging patterns, and 2) to offer conclusions to guide IDFG's habitat restoration program. The first objective will briefly describe the restoration projects completed or in progress, along with how IMW results have influenced them. As part of the second objective, we will revisit expectations to allow for adaptive management towards completion of the IMW projects. The purpose of this step is to build towards a formal analysis framework to give rigor and credibility to IMW products.

PART 1. THE LEMHI RIVER INTENSIVELY MONITORED WATERSHED PROJECT

INTRODUCTION

The Lemhi River basin was historically one of the most important spawning areas for migratory salmonids in the upper Salmon River basin (Bjornn 1978). The Interior Columbia Basin Technical Recovery Team (ICBTRT) classified the intrinsic steelhead (anadromous *Oncorhynchus mykiss*) population size in the Lemhi River basin as intermediate in size; the Chinook Salmon (*O. tshawytscha*) population size was classified as very large, the largest Chinook Salmon population in the Upper Salmon River major population group (ICBTRT 2005). The U.S. Fish and Wildlife Service considered the Lemhi River Critical Habitat Subunit essential to Bull Trout (*Salvelinus confluentus*) recovery because of the large population size, quantity of habitat, and diversity of life history forms (USFWS 2010). For a full historical perspective of the upper Salmon River basin, see Uthe et al. (2017).

The Lemhi River is a relatively low-gradient, 4th-order system located in east-central Idaho with a drainage basin encompassing approximately 3,290 km^2 (Figure 1.1). The river originates at the confluence of Eighteenmile Creek and Texas Creek near Leadore, Idaho and flows in a northwesterly direction for 90 km before entering the Salmon River near Salmon, Idaho. Although most of the basin is public land (82.2%), most of the valley and mainstem riparian areas are on private lands (BLM 1998). The basin contains 31 major tributaries, most of which originate in the surrounding mountains and enter the valley across alluvial fans, where they naturally lose some discharge to the aquifer. However, some of the major tributaries in the upper portion of the valley are spring creeks and thus the upper Lemhi River is influenced by groundwater.

The current restoration program is largely driven by habitat improvements such as offsite mitigation for Columbia River hydrosystem impacts. Most habitat improvements were made following Endangered Species Act (ESA) listing of spring/summer Snake River Chinook Salmon in 1992 and steelhead in 1997 (NMFS 2008), although there is a history of restoration dating to the 1950s (see Uthe et al. 2017 for a complete review). Specific recovery objectives for the Lemhi River included a 7% increase in freshwater productivity of Chinook Salmon and a 3% increase in freshwater productivity of steelhead (NMFS 2008). The current recovery plan seeks to achieve a population status of “viable,” which constitutes a minimum abundance threshold (MAT) of 2,000 adult Chinook Salmon with 1.34 recruits per spawner and a MAT of 1,000 adult steelhead with 1.14 recruits per spawner (NOAA 2017).

Bull Trout were listed under the ESA in 1999. The Bull Trout recovery plan identifies the Lemhi River as a core area within the Upper Snake River Recovery Unit (USFWS 2015). The Bull Trout recovery plan has many similarities with the Chinook Salmon and steelhead recovery plans and recommends an integrated recovery effort with anadromous fish recovery in the Salmon River drainage. The plan does not identify quantitative conservation objectives, but does recommend implementing projects that remove passage barriers, restore riparian areas, and reverse the negative effects of degradation associated with historic and contemporary land-use practices. It also recommends maintenance of long-term data sets and continued monitoring of the abundance and distribution of Bull Trout in the Lemhi River basin.

A major emphasis of conservation efforts in the Lemhi River prior to the period covered by this report was providing sufficiently conducive fish migration conditions. Currently, all major irrigation ditches on the mainstem Lemhi River are screened and have bypass systems to prevent entrainment of migrating fish. A minimum flow agreement was implemented through the Idaho Water Resources Board, such that permanent water acquisitions and annual irrigation agreements were established to increase flow below the L6 diversion, a Lemhi River segment that historically dewatered during the irrigation season. These efforts acquired a minimum of 35 cfs for 80% of the irrigation days and 25 cfs for 20% of the days between March 15 and June 30. The remainder of the irrigation season (July 1 through November 15) has a minimum flow agreement of 25 cfs.

Limiting Factors in the Lemhi River Watershed

Current habitat implementation strategies are informed by the 2017 Upper Salmon Subbasin Integrated Rehabilitation Assessment (USSIRA 2018). This assessment provided a science-based foundation based on fish population, geomorphic, and hydrologic data for project implementers to reference when designing habitat projects in the Lemhi River. The main factors limiting Chinook Salmon production identified in the Lemhi River were overwinter capacity (high priority) and summer (parr) rearing capacity (medium priority). The assessment also prioritized specific reaches of the Lemhi River to focus habitat action implementation. These areas in priority order are 1) mainstem Lemhi River downstream of Hayden Creek, 2) mainstem Lemhi River upstream of Hayden Creek, and 3) Hayden Creek.

The Integrated Rehabilitation Assessment team also completed companion reach-specific assessments to provide even finer scale guidance to project implementers, the upper and lower Lemhi River Multiple Reach Assessments (LLRMRA 2021; ULRMRA 2021). The lower Lemhi River was specifically identified as a bottleneck for juvenile Chinook Salmon survival because of the lack of quantity and quality rearing habitat and has been the main focus for restoration since 2017. Habitat in the lower Lemhi River is characterized by perched floodplains, a single-threaded armored straight channel, and limited habitat complexity with reference to water velocity, physical structure, and riparian vegetation. Therefore, the lower Lemhi River Multiple Reach Assessment's primary biological objective is to increase rearing habitat capacity for Chinook Salmon during summer and winter. The lower Lemhi River Multiple Reach Assessment biological and geomorphic objectives include the following:

Biological objectives:

1. Increase braided channels in the lower Lemhi River valley segment.
2. Increase the frequency of channel units (i.e., pools, riffles, glides).
3. Improve and increase juvenile fish cover quantity and quality.
4. Increase the structural and hydraulic diversity of available foraging locations (i.e., off-channel and/or side channel habitats).
5. Increase availability of reduced water velocity to decrease bioenergetic demands.
6. Maintain or improve tributary connection and maintain or increase base flow of the mainstem Lemhi River.
7. Mediate temperatures to increase hyporheic flow and/or riparian cover.

Geomorphic objectives:

1. Distribution of stream flow and energy among multiple channels and/or floodplain.

2. Improved channel geometry.

3. Increase floodplain connectivity and activation of secondary channels.

4. Increase secondary channel abundance and diversity.

5. Increase hydraulic and structural diversity and complexity.

6. Increase density of native riparian plant communities.

Lemhi River Intensively Monitored Watershed Project Overview

A comprehensive effectiveness monitoring strategy was implemented by IDFG in 2007 under the Intensively Monitored Watershed (IMW) program to evaluate fish response to habitat restoration actions in the Lemhi River basin. The overarching goal of an IMW project is to understand the linkage between habitat actions and fish responses at the watershed scale (Bennett et al. 2016). In the Lemhi River basin, the IMW study design is specific to the limiting factors being addressed and the types of habitat restoration actions being implemented. The main objectives of the Lemhi River IMW study are:

1. Monitor changes in the distributions of adult Chinook Salmon, steelhead, and resident/fluvial salmonids in the Lemhi River, Hayden Creek, and candidate tributaries for reconnection.

2. Measure changes in productivity of Chinook Salmon and steelhead at watershed and priority area scales.

3. Monitor fish population and habitat responses to individual restoration projects and specific habitat treatment types.

The Lemhi River IMW study consists of a nested spatial design that enables assessment of juveniles and adults at the tributary, mainstem river, and watershed scales. Hayden Creek serves as a reference watershed in the study design to enable comparisons among restored areas in two treatment watersheds (i.e., the reaches of the Lemhi River upstream of Hayden Creek [upper] and downstream [lower], including the tributaries to those reaches). This design allows investigators to provide results at the scale necessary for addressing IMW objectives, but also at the finer scales most relevant to restoration practitioners. Monitoring efforts have recently expanded to include site-specific evaluations of restoration projects that use existing infrastructure and sampling designs to simultaneously meet the broader objectives of the IMW project. We developed the following hypotheses associated with the primary restoration strategies:

1. <u>*Tributary reconnections and off-channel habitats should increase the amount of spawning and rearing habitat accessible to migratory salmonids.*</u> We consider tributaries functionally connected when fish can migrate into, out of, and through tributaries without delay. Therefore, upon completion of multiple actions to achieve reconnection status, we expect to observe adults migrating into tributaries for spawning activity, juveniles produced in other areas to migrate into and out of tributaries for seasonal rearing opportunities, and an increase in the upstream extent of pioneering individuals (i.e., expanded species and redd distributions). We also expect to see an increase in the number of juvenile fish occupying reconnected tributaries over time (i.e., standing stock). Additionally, off-channel habitats (e.g., side channels and floodplains) in the mainstem Lemhi River will provide habitat for staging

adult fish before spawning, and seasonal rearing opportunities for juvenile fish migrating downstream.

2. *The combination of tributary reconnections and mainstem upper and lower Lemhi River habitat improvement projects should improve rearing conditions during all seasons.* Benefits should occur primarily in summer and autumn when irrigation diversion and passage barriers would have rendered those habitats inaccessible. Benefits should also occur in the winter when the lack of overwintering habitat limits fall parr survival. In addition to the tributary-specific and off-channel habitat responses (see above), we expect to see increased productivity of fall parr and spring age-1 smolts per redd emigrating from the upper Lemhi River. When fry and parr have access to newly available habitat in the Lemhi River, we expect them to use those habitats and not emigrate until the fall parr or smolt life stage. Increased duration of use of natal reaches by fry and parr or higher survival rates should translate into an increase in fall parr and smolt abundance measured at the downstream boundaries.

The current research, monitoring, and evaluation studies began in 2010. The first phase continued until 2016 and was evaluated by Uthe et al. (2017). The first phase emphasized monitoring the effects of reconnecting tributaries to the Lemhi River. The second phase began in 2017 and was partly a continuation of the first phase but with more emphasis on habitat rehabilitation in the mainstem Lemhi River.

During 2017-2021, research, monitoring, and evaluations focused on six reconnected tributaries and several projects in the mainstem Lemhi River. Six priority tributaries were identified for reconnection and three were functionally reconnected by 2016 (Kenney Creek, Little Springs Creek, and Canyon Creek; Uthe et al. 2017). During this reporting period, three additional tributaries were functionally reconnected (Bohannon Creek, Big Timber Creek, and Hawley Creek). Additionally, we monitored fish response to mainstem Lemhi River habitat projects. The Lower Lemhi River Rehabilitation Project started in 2015 and ultimately will treat or restore over 5 km of stream while adding additional stream length through side channel complexes. Since the scope of the project is large, the river is divided into four subreaches and each subreach consists of multiple phases of restoration. We anticipate this project to be fully completed by the end of 2024. For this reporting period, one of four subreaches were completed with an additional subreach partially completed.

The partial completion of the Lower Lemhi River Rehabilitation Project increased the quantity and improved the quality of rearing habitat in the lower Lemhi River. Numerous complex and diverse habitat features are now available that provide ideal habitat for fish of various life stages, with a focus placed on overwintering habitat for juvenile Chinook Salmon and steelhead. These completed mainstem Lemhi River projects address the lower Lemhi River biological objectives 1-5 and 7, as well as the geomorphic objectives 1-6. Preliminary results on fish response to these habitat actions are evaluated in this report.

In this report, we present the results of monitoring activities completed during 2017-2021 and report on the data series that this work builds on. We organized report sections by life-stage to link methods and results to the life cycle monitoring approach. To date, primary restoration efforts focused on reconnecting six priority tributaries (Big Timber Creek, Canyon Creek, Hawley Creek, and Little Springs Creek in the upper Lemhi watershed; and Kenney Creek and Bohannon Creek in the lower Lemhi watershed; Figure 1.2), and the rehabilitation of the lower mainstem Lemhi River by re-establishing multiple channels and reconnecting the floodplain. Therefore, these restoration efforts are the focus of this report. We discuss the fish and habitat results in the context of restoration actions completed through the monitoring period (i.e., tributary, mainstem river, and watershed research, monitoring, and evaluations). Furthermore, we make recommendations about future directions for the Lemhi River IMW and discuss how results can inform adaptive management of restoration actions in the basin and elsewhere.

METHODS

Monitoring Design

Fish and habitat monitoring for the Lemhi River IMW is conducted within a spatially nested sampling framework to provide results at the tributary, mainstem river reach, and watershed scales. This hierarchy enables elucidation of the effects of specific restoration treatment types. Monitoring occurs within three key areas: the upper Lemhi River basin, lower Lemhi River basin, and Hayden Creek. A critical element of the monitoring framework is the use of Hayden Creek as a reference tributary to other tributaries influenced by water runoff and as a comparison to mainstem river habitats. Hayden Creek is the larger of the two tributaries that maintained a perennial connection with the Lemhi River following agricultural development in the basin. A large proportion of adult Chinook Salmon spawning also occurs in Hayden Creek. Therefore, it provides insight into the historical importance of tributaries in the Lemhi River basin and serves as a reference system for use in statistical comparisons of fish population response elsewhere in the watershed. The spatially explicit, quantitative monitoring of adult and juvenile abundance allows comparisons of productivity and abundance among the key areas.

Tributary reconnections have been a major focus of the restoration actions in the basin. Therefore, intensive tributary monitoring occurs within the six priority tributaries that have been reconnected (Figure 1.2). Monitoring surveys were conducted in Big Timber Creek, Canyon Creek, Hawley Creek, Little Springs Creek, Kenney Creek, and Bohannon Creek to provide pre- and post-reconnection fish information. Note that the number of years of pre- and post-treatment data available per tributary varies because reconnection efforts began in different years and varied in time necessary to complete habitat treatment.

In more recent years, results from the Integrated Rehabilitation Assessment and Multiple Reach Assessments have shifted restoration efforts to the mainstem Lemhi River. The Lower Lemhi River Rehabilitation Project was evaluated using a before-after-control-impact (BACI) study design to assess project-level monitoring (Roni et al. 2005). Sites within the BACI study design consisted of control, reference, and treatment reaches. The control reach is immediately downstream of Hayden Creek and represents the channelized and simplified habitat associated with much of the degraded lower Lemhi River. The reference reach is downstream of the project site and contains the historic channel complexity and intact riparian forest that was prevalent throughout the Lemhi River in a pre-degraded state. The treatment reach is where habitat restoration occurred (Figure 1.3).

Reach-level monitoring activities were designed to assess the efficacy of specific restoration actions and identify specific locations where restoration actions should be directed. Monitoring was conducted within the four subreaches of the Lower Lemhi River Rehabilitation project to understand fish and habitat responses consisting of re-meandering of the main channel, creating a more active floodplain, activating historic channels, constructing side channels, and placement of large woody debris (Figure 1.3).

Fish response to habitat actions are reported for the anadromous life cycle from adult spawning to juvenile emigration using a variety of fish metrics. Fish metrics include adult and juvenile abundance (fish into the basin and fish out of the basin), juvenile standing stock and distribution, movement, and survival at various scales. Results will help us better understand how tributary reconnections increase accessible habitat to fish of all life stages and how tributary and mainstem Lemhi River habitat projects improve summer and winter rearing conditions to increase survival of anadromous fish at the watershed scale.

Fish Monitoring

The monitoring framework was developed to evaluate watershed-scale fish population responses to habitat actions, and to understand how fish populations are responding to individual habitat projects and specific treatment types (i.e., side channels and floodplain habitats). Fish population responses are based on data from three primary sources: (1) juvenile salmon and steelhead emigration estimates at rotary screw traps, (2) interrogations at in-watershed and out-of-watershed antennas set up to detect Passive Integrated Transponder tags (PIT tags) placed during electrofishing surveys and rotary screw trap operations, and (3) redd counts.

Monitoring Infrastructure

Three rotary screw traps (RSTs) are positioned to monitor juvenile salmon and steelhead production from the three priority areas identified in our study design. A subsample of fish captured at the screw traps are implanted with PIT tags to estimate trapping efficiency, to estimate within watershed or within reach survival rates, and to estimate survival to Lower Granite Dam (LGR). The lower Lemhi RST is at rkm 7, the upper Lemhi RST is at rkm 49, and the Hayden Creek RST is 1 km upstream of the confluence with the Lemhi, which is just downstream of the upper Lemhi RST (Figure 1.4). The purpose of the trap closest to the mouth of the Lemhi River is to estimate total production in the Lemhi River basin. The Hayden Creek RST enables us to treat Hayden Creek as a reference for investigations of changes in upper Lemhi River production.

Tandem instream PIT-tag detection systems (IPTDSs) were installed in the Lemhi River basin to document movement patterns of PIT-tagged fish, estimate spatially-explicit survival rates, and estimate adult escapement. The first IPTDS installations were in locations associated with tributary reconnections and with existing RST infrastructure to provide juvenile survival and adult escapement information that enables adult abundance to be linked to specific brood year juvenile production. Two additional IPTDSs were installed in the lower Lemhi River in 2019 to monitor movement, overwinter use, and estimate spatially-explicit survival rates of juvenile anadromous fish within a specific reach in the Lower Lemhi River Rehabilitation project. Within the Lower Lemhi River Rehabilitation project, there are two floating antennas that were installed at the inlet and outlet of constructed side channels to further monitor fish use (e.g., proportional use and residence time; Figure 1.4). Additional IPTDS installations were completed in 2010 and 2011, in priority candidate tributaries for reconnection, to document juveniles migrating into reconnected rearing habitat and adults pioneering into newly available spawning habitat (Bowersox and Biggs 2012).

In tributaries, IPTDSs were installed as close to the mouth as possible to account for as much production in the tributary as possible (Table 1.1). They were also installed within tributaries to more accurately measure reach-specific distribution and abundance response. All detections were uploaded to the PIT Tag Information System (PTAGIS, www.ptagis.org). Tagging at key locations facilitated IPTDS function (i.e., generating interrogations), as explained further in this document.

Reconnected Tributaries

Distribution and Movement

Movement of fish in and out of tributaries was assessed using interrogations at tributary IPTDSs for insight on the distribution and abundance of fish that access these habitats for juvenile rearing. To investigate fish moving out of tributaries, the PIT-tag codes of fish tagged in electrofishing surveys were queried in PTAGIS to determine if they were detected on the IPTDS at the mouth of the tributary in which they were tagged, beginning the year of tagging and afterwards. We included all tags implanted during electrofishing surveys within the tributary. Conversely, to assess movements into tributaries, the IPTDSs at the mouths of priority tributaries were queried for all detections generated by fish tagged outside of that specific tributary (e.g., all detections on LLS from fish tagged in the Lemhi River basin, except those with a mark site of Little Springs Creek). This included fish marked during electrofishing surveys and at RSTs. We used records from the reference system, Hayden Creek, as a benchmark against which to compare the reconnected priority tributaries.

Interrogations at tributary and mainstem river IPTDSs were used to investigate movement of fluvial Bull Trout into reconnected tributaries. Weirs were installed to capture post-spawn Bull Trout to increase the number of fluvial Bull Trout tagged in the Lemhi River basin. Hayden Creek has been identified as a potential source population for fluvial Bull Trout and has maintained connection to the mainstem Lemhi River. Therefore, Bull Trout weirs were installed on Hayden Creek and Bear Valley Creek (tributary of Hayden Creek) and operated during the month of September from 2013 to 2021. The IPTDS on Hayden Creek was monitored closely to ensure that adult Bull Trout were moving upriver to their spawning grounds prior to installing the weirs so that fish were not impeded from reaching their spawning grounds. Weirs were checked daily. All fish captured at the weirs were scanned for previous PIT tags. Tags were implanted in the dorsal sinus of newly captured fish. Fin clips, length, and weight were collected from all fish prior to their release immediately downstream of the weirs.

Summer Juvenile Salmonid Standing Stock

Electrofishing surveys were conducted in the six priority tributaries to estimate juvenile salmonid standing stock during the summer (June through September). We refer to tributary-specific abundance as standing stock to differentiate it from emigrant abundance estimated over time at RSTs (see below). We also used these surveys to investigate changes in distribution associated with restoration actions, and to deploy PIT tags (see Appendix A for sampling details). We used records from the reference system, Hayden Creek, as a benchmark against which to compare the reconnected priority tributaries.

The electrofishing survey design is a continuous sampling framework in which multiple kilometers of each tributary are surveyed using mark-recapture electrofishing techniques. This survey design adheres to the closure assumptions of mark-recapture estimators and provides an opportunity to implant PIT tags in fish. Electrofishing surveys were conducted in each tributary from either 2009 or 2010 through 2021 (Appendix A).

During electrofishing surveys, operators used one or two backpack electrofishing units depending on tributary width to ensure that maximum distance between two units was generally less than 3 m. We measured fork lengths (± 1 mm) and weights (± 0.1 g) of all captured salmonids and implanted PIT tags in a subsample of salmonids ≥60 mm. We tagged salmonids at a rate of at least 50 individuals per kilometer (per species) to distribute tags throughout the sampled areas.

We clipped a portion of the upper caudal fin of all captured salmonids to serve as a mark for use in mark-recapture analyses. Although we marked all salmonids, we only consider standing stock estimates of Chinook Salmon, steelhead, and Bull Trout in this report because of the low numbers and limited distribution of Cutthroat Trout (*O. clarkii*) in sampled areas. Fish locations on the mark event were associated with tagging locations distributed along each site at a maximum distance of 250 m. On the recapture event, fish locations were recorded at the approximate location of capture. We stored survey data in the IDFG Lakes and Streams Survey Database and uploaded records of tagging and recapture events to PTAGIS.

Standing stock estimates were estimated for each tributary by calculating densities within sampled areas and extrapolating to non-sampled areas. To standardize estimation across years, the farthest upstream sampling location (all years) was chosen as the upstream extent of standing stock estimation. Therefore, estimates of total juvenile steelhead standing stock are biased low, because we did not locate the upper extent of *O. mykiss* distribution within any tributary. To interpolate through non-sampled areas that were bound by sampled reaches on the upstream and downstream ends, the average density from the two sampled reaches was multiplied by the total non-sampled reach length. To extrapolate upstream or downstream through a non-sampled reach, the density within the adjacent sampled reach was multiplied by the non-sampled reach length. Reach lengths were measured by plotting bottom and top of site coordinates in ArcMap 10.3 and measuring along the NHDPlus 1:24,000 hydrography shapefile. Standing stock estimates will help us evaluate how tributary reconnections have provided rearing opportunities for juvenile fish.

Standing stock at sites were calculated using the FSA package (Ogle 2017) in Program R (R Development Core Team 2017). Multi-pass depletion estimates were calculated using the removal function with the Carle-Strub method for the first few years. After 2013, mark-recapture estimates were calculated using the *mrClosed* function with the Chapman-modified Lincoln Petersen estimator. Juvenile Chinook Salmon of all lengths were included in the analyses, but only steelhead less than 365 mm were included. This size criterion was chosen because a PTAGIS mainstem interrogation query revealed that only one individual from the Lemhi basin larger than this threshold (390 mm) has ever been detected on a mainstem Columbia River interrogation site. Thus, this maximum size threshold is sufficient for including most of the probable anadromous component of the *O. mykiss* population encountered during electrofishing surveys.

Redd Counts

Multi-pass ground surveys were conducted for spawning by the target species (Chinook Salmon, steelhead, and Bull Trout) in selected tributaries following standard IDFG redd survey protocols (Copeland et al. 2019). Each transect usually had two viewers walking on opposite stream banks, but a few surveys in the small tributaries were conducted with one observer. Duration between passes was typically one week. All survey data were stored in the IDFG Spawning Ground Survey database. Spawning data will help us better understand how tributary reconnection efforts and habitat projects have influenced spawning distribution. We used records from the reference system, Hayden Creek, as a benchmark against which to compare the reconnected priority tributaries.

Mainstem Lemhi River

Distribution and Movement

Interrogations at mainstem Lemhi River IPTDSs were used to understand how fish respond to large-scale mainstem Lower Lemhi River Rehabilitation efforts to evaluate timing of fish use, residence time, and overwinter survival. These IPTDSs were installed in August 2019. Fish tagged as juveniles were used to investigate the timing of fish entering the reach and the amount of time spent residing in that reach. To assess movement and retention, the IPTDSs at the furthest upstream and downstream locations were queried for all fish detections (Figure 1.4). We only considered the first interrogation record of each fish at the upstream antenna and the first detection of that fish that the downstream antenna. The number of days an individual fish resided within the reach was calculated by subtracting the downstream antenna detection date from the upstream antenna detection date. Fish that were only detected by an individual antenna (either upstream or downstream) were excluded from the summary.

Summer Juvenile Salmonid Standing Stock

Electrofishing surveys were also conducted in the mainstem Lemhi River to implant tags in fish for estimating reach-based survival in the mainstem river (ISEMP and CHaMP 2017), as well as monitoring fish moving from the mainstem river into tributaries. The mainstem surveys also provided the opportunity for reach-level, project-specific effectiveness monitoring.

Standing stock estimates were calculated control, reference, and treatment sites from 2016 to 2020. All three sites were sampled via mark-recapture electrofishing surveys pre- and post-restoration to evaluate fish response to restoration actions. Standing stock estimates for each site were divided by the length of the stream sampled to estimate linear fish densities. Densities were compared among control, reference, and treatment reaches across years. A two-way analysis of variance (ANOVA) test was used to model juvenile Chinook Salmon and steelhead densities as a function of location (e.g., control, reference, and treatment) and period (e.g., before and after restoration). The two-way ANOVA was as follows:

$$X = \alpha + \beta_i + \alpha\beta_i$$

where X is the response variable, fish density measured as fish per linear km, α is the reach location, β_i is the period ($i =$ before and after), before was 2016-2019 and after was 2020, and $\alpha\beta_i$ is the interaction between the two explanatory variables. A significant interaction ($p \leq 0.05$) indicates an effect by restoration (Smith 2002).

Juvenile Chinook Salmon Survival

Survival rates within the Lemhi River basin by priority area and season were estimated to infer how habitat restoration influences juvenile survival. We hypothesized better quality rearing habitat will allow fish to rear in the Lemhi River basin through the winter into the smolt stage and to grow larger in size, increasing survival to the ocean (Zabel and Achord 2004). Survival rates for this analysis were estimated using TribPit (Lady et al. 2014; Buchanan et al. 2015). Program TribPit estimates cohort-based survival rates using a release-recapture model that accounts for fish exhibiting multiple winter and rearing strategies during their downstream migration (Lady et al. 2014). Marking events at RSTs and roving electrofishing surveys were included in the analyses, as were recapture events at RSTs and live-resights at IPTDSs (ISEMP and CHaMP 2017). Survival was estimated for the following combinations: (1) upper Lemhi

River*Summer/Fall; (2) upper Lemhi River*Winter; (3) Hayden Creek*Summer/Fall; (4) Hayden Creek*Winter; and (5) lower Lemhi River*Winter. Survival rates above the upper Lemhi RST and the Hayden Creek RST were based on fish tagged during summer electrofishing surveys. Therefore, two seasonal timeframes are considered: time of release through December 31 for summer/fall survival rates, and January 1 through the following spring (time of age-1 smolt migration) for winter survival rates. The upper Lemhi River subpopulation was sampled during late May through mid-June, whereas the Hayden Creek subpopulation was sampled in mid-September, so the summer/fall seasonal durations are different between the two groups. The electrofishing release groups were supplemented with fish tagged at RSTs to increase precision of parameter estimates in the lower Lemhi River stratum. Winter survival rates in the lower Lemhi River were estimated from electrofishing release groups that moved downstream from the upper Lemhi River or Hayden Creek before December 31, or were released at the upper Lemhi RST or the Hayden Creek RST in the fall.

Lemhi River Watershed

Juvenile Chinook Salmon and Steelhead Emigration

Emigration from the three priority areas was estimated from data collected using the three RSTs. The RSTs were operated according to protocols established for anadromous emigrant monitoring by IDFG (Copeland et al. 2021). All traps were checked daily when in operation. We anesthetized captured fish with Aqui-S and scanned for PIT tags, measured weights to the nearest 0.1 g, and measured fork lengths to the nearest 1 mm. All fish captured were implanted with PIT tags except in the fall when a subsample of fish was tagged due to a limited number of tags. Chinook Salmon ≥60 mm and steelhead ≥80 mm were tagged with 12-mm PIT tags. Steelhead between 60-79 mm were tagged with 9-mm PIT tags. We released a known number of tagged fish above the screw trap each day to estimate trap efficiency. Marked fish released above the screw trap that were subsequently captured within five days of the initial marking event were considered recaptures for efficiency calculations. Trap information was archived in the JTRAP database, and all PIT-tag records were uploaded to PTAGIS.

Abundance estimates of juvenile Chinook Salmon and steelhead emigrating past RSTs were calculated using the Bailey-modified Lincoln-Peterson estimator:

$$N = \sum_{i=1}^{k} c_i \, (m_i + 1)/(r_i + 1)$$

where *N* is abundance of juveniles emigrating in a given year, *i* is season (defined below for each species), c_i is the number of all unique fish captured in season *i*, m_i is the number of tagged fish released in season *i*, and r_i is number of recaptures in season *i*. The estimator was computed using software specifically developed for use with screw trap data that uses an iterative maximization of the log likelihood (Steinhorst et al. 2004). The 95% confidence intervals were estimated with a bootstrap method with 10,000 iterations.

To estimate Chinook Salmon abundance, the trapping season was stratified according to life-stage intervals, which generally coincided with changes in trapping efficiency associated with changing hydrologic conditions. The start of the trapping season through June 30 was considered the spring period when the catch is predominately age-1 smolts. During periods of simultaneous capture of age-0 and age-1 Chinook Salmon, individuals were assigned to cohorts based on body size and appearance (Apperson et al. 2016). For the summer period from July 1 through August

31, the catch was predominately age-0 parr. The fall period was considered September 1 through the end of trapping season, which is the period when age-0 parr were actively migrating past RSTs.

Chinook Salmon life stages were summed into cohorts by brood year, the year that the fish were produced. For example, the total abundance estimate for brood year 2017 is calculated as the sum of age-0 fry caught in the spring period during 2018, age-0 parr caught during the summer of 2018, age-0 parr caught during the fall of 2018, and age-1 smolts captured during the spring of 2019. However, we excluded the fry life stage from our total brood year abundance estimates because not enough fry were captured to generate a precise estimate.

For steelhead abundance estimation, the trapping season was divided into two strata: 1) the start of trapping through May 31 (spring); 2) June 1 through the end of trapping season (summer/fall). The abundance estimates from the two trapping periods were summed into cohorts by trapping year to evaluate the number of fish migrating out of the basin and relate that to the number of spawning adults to estimate productivity which will help us better understand survival.

Juvenile Chinook Salmon Survival

Survival rates from the RST to hydrosystem entry at LGR were estimated based on detections of PIT-tagged fish through the spring 2021 emigration (brood year 2019). Survival to LGR was estimated separately for fall parr and age-1 smolts using Survival Under Proportional Hazards (SURPH) 2.2 software (Lady et al. 2001). This program uses a Cormack-Jolly-Seber model to estimate survival rates and detection probabilities based on interrogation histories at Lower Granite, Little Goose, Lower Monumental, McNary, John Day, and Bonneville dams and the estuary towed detector array. Survival rates of juvenile steelhead from RSTs were not estimated because of the large variation in duration of freshwater rearing strategies, which is problematic for distinguishing mortality from lack of downstream movement. This is especially challenging in the Lemhi River basin, because a significant proportion of the *O. mykiss* population exhibits resident or fluvial life history strategies.

Adult Chinook Salmon and Steelhead Escapement

We estimated escapement to locations in the Lemhi River basin for wild Chinook Salmon and steelhead using detections of fish tagged as adults at LGR analyzed by the Dam Adult Branch Occupancy Model (DABOM model; Kinzer et al. 2020; See et al. 2021). The DABOM model estimates the probability that a given fish moved along a certain corridor of the stream network and escaped to a specific location (Waterhouse et al. 2020). In this report, we use estimates at IPTDSs located near each RST.

Redd Counts

Multi-pass ground surveys were conducted for Chinook Salmon following the same protocols explained previously for redd counts in tributaries.

Chinook Salmon Productivity

The restoration efforts in upper Lemhi River tributaries (Big Timber Creek, Hawley Creek, Little Springs Creek, and Canyon Creek) should provide important rearing habitat for Chinook Salmon parr, increasing the productivity of that subpopulation. To test this hypothesis, productivity measured as the number of fall parr and age-1 spring smolts emigrating past the upper Lemhi

River RST per redd was compared to productivity measured at the Hayden Creek RST before and after reconnection efforts began. We chose this measurement of productivity, rather than total brood year emigrants per redd, because we predicted restoration efforts to have the largest effect on individuals that rear in those areas as summer or fall parr before emigrating the following spring as age-1 smolts. We used redd counts from spawning ground surveys as the measure of adult abundance in productivity calculations. For 2007, we used aerial redd counts for the Lemhi River because ground counts were not conducted that year because access to some properties was denied. Although the spawning ground transects in the upper Lemhi River do not cover the entire area above the RST, they encompass most spawning activity, so negative bias associated with redd abundances should be minimal. Productivity is the number of juveniles divided by the number of redds that produced the cohort.

We conducted the statistical analysis using the equation:

$$PL = PH + T$$

where PL is productivity of the upper Lemhi subpopulation, PH is the productivity of Hayden Creek subpopulation, and T is time period. We set the before and after periods for the analysis as follows. Initial reconnection efforts opened some summer rearing habitats as early as 2009 (i.e., Big Timber Creek). Therefore, 2009 was chosen as the initial year for the “post treatment” period. The productivity of the upper Lemhi was regressed on the productivity of Hayden Creek using the lm function in Program R (R Development Core Team 2017). We considered the treatment effect significant if the coefficient of time period variable had a *p*-value less than 0.05.

RESULTS

Fish Monitoring

Reconnected Tributaries

Distribution and Movement

The distribution of steelhead within streams remained consistent during the last five years. Steelhead occupied most locations sampled, which is a pattern consistently observed in Hayden Creek (Appendix B7). Steelhead were detected emigrating from all priority tributaries (Appendix C). Canyon Creek had a highly variable proportion of steelhead detected emigrating each year, ranging from 5% to 74% of a tagging cohort detected on CAC. Little Springs Creek also had a relatively high proportion of steelhead emigrating, with all but one tagging cohort emigrating at rates greater than 20.5% and an average of 28.9% (SD = 20.8%), followed by Bohannon Creek (average = 20.1%, SD = 8.5%), Kenney Creek (average = 17.3%, SD = 8.9%), and Big Timber Creek (average = 5.4%, SD = 4.3%). The pattern of steelhead emigration out of Hayden Creek was similar to Kenney Creek, with an average of 13.2% (SD = 9.0%, Appendix C11). Hawley Creek did not have any tagged steelhead that were detected leaving; however, the HEC IPTDS had periods where it was inoperable or had low detection efficiency.

The rate at which Chinook Salmon juveniles emigrated the year after tagging varied considerably (Appendix C). The proportion of Chinook Salmon detected leaving Hayden Creek the following year after tagging, as age-1 smolts in the spring, averaged 8% (SD = 6.3%), suggesting that most fish were emigrating out of Hayden Creek as summer parr or fall parr (Appendix C11). Most juvenile Chinook Salmon in the reconnected tributaries migrated out the year of tagging. Big Timber Creek had a consistently high proportion of juvenile Chinook Salmon emigrate the year of tagging (average = 35%) as well as Chinook Salmon in Canyon Creek (average = 63%). In Little Springs Creek, 96% of the fish detected leaving left the year they were tagged. These results show that few juvenile Chinook Salmon use the reconnected tributaries for winter habitat.

Bull Trout distribution within tributaries after reconnection was primarily in the reaches farthest upstream except in Kenney Creek, where Bull Trout were found in all sampled reaches. However, in the last five years, Bull Trout have been observed in most reaches sampled throughout Big Timber Creek, Canyon Creek, and Hawley Creek (Appendix B). Likewise, Bull Trout distribution in Hayden Creek extended throughout all reaches sampled (Appendix B19). However, sampling locations varied in years prior to 2013, so trends should be viewed with caution (Appendix A). We detected Bull Trout leaving some tributaries. Hayden Creek had the largest proportion of Bull Trout detected leaving the tributary with an average of 16.4% (SD = 7.8%, Appendix C11). Of the priority tributaries, Kenney Creek had the highest proportion of Bull Trout leaving, with an average of 4.5% (SD = 3.1%, Appendix C7). Bull trout were first tagged in Kenney Creek in 2011, but none were detected leaving until 2012. Bull Trout were tagged every year in Bohannon Creek since 2010, but none were detected leaving until one fish tagged in 2016 was detected at BHC in the autumn of 2016. Although Bull Trout were tagged in Canyon Creek, Big Timber Creek, Little Springs Creek, and Hawley Creek, none were detected on the IPTDSs at the mouths of those tributaries. Given the small number of tagged Bull Trout in Little Springs Creek, Canyon Creek, and Hawley Creek, IPTDSs may have failed to detect emigrating fish, or most of the fish tagged were resident fish rather than fluvial. In addition, Hawley Creek was seasonally disconnected due to low flows.

The number of fish detected moving from the Lemhi River into reconnected tributaries in the last five years was highly variable (Appendix C). During that time, we observed Bull Trout moving into Bohannon Creek, with the first detection in 2019. Chinook Salmon and steelhead were first detected moving into Bohannon Creek in 2015, three years after reconnection. In the other reconnected tributaries, Chinook Salmon, steelhead, and Bull Trout were detected entering prior to 2017. Post tributary reconnection, it typically took a few years to observe fish moving from the Lemhi River into the tributaries.

At Hayden Creek and Bear Valley Creek weirs, 196 post-spawn Bull Trout were captured and tagged between 2017 and 2021 (mean TL = 499 mm, range 122-790 mm). Between 2013 and 2021, in total, 354 adult Bull Trout (mean TL = 489 mm, range 122-790 mm) were captured and tagged (Appendix D). The number of fish captured each year varied and included both newly captured fish as well as recaptured tagged fish. On average, 34% of these fish returned to the same weir in the Hayden Creek drainage, (range: 4 fish (12%) in 2016 to 8 fish (44%) in 2017). Recaptured Bull Trout were documented returning to their spawning grounds up to five times. Time between spawns varied from every other year up to four years. However, most fish captured at the weirs were only captured once. Bull Trout that spawn in the Hayden Creek drainage and displayed fluvial characteristics were detected moving into Big Timber Creek, Kenney Creek, and Little Springs Creek.

Summer Juvenile Salmonid Standing Stock

Juvenile steelhead were the most abundant species in all tributaries but showed no consistent pattern in abundance over time (Figure 1.5). Prior to 2017, standing stock was variable across years and within streams, such that few overall trends were apparent (Figure 1.5). Likewise, standing stock in Hayden Creek does not show a distinct trend (Figure 1.6). However, sampling design and effort in tributaries were much different before 2013 (Appendix A), so these results should be viewed with caution. Hawley Creek and Kenney Creek were not sampled in 2020 and 2021.

Juvenile Chinook Salmon standing stock varied by year in each reconnected tributary during 2017–2021 (Figure 1.7). Similarly, from time of reconnection to 2021, Chinook Salmon standing stock varied but standing stock in all tributaries sampled was relatively high in 2015 compared to other years. Although standing stock exhibited considerable inter-annual variability, upper Lemhi River tributaries had consistently higher standing stocks than lower Lemhi River tributaries. In comparison, standing stock of juvenile Chinook Salmon in Hayden Creek was an order of magnitude higher than in reconnected tributaries (Figure 1.7; Figure 1.8). In reconnected tributaries where Chinook Salmon were present, as standing stock increased, so did the upstream extent of distribution (as illustrated in Little Springs Creek, Appendix B9). However, Chinook Salmon were not as broadly distributed in the priority tributaries as in Hayden Creek (Appendix B13).

Bull Trout standing stock varied among years and across priority tributaries during 2017-2021 (Figure 1.9). Bull Trout standing stock varied considerably from year to year from the time of tributary reconnection to 2021 for all priority tributaries. Standing stock in Little Springs Creek could not be estimated due to the small sample size. Standing stock was generally the highest in the reference tributary of Hayden Creek but standing stock estimates of Bull Trout in Big Timber Creek exceeded those of Hayden Creek in some years (Figure 1.9; Figure 1.10).

Redd Counts

No Chinook Salmon redds have been documented in any of the priority tributaries since tributary reconnection.

Steelhead spawning ground surveys did not reveal any clear trends in redd counts in reconnected tributaries or Hayden Creek during 2017-2021. Similarly, there were no apparent trends in redd counts from the time of tributary reconnections to 2021. We did not observe any redds in Bohannon Creek during the first year of redd counts in 2013 (Figure 1.11). Redd counts in Bohannon Creek were highest in 2014 (33 redds) and the fewest redds were observed in 2017 (2 redds). We did not survey Kenney Creek consistently through the years. From what was observed, the highest spawning activity occurred in 2008 with 22 redds and the fewest redds occurred in 2016 with 1 redd (Figure 1.11). No steelhead redds were observed in Big Timber Creek or Canyon Creek during any years. One steelhead redd was observed in Little Springs Creek during 2017, 2019, and 2021 (Figure 1.11, Appendix E5).

Mainstem Lemhi River

Distribution and Movement

Movement and retention time of juvenile Chinook Salmon in the mainstem Lemhi River within the Lower Lemhi Rehabilitation Project varied by year and month. Between 2019 and 2021, 22% of the juvenile Chinook Salmon that passed through the project site were detected at both the Eagle Valley Upper (EVU) and Eagle Valley Lower (EVL) IPTDSs. Juvenile Chinook Salmon were detected passing the EVU IPTDS shortly after installation, with most fish first detected in the project during October (Figure 1.12). After the first year, a spring/fall bimodal distribution is evident. In 2020, a higher abundance of fish was detected entering the project in May and even higher in October. Similarly, in 2021, fish moved into the reach primarily in May and October.

Juvenile Chinook Salmon retention varied from less than 1 day to 210 days. Between 2019 and 2021, fish resided in the reach for less than 1 day (35-56%), between 1-15 days (38-55%), between 16-30 days (2%), and greater than 30 days (4-7%), respectively. Over the years, the proportion of fish that resided for less than one day decreased and fish that resided for greater than one day increased.

Movement and retention time of juvenile Chinook Salmon in the side channels of the Lower Lemhi Rehabilitation Project depended on month. Most fish were detected moving into the side channel in the fall (Figure 1.13). Fish entering in October typically resided in the side channel for greater than 31 days and fish entering in November typically resided for 1-30 days. Fish were also detected entering the side channel in May but moved out in less than 1 day.

Summer Juvenile Salmonid Standing Stock

Juvenile Chinook Salmon and steelhead density estimates within the Lower Lemhi River Rehabilitation Project varied through time as restoration proceeded. Chinook Salmon densities declined in the treatment and reference reach but increased in the control reach from 2016-2017 (Figure 1.14). Following habitat enhancements in 2018, there wasn't an obvious fish response in any of the reaches. In 2019, additional habitat enhancement projects were completed that coincided with an observed increase in Chinook Salmon densities in the control and treatment reaches. However, Chinook Salmon densities continued to decrease in the reference reach. Chinook Salmon densities were the highest in the control reach (p = 0.0009), suggesting that variations in juvenile Chinook Salmon densities are a response of location (control, reference, and treatment reaches) and not of period (before or after habitat enhancement; Table 1.2). There was no interaction effect (p = 0.76); therefore, no restoration effect on Chinook Salmon standing stock was detected. Following habitat enhancements in 2019, steelhead densities increased in all three reaches, with the highest density in the treatment reach (p = 0.002), suggesting that variations in juvenile steelhead densities are a response of location (control, reference, and treatment reaches), period (before or after habitat enhancement), and the interaction of location and period (Table 1.2). For steelhead, the interaction is significant (p = 0.002), which means that the treatment reach responded differently during the After period. The change in steelhead standing stock for the treatment reach was more positive than the change in either of the other reaches.

Juvenile Chinook Salmon Survival

Survival rates within the Lemhi River basin for the last five brood years were highly variable in the upper Lemhi and Hayden priority areas (Figure 1.15). In all reaches, summer survival rates of parr were much higher than winter survival rates. Summer survival rates were generally higher in Hayden Creek than in the Lemhi River, with mean survival in Hayden Creek of 0.47 (SE = 0.09) and mean survival in the Lemhi River of 0.28 (SE = 0.06). Winter survival in the lower Lemhi priority area was more constant and usually higher than for fish staying in their natal reaches. Hayden Creek parr had an average winter survival rate in the lower Lemhi River of 0.22 (SE = 0.06), and upper Lemhi River parr had an average winter survival rate of 0.22 (SE = 0.05). The average winter survival rate of upper Lemhi River salmon in the upper Lemhi River was 0.08 (SE = 0.04). Winter survival of Hayden Creek salmon in Hayden Creek was higher than that of their cohort wintering in the lower Lemhi River only once (BY2016; Figure 1.15).

Lemhi River Watershed

Juvenile Chinook Salmon and Steelhead Emigration

Abundance of Chinook Salmon migrating past the RSTs by brood year declined during the last 5 years and that trend was similar among RSTs (Figure 1.16). Total brood year abundance estimated at the Upper Lemhi RST ranged from 4,376 fish (SE = 421) for brood year 2007 to 80,386 fish (SE = 3,317) for brood year 2014. Brood year abundance in Hayden Creek ranged from 3,369 fish (SE = 125) for brood year 2005 to 77,221 fish (SE = 5,599) for brood year 2014. Total emigrants from the basin estimated at the Lower Lemhi River RST ranged from 7,656 fish (SE = 745) for brood year 2006 to 79,130 fish (SE = 1901) for brood year 2014.

Abundance estimates of juvenile steelhead migrating past RSTs were relatively stable in the last five years. Over the last 15 years, abundances at the upper Lemhi and Hayden Creek RSTs were more variable than at the lower Lemhi River trap (Figure 1.16). Steelhead abundance at the upper Lemhi RST ranged from 6,825 fish (SE = 1,266) in 2019 to 35,715 fish (SE = 2,482) in 2008. Abundance at the Hayden Creek RST was lowest in 2006 with 469 steelhead (SE = 169), but that was the first year of trap operations, so the estimate only included the fall migration period. Considering all years with full operation, abundance estimates at the Hayden Creek RST ranged from 3,472 fish (SE = 405) in 2010 to 18,311 fish (SE = 971) in 2013. Basin-wide abundance estimates of steelhead migrating past the lower Lemhi River RST ranged from 5,501 fish (SE = 741) in 2014 to 47,485 fish (SE = 8,829) in 2008. Although no clear trend existed in abundance estimates at the upper Lemhi River and Hayden Creek RSTs, a negative trend was observed at the lower Lemhi River RST between brood year 2006 and 2014 and then remained relatively stable from 2015-2021.

Juvenile Chinook Salmon Survival

Survival from RST to LGR varied among subpopulations and brood years, but age-1 smolt survival rates were always greater than fall parr survival rates for both the upper Lemhi River and Hayden Creek subpopulations (Figure 1.17). Survival rates were relatively consistent over the full record although individual smolt survivals were sometimes unexpectedly low. The difference in survival between fall parr and smolts ranged from 0.10 to 0.52 (mean = 0.31) for the upper Lemhi River subpopulation and from 0.11 to 0.49 (mean = 0.36) for the Hayden Creek subpopulation. Mean age-1 smolt survival rates were 0.65 (SE = 0.07) for the upper Lemhi River smolts and 0.62 (SE = 0.11) for the Hayden Creek smolts. Fall parr survival rates were considerably lower at 0.34 (SE = 0.03) for upper Lemhi River fish and 0.27 (SE = 0.02) for Hayden Creek fish.

Adult Chinook Salmon and Steelhead Escapement

Chinook Salmon escapement past the downstream-most IPTDS into the Lemhi River ranged from 82 to 216 fish in the last 5 years, which is lower than average (Figure 1.18). The current level of low escapement began in 2016 and is similar to the 2010-2012 escapement. The upper Lemhi River spawning aggregate had higher estimates than Hayden Creek 2010-2019; but, in 2015, Hayden Creek had an estimated 361 adults (SE = 541) return, compared to the 325 adults (SE = 53) in the upper Lemhi River (Figure 1.19). The percentage of total escapement to the Lemhi River basin that migrated into Hayden Creek ranged from 14.1% in 2014 to 49.5% in 2015, with an average percentage of 30.2% from 2010 through 2019. Although the majority of fish spawn in Hayden Creek or the mainstem Lemhi River upstream of the Hayden Creek confluence, some Chinook Salmon spawn downstream of this area. The percentage of total escaping Chinook Salmon that spawned in the Lemhi River downstream of the Hayden Creek confluence ranged from 11% in 2010 and 2014 to 26.8% in 2019, with a 2010-2019 average of 13.6%.

Steelhead escapement past the IPTDS into the Lemhi River in the last five years ranged from 62 to 158 fish, which is less than previous levels (Figure 1.18). Escapement estimates ranged from 62 adults (SE = 15) in 2019 to 518 adults (SE = 47) in 2010. No clear trend was observed from 2010-2016. However, estimated steelhead returns decreased in 2017 and remained relatively low through 2020. The upper Lemhi River and Hayden Creek spawning aggregates were relatively similar among years (Figure 1.19). Tributary specific escapement estimates revealed steelhead spawned in Hayden Creek, as well as tributaries downstream from the mouth of Hayden Creek (Figure 1.20). Only a small proportion of total steelhead escaping to the basin spawned in the Lemhi River upstream of the Lemhi River Weir (LRW) IPTDS. The proportion of total escapement that spawned in the basin upstream of LRW ranged from 3.4% in 2010 to 23.3% in 2011, with a 2010-2020 average of 13.2%. Comparison of Figures 1.18, 1.19 and 1.20 suggests that most steelhead spawning likely occurs within the main stem in the lower Lemhi priority area.

Redd Counts

Chinook Salmon redd counts in the upper Lemhi River and Hayden Creek ranged from 41-85 and 12-40 in the last 5 years, respectively, which is lower than the long-term average (Figure 1.21). Since 2001, when redd counts were conducted in both Hayden Creek and the upper Lemhi River, the highest redd count in the basin was 426 in 2001. The lowest observed basin-wide total was 40 redds in 2004. In general, the number of redds declined from 2001 to 2007, then increased through 2015 when 310 total redds were observed. In years to follow, redd counts were much lower with a low of 55 total redds in 2017. The basin-wide average number of redds observed between 2001 and 2021 was 129 redds. Between 2001 and 2021, the proportion of redds observed in Hayden Creek ranged from 19% to 51% of the redds found in the Lemhi River basin. Redd counts provide a long time series of spawning activity and show a similar trend during the years that PIT-tag-based escapement estimates are available.

Spawning distribution did not increase substantially through time. In years with more spawning activity, the density of redds increased within certain areas of the Lemhi River and Hayden Creek (Appendix E), rather than in newly occupied reaches. The majority of redds in Hayden Creek were within 13 km of the confluence with the Lemhi River.

Chinook Salmon Productivity

Productivity varied considerably across years within the upper Lemhi River and Hayden Creek, and between spawning areas within years. Since 2017, productivity in the upper Lemhi River ranged from 32 smolts/redd for brood year 2017 to 84 smolts/redd for brood year 2019 (Figure 1.22). Productivity in Hayden Creek ranged from 10 smolts/redd from brood year 2018 to 76 smolts/redd for brood year 2019. In the last fifteen years, smolts per redd in the upper Lemhi River and Hayden Creek ranged from 6-84 and 10-130. When comparing productivity as the combined number of fall parr and smolts per redd, there was not a significant treatment effect (p = 0.873, Table 1.3). However, there was a significant treatment effect (p = 0.003) when expressing productivity as the number of age-1 smolts per redd. In years after juvenile Chinook Salmon had access to summer rearing areas in priority tributaries, more age-1 smolts were produced per redd (Mean = 55 smolts/redd) compared to years prior to restoration efforts (Mean = 21 smolts/redd, Figure 1.22).

DISCUSSION

Declines in anadromous fish abundance prompted significant investment in habitat restoration as a method to support fish recovery efforts in the Pacific Northwest (Barnas et al. 2015). Restoration actions in the Lemhi River basin led to an increase in the amount of accessible rearing habitat, particularly during low summer flows and during winter when specific habitat features are crucial for survival. These actions provide opportunities for fish to migrate in and out of habitats without delay and to increase their distribution. When anadromous fry and parr have access to newly available habitat, we expect them to use those habitats and not emigrate until the fall parr or smolt life stage, which we hypothesize will increase survival rates (Chapman 1966). Results from the Lemhi River IMW show that restoration actions have elicited detectable responses from habitat and fish. We documented three main results: 1) fish (mostly juvenile steelhead) moving into and out of reconnected tributaries; 2) the Lower Lemhi Rehabilitation Project is being used by juvenile Chinook Salmon and has induced a positive change in juvenile steelhead standing stock; and 3) a sustained increase in number of age-1 smolts per redd from the upper Lemhi priority area relative to Hayden Creek, the reference system. We discuss key results at the tributary, mainstem river, and watershed scale and relate them to restoration actions completed through 2021. Furthermore, we show how IMW results were used to adapt restoration strategies in the Lemhi River basin and the monitoring studies designed to evaluate them.

Reconnected Tributaries

Responses to restoration actions were exhibited primarily by juvenile salmonids. Prior to restoration efforts in the six priority tributaries, no juvenile Chinook Salmon were encountered during electrofishing surveys in the early to mid-2000s. However, intensive pretreatment monitoring surveys were limited. We documented juvenile Chinook Salmon during summer electrofishing surveys in all reconnected tributaries except for Hawley Creek. Juvenile Chinook Salmon were observed in tributaries following barrier removal projects and re-watering dry stream segments during 2009-2021. They have been observed in all subsequent years, indicating that juveniles produced in the upper mainstem river can access newly connected tributary habitats for rearing opportunities.

Tributaries can play an important role in Chinook Salmon rearing. The individuals we documented in tributaries during the spring may be avoiding deleterious effects of sediment loads during mainstem high flows (Scrivener et al. 1994). Fish observed in tributaries in the summer months are likely using cold water tributaries as thermal refugia when water temperatures in the Lemhi River exceed optimum thresholds (USSIRA 2018). Juvenile Chinook Salmon migrating into tributaries in the fall are likely seeking overwinter habitat (Swales et al. 1986; Bradford et al. 2001). A prime example is Little Springs Creek, where tagged juvenile Chinook Salmon immigrate into the tributary during October and November and do not emigrate out of the tributary until the following spring. This is consistent with other studies that have documented use of groundwater-influenced habitats by salmonids in winter (Swales et al. 1986; Cunjak 1996; Bradford et al. 2001; Giannico and Hinch 2003). Therefore, tributary habitats provide important refugia throughout the entire year to all juvenile life stages of Chinook Salmon.

We further evaluated the use of rearing habitat for juvenile Chinook Salmon by estimating standing stock in reconnected tributaries. In recent years, standing stock of juvenile Chinook Salmon has increased in some prioritized tributaries. Some of the increases in standing stock followed water conservation measures. However, adult escapement has not increased in the Lemhi River basin; therefore, an increase in juvenile Chinook Salmon standing stock is likely attributed to fish seeking rearing opportunities in good quality habitat. We also observed more juvenile Chinook Salmon in upper Lemhi River tributaries than in the lower Lemhi River tributaries. Very little Chinook Salmon spawning occurs in the Lemhi River downstream of the Hayden Creek confluence. As such, we would expect to observe fewer juvenile Chinook salmon in the lower Lemhi River tributaries than upper Lemhi River tributaries.

Reconnected tributaries also provide habitat essential for the conservation and recovery of Bull Trout (USFWS 2015) Historically, Bull Trout were widely distributed throughout the Lemhi River basin in fluvial and resident forms. The fluvial form is well documented in the Hayden Creek drainage, but over the past century, has likely been extirpated from the remaining Lemhi River tributaries. Bull Trout that spawn in Hayden Creek and its tributaries display strong site fidelity, returning to spawn every year or every other year. Since tributary reconnection efforts were implemented, fluvial adult Bull Trout have expanded their distribution, suggesting that tributary connectivity provides an opportunity for the fluvial life history to be re-expressed or fish are recolonizing newly available habitats. Previous studies in the upper Salmon River basin have suggested that interconnected stream habitats are important for the completion of various life history requirements (Schoby and Keeley 2011). Moreover, results from our Bull Trout monitoring are similar to that of anadromous fish and thus Bull Trout could be a potential surrogate to evaluate fish response to habitat actions during poor escapement years for steelhead and Chinook Salmon.

Adult steelhead escapement to the Lemhi River basin has remained low but adult fish were observed spawning in some reconnected tributaries. In Kenney Creek, steelhead spawning occurred in the lower 1.75 km prior to reconnection. We continue to document redds in this creek after reconnection occurred. In 2018, the completion of a source switch for a diversion in Bohannon Creek restored water flow to the lower portion of the creek and provided additional flow for fish migration. Following reconnection of Bohannon Creek, we did not observe an increase in the number of redds. This is likely attributed to the drought that occurred in 2020 and 2021 and low adult steelhead escapement to the Lemhi River basin. Adult steelhead have been detected on the Big Timber and Canyon creek IPTDSs, but we did not observe any steelhead redds during spawning ground surveys. Visibility of redds in tributaries during spring is typically good because high flows occur over a relatively short time; furthermore, some tributaries are spring-fed and flows relatively consistent. However, we have observed several adult steelhead spawning in Little Springs Creek. The first steelhead redd in Little Springs Creek was documented six years after the tributary was reconnected. One plausible explanation is that fish response to habitat restoration is not always immediate. Long-term monitoring is necessary to observe fish response to restoration actions (Minns et al. 1996; Louhi et al. 2016). Although total escapement of steelhead to the basin has decreased and no significant changes in tributary-specific escapement have occurred, adult steelhead have the opportunity to use habitat in newly restored locations.

Adult Chinook Salmon have not been observed spawning in reconnected tributaries in the Lemhi River basin. Although habitat quantity and connectivity are important predictors of salmon redd occurrence (Isaak et al. 2007), the relatively low abundance of adult Chinook Salmon may explain the lack of response to the reconnection of historically important spawning areas. Some studies have documented rapid recolonization of reconnected habitats by anadromous salmonids (Bryant et al. 1999; Anderson and Quinn 2007), but those have primarily been in drainages closer to the ocean and within proximity of a large source population that may have facilitated the exploitation of reconnected habitats (Kiffney et al. 2009). Where rapid recolonization has occurred following removal of passage barriers, spawning distribution was a function of distance from the source population, with highest redd densities occurring in the nearest suitable spawning habitat above the circumnavigated barrier (Kiffney et al. 2009). We would expect adults to occupy the best available habitat first and the mainstem Lemhi River has the highest intrinsic spawning potential (NOAA 2017). Therefore, we may not observe spawning in reconnected areas until escapement approaches or exceeds redd capacity in currently occupied spawning reaches and there is a greater impetus for colonizing new habitats.

Reconnected tributaries in the Lemhi River basin are not only important to fish production but also provide additional flow to the mainstem Lemhi River, which has multiple benefits. Early season tributary flow during the period of runoff (typically late-May) is strongly correlated with early life stage survival and egg-adult return rates in the Lemhi River basin (Arthaud et al. 2010). Cold-water inputs from tributaries improve rearing conditions in mainstem river reaches where temperatures often exceed optimal threshold levels during late summer (Null et al. 2009; IDEQ 2012; Ebersole et al. 2015). Most Lemhi River tributaries drain high-elevation, mountainous areas, which contribute cold water throughout the year. Furthermore, because changes in flow regime from climate change are anticipated (Isaak et al. 2012), the benefits of cold-water inputs are particularly important in the future (Justice et al. 2017; White et al. 2017). Beechie et al. (2013) further supports this concept, suggesting that even if the benefits are not evident now, they may become apparent in the future. Thus, cold-water tributary inputs may bolster the effects that mainstem restoration projects (e.g., floodplain connectivity and side channel habitats) have on fish in the Lemhi River.

Mainstem Lemhi River

Side channels and restored habitat in the Lower Lemhi River Rehabilitation Project provided opportunities for overwintering. Juvenile Chinook Salmon retention increased in side-channel habitats and the mainstem river since project implementation. In 2019, fall parr Chinook Salmon migrating downstream spent little time in newly constructed habitats. However, in 2020-2021, we observed fall migrants residing in constructed side-channel habitats in the treatment reach for longer durations of time. Side-channel habitats are important to early life stage rearing of salmon due to lower water velocities, riparian cover, and pool habitat (Swales et al. 1986; Morley et al. 2005; Carmichael et al. 2020). At a larger scale, juvenile Chinook Salmon were also observed rearing in the rehabilitated mainstem river in the treatment reach during the same period as more restored habitat became available. These are signs of increased juvenile retention in the Lemhi River basin through winter. Further evaluation is needed to assess if juvenile retention will increase with time and if it will result in increased standing stock, survival, and productivity.

Results of the BACI analysis of fish response to the Lower Lemhi River Rehabilitation Project were puzzling. Juvenile steelhead positively responded to the completion of the first few phases of habitat restoration in the lower Lemhi River in a time span of just four years. Juvenile steelhead densities increased in the treatment reach after multiple stages of habitat project enhancements (e.g., off channel habitat, large wood debris). Simultaneously, densities of both species in the reference reach decreased. The reference reach is downstream of the treatment reach and the main spawning reach for Chinook Salmon is upstream of both; hence, it is likely that the treatment reach retained juvenile salmon that otherwise might have moved to the reference reach, creating spatial variability that obfuscated the restoration effects (Rogers et al. 2022) and violating the assumption that the reaches are independent units. While some conclusions can be drawn from our BACI study design, the results are somewhat ambiguous. We do not have enough years pre- and post-restoration to observe a significant change in fish distribution, density, and abundance, which directly affects the power of a BACI design to detect a difference (Rogers et al. 2022). In more recent years, our reference and control sites have become treatment sites. One of the reasons the control site was chosen for restoration was because of the large numbers of juvenile Chinook Salmon emigrating into that reach from directly upstream. Also, during the study we observed years of poor adult escapement and thus fewer juvenile fish were sampled. Furthermore, most adult steelhead were likely spawning in the lower Lemhi River. Therefore, the positive steelhead response in relation to the Lower Lemhi River Rehabilitation project within the BACI design is likely explained by spawning distribution. Similarly, juvenile Chinook Salmon spawning occurs primarily in the upper Lemhi River and may explain why we did not see the same positive response in our BACI study.

Suitable winter habitat is a major limiting factor throughout the Lemhi River watershed. Parr that emigrated from Hayden Creek in the fall and overwintered in the lower Lemhi River had similar survival rates as the upper Lemhi River parr that overwintered in the same area ($S = 0.22$). These rates are very low; Smith and Griffith (1994) reviewed 24 studies of juvenile salmonids exposed to prolonged periods of 0°C temperatures that were not affected by winter floods and found average survival was 0.50 (SD 0.18). In comparison, Mitro and Zale (2002) found winter survival was 0.18-0.23 for age-0 *O. mykiss* in the Box Canyon reach of Henry's Fork, which is about 1900 m above sea level. Our overwinter survival results are consistent with a wealth of literature that identify the winter season as a potential seasonal survival bottleneck of salmonids (Mitro and Zale 2002; Letcher et al. 2002). Overwinter survival of juvenile fish can be influenced by a variety of factors including groundwater, water temperature, elevation, snowfall, channel type, and channel size (Annear et al. 2002; Brown et al. 2011). Interestingly, in the peer-reviewed literature, there is a lack of consistency on which riverine variables influence overwinter survival of juvenile fish, supporting the assumption that survival rates are habitat dependent, varying from system to system, and need to be viewed in that context (Huusko et al. 2007). Our results demonstrate low winter survival throughout the entire basin, indicating the need for mainstem habitat rehabilitation projects to bolster winter survival of juvenile salmonids.

Lemhi River Watershed

Chinook Salmon survival rates from the Lemhi River basin to LGR provide valuable information about life history diversity and help identify areas where recovery efforts should be directed. Age-1 smolts from the upper Lemhi River and Hayden Creek had much higher survival rates to LGR than did fall migrants from the same production areas. This is consistent with results from previous studies in the Lemhi River basin as well as other drainages in Idaho (Copeland et al. 2014). Our reach-specific survival results indicate that overwinter survival in the Lemhi River basin is low, and overwinter survival of fish in the upper Lemhi River is lower than fish that winter in the lower Lemhi River. In contrast, Uthe et al. (2017) found that overwinter survival in upper Lemhi River was greater than in the lower Lemhi River. Further evaluation is needed to better understand mechanisms that influence overwinter survival of fish in the upper and lower Lemhi River. What remains unknown is how overwinter survival in the Lemhi River compares to overwinter survival in the Salmon River for emigrants that overwinter downstream of the Lemhi River.

Survival of fall emigrants is a composite estimate of overwinter survival downstream of natal areas with survival during the final stage of migration to LGR in the following spring, whereas survival rate of spring emigrants is a true measure of survival from rearing areas to LGR. Partitioning survival to LGR into an overwinter component and spring migration component will enable us to understand if a true migration advantage exists for those individuals that rear in the Lemhi River for a full year and emigrate as age-1 smolts. Unfortunately, estimating overwinter survival of fall parr in the Salmon River is challenging due to limited monitoring infrastructure and lack of access. Therefore, we placed emphasis on mainstem Lemhi River rehabilitation to provide optimal winter habitat for juvenile Chinook Salmon retention and survival.

Improved habitat conditions through reconnected tributaries and mainstem Lemhi River rehabilitation have elicited a population-level productivity response. We hypothesized that reconnection and flow enhancement efforts in the upper Lemhi River basin would increase productivity of Chinook Salmon. Our results suggest that productivity of Chinook Salmon age-1 smolts increased throughout the monitoring period. Interestingly, we detected a significant treatment effect when considering only age-1 smolts per redd, but not when considering total emigrants per redd. This pattern suggests that the increase in smolt productivity may be the result of more fish remaining in the natal reach through winter, or higher winter survival of the fish that do stay, or a combination of both. To confirm, we plan to further investigate other metrics to develop a mechanistic basis for this relationship. We expect that fish using restored rearing areas will have improved fitness, given that increased streamflow reduces density-dependent constraints on growth and survival rates (Hartson and Kennedy 2015; Myrvold and Kennedy 2016). As the number of tagged juvenile Chinook Salmon immigrating into tributaries and restoration areas increases, we can test our prediction that rearing in newly connected habitats confers fitness advantages in the form of increased in-basin survival, as well as improved survival during downstream migrations through the hydrosystem.

The benefits we documented during the first 15 years of the IMW project (e.g., juvenile salmonids using reconnected habitats; Uthe et al. 2017) will likely take more time to accrue at the watershed-level. Low adult Chinook Salmon and steelhead escapement directly affects our ability to effectively estimate juvenile standing stock as it relates to habitat actions. It is unlikely that we will observe an increase in juvenile abundance if adult escapement to the Lemhi River basin remains low due to out-of-basin factors (e.g., ocean conditions and hydrosystem effects). While there is much to learn about fish response to reconnected tributaries, the more recent focus on improving quality of currently used habitat in the mainstem Lemhi River to address overwinter survival limitations may elicit a more rapid fish response in years of poor adult escapement.

Adaptive Management

Results from the Lemhi River IMW have been integral in shaping the monitoring framework as well as guiding prioritization and implementation of restoration projects in the basin. We frequently disseminated key monitoring results at Upper Salmon Basin Watershed Project technical team meetings to influence restoration implementation and provide up-to-date information for project ranking and planning efforts. We also attended annual coordination meetings among other IMW programs from the Pacific Northwest to assess how well sampling protocols were meeting objectives and develop sampling modifications to address shortfalls. Overall, communication and collaboration has been essential to adaptively managing the restoration and monitoring programs in the Lemhi River basin.

The flexibility of the IMW project, particularly the spatial hierarchy with effectiveness monitoring occurring at different scales, enabled us to add new restoration project components as results highlighted novel or previously unknown issues. Not only do our results provide information to guide restoration actions, but as new restoration projects and actions are implemented, we respond with changes in our monitoring framework. In the last five years, numerous habitat rehabilitation projects have been implemented, and our monitoring framework has adapted to include these projects in our evaluations (e.g., by additional IPTDSs and electrofishing surveys).

Adaptations in monitoring efforts occurred initially at the tributary scale. After reconnecting all priority tributaries, we modified our sampling design to further evaluate fish response to tributary reconnection. We continued to monitor fish movement, distribution, and abundance in tributaries but wanted to understand life-stage-specific habitat use. During our continuous backpack electrofishing efforts, we partnered with Merck Animal Health, Aqua (formerly Biomark, Inc.) to describe specific habitat metrics (e.g., pool, riffle, discharge, substrate, large woody debris) associated with the location of individual fish sampled. Sampling efforts took place over a two-year period and data are currently being analyzed. Information gathered from this study will help us better understand life-stage specific habitat requirements to guide future habitat project implementation.

In recent years, monitoring efforts focused on large-scale restoration projects in the mainstem river to evaluate use and associated survival. In doing so, we implemented a similar adaptive management approach. The sample design to monitor fish response to the early stages of the Lower Lemhi River Rehabilitation project (2016-2020) used a BACI design, where we used electrofishing methods to sample fish. However, due to the number, location, and complexity of habitat projects in the lower Lemhi River, the monitoring study design was recently modified to use multiple sampling methods to evaluate juvenile fish rearing (specifically overwintering) and survival in specific habitat projects. We are particularly interested in the use of off-channel habitats (i.e., side channels and floodplains) for rearing and are primarily monitoring fish distribution using IPTDSs. Habitat projects included in this modified study design include Lower Lemhi Rehabilitation Project and the Henry's Project (located in the former control reach downstream from the confluence of the Lemhi River and Hayden Creek). Project-level monitoring is crucial to determine if restoration actions have been effective in eliciting a fish response (Roni et al. 2005). Furthermore, our current monitoring efforts will help us better understand how habitat actions influence fish response at a watershed scale.

Other project-specific monitoring studies include evaluating the ecological benefits and risks to salmonids from implementing beaver dam analogs (BDAs). Using BDAs as a restoration tool in the upper Salmon River watershed has become a popular method to reduce water velocity, increase floodplain connectivity, activate secondary side channels, and increase juvenile fish rearing habitat. In Hawley Creek, BDAs were installed to improve habitat conditions for native salmonids. However, there is a concern that BDAs could provide the opportunity for Brook Trout (*Salvelinus fontinalis*) expansion, which are a threat to Bull Trout (USFWS 2015). Therefore, this study focuses on fish response to BDAs to describe and quantify species abundance, densities, distribution, and movement. The study will also compare growth and survival of salmonids and describe habitat preference among species. Results from this study will help inform project managers and guide future restoration as to the location, scale, and extent of BDA implementation.

The Lemhi River IMW results have been critical to restoration planning, prioritization, and implementation. The knowledge gained in the Lemhi River basin has been applied to restoration planning in the Pahsimeroi River and upper Salmon River drainages (USSIRA 2018). Overwinter survival in the Lemhi River is the primary limiting factor for Chinook Salmon (ISEMP and CHaMP 2015). Tributary reconnections and improving summer rearing habitat remain a major focus of restoration actions, but recent projects provide overwintering habitats in the lower mainstem Lemhi River, where habitat quality is poor. Given that tributary reconnections are increasing habitat quantity and fish must use that new habitat to receive the benefits of restoration, it is unlikely that we will observe this benefit if adult escapement remains low. Therefore, a focus on improving quality of currently-used habitat should provide a more immediate benefit to the fish, and thus a more immediate response. Overall, the Lemhi River IMW helps identify limiting factors and those results are integrated into the planning process to guide specific project actions. Further, we modify and add to our monitoring framework to assess the effectiveness of those projects and treatment types.

Over the next 10 to 15 years, the Lemhi River IMW will provide valuable insight into the effectiveness of freshwater habitat restoration actions in the Upper Salmon basin. Restoration planned within the next five years should result in the completion of the Lower Lemhi Rehabilitation project and Henry's Project. Post-treatment monitoring should occur for a minimum of five years to better understand how species respond to different treatment types, what specific life stages should we focus on creating habitat for, and where spatially should we focus our restoration efforts. Therefore, we need to continue and adaptively manage monitoring for another 10 to 15 years to enable sufficient post-treatment evaluation following successful completion of this suite of major restoration milestones.

CONCLUSION

Our results demonstrate that restoration efforts in the Lemhi River basin have been substantial enough to elicit local responses of multiple species and life stages of salmonids. Clearly, restoration actions have increased the abundance and distribution of salmonids at varying spatial scales and have provided life stage opportunities in higher quality habitats. The indication that survival of age-1 Chinook Salmon smolts may be increasing as a result of habitat actions in the upper Lemhi River underscores the importance of maintaining the existing IMW monitoring framework in the future. To date, the responses to restoration that we have documented are encouraging, but full understanding of fish population and habitat responses in the Lemhi River will require monitoring multiple anadromous fish generations for an additional 10 to 15 years.

<u>PART 1 TABLES</u>

Table 1.1. Site metadata for instream PIT tag detection systems in the Lemhi River basin. The site acronyms for sites used in PTAGIS are given (PTAGIS code).

Stream	Site Location	Installation Date	PTAGIS Code	Latitude	Longitude
Lemhi River	Lower Lemhi River	8/18/2009	LLR	45.176	-113.885
Lemhi River	Lemhi River Weir	8/18/2009	LRW	44.866	-113.625
Hayden Creek	Mouth	8/19/2009	HYC	44.861	-113.632
Big Timber Creek[a]	Upstream of Hwy 28	2/25/2010	BTC	44.688	-113.37
Kenney Creek	Mouth	6/1/2010	KEN	45.027	-113.658
Canyon Creek	Mouth	11/12/2010	CAC	44.692	-113.355
Little Springs Creek	Mouth	6/14/2011	LLS	44.781	-113.545
Bohannon Creek	Mouth	12/6/2011	BHC	45.112	-113.747
Wimpy Creek[d]	Mouth	10/21/2013	WPC	45.098	-113.721
Agency Creek[c]	Mouth	10/21/2013	AGC	44.957	-113.639
Big Timber Creek	Mouth	10/21/2013	BTL	44.698	-113.374
Hawley Creek[b]	Hawley/Eighteenmile Confluence	10/21/2013	HEC	44.669	-113.311
Lee Creek[d]	Mouth	12/21/2013	LCL	44.747	-113.475
Hayden Creek	Hayden/Bear Valley Confluence	9/15/2014	HYB	44.772	-113.708
Big Eightmile Creek[d]	Mouth	9/15/2014	LB8	44.738	-113.463
Big Springs Creek	Mouth	9/15/2014	LBS	44.727	-113.433
Big Timber Creek[d]	Upper section of creek	9/15/2014	BTU	44.614	-113.397
Eighteenmile Creek	Upstream of Hwy 29	9/15/2014	18M	44.683	-113.353
Big Timber Creek[d]	Middle section of creek	5/15/2015	BTM	44.661	-113.378
Eagle Valley Upper	Lower Lemhi River	7/18/2019	EVU	45.1	-113.726
Eagle Valley Lower	Lower Lemhi River	9/21/2019	EVL	45.115	-113.774

[a] Discontinued 2013.
[b] Discontinued 2015.
[c] Discontinued 2016.
[d] Discontinued 2018.

Table 1.2 Analysis of Before-After Control Impact design in the lower Lemhi River as a measure of juvenile Chinook Salmon (top) and steelhead (bottom) densities in the reference reach, treatment reach, and control reach before and after habitat restoration implementation.

Factor	**df**	**Sum square**	**Mean square**	**F-value**	***P***
		Chinook Salmon			
Reach	2	8389735	4194867	16.914	0.00089
Period	1	8520	8520	0.034	0.85706
Reach:Period	2	140351	70176	0.283	0.76002
Residuals	9	2232125	248014		
		Steelhead			
Reach	2	2189492	1094746	13.96	0.00174
Period	1	1955537	1955537	24.94	0.00075
Reach:Period	2	2010811	1005406	12.82	0.00232
Residuals	9	705748	78416		

Table 1.3 Analysis of productivity in the upper Lemhi River as measure of total fall parr and age-1 smolts per redd (top) and a measure of age-1 smolts per redd (bottom).

Factor	**Estimate**	**SE**	***df***	***t*-value**	***P***
		Fall Parr and Smolts			
Intercept	284.223	46.644	12	6.093	5.39E-05
Slope	-0.071	0.121	12	-0.587	0.568
Time period	-12.892	79.011	12	-0.163	0.873
		Smolts			
Intercept	48.349	7.668	12	6.305	3.92E-05
Slope	0.242	0.167	12	1.45	0.172
Time period	-40.481	11.093	12	-3.649	0.003

PART 1 FIGURES

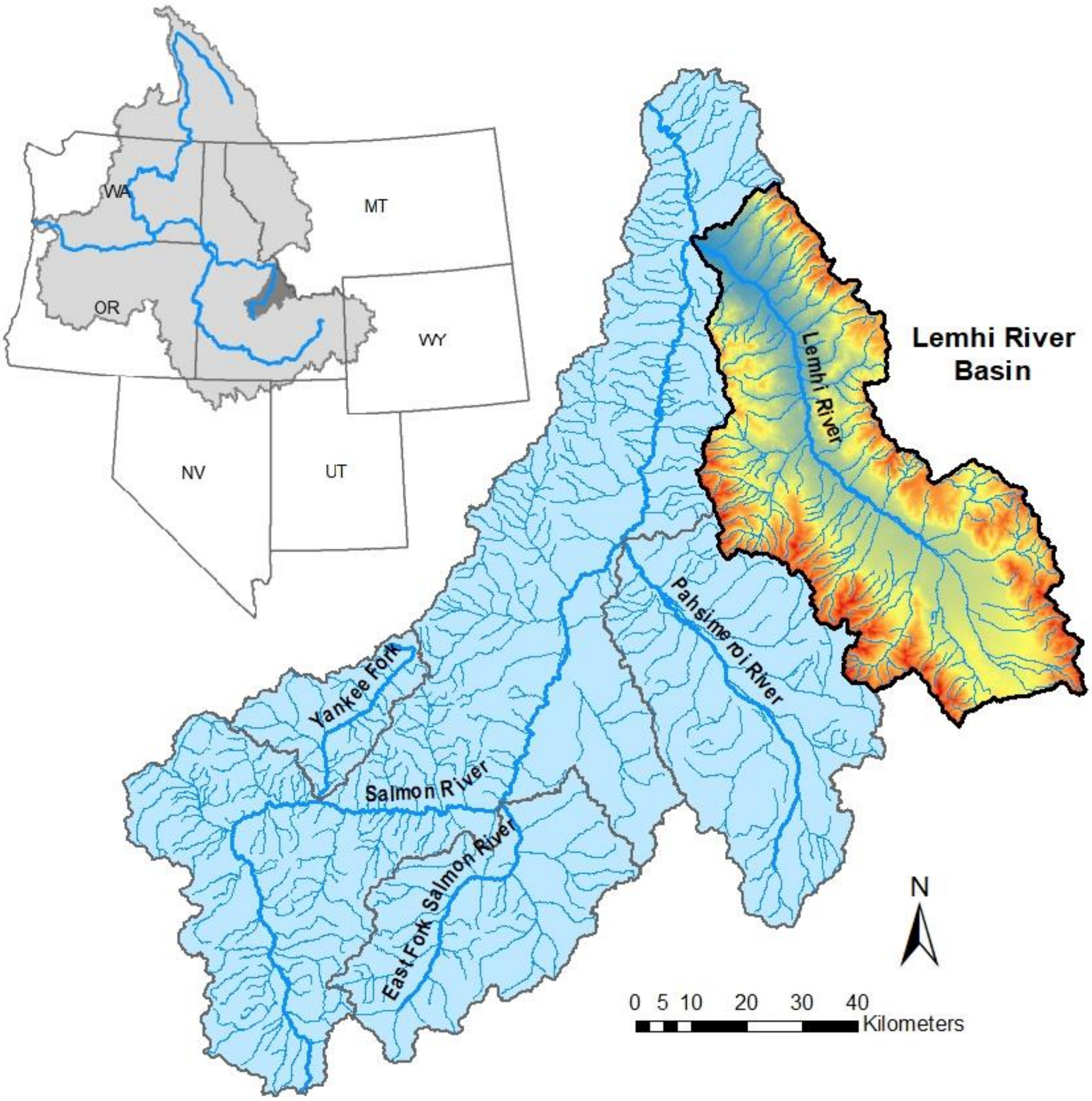

Figure 1.1. Location of the Lemhi River basin in the upper Salmon River drainage, Idaho.

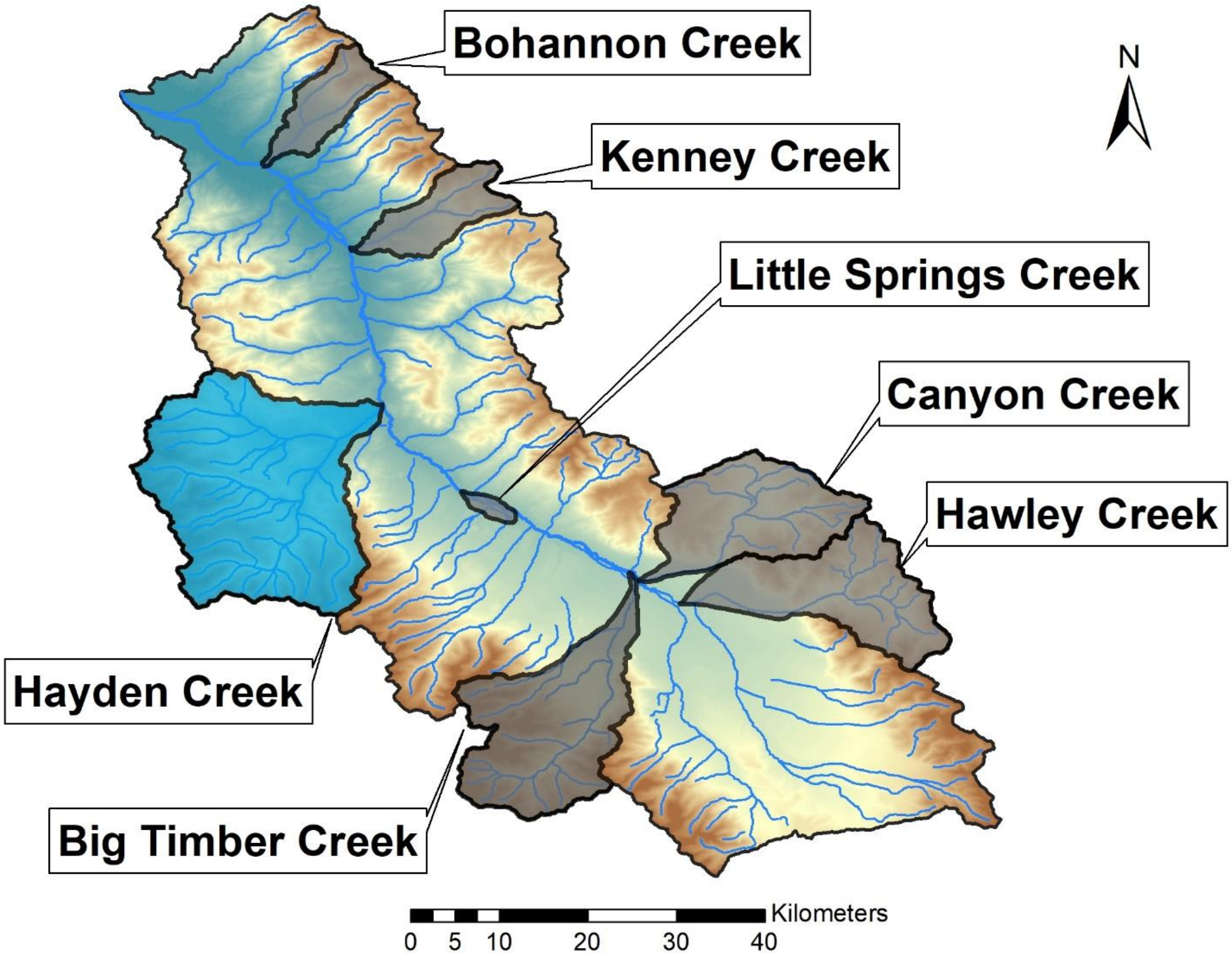

Figure 1.2. Priority tributaries that have been reconnected (grey) and the reference tributary, Hayden Creek (blue).

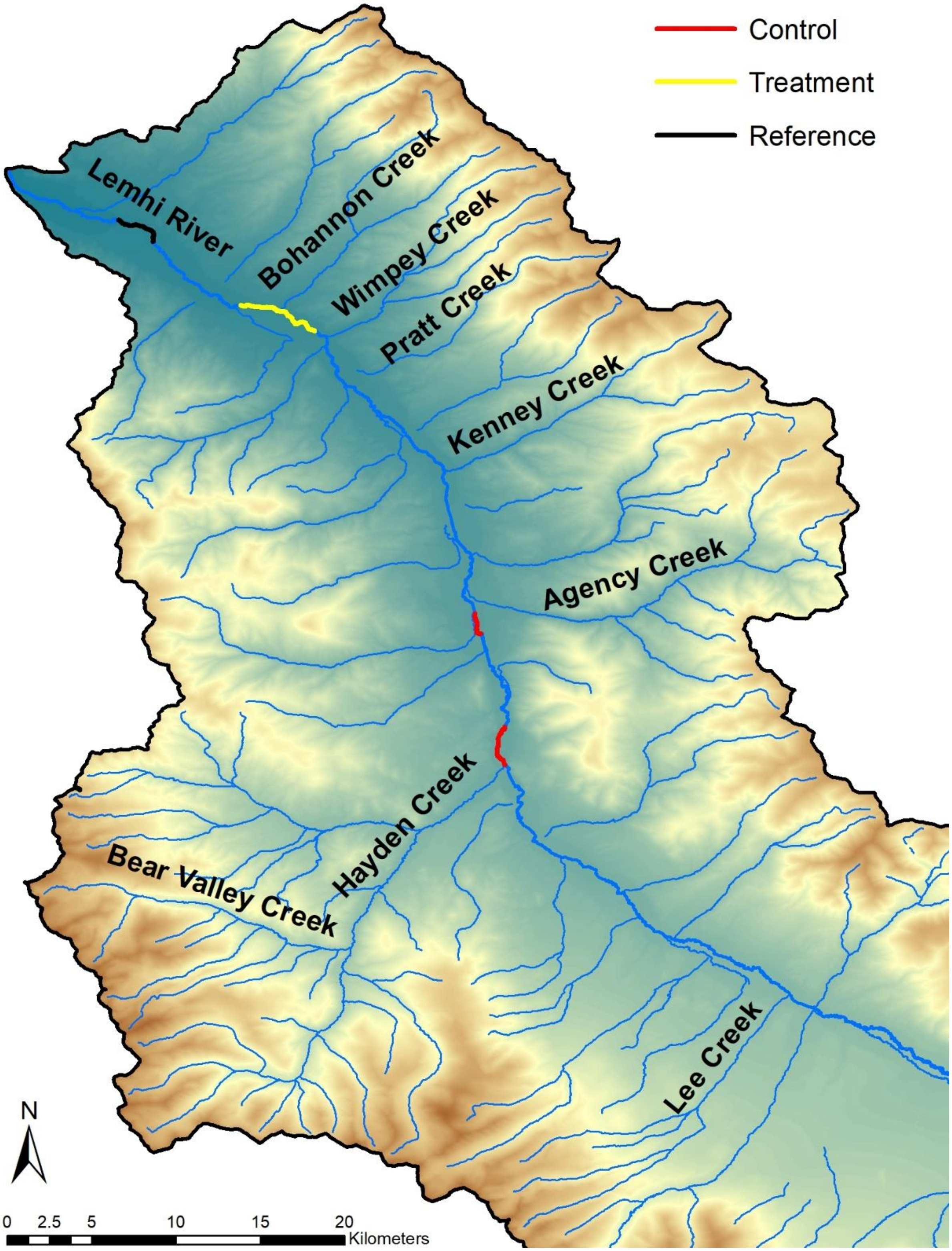

Figure 1.3. Control (red), treatment (yellow), and reference (black) reaches surveyed in the lower Lemhi River as part of the study design for project-level monitoring.

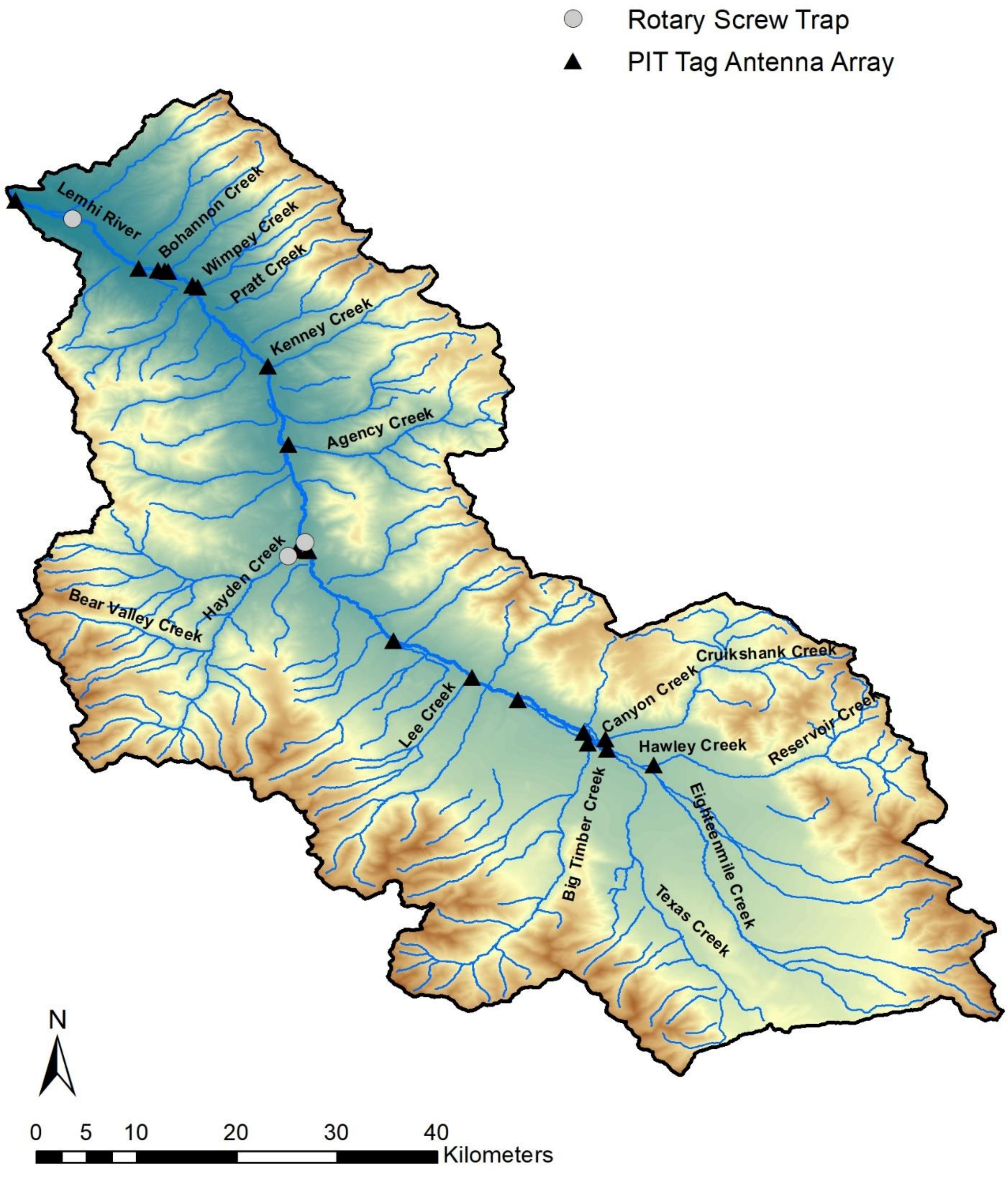

Figure 1.4. Locations of instream PIT tag detection systems (triangles) and rotary screw traps (circles) installed in the Lemhi River basin.

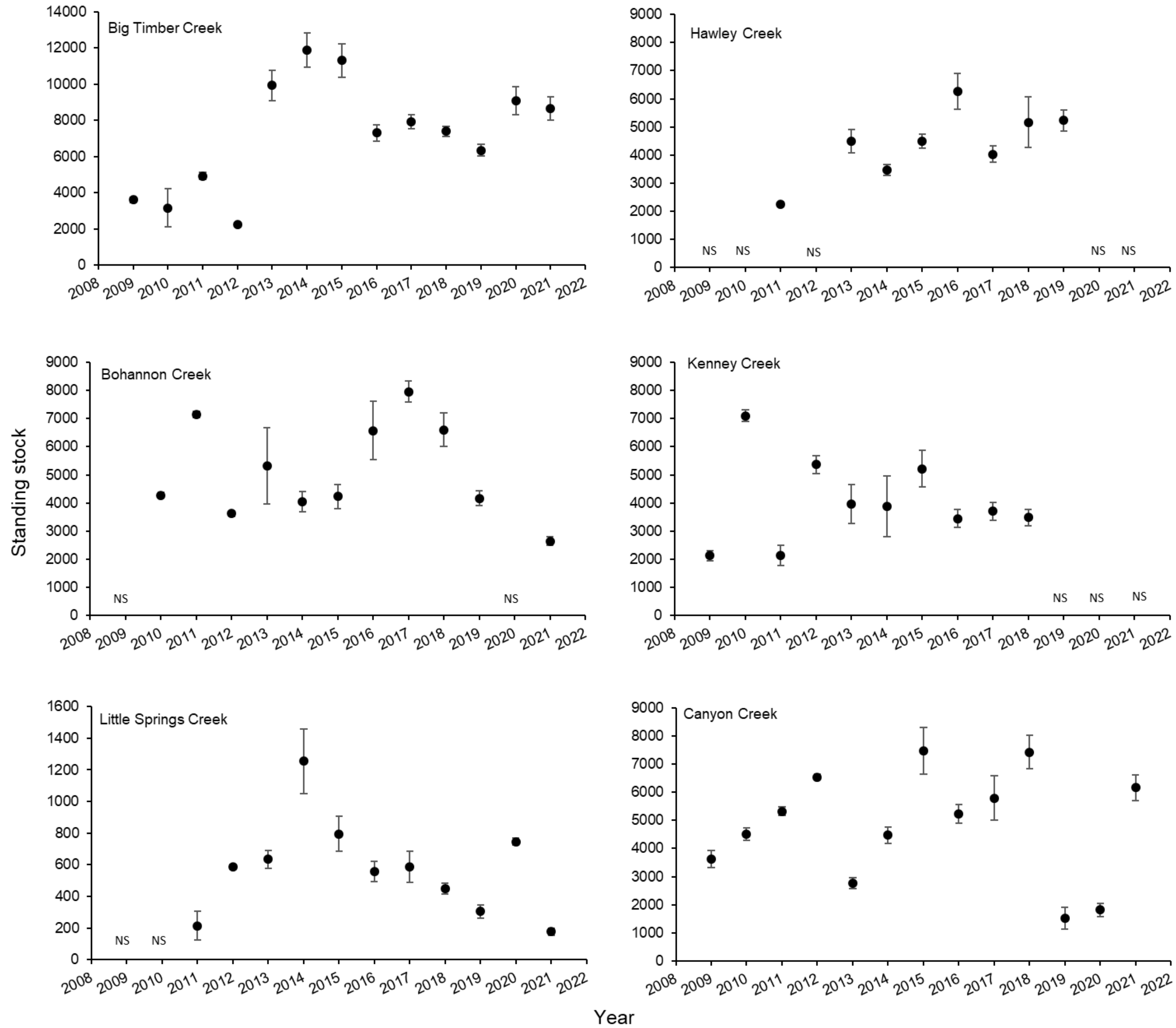

Figure 1.5. Estimates of juvenile steelhead standing stock in the six priority tributaries. Estimates are shown with standard error. NS = not sampled or sample size too small to estimate standing stock.

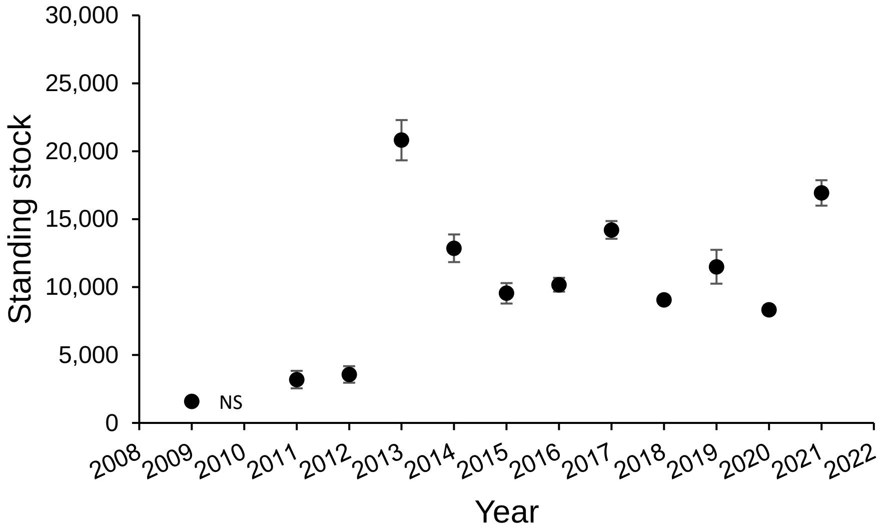

Figure 1.6. Estimates of juvenile steelhead standing stock in Hayden Creek, the reference tributary. Estimates are shown with standard errors. NS= not sampled.

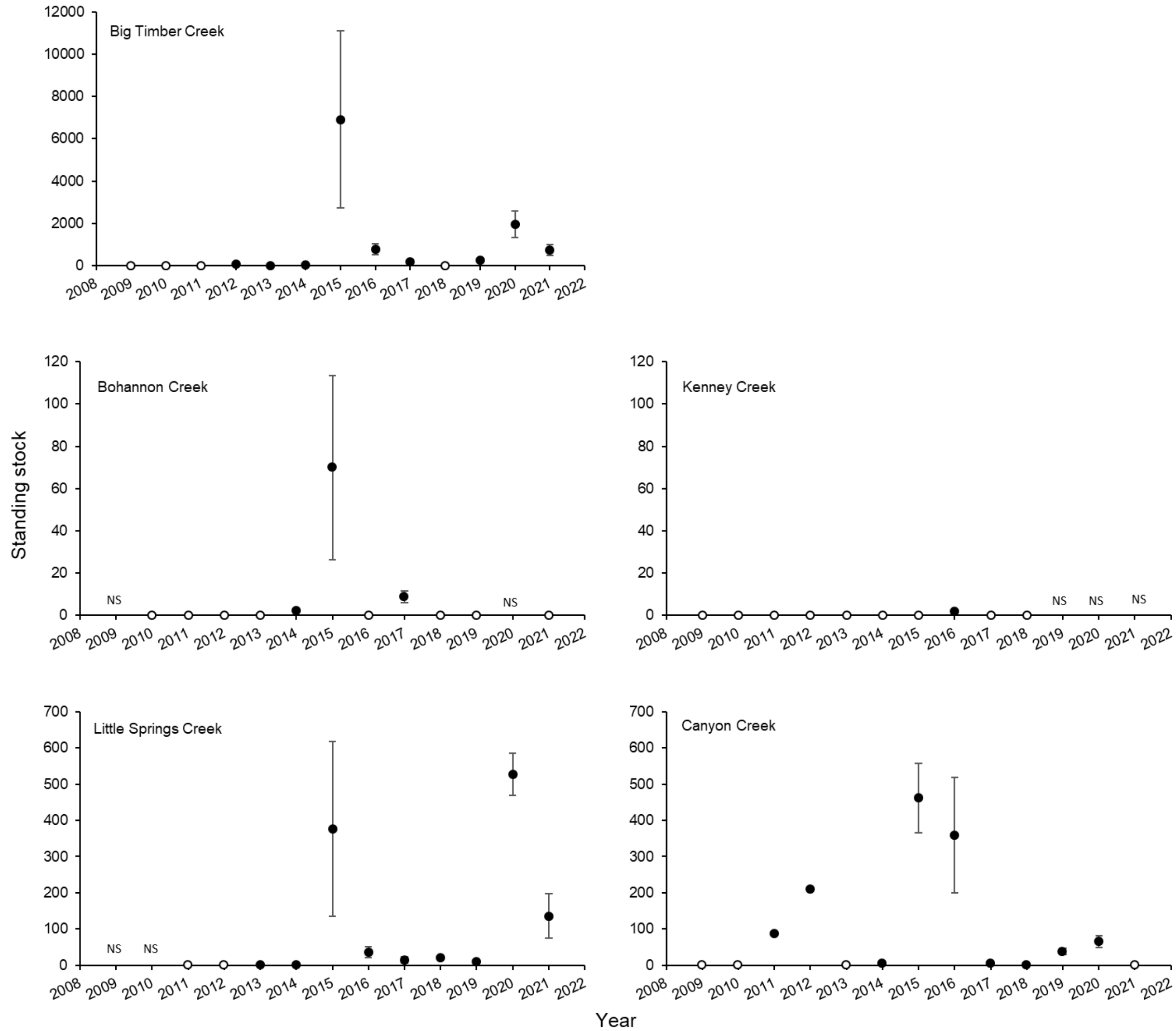

Figure 1.7. Estimates of juvenile Chinook Salmon standing stock in five of the six priority tributaries. Estimates are shown with standard error. NS = not sampled or sample size too small to estimate standing stock.

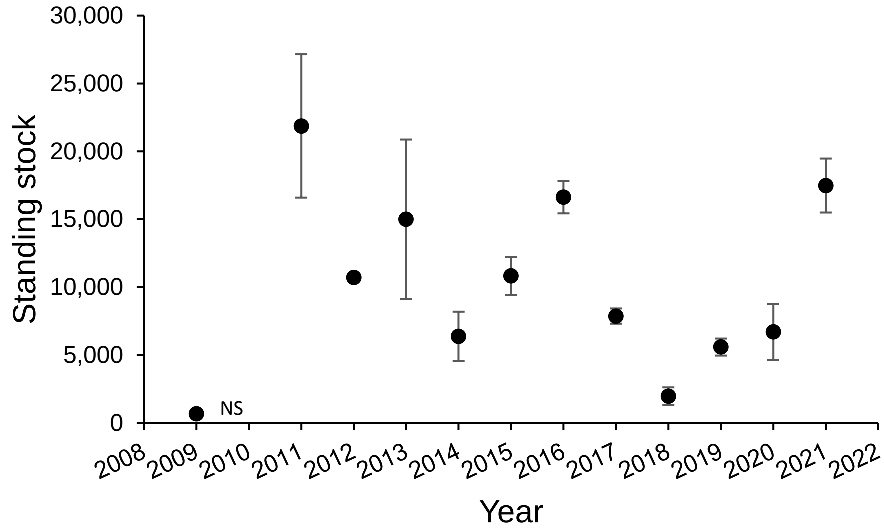

Figure 1.8. Estimates of juvenile Chinook Salmon standing stock in Hayden Creek, the reference tributary. Estimates are shown with standard errors. NS= not sampled.

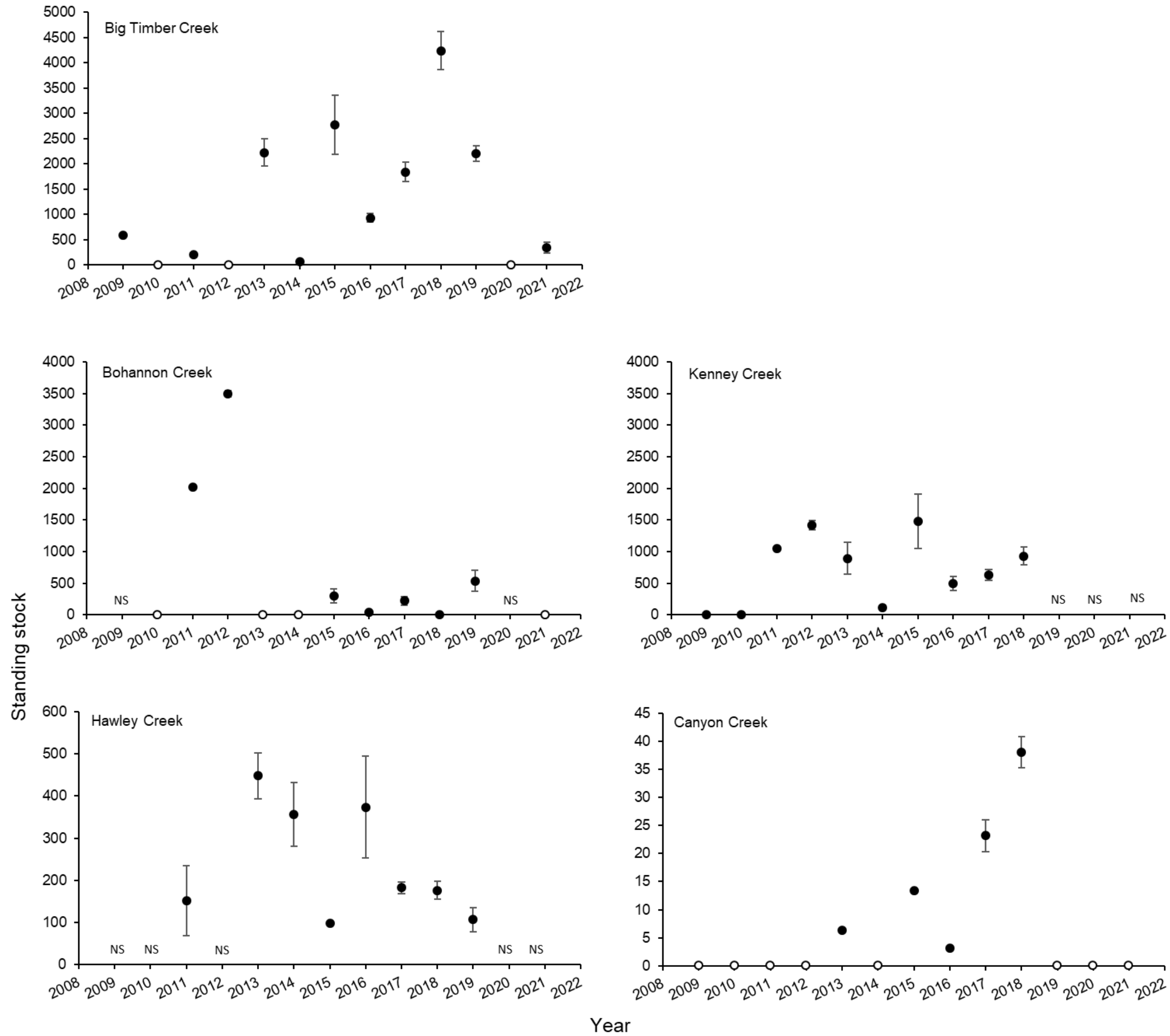

Figure 1.9. Estimates of Bull Trout standing stock in five of the six priority tributaries. Estimates are shown with standard error. NS = not sampled.

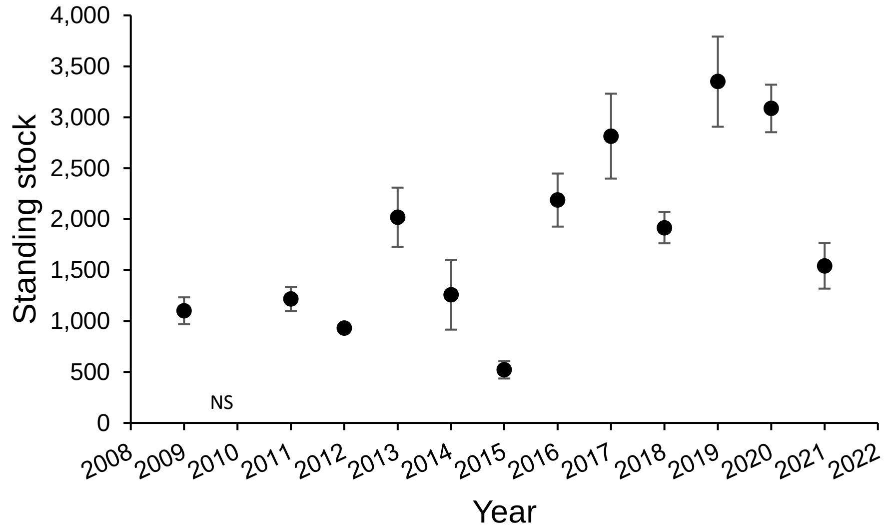

Figure 1.10. Estimates of Bull Trout standing stock in Hayden Creek, the reference tributary. Estimates are shown with standard errors. NS= not sampled.

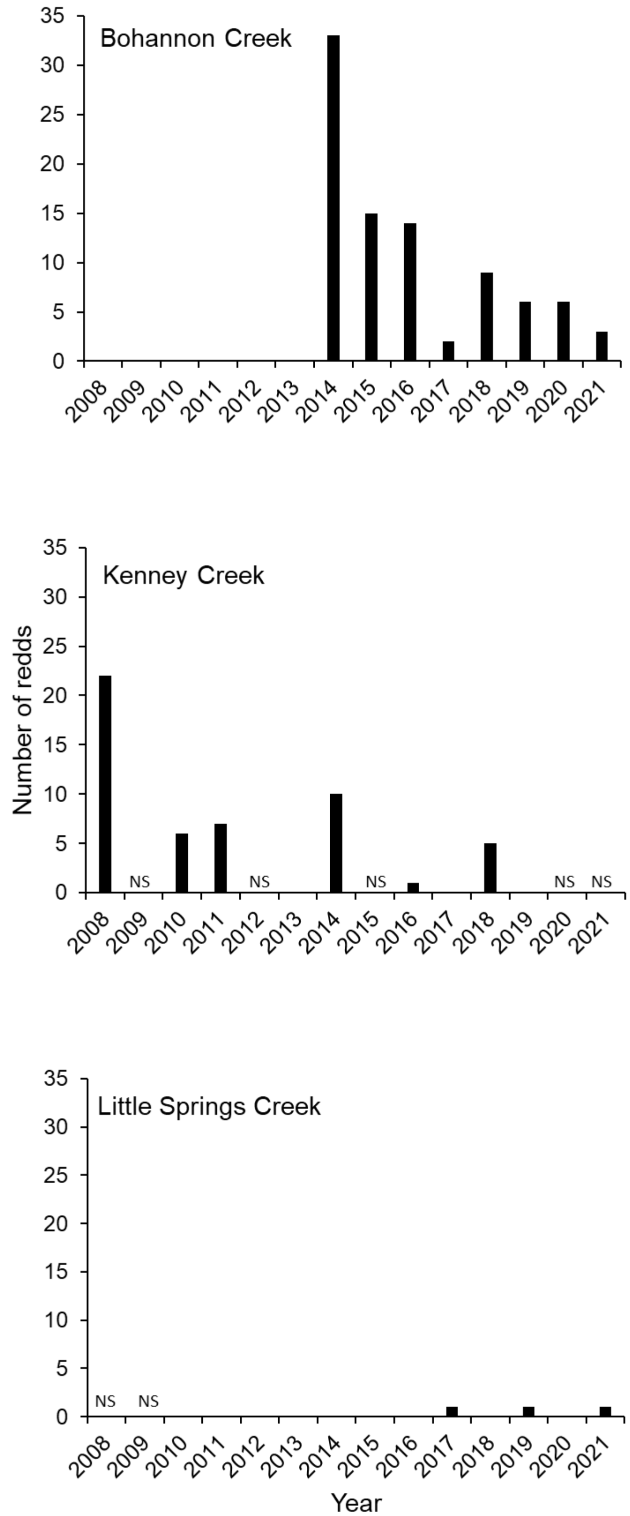

Figure 1.11. Redd counts from annual steelhead spawning ground surveys in three of the priority tributaries. Canyon, Big Timber, and Hawley were omitted because no steelhead redds were observed. NS = not sampled.

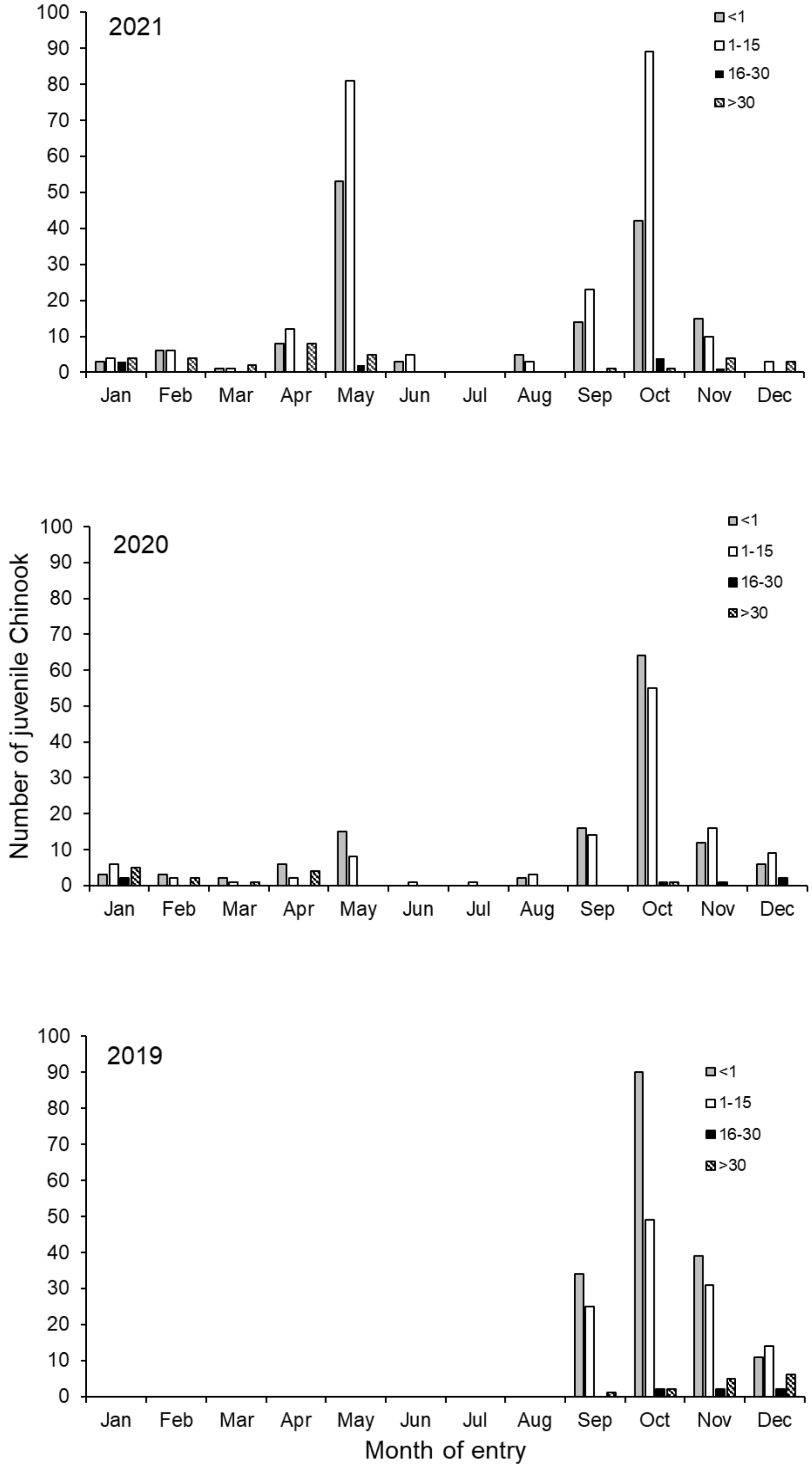

Figure 1.12. The number of tagged juvenile Chinook Salmon that resided between the Eagle Valley Upper and Eagle Valley Lower instream PIT tag detection systems from 2019-2021 starting with the month of entry and the retention time. Chinook Salmon retention included less than one day, one to 15 days, 16 to 30 days, and greater than 30 days.

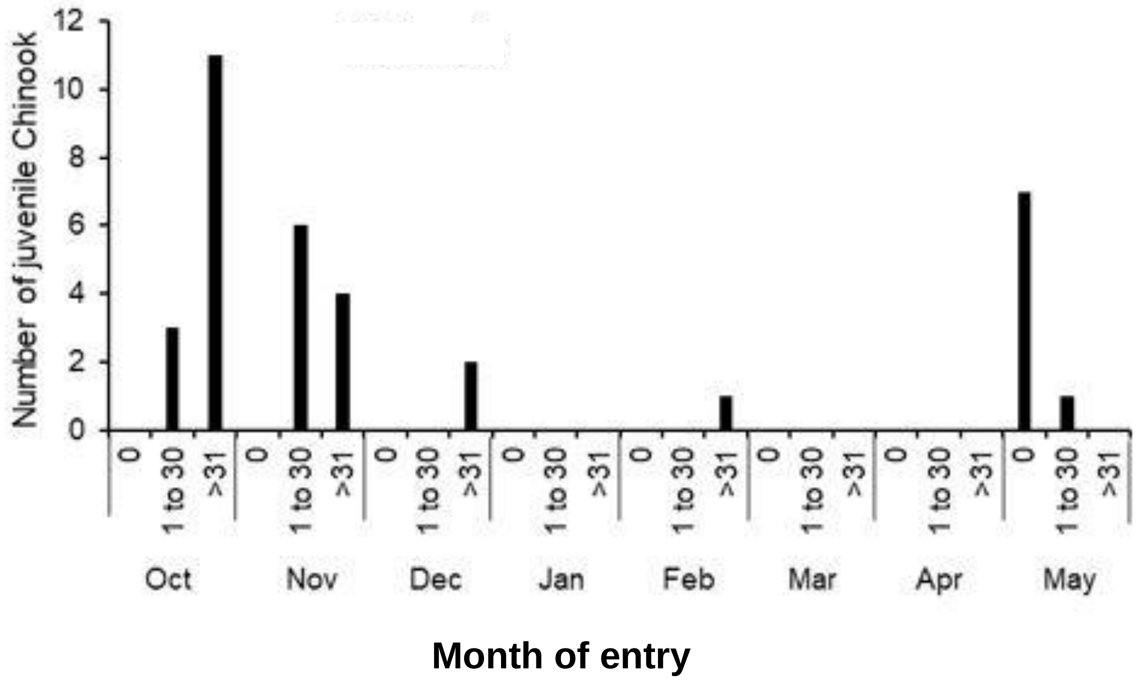

Figure 1.13. Juvenile Chinook Salmon that resided in a side channel in the lower Lemhi River between October 2020 and May 2021. The figure shows the number of Chinook Salmon that entered the side channel by month and the amount of time spent in the side channel (0, 1 to 30, and >31 days).

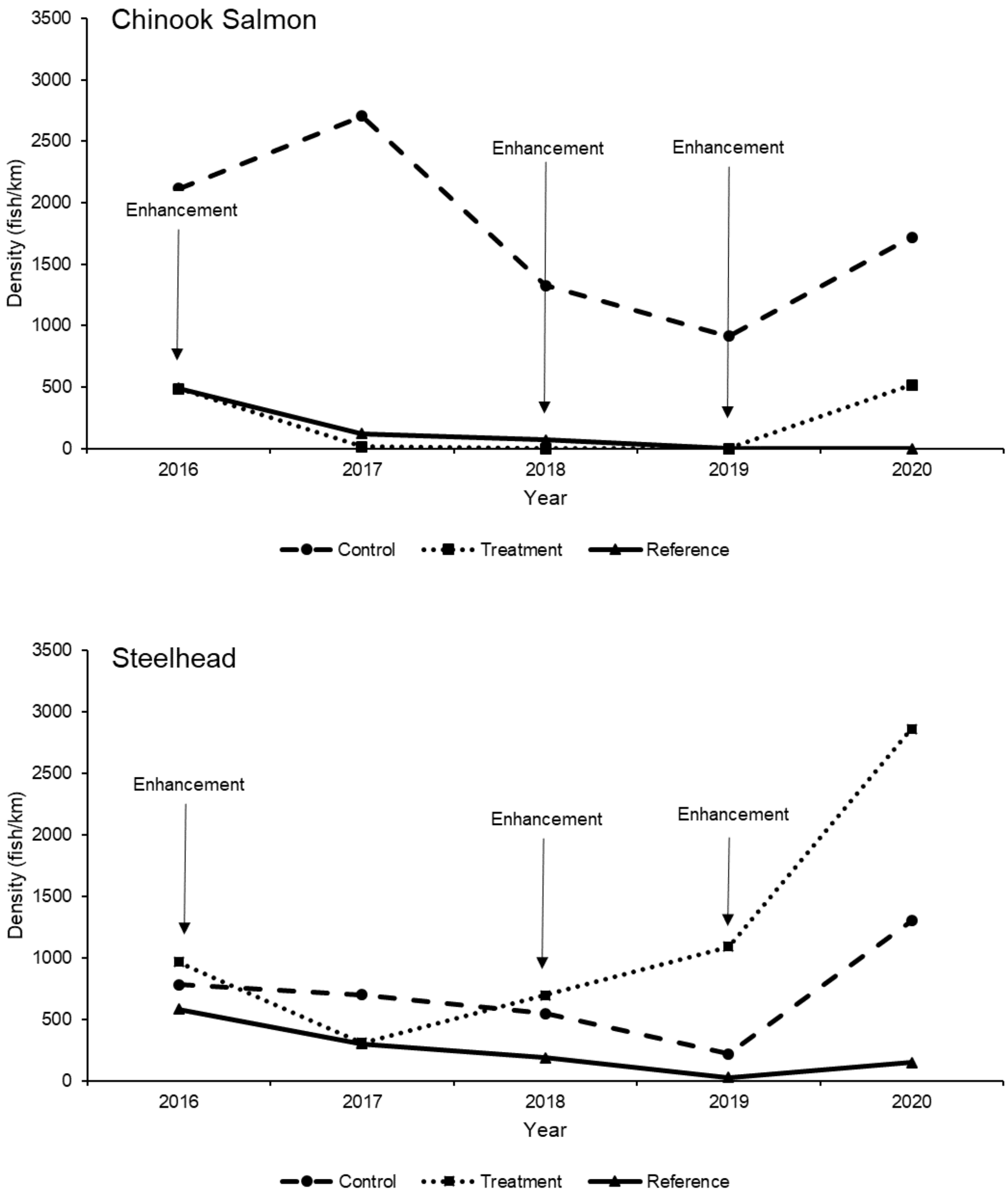

Figure 1.14. Juvenile Chinook Salmon (top panel) and steelhead (bottom panel) densities (fish/km) in the control, treatment, and reference reaches surveyed via electrofishing in the lower Lemhi River from 2016-2020. Multiple habitat enhancements (i.e., habitat structures, expanded floodplain, and constructed side channels) were completed in the treatment reach in 2016, 2018, and in 2019.

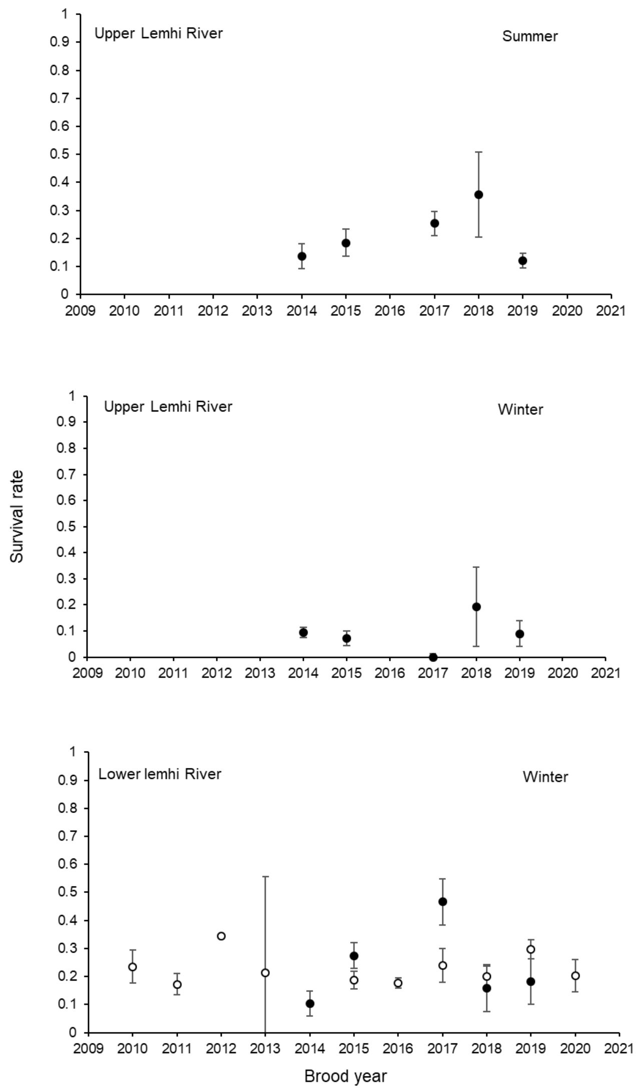

Figure 1.15. Survival of juvenile Chinook Salmon tagged in the upper Lemhi River (filled circles) and Hayden Creek (hollow circles) by brood year during the summer (top row) and winter (middle row). The lowest panel shows winter survival in the lower Lemhi River of fish that left their natal reaches. Error bars show standard error.

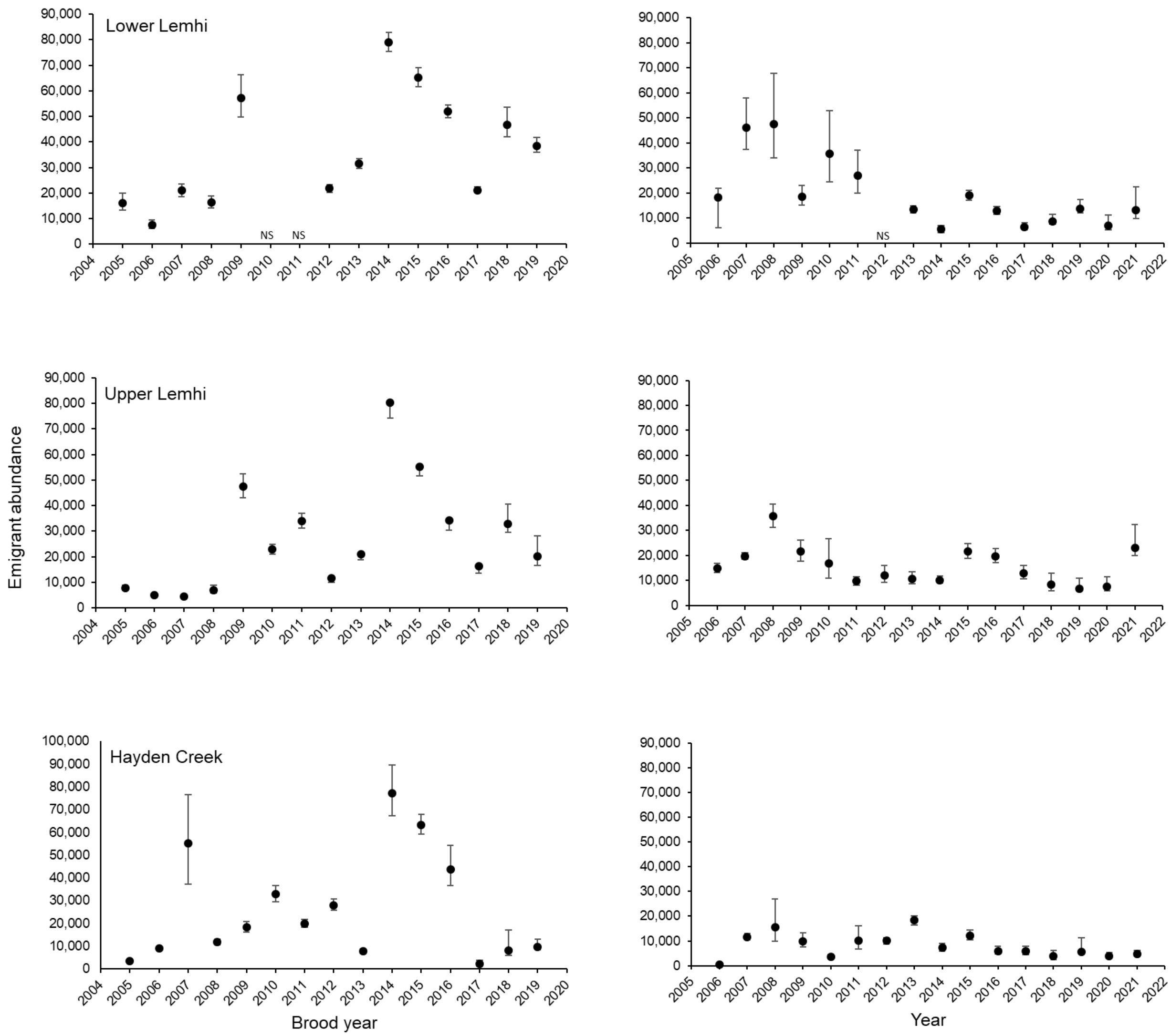

Figure 1.16. Estimates of emigrant abundance at rotary screw traps for Chinook Salmon (left panel, by brood year) and steelhead (right panel, by calendar year) migrating from the lower Lemhi River (top row), upper Lemhi River (middle row), and Hayden Creek (bottom row). Estimates are shown with 95% confidence intervals. NS = not sampled.

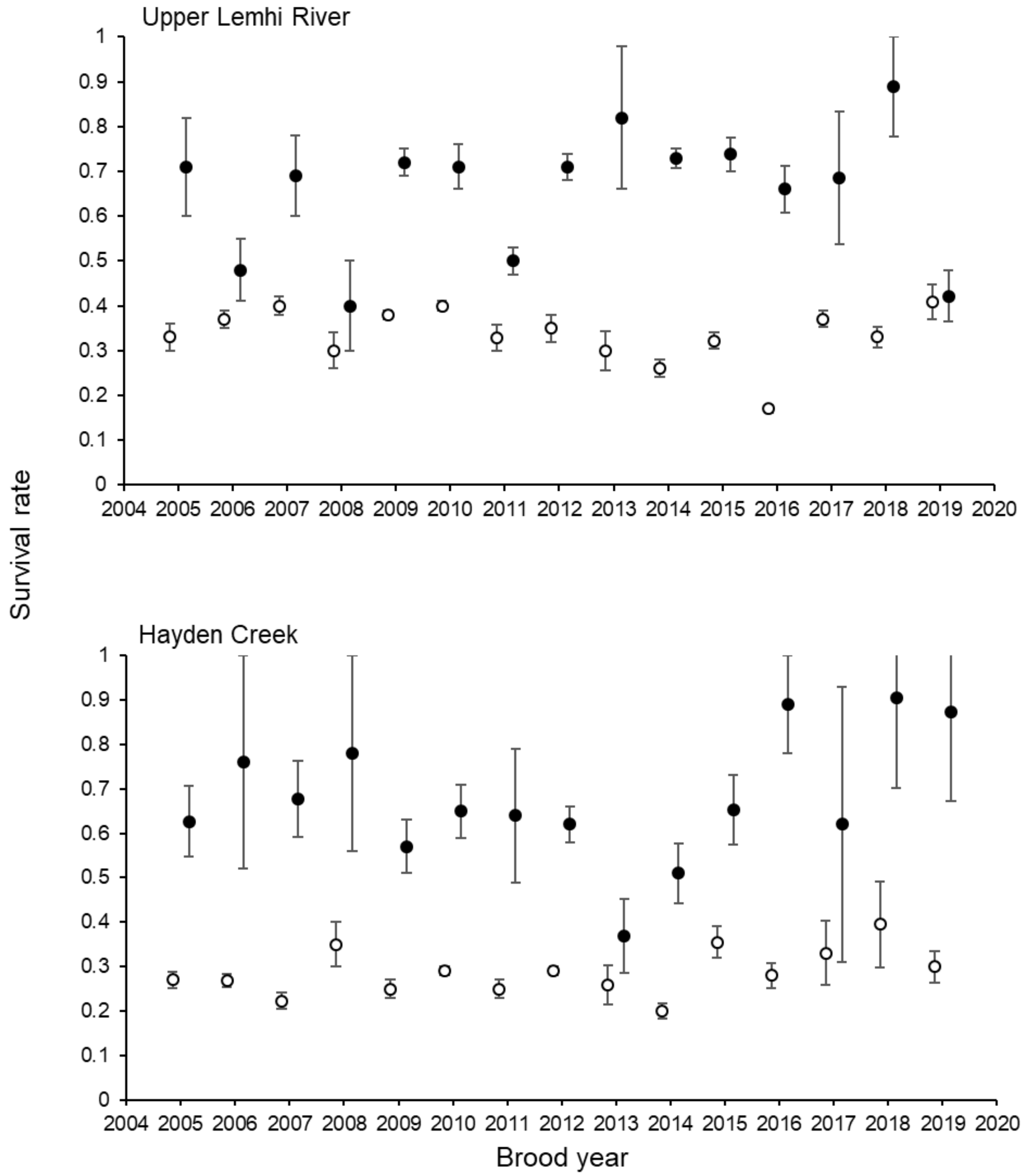

Figure 1.17. Survival rates to Lower Granite Dam of juvenile Chinook Salmon by life stage at rotary screw traps for brood years 2005-2019. Survival of fall parr (open) and age-1 smolt (closed) were estimated from the upper Lemhi River (top) and Hayden Creek (bottom). Estimates are shown with standard errors.

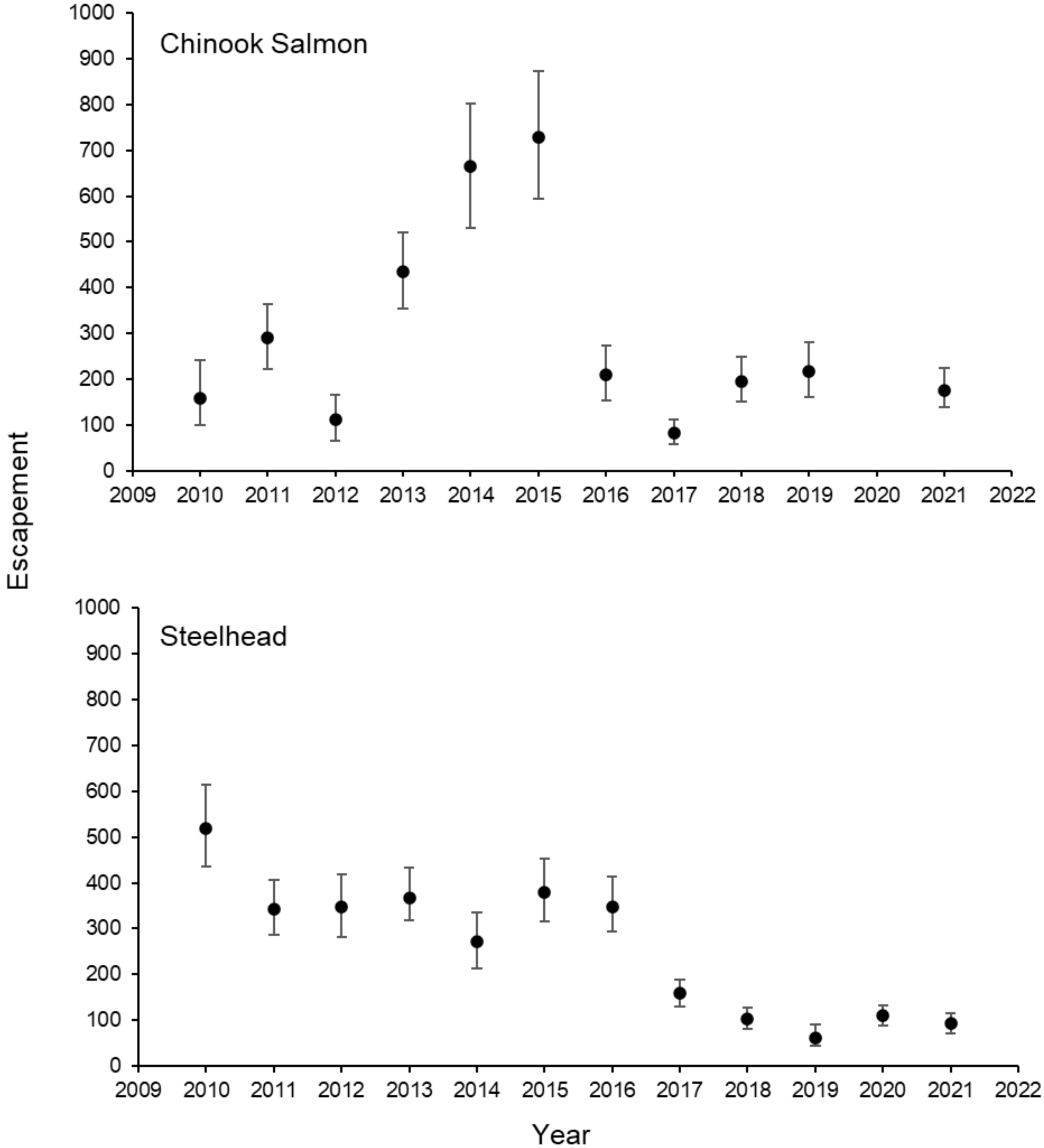

Figure 1.18. Estimates of escapement by spawn year to the lower Lemhi River instream PIT tag detection system of adult Chinook Salmon (top) and steelhead (bottom). Estimates are shown with 95% confidence intervals (Kinzer et al. 2020). There were no Chinook Salmon escapement estimates for 2020 because the Lower Granite Dam adult trap did not operate.

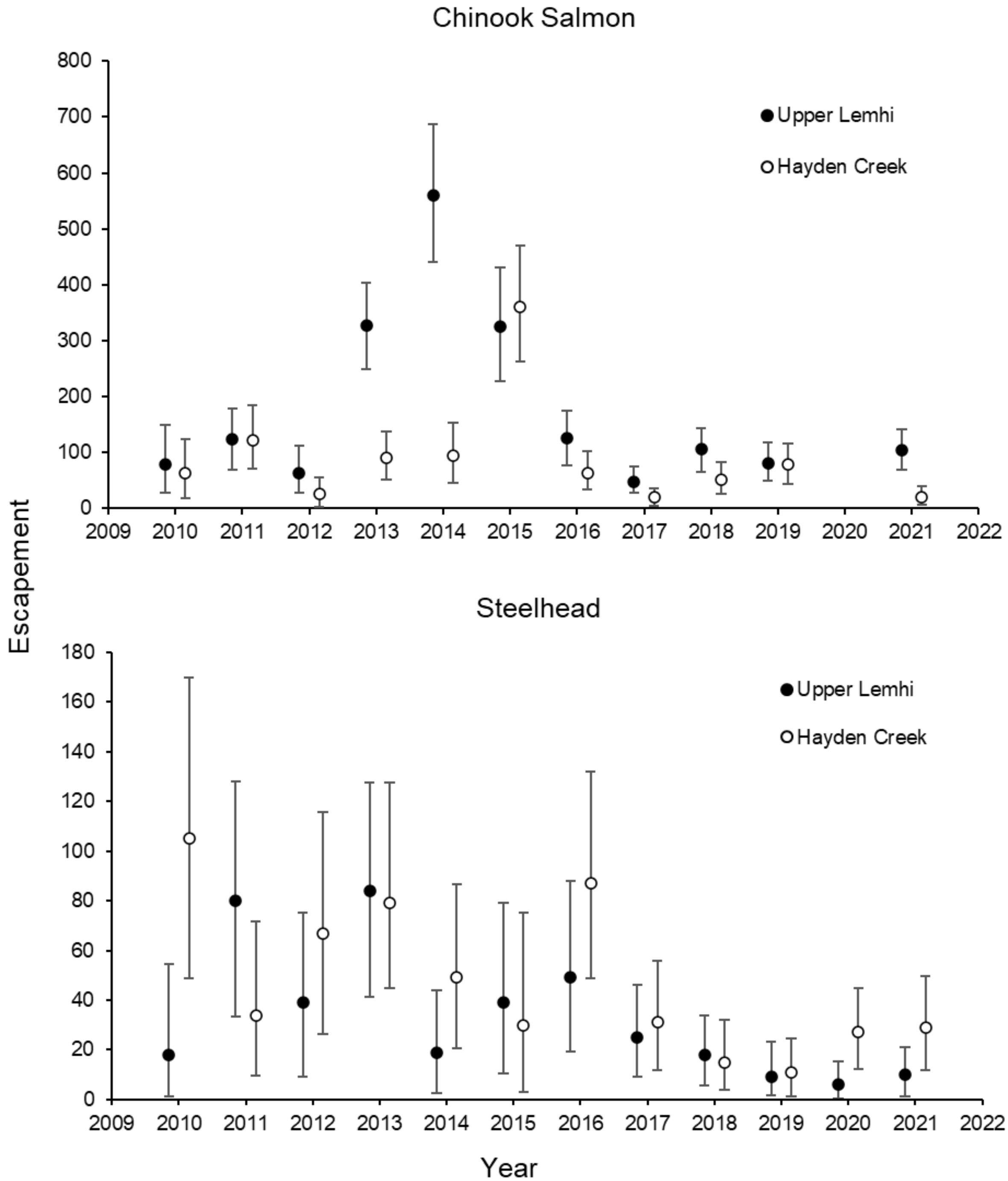

Figure 1.19. Estimates of escapement by spawn year of adult Chinook Salmon (top) and steelhead (bottom) to the upper Lemhi River (closed) and Hayden Creek (open) instream PIT tag detection systems. Estimates shown with 95% confidence intervals (Kinzer et al. 2020). There were no Chinook Salmon escapement estimates for 2020 because the Lower Granite Dam adult trap did not operate.

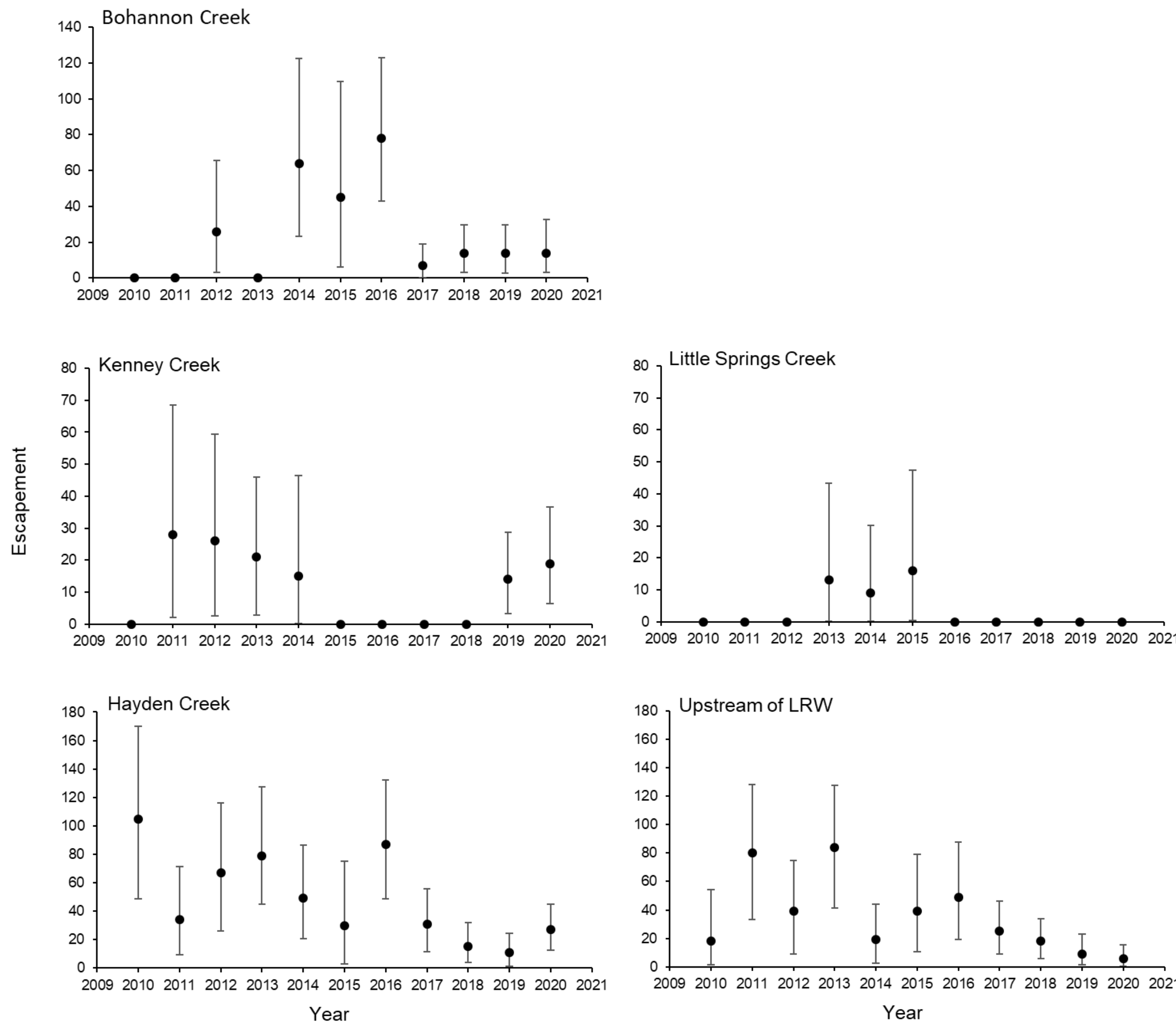

Figure 1.20. Estimates of escapement by spawn year of adult steelhead to instream PIT tag detection systems in priority tributaries and the reference tributary, Hayden Creek (Kinzer et al. 2020). Estimates shown with 95% confidence intervals. Hawley Creek, Canyon Creek, and Big Timber Creek are omitted because little to no adult steelhead were detected. Upstream of LRW = mainstem Lemhi River upstream of Hayden Creek.

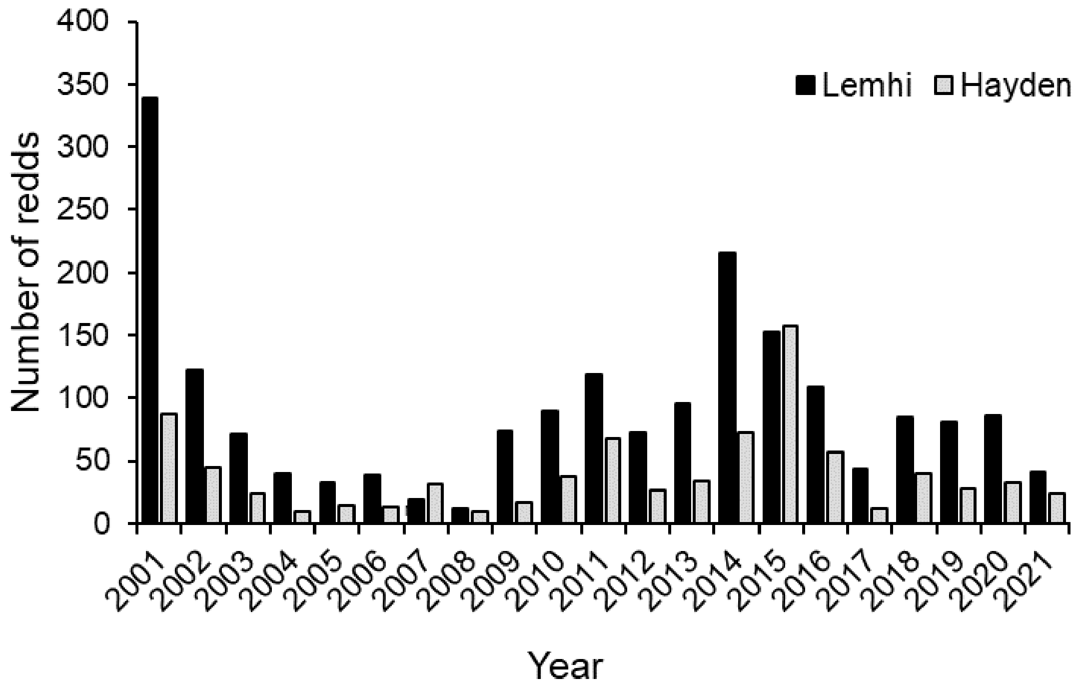

Figure 1.21. Redd counts from annual Chinook Salmon spawning ground surveys in the upper Lemhi River (black) and Hayden Creek (grey).

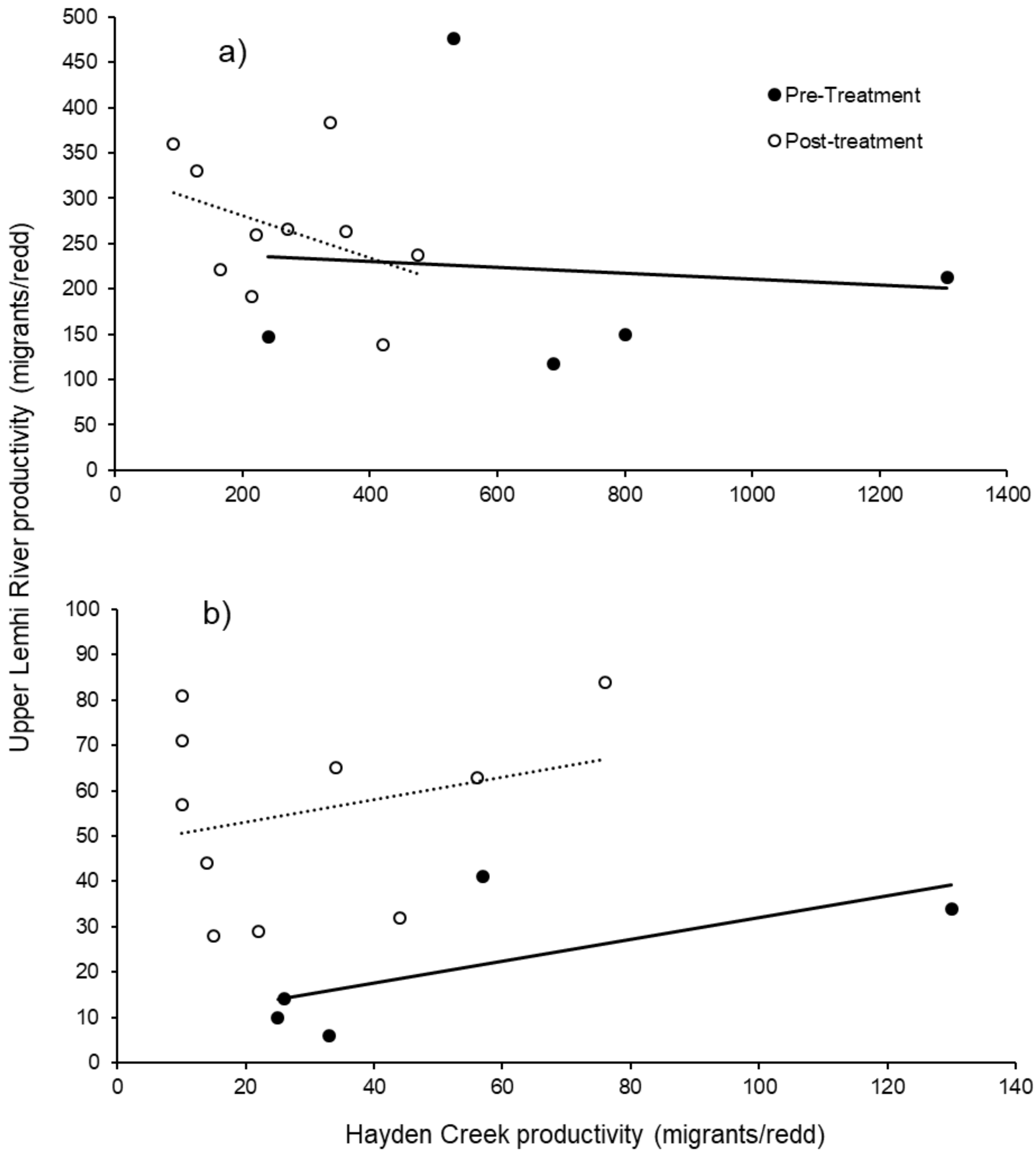

Figure 1.22. Relationship between upper Lemhi River productivity and Hayden Creek productivity before and after reconnection efforts commenced. Brood years 2005-2009 are considered pretreatment and brood years 2010-2019 are considered post-treatment. Productivity is measured as total fall parr and age-1 smolts per redd (panel a) and age-1 smolts per redd (panel b).

PART 2: THE POTLATCH RIVER INTENSIVELY MONITORED WATERSHED

INTRODUCTION

The Potlatch River watershed supports the largest spawning area of wild steelhead (*Oncorhynchus mykiss*) in the Lower Mainstem Clearwater River steelhead population (ICBTRT 2003; Bowersox et al. 2009). The Lower Mainstem Clearwater River steelhead population, which includes the Potlatch River watershed, is genetically distinct from other wild Clearwater River steelhead groups (Nielsen et al. 2009; Ackerman et al. 2016; Bowersox et al. 2023). Furthermore, the Lower Mainstem Clearwater River steelhead population comprises the only "large" independent population in the Clearwater River major population group (ICBTRT 2003) and must achieve viability in order for Snake River steelhead to be viable (NOAA 2017).

The Potlatch River watershed 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 2.1) 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. The upper Potlatch River watershed encompasses the drainage area upstream of Boulder Creek (Figure 2.1) 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 cool water temperatures (Bowersox and Brindza 2006). Landownership in the upper Potlatch River watershed is a mix of public U.S. Forest Service lands and large tracts of private timber lands used for timber production.

Land-use practices, primarily large commercial agriculture and timber harvest have altered the aquatic habitat and hydrograph such that limiting factors differ between the lower and upper watersheds. Primary limiting factors in the lower watershed are low summer base flows and fish passage barriers which restrict juvenile steelhead rearing habitat (Johnson 1985; Bowersox and Brindza 2006). The Potlatch River watershed receives the bulk (95%) of its annual precipitation from December to June (USDA SCS 1994). Thus, there is a seasonal pattern of high flow periods in the late winter/early spring followed by decreasing flows through the summer. However, the conversion of thousands of acres of timbered and meadow terrain into cropland in uplands and headwaters of tributaries has altered the hydrograph and resulted in acute high intensity 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; Knoth et al. 2021). Fish passage barriers, such as road culverts, also limit the quantity of juvenile steelhead rearing habitat in the lower watershed.

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 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.

Strategies to address limiting factors are 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 restoration of wetlands in headwater tributaries. 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.

Habitat restoration treatments are guided by the 2007 Potlatch River Watershed Management Plan and the 2019 Amendment (Latah SWCD 2007; Latah SWCD 2019). Treatments are developed in coordination with the Potlatch Implementation Group and funding agencies. Key watersheds, limiting factors, and restoration strategies were identified and incorporated into the management plan based on monitoring data, professional knowledge, and guidance from the ESA Recovery Plan for Snake River Spring/Summer Chinook Salmon and Snake River Basin Steelhead (NOAA 2017). Improving freshwater habitat conditions in the Potlatch River watershed is a priority within Idaho Department of Fish and Game (IDFG) Fisheries Management Plan (2019–2024) and the FY 2020–2023 Annual Strategic Plan (IDFG 2019, 2020).

A collaborative partnership among private landowners, local communities, and government agencies plans and implements restoration projects in the Potlatch River watershed. The primary agencies involved include the Latah Soil and Water Conservation District (Latah SWCD), IDFG, Idaho Office of Species Conservation, Natural Resources Conservation Service (NRCS), National Oceanic and Atmospheric Administration (NOAA), and the U.S. Forest Service (USFS). Implementation funding is provided by many sources including: USFS, NRCS, Bonneville Power Administration, and Pacific Coastal Salmon Recovery Funds. The IDFG conducts the monitoring and evaluation work that assesses the effectiveness of restoration treatments and guides future work.

Intensively Monitored Watershed (IMW) experiments were established across the Pacific Northwest to develop monitoring frameworks to assess the effectiveness of watershed-scale restoration treatments for increasing fish populations (Bilby et. al 2004). The Potlatch River IMW began in 2008 to rigorously test the effectiveness of stream restoration treatments aimed at increasing freshwater production of steelhead. The study contributes valuable knowledge on fish-habitat relationships and wild steelhead life history, which enable managers to improve natural spawning populations of salmonids in the Clearwater River basin. The IMW 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). Given the limiting factors and restoration projects being implemented, we developed the following hypotheses associated with the primary restoration strategies:

1. Barrier removals should increase the amount of spawning and rearing habitat. Improved passage will result in the expansion of adult spawning and juvenile rearing distribution (Anderson et al. 2008). Upstream distribution of steelhead spawners may also increase the number of emigrants through an increase in rearing habitat available to juveniles and a reduction of density dependent effects.

2. Flow supplementation should increase the quantity of juvenile rearing habitat (increased available wetted habitat and pool abundance) and improve the quality of existing rearing habitat (improved temperature and dissolved oxygen) for juvenile steelhead. In the short-term, flow supplementation is expected to increase growth and condition of juvenile steelhead. In the long-term, parr-to-smolt survival is expected to

change in response to flow supplementation; ultimately resulting in increased steelhead productivity within the system.

3. LWD treatments should increase the quantity of instream rearing habitat (i.e. pool formation) and increase hyporheic exchange between the river and surrounding aquifer (Sawyer et al. 2011). Expected fish responses include increased parr abundance and parr-to-smolt survival in treatment reaches compared to control reaches (Solazzi et al. 2000). Other potential responses include changes in emigrant age structure and/or length-at-age (Hunt 1988).

In 2017, we produced a 10-year summary report (Uthe et al. 2017) detailing monitoring, restoration, and scientific findings from project activities completed from 2008 to 2017. The current report provides updates to the restoration, monitoring, and research activities of the Potlatch River IMW project from 2017 to 2021. We focused the fish and habitat results in the context of restoration projects completed in the index watersheds during this timeframe. We begin with an inventory of the restoration work, then proceed to the monitoring methods and results. We are currently in the treatment phase of habitat restoration in the index watersheds and preliminary results presented here include both pretreatment data and partial data from the treatment phase.

Restoration Inventory

The current habitat restoration program began in 2009 in the upper watershed and 2013 in the lower watershed (Figures 2.2 and 2.3). Initial restoration treatments were conducted opportunistically with willing landowners throughout the watershed; however, since 2008 efforts have been made to direct restoration activities within two index watersheds: Big Bear Creek (BBC) in the lower watershed and the East Fork Potlatch River (EFPR) in the upper watershed. In addition, there have been significant restoration treatments implemented in Corral Creek (CORC) in the lower watershed. Restoration treatments in CORC have focused primarily on wetland and meadow restoration to improve low summer baseflows and information on habitat monitoring activities in CORC can be found under Habitat Surveys header below. Since the bulk of monitoring activities of the Potlatch River IMW study are focused in the BBC and EFPR index watersheds, they are the primary focus of this report.

Previous Restoration

The first of the current restoration treatments (2009–2016) in the BBC and EFPR watersheds were documented in Uthe et al. (2017). Briefly, restoration treatments in BBC involved barrier removals, riparian plantings, and a flow supplementation project. Ten passage barriers were removed which opened up access to an additional 23 km of spawning and rearing habitat and >55,000 trees/shrubs were planted. The flow supplementation project was a feasibility study that was completed in 2016 (Hand et al. 2020). In the EFPR, restoration treatments involved LWD installations, riparian fencing and plantings, and projects associated with road best management practices (i.e., road rocking or decommissioning for sediment reduction). Approximately 4.0 km of stream were treated with LWD installations (218 instream wood structures were installed), >11,200 shrubs/trees were planted, >16,600 linear feet of fencing were installed, and >35 km of roads were treated (Uthe et al. 2017).

Recently Implemented Projects

Restoration treatments in the BBC watershed during 2017–2021 involved primarily passage barrier and meadow restoration projects (Table 2.1). Four passage barriers were removed or modified on Big Meadow Creek, a tributary to the West Fork Little Bear Creek (WFLBC), which opened up access to an additional 10 km of spawning and rearing habitat. In addition, four meadow restoration projects were completed on approximately 5.5 km of stream in BBC and Nora Creek, a tributary to Little Bear Creek (LBC). The meadow restoration projects occurred in the upper reaches of BBC and LBC where steelhead are either currently blocked from accessing or do not sustain perennial flows. Therefore, the full impact of these projects will not be achieved until either passage or flow is restored to these systems. Funding and permitting issues impacted the implementation of large-scale projects in the watershed. In particular, the Big Bear Falls passage barrier project was slated for implementation in 2021 but was delayed due to permitting issues.

Restoration treatments in the EFPR watershed during 2017–2021 involved primarily LWD installations and floodplain reconnection projects. Recent LWD projects differed from previous work in terms of treatment size and intensity (i.e., the number of individual trees and structures). More specifically, recent treatments include using more channel spanning logjams using both anchored and unanchored logs, root wads, and engineered log jams designed to capture natural wood pieces moving downstream. Approximately 8.3 km of stream were treated with LWD installations including 89 complex wood structures, >350 trees, and 78 beaver dam analogs (BDAs; Pollock et al. 2007) to increase instream habitat complexity and promote floodplain reconnection. The majority of these projects were focused on the mainstem EFPR.

In summary, there have been significant restoration treatments implemented throughout the Potlatch River watershed. In the lower Potlatch River watershed, restoration treatments were concentrated primarily in the WFLBC, where 14 passage barrier projects were implemented, which opened up access to an additional 33 km of potential spawning and rearing habitat. In the upper Potlatch River watershed, restoration treatments have concentrated primarily on the mainstem EFPR, where approximately 12.3 km of stream were treated with LWD installations and BDAs to increase habitat complexity. Although additional restoration treatments were completed in each watershed, these are the treatments that should generate a fish response detectable by appropriate monitoring.

METHODS

Study Design

The Potlatch River IMW study is designed to assess the effectiveness of restoration treatments and potential response in steelhead production and productivity at multiple scales: 1) a broad-scale monitoring effort to document steelhead response within index watersheds (BBC and EFPR); 2) a finer-scale effort to assess habitat and fish response to restoration projects at the tributary level; and 3) reach-scale monitoring to assess whether individual projects produced the intended outcome (e.g., LWD installation altered stream hydrology and was used by fish). This study design allows managers to better understand the relationship between a habitat treatment and fish response and how localized responses to restoration propagate up to a higher, management-scale level. Detailed information on the Potlatch River IMW study goals, objectives, and hypotheses can be found in Uthe et al. (2017). This report covers watershed and tributary level monitoring activities in the last five years as no reach scale evaluations were completed during this timeframe.

Index-watershed monitoring was conducted to measure the total restoration benefits (sum of all projects) in the two index watersheds, which have different limiting factors. Multiple life stages were monitored to allow assessment of size- or age- specific responses. Index-watershed monitoring was set up as a before/after comparison of juvenile steelhead abundance and emigrant productivity in BBC and the EFPR. The primary response metrics included adult escapement, juvenile emigration, and smolt-per-female productivity.

Tributary-scale monitoring was conducted to isolate and measure fish and habitat responses by restoration type. The primary habitat response metrics examined included the amount of wetted habitat and pool density (lower watershed) and large woody debris (LWD) quantity (number of pieces per km), pool density (number per km), and percent canopy cover (upper watershed). The primary fish response metrics examined included juvenile steelhead density, survival, and growth, which are necessary for identifying the causal mechanisms of a response (Bennett et al. 2016). Tributary-scale monitoring used a Before/After/Control/Impact (BACI) study design (Roni et al. 2005) to examine habitat conditions and juvenile steelhead production in treatment and control areas in each watershed (Figure 2.4). In the lower watershed, treatment tributaries included BBC, LBC, WFLBC, and the control tributary was Pine Creek (PNC). For habitat surveys in the lower Potlatch River watershed, Corral Creek (CORC) was monitored as a treatment tributary to evaluate changes in base flow conditions in response to wetland restoration projects in the drainage and Cedar Creek (CEDC) was monitored as an additional control tributary. In the upper watershed, the treatment area was the mainstem EFPR downstream of Pivash Creek (lower 22 km) and control areas were the mainstem EFPR upstream of Pivash Creek and the West Fork Potlatch River (WFPR).

Adult Steelhead Escapement

Adult steelhead escapement into index watersheds was monitored using weirs and IPTDSs (Figure 2.1). The BBC IPTDS has undergone several iterations since 2013, in terms of both antenna design and location. Currently, the BBC IPTDS is located approximately 1.3 km from the mouth of BBC and consists of two 40 ft. litz cord antennas in series powered by two stand-alone IS1001 readers (https://www.ptagis.org). Each year, the IPTDS was operated from early January until the kelt (post-spawn fish) outmigration was complete in late-May or early June. The IPTDS was not operated in the summer and fall due to low flow conditions. A resistance-board weir has been operated in the EFPR since 2008 to monitor adult steelhead escaping into the watershed to spawn. Weirs were installed as early as conditions allowed, typically mid-February (BBC) or March (EFPR), and operated until the kelt outmigration was complete. Abundances were estimated using a mark-recapture method. 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, 2022a) and Idaho Adult Steelhead Monitoring Annual Reports (Dobos et al. 2019, 2020a; Knoth et al. 2018; Smith et al. 2021).

Juvenile Steelhead Emigration, Diversity, and Survival

Juvenile emigrant metrics (estimates of abundance, emigrant age composition, and size-at-age) from the index watersheds were monitored using RSTs (Figure 2.1). Annual operations began as early as conditions allowed, typically late January through February (BBC) or March (EFPR) and continued until early June in most years when low flows prevented RSTs from operating. Operations resumed during the fall at both sites when sufficient flows and personnel allowed. However, flows during the fall were generally not high enough to have continuous operations of RSTs at either site; therefore, abundances in the fall often could not be estimated (Uthe et al. 2017). Even when flows allow, fall emigration estimates were much smaller than spring estimates (median fall estimate = 8.5% of spring estimate). Detailed information on methods, data analysis, and annual operations for BBC and EFPR RSTs can be found in Potlatch River Steelhead Monitoring and Evaluation Reports (Knoth et al. 2021, 2022a), Idaho Anadromous Emigrant Monitoring annual reports (Belnap et al. 2018; Poole et al. 2019; Feeken et al. 2020, McClure et al. 2021), and Idaho Protocols for Trapping Anadromous Emigrants report (Copeland et al. 2021).

We examined survival rates of tagged juvenile steelhead emigrants from each RST to Lower Granite Dam (LGR). Estimating steelhead smolt survival is challenging because steelhead emigrate at different ages, size distributions of age groups overlap, and some fish have a tendency to rear an additional winter (or two) in freshwater before migrating to the ocean (Feeken et al. 2020). Previously, we estimated apparent survival as a proxy for actual cohort survival, but apparent survival estimates do not account for delayed emigration and thus are biased low since some individuals will not emigrate until subsequent years (Knoth et al. 2022a). The Lowther-Skalski model is a multistate release-recapture model that allows flexibility for delayed migration and having multiple tributary releases (i.e., years tagged at a RST) for a given cohort (i.e., brood year; Buchanan et al. 2015). In this report, we used detection and age data of juvenile steelhead from the index watersheds and applied the Lowther-Skalski model through the Basin TribPIT program (Lady et al. 2017) to estimate cohort survival of wild juvenile steelhead from each RST to LGR.

The Basin TribPIT program was downloaded from the Columbia Basin Research website. Mainstem observation history and age data were the two inputs for the model. For the mainstem observation history, a list of all known PIT tags implanted in juveniles at each RST (Big Bear Creek and East Fork Potlatch River) across brood years was generated from the PTAGIS website. The list was uploaded at http://www.cbr.washington.edu/dart/query/pit_tagids using the Basin TribPIT 18 “Observation File” option to generate the observation history for all juveniles tagged at the RST. Ages of tagged juvenile steelhead were determined either from scales or assigned from an age-length key if scales were not sampled (Dobos et al. 2023).

Productivity Estimates

Freshwater productivity estimates (juveniles at the RST per female spawner) were computed for each index watershed. Annual abundances of female spawners were calculated by applying the observed sex ratio at the weir or IPTDS to the total adult escapement estimate. Juvenile age proportions based on juvenile scale samples were applied to annual emigration estimates at RSTs 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.

Habitat Surveys

We surveyed low water habitat availability in the lower Potlatch River watershed to evaluate the amount of wetted habitat and pool density within treatment and control tributaries using methods described in Bowersox et al. (2009) and Knoth et al. (2021, 2022a). Corral Creek (CORC) was also monitored as a treatment tributary to evaluate changes in base flow conditions in response to wetland restoration projects in the drainage and Cedar Creek (CEDC) was monitored as an additional control tributary. The 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. We calculated the average proportion of wetted habitat (linear %) and pool density (number per 100 m) for each tributary annually.

We conducted surveys 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) count (number of pieces per km), pool density (number per km), and percent canopy cover. Detailed information on habitat survey methodology and site selection can be found in Uthe et al. (2017) and Knoth et al. (2021, 2022a). 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 densitometer 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.

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

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. Detailed information on electrofishing methodology and site selection can be found in Uthe et al. (2017) and Knoth et al. (2021, 2022a). Mean annual density estimates (fish per 100 m^2) were calculated for each tributary by averaging density from each site within a tributary.

Apparent survival of tagged steelhead from summer, to LGR the following spring, was used as an index of parr-to-smolt survival. We did not use the Basin TribPIT model to estimate parr-to-smolt survival due to the low number of tagged fish detected in the hydrosystem. Detailed methods for estimating apparent survival can be found in Uthe et al. (2017) and Knoth et al. (2021, 2022a). Briefly, we conducted roving electrofishing surveys in addition to single-pass surveys during the summer months to increase the number of 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 tagged. We assumed all steelhead tagged during the summer would emigrate the following spring. In the upper watershed, parr-to-smolt survival estimates were generated for the EFPR in aggregate. Too few juvenile steelhead were tagged in the EFPR treatment and control areas separately, and the WFPR, to generate separate survival estimates for this report.

We also monitored juvenile steelhead growth (summer to fall) as a response to restoration treatments in select tributaries in the lower watershed. Growth was monitored in LBC, WFLBC, and PNC because these were the only tributaries where we were able to successfully recapture an adequate number of fish to estimate growth. Electrofishing surveys were conducted annually during late October through early November, to recapture previously tagged juvenile steelhead. Detailed information on fall electrofishing surveys can be found in Uthe et al. (2017) and Knoth et al. (2021, 2022a). Growth (mm per d) was calculated as the change in fork length between time of tagging and time of recapture for each recaptured tagged fish. Annual means and standard deviations were calculated for each tributary to allow for trend comparison across years.

Data Analyses

We are currently in the treatment phase of habitat restoration in the index watersheds (Figures 2.2 and 2.3) and preliminary results presented here include both pretreatment data and partial data from the treatment phase.

In the index-watershed analysis, we used graphical comparisons for inference, with an emphasis on trends over the past five years (2017-2021) and pretreatment and treatment comparisons. These analyses continue those previously reported (Uthe et al. 2017). There were nine years of pretreatment data and eight years of treatment data used in the BBC watershed analyses and two years of pretreatment data and twelve years of treatment data used in the EFPR watershed analyses.

In the tributary-scale analysis, we used graphical comparisons for inference, with an emphasis on trends over the past five years (2017-2021) in treatment and control areas. We also analyzed each parameter (habitat metrics, juvenile steelhead density, growth, and survival) as a ratio (treatment:control) 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. For each habitat or fish parameter, we calculated the ratio of treatment to control for each year, estimated the mean ratios for the pretreatment and treatment periods, and used a *t*-test to compare the means. The formal two sample *t*-test analysis was only conducted for select treatment tributaries where sufficient restoration treatments were completed that could generate a habitat or fish response. The tributaries included the WFLBC (juvenile density, growth, and survival), CORC (habitat metrics), and BBC (habitat metrics) in the lower watershed and the EFPR treatment area (habitat metrics and juvenile density) in the upper watershed. Pretreatment and treatment periods varied by tributary and parameter (Table 2.2).

RESULTS

Adult Steelhead Escapement

Big Bear Creek

Adult steelhead escapement into BBC during 2017–2021 was below the range of previous estimates (Figure 2.5). The mean number of adult steelhead (2017–2021) returning to BBC was 26 fish (range = 20–35 fish). These are minimums because detection probability could not be calculated at the IPTDS during 2017–2021 because of the low number of detections at the IPTDS (average number of detections = 5 fish). Overall, adult steelhead escapement into BBC varied 15-fold across years and the mean number of adult steelhead returning to BBC during pretreatment years (2008–2013) was 159 fish (range = 50–317 fish) and treatment years (2014–2021) was 93 fish (range = 20–254 fish).

East Fork Potlatch River

Adult steelhead escapement into the EFPR during 2017–2021 was below the range of previous estimates (Figure 2.5). The mean number of adult steelhead (2017–2021) returning to the EFPR was 16 fish (range = 6–25 fish). Estimates in four of these years (2017, 2018, 2019 and 2021) were likely biased low due to the small sample size of kelts to establish the mark rate within the population (Dobos et al. 2020a). The mean number of kelts captured at the weir was 6 fish (range = 2–12 fish) and capture probability was 69% (range = 0–100%) during these years. Overall, adult steelhead escapement into the EFPR varied 23-fold across years and the mean number of adult steelhead returning during pretreatment years (2008–2009) was 116 fish (range = 92–140 fish) and treatment years (2010–2021) was 55 fish (range = 6–105 fish).

Juvenile Steelhead Emigration, Diversity, and Survival

Big Bear Creek

Spring emigration from BBC during 2017–2021 fell within the range of previous estimates (Figure 2.6). The mean number of spring emigrants (2017–2021) was 7,957 fish (range = 3,905–10,928 fish). There were no fall emigration estimates for BBC during this timeframe. Overall, spring emigration from BBC has remained relatively stable across years (six-fold variation). The mean number of spring emigrants during pretreatment years (2005–2013) was 9,976 fish (range = 3,837–22,649 fish) and treatment years (2014–2021) was 7,834 fish (range = 3,905–10,928 fish).

Emigrant age composition during the spring season at BBC during 2017–2021 was dominated by age-2 emigrants, which is consistent with previous estimates (Figure 2.7). Mean emigrant age composition (2017–2021) was 42.1% age-1 fish, 54.2% age-2 fish, and 3.8% age-3 fish. Emigrant age composition at BBC has remained consistent across years with the mean emigrant age composition during pretreatment years (2005–2013) at 33.1% (range = 8.8–51.2%) age-1 fish, 62.0% (range = 47.3–86.8%) age-2 fish, and 4.9% (range = 1.5–13.6%) age-3 fish and treatment years (2014–2021) at 36.9% (range = 11.1–70.7%) age-1 fish, 58.2% (range = 28.6–82.7%) age-2 fish, and 4.9% (range = 0.6–10.9%) age-3 fish.

Emigrant length-at-age at BBC was variable during 2017–2021 (Figure 2.8). Mean length-at-age (2017–2021) was 142.1 mm (SE = 3.8) for age-1 emigrants, 160.2 mm (SE = 2.7) for age-2 emigrants, and 177.5 mm (SE = 3.3). There was no discernable trend in emigrant length-at-age across years. During pretreatment years (2008–2013), mean FL of age-1 emigrants was 134.1 mm (SE = 6.5), age-2 emigrants was 170.5 mm (SE = 2.8), and age-3 emigrants was 188.2 mm (SE = 5.4) and during treatment years (2014–2021), mean FL of age-1 emigrants was 122.8 mm (SE = 3.0), age-2 emigrants was 160.2 mm (SE = 2.2), and age-3 emigrants was 172.3 mm (SE = 3.2).

The number of fish tagged at the BBC screw trap and detected in the hydrosystem during 2017–2021 was above the range of previous estimates (Table 2.3). The mean number of fish tagged (2017–2021) was 1,800 (range 390–4,134 tagged fish) and detected was 1,217 (range 127–3,175 unique detections). Tag distribution at the BBC screw trap varied 11-fold across years and the mean number of tags distributed during pretreatment years (2005–2013) was 1,376 (range = 562–1,915 tags) and treatment years (2014–2021) was 1,940 (range = 390–4,134 tags). Tag detection in the hydrosystem varied 25-fold across years and the mean number of tags detected during pretreatment years (2005–2013) was 739 (range = 198–1,569 unique detections) and treatment years (2014–2021) was 1,279 (range = 127–3,175). The majority (97%) of the tags detected in the hydrosystem were detected the following spring after tagging.

Estimated survival of emigrants from BBC screw trap to LGR during brood years 2017 and 2018 fell within range of previous estimates (Figure 2.9). Mean emigrant survival (2017–2018) was 38.0% for age-1 emigrants, 49.0% for age-2 emigrants, and 67.0% for age-3 emigrants. During pretreatment brood years (2007–2013) mean emigrant survival was 33.4% for age-1 emigrants (range = 17.6-59.2%), 59.2 % for age-2 emigrants (range = 40.3-85.2%), and 56.1%

for age-3 emigrants (range = 31.0-100.0%) and during treatment brood years (2013–2018) mean emigrant survival was 30.0% for age-1 emigrants (range = 12.3-41.2%), 52.3% for age-2 emigrants (range = 47.6-58.3%), and 58.4% for age-3 emigrants (range = 12.5-100.0%).

East Fork Potlatch River

Spring emigration from the EFPR during 2017–2021 was below the range of previous estimates (Figure 2.6). The mean number of spring emigrants during 2017–2021 was 6,606 fish (range = 2.184–15,210 fish). Of note, emigration estimates from 2019–2021 were the lowest on record. There were no fall emigration estimates for EFPR from 2017–2021. Spring emigration from EFPR varied 18-fold across years and the mean number of emigrants during pretreatment years (2008–2009) was 10,588 fish (range = 10,106–11,069 fish) and treatment years (2010–2021) was 13,662 fish (range = 2,184–40,224 fish).

Emigrant age composition during the spring season at the EFPR during 2017–2021 consisted of higher proportions of older fish relative to previous estimates (Figure 2.7). Mean emigrant age composition (2017–2021) was 48.2% age-1 fish, 44.9% age-2 fish, and 6.9% age-3 fish. Overall, there has been a shift towards older emigrants across years with the mean emigrant age composition during pretreatment years (2008–2009) at 74.7% (range = 63.3–86.1%) age-1 fish, 24.4% (range = 12.1–36.7%) age-2 fish, and 4.3% (range = 0.0–1.8%) age-3 fish and treatment years (2010–2021) at 59.9% (range = 25.0–83.5%) age-1 fish, 35.2% (range = 14.8–66.7%) age-2 fish, and 4.9% (range = 0.0–15.4%) age-3 fish.

Emigrant length-at-age at the EFPR during 2017–2021 was larger than previous estimates (Figure 2.10). Mean length-at-age (2017–2021) was 106.4 mm (SE = 5.0) for age-1 emigrants, 154.3 mm (SE = 3.7) for age-2 emigrants, and 179.7 mm (SE = 5.8) for age-3 emigrants. Overall, emigrant size for all age classes has increased across years in the EFPR. During pretreatment years (2008–2009), mean FL of age-1 emigrants was 91.2 mm (SE = 9.3), age-2 emigrants was 132.4 mm (SE = 4.0), and age-3 emigrants was 146.6 mm and during treatment years (2010–2021), mean FL of age-1 emigrants was 98.0 mm (SE = 3.0), age-2 emigrants was 145.1 mm (SE = 3.0), and age-3 emigrants was 172.3 mm (SE = 3.3).

The number of fish tagged at the EFPR screw trap and detected in the hydrosystem during 2017–2021 was below the range of previous estimates (Table 2.3). The mean number of fish tagged (2017–2021) was 405 (range 78–1,096 tagged fish) and detected was 73 (range 20–156 unique detections). Tag distribution at the EFPR screw trap varied 25-fold across years and the mean number of tags distributed during pretreatment years (2005–2013) was 838 (range = 432–1,243 tags) and treatment years (2014–2021) was 815 (range = 78–1,980 tags). Tag detection in the hydrosystem varied 54-fold across years and the mean number of tags detected during pretreatment years (2005–2013) was 131 (range = 107–154 unique detections) and treatment years (2014–2021) was 298 (range = 20–1,086). Of the tags detected, 87% were detected the following spring after tagging and 13% were detected two or more years after tagging.

Estimated survival of emigrants from the EFPR screw trap to LGR during brood years 2017 and 2018 fell within range of previous estimates (Figure 2.11). Mean emigrant survival (2017–2018) was 14.2% for age-1 emigrants, 32.2% for age-2 emigrants, and 14.1% for age-3 emigrants. During pretreatment brood years (2007–2009) mean emigrant survival was 17.6% for age-1 emigrants (range = 9.1-25.7%), 51.8% for age-2 emigrants (range = 38.4-60.5%), and 42.1% for age-3 emigrants (range = 30.6-54.0%) and during treatment brood years (2010–2018) mean emigrant survival was 8.0% for age-1 emigrants (range = 0.4-19.5%), 38.0% for age-2 emigrants (range = 26.4-65.1%), and 36.0% for age-3 emigrants (range = 14.1-71.4%).

Population Productivity

Big Bear Creek

Mean BY productivity (2017–2018) in BBC was 350 recruits per spawner (range = 213–487 recruits per spawner), which was above the range of previous estimates (Figure 2.12). Productivity estimates for these BYs were biased high because minimums were used in place of expanded adult escapement. Complete BY productivity estimates have been generated for 13 BYs in BBC and mean BY productivity for pretreatment years (2005–2013) was 131 recruits per spawner (range = 53–277 recruits per spawner) and treatment years (2014–2018) was 191 recruits per spawner (range = 48–487 recruits per spawner). Productivity estimates for BBC displayed a strong density-dependent relationship (Figure 2.13).

East Fork Potlatch River

Mean BY productivity (2017–2018) in the EFPR was 631 recruits per spawner (range = 294–967 recruits per spawner), which was above the range of previous estimates (Figure 2.12). Productivity estimates for these BYs were biased high because minimums were used in place of expanded adult escapement. Complete BY productivity estimates have been generated for 10 BYs in the EFPR and mean BY productivity during pretreatment years (2008–2009) was 480 recruits per spawner (range = 364–596 recruits per spawner) and treatment years (2014–2018) was 432 recruits per spawner (range = 127–967 recruits per spawner). Productivity estimates in the EFPR also displayed a weak density-dependent relationship (Figure 2.13).

Habitat Surveys

Lower Potlatch River Watershed

Low water habitat surveys during 2017-2021 highlighted the extent of low summer base flows in the lower watershed tributaries, especially in the upland reaches (Figures 2.14 and 2.15). The mean amount of wetted habitat (2017–2021) across all tributaries was 85.6% (range = 34–100%) in canyon reaches and 56.8% (range = 19–100%) in upland reaches. Mean pool density (2017–2021) across all tributaries was 3.9 pools per 100 m (range = 0.2–10.6 pools per 100 m) in canyon reaches and 1.7 pools per 100 m (range = 0.7–2.4 pools per 100 m) in upland reaches. Recent wetted habitat and pool density estimates (2017–2021) were below previous estimates in the treatment tributaries but were similar to or above previous estimates in the control tributaries.

Base flow conditions in select treatment tributaries (CORC and BBC) were relatively constant in relation to the control tributary (PNC) across years (Figures 2.16 and 2.17). In CORC, there were no significant differences in mean ratios between pretreatment (2008–2010) and treatment (2012–2021) periods in terms of wetted habitat (P = 0.732) or pool density (P = 0.710). In BBC, there was a significant decrease (P = 0.032) in wetted habitat between pretreatment (2008–2016) and treatment (2017–2021) periods, but no change in pool density (P = 0.170).

Upper Potlatch River Watershed

Habitat metrics in treatment and control areas displayed similar trends during 2017–2021 (Figure 2.18). Mean canopy cover (2017–2021) was 46% (range = 28–71%), LWD density was 194 pieces per km (range = 42–554 pieces per km), and pool density was 7 pools per km (range = 1–12 pools per km) across all areas. Recent (2017–2021) canopy cover and LWD estimates were higher than previous estimates while pool density estimates were lower than previous estimates. The EFPR control area contained the highest estimates of canopy cover, LWD, and pool density among areas. Conversely, the EFPR treatment area had the lowest percent canopy cover and the WFPR control area had the lowest LWD and pool density across years.

Results of the ratio analyses of habitat metrics in the upper watershed were mixed (Figures 2.19–2.21). Canopy cover in the EFPR treatment area has not changed relative to the EFPR control (P = 0.086) but increased significantly (P = 0.017) relative to the WFPR control area between the pretreatment and treatment periods (Figure 2.19). Large wood density in the EFPR treatment area decreased significantly (P = 0.020) relative to the EFPR control area but has not changed relative to the WFPR control area (P = 0.826) between the pretreatment and treatment periods (Figure 2.20). There was no significant change (P = 0.310) in pool density in the EFPR treatment area relative to the WFPR control area between the pretreatment and treatment periods (Figure 2.21). Pool density in the EFPR treatment area has remained stable relative to the EFPR control area during the treatment period.

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 during 2017– 2021, with peaks in 2017 and 2021 and a low in 2019 (Figure 2.22). Note that there has not been any restoration in Big Bear Creek that would influence densities and all years are regarded as pretreatment. Mean juvenile steelhead density estimates were highest in LBC and lowest in PNC. Recent density estimates generally fell within the range of previous estimates, except for PNC and LBC in 2019, which were the lowest estimates on record for each tributary.

Juvenile steelhead density in the WFLBC (treatment tributary) increased relative to PNC (control tributary) between pretreatment and treatment years, though not significantly (P = 0.689; Figure 2.23). The mean density ratio was 1.87 during treatment years (1996, 2013) and increased to 2.32 in treatment years (2014–2021). The three highest ratio values (2019, 2020, and 2021) were influenced in part by the record low density estimates in PNC in 2019 and 2020 (Knoth et al. 2022a), but also by the high juvenile density estimate in WFLBC in 2021.

Parr-to-Smolt Survival

The numbers of fish tagged and of tags detected in the hydrosystem during 2017–2021 varied across tributaries and years in the lower watershed (Table 2.4). The mean number of fish tagged was 296 (range 124–424 tagged fish). On average, the WFLBC had the highest number of fish tagged and BBC the lowest number of fish tagged over this timeframe. Tag distribution in PNC in 2019–2021 was below average due to low densities of juvenile steelhead captured in the drainage. The majority (>95%) of the tags detected in the hydrosystem were detected the following spring after tagging.

Apparent survival from tributary to LGR the following spring during 2017–2021 varied among tributaries (Figure 2.24). Mean apparent survival (ranged from 9.1–14.1% and was highest in the WFLBC and lowest in LBC. Recent (2017–2020) apparent survival estimates fell within the range of previous estimates. Overall, 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.

Apparent survival in the WFLBC (treatment tributary) has not changed significantly relative to PNC (control tributary) between pretreatment and treatment years (P = 0.728; Figure 2.25). The mean survival ratio was 0.99 during treatment years (2008–2013) and increased to 1.16 in treatment years (2014–2020).

Growth Rates

The number of fish recaptured during 2017–2021 fall electrofishing surveys varied across the three tributaries (WFLBC, LBC, and PNC). The mean number of steelhead recaptured during 2017–2021 was 28 fish (range = 0–54 fish). Recaptures were the highest in the WFLBC and lowest in PNC. No fish were recaptured in PNC during 2019 and 2020 due to the low number of tags distributed in the drainage. Mean time at large (2017–2021) was 119 days (range = 92–141 days) and was greatest in PNC and lowest in WFLBC (Table 2.5).

Daily summer growth rates varied among tributaries during 2017–2021 but were generally higher in the control tributary (PNC) relative to the treatment tributaries (LBC and the WFLBC) (Figure 2.26, Table 2.5). Mean growth rates were 0.068 mm per d (range = 0.058–0.083 mm per d) for PNC, 0.034 mm per d (range = 0.009–0.068 mm per d) for the WFLBC, and 0.028 mm per d (range = 0.008–0.049 mm per d) for LBC. Recent estimates (2017–2021) fell within range of previous estimates. Of note, both the WFLBC and LBC experienced the highest growth rates on record in 2020. Growth rates could not be calculated for PNC in 2019 and 2020 since no fish were recaptured.

Juvenile steelhead growth in the WFLBC (treatment tributary) fluctuated relative to the PNC (control tributary) with no discernable trend during treatment years (Figure 2.27). The mean growth ratio was 0.40 (range = 0.15–1.03) during the treatment years.

Upper Potlatch River Watershed

Juvenile Steelhead Density

Juvenile steelhead density estimates in the treatment and control areas displayed relatively similar trends during 2017–2021, with peaks in 2017 and 2021 and a low in 2019 (Figure 2.28). Mean juvenile steelhead density ranged from 1.06 fish per 100 m^2 (range = 0.09–2.01) in the WFPR to 8.3 fish per 100 m^2 (range = 4.8–11.2 fish per 100 m^2) in the EFPR control area. Recent density estimates generally fell within the range of previous estimates for each area.

Juvenile steelhead density in the EFPR treatment area remained comparatively stable relative to control areas over time (Figure 2.29). Steelhead density in the EFPR treatment area has not changed relative to WFPR between pretreatment and treatment years (P = 0.624). The density ratio was 1.9 during the pretreatment year (2004) and increased to a mean of 3.9 in treatment years (2013–2021). The two highest ratio values in 2015 and 2021 were the result of below average density estimates in the WFPR in those years. The density ratio values between the EFPR treatment and control areas averaged 0.52 with minimal variation (range = 0.25–0.82) during the treatment years.

Parr-to-Smolt Survival

The numbers of fish tagged and of tags detected in the hydrosystem during 2017–2021 varied across years in the EFPR (Table 2.4). The mean number of fish tagged was 307 fish (range = 272–381 tagged fish). The mean number of tags detected in the hydrosystem was 22 (range = 18–29 tags), % of which were detected the following spring after tagging. There were minimal steelhead tagged in the WFPR across the years (26 fish total) and three fish were subsequently 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) was relatively consistent in recent years (Figure 2.30). Mean apparent survival was 6.9% (range = 5.1–11.3%) and estimates in tag years 2019 and 2020 were the lowest on record at 4.8% and 5.0%, respectively. Estimates could not be generated for tag groups in 2010, 2014, and 2015 because of low detections in the hydrosystem.

DISCUSSION

Results from the Potlatch River IMW study provide the necessary data to evaluate the success of restoration efforts. Recent results underscore the need for continued implementation of habitat restoration and highlight the challenges that out-of-basin factors can play on assessing the effectiveness of restoration efforts and achieving the full benefits of such efforts. Funding and permitting limitations continue to impact the pace of project implementation, especially in regard to large-scale, high impact projects in the BBC watershed. As a result, project implementation in the BBC watershed has not generated a sustained positive response in juvenile steelhead production. Conversely, recent shifts in the EFPR emigrant age composition and length-at-age suggest improved rearing conditions in the drainage, though the underlying mechanisms remain unclear.

We begin by highlighting key 2017–2021 findings and trends observed in the monitoring data. Next, we cover some of the recent challenges and lessons learned in the planning, restoration, and monitoring components of the study, including adaptive management of the restoration and monitoring programs.

Recent Findings and Trends

We observed sharp declines in adult steelhead returns to the index watersheds during 2017–2021. On average, adult steelhead returns declined by 86% in BBC and 82% in the EFPR relative to previous years. The magnitude of declines was likely exaggerated to some extent since minimum escapement estimates were generated for the past five years in BBC and three of the past five years in the EFPR. Nonetheless, a commensurate decline was observed in steelhead populations throughout Idaho and the rest of the Snake River basin (Knoth et al. 2018; Dobos et al. 2019, 2020a; Smith et al. 2021; Baum et al. 2022). In fact, all Snake River steelhead populations showed significant declines in recent years (Ford 2022). Declines across steelhead populations over a wide geographical range suggest large-scale biological and environmental factors may be the cause. Work is ongoing to relate ocean condition indices to steelhead population productivity, but evidence suggests sea surface temperatures during the first summer of ocean entry may be a key factor influencing steelhead marine survival (Kendall et al. 2017). The recent decline in adult steelhead escapement in the Potlatch River is troubling and has implications in both monitoring assessment and response to restoration projects (discussed below).

We have observed different patterns in emigrant production in the index watersheds during recent years with low adult escapement. We have observed a sharp decline in juvenile emigration in the EFPR watershed, but a more subtle decline in BBC. On average, juvenile emigration declined by 61% in the EFPR, but only 15% in BBC relative to previous years. The 2019-2021 EFPR emigration estimates were the lowest on record. Similar declines in juvenile emigration were observed in steelhead populations across Idaho during recent years (e.g., Heller et al. 2022) and were likely the result of low adult escapement into the various drainages. It may also speak to differing levels of density dependence in the two watersheds. Population productivity in BBC is highly density dependent as a result of limited juvenile rearing capacity (Uthe et al. 2017). 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. Although the low emigrant production in the EFPR during recent years is discouraging, we have documented recent positive shifts in emigrant age and growth that suggest rearing conditions are improving in the drainage.

Restoration efforts that improve the quality of rearing habitat should increase the carrying capacity of the system, thereby reducing competition for resources (i.e., food or habitat) and improving freshwater productivity. Improvements to freshwater productivity may manifest in higher juvenile abundance or in fitter emigrants (i.e., larger or older), resulting in higher overall population productivity. In the EFPR, restoration efforts involve primarily the addition of wood structures to improve instream habitat complexity. The addition of wood and wood structures not only provides habitat for juvenile salmonids (Roni et al. 2002, 2008, Pess et al. 2012) but can also improve food resources by trapping organic material and increasing aquatic insect production (Coe et al. 2009). To date, approximately 18% of the mainstem EFPR has been treated with large wood and floodplain restoration projects. We hypothesized that as rearing conditions improve in the EFPR, emigrant age structure may shift older as more juveniles rear longer in the drainage versus emigrating early and rearing downstream (Uthe et al. 2017). Emigrant length-at-age may also increase as a result of improved feeding resources or opportunities. We have documented a shift in the EFPR emigrant age structure in recent years, with higher proportions of older fish (age-2 and age-3 fish) emigrating from the drainage. We have also documented positive trends in emigrant length-at-age, most notably in age-2 and age-3 fish. These findings are consistent with our hypothesis and suggest that rearing conditions are improving in the EFPR. In addition to restoration efforts, other processes such as natural beaver colonization may also be benefitting rearing conditions in the drainage. It is worth noting there is likely less competition in the system because of low densities of steelhead in recent years that may be contributing to the observed changes in age structure and growth. Nonetheless, these findings are encouraging and continued monitoring of the EFPR emigrant population will help further elucidate the relationship between habitat actions and steelhead response in the upper Potlatch River watershed.

We hypothesized barrier removal projects would initially result in the expansion of steelhead spawning and rearing distribution and ultimately benefit emigrant production through an increase in available rearing habitat and a reduction of density dependent effects. To date, most barrier removals have been in WFLBC, where 14 barriers were addressed to open access to an additional 33 km of potential spawning and rearing habitat. We previously documented a rapid recolonization of upstream habitats by adult steelhead following the implementation of barrier removal projects in WFLBC (Uthe et al. 2017; Knoth et al. 2021, 2022b). However, we have not observed any sustained improvements in juvenile steelhead production in WFLBC following the implementation of these projects. Barrier removal projects are one of the most successful habitat improvement actions for salmon and steelhead in the Pacific Northwest (Bilby et al. 2022), but the effectiveness of these projects on improving juvenile fish production is dependent not only on the amount and quality of the habitat upstream of the barrier but also on the number of returning fish to colonize the new habitat (Anderson et al. 2014; Roni et al. 2002, 2008, 2014). The barrier removal projects in the WFLBC were located primarily in the upper reaches of the drainage where habitat conditions upstream of the barriers suffer from intermittent base flows, degraded riparian conditions, and poor instream complexity. Furthermore, adult steelhead returns have declined substantially over the past five years, and it is possible there are not enough fish to fully colonize the restored habitat. Improving connectivity was a critical first step in the restoration process in the WFLBC, but the full benefit of these projects will not be achieved until upstream habitat conditions are addressed and adult returns improve to the drainage.

Challenges, Lessons Learned, and Adaptive Management

The following section covers the key challenges and lessons learned in the Potlatch River IMW project over the past five years, including adaptive management of the restoration and monitoring programs. The section begins with recent challenges related to the planning and implementation of restoration activities in the Potlatch River, including our efforts to address these issues moving forward. Next, the section highlights issues that we have encountered in the monitoring program over the past five years, especially regarding the low adult steelhead returns. The section ends with a discussion of the adaptive management approaches we incorporated to improve the restoration and monitoring programs.

Restoration Program

We continued to face obstacles to keystone restoration projects in the Potlatch River, in particular the Big Bear Falls passage project and the Spring Valley Creek flow supplementation project. Combined, these projects would treat >30 km of critical habitat and could significantly increase (>70% increase) juvenile steelhead production in the lower watershed (Uthe et al. 2017). However, these are large-scale, expensive projects that fall outside the traditional approach to habitat restoration, such as restoring and enhancing steelhead passage at Big Bear Falls, a natural barrier altered by degraded flow conditions (Bowersox et al. 2016). As a result, they require a great deal of coordination among funders, implementers, and permitting agencies to implement. During 2016-2020, IDFG personnel worked closely with engineers and funders to develop a project to physically modify Big Bear Falls to enhance upstream adult steelhead passage. The project was scheduled for implementation in 2021 but was cancelled due to cultural resource concerns. In light of this setback, we are exploring alternative options to transport adult steelhead upstream of the falls (more details below). The implementation of the Spring Valley Creek flow supplementation project has also been delayed. We previously documented the feasibility and benefits of flow supplementation in Spring Valley Creek during a pilot study in 2015 and 2016 (Hand et al. 2020; Knoth et al. 2021). However, the implementation of this project has been stalled due to concerns over protecting the downstream water releases and funding improvements to the dam and reservoir infrastructure. IDFG is in the process of securing a water right to protect the downstream releases from unauthorized withdrawal and project engineers recently delivered preliminary designs for the dam modifications with updated cost estimates. These efforts are promising and should allow us to secure funding for this project. Although these keystone projects have proven to be challenging to implement, they are necessary to generate a population-level benefit to steelhead production in the watershed.

As mentioned previously, we plan to take an alternative approach to address passage issues at Big Bear Falls. We will conduct a pilot study to assess the feasibility of using trap-and-haul techniques to transport adult steelhead upstream of the falls. In the Pacific Northwest, trap and haul is an important management tool for providing salmonids access to historical habitats, especially in cases where the barrier cannot be modified or removed (Kock et al. 2020). Our plan is to transport prespawn adult steelhead upstream of Big Bear Falls for three consecutive years (2024-2026). We will operate a picket weir at the mouth of BBC each spring to capture prespawn adult steelhead. Each fish captured will be transported via truck to the release location upstream of the falls. We will utilize radiotelemetry to assess the movement patterns and spawning distribution of each adult steelhead we transport upstream of the falls. In addition, we plan to collect a tissue sample from each adult steelhead, as well as tissue samples from all steelhead smolts captured at the BBC RST in the years following the transportation of adults. We can use parentage-based tagging (PBT) techniques to assign the steelhead offspring collected at the BBC screw trap to the parents we transported upstream of the falls. This will allow us to quantify the smolt production from the translocated adults relative to production from adults that spawned elsewhere in the watershed. Information gained from this pilot study will allow us to assess the production potential of habitat upstream of Big Bear Falls and determine if trap and haul is a viable, long-term management solution to passage issues in BBC.

The design and implementation of LWD projects in the EFPR has evolved over the course of the project to promote greater floodplain connectivity and habitat complexity. Initial LWD projects (2008-2016) were generally in-channel projects that used design techniques to create static features such as log weirs and logjams constructed of cut logs. These projects were generally small in overall size and intensity (i.e., the number of individual trees and structures), and obstructed a relatively small percentage of the stream channel cross-sectional area. In addition, some treatments were used to provide bank stabilization. Beginning in 2017, the design of LWD projects evolved to include more channel-spanning logjams using both anchored and unanchored logs, root wads, and engineered log jams designed to capture natural wood pieces moving downstream. Recent LWD projects contain more wood pieces that interact with the stream under various flows and are strategically placed to help reconnect floodplain habitats. In 2019, IDFG implemented a reach-scale monitoring plan in the EFPR to assess the effectiveness of these projects in promoting floodplain connectivity and habitat complexity. The monitoring design incorporated three treatment sites and one control site and protocols were adapted from Bonneville Power Administration’s Action Effectiveness Monitoring protocol for monitoring the effectiveness of off-channel habitat/floodplain enhancement projects (O’Neal et al. 2016). Data will be collected bi-annually through 2025 to allow for multiple channel-forming flow events to alter habitat conditions at the project sites. Results from this reach-scale evaluation will inform us how the latest design of LWD projects modify physical habitat features in the EFPR.

Monitoring Program

One of the major challenges we encountered is our inability to accurately monitor adult escapement into the index watersheds in recent years. This issue is more pronounced in BBC where we utilize a IPTDS to monitor adult escapement into the watershed. Sparse adult steelhead detections and frequent antenna outages at the BBC IPTDS site hampered our ability to estimate detection efficiency at the site and precluded us from producing expanded escapement estimates (Knoth et al. 2021, 2022a; Smith et al. 2021). We upgraded the BBC IPTDS in 2019 and 2020 by installing lower-profile, corded IPTDS to reduce antenna outages during high flow events. Nonetheless, we are still not getting at least one unique detection on each antenna span, preventing estimation of detection efficiency. One possible solution to increase detection efficiency at the BBC site would be to install a third antenna span upstream of the existing site.

Connolly et al. (2008) examined the detection efficiency of multiple antenna configurations and found overall detection efficiency was higher with configurations with the most antenna spans. Furthermore, the authors noted the built-in redundancy of multiple span configurations allowed the sites to perform even with a loss of one or more antennas. Another possible approach would be to utilize a hierarchical Bayesian model to estimate missing detections at the site based on detection histories from previous years (Jasper et al. 2018). Improving our ability to estimate spawner abundance will enhance our ability to assess the effects of habitat restoration on adult production and population productivity in the watershed.

Evaluation of fish response to habitat treatments in the Potlatch River has been compromised to some extent by out-of-basin influences on fish populations. These factors may have limited the capacity for fish populations to respond to habitat restoration treatments (Anderson et al. 2019; Bilby et al. 2022). The low adult steelhead returns in recent years, potential causes, and associated monitoring and evaluation problems have been discussed previously in this report. Nonetheless, restoration treatments being implemented in the Potlatch River and elsewhere require a certain level of fish abundance to be beneficial. For example, the effectiveness of barrier removal projects on improving juvenile fish production is dependent not only on the amount and quality of the habitat upstream of the barrier but also on the number of returning fish to colonize the new habitat (Roni et al. 2002, 2008, 2014). In the WFLBC, the low abundance of adult steelhead was likely below the capacity needed to fully capitalize on the potential increase in production capacity of the system. Similarly, LWD treatments in the EFPR would enhance fish abundance/production in the system if the treatments increase capacity by creating more resources (i.e., food or space). However, this requires an adequate number of spawners so that juveniles occupy most of the suitable habitat in order to reap the benefits from increased capacity. The low abundance of juvenile fish in the EFPR (below carrying capacity) during recent years is likely masking the potential benefits of the LWD treatments at this time.

Adaptive Management

Data from the Potlatch River IMW study have guided the coordination and implementation of restoration projects. For example, we used the data to help better align restoration actions with monitoring priorities in the IMW. In 2016, the Idaho Office of Species Conservation formed the Potlatch River Implementation Group, comprised of restoration partners and management agencies (state and federal), to better coordinate and plan restoration activities within the watershed. In 2019, the group amended the Potlatch River Watershed Management Plan based on guidance from the ESA Recovery Plan for Snake River Spring/Summer Chinook Salmon and Snake River Basin Steelhead (NOAA 2017) and on data collected from the IMW study monitoring efforts. The 2019 amendment formally identified priority watersheds and provided structure for project planning, decision making, and monitoring efforts. To facilitate project planning and development, the group works collaboratively with state and federal agencies to develop five-year implementation plans for the priority watersheds. Projects are chosen based on current steelhead monitoring data, limiting factors being addressed, landownership, and evaluation of previous project effectiveness. This group has also fostered increased communication and coordination among funders, partners, and landowners through field tours, pre-project meetings, public presentations, peer reviewed publications, and social media products. Together, these efforts have created efficiencies in restoration implementation and increased support in the local communities.

We used an adaptive management approach to improve our ability to detect habitat and fish responses to restoration treatments. In 2017, we re-designed the tributary-level monitoring framework in the upper watershed by incorporating an additional control reach within the upper

EFPR watershed. The intent of the re-designed framework was to reduce the post-treatment monitoring time needed to detect a significant response in juvenile steelhead density estimates to restoration treatments (Uthe et al. 2017). Over the past five years, we have observed relatively similar trends in juvenile steelhead density estimates among the treatment reach and both control reaches, which indicate the reaches experience similar environmental phenomena (Roni et al. 2005). However, certain habitat metrics, such as LWD density, are increasing at a faster rate in the EFPR control reach relative to other areas, which suggests other factors such as beaver activity, reduced grazing pressure, or reduced logging activities may be acting on the system. In addition, the low number of juvenile steelhead captured in the EFPR treatment reach has made it challenging to evaluate trends in juvenile growth and survival among the treatment and control areas. To address this issue, we have extended our electrofishing efforts in the EFPR treatment reach to capture more fish and are working to establish new tagging goals in the upper watershed. However, an alternative method to estimate growth rates, such as taking scales and back-calculating growth, may be warranted due to the low density of juvenile steelhead in the area. In 2019, we installed a new IPTDS near the mouth of the EFPR to better evaluate parr-to-smolt survival between the EFPR treatment and control reach moving forward. Together, these improvements will allow us to more accurately assess steelhead response to restoration efforts in the upper watershed.

We also took steps to estimate steelhead smolt survival to LGR more accurately, which was problematic in the past. 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 in Idaho, with the tendency of some to delay migration for an additional year or multiple years after leaving their natal tributary (Dobos et al. 2020b). Previously, we utilized methods to estimate smolt survival that could not account for delayed migration and therefore, underestimated steelhead survival to LGR. For this report, we updated our smolt survival estimates for each index watershed by using the Basin TribPIT model developed by University of Washington researchers (Lady et al. 2014; Buchanan et al. 2015). The model estimates brood year survival by accommodating the variation in age at migration of steelhead (Lady et al. 2014; Buchanan et al. 2015). The model was previously 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. Estimating age-specific survival of juvenile steelhead will add to our understanding of the response of fish production to restoration efforts. This will be highly valuable for the EFPR because a large proportion of emigrants are age-1 fish that rear an additional year downstream within the Potlatch River before ocean migration (Bowersox et al. 2011). Accurately estimating age-specific survival will allow us to understand if the observed shifts in EFPR emigrant age structure and growth improve smolt survival through the hydrosystem. Furthermore, it will allow us to 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.

CONCLUSION

We achieved a diversity of successes in the planning, restoration, and monitoring components of the Potlatch River IMW project over the past five years. Improvements in the planning and prioritization of projects have led to more focused efforts in the index watersheds. Overall, the pace of restoration implementation in the Potlatch River has improved, but we still face challenges implementing key large-scale projects in the lower watershed. In particular, the cancellation of the Big Bear Falls restoration project was a major setback. Recent fish passage projects in the lower watershed have elicited positive responses in steelhead in terms of increased connectivity and distribution. However, we have not observed any increase in juvenile production as a result of these projects, likely due to the degraded condition of the restored habitat and low abundance of fish to fully occupy these areas. In the upper watershed, we continue to document a positive shift towards larger and older steelhead smolts emigrating from the EFPR which is encouraging. Although the underlying mechanisms for the observed shifts require further study, the evidence suggests that rearing conditions are improving in the EFPR. The low adult steelhead returns over the past five years are disconcerting and have highlighted shortcomings in our monitoring infrastructure and design that need to be addressed.

The next 10 to 15 years will be critical to the success of the Potlatch IMW project. Plans are in place to address the shortcomings in our monitoring infrastructure and design. Results from the Big Bear Falls trap and haul pilot project will be vital in determining future restoration efforts in the Big Bear Creek drainage. In addition, the reach-scale evaluation of large wood projects in the EFPR will allow us to better understand the effectiveness of these techniques in promoting floodplain connectivity and habitat complexity. Continued evaluation of fish response to these large wood projects should provide a better understanding of the mechanisms influencing the emigrant life history changes we are observing in the EFPR. We have built a solid foundation of restoration and monitoring over the course of this study and these efforts will continue into the future to aid in the recovery of Potlatch River wild steelhead.

PART 2 TABLES

Table 2.1. Year, location, project type, and amount of stream treated in the Big Bear Creek (BBC) and East Fork Potlatch River (EFPR) watersheds from 2017-2021.

Year	Tributary	Project type	Stream treated (km)
	Big Bear Creek		
2017	Nora Creek	Meadow/wetland restoration	2.4
2020	West Fork BBC	Meadow/wetland restoration	0.73
2020	BBC	Meadow/wetland restoration	0.35
2021	Middle Fork BBC	Meadow/wetland restoration	2.06
2018	Big Meadow Creek	Passage barrier modification (baffles)	11
2017/2018	Big Meadow Creek	Passage barrier removal	
2017-2021	BBC	Passage barrier modification	≥ 20
2018-19	BBC	Sediment reduction	0.13
	East Fork Potlatch River		
2018	Fry Creek	Floodplain reconnection	0.3
2021	Fry Creek	Floodplain reconnection	1.8
2017-2019	EFPR	Floodplain reconnection/riparian planting	2.8
2017	EFPR	LWD installation/floodplain reconnection	0.4
2019	EFPR	LWD installation/floodplain reconnection	0.9
2020	EFPR	LWD installation/floodplain reconnection	1.6
2021	EFPR	LWD installation/floodplain reconnection	1.6
2021	EFPR	LWD installation/floodplain reconnection	1
2017	Mallory Creek	Passage barrier removal and road BMP	

Table 2.2. Analytical layout of the fish and habitat parameters measured in select treatment areas in the Potlatch River watershed, Idaho.

Study area	Parameter	Pretreatment years	Treatment years
Big Bear Creek	Wetted habitat	2008–2016	2017–2021
	Pool density	2008–2016	2017–2021
Corral Creek	Wetted habitat	2008-2010	2011-2021
	Pool density	2008-2010	2011-2021
West Fork Little Bear Creek	Juvenile steelhead density	1996, 2013	2014-2021
	Parr-to-smolt survival	2008-2013	2014-2020
	Juvenile steelhead growth	na	2014-2021
East Fork Potlatch River (treatment area)	Canopy cover	2003/04, 2008	2013-2021
	Large wood density	2003, 2008	2013-2021
	Pool density	2003/04	2013-2021
	Juvenile steelhead density	2004	2013-2021

Table 2.3. Number of juvenile steelhead (>80 mm) tagged at rotary screw traps in Big Bear Creek and East Fork Potlatch River and subsequently detected in the hydrosystem for brood year survival analyses from 2008-2021.

Watershed	**Year**	**Number of fish tagged**	**Number of detections in hydrosystem**
Big Bear Creek	2005	2,318	1,501
	2006	567	198
	2007	1,281	320
	2008	1,062	575
	2009	968	474
	2010	1,915	1,569
	2011	562	446
	2012	1,842	584
	2013	1,865	983
	2014	2,329	1,709
	2015	2,912	1,910
	2016	1,283	528
	2017	2,183	1,499
	2018	4,134	3,175
	2019	1,183	620
	2020	1,109	664
	2021	390	127
East Fork Potlatch River	2008	432	107
	2009	1,243	154
	2010	1,980	1,086
	2011	1,132	457
	2012	968	156
	2013	1,505	790
	2014	722	298
	2015	815	315
	2016	634	106
	2017	1,096	156
	2018	450	77
	2019	237	59
	2020	78	20
	2021	162	55

Table 2.4. Number of juvenile steelhead (>80 mm) tagged in Potlatch River tributaries for parr-to-smolt survival analyses from 2008-2021. Values in parenthesis indicate number of tagged fish subsequently detected in the hydrosystem the following spring.

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 (28)	38 (3)	302 (29)	18 (2)
2021	212 (5)	260 (21)	385 (25)	409 (16)	302 (17)	2 (0)

Table 2.5. Summer-fall growth of juvenile steelhead (≥80 mm) in select tributaries in the lower Potlatch River watershed, Idaho from 2014-2021. 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)	S.E.
2013	PNC	0	--	--	--	--	--
	WFLBC	6	8.7	6.7	142	0.061	0.019
	LBC	0	--	--	--	--	--
2014	PNC	29	8.3	8.0	147	0.057	0.010
	WFLBC	26	7.5	7.7	123	0.059	0.012
	LBC	0	--	--	--	--	--
2015	PNC	17	12.4	16.2	141	0.086	0.028
	WFLBC	10	3.7	4.3	132	0.028	0.010
	LBC	2	17.0	5.7	138	0.124	0.032
2016	PNC	62	5.7	8.4	130	0.044	0.008
	WFLBC	45	0.8	4.4	117	0.007	0.006
	LBC	77	2.9	5.7	127	0.023	0.005
2017	PNC	54	8.3	7.3	133	0.062	0.007
	WFLBC	41	1.1	4.8	118	0.009	0.006
	LBC	18	1.1	4.2	124	0.008	0.008
2018	PNC	29	11.2	6.8	134	0.083	0.009
	WFLBC	33	2.9	4.2	116	0.025	0.006
	LBC	32	3.8	5.5	141	0.026	0.007
2019	PNC	0	--	--	--	--	--
	WFLBC	31	4.5	5.1	110	0.040	0.008
	LBC	16	2.4	2.7	131	0.018	0.005
2020	PNC	0	--	--	--	--	--
	WFLBC	29	7.7	4.7	113	0.068	0.008
	LBC	51	5.1	4.0	105	0.049	0.005
2021	PNC	19	7.1	9.6	120	0.058	0.013
	WFLBC	36	2.3	5.0	92	0.025	0.011
	LBC	29	4.2	5.1	109	0.039	0.007

PART 2 FIGURES

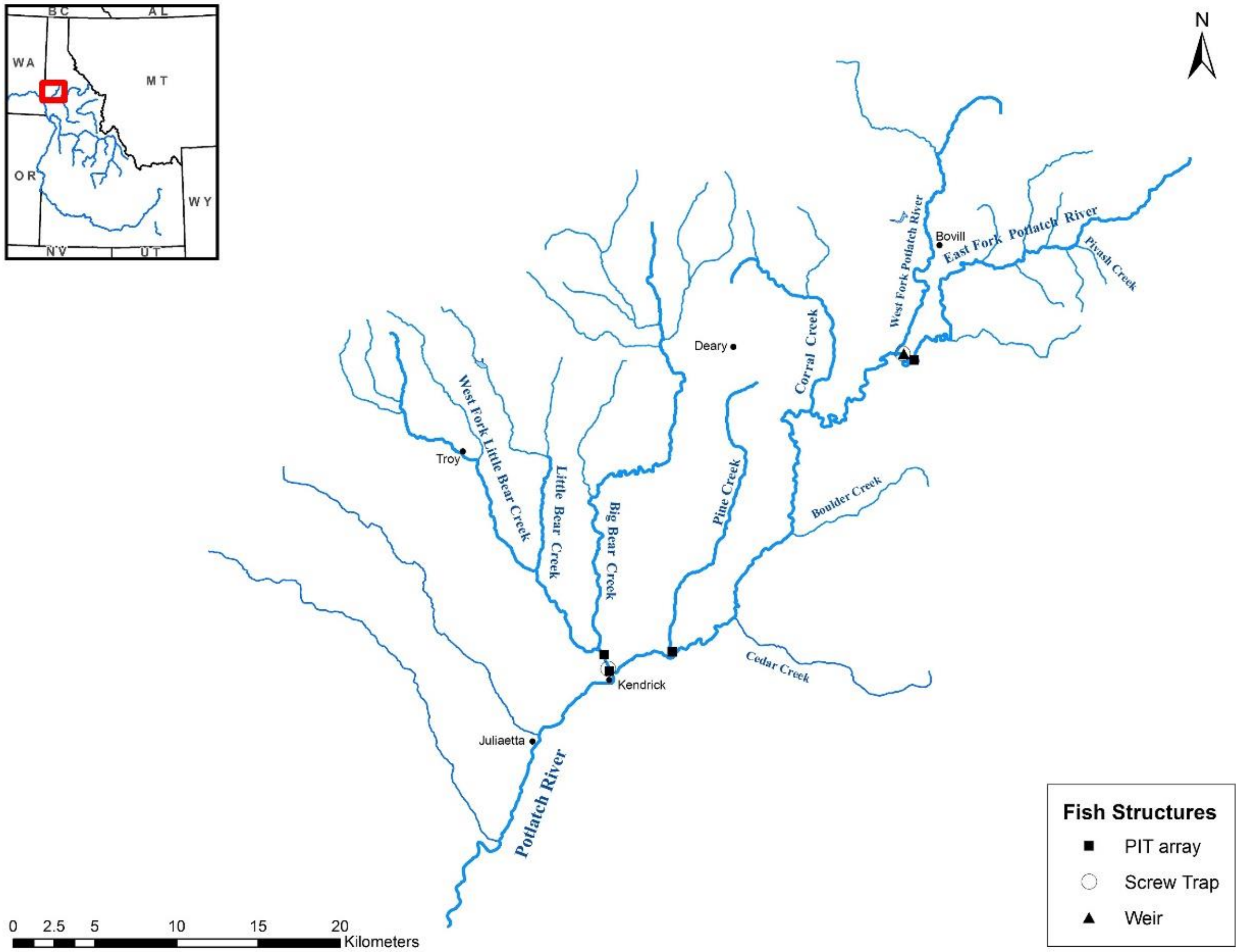

Figure 2.1. Key features and monitoring infrastructure in the Potlatch River watershed in northern Idaho.

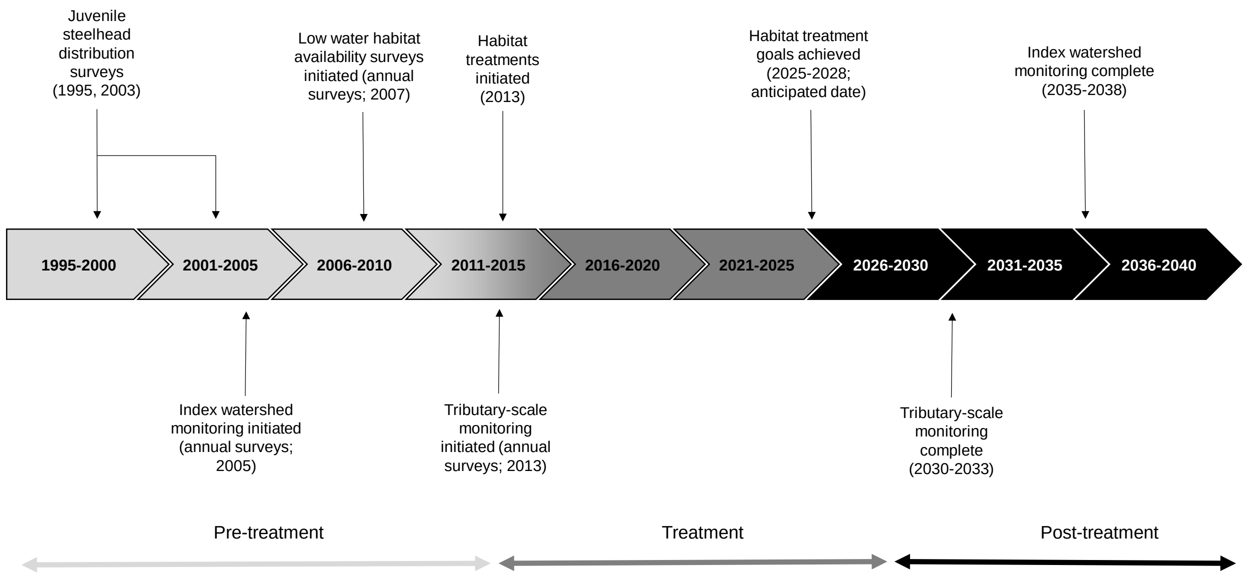

Figure 2.2. Timeline of project monitoring and restoration implementation in Big Bear Creek watershed in five-year increments.

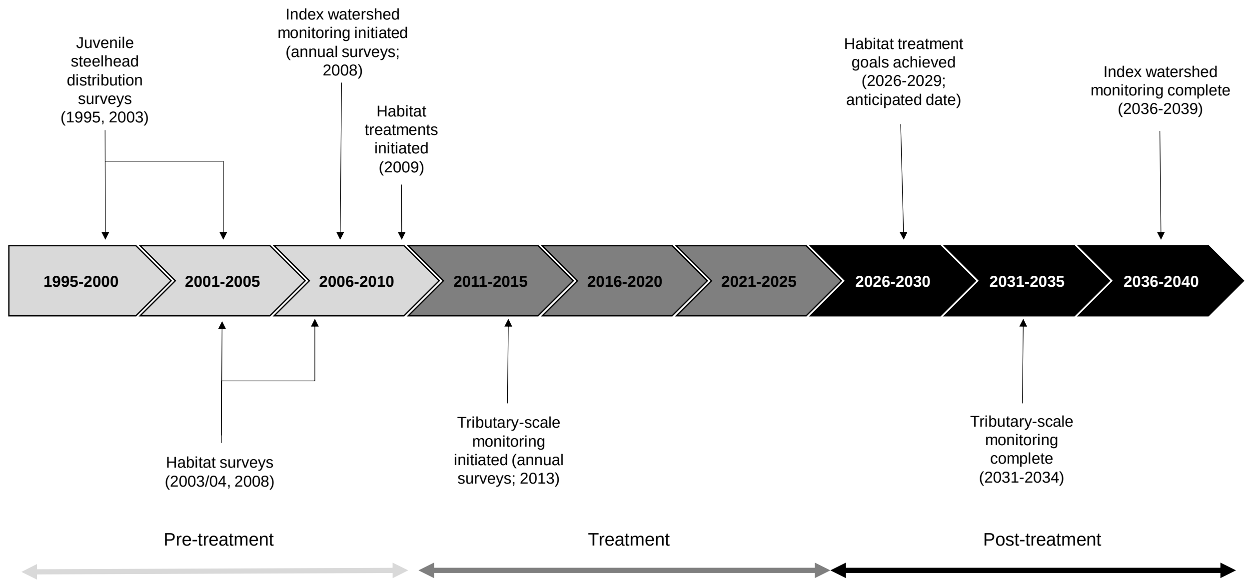

Figure 2.3. Timeline of project monitoring and restoration implementation in the East Fork Potlatch River watershed.

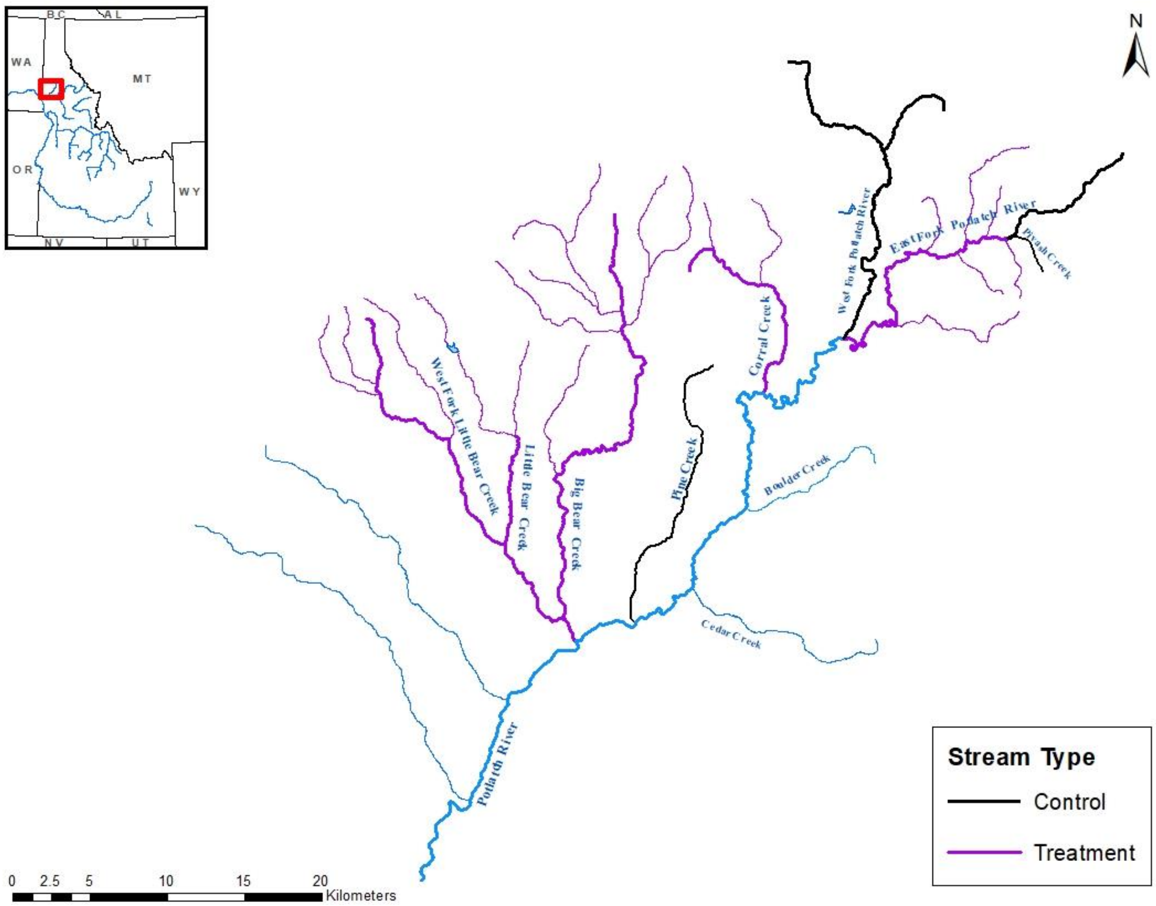

Figure 2.4. Study area map showing treatment and control reaches in the lower and upper Potlatch River watersheds.

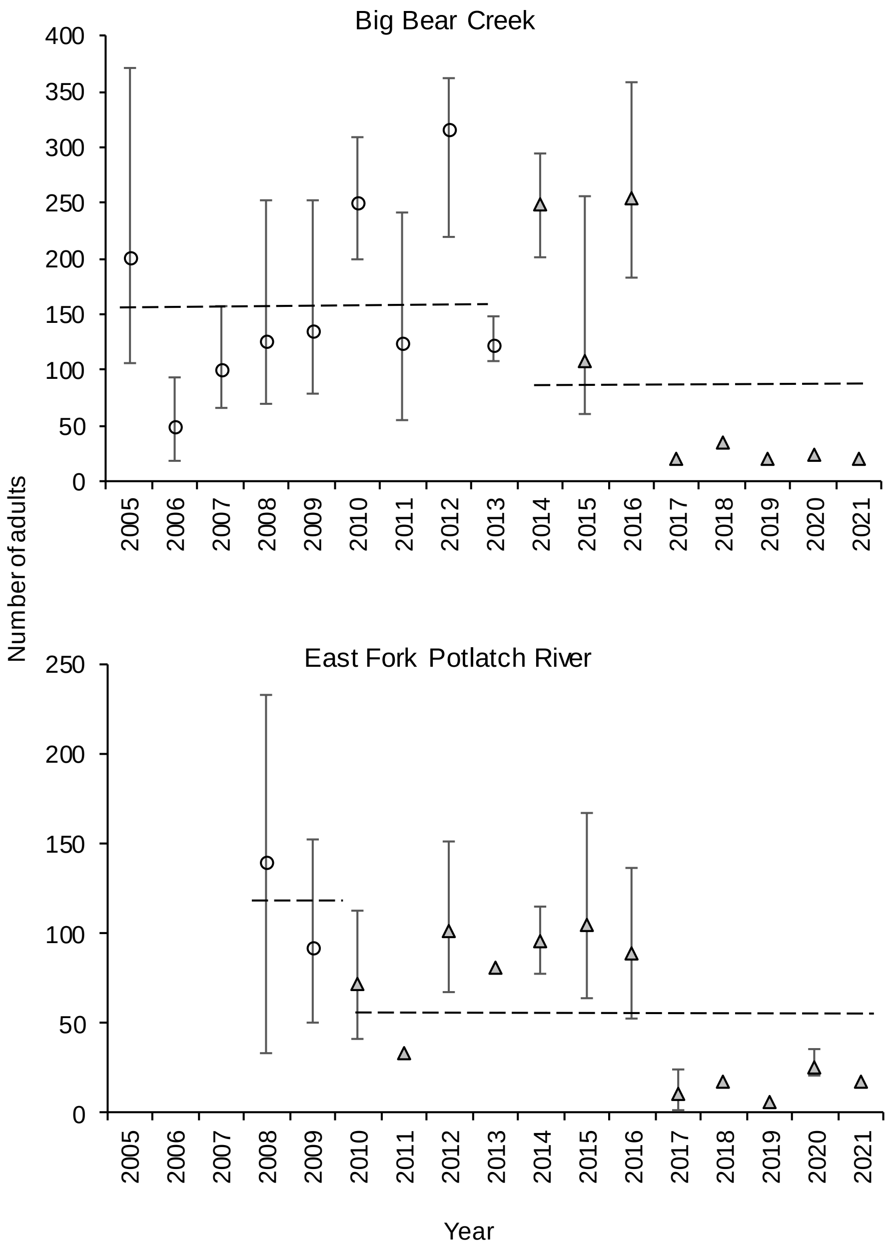

Figure 2.5. Abundance of wild adult steelhead in Big Bear Creek and the East Fork Potlatch River watersheds, 2005-2021. East Fork Potlatch River estimates begin in 2008. Error bars are at 95% confidence intervals, but could not be calculated in some years due to low detections or captures at sites. Open circles indicate pretreatment years and filled triangles indicate treatment years. Dashed lines indicate mean abundance for each time period.

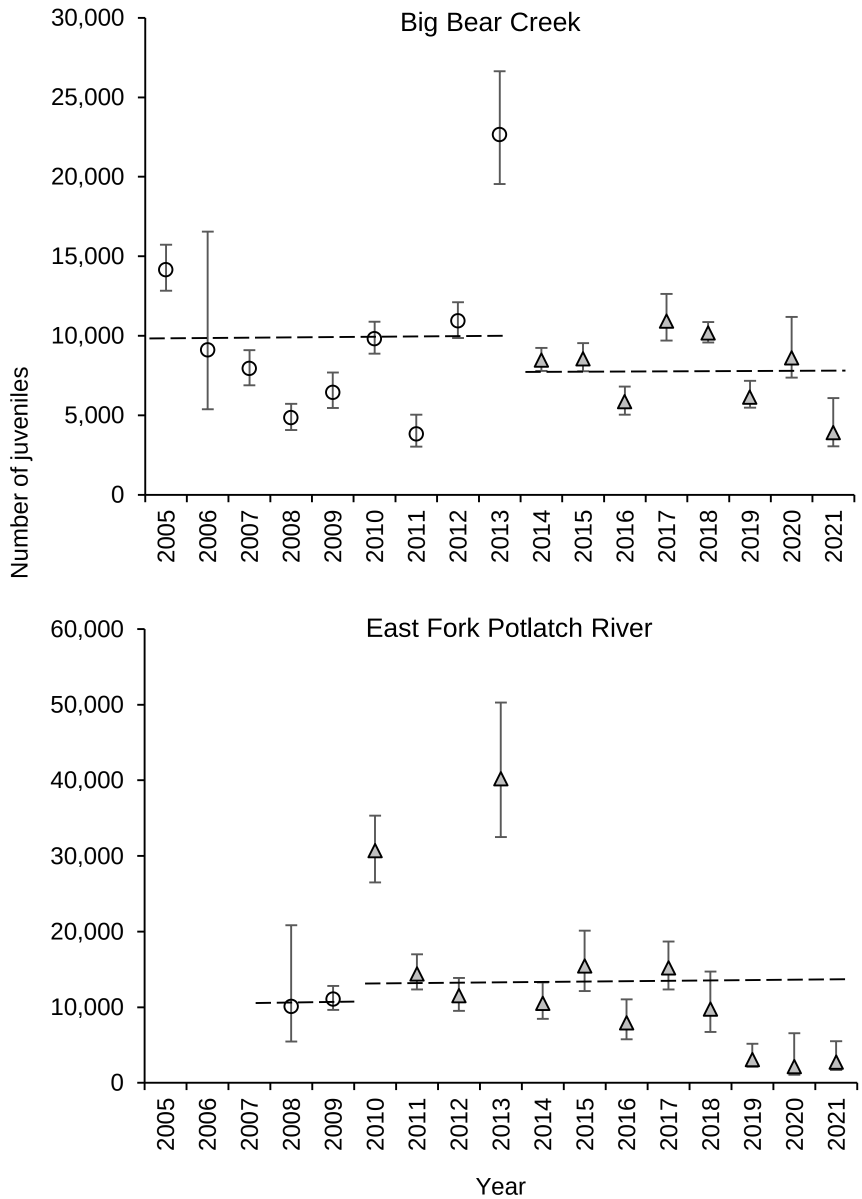

Figure 2.6. Abundance of wild juvenile steelhead emigrants during the spring season in the Big Bear Creek and East Fork Potlatch River watersheds, 2005-2021. Error bars are 95% confidence intervals. East Fork Potlatch River estimates begin in 2008. Open circles indicate pretreatment years and filled triangles indicate treatment years. Dashed lines indicate mean abundance for each time period.

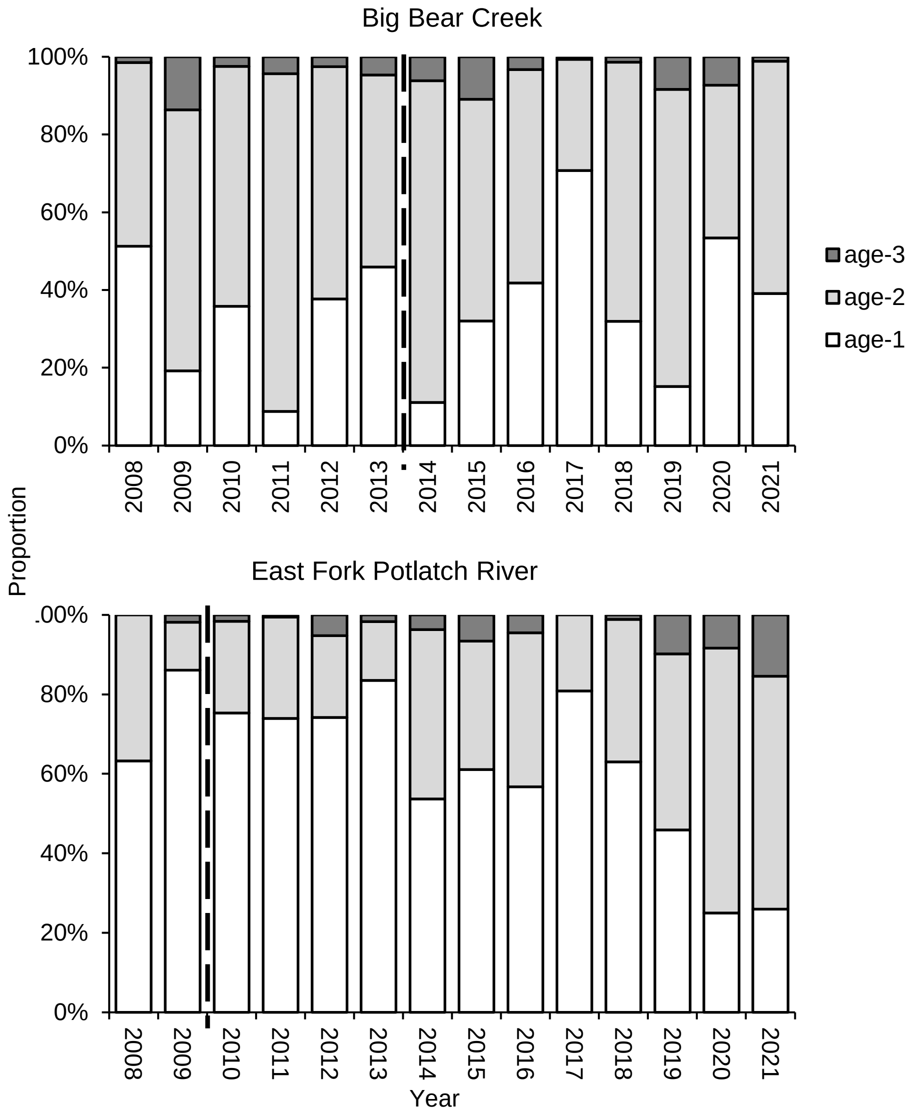

Figure 2.7. Age composition of wild juvenile steelhead emigrants captured at rotary screw traps during the spring season in the Big Bear Creek and East Fork Potlatch River watersheds, 2008-2021. Dashed lines indicates year of first restoration treatment. Years prior to the dashed line are pretreatment years and years after dashed line are treatment years in each watershed.

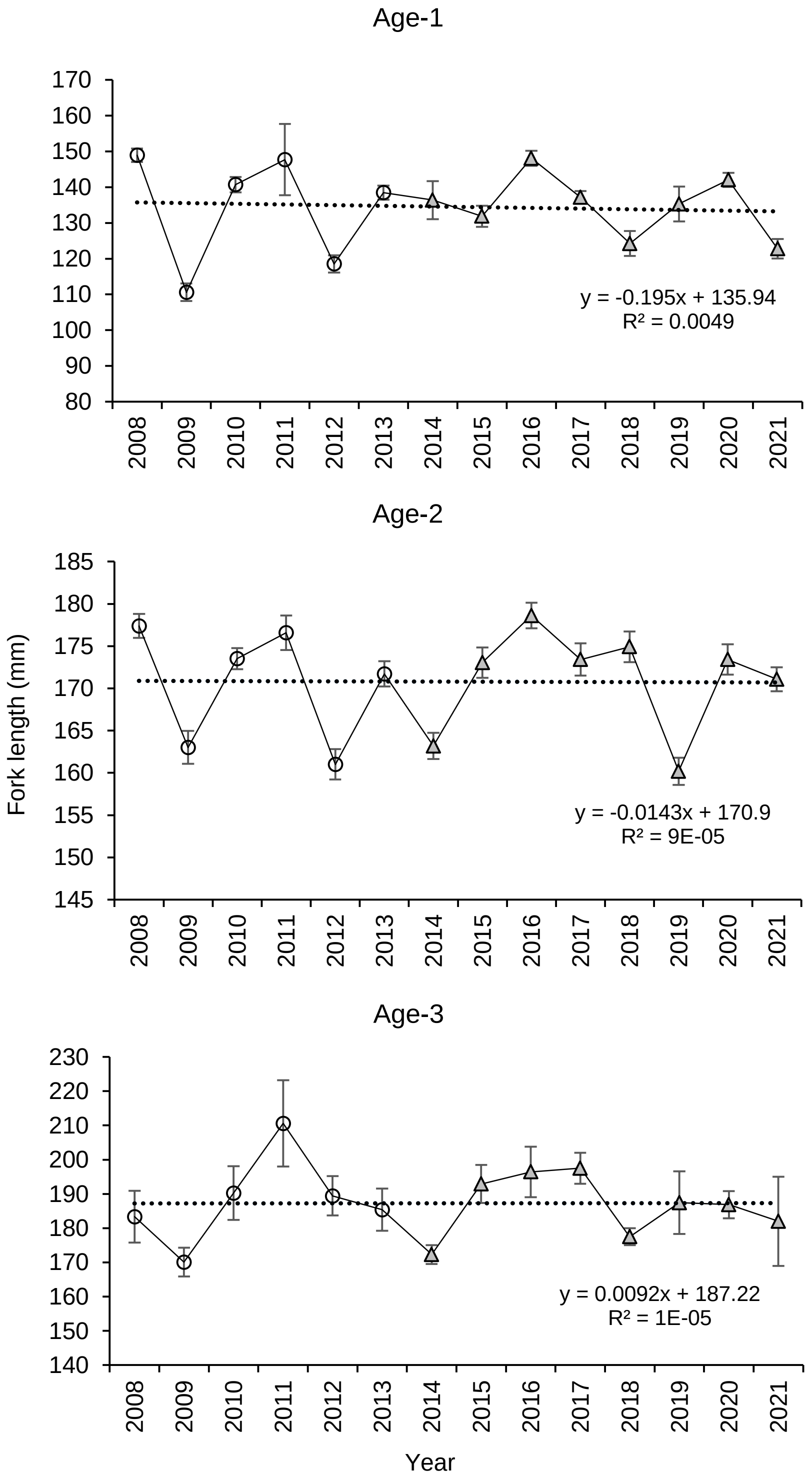

Figure 2.8. Mean length at age of wild juvenile steelhead emigrants captured at the rotary screw trap during the spring season in the Big Bear Creek watershed, 2008-2021. Error bars are S.E. Open circles indicate pretreatment years and filled triangles indicate treatment years.

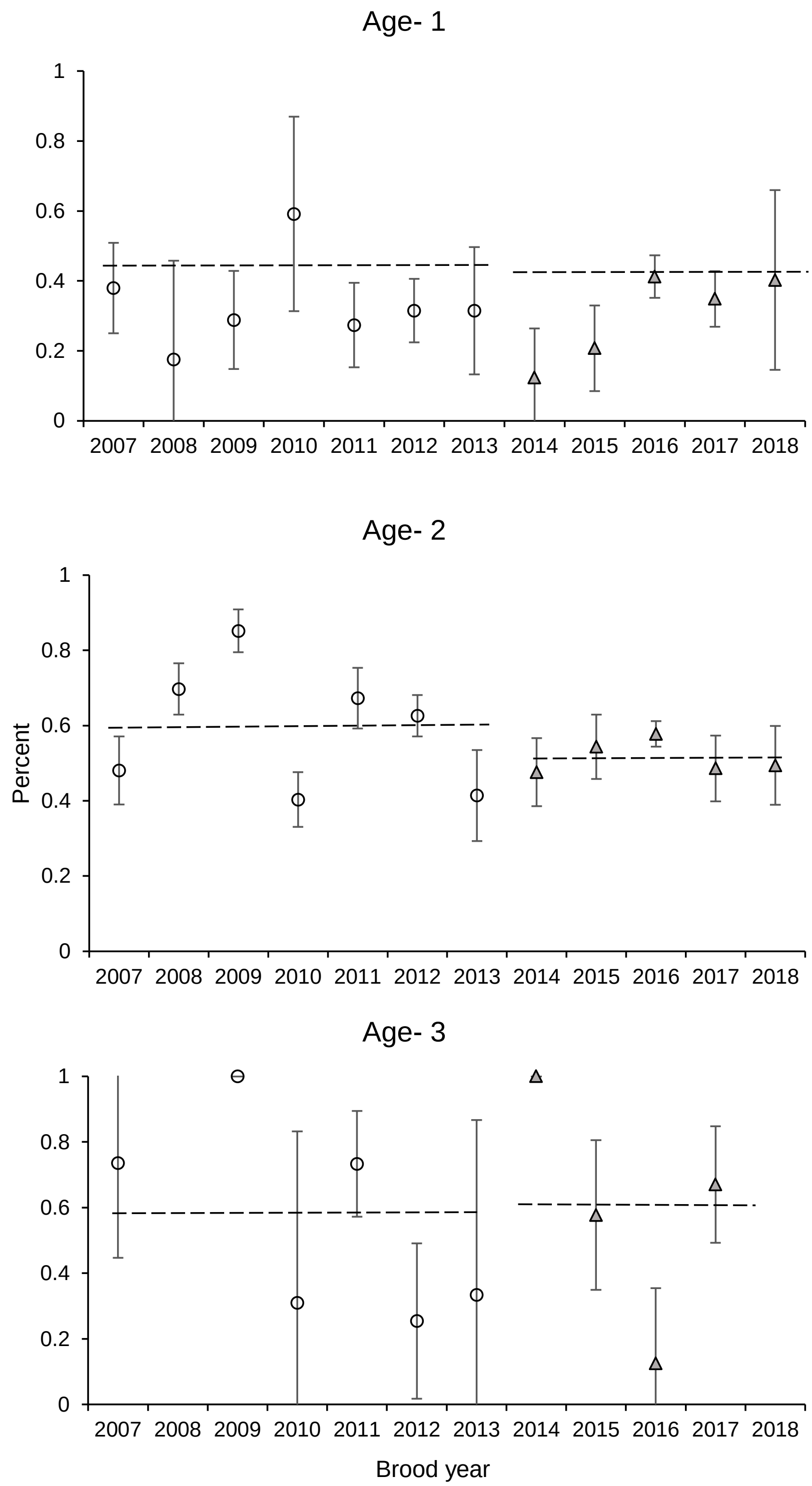

Figure 2.9. Age based brood year survival estimates for spring emigrants from Big Bear Creek rotary screw trap downstream to Lower Granite Dam, 2007-2018. Error bars are 95% C.I.

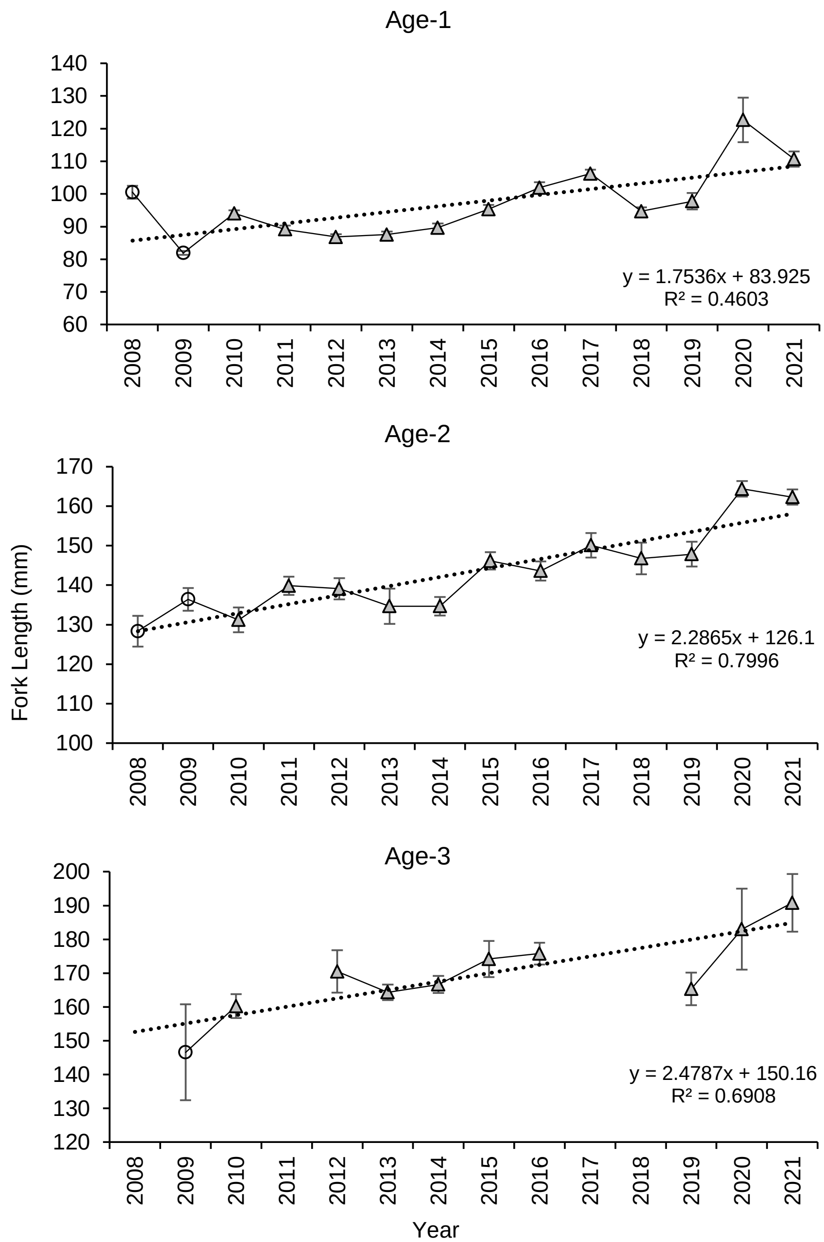

Figure 2.10. Mean length at age of wild juvenile steelhead emigrants captured at the rotary screw trap during the spring season in the East Fork Potlatch River watershed, 2008-2021. Error bars are S.E. Open circles indicate pretreatment years and filled triangles indicate treatment years.

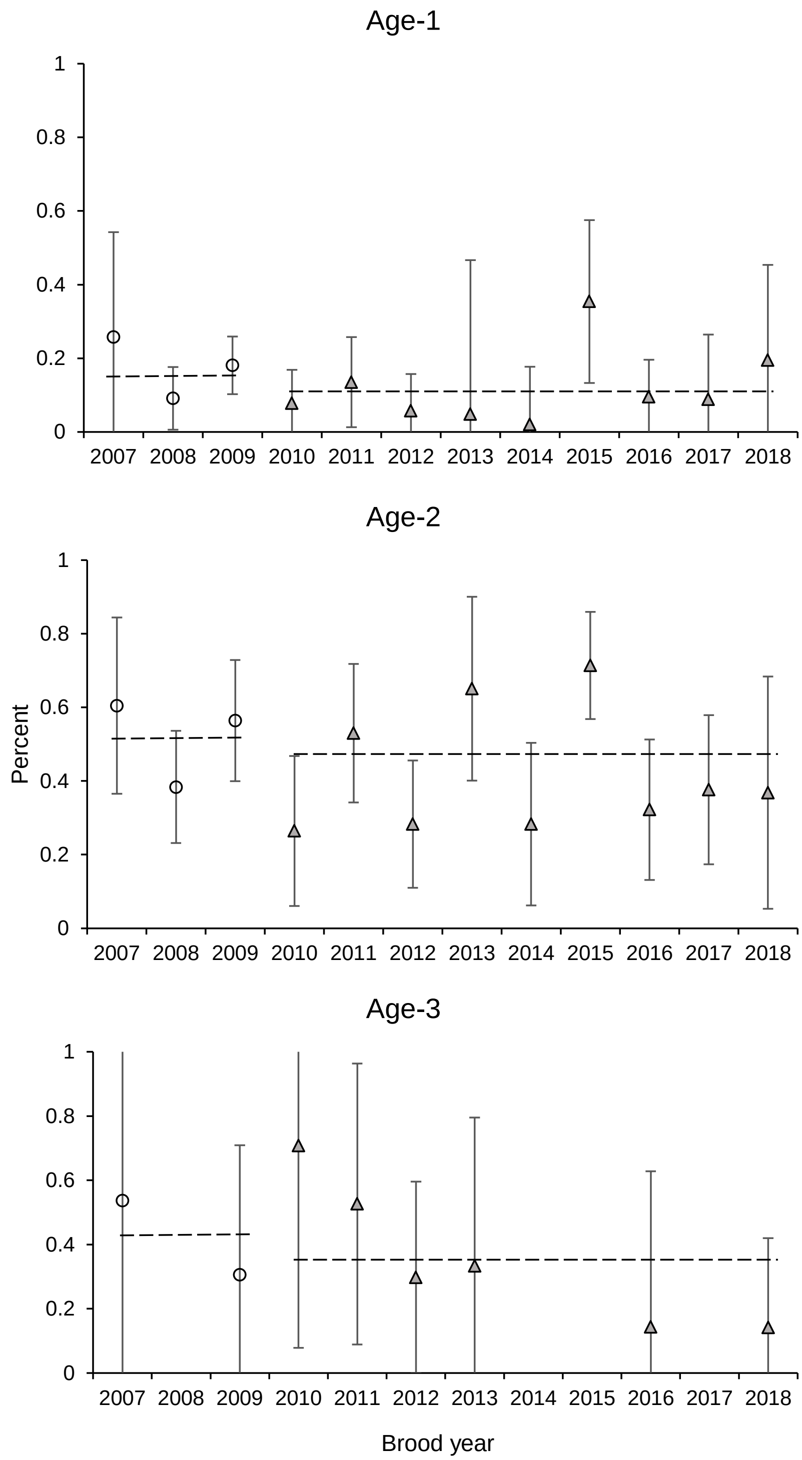

Figure 2.11. Age based brood year survival estimates for spring emigrants from the East Fork Potlatch River rotary screw trap downstream to Lower Granite Dam, 2007-2018. Error bars are 95% C.I.

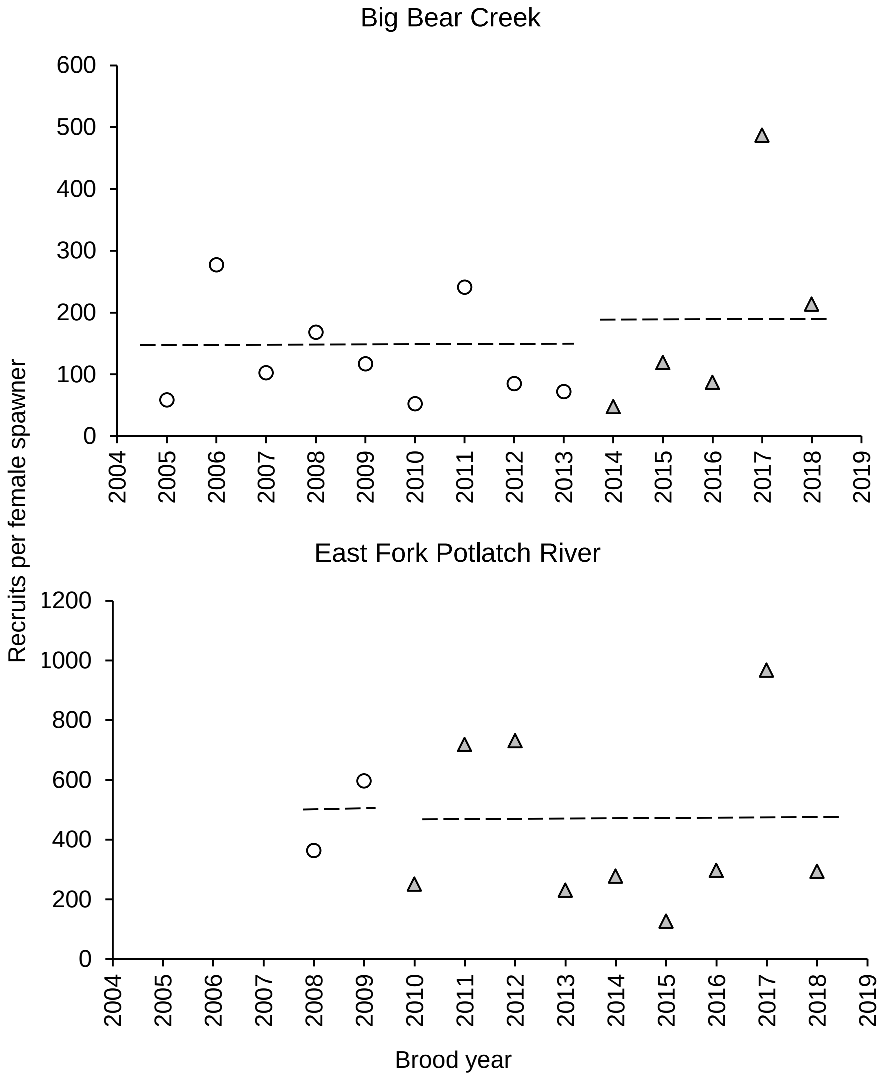

Figure 2.12. Productivity (juvenile recruits per female spawner) for the Big Bear Creek and East Fork Potlatch River watersheds. Big Bear Creek data are BYs 2005-2018 and the East Fork Potlatch River data are BYs 2008-2018. Open circles indicate pretreatment years and filled triangles indicate treatment years. Dashed lines indicate mean productivity for each time period.

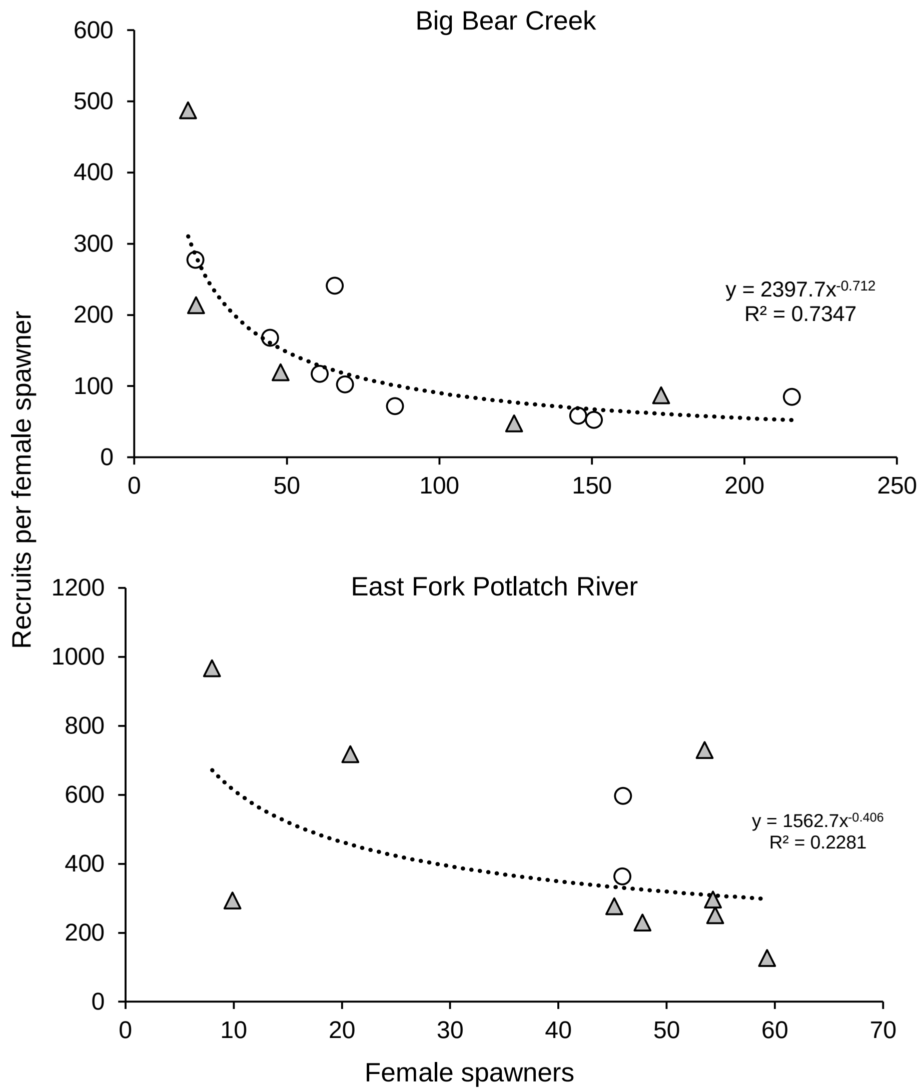

Figure 2.13. Productivity (juvenile recruits per female spawner) versus number of female spawners for the Big Bear Creek and East Fork Potlatch River watersheds. Big Bear Creek data are BYs 2005-2018 and the East Fork Potlatch River data are BYs 2008-2018. Open circles indicate pretreatment years and filled triangles indicate treatment years.

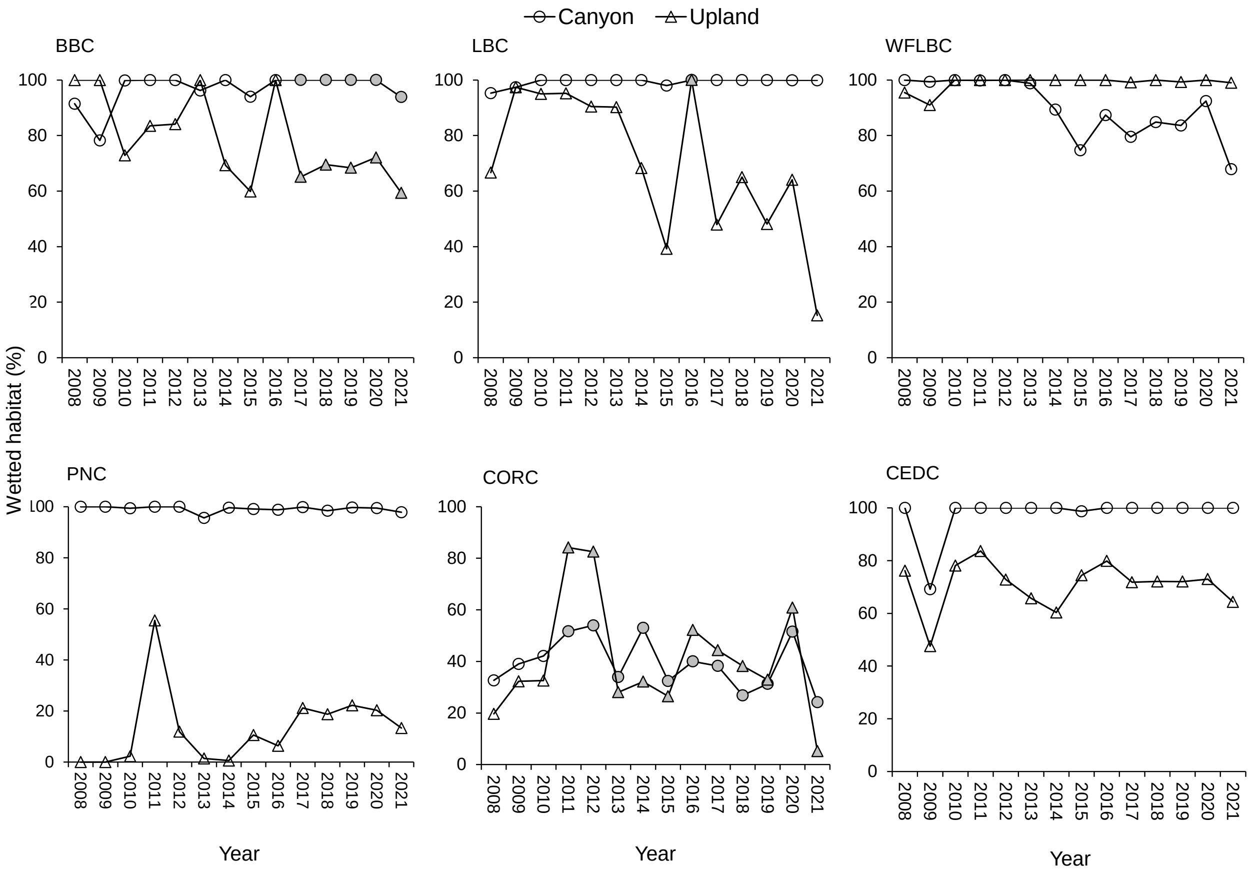

Figure 2.14. The amount of wetted habitat in the canyon and upland sections of four treatment tributaries (BBC-Big Bear Creek, LBC-Little Bear Creek, WFLBC-West Fork Little Bear Creek, and CORC-Corral Creek) and two control tributaries (PNC-Pine Creek and CEDC-Cedar Creek) in the lower Potlatch River watershed from 2008-2021. Shaded symbols indicate treatment periods and open symbols indicate non-treatment periods.

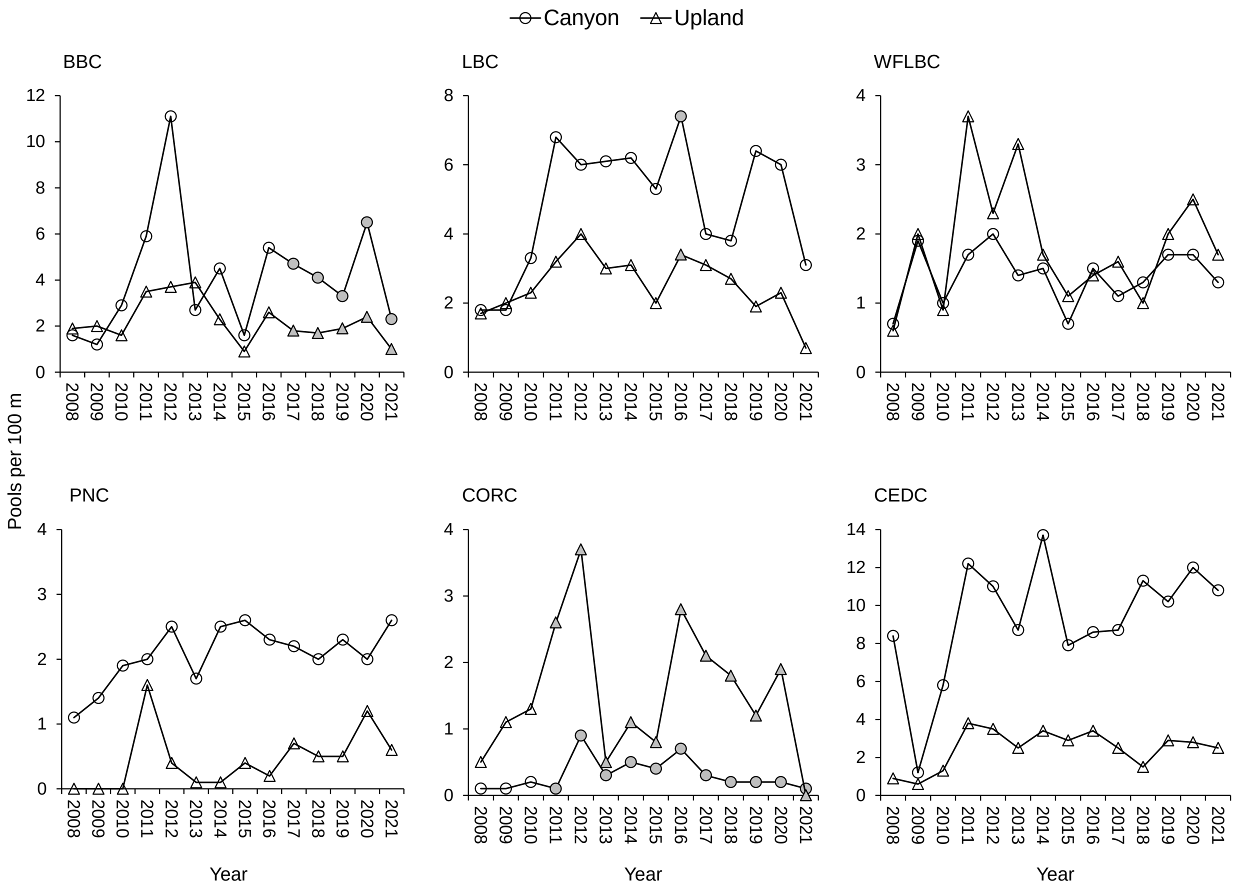

Figure 2.15. Pool density in the canyon and upland sections of four treatment tributaries (BBC-Big Bear Creek, LBC-Little Bear Creek, WFLBC-West Fork Little Bear Creek, and CORC-Corral Creek) and two control tributaries (PNC-Pine Creek and CEDC-Cedar Creek) in the lower Potlatch River watershed from 2008-2021. Shaded symbols indicate treatment periods and open symbols indicate non-treatment periods).

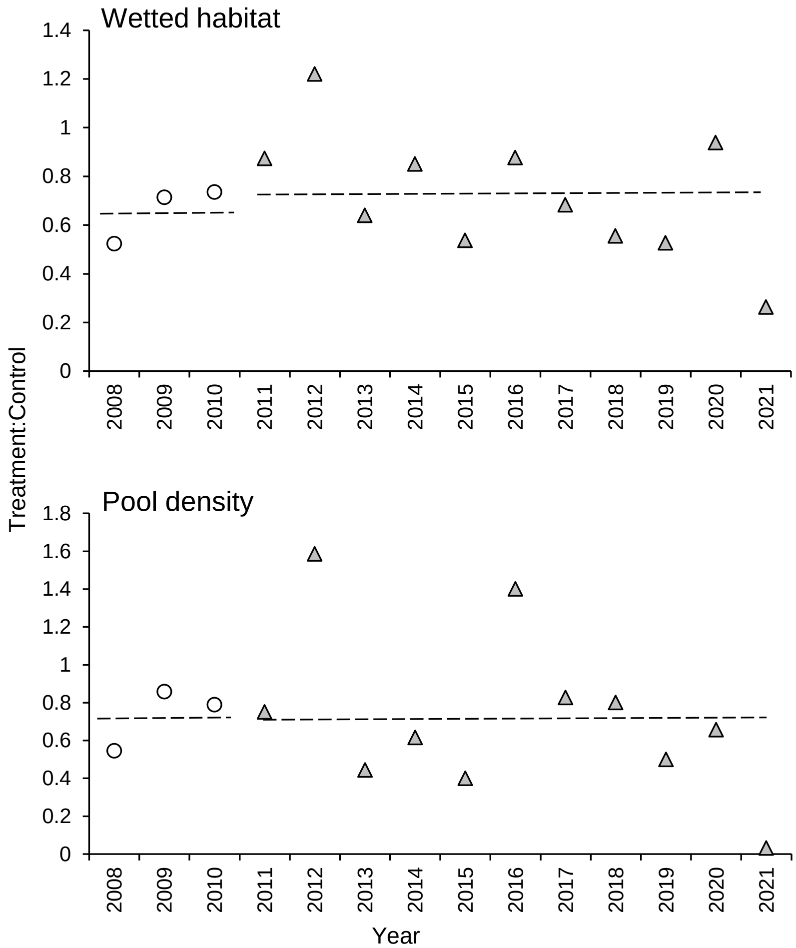

Figure 2.16. Habitat metric ratio values (treatment:control) for Corral Creek (treatment tributary) and Pine Creek (control tributary) in the lower Potlatch River watershed during 2008-2021. The upper and lower panels indicate wetted habitat and pool density relationships respectively. Open circles are pretreatment years and shaded triangles are treatment years. Dashed lines indicate mean ratio values for each time period.

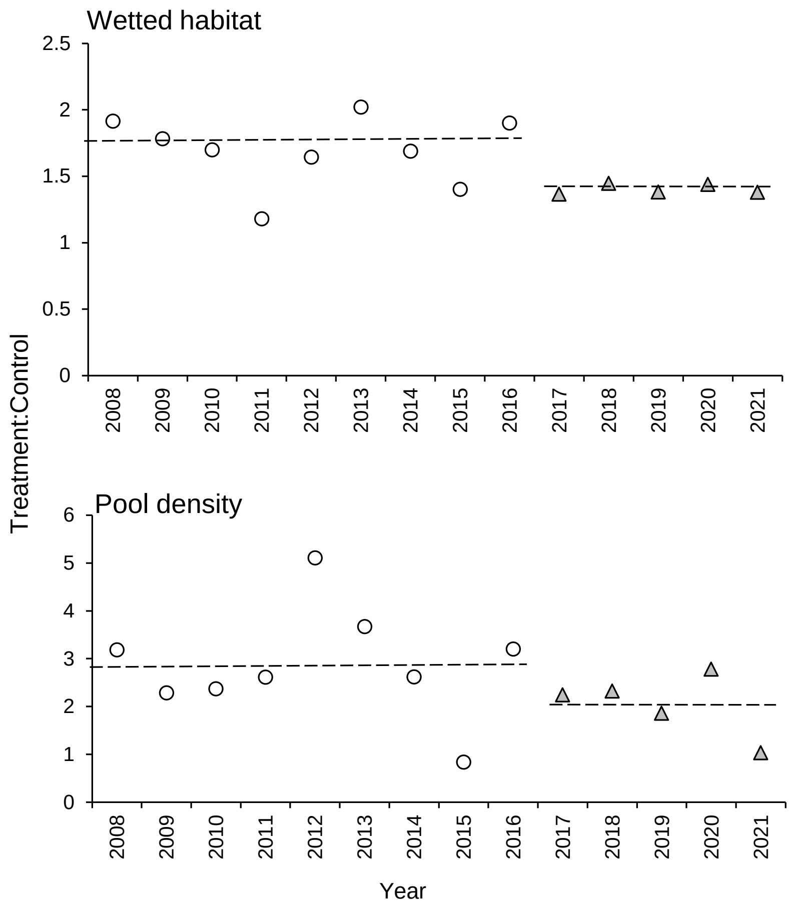

Figure 2.17. Habitat metric ratio values (treatment:control) for Big Bear Creek (treatment tributary) and Pine Creek (control tributary) in the lower Potlatch River watershed during 2008-2021. The upper and lower panels indicate wetted habitat and pool density relationships respectively. Open circles are pretreatment years and shaded triangles are treatment years. Dashed lines indicate mean ratio values for each time period.

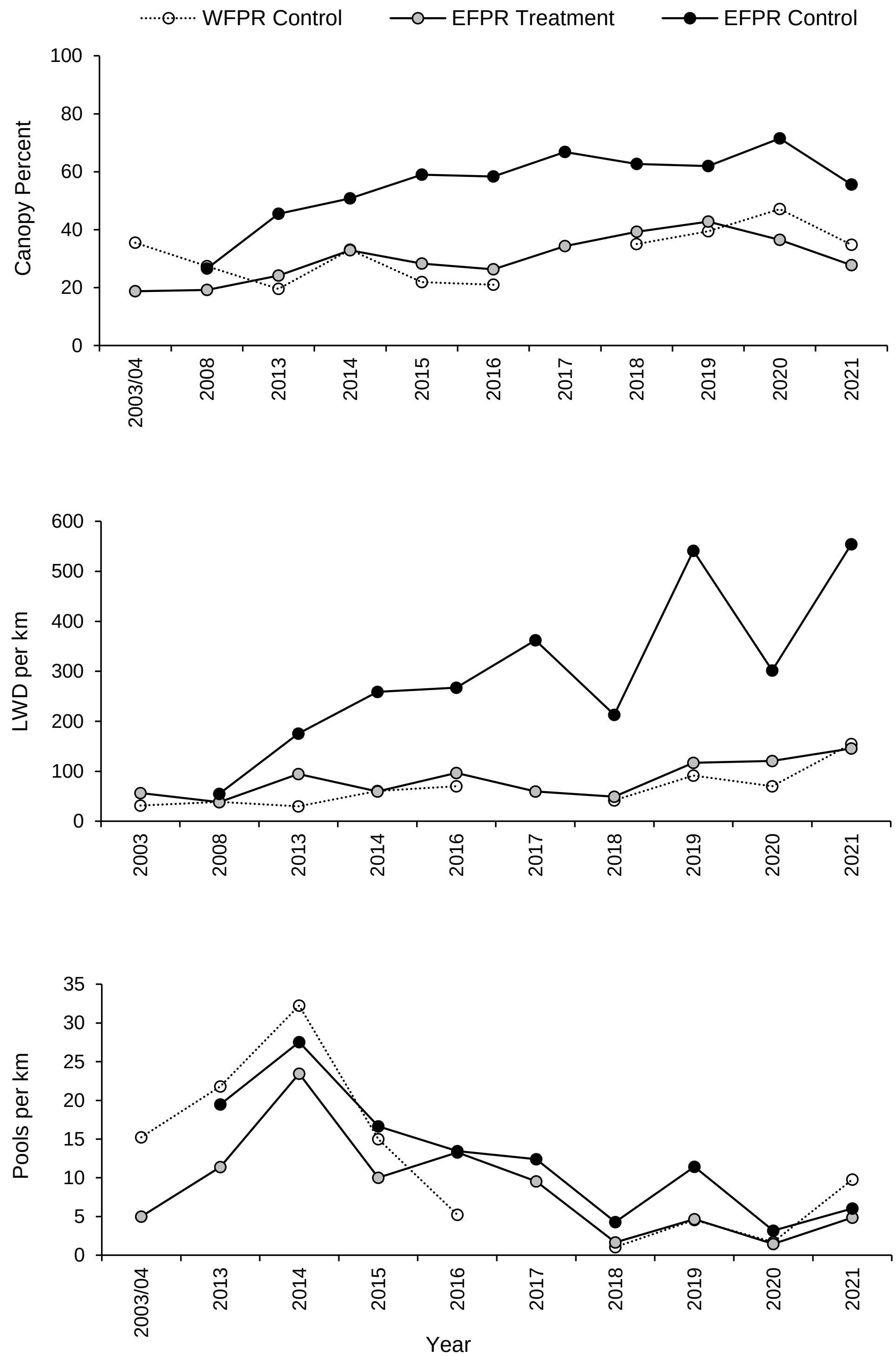

Figure 2.18. Canopy cover, large wood density (LWD), and pool density and within the treatment area (EFPR Treatment) and control areas (WFPR and EFPR Control) in the upper Potlatch River watershed during 2003–2021. Restoration treatments began in 2009.

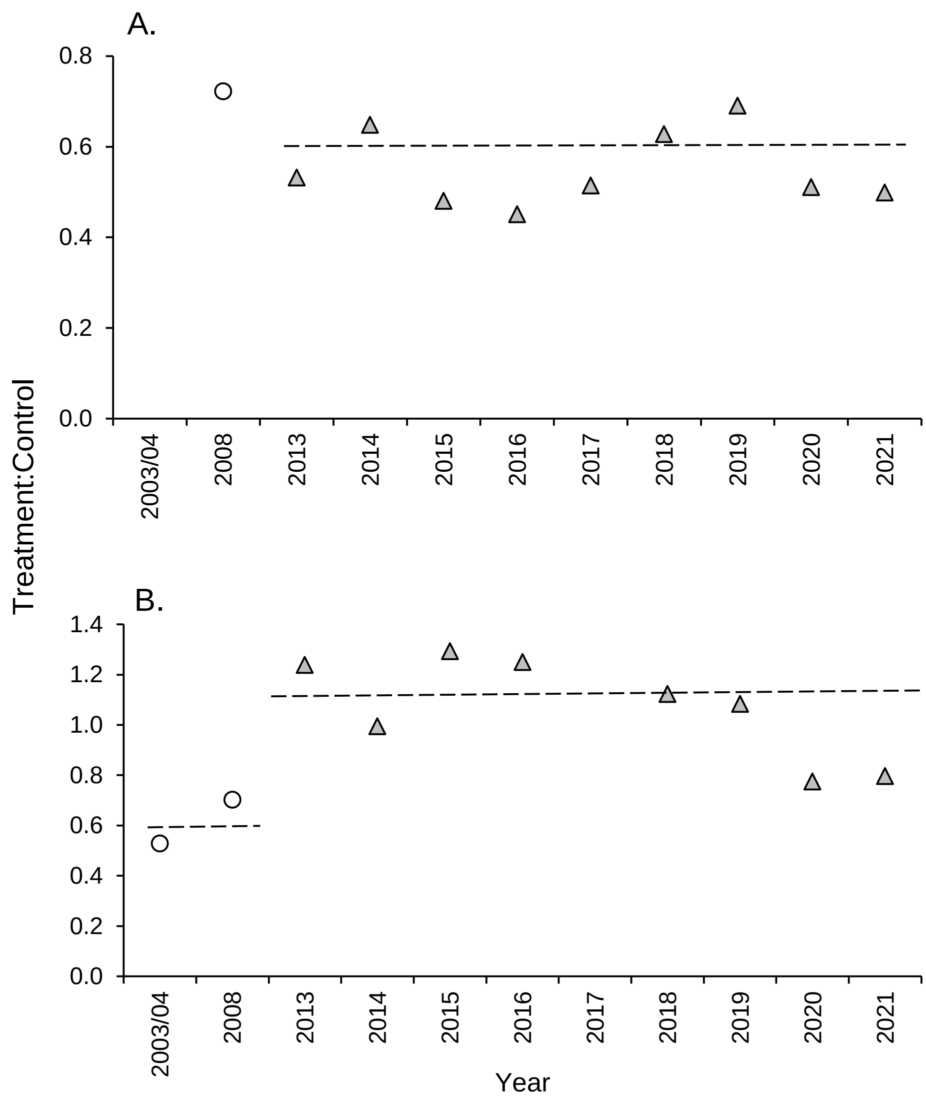

Figure 2.19. Habitat metric ratio values (treatment:control) for percent canopy cover in the upper Potlatch River watershed during 2003-2021. Panel A shows the relationship between the EFPR treatment and EFPR control areas, whereas Panel B shows the relationship between the EFPR treatment and WFPR control areas. Open circles are pretreatment years and shaded triangles are treatment years. Dashed lines indicate mean ratio values for each time period.

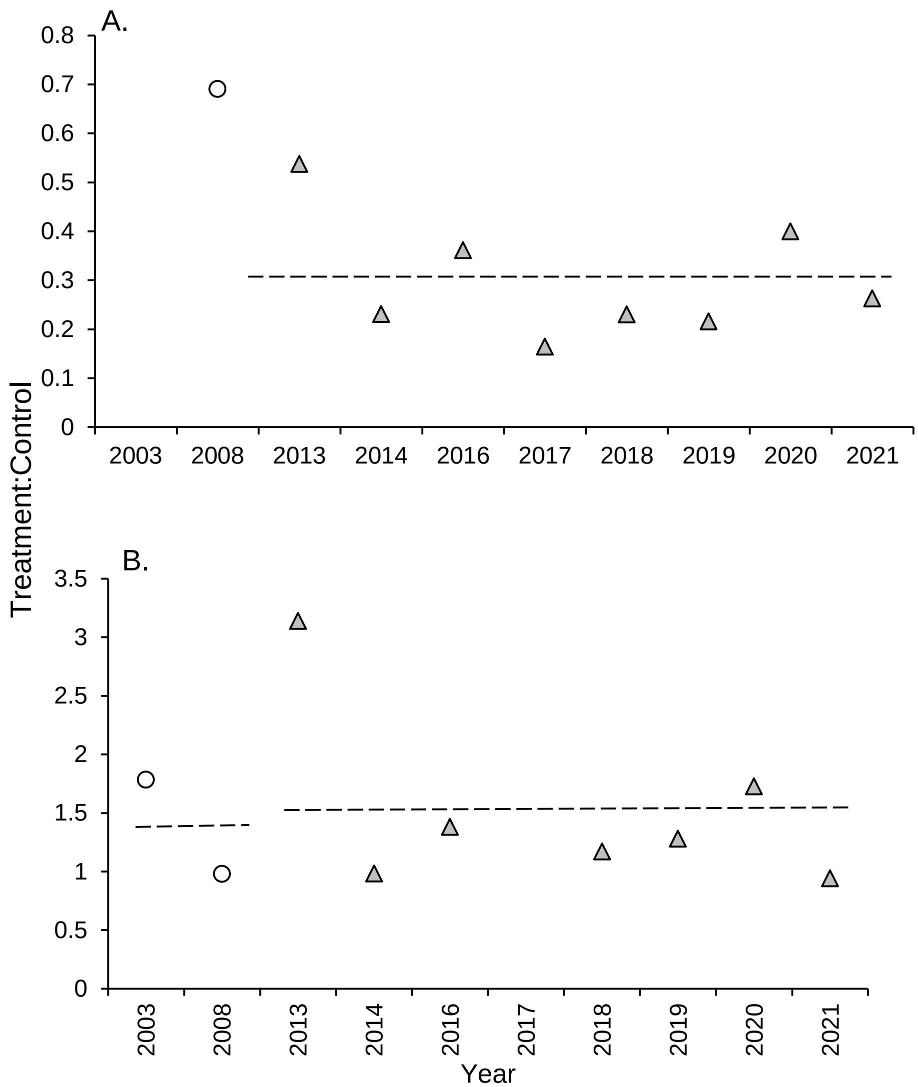

Figure 2.20. Habitat metric ratio values (treatment:control) for large wood density in the upper Potlatch River watershed during 2003-2021. Panel A shows the relationship between the EFPR treatment and EFPR control areas, whereas Panel B shows the relationship between the EFPR treatment and WFPR control areas. Open circles are pretreatment years and shaded triangles are treatment years. Dashed lines indicate mean ratio values for each time period.

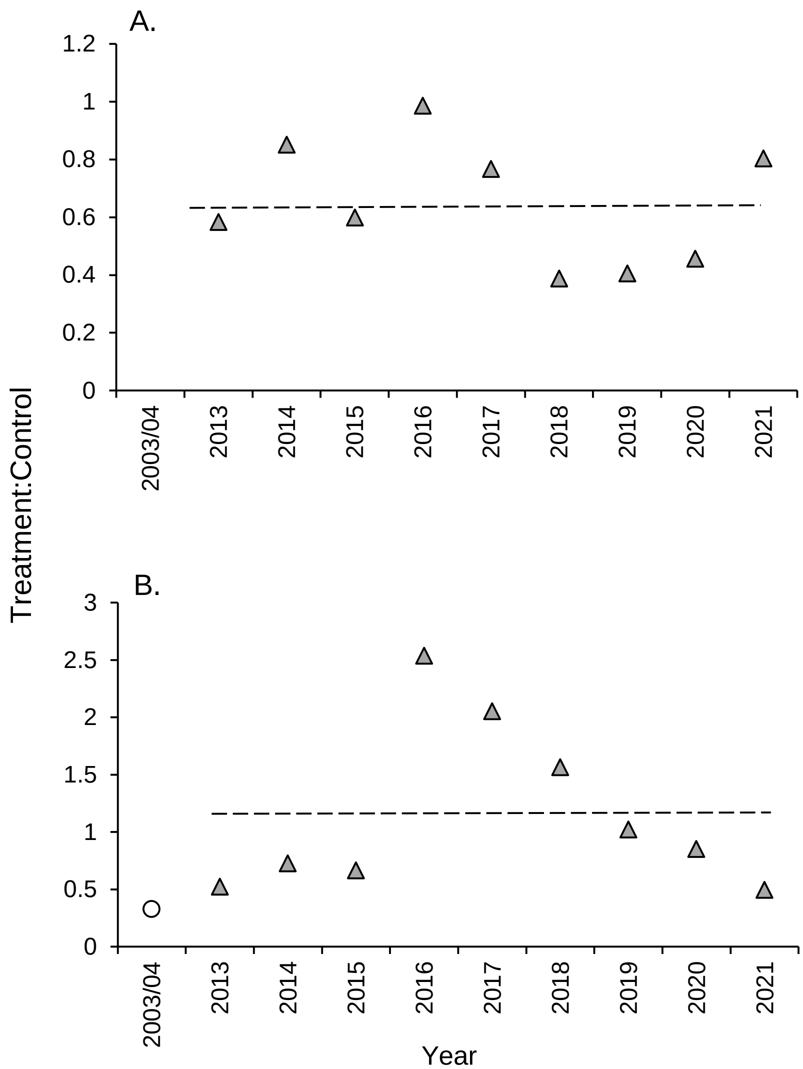

Figure 2.21. Habitat metric ratio values (treatment:control) for pool density in the upper Potlatch River watershed during 2003-2021. Panel A shows the relationship between the EFPR treatment and EFPR control areas, whereas Panel B shows the relationship between the EFPR treatment and WFPR control areas. Open circles are pretreatment years and shaded triangles are treatment years. Dashed lines indicate mean ratio values for each time period.

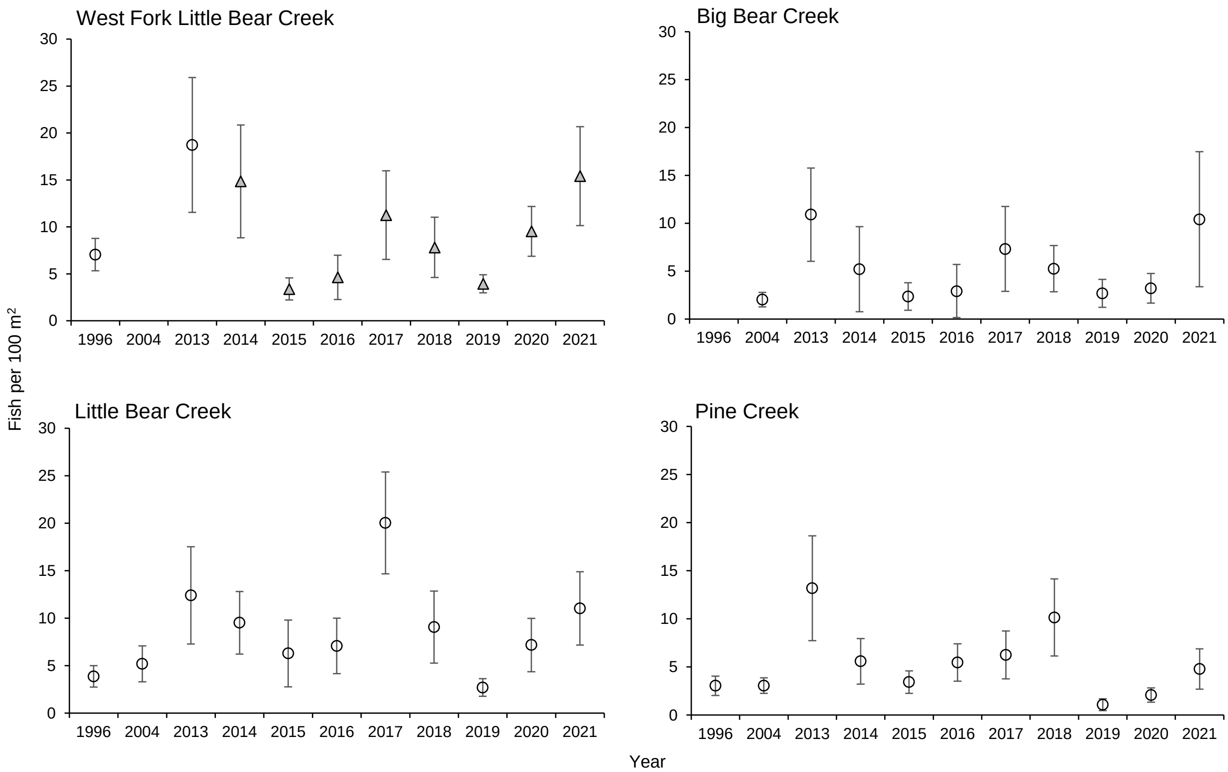

Figure 2.22. Density of juvenile steelhead ≥80 mm based on single-pass electrofishing surveys in Big Bear Creek, Little Bear Creek, West Fork Little Bear Creek (treatment tributaries) and Pine Creek (control tributary) in the lower Potlatch River watershed during 1996-2021. Open circles indicate pretreatment years and filled triangles indicate treatment years. Error bars are standard error.

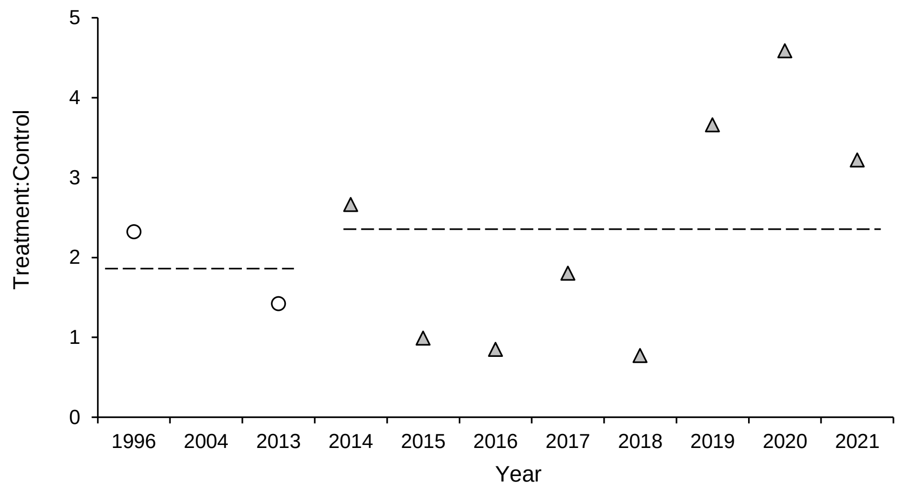

Figure 2.23. Juvenile steelhead density ratio values (treatment:control) for the West Fork Little Bear Creek (treatment tributary) and Pine Creek (control tributary) in the lower Potlatch River watershed during 1996-2021. Open circles are pretreatment years and shaded triangles are treatment years. Dashed lines indicate mean ratio values for each time period.

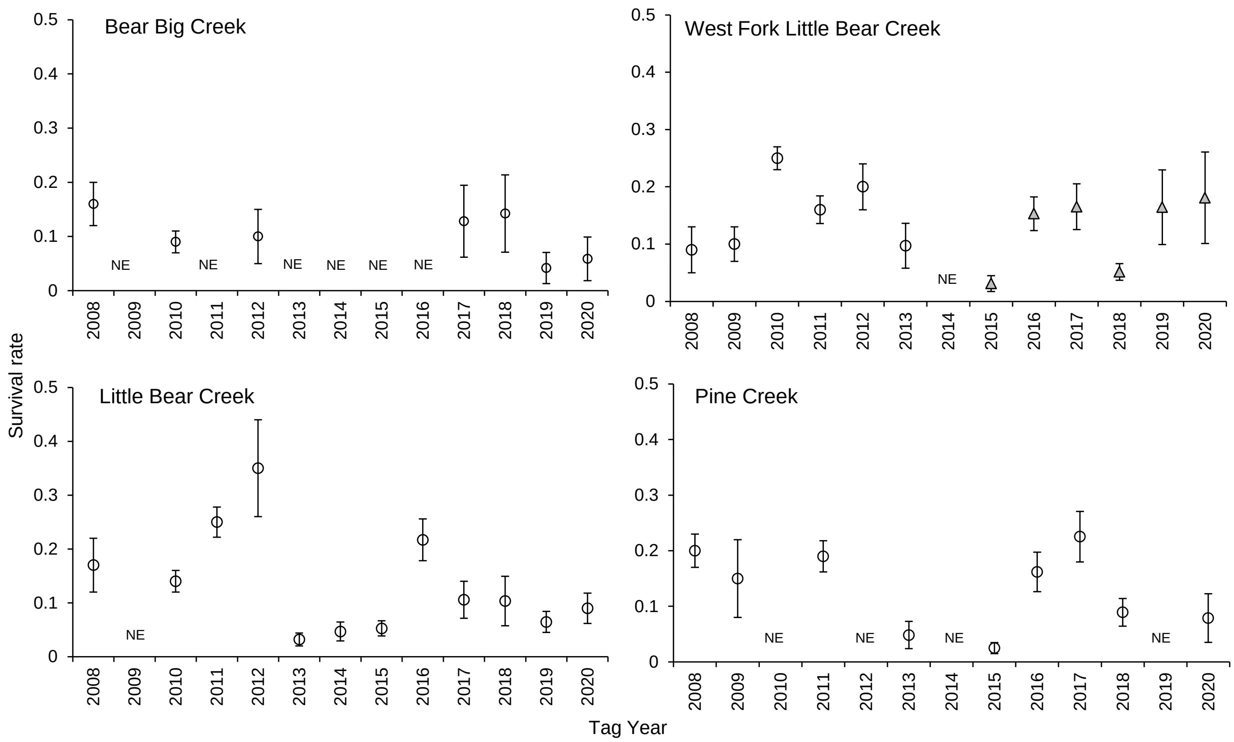

Figure 2.24. Juvenile steelhead apparent survival estimates to Lower Granite Dam from Big Bear Creek, Little Bear Creek, and the West Fork Little Bear Creek (treatment tributaries) and Pine Creek (control tributary) in the lower Potlatch River watershed during tag years 2008-2020. No estimate (NE) indicates insufficient detections to generate an estimate.

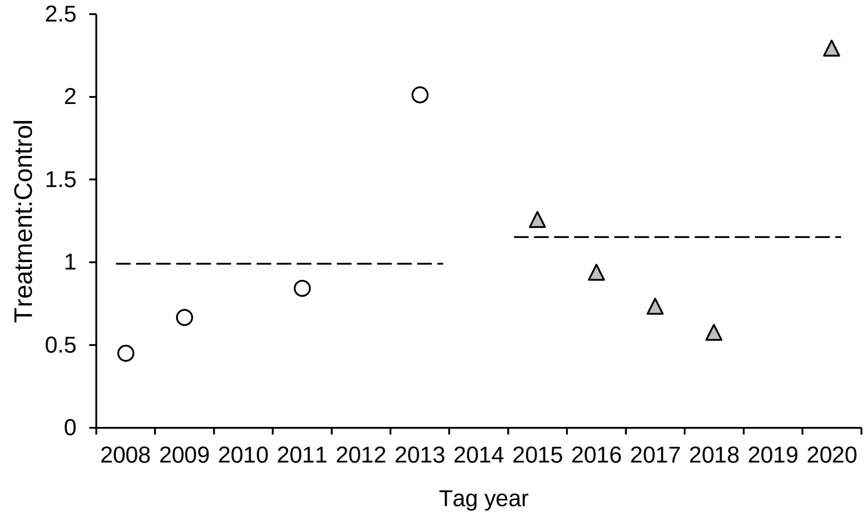

Figure 2.25. Juvenile steelhead apparent survival ratio values (treatment:control) for the West Fork Little Bear Creek (treatment tributary) and Pine Creek (control tributary) in the lower Potlatch River watershed during 2008-2021. Open circles are pretreatment years and shaded triangles are treatment years. Dashed lines indicate mean ratio values for each time period.

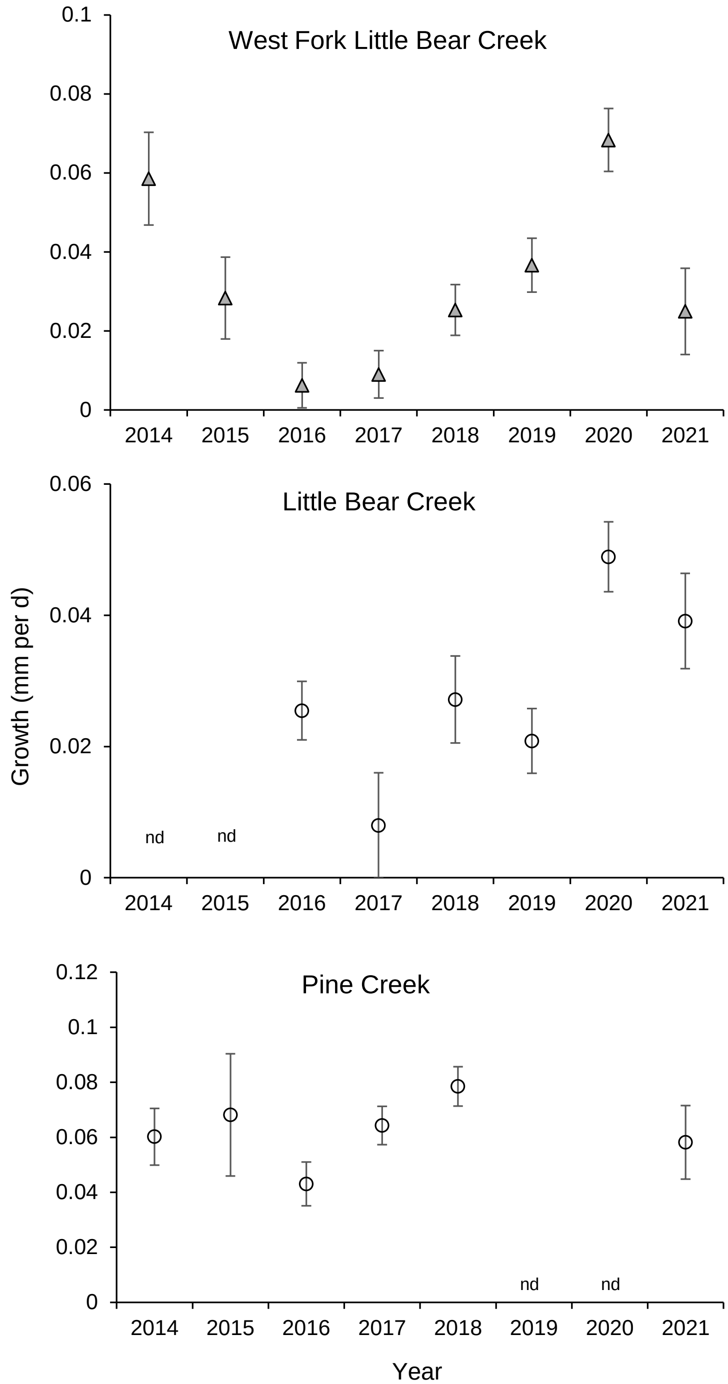

Figure 2.26. Summer-fall growth rates (mm per d) of juvenile steelhead (≥80 mm) in select treatment (West Fork Little Bear Creek and Little Bear Creek) and control (Pine Creek) tributaries in the lower watershed of the Potlatch River, Idaho from 2014-2021.

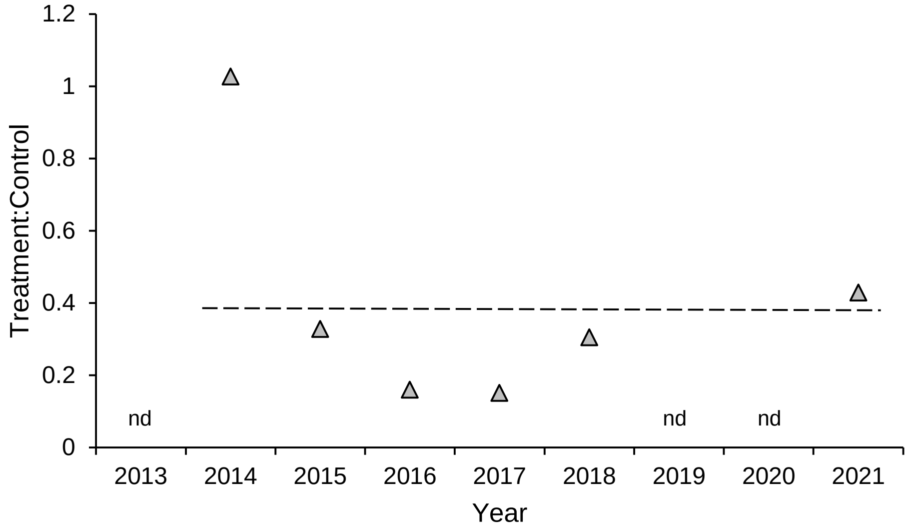

Figure 2.27. Juvenile steelhead growth ratio values (treatment:control) for the West Fork Little Bear Creek (treatment tributary) and Pine Creek (control tributary) in the lower Potlatch River watershed during 2008-2021. Shaded triangles indicate treatment years. Dashed line indicates mean ratio value for the time period.

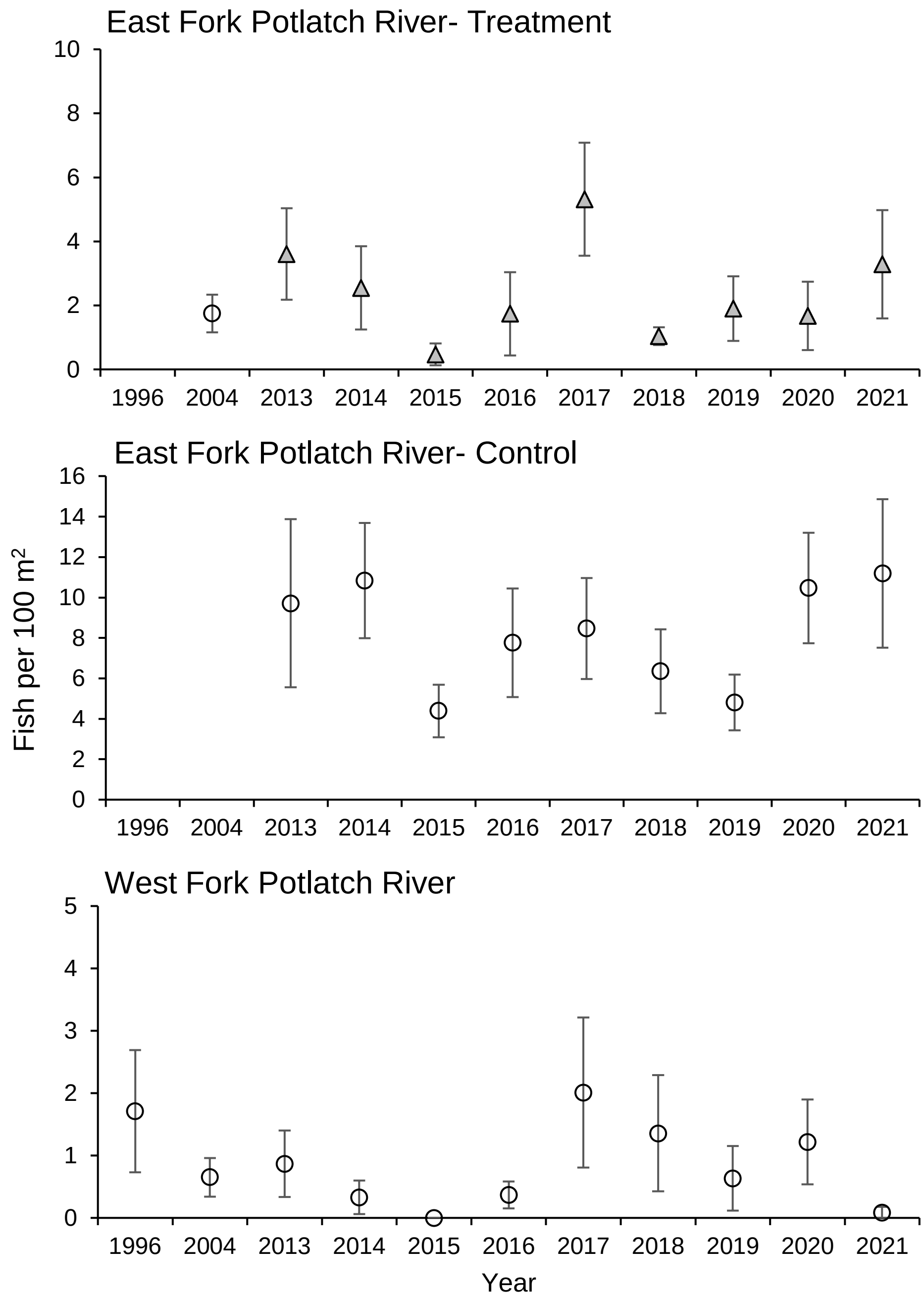

Figure 2.28. Density of juvenile steelhead ≥80 mm based on single-pass electrofishing surveys in the East Fork Potlatch River treatment area, the East Fork Potlatch River control area, and the West Fork Potlatch River control area in the upper Potlatch River watershed during 1996–2021. Open circles indicate pretreatment years and filled triangles indicate treatment years. Error bars are standard error.

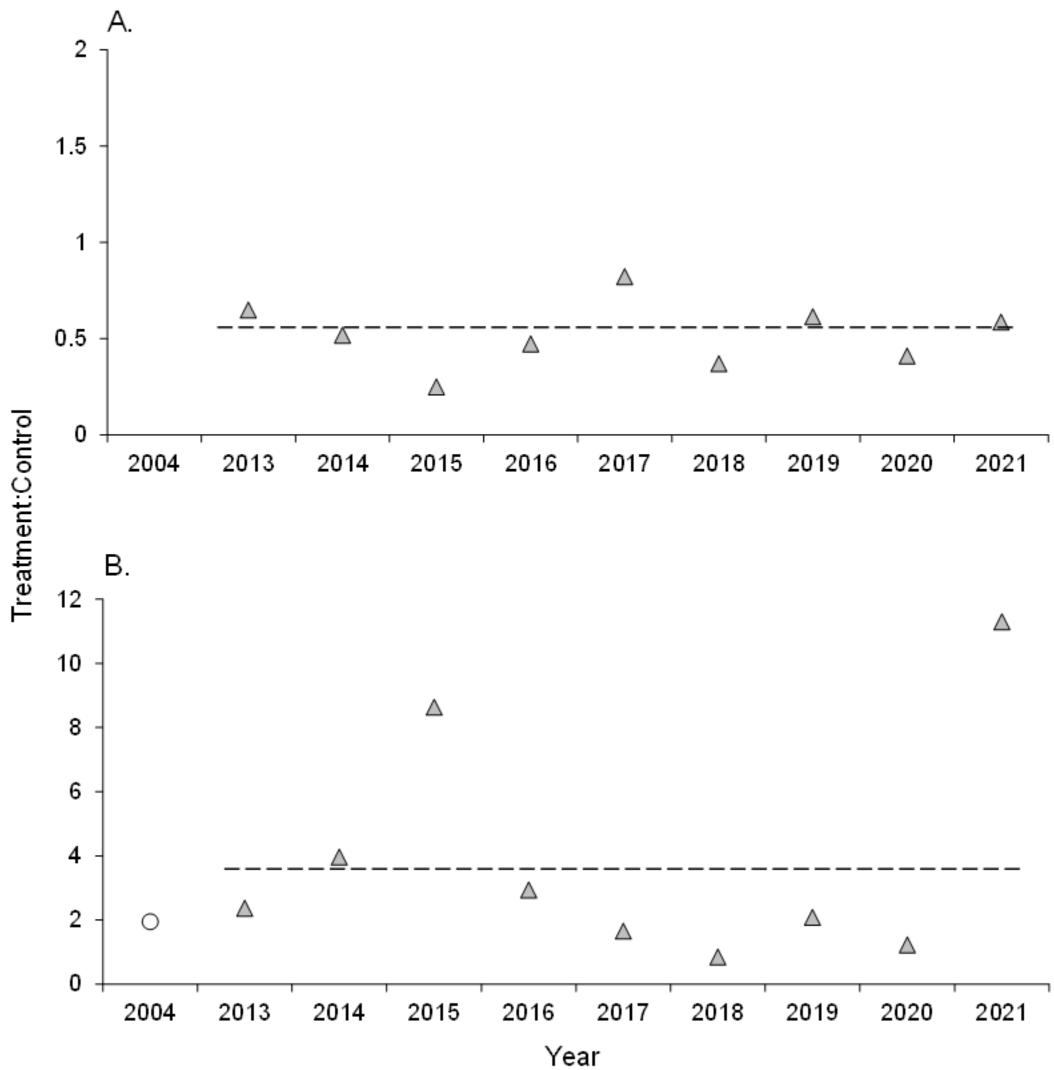

Figure 2.29. Juvenile steelhead density ratio values (treatment:control) for the EFPR Treatment and EFPR Control areas (upper panel) and EFPR Treatment and WFPR Control areas (lower panel) in the upper Potlatch River watershed from 2004-2021. Open circles are pretreatment years and shaded triangles are treatment years. Dashed lines indicate mean ratio values for each time period.

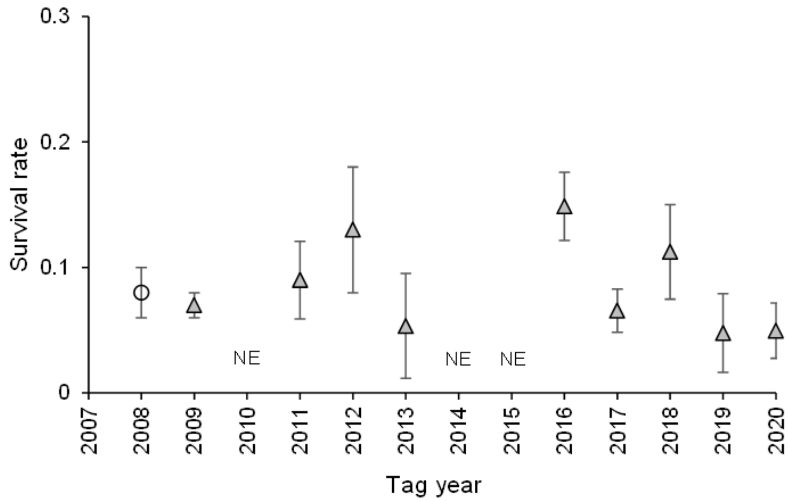

Figure 2.30. Juvenile steelhead apparent survival estimates to Lower Granite Dam from the East Fork Potlatch River during tag years 2008-2020. No estimate (NE) indicates insufficient detections to generate an estimate.

REPORT SYNTHESIS

The intensive monitoring projects in the Lemhi River and Potlatch River watersheds have now been operating for more than 15 years. The restoration strategies and specifics of the monitoring are different between the watersheds, but many experiences are similar. In this synthesis, we emphasize the key features of the two IMWs. There have been several important events that have happened in the past five years that have influenced the trends in the focal fish populations that have implications for the future. Commonalities in the lessons learned by these projects will help guide restoration within the project areas and elsewhere in Idaho. We discuss the overall trajectories of restoration and population status, lay out considerations for the short- and long-term future, and finish with concluding thoughts on adaptive management of habitat restoration.

Restoration & Population Status

In the last five years, we have seen two trends affecting stream restoration in our study areas: emphasis on larger-scale restoration projects and declining anadromous adult returns. Both IMWs have seen some successes but those are largely compromised by the latter trend. Reconnections of tributary and off-channel habitats have yielded benefits. In the Lemhi watershed, we have seen more fluvial bull trout movement and an increase in productivity by Chinook Salmon that stay in their natal reach (i.e., the upper Lemhi reach) until they become a smolt. In Big Bear Creek, distribution of steelhead spawning has increased but population-level response has not yet been seen. Recently, there is greater emphasis on restoration of off-channel habitats and floodplain reconnection in the Lemhi and East Fork Potlatch watersheds to provide better winter shelter. Floodplain reconnections could also benefit parr in the summer (Bond et al. 2019; Fogel et al. 2022). These efforts have been informed by monitoring and evaluation efforts conducted by the IMWs, which helps tailor location and type of restoration projects. The Lemhi IMW has had several ambitious projects implemented or in progress. However, only one of three keystone projects in the Big Bear Creek watershed has been implemented to date. A major factor affecting both IMWs was the decline in adult anadromous returns throughout the last 5 years.

The current state of the art in stream habitat restoration can be summed by the phrase, “Go big or go home.” This attitude affects restoration direction and project size. In many cases, the easy projects were already done; therefore, larger and more complex projects are needed to address watershed-level processes and yield demonstrable, significant benefits (NMFS 2020). There has been an evolution of LWD treatments in the EFPR watershed toward more intense treatments, larger wood, and more pieces interacting with the stream. However, there are significant difficulties getting big projects done (Big Bear Creek). Further, large projects take a long time to implement and complete (Lemhi). Some restoration projects address off-channel habitats and floodplain reconnection, which can be complicated to monitor for effects on fish performance. To address these difficulties, there is increasing reliance on and incorporation of novel IPTDS configurations.

The success of habitat restoration to benefit anadromous salmonids is affected by out-of-basin factors. Habitat restoration should be viewed as part of full life-cycle management; something that adds resilience to the focal populations. However, salmon and steelhead populations need spawning adults to achieve full benefits from restored and improved habitats. Recent trends in adult abundance across the Snake and Columbia river basins are declining due to poor downstream and ocean survival (Ford 2022; NOAA 2022). Improved habitat needs young recruits for benefits to be achieved; thus, out-of-basin effects compromise the goals of restoration programs and the ability of monitoring to detect real effects.

For example, consider a case in which restoration increases intrinsic productivity by 25% versus the current state. We use the recent Beverton Holt parameters based on abundance of Chinook Salmon emigrants past lower Lemhi River trap (Heller et al. 2022; Figure S1). The width of the 95% CI about the BY2019 abundance in the data set was approximately 10,500 emigrants at the trap. Assume this precision applies to the stock-recruit function, thus giving a minimum detectable difference between production regimes. In this simple example, number of redds would need to exceed 265 before a statistically significant difference between production regimes could be detected. If we consider the difference between the curves to be the benefit of restoration, then one can see how the benefit is rapidly reduced as redds decline below 265 and is increased as redds exceed that threshold. In reality, the redd threshold needed for detection of differences between pre- and post-treatment smolt production will be greater than this simple example shows because of annual variability about the trend.

Considerations for the Future

Both IMWs should consider the means for formal analysis and evaluation as they are well into the implementation stage. A formalized evaluation framework and model strengthens the ability to adaptively manage. Caution should be employed when using, or interpreting results from, a BACI design, although a properly executed BACI is still one of the best means for detecting effects in a variable environment (Smokorowski and Randall 2017). Our experience has been decidedly mixed and points to several lessons. In a BACI design, power is related to the number of years of sampling to characterize variation within each period, as well as the effect size to be detected. The need to control temporal variability to increase statistical power is more important than synchrony between restoration and control sites (Rogers et al. 2022). Another problem we encountered was a lack of independence among sites with respect to the metric being tested (e.g., source of juvenile salmon in the Lemhi River watershed). The processes affecting the variables measured need to be carefully considered when selecting sites and when interpreting the results. Finally, there was a decreasing trend in adult spawners as treatments were being implemented, essentially reducing the ability of the fish populations to respond. Some of these issues can be addressed by a model-driven framework for internal analysis paired with external comparisons. Spatial variability among sites within the watershed may present an analytical problem for analyzing changes in fish density that appropriate modeling can handle, whereas smolt production will not be affected by such variability (Rogers et al. 2022). The ultimate effects of restoration programs within the IMWs on fish populations can be detected in comparison to a range of other populations, i.e. an external comparison. Because any particular framework has its strengths and weaknesses, evaluation should take a weight-of-evidence approach with several lines of assessment based on explicit hypotheses (Diefenderfer et al. 2016).

There are two types of general hypotheses for IMWs to test: population-level effects and effects of specific actions (by type or by reach). Choice of metric is important to success of the study design (Rogers et al. 2022). The target population-level metrics for anadromous species are smolt abundance and productivity (smolts per female spawner), whereas for bull trout they are the resident standing stock and number of fluvial migrants. Productivity analyses are a measure of resilience and a focus on bull trout may help control the effects of out-of-basin losses and other sources of temporal noise. The core study design must be maintained for productivity metrics. The study design for other metrics can be more flexible and there may be opportunities to be more efficient. These may include parameters such as growth, movement, occupancy, and survival. For example, annual measurements might not be needed for some metrics if resources are limited. Each IMW should frame key metrics into hypotheses about 1) opening access to blocked habitats, and 2) improving performance in accessible habitats.

Climate change could be an additional stressor by changing the hydrograph and elevating summer stream temperatures. Years of drought will reduce yield from the snowpack to the spring freshet, lowering hydrograph peaks and extending periods of base flow. As flows drop, mean summer stream temperatures will increase if no actions are taken, stressing stream-dwelling salmonids. In particular, Bull Trout are dependent on cold stream temperatures and their distribution is predicted to shrink as the climate warms (Falke et al. 2015; Isaak et al. 2022). Idaho's spring/summer Chinook Salmon and steelhead populations are likely to be vulnerable to changes in ocean conditions and to temperature increases in their mainstem migratory corridors as immigrating adults (Crozier et al. 2019, 2020, 2021), exacerbating out-of-basin effects. However, effective restoration can ameliorate climate effects on rearing habitats. Riparian restoration could increase parr abundance and distribution to offset climate changes (Justice et al. 2017; White et al. 2017). To restore resiliency to stream habitats there is greater emphasis on habitat-forming processes (Beechie et al. 2010). One example is valley restoration by re-activation of floodplain and allowing anastomosing stream channels to re-establish (e.g., the Lower Lemhi River Rehabilitation project; see Powers et al. 2019, Flitcroft et al. 2022). Another example is to encourage beaver colonization or to mimic them (Nash et al. 2021; Dittbrenner et al. 2022; Roper 2022). The aim of these strategies is to restore moderating processes, which will increase system resilience (Skidmore and Wheaton 2022) and thus resilience of the target populations (Falke et al. 2015; Crozier et al. 2021; Fogel et al. 2022).

The IDFG has adult abundance goals for salmon and steelhead populations (IDFG 2019). Habitat restoration is one management tool to achieve these goals. These goals were based on goals from Columbia Basin Partnership (NMFS 2020) in low, medium, high ranges for each population. Low corresponds with the minimum abundance threshold for a viable population. The high range corresponds with IDFG goals, signifying healthy populations that can sustain desired levels of harvest while adequately seeding the available habitat. Medium is about halfway between. These goals can be cross-referenced to juvenile production to provide benchmarks for Idaho's IMWs.

How many smolts are needed to achieve IDFG management goals? The answer largely depends on smolt-to-adult return (SAR) rate, which is the rate at which adults are produced from smolts. The goal for the Columbia Basin Fish and Wildlife Program is an average 4% SAR with a range of 2%-6% (NPCC 2020). The geometric means of SARs for Snake River spring/summer Chinook Salmon and steelhead since the 1990s have been about 1% (McCann et al. 2022). We used three SARs (1%, 2%, and 4%) to convert goal ranges to smolt production by the study populations at Lower Granite Dam. We make three further assumptions to cover the complete life cycle. First, we assume an adult conversion from Lower Granite Dam to the tributary, based on the Columbia Basin Partnership's schedule, by species. We further assume that tributaries between the Lemhi and North Fork Salmon rivers (e.g., Carmen Creek) and mainstem in that reach do not produce meaningful numbers of steelhead (i.e., only production from the Lemhi River itself counts). Lastly, we prorate goals for the Potlatch watershed from the Lower Clearwater population by the weighted intrinsic potential habitat area (24% of currently accessible habitat in the population is in the Potlatch watershed). These calculations generate reasonable benchmarks for restoration programs to achieve and the IMW programs to detect. Results of these calculations and assumptions are in Table S1.

Smolt production needed to achieve adult goals depends in large part on SAR levels. In the Potlatch River watershed, most of the best habitat is in the mainstem upstream of Big Bear Creek and downstream of EFPR. This observation points to a need to understand what is going on in that reach for the Potlatch IMW. As reported by McCann et al. (2022, see page 246), the

Chinook Salmon population in the Lemhi River needs an SAR of 4.7% to attain the low goal. Using the analysis in Figure S1, the current Lemhi population could achieve 64,000 smolts at 705 redds or about 395 redds if productivity is increased by 25%; the latter redd count was exceeded in 2001. Bond et al. (2019) estimated side channel restoration in the Lemhi watershed could increase parr capacity by 81%; applying that increase to Heller et al.'s (2022) asymptotic smolt production gives 199,357 smolts. These two examples show that restoration could help achieve NMFS (2020) goals but not IDFG (2019) goals, unless out-of-basin issues are also ameliorated.

Bull Trout are a focal species for restoration because they were federally listed as a threatened species under the Endangered Species Act in 1998 by the U.S. Fish and Wildlife Service. Similar to other native trout species in Idaho, IDFG has expended considerable effort to promote the conservation of Bull Trout. Recreational angling for Bull Trout has been managed under a no-harvest regulation since 1994 (IDFG 2019). IDFG advocates for de-listing those portions of the Bull Trout range where populations are secure and no longer in need of ESA protection. Bull Trout currently do not have numerical management goals for abundance needed to achieve recovery; however, threats-based goals for recovery are contained within the recovery plan (USFWS 2015). Although there are no demographic benchmarks, the expectations are distribution across representative habitats and demographically stable populations; conservation of genetic and life history diversity; and protection and connection of essential coldwater habitats. The implications for restoration are: 1) to take actions to protect, restore, and maintain suitable habitat conditions; 2) to minimize demographic threats to Bull Trout by restoring connectivity; and 3) to prevent and reduce negative effects of nonnative fishes. Current Bull Trout spawning and rearing reaches in Idaho tend to be in higher elevations on Federal lands. For IDFG's Fish Habitat Program, which is focused on private lands in lower elevations, actions to benefit Bull Trout should maintain or restore connectivity between main stem migratory and upstream spawning and rearing reaches, which implies a water temperature range that Bull Trout tolerate. Monitoring should demonstrate that these benefits are achieved.

Adaptive Management and Conclusion

The IMWs provide value to IDFG's fisheries managers by producing the information needed to adaptively manage habitat restoration. The information generated by the IMWs increasingly has been used to locate and design restoration projects (e.g., lower Lemhi and Henry reaches). It takes time for monitoring to help build the foundation for adaptive management and for that relationship to emerge and mature (e.g., Littles et al. 2022). The adaptive management of restoration in the Lemhi and Potlatch IMWs shows that there is a properly functioning relationship between the monitoring and the restoration programs. Both IMWs are working on ideas and techniques that may prove to be helpful in the next five years, such as investigating concerns around the use of beaver dam analogs and testing novel uses and configurations of IPTDS antennae. These products will be useful across IDFG's Habitat program. Improvements in adaptive management for habitat restoration could be made by keeping the following three best practices in mind: 1) a clear purpose for restoration design, 2) hypotheses for expected effects, and 3) metrics linked to near-term benchmarks and expected long-term outcome (Oakes et al. 2022).

Across the Pacific Northwest, IMWs often have similar experiences (summarized by Bilby et al. 2022). Many have found that habitat usually responds to restoration, but fish responses are detected less frequently because there are many complexities that mitigate against easy detection of fish responses. However, the fish responses most frequently detected are related to remediation of passage barriers and to enhancement of floodplain access. The responses tend to be in the juvenile stages because of out-of-basin effects on adults. Therefore, accounting for out-of-basin effects is important to set realistic expectations for restoration and monitoring. A key message is that high quality habitats will enhance resilience to severe disturbance or climate change; therefore, in the face of bad years, look for resilience, not abundance. Lastly, the time required for evaluation is affected by restoration pace and variable fish response times, especially in light of out-of-basin effects. These lessons are important for setting expectations and goals for Idaho IMWs in the next five years and for IDFG's Fish Habitat Program in general.

Table 2.6. Numbers of smolts at Lower Granite Dam needed to produce adults that achieve low and high goal levels at three smolt-to-adult return (SAR) levels.

	Low goals SAR			High goals SAR		
Population/Species	**1%**	**2%**	**4%**	**1%**	**2%**	**4%**
Lemhi						
Chinook Salmon	256,716	128,358	64,179	1,178,683	589,342	294,671
Steelhead	136,986	68,493	34,247	657,895	328,947	164,474
Potlatch						
Steelhead	49,726	24,863	12,432	239,079	119,539	59,770

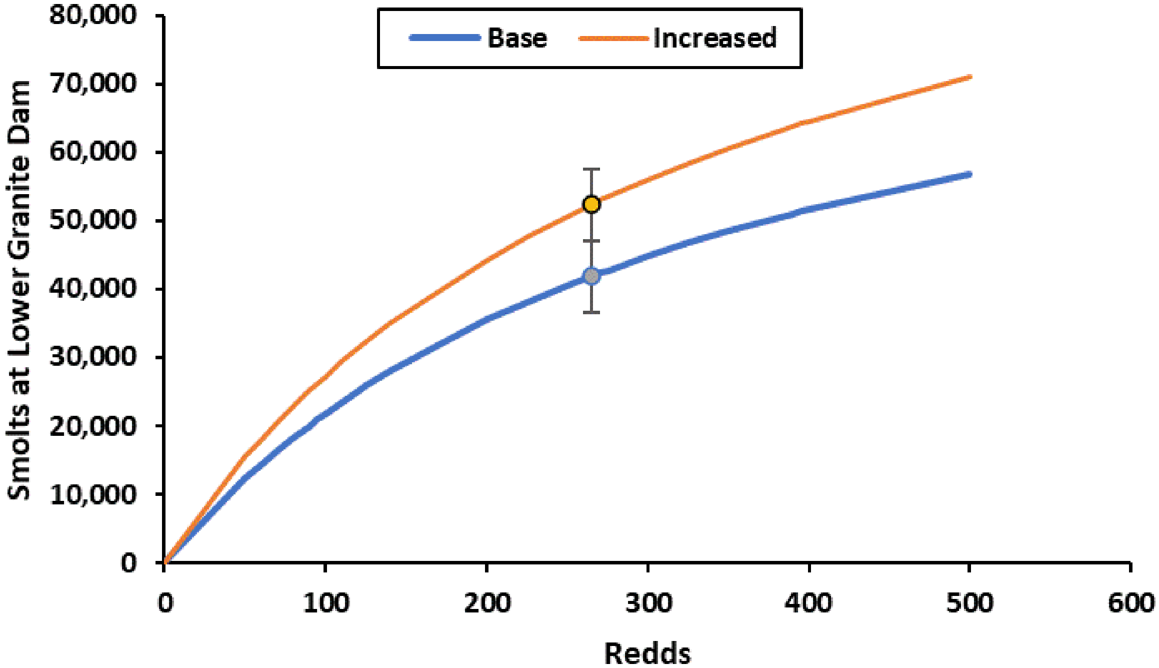

Figure 2.31. Predicted number of Chinook Salmon smolts produced per number of redds under current and increased productivity. The base model is a Beverton-Holt curve with parameters α=283.7 and β=0.003 (Heller et al. 2022). In the increased model, α was increased by 25%. Points show predicted smolts at 265 redds with 95% confidence intervals.

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APPENDICES

Appendix A. Electrofishing sampling effort in Lemhi River tributaries.

Table A1. Sampling effort of electrofishing surveys in tributaries of the Lemhi River. The standing stock estimation length represents the total stream length over which the standing stock estimates were calculated.

Year	Sampling method	Cumulative length sampled (km)	Abundance estimation length (km)	Proportion sampled (%)
Big Timber Creek				
2009	Depletion	1.06	23.28	4.55
2010	Depletion	0.51	23.28	2.21
2011	Depletion	1.17	23.28	5.04
2012	Depletion	0.33	23.28	1.43
2013	Mark Recapture	12.93	23.28	55.54
2014	Mark Recapture	6.87	23.28	29.51
2015	Mark Recapture	8.64	23.28	37.12
2016	Mark Recapture	7.99	23.28	34.32
2017	Mark Recapture	8.42	23.28	36.17
2018	Mark Recapture	8.01	23.28	34.41
2019	Mark Recapture	10.86	23.28	46.65
2020	Mark Recapture	2.60	23.28	11.17
2021	Mark Recapture	4.62	23.28	19.85
Bohannon Creek				
2010	Depletion	0.15	14.79	1.01
2011	Depletion	0.48	14.79	3.25
2012	Depletion	0.41	14.79	2.77
2013	Mark Recapture	7.82	14.79	52.87
2014	Mark Recapture	8.10	14.79	54.77
2015	Mark Recapture	13.25	14.79	89.59
2016	Mark Recapture	11.68	14.79	78.97
2017	Mark Recapture	6.99	14.79	47.26
2018	Mark Recapture	3.25	14.79	21.97
2019	Mark Recapture	11.69	14.79	79.04
2020[a]	Mark Recapture	--	--	--
2021	Mark Recapture	2.64	14.79	17.85
Canyon Creek				
2009	Depletion	0.45	19.82	2.27
2010	Depletion	0.49	19.82	2.47
2011	Depletion	0.61	19.82	3.08
2012	Depletion	0.42	19.82	2.12
2013	Mark Recapture	13.25	19.82	66.85
2014	Mark Recapture	8.81	19.82	44.45
2015	Mark Recapture	6.92	19.82	34.91

2016	Mark Recapture	9.84	19.82	49.65
2017	Mark Recapture	7.76	19.82	39.15
2018	Mark Recapture	8.31	19.82	41.93
2019	Mark Recapture	0.99	19.82	4.99
2020	Mark Recapture	0.81	19.82	4.09
2021	Mark Recapture	1.61	19.82	8.12
Hawley Creek				
2009	Depletion	0.6	26.21	2.29
2010	Depletion	0.45	26.21	1.72
2011	Depletion	0.67	26.21	2.56
2012	Depletion	0.26	26.21	0.98
2013	Mark Recapture	7.83	26.21	29.86
2014	Mark Recapture	8.06	26.21	30.73
2015	Mark Recapture	8.44	26.21	32.18
2016	Mark Recapture	10.75	26.21	41.01
2017	Mark Recapture	9.94	26.21	37.92
2018	Mark Recapture	9.06	26.21	34.57
2019	Mark Recapture	10.59	26.21	40.40
2020[a]	Mark Recapture	--	--	--
2021	Mark Recapture	--	--	--
Hayden Creek				
2009	Depletion	0.75	18.76	4.01
2010[a]	--	--	--	--
2011	Mark Recapture	1	18.76	5.34
2012	Depletion	0.59	18.76	3.17
2013	Mark Recapture	9.82	18.76	52.34
2014	Mark Recapture	10.6	18.76	56.5
2015	Mark Recapture	13.3	18.76	70.87
2016	Mark Recapture	8.71	18.76	46.41
2017	Mark Recapture	8.01	18.76	42.70
2018	Mark Recapture	8.02	18.76	42.75
2019	Mark Recapture	7.10	18.76	37.85
2020	Mark Recapture	6.71	18.76	35.77
2021	Mark Recapture	4.20	18.76	22.39
Kenney Creek				
2009	Depletion	0.15	8.65	1.73
2010	Depletion	0.27	8.65	3.16
2011	Depletion	0.46	8.65	5.32
2012	Depletion	0.66	8.65	7.63
2013	Mark Recapture	8.31	8.65	96.07
2014	Mark Recapture	5.73	8.65	66.24
2015	Mark Recapture	3.35	8.65	38.73
2016	Mark Recapture	7.18	8.65	83.01

2017	Mark Recapture	5.91	8.65	68.32
2018	Mark Recapture	6.37	8.65	73.64
2019	Mark Recapture	5.16	8.65	59.65
2020[a]	Mark Recapture	--	--	--
2021[a]	Mark Recapture	--	--	--
Little Springs Creek				
2009	Depletion	0.29	5.34	5.43
2010[a]	--	--	--	--
2011	Depletion	0.31	5.34	5.81
2012	Depletion	0.31	5.34	5.81
2013	Mark Recapture	5.31	5.34	99.44
2014	Mark Recapture	5.31	5.34	99.44
2015	Mark Recapture	4.91	5.34	91.95
2016	Mark Recapture	4.91	5.34	91.95
2017	Mark Recapture	5.24	5.34	98.13
2018	Mark Recapture	5.24	5.34	98.13
2019	Mark Recapture	5.24	5.34	98.13
2020	Mark Recapture	0.97	5.34	18.16
2021	Mark Recapture	5.34	5.34	100.00

[a] Not sampled

Appendix B. Focal species distribution in priority tributaries and Hayden Creek in the Lemhi River basin.

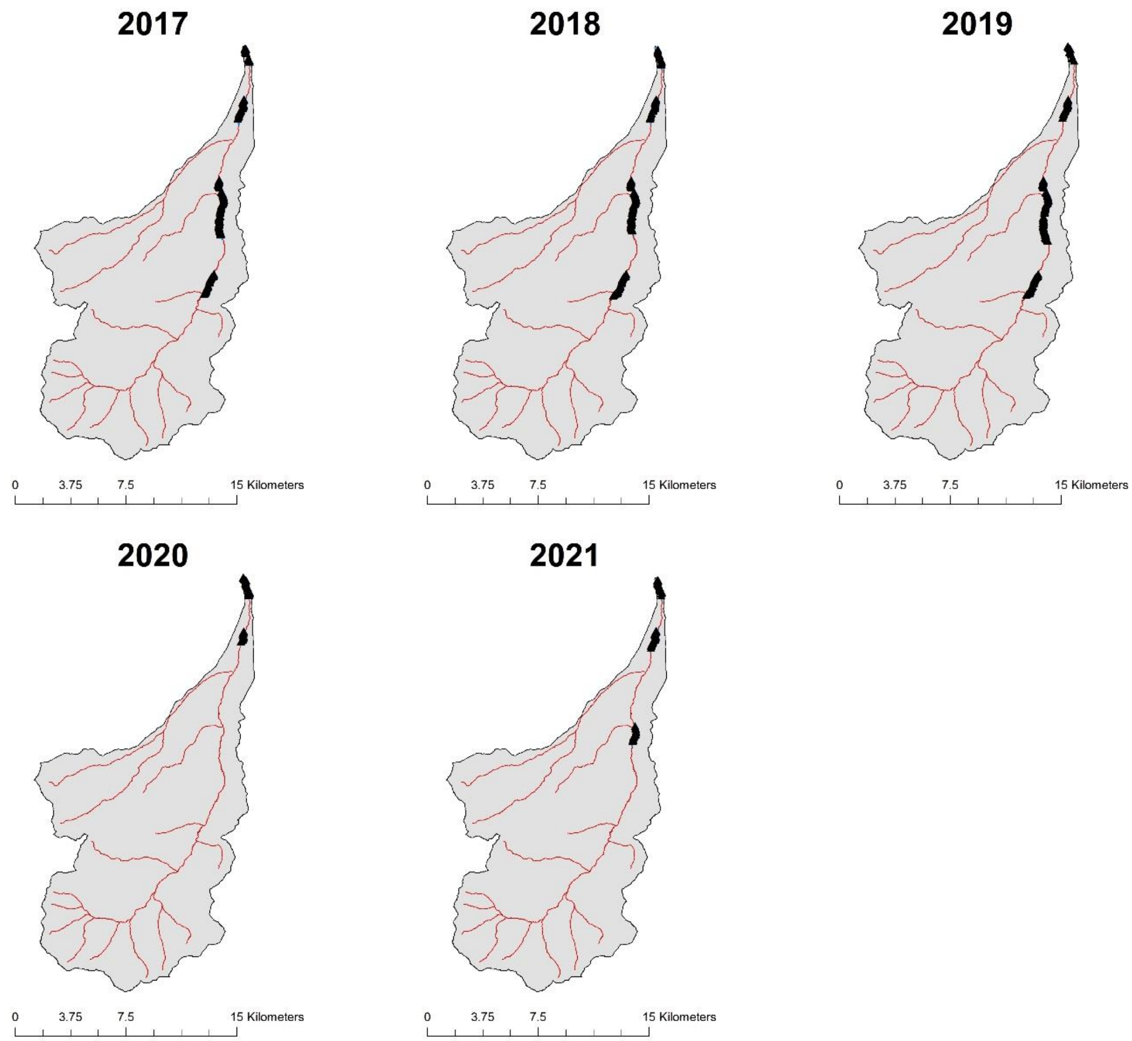

Figure B1. Distribution of steelhead encountered during annual summer electrofishing surveys in Big Timber Creek, 2017-2021.

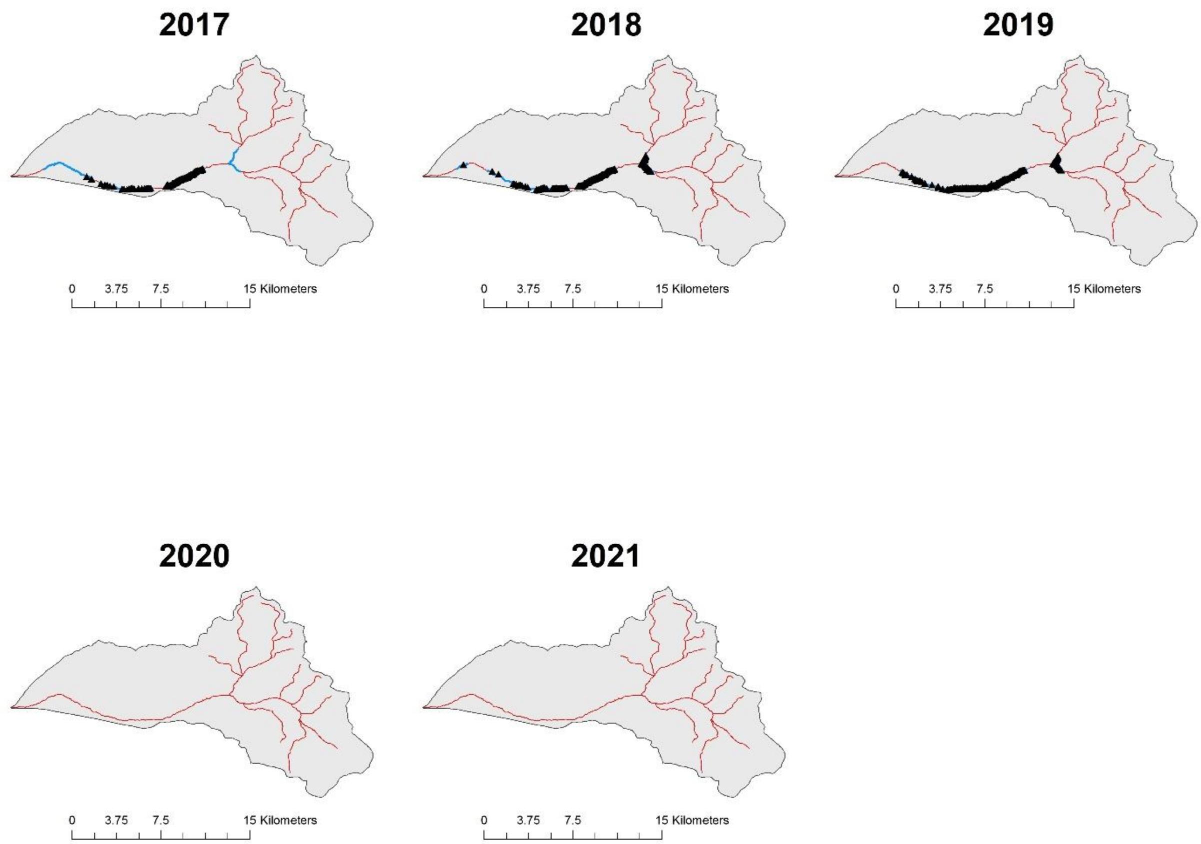

Figure B2. Distribution of steelhead encountered during annual summer electrofishing surveys in Hawley Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches. No sampling was conducted in 2020 and 2021.

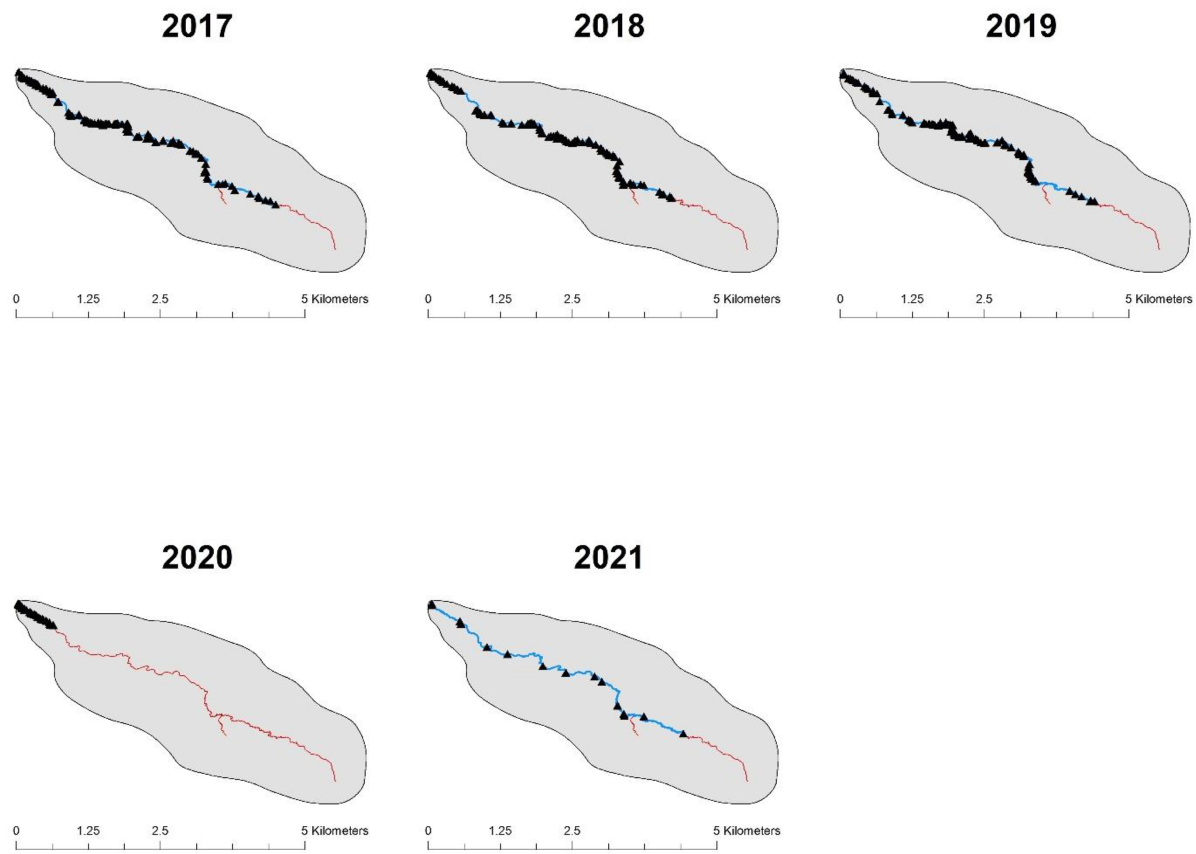

Figure B3. Distribution of steelhead encountered during annual summer electrofishing surveys in Little Springs Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches. Note, in 2021 waypoint locations were only taken at locations where fish were processed (numerous fish per point) not where individual fish were captured due to technical difficulties.

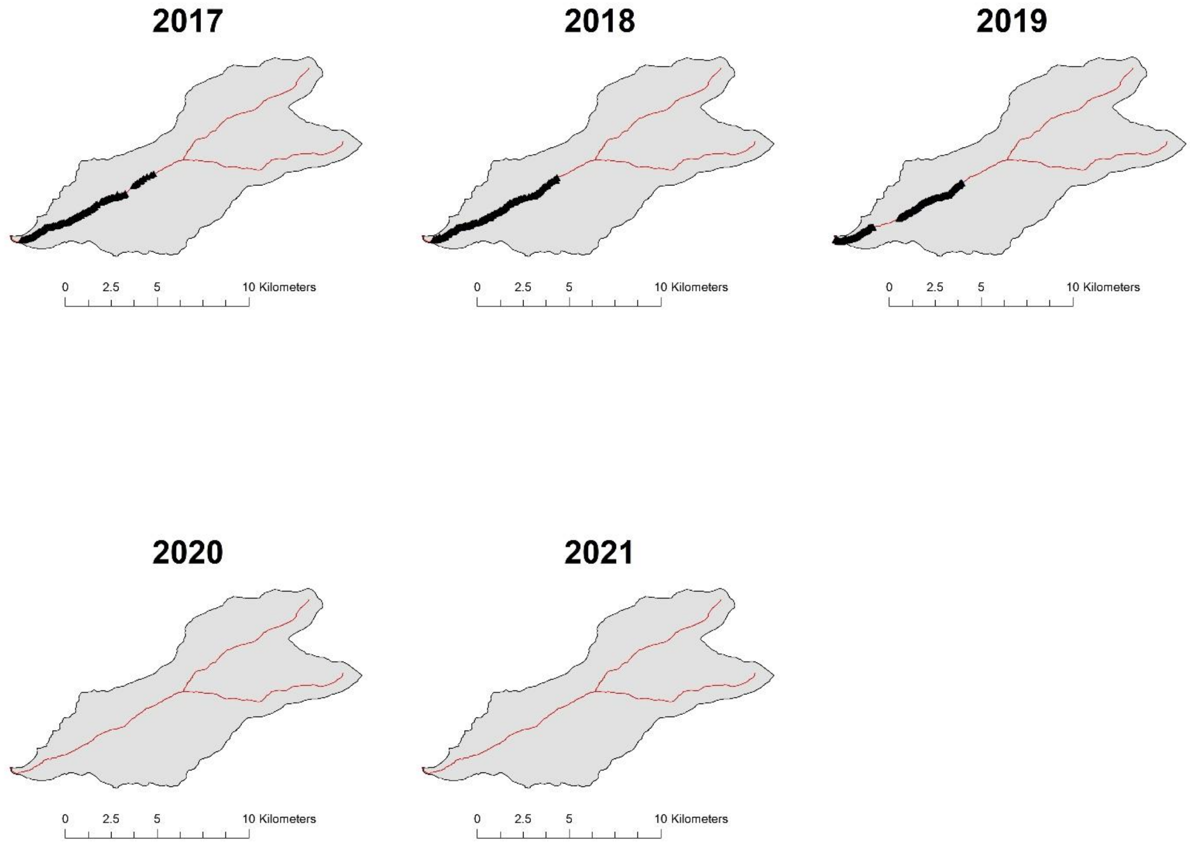

Figure B4. Distribution of steelhead encountered during annual summer electrofishing surveys in Kenney Creek, 2017-2021. No sampling was conducted in 2020 and 2021.

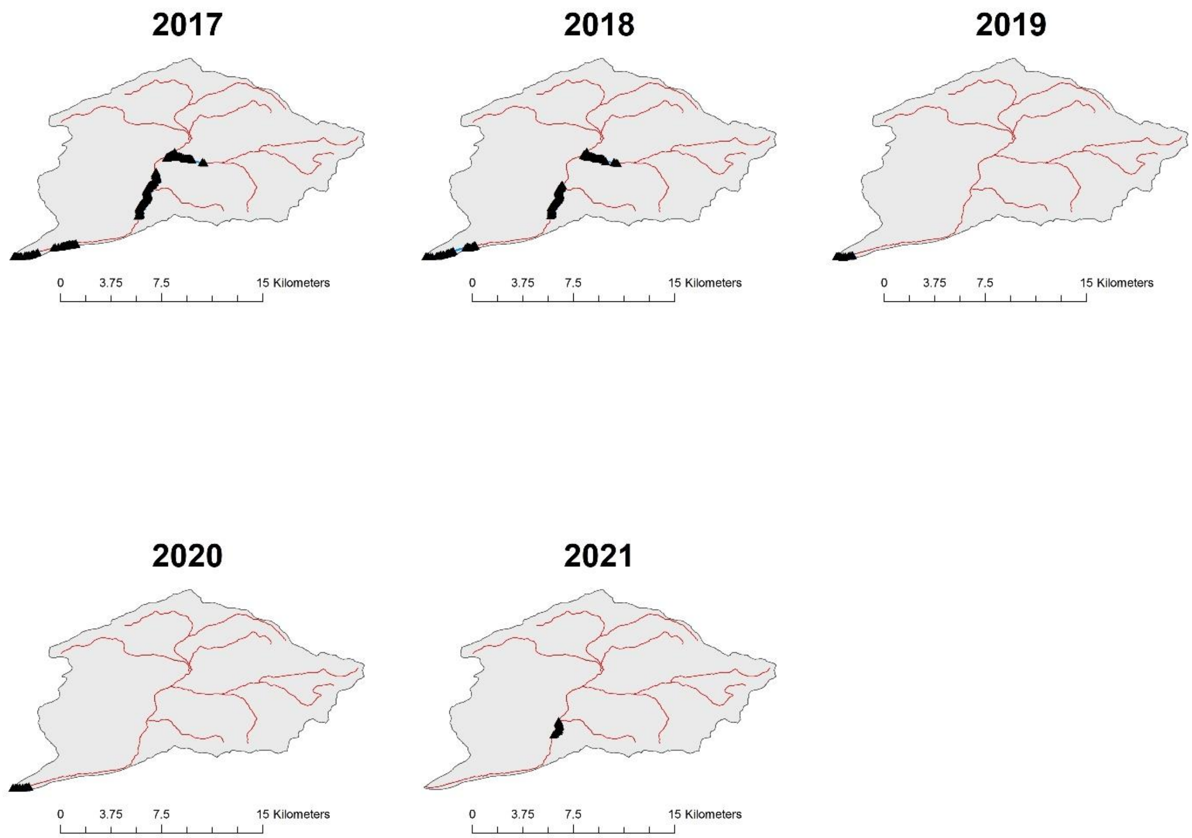

Figure B5. Distribution of steelhead encountered during annual summer electrofishing surveys in Canyon Creek, 2017-2021. Red lines represent non-sampled reaches.

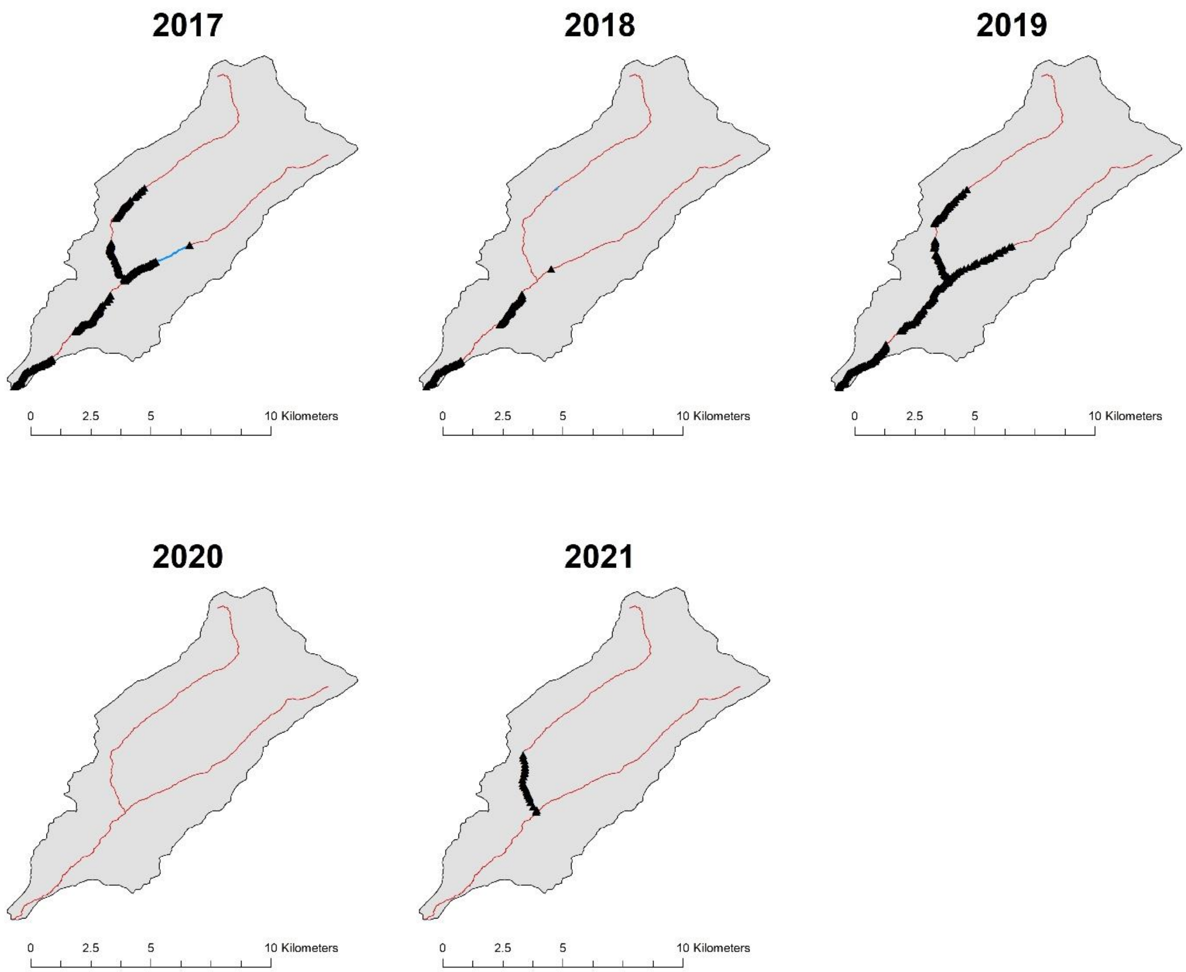

Figure B6. Distribution of steelhead encountered during annual summer electrofishing surveys in Bohannon Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches. No sampling was conducted in 2020.

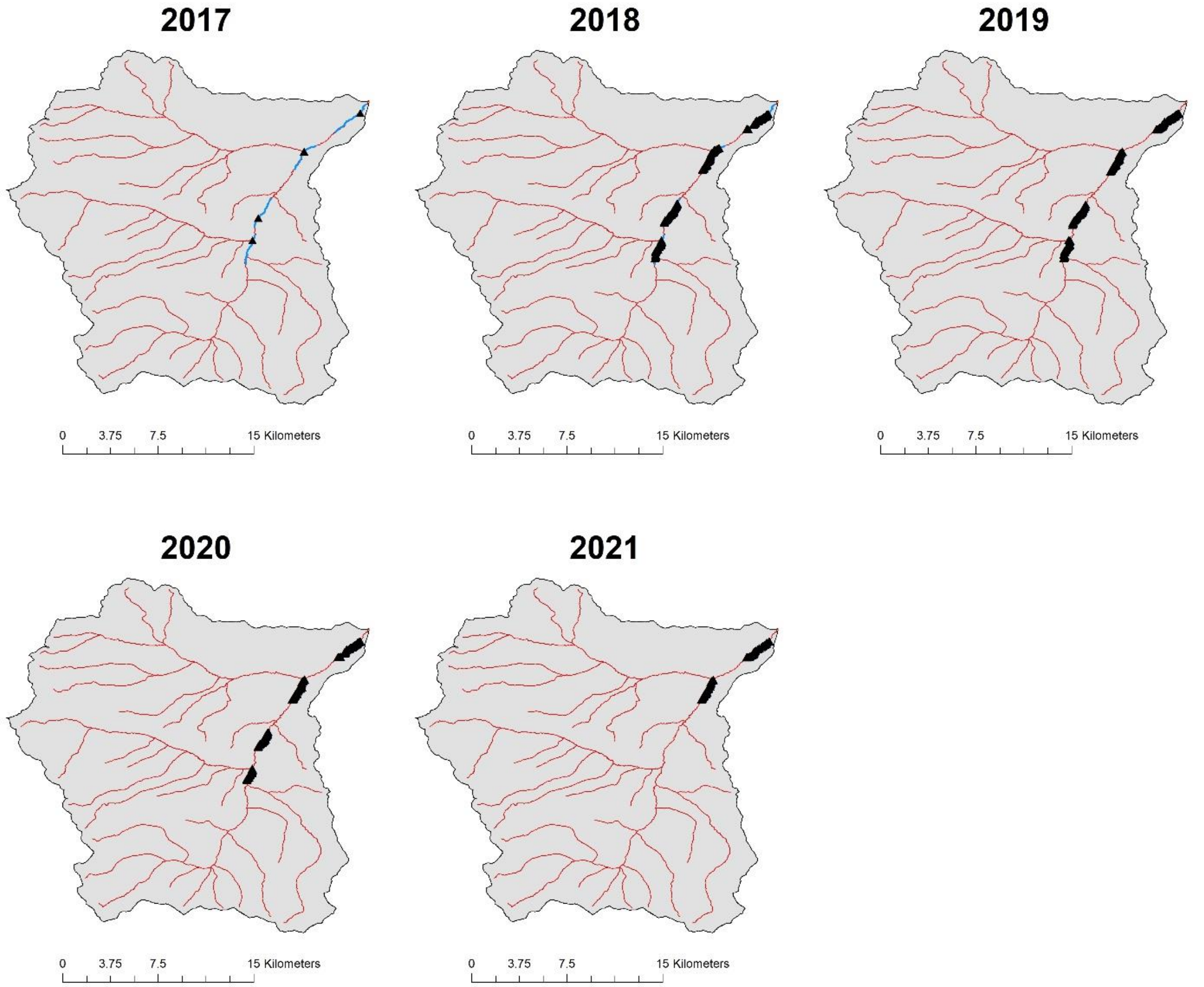

Figure B7. Distribution of steelhead encountered during annual summer electrofishing surveys in Hayden Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches. Note, in 2017 waypoint locations were only taken at locations where fish were processed (numerous fish per point) not where individual fish were captured due to technical difficulties.

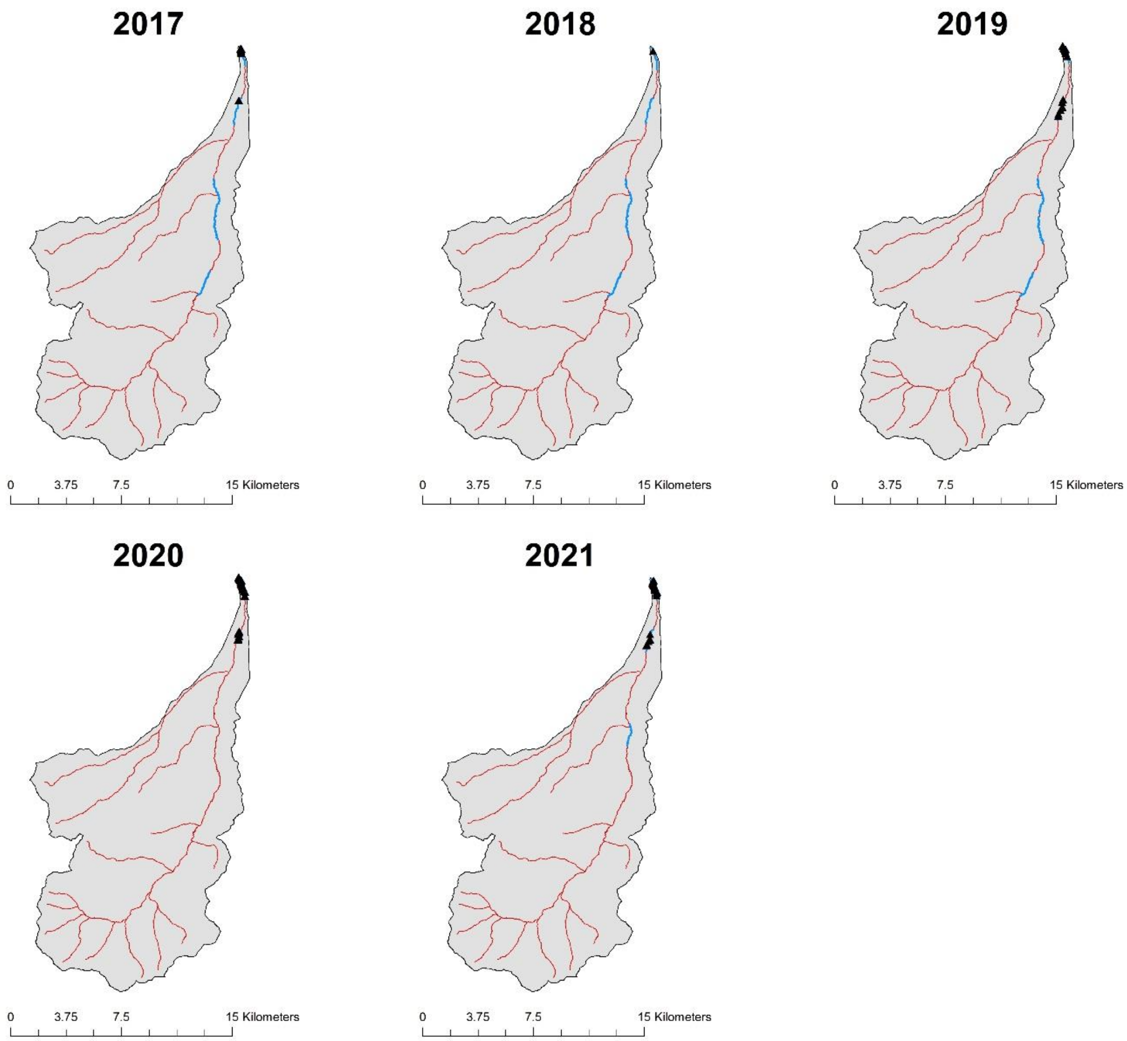

Figure B8. Distribution of juvenile Chinook Salmon encountered during annual summer electrofishing surveys in Big Timber Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

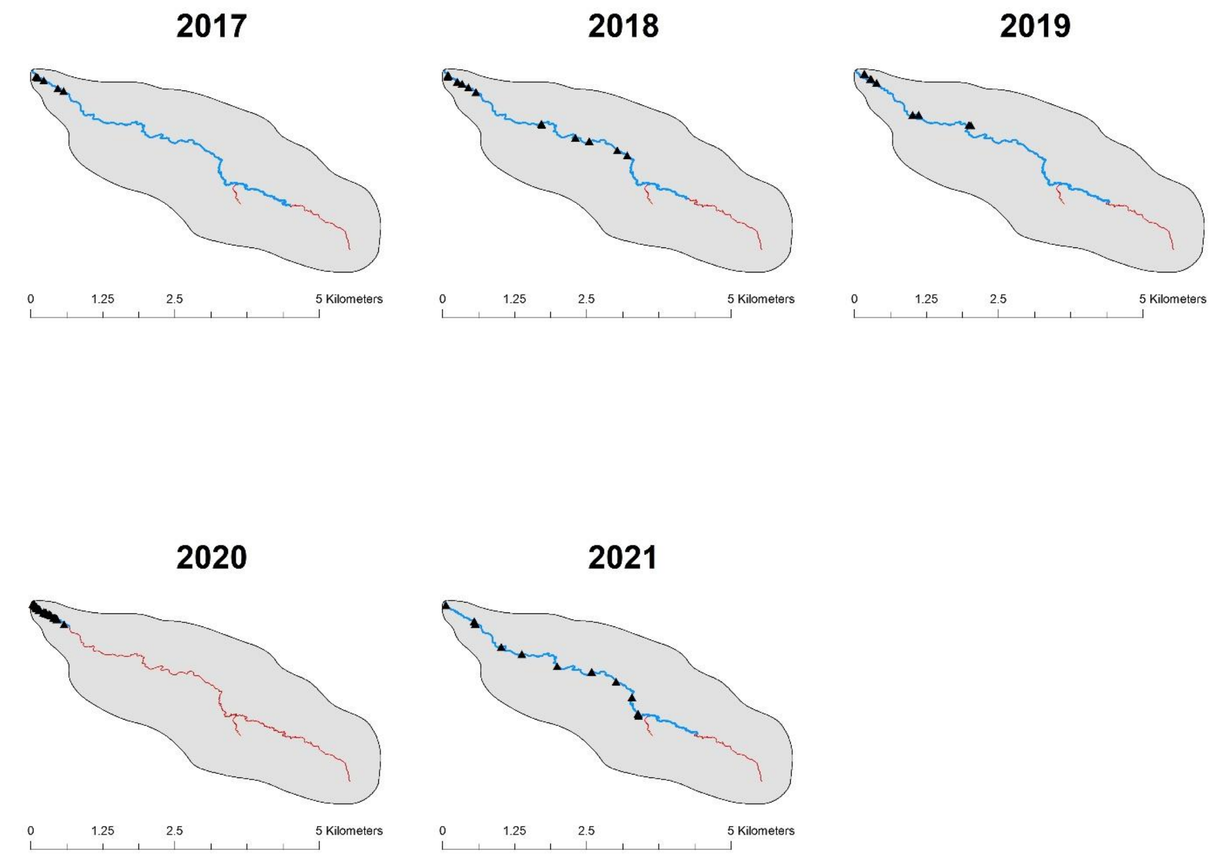

Figure B9. Distribution of juvenile Chinook Salmon encountered during annual summer electrofishing surveys in Little Springs Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches. Note, in 2021 waypoint locations were only taken at locations where fish were processed (numerous fish per point) not where individual fish were captured due to technical difficulties.

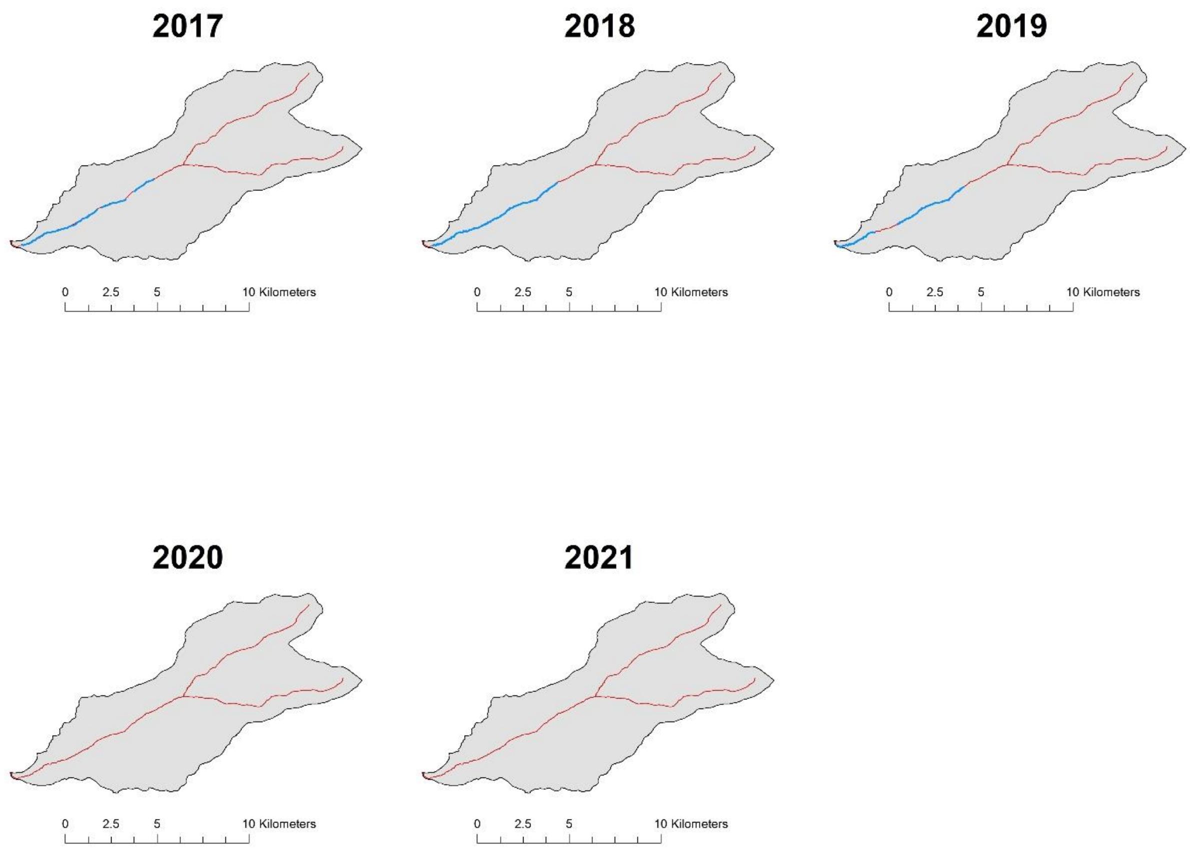

Figure B10. Distribution of juvenile Chinook Salmon encountered during annual summer electrofishing surveys in Kenney Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches. No sampling was conducted in 2020 and 2021.

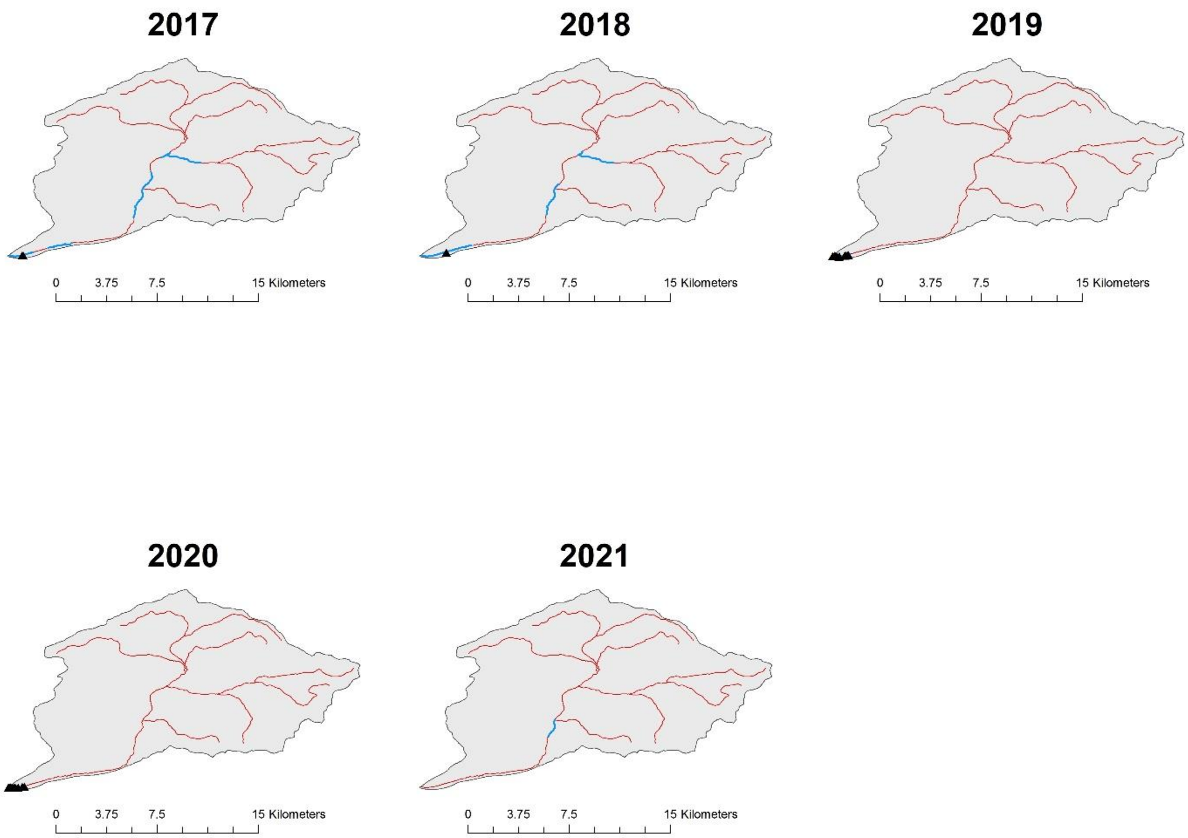

Figure B11. Distribution of juvenile Chinook Salmon encountered during annual summer electrofishing surveys in Canyon Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

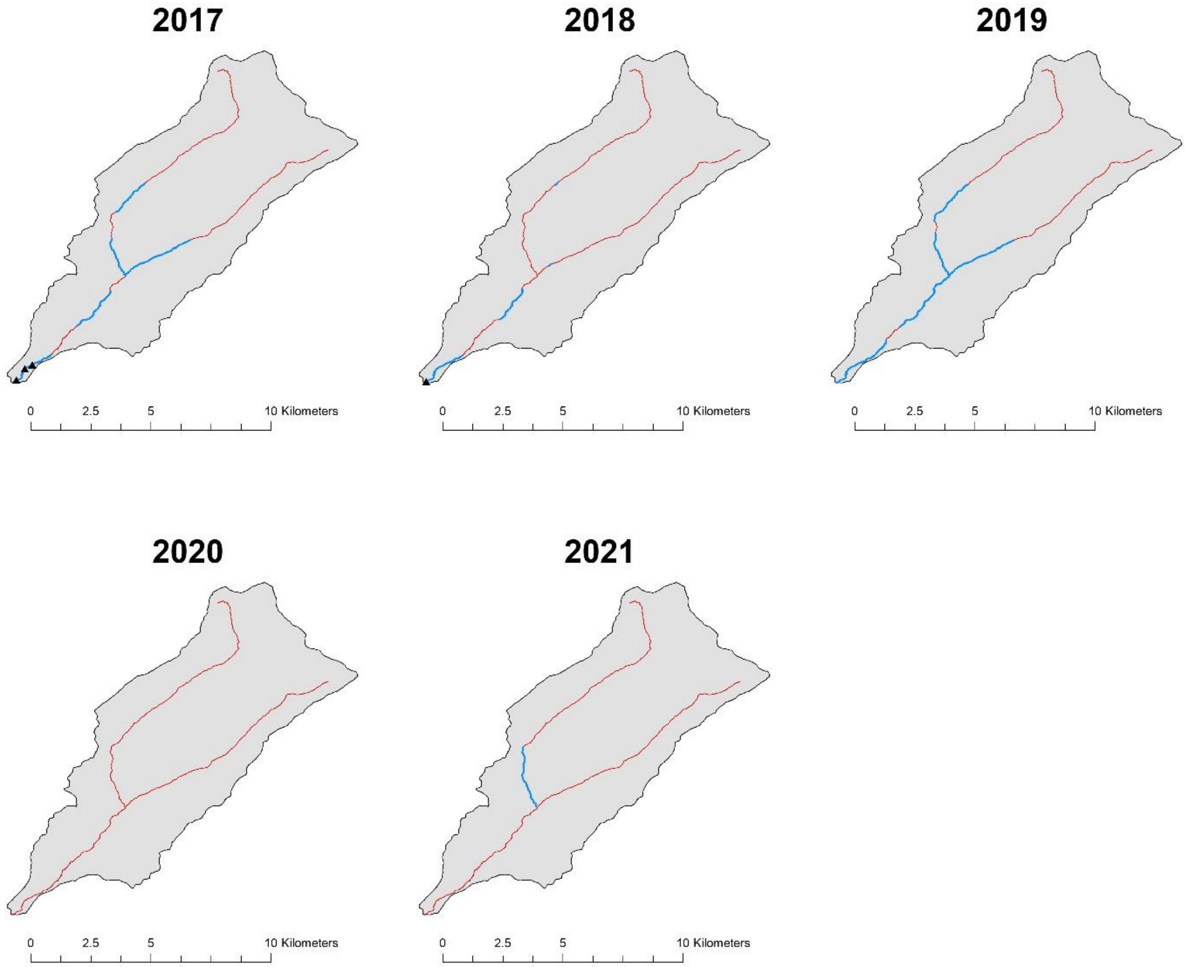

Figure B12. Distribution of juvenile Chinook Salmon encountered during annual summer electrofishing surveys in Bohannon Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches. No sampling was conducted in 2020.

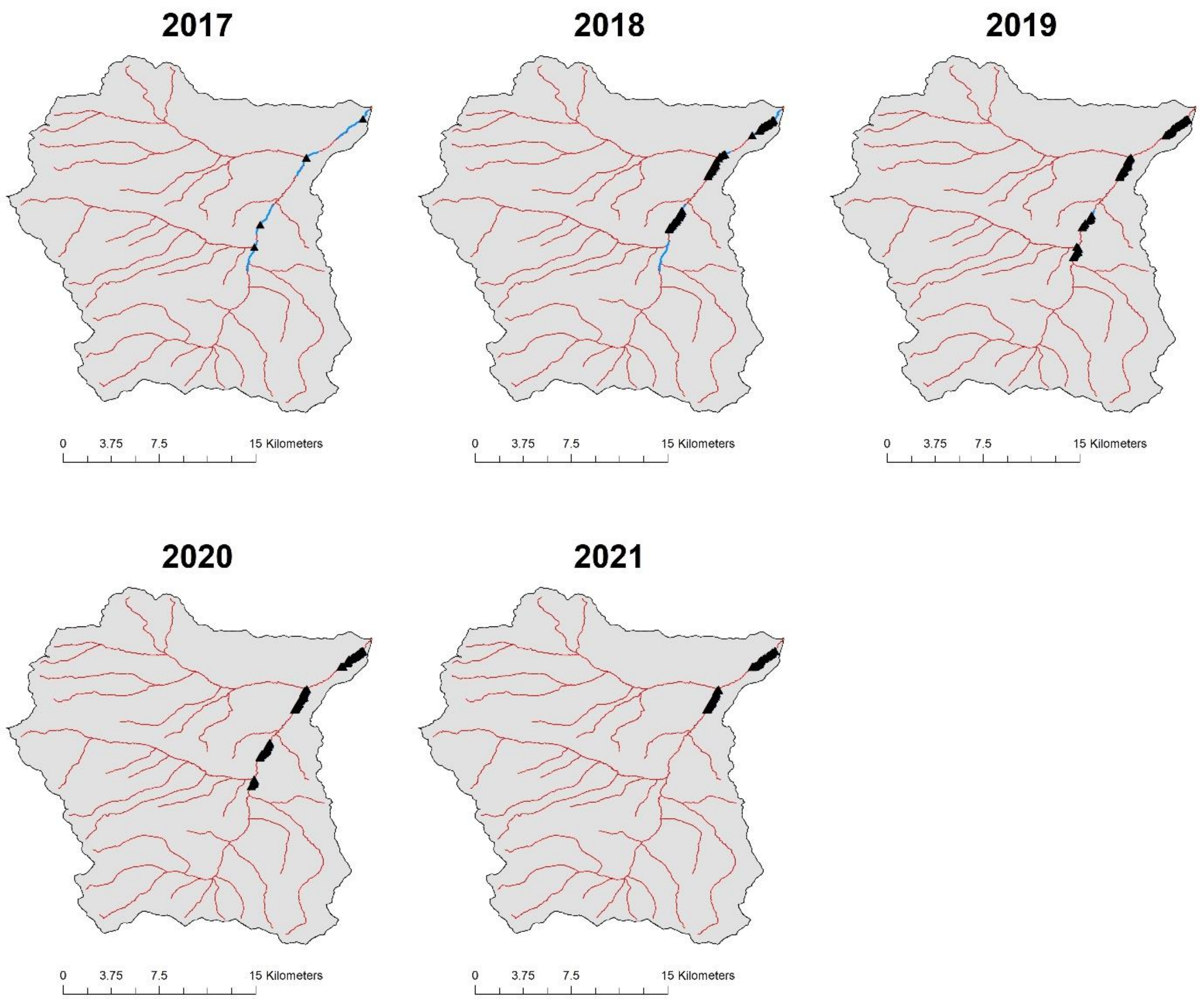

Figure B13. Distribution of juvenile Chinook Salmon encountered during annual summer electrofishing surveys in Hayden Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

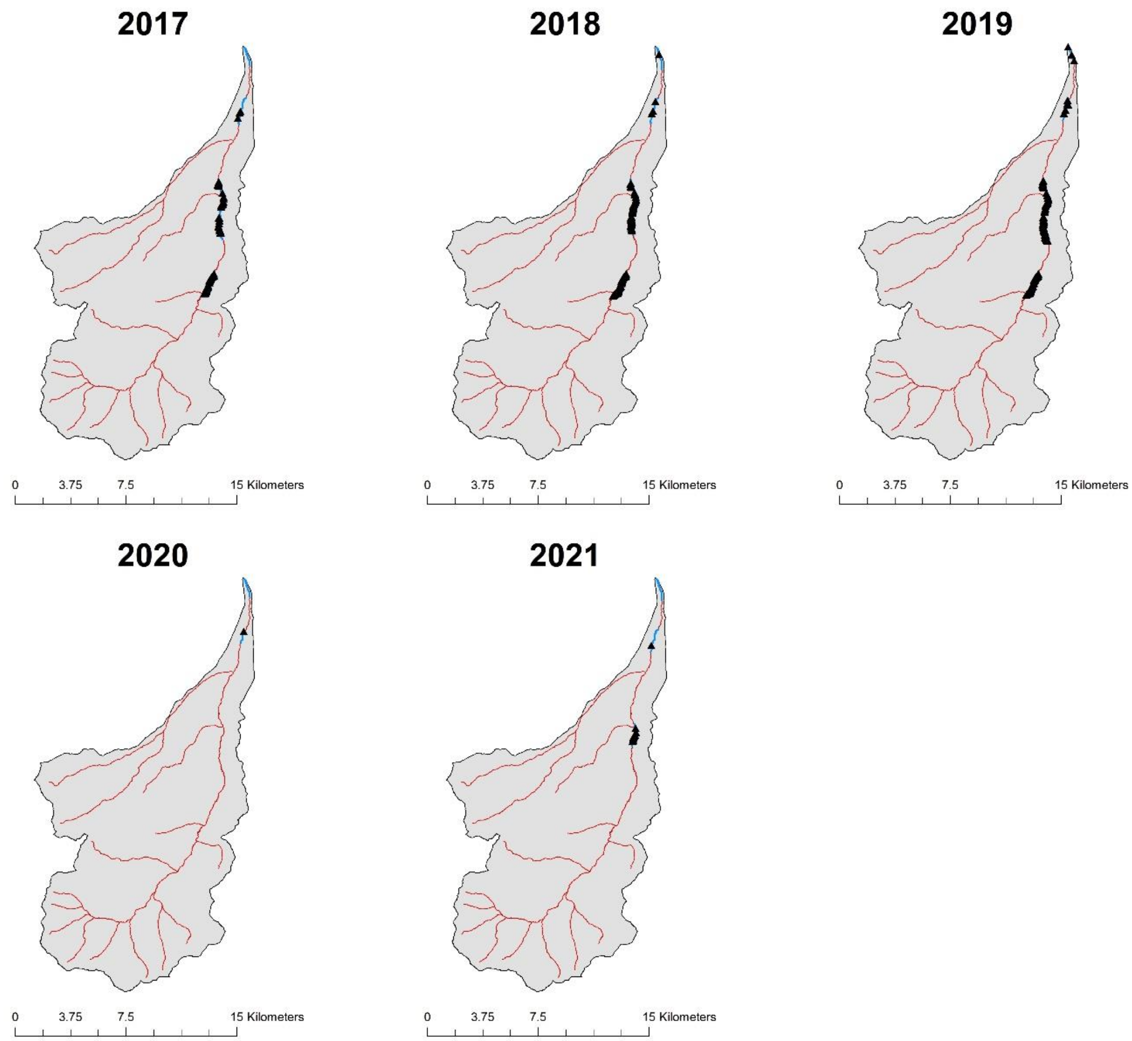

Figure B14. Distribution of Bull Trout encountered during annual summer electrofishing surveys in Big Timber Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

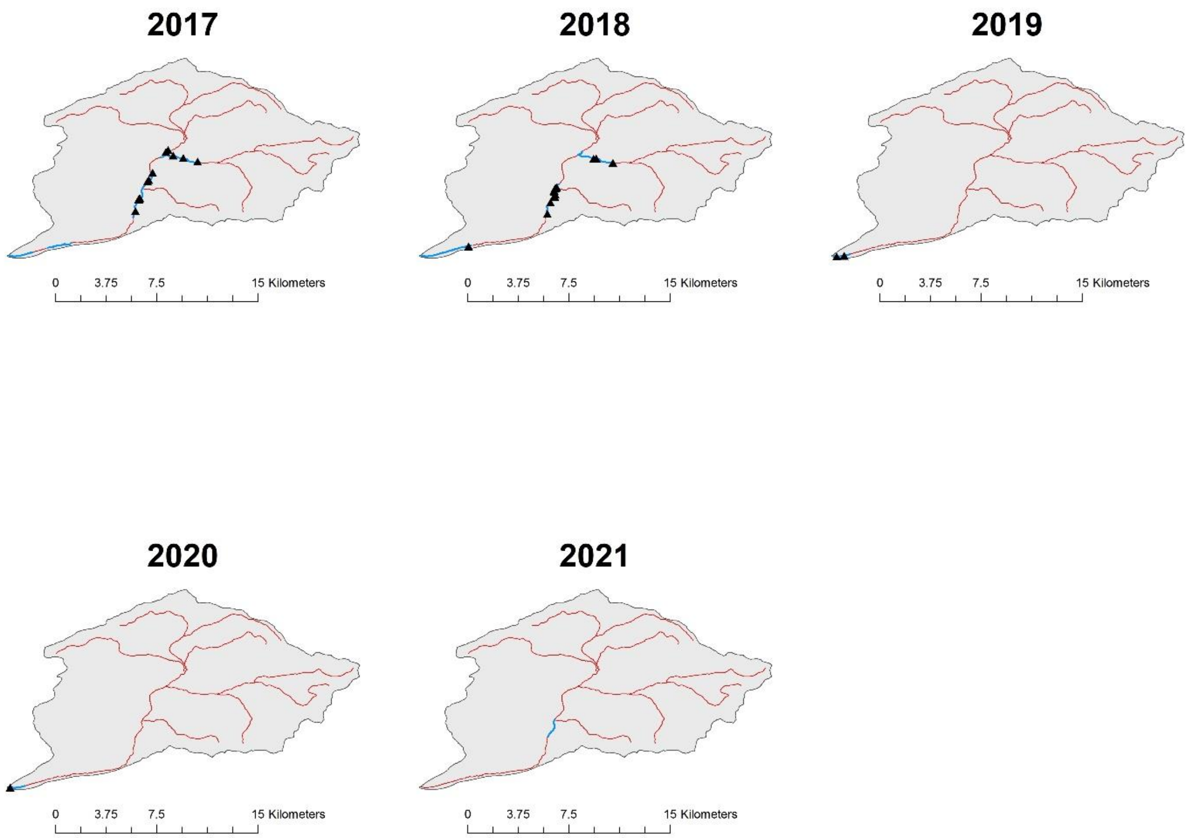

Figure B15. Distribution of Bull Trout encountered during annual summer electrofishing surveys in Canyon Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

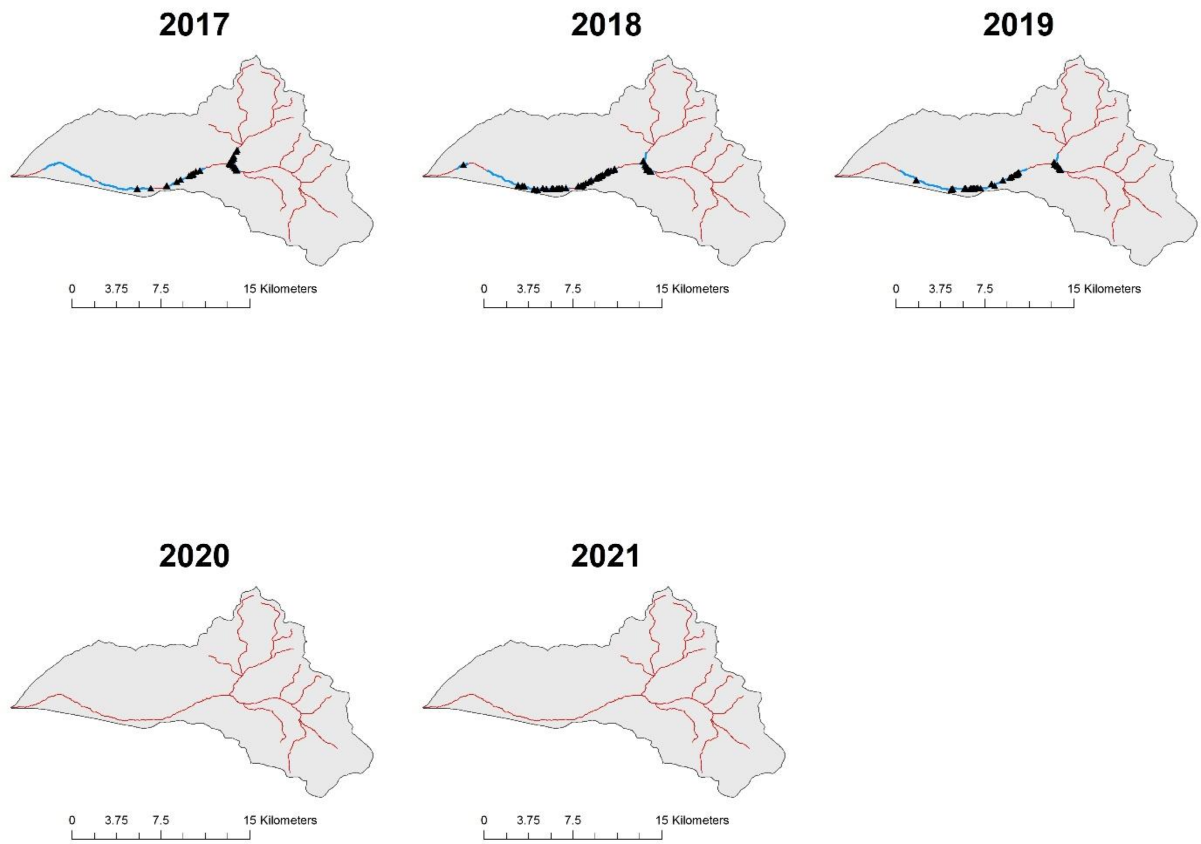

Figure B16. Distribution of Bull Trout encountered during annual summer electrofishing surveys in Hawley Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches. No sampling was conducted in 2020 and 2021.

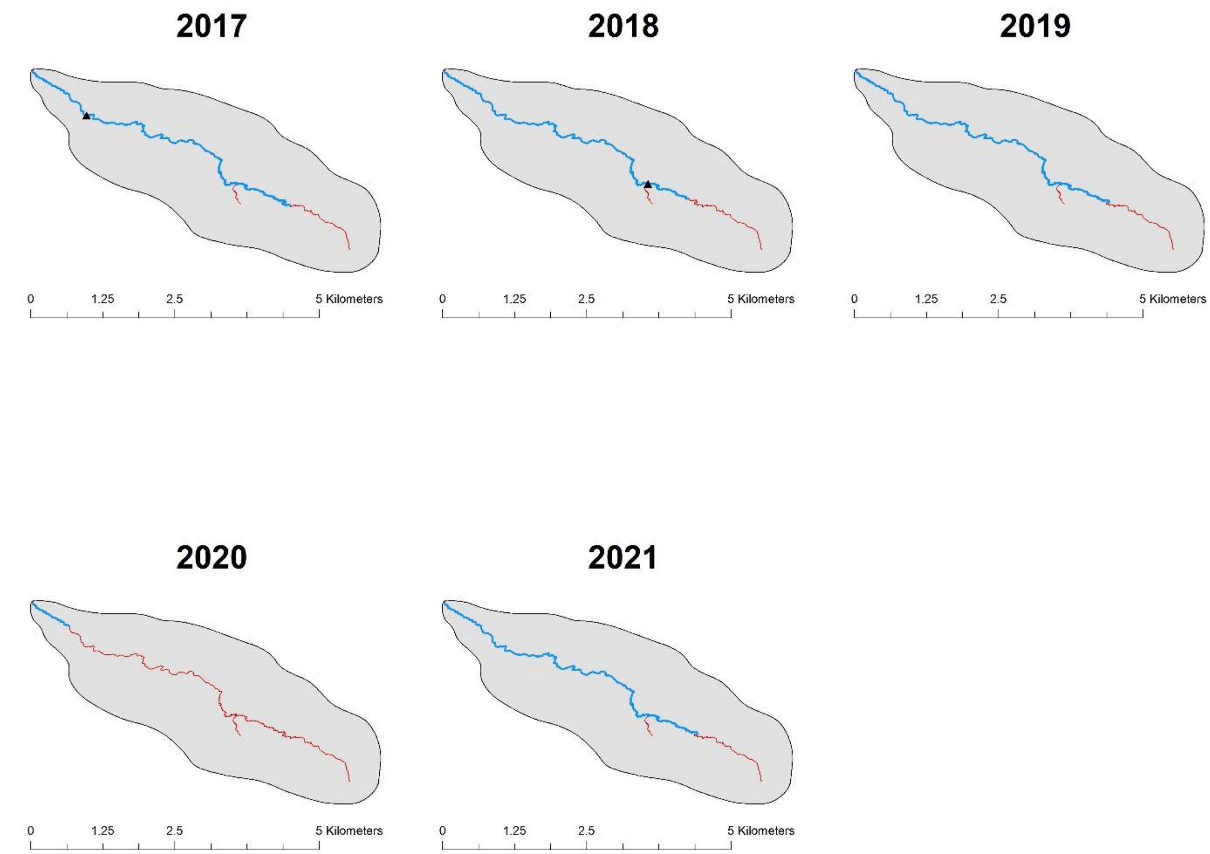

Figure B17. Distribution of Bull Trout encountered during annual summer electrofishing surveys in Little Springs Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

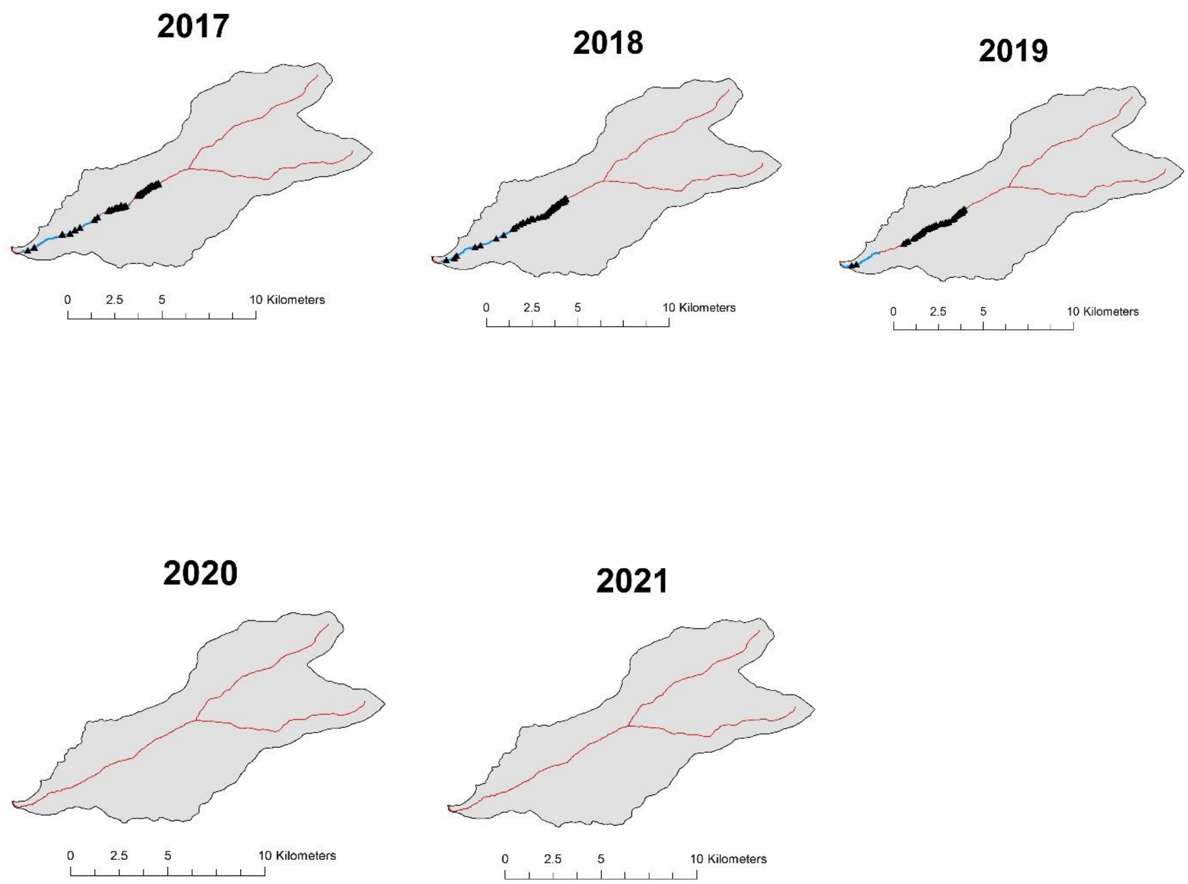

Figure B18. Distribution of Bull Trout encountered during annual summer electrofishing surveys in Kenney Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches. No sampling was conducted in 2020 and 2021.

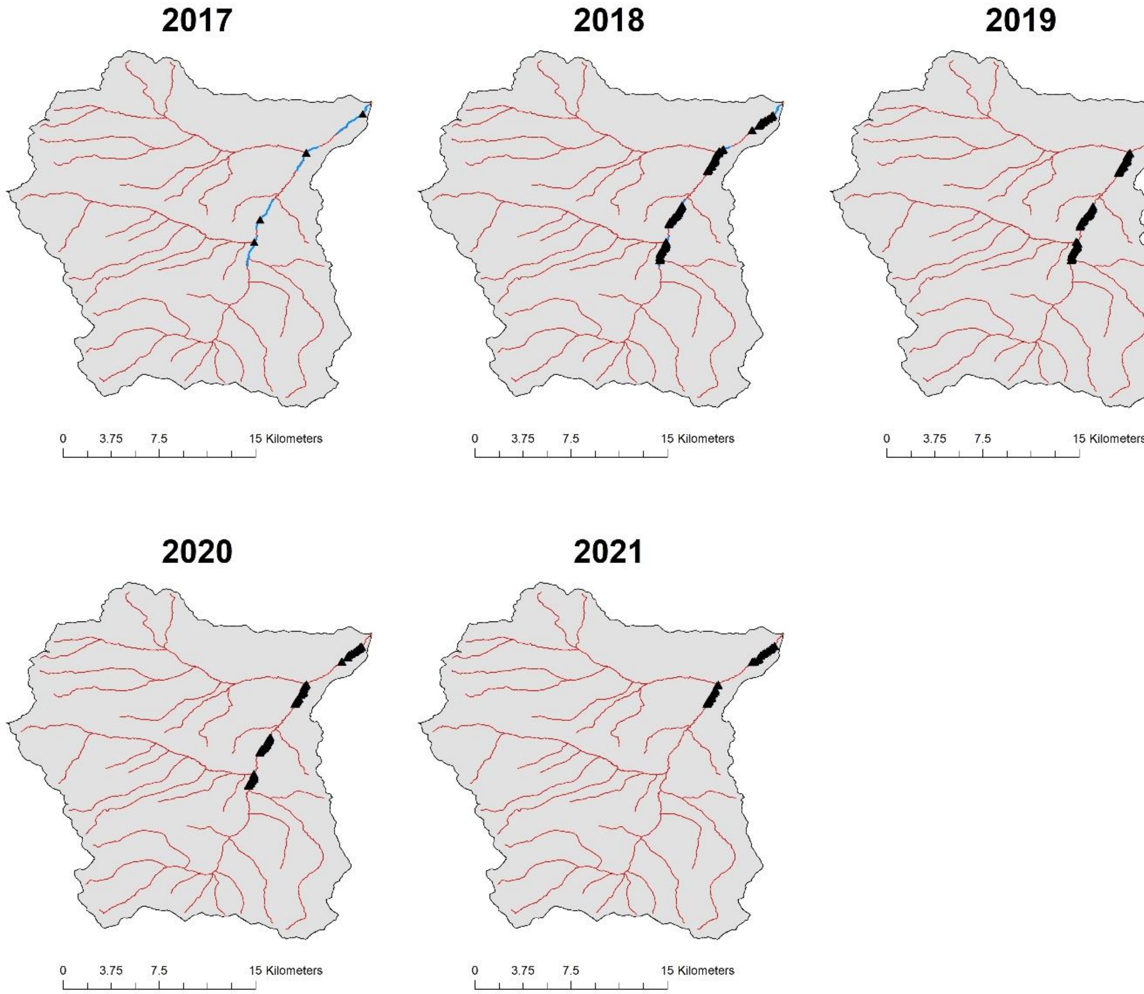

Figure B19. Distribution of juvenile Bull Trout encountered during annual summer electrofishing surveys in Hayden Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

Appendix C. Lemhi River priority tributary and Hayden Creek instream PIT tag detector system interrogation tables.

Table C1. Number of steelhead and Chinook Salmon tagged in Big Timber Creek that were subsequently detected on the instream PIT tag detection system near the mouth of Big Timber Creek (BTC). Proportion of tagging cohort shown in parentheses.

Tag year	Number tagged	Number detected by year 2017	2018	2019	2020	2021	Total
				Steelhead			
2017	555	11 (1.9)	10 (1.8)	2 (0.3)			23 (4)
2018	566		14 (2.4)	6 (1.0)	1 (0.1)		21 (3.5)
2019	702			5 (0.7)	16 (2.2)	2 (0.2)	23 (2.4)
2020	151				17 (11.2)	12 (1.7)	29 (12.9)
2021	355					15 (4.2)	15 (4.2)
				Chinook Salmon			
2017	8	5 (62.5)	1 (12.5)	0	0	0	6 (75)
2018	1		0	0	0	0	0
2019	39			4 (10.2)	12 (30.7)	0	16 (40.9)
2020	129				41 (31.7)	21 (16.2)	62 (47.9)
2021	24					9 (37.5)	9 (37.5)

Table C2. Number of steelhead, Chinook Salmon, and Bull Trout tagged in the Lemhi River basin outside of Big Timber Creek that were subsequently detected on the instream PIT tag detection system near the mouth of Big Timber Creek (BTC).

Species	Number detected by year 2017	2018	2019	2020	2021	Total
Steelhead	6	31	4	11	2	54
Chinook Salmon	0	0	0	9	0	9
Bull Trout	0	1	0	2	0	3

Table C3. Number of steelhead and Chinook Salmon tagged in Canyon Creek that were subsequently detected on the instream PIT tag detection system near the mouth of Canyon Creek (CAC). Proportion of tagging cohort shown in parentheses.

Tag year	Number tagged	Number detected by year 2017	2018	2019	2020	2021	Total
				Steelhead			
2017	535	10 (1.8)	16 (2.9)	2 (0.3)	0	0	28 (5)
2018	527		22 (4.1)	22 (4.1)	2 (0.3)	0	46 (8.5)
2019	26			9 (34.6)	4 (15.3)	0	13 (49.9)
2020	38				15 (39.4)	3 (7.8)	18 (74.2)
2021	147					0	0
				Chinook Salmon			
2017	1	0	0	0	0	0	0
2018	1		1 (100.0)	0	0	0	1 (100.0)
2019	23			14 (60.8)	2 (8.6)	0	16 (69.4)
2020	32				21 (65.6)	4 (12.5)	25 (78.1)
2021	0					0	0

Table C4. Number of steelhead, Chinook Salmon, and Bull Trout tagged in the Lemhi River basin outside of Canyon Creek that were subsequently detected on the instream PIT tag detection system near the mouth of Canyon Creek (CAC).

Species	Number detected by year 2017	2018	2019	2020	2021	Total
Steelhead	1	6	12	19	4	42
Chinook Salmon	0	3	3	7	0	13
Bull Trout	0	0	1	3	0	4

Table C5. Number of steelhead and Chinook Salmon tagged in Little Springs Creek that were subsequently detected on the instream PIT tag detection system near the mouth of Little Springs Creek (LLS). Proportion of tagging cohort shown in parentheses. NA = creek was not sampled.

Tag year	Number tagged	Number detected by year 2017	2018	2019	2020	2021	Total
				Steelhead			
2017	134	51 (38.0)	12 (8.9)	0	0	0	63 (46.9)
2018	162		84 (51.8)	2 (1.2)	0	0	86 (53.0)
2019	125			27 (21.6)	0	0	27 (21.6)
2020	40				0	1 (2.5)	1 (2.5)
2021	78					16 (20.5)	16 (20.5)
				Chinook Salmon			
2017	2	1 (50.0)	1 (50.0)	0	0	0	2 (100.0)
2018	13		8 (61.5)	0	0	0	8 (61.5)
2019	8			5 (62.5)	0	0	5 (62.5)
2020	62				10 (16.1)	0	10 (16.1)
2021	20					0	0

Table C6. Number of steelhead, Chinook Salmon, and Bull Trout tagged in the Lemhi River basin outside of Little Springs Creek that were subsequently detected on the instream PIT tag detection system near the mouth of Little Springs Creek (LLS).

Species	Number detected by year 2017	2018	2019	2020	2021	Total
Steelhead	21	13	4	0	5	43
Chinook Salmon	0	1	2	0	1	4
Bull Trout	0	2	1	0	3	6

Table C7. Number of steelhead and Bull Trout tagged in Kenney Creek that were subsequently detected on the instream PIT tag detection system near the mouth of Kenney Creek (KEN). Proportion of tagging cohort shown in parentheses. NA = creek was not sampled.

Tag year	Number tagged	Number detected by year					Total
		2017	2018	2019	2020	2021	
				Steelhead			
2017	417	0	10 (2.3)	16 (3.8)	3 (0.7)	1 (0.2)	30 (7)
2018	376		17 (4.5)	52 (13.8)	12 (3.1)	2 (0.5)	83 (21.9)
2019	823			49 (5.9)	111 (13.4)	33 (4.0)	193 (23.3)
2020	NA				0	0	0
2021	NA					0	0
				Bull Trout			
2017	49	0	1 (2.0)	0	0	0	1 (2.0)
2018	38		3 (7.0)	1 (2.0)	0	0	4 (9)
2019	51			1 (1.0)	1 (1.0)	3 (5.0)	5 (2.5)
2020	NA				0	0	0
2021	NA					0	0

Table C8. Number of steelhead, Chinook Salmon, and Bull Trout tagged in the Lemhi River basin outside of Kenney Creek that were subsequently detected on the instream PIT tag detection system near the mouth of Kenney Creek (KEN).

Species	Number detected by year					Total
	2017	2018	2019	2020	2021	
Steelhead	0	2	12	10	12	36
Chinook Salmon	0	0	0	1	1	2
Bull Trout	0	1	3	2	2	8

Table C9. Number of steelhead and Chinook Salmon tagged in Bohannon Creek that were subsequently detected on the instream PIT tag detection system near the mouth of Bohannon Creek (BHC). Proportion of tagging cohort shown in parentheses.

Tag year	Number tagged	Number detected by year 2017	2018	2019	2020	2021	Total
				Steelhead			
2017	1,002	93 (9.2)	22 (2.1)	32 (3.1)	2 (0.1)	1 (0.0)	150 (14.5)
2018	320		32 (10.0)	61 (19.0)	3 (0.9)	0	96 (29.9)
2019	500			45 (9.0)	29 (5.8)	5 (1.0)	79 (15.8)
2020	NA				0	0	0
2021	345					5 (1.4)	5 (1.4)
				Chinook Salmon			
2017	2	0	0	0	0	0	0
2018	1		1 (100.0)	0	0	0	1 (100.0)
2019	0			0	0	0	0
2020	NA				0	0	0
2021	0					0	0

Table C10. Number of steelhead, Chinook Salmon, and Bull Trout tagged in the Lemhi River basin outside of Bohannon Creek that were subsequently detected on the instream PIT tag detection system near the mouth of Bohannon Creek (BHC).

Species	Number detected by year 2017	2018	2019	2020	2021	Total
Steelhead	10	14	23	17	29	93
Chinook Salmon	2	0	3	1	3	9
Bull Trout	0	0	1	5	2	8

Table C11. Number of steelhead and Chinook Salmon tagged in Hayden Creek that were subsequently detected on the instream PIT tag detection system near the mouth of Hayden Creek (HYC). Proportion of tagging cohort shown in parentheses.

Tag year	Number tagged	Number detected by year					Total
		2017	2018	2019	2020	2021	
				Steelhead			
2017	771	87 (11.2)	81 (10.5)	16 (2.0)	4 (0.5)	0	188 (24.2)
2018	795		61 (7.6)	77 (9.6)	27 (3.3)	2 (0.2)	167 (20.7)
2019	1045			53 (5.0)	80 (7.6)	6 (0.5)	139 (8.6)
2020	449				22 (4.8)	22 (4.8)	44 (9.6)
2021	843					23 (2.7)	23 (2.7)
				Chinook Salmon			
2017	886	391 (44.1)	25 (2.8)	0	0	0	416 (46.9)
2018	282		52 (10.7)	53 (18.7)	0	0	105 (29.4)
2019	483			115 (23.8)	24 (4.9)	0	139 (28.7)
2020	1032				186 (18.0)	58 (5.6)	244 (23.6)
2021	387					49 (12.6)	49 (12.6)
				Bull Trout			
2017	147	14 (9.5)	19 (12.9)	5 (3.4)	1 (0.6)	0	39 (26.4)
2018	167		12 (7.1)	14 (8.3)	5 (2.9)	0	31 (18.3)
2019	242			11 (8.3)	32 (1.3)	7 (2.8)	50 (12.4)
2020	131				16 (12.2)	12 (9.1)	28 (21.3)
2021	81					3 (3.7)	3 (3.7)

Table C12. Number of steelhead, Chinook Salmon, and Bull Trout tagged in the Lemhi River basin outside of Hayden Creek that were subsequently detected on the instream PIT tag detection system near the mouth of Hayden Creek (HYC).

Species	Number detected by year					Total
	2017	2018	2019	2020	2021	
Steelhead	9	14	36	28	29	116
Chinook Salmon	20	23	38	38	47	166
Bull Trout	11	24	34	42	24	135

Appendix D. Hayden Creek and Bear Valley Creek Bull Trout Weirs

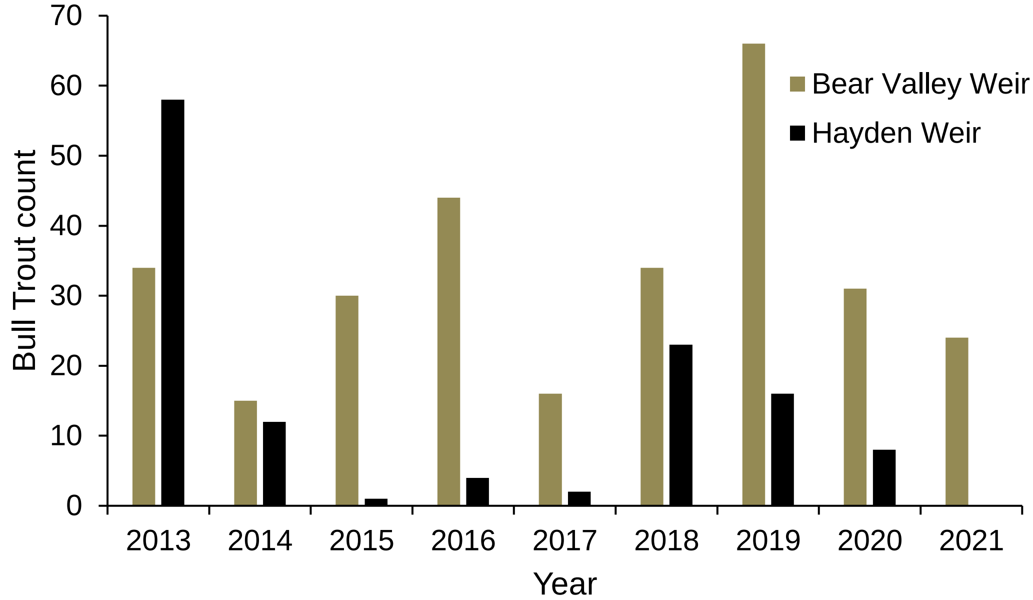

Figure D1. The number of adult Bull Trout captured at the Hayden Creek and Bear Valley Creek weirs between 2013 and 2021.

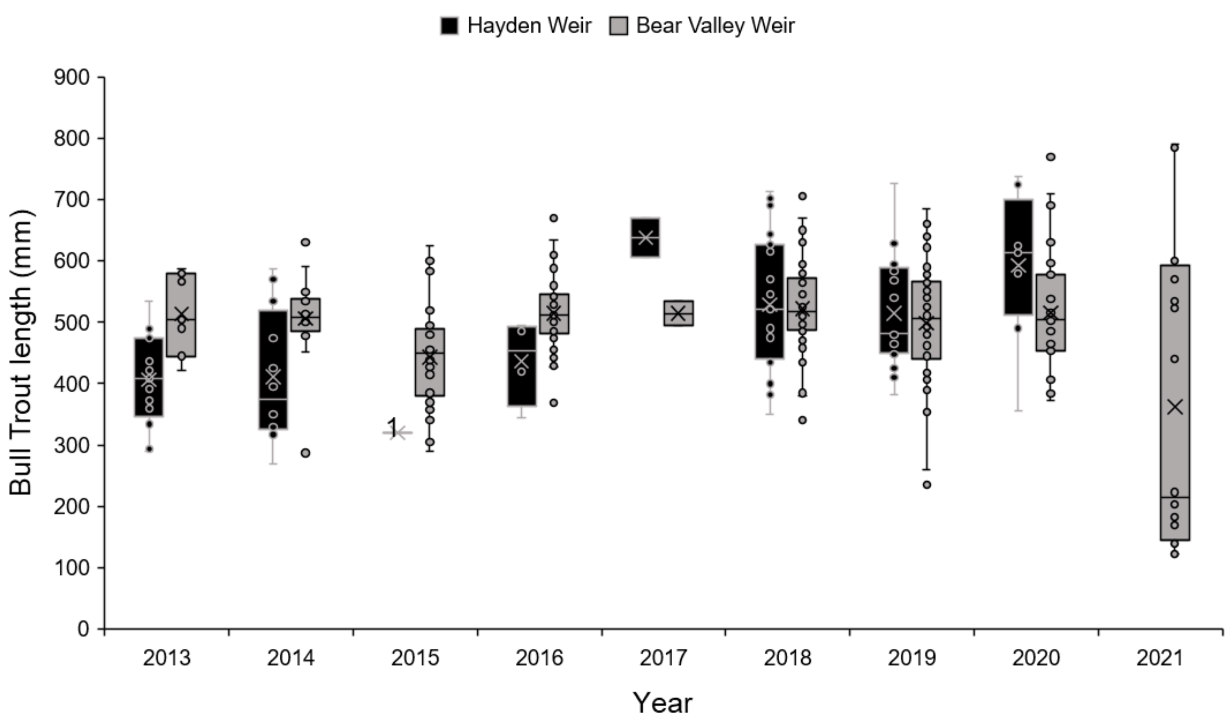

Figure D2. The length distribution of adult Bull Trout captured at the Hayden Creek and Bear Valley Creek weirs between 2013 and 2021.

Appendix E. Locations of Chinook Salmon and Steelhead Redds Observed During Annual Spawning Ground Surveys in the Lemhi River Basin.

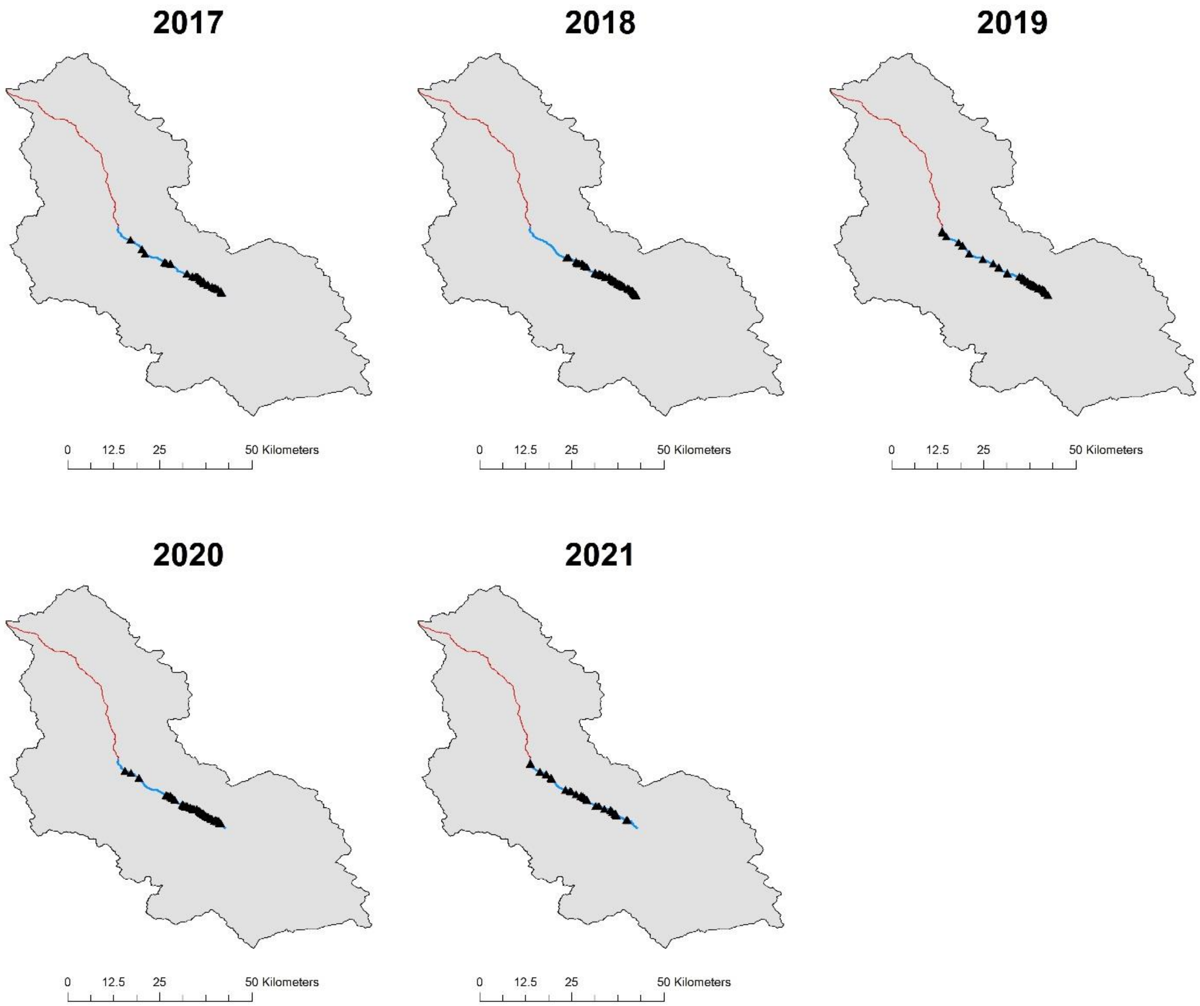

Figure E1. Locations of Chinook Salmon redds observed during annual spawning ground surveys in the upper Lemhi River, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

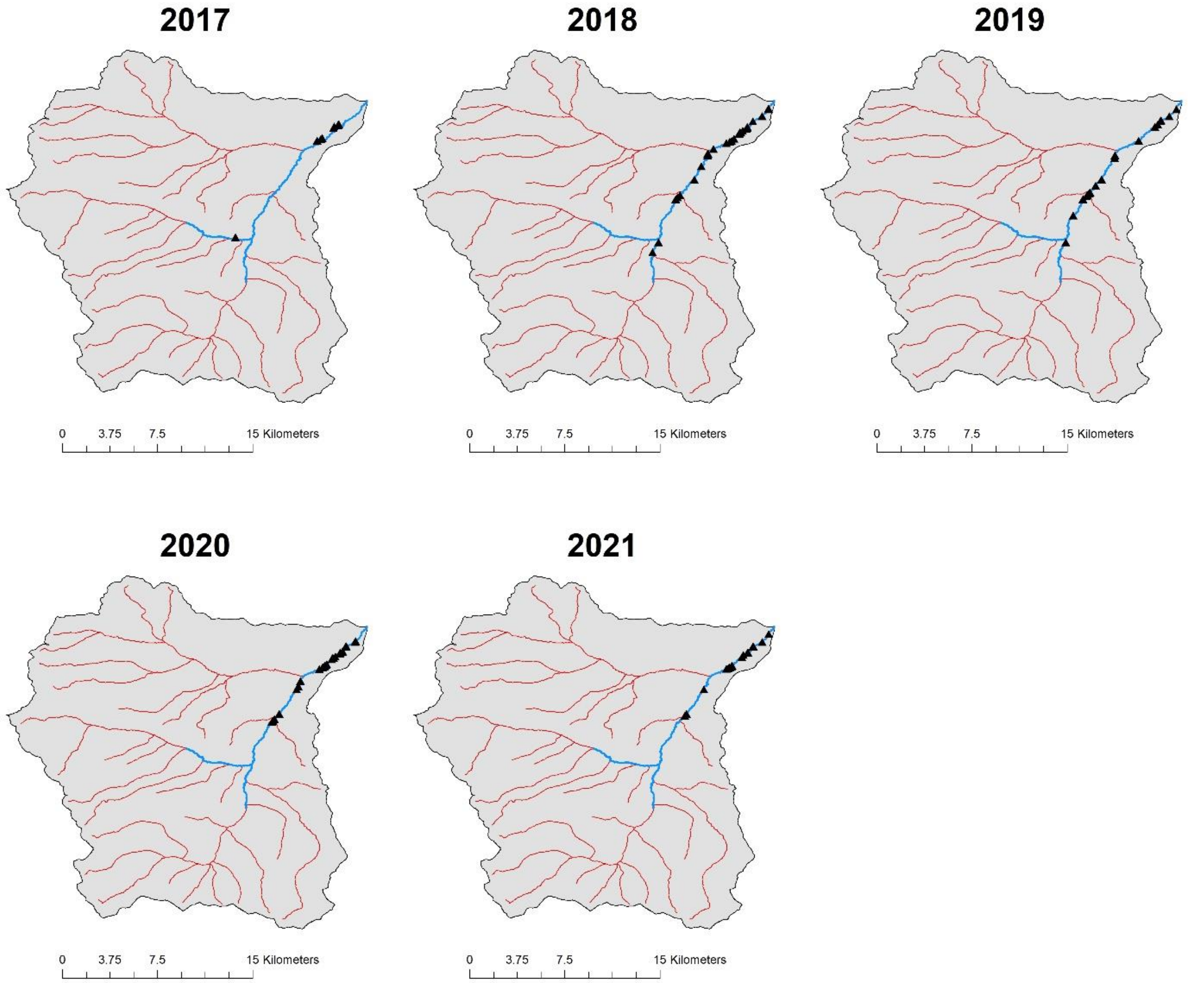

Figure E2. Locations of Chinook Salmon redds observed during annual spawning ground surveys in the Hayden Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

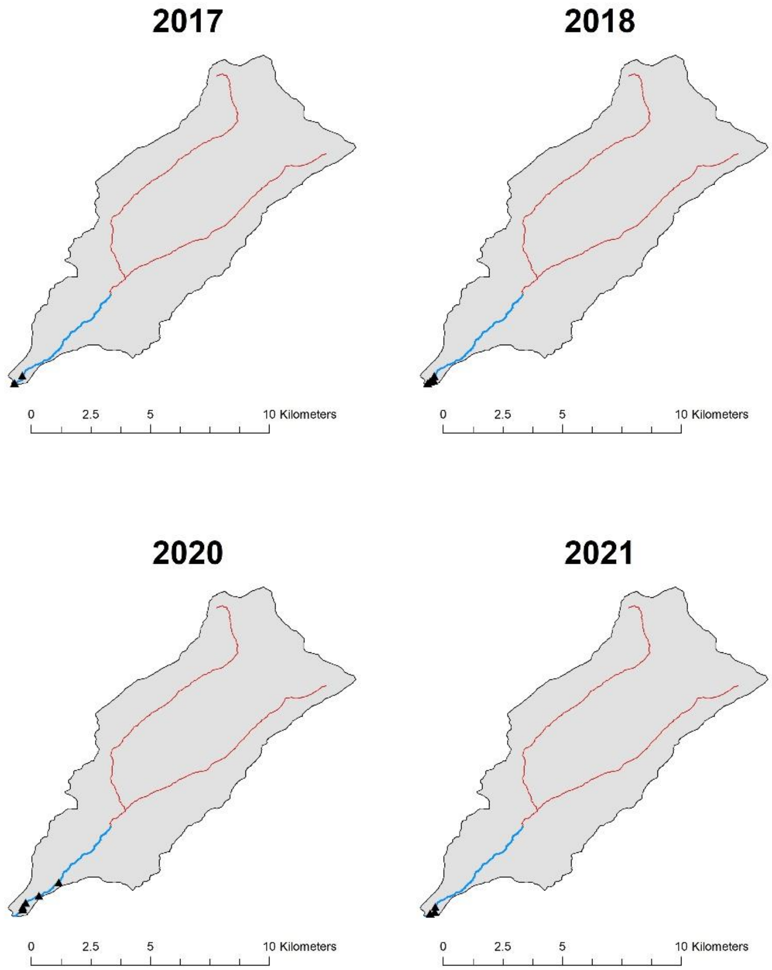

Figure E3. Locations of steelhead redds observed during annual spawning ground surveys in the Bohannon Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

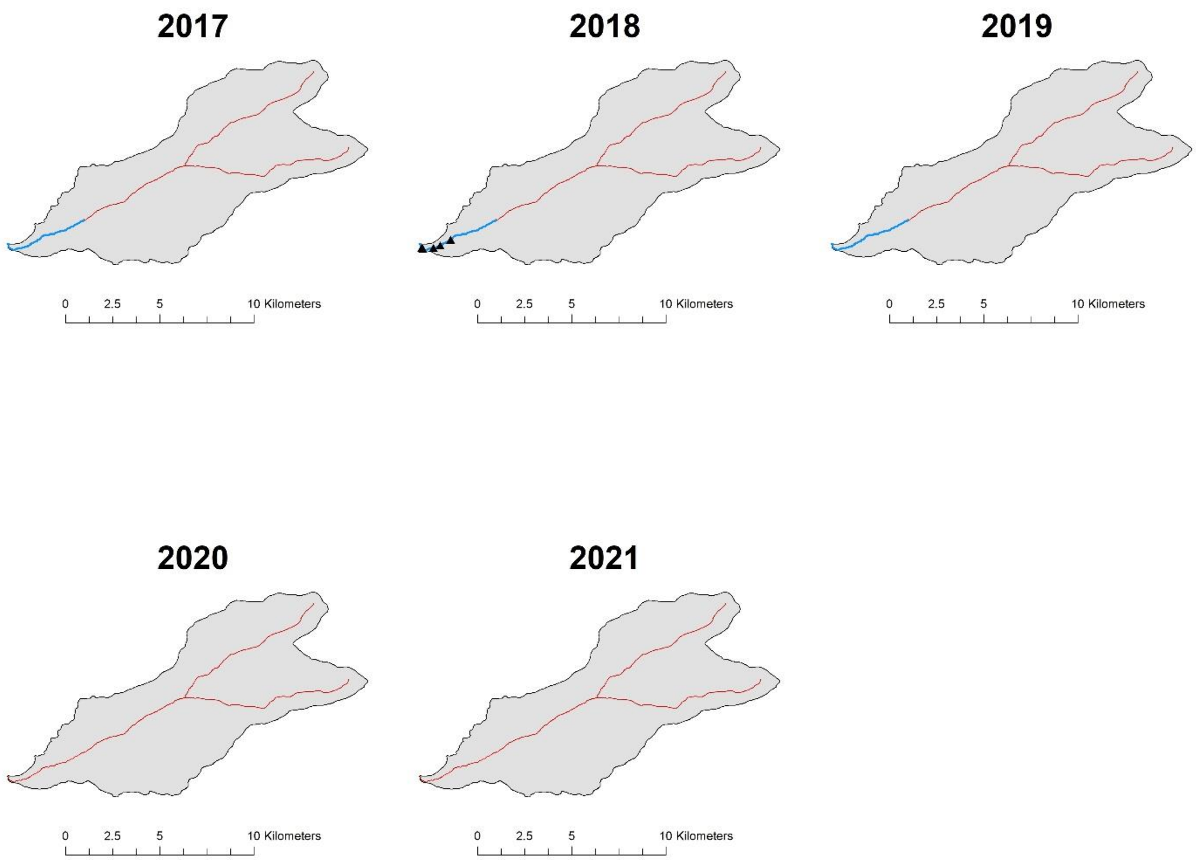

Figure E4. Locations of steelhead redds observed during annual spawning ground surveys in the Kenney Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches. No surveys were conducted in 2020 and 2021.

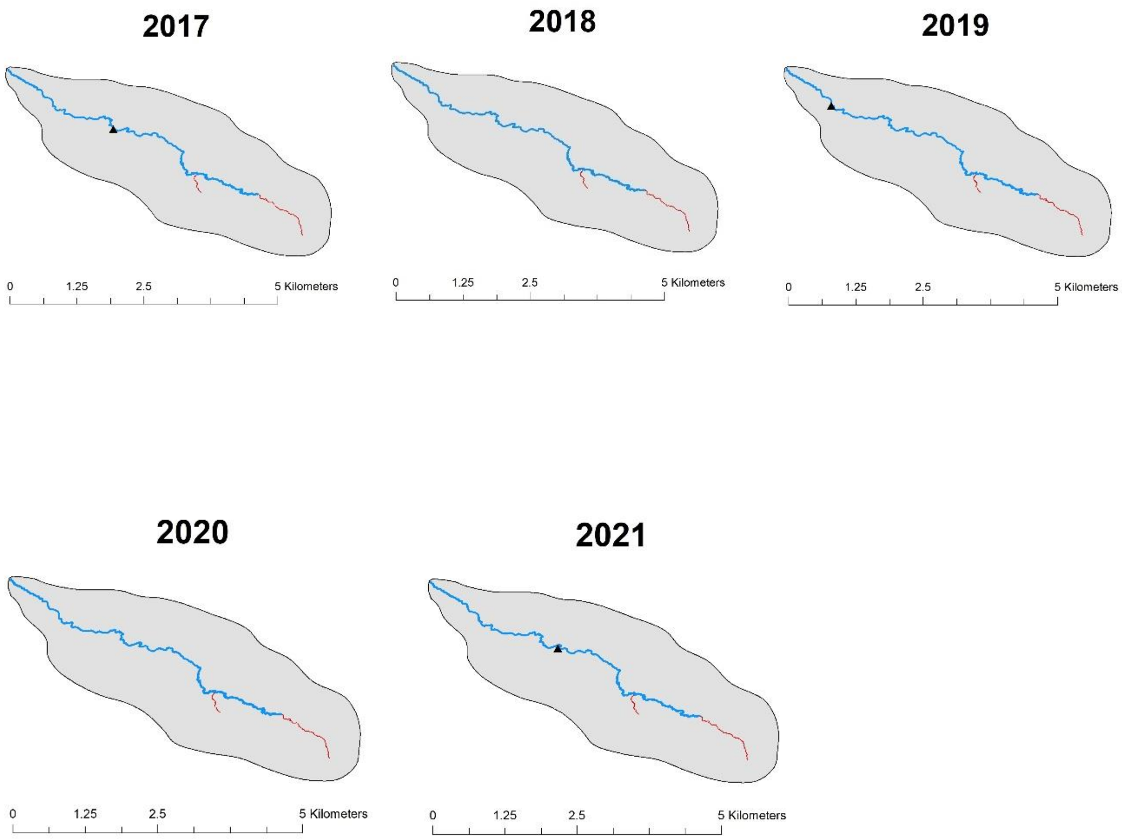

Figure E5. Location of steelhead redds observed during annual spawning ground surveys in Little Springs Creek, 2017-2021. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

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