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

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ABSTRACT

It has proven difficult to demonstrate the benefits of stream habitat restoration on fish populations. Therefore, management experiments have been implemented at the watershed scale with well-developed, long-term monitoring programs to determine watershed-scale fish and habitat responses to restoration actions via causal mechanisms. These are termed Intensively Monitored Watershed (IMW) projects. The IMW projects in Idaho are located in the Lemhi River and the Potlatch River watersheds. The goal of this report is to summarize the work conducted by the Idaho IMW projects since initiation in the mid-2000s. Report objectives are 1) to offer preliminary conclusions to guide restoration in Idaho and elsewhere; and 2) to revisit expectations and allow for adaptive management towards completion of the IMWs.

The Lemhi River IMW is located in the Salmon River basin in east-central Idaho and is focused on Chinook Salmon, steelhead trout, 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 main stem. To date, three of six priority tributaries have been reconnected, allowing access to >38 km of spawning and rearing habitat. A minimum flow agreement keeps sufficient water in the lower Lemhi River during the summer for fish passage, and water conservation measures have been taken to increase rearing habitat in selected reaches and tributaries. Restoration projects increased habitat complexity by adding large woody debris (>5 projects) and established floodplain and lateral habitat connections (4 projects). Chinook Salmon have not yet spawned in newly accessible habitats, but adults have been detected migrating into a reconnected tributary. Steelhead Trout spawning has been documented in two priority tributaries, and fluvial Bull Trout have been documented using one reconnected tributary. There has been an increase in abundance and the upstream occurrence of juvenile Chinook Salmon in priority tributaries. We have documented survival advantages for Chinook Salmon using reconnected tributaries. Further, there has been an increase in the number of smolts per redd emigrating from the upper reach of the Lemhi River relative to Hayden Creek, the reference system. Results have identified life-stage-specific limiting factors (e.g., winter survival), which are being used to guide restoration in the Lemhi River and throughout the upper Salmon River basin. These results are also being used to develop fish-habitat relationships and monitoring techniques for use by restoration practitioners more generally. Significant amounts of restoration remain to be implemented.

The Potlatch River IMW is located in the Clearwater River basin in northern Idaho and is focused on steelhead trout. Restoration in the Potlatch River addresses tributary blockages and dewatered reaches in the lower watershed and simplified habitat in the upper watershed. In the lower watershed, 10 barriers have been removed, opening >10 km. A flow supplementation project treated >16 km during summer low flows, creating 8 km of additional wetted habitat, reducing water temperature, and increasing dissolved oxygen through treatment reach. In the upper watershed, 4.0 km have been treated with large woody debris structures, increasing habitat complexity and enhancing hydrologic function. Steelhead trout spawning and fry production was observed in a blocked reach after the barrier was removed in the lower watershed. Seasonal use of instream structures by juveniles was documented. Pretreatment data shows density dependence in smolt/female productivity, demonstrating the need for habitat restoration and providing a good baseline against which to evaluate it. Observed correlation between flow conditions and juvenile survival and changes in emigrant age composition and size justify the restoration approaches being implemented. The Potlatch River dataset provides a valuable monitoring component for the Lower Clearwater River Main-stem steelhead trout population, and the IMW has contributed information included in the draft recovery plan for the population. Most

of the planned restoration projects in the Potlatch River watershed have yet to be implemented, but modeling exercises show the potential for them to generate a watershed-level response.

Experiences from 10 years of monitoring in the Lemhi and Potlatch watersheds have valuable lessons for habitat restoration. Both projects show how monitoring information can highlight needs not anticipated and thus focus restoration projects towards program goals more efficiently. Clear guidance documents and transparency can foster effective working relationships and make adjustments easier to implement. Both projects have had to reconsider study designs necessary to evaluate restoration objectives, highlighting the need for review, feedback, and adaptation in monitoring design. Reviews can catch errors or notice trends that may not have been apparent at the outset. Thus, strong monitoring designs can accommodate change while protecting the quality and consistency of the data. With this deeper understanding, results can be applied more generally to shape restoration to suit other scenarios, setting the scope for expectations of restoration effectiveness. These lessons will help Idaho's habitat restoration program to be more efficient and strategic and build credibility with landowners and cooperating agencies.

INTRODUCTION

A major underpinning of recovery efforts for Pacific salmon listed under the Endangered Species Act (ESA) is that there is a strong relationship between freshwater habitat quantity and quality and salmon abundance, survival, and productivity in the fresh water environment (Roni et al. 2014). In the Columbia River basin, management under the 2008 Federal Columbia River Power System Biological Opinion (NMFS 2008) includes an expanded habitat program to protect and improve tributary and estuary environments and reduce limiting factors, based on the biological needs of listed fish. The tributary habitat program requires implementation of habitat improvement actions, including actions to protect and improve main-stem and side-channel habitat for fish migration, spawning and rearing, and to restore floodplain function. In the Pacific Northwest, tens of thousands of restoration actions have been implemented in recent decades, primarily focusing on anadromous salmonids (Katz et al. 2007).

It has proven difficult to demonstrate the benefits of stream habitat restoration on fish populations. Several studies have questioned the benefits of habitat restoration measures (e.g., Thompson 2006; Bernhardt and Palmer 2011) or found equivocal evidence (Roni et al. 2008; Stewart et al. 2009). There are many confounding factors that complicate evaluation of habitat restoration for anadromous salmonids. Riverine environments are complex and our knowledge of them is uncertain (Wissmar and Bisson 2003). Anadromous salmonids have complex life histories crossing freshwater and oceanic habitats and have generation times that take several years to complete (Groot and Margolis 1991; Quinn 2005). The combination of environmental and biological variability with the scale at which management is focused (watershed or population) means that proper assessments are difficult and time-consuming to do.

To address these issues, the concept of the intensively monitored watershed (IMW) was developed. The basic premise is that the complex relationships controlling salmon response to habitat conditions can best be understood by concentrating monitoring and research at a few locations such that sampling intensity is sufficiently high to evaluate complicated biological responses to management actions (Bilby et al. 2004; Waste 2007). Effective management in the presence of uncertainty requires learning and flexibility; within this context, monitoring is important as the feedback loop whereby management is evaluated (Kershner 1997; Bisbal 2001). Hence, an IMW project requires an explicit monitoring design at the watershed scale, deliberate restoration actions that can be tied to expected salmon responses, and a long-term commitment so that responses can be detected by the monitoring. In other words, 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). The IMW concept evolved in western Washington State (Bilby et al. 2004), but IMWs have been implemented across the Pacific Northwest (Bennett et al. 2016) because they are the best option to deliver integrative watershed-scale evaluations that assess fish population responses to habitat actions and adequately address confounding factors (Waste 2007).

As the responsible party for developing recovery plans, NOAA is interested in evaluating population-level effects of habitat restoration by the states of Washington, Oregon, and Idaho. Therefore, a grant program to the states entitled ‘Monitoring State Restoration of Salmon Habitat in the Columbia Basin’ was initiated by NOAA through the Pacific States Marine Fisheries Commission. Goals of this program are to provide landscape applicable results that will inform decision makers at NOAA and in the states on the effectiveness of restoration actions taken to date, as well as how to address factors that limit specific population processes and how these

factors can be mitigated with certain restoration actions. Further, IMWs should show progress in the Columbia Basin toward recovery of ESA listed salmon and steelhead trout.

There are two IMW projects in Idaho: the Lemhi River and the Potlatch River basins. These projects are located where habitat restoration efforts are being focused in order to determine the effectiveness of that work toward increased fish production and to provide guidance on future habitat work (IDFG 2013). The Chinook Salmon and steelhead trout populations that spawn and rear in these watersheds are considered priority populations by NOAA (2016). The Idaho Department of Fish and Game (IDFG) had conducted varying amounts of monitoring and research in these basins historically but past efforts were not focused on the links between local habitat conditions and the populations of interest. Hence, an IMW approach is desirable to elucidate these relationships and establish a cause-and-effect link between restoration work and population-level fish response. Implementation of IMWs will ensure that results will be useful elsewhere in Idaho and the Pacific Northwest because of rigorous experimental design and monitoring. Therefore, NOAA has funded IDFG to conduct monitoring and research through a grant to Pacific States Marine Fisheries Commission entitled 'Monitoring State Restoration of Salmon Habitat in the Columbia Basin'.

The goal of this report is to summarize the work conducted by the Idaho IMW projects to date. The Lemhi and Potlatch IMWs were first implemented in 2007 and 2008, respectively, and the first five years of results were summarized by Bowersox and Biggs (2012). The current NOAA grant expires July 1, 2017. Hence, in this report we summarize progress since the earlier report as well as synthesize results over the entire project term. The objectives of this summary are 1) to revisit expectations and allow for adaptive management towards completion of the IMW projects; and 2) to offer preliminary conclusions to guide restoration elsewhere. This document will have two large chapters detailing the work and results from each Idaho IMW followed by a brief synthesis to summarize the broader lessons learned.

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. Early accounts by explorers highlighted the abundance of Chinook Salmon *Oncorhynchus tshawytscha* in the basin and compared the Lemhi Shoshone-Bannock tribes' reliance on salmon to that of the Plains tribes' reliance on bison (Walker 1993). A Bureau of Indian Affairs agent made specific mention of the harvest of wagonloads of Chinook Salmon and steelhead trout *O. mykiss* (hereafter steelhead) from the Lemhi River and its tributaries (Walker 1993). An early settler of the Lemhi River valley constructed a weir at the mouth of the Lemhi River that operated from 1862-1879 and captured Chinook Salmon weighing 6-20 pounds and steelhead weighing 5-10 pounds (Walker 1993). Accurate estimates of historical production are scarce, but as late as 1926 approximately 20 million eggs were collected from 5,000 female Chinook Salmon in the upper Lemhi River by the US Bureau of Commercial Fisheries, representing a total adult escapement of around 10,000 fish (Gebhards 1959). Average Chinook Salmon escapement in the period of 1954-1958 was 1,300 salmon, with a high of 2,558 salmon in 1957 (Gebhards 1959). The Interior Columbia Basin Technical Recovery Team (ICBTRT) classified the intrinsic steelhead population size in the Lemhi River basin as intermediate and the Chinook Salmon population size as very large, which the latter ranked highest among all of the upper Salmon River basin subpopulations (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).

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 and Texas creeks near Leadore, Idaho and flows in a northwesterly direction for 90 km before entering the Salmon River near Salmon, Idaho. The climate of the region is characterized by dry summers and cold winters with most annual precipitation occurring as snowfall. Annual precipitation in the Lemhi River valley is less than 25 cm per year (Gebhards 1959). River hydrology is heavily influenced by spring creeks and ground-water inputs, but also receives considerable input from snowmelt dominated tributaries. The Lemhi River historically had many beaver dam complexes, and an extensive riparian area consisting of willows and cottonwoods (BLM 1998). The relatively broad river valley resulted in a historically complex and anabranching channel structure. The river contained many side channels and braided reaches, as evidenced by the Shoshone-Bannock fishing weirs, which spanned four separate channels when the Lewis and Clark Expedition recorded their observations in August 1805 (Walker 1993). Significant topographic diversity exists in the drainage, with elevations ranging from 1,189 m to 3,395 m. Although most of the basin is public land (82.2%), the majority of the valley and main-stem riparian areas are on private lands (BLM 1998). The basin contains 31 major tributaries, most of which originate in the surrounding mountains, enter the valley across alluvial fans, and naturally lose some discharge to the aquifer.

The basin currently supports a diverse salmonid community including naturally produced Chinook Salmon, steelhead trout, Westslope Cutthroat Trout *O. clarkii lewisi*, Bull Trout *Salvelinus confluentus*, Mountain Whitefish *Prosopium williamsoni*, and non-native Brook Trout *S. fontinalis*. Bull Trout, Westslope Cutthroat Trout, and the non-anadromous component of the *O. mykiss* population exhibit resident and fluvial life histories in the basin. Resident individuals complete all aspects of their life cycle within a particular stream, making limited migrations between seasonal habitats. Conversely, fluvial individuals make extensive migrations and typically spawn within

tributaries, followed by downstream migrations to larger river sections lower in the watershed (i.e., main-stem Lemhi River or Salmon River).

Limiting Factors

Multiple factors in the Lemhi River basin have contributed to the significant decline in fish production from historic conditions. Factors within the watershed, such as fish entrainment, lack of connectivity between the Lemhi River and tributaries, loss of channel form and structure, and reduced flow, have decreased the quality and quantity of spawning and rearing habitat (Gebhards 1958a, 1958b, 1959). Decreed water rights in tributaries often exceed summer stream flows, causing complete dewatering in the lower reaches and loss of habitat connectivity between tributaries and the Lemhi River (Dorratcaque 1986). Some irrigation structures block fish migration regardless of streamflow levels. Of the 31 tributaries to the Lemhi River, all but two (Hayden and Big Springs creeks) have been functionally disconnected from the main-stem river during part or all of the irrigation season (March 15-November 15) since water development began over 100 years ago. In some tributaries, water withdrawals may not entirely dewater a stream reach, but flows are reduced to levels during certain periods of the irrigation season that effectively create thermal or hydraulic barriers to certain life stages of fish. This situation prevents anadromous and resident/fluvial salmonids from accessing high quality habitat for spawning and rearing. Chinook Salmon production is currently limited to the upper main-stem Lemhi River and Hayden Creek, even though several other tributaries historically supported Chinook Salmon production (Walker 1993). Furthermore, juvenile salmonids rearing in the Lemhi River are unable to seek thermal refuge in the tributaries when reductions in main-stem flows impact habitat conditions. Irrigation has also reduced the high-volume flows necessary for maintaining the complexity of stream channels, removing fine sediments, and providing loose gravels for spawning. These issues negatively affect the four important parameters necessary for salmonid persistence: abundance; productivity; spatial distribution, and diversity (McElhany et al. 2000).

Other land-use practices have also had negative effects on fish populations in the Lemhi River. The Idaho Power Company operated a diversion dam about 0.6 km upstream from the mouth of the Lemhi River during 1908-1954 that was only passable to Chinook Salmon and steelhead during periods of high flow (Parkhurst 1941). This barrier, in addition to the many irrigation diversion structures that were impassable after peak flows subsided, likely caused the summer-run Chinook Salmon to become extirpated from the basin, as they would have attempted to enter the Lemhi River in late summer (Bjornn 1978). In the mid-20th century, highway development and flood abatement actions straightened and disrupted large sections (21%) of the main-stem river, which significantly reduced the availability of spawning and rearing habitats (Gebhards 1958b). Land development to support agriculture has further reduced channel complexity and degraded large areas of riparian and floodplain habitats (BLM 1998). Out-of-basin factors, such as commercial harvest and hydroelectric development in the Columbia and Snake River basins, have had major effects on salmon and steelhead, and remain some of the primary factors influencing their viability (NOAA 2016). Despite declines, the Lemhi River basin is still considered a stronghold for Chinook Salmon, steelhead, and Bull Trout because of the intrinsic production potential of the river and its tributaries.

Habitat Restoration Plans

In response to declines in migratory salmonid abundance and production, conservation planning documents and restoration frameworks have been developed, implemented, and refined since the 1950s. Within-basin limiting factors were first addressed as early as the 1950s, when anadromous fish losses to entrainment in irrigation ditches was first quantified (Gebhards 1958a),

and installation of fish screens began in 1958 (Schill 1984). In 1984, the Northwest Power and Conservation Council (NPCC) directed the Bonneville Power Administration to fund the Lemhi River Habitat Improvement Program to address the lack of migration, spawning, and rearing flows in the Lemhi River (Dorratcaque 1986). An extensive basin-wide survey was conducted to compile existing knowledge about habitat conditions, identify potential habitat restoration actions, and evaluate the feasibility to implement measures to address the limiting factors in the basin (Dorratcaque 1986).

Increased attention was given to freshwater habitat improvements as a form of offsite mitigation for Columbia River power system impacts following Endangered Species Act (ESA) listing of spring/summer Snake River Chinook Salmon in 1992 and steelhead in 1997 (NMFS 2008). Specific recovery objectives for the Lemhi River in the Federal Columbia River Power System Biological Opinion include a 7% increase in freshwater productivity of Chinook Salmon and a 3% increase in freshwater productivity of steelhead (NMFS 2008). The current draft recovery plans seek to achieve a population status of "viable," which constitutes an abundance threshold of 2,000 adult Chinook Salmon with 1.34 recruits per spawner and an abundance threshold of 1,000 adult steelhead with 1.14 recruits per spawner (NOAA 2016).

Bull Trout were listed under the ESA in 1999 and the Bull Trout recovery plan identifies the Lemhi River as a core area within the Upper Snake River Recovery Unit (USFWS 2015). The recovery plan has many similarities with 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 maintaining the long-term data sets and continuing monitoring the abundance and distribution of Bull Trout in the Lemhi River basin.

Several collaborative efforts were initiated in the 1990s to identify and prioritize major limiting factors in the Lemhi basin. Irrigation stakeholders in the basin developed a plan in 1992 to improve fish passage in the Lemhi River for Chinook Salmon (LID and WD74 1992). Formation of the Model Watershed Project in 1993 led to the development of the Model Watershed Plan, which identified specific restorative actions to benefit fish in the Lemhi, Pahsimeroi, and East Fork Salmon rivers (ISCC 1995). The scope was later expanded to include the entire Upper Salmon River basin and the entity name was changed to the Upper Salmon Basin Watershed project, and ultimately to its current form, the Upper Salmon Basin Watershed Program (hereafter USBWP). The USBWP (www.modelwatershed.org) provides the forum for the Upper Salmon Basin Technical Team, which consists of a variety of state, federal, and non-governmental organizations that develop, review, rank, and prioritize habitat restoration actions throughout the basin. The Technical Team also makes recommendations to the Upper Salmon Basin Advisory Board, which is comprised of representatives from select state agencies, local conservation and water districts, and private landowners. Both entities are integral to the success of the restoration efforts occurring in the Lemhi basin, as well as the entire Upper Salmon River basin. As part of the early stages of the Model Watershed Project, a stream habitat inventory was completed in the upper Salmon River basin that included a fine-scale documentation of habitat status, and identified reach-specific restoration objectives aimed at addressing specific life-stage limiting factors (Trapani 2002).

These prioritization efforts and planning documents provided a critical foundation for the development of more formal restoration guidance in the Lemhi River basin. In 2002, The State of Idaho, Idaho Department of Water Resources (IDWR), USBWP, Lemhi Irrigation District (LID),

Water District 74 (WD74), National Marine Fisheries Service (NMFS), and U.S. Fish and Wildlife Service (USFWS) pursued a conservation agreement in the Lemhi River basin to address the conflicts between water withdrawals and the viability of ESA-listed fish. These agreements outlined specific limiting factors to address and set forth the goal of developing a long-term conservation agreement. The Lemhi Monitoring Plan was executed in conjunction with the agreement, primarily as implementation monitoring. This plan included temperature monitoring, mapping critical reaches in the lower Lemhi River during irrigation season, and flow monitoring to assess if targets were met and determine if successful fish passage was occurring.

Recommendations from interim conservation agreements were incorporated by the NPCC into the Salmon Subbasin Management Plan (Ecovista 2004). Although the long-term conservation agreement (i.e. Lemhi Habitat Conservation Plan) was never formally adopted as an ESA Section 6 Agreement, many of the recommendations were incorporated into the Memorandum of Agreement (MOA) reached during the Snake River Basin Adjudication (SRBA). The SRBA reviewed all existing water rights in the Snake River basin and placed a moratorium on future development in response to the Nez Perce Tribe filing claims at more than 1,000 locations based on their treaty-reserved fishing rights. The Snake River Water Rights Act of 2004 ratified the MOA established among the Nez Perce Tribe, the State of Idaho, and the United States. As a condition of the settlement, the Act established the Salmon and Clearwater River Basins Habitat Fund, which provided funding opportunities for restoration work in the Lemhi River basin, and developed a restoration and prioritization framework for actions in the basin. The Screening and Habitat Improvement Prioritization for the Upper Salmon Subbasin document prepared by the Upper Salmon Basin Watershed Project Technical Team provides additional restoration planning and project prioritization support (USBWP TT 2005). The most recent Fisheries Management Plan by the Idaho Department of Fish and Game (IDFG) has incorporated many of the SRBA recommendations into the plan and specifically identifies flow enhancement, fish passage improvement, habitat restoration, and screening as major objectives in the Lemhi River basin management (IDFG 2013).

Restoration implementation in the Lemhi River basin is a collaborative process with state, federal, and non-governmental agency partners working together on individual projects, as well as implementing separate, complementary projects that make progress towards recovery goals. Project sponsors in the basin include IDFG, US Forest Service, Bureau of Land Management, USBWP, Trout Unlimited, The Nature Conservancy, Idaho Department of Water Resources, Idaho Governor's Office of Species Conservation, Bureau of Reclamation, Natural Resource Conservation Service, Lemhi County Soil and Water Conservation District, and the Lemhi Regional Land Trust. Major funding sources include Bonneville Power Administration, SRBA Habitat Trust Fund, and Pacific Coastal Salmon Recovery Funds.

Reconnecting productive tributaries to the Lemhi River was the top priority for ongoing habitat conservation efforts in the Lemhi basin. The SRBA established a goal of reconnecting ten tributaries in the first 20 years of the MOA. Projects implemented in select tributaries are designed to remove or modify fish migration barriers, maintain or enhance riparian conditions, improve in-stream habitat, and enhance stream flows through diversion consolidation and irrigation efficiency improvements. Improving connectivity between the Lemhi River and historically-accessible high-quality spawning and rearing habitat is expected to increase anadromous and resident/fluvial salmonid production, growth, and survival.

Restoration is also occurring in the main-stem Lemhi River and associated riparian habitats that have been degraded. Habitat actions are not limited to instream habitat improvements, but also extend to riparian areas and floodplains. Restoration actions range from

development of side channels and lateral habitats to complete rehabilitation and relocation of large river segments. Currently, all major irrigation ditches on the main-stem Lemhi River are screened and have bypass systems to prevent entrainment of migrating fish (IDFG 2013). A minimum flow agreement was first implemented in the lower Lemhi River in 2000 to maintain surface water connection in a previously dewatered reach immediately downstream of the L6 diversion, which is a major irrigation diversion serving several senior water rights. The initial agreement maintained 10 cfs to aid in downstream migrations of juveniles and included pulses of 35 cfs to aid in adult upstream migrations. In 2002, a new agreement was implemented through permanent easements and annual irrigation agreements at the L6 diversion, which achieved a minimum of 35 cfs for 80% of the 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.

Fish Population Monitoring

Fish population surveys and research have occurred in the Lemhi basin since the early 1900s, which provide critical long-term abundance trends and the ability to view current status in a historical context. Early monitoring activities in the Lemhi River basin ranged from spawning investigations (Parkhurst 1941) to quantitative evaluations of habitat degradation impacts (Gebhards 1958b) and life history investigations (Bjornn 1969). IDFG began conducting Chinook Salmon redd counts in the Lemhi River in 1947 (standardized after 1956) and these data served as important indicators of adult abundance during ESA listing and have been integral to every status review thereafter.

Following a major flood in 1957, an extensive assessment of the entire Lemhi River was completed and established one of the first quantitative linkages between habitat degradation and fish population decreases in the Lemhi River basin (Gebhards 1958b). To aid in future flood control, large sections of the river were deepened and straightened by bulldozing from the middle of the channel to the banks and using the substrate to create levees ranging in height from 2 to 15 feet. Gebhards (1958b) estimated that 17% of the entire length of the Lemhi River had been channelized and deepened in one year. The most extensive disruptions occurred in the lower 13 miles of the river between the mouth and the community of Baker, where 35% of the channel had been straightened. The direct effects of channel modification during 1957 were estimated at 500,000 eggs killed, but the long-lasting effects from channel alterations reduced rearing habiat and casued over 12% of the prime Chinook Salmon spawning areas to be unsuitable for spawning activity.

Intensive studies of fish production were conducted in the Lemhi River from 1962 to 1975 (Bjornn 1978). A research facility was constructed in the upper Lemhi River just upstream of the mouth of Hayden Creek in 1964 that included a weir to capture migrating adult salmonids and a trap to capture juveniles migrating downstream past the location. Operation of this facility established an extensive long-term data set on Chinook Salmon and steelhead escapement, juvenile emigrant abundance, upper Lemhi River productivity, and smolt-to-adult return (SAR) rates.

The Lemhi River was used as a reference system in a long-term, statewide study of hatchery supplementation of Chinook Salmon populations (Venditti et al. 2015). The Idaho Supplementation Studies (ISS) project began in 1992, and estimated annual abundance trends of juvenile Chinook Salmon migrating past the old Bjornn weir location (Bowles and Leitzinger 1991). The ISS project also implemented more intensive, multi-pass grounds counts of Chinook

Salmon spawning grounds in the upper Lemhi River to provide more precise estimates of salmon production.

Additional fisheries investigations increased in the tributary habitats as the IDFG Screen Program conducted extensive tributary surveys throughout the late 1990s and early 2000s (e.g., Murphy and Horsmon 2003, 2004; Murphy and Yanke 2003; Warren and Bliss 2006). These surveys are integral to planning and prioritization because they contain diversion and barrier inventories and comprehensive fisheries information. They also provide robust baseline fish data that can be used as pretreatment data for many of the focal areas in the Lemhi River basin.

The Integrated Status and Effectiveness Monitoring Program (ISEMP) began operations in 2009 in the Lemhi watershed because of the substantial number of planned restoration activities and the wealth of relevant fisheries and habitat information. The ISEMP project was implemented across the Columbia River basin to develop standardized monitoring, sampling, and analytical tools in target watersheds that can be used across the entire Columbia River basin to understand the benefits of tributary restoration actions (ISEMP and CHaMP 2017a). Although the ISEMP project has broad-reaching goals across the Columbia River basin, the study design developed specifically for the Lemhi River basin addresses the following objectives (QCI 2005):

1. A watershed model that evaluates productivity and carrying capacity by life cycle stage as a function of habitat availability and quality, and then simulates expected life-stage specific benefits from increased habitat availability or quality.

2. Reach-specific empirical measures of juvenile productivity, survival, and condition to determine whether tributary reconnection has provided high quality habitat that benefits fish vital rates (survival, growth, etc.).

3. Measures of the movement and distribution of anadromous and resident fish to address the following questions:
   a) Are anadromous fish utilizing newly available habitat?
   b) Have reconnections changed the distribution and connectivity of resident fish?

Intensively Monitored Watershed Project Overview

In 2007, a comprehensive effectiveness monitoring strategy was implemented by IDFG under the Intensively Monitored Watershed (IMW) program to evaluate fish response to habitat restoration actions in the Lemhi River basin. The level of monitoring proposed for the Lemhi basin made the ISEMP project a logical partnership, which enabled the pooling of resources to develop a more intensive and larger scale IMW program, and meet the mutual objectives shared between the two projects. 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.

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 watershed, tributary, and project 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. Because of the types of limiting factors being addressed in the Lemhi River basin, as well as the strategic implementation of restorative actions, we have developed the following hypotheses associated with the primary restoration strategies:

1. Tributary reconnections should increase the amount of spawning and rearing habitat accessible to migratory salmonids. We consider tributaries functionally connected when fish have the ability to 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).

2. Flow improvements in the main-stem river, as a result of tributary reconnections and main-stem water conservation projects, should reduce passage impediments for adult salmonids. We expect to see successful upstream migrations of adult Chinook Salmon when the lower Lemhi River minimum flow agreement is met. The efficacy of the minimum flow agreement will be assessed using radiotelemetry equipment to monitor radio-tagged adult Chinook Salmon to assess fine-scale movements from entry to the Lemhi River until they reach their spawning locations.

3. The combination of tributary reconnections and main-stem upper Lemhi River habitat improvement projects should improve rearing conditions during all seasons, but in particular, summer and autumn when irrigation impacts and other passage barriers would have rendered those habitats inaccessible. In addition to the tributary-specific responses (see above), we expect to see increased productivity measured in terms of the number 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 upper basin, we expect them to use those habitats and not emigrate until the presmolt or smolt life stage. An increase in rearing duration by fry and parr or higher survival rates should translate into an increase in presmolt and smolt abundance measured at the upper Lemhi River rotary screw trap.

In this report, we present the results of monitoring activities completed from 2008 through 2016. We organize report sections by life-stage to link methods and results to the life cycle monitoring approach. To date, primary restoration efforts have focused on reconnecting six priority tributaries (Big Timber, Canyon, Hawley, and Little Springs creeks in the upper Lemhi watershed; and Kenney and Bohannon creeks in the lower Lemhi watershed), and those are the focus of the current synthesis of IMW results. We discuss the fish and habitat results in the context

of restoration actions completed through the monitoring period. 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 watershed, tributary, and project scales. This hierarchy enables elucidation of the effects of specific restoration treatment types. The sampling framework also provides estimates of life-stage specific vital rates (e.g. survival and growth) by habitat type to parameterize the Lemhi River life cycle model (QCI 2005). Watershed-level monitoring occurs within three key areas: the upper Lemhi River main-stem, lower Lemhi River main-stem, and Hayden Creek. A critical element of the monitoring framework is the use of Hayden Creek as a reference tributary in the basin. It is the larger of the two tributaries that maintained a perennial connection with the Lemhi River following agricultural development in the basin. 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 basin. The spatially explicit, quantitative monitoring of adult and juvenile abundance allows comparisons of productivity and abundance among the key areas.

Because tributary reconnections are a major focus of the restoration actions in the basin, intensive tributary monitoring occurs within the six priority candidate tributaries for reconnection identified in the SRBA Lemhi River Basin Habitat Action Framework (Figure 1.2). Electrofishing and habitat surveys were conducted in Big Timber, Canyon, Hawley, Little Springs, Kenney, and Bohannon creeks to provide pre-reconnection fish information, and document habitat quality and quantity to understand changes in capacity associated with reconnection efforts. The number of years of pretreatment data available per tributary varies because reconnection efforts began in different years and some tributaries are still in various phases of restoration (for complete description of tributaries and habitat actions, see Appendix A).

In addition to watershed- and tributary-scale evaluations of fish and habitat response to restoration actions, some monitoring activities conducted by the Lemhi IMW are designed to assess efficacy of specific restoration actions and identify specific reaches where restoration actions should be directed. Project-level effectiveness monitoring of a multi-phase project occurred in the lower main-stem Lemhi River 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 (Appendix C). A multi-year radiotelemetry study was implemented to understand fine-scale movement patterns of adult Chinook Salmon to assess if passage barriers were impeding movements in the main-stem Lemhi River and Hayden Creek.

Habitat Monitoring

Habitat monitoring in the Lemhi River basin was conducted to estimate baseline habitat conditions prior to restoration efforts, document changes in habitat associated with restoration efforts, and develop fish-habitat relationships. Quantitative Consultants, Inc. (QCI) was responsible for implementing the habitat monitoring project in the Lemhi River basin under the ISEMP program. Habitat monitoring began in 2009 using the U.S. Forest Service

PACFISH/INFISH Biological Opinion (PIBO) protocol, but the project adopted the Columbia Habitat Monitoring Program (CHaMP) protocol in 2011 to utilize a sampling methodology that measures habitat characteristics most relevant to salmon and steelhead production (Bouwes 2011). Although CHaMP objectives and methodologies differ from PIBO, there are many similarities that enable crosswalking of metrics to facilitate collaboration among regional monitoring programs (ISEMP and CHaMP 2015). The primary goal of CHaMP is to measure the quantity and quality of anadromous salmonid habitat and document changes through time (Bouwes 2011). The monitoring program evaluated habitat responses at multiple scales, ranging from individual habitat units within reaches up to the basin scale. In addition to basin-scale habitat trends, the program was developed to evaluate habitat responses to specific land management, restoration, and conservation actions in the Columbia River basin (CHaMP 2016).

The habitat surveys were distributed in the Lemhi River basin using a Generalized Random Tessellation Stratified (GRTS) design (Stevens and Olsen 2004). The spatially-balanced GRTS sampling design distributes sites along the stream network, enabling the collection of representative and unbiased site-based data that can be scaled up to estimate status and trends at the basin scale (Bouwes 2011). The habitat monitoring sites overlap fish monitoring sites to enable the development of fish-habitat relationships to populate a basin-scale life cycle model (ISEMP and CHaMP 2015). Many stream characteristics were measured during CHaMP surveys, but in general, all are related to the categories of stream temperature, stream production, and stream morphology because of their influence on salmonid performance (Bouwes 2011).

Information collected by CHaMP surveys are used to calculate a variety of metrics, but the ones contained in this report are limited to those most relevant to the restoration strategies enacted in the Lemhi River basin. These include sinuosity, slower water habitat frequency, fish cover, substrate embeddedness, slow water habitat, large woody debris frequency, wetted volume, wetted area, and the 7-day average of daily maximum temperature (see Bouwes 2011 for a detailed description of all metrics calculated by CHaMP). We used 95% confidence intervals to make statistical inferences about annual changes in watershed-level metrics.

Fish Monitoring

Fish sampling activities in the Lemhi River basin monitor the effectiveness of ongoing habitat conservation actions in the main-stem Lemhi River and tributaries of the Lemhi River that are prioritized for reconnection. The monitoring framework was developed to evaluate watershed-scale fishery responses, but also to understand how fish are responding to individual habitat projects and specific treatment types. Monitoring activities include tagging with passive integrated transponders (PIT tags), operating PIT-tag detector arrays and rotary screw traps, as well as electrofishing, radio-tagging and tracking, and salmonid redd counts.

Infrastructure—Three rotary screw traps (RST) are operated in the Lemhi River basin to estimate juvenile salmon and steelhead production with useful precision. A subsample of fish captured at the screw traps are implanted with PIT tags to estimate survival rates within the basin by life stage and rearing area (watershed), as well as estimate survival to Lower Granite Dam (LGR). The ISS project began operating the upper Lemhi (LEMHIW) RST in 1992 (Walters et al. 1999). At the end of the ISS project in 2014, the IMW project took over operations to maintain the long-term data set and the spatial distribution of screw traps, which is integral to the monitoring study design. The lower Lemhi (L3AO) RST was installed in 2004 near the L3A diversion and operated in that location through 2011. It was inoperable in 2012, but ISEMP personnel began operating it in 2013 at the L3A0 diversion, approximately 2 km downstream of the previous location. The purpose of this trap is to estimate total production in the Lemhi River basin. The

Hayden Creek (HAYDNC) RST is located 1 km upstream of the confluence with the Lemhi River. It began operating in September 2006. This screw trap enables us to treat Hayden Creek as a reference for investigations of changes in upper Lemhi River production.

Tandem PIT-tag arrays have been 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 PIT-tag array installations were in locations associated with existing RST infrastructure to provide juvenile survival and adult escapement information within the life cycle monitoring design that enables adult abundance to be linked to specific brood year juvenile production (Figure 1.3). Additional 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).

PIT-tag arrays in tributaries were installed as close to the mouth as possible to prevent undocumented spawning from occurring downstream of the array location (Table 1.1). However, not all tributaries had ideal locations near the mouths to provide sufficient solar or grid power to operate arrays, so distances between arrays and mouths of tributaries ranged from 0.04 km-1.3 km (mean = 0.45 km). All detections were uploaded to the PIT Tag Information System (PTAGIS, www.ptagis.org). As the ISEMP project matured, PIT-tag arrays were also installed in many tributaries to account for movements and production in the tributaries that were not the main focus of restoration efforts (ISEMP and CHaMP 2015). As the IMW and ISEMP projects modified sampling designs and goals, some PIT-tag arrays were removed as they were deemed superfluous. As restoration has progressed in certain tributaries, some arrays have been relocated to improve project effectiveness monitoring. PIT-tag arrays were also installed at several locations within some tributaries to identify responses within specific reaches and provide finer resolution migration data to restoration practitioners (i.e., Big Timber Creek). PIT array function (i.e. generating interrogations) is facilitated by tagging at key locations, as explained further in this document.

Adult Chinook Salmon and Steelhead Escapement—Escapement of Chinook Salmon and steelhead from Lower Granite Dam to the Lemhi River basin is estimated using a two-step process (See et al. 2016). The analysis partitions escapement over Lower Granite Dam into basin-specific estimates using a hierarchical patch-occupancy model. Counts of Chinook Salmon and steelhead ascending the LGR fish ladder are observed through a window on a daily basis for 10-16 hours per day. Additionally, an adult trap captures a subsample of fish ascending the ladder, and those individuals are sampled for genetic samples and implanted with PIT tags. Weekly PIT-tag interrogations at LGR are used to estimate the number of fish ascending the ladder while the windows are not being observed. The window counts are adjusted by this rate to estimate the total weekly escapement over LGR. The hierarchical patch-occupancy model is used to estimate transition and detection probabilities at PIT-tag arrays and weirs upstream of LGR using PIT-tag detection histories of fish tagged at the LGR trap and previously-tagged fish that ascended the fish ladder. The transition probabilities from LGR to the Lemhi River are multiplied by the total escapement over LGR to estimate Lemhi River basin escapement estimates.

Spawning Surveys—Multi-pass spawning ground surveys were conducted for target species in upper Lemhi River and selected tributaries following standard IDFG redd survey protocols (Hassemer 1993). 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. Chinook Salmon redd counts were conducted in 2014-2015 in Kenney and Bohannon creeks; in 2011 and 2013-2016 in Big Timber, Canyon, and Little Springs creeks; and in 2008-2016 in the upper Lemhi River and Hayden Creek. Steelhead redd

counts were conducted in 2013-2017 in Bohannon, Big Timber, and Canyon creeks; in 2008, 2010-2011, 2013-2014, and 2017 in Kenney Creek; in 2011 and 2013-2017 in Little Springs Creek; and in 2011 and 2013-2015 in Hayden Creek. Chinook Salmon surveys were consistent across years because they were conducted during periods of base flow in August and September. Steelhead spawning surveys conducted in April and May were sparse across years due to spring runoff conditions that often created poor visibility conditions. All survey data were stored in the IDFG Spawning Ground Survey database.

Adult Chinook Salmon Migration Characteristics—We used radiotelemetry equipment to investigate fine-scale movement patterns of returning adult Chinook Salmon and identify passage barriers in the Lemhi River and tributaries to help prioritize restoration actions (Biggs 2014). A proportion of Lemhi-origin adult Chinook Salmon PIT-tagged as juveniles at RSTs or electrofishing sites in the Lemhi River basin were diverted using sort-by-code (SbyC) at LGR and orally implanted with radio tags at LGR adult trap during migration years 2008-2016. The number of radio-tagged adults ranged from 1 to 18 fish per year. Tag deployment was spread out over the entire annual run to prevent bias of movement patterns associated with fish returning at different times.

Tracking commenced when fish migrated upstream near the Corn Creek boat ramp on the Salmon River (111 km downstream of the mouth of the Lemhi River), where personnel could use vehicle-mounted telemetry equipment. Tracking usually occurred daily, but some fish had more than a week duration between subsequent relocations because of failing to detect every tag during some surveys and reduced tracking effort in some years. All tracking occurred from the road, but some radio tags were recovered from carcasses on foot. All points recorded from the road were later repositioned to points along the stream by manually selecting the nearest point on the stream perpendicular to the road detection location. Tag retention was estimated as the number of fish detected on the PIT-tag arrays in the Lemhi River basin that were not relocated with telemetry equipment. Tag retention varied across years, with as much as 50% of a tagging cohort not being detected with telemetry equipment after release.

Summer Juvenile Salmonid Standing Stock and Distribution—Electrofishing surveys were conducted in the six priority tributaries to estimate juvenile salmonid standing stock during the summer, investigate changes in distribution associated with restoration actions, and deploy PIT tags to document tributary-specific survival rates and compare downstream performance among fish rearing in different areas (QCI 2005; see Appendix B for sampling details). We refer to tributary-specific abundance as standing stock to differentiate between emigrant abundance estimated at rotary screw traps (see below). Electrofishing surveys were also conducted in the main-stem Lemhi River to implant tags in fish for estimating rearing capacities and reach-based survival in the main-stem river (ISEMP and CHaMP 2017a), as well as monitoring fish moving from the main-stem river into tributaries. The main-stem surveys also provided the opportunity for reach-level, project-specific effectiveness monitoring (see Appendix C).

Electrofishing surveys started in most tributaries in 2009, but some limited sampling occurred in 2008 in Big Timber and Kenney creeks. Bohannon Creek was not surveyed until 2010. Electrofishing sites were distributed throughout tributaries using a GRTS design (ISEMP and CHaMP 2015). Sampling occurred at the same locations surveyed by the habitat monitoring program. The lengths of sites were established by measuring the bankfull width and multiplying that value by 20 (Bouwes 2011), resulting in site lengths from 40 m to 300 m.

In 2013, the electrofishing survey design was changed to a continuous sampling framework in which multiple kilometers of each tributary were surveyed using mark-recapture

electrofishing techniques to increase precision of abundance estimates and address the patchy distribution and overall low density of target species (ISEMP and CHaMP 2016a). The increased sampling coverage associated with the continuous surveys helps adhere to the closure assumptions of mark-recapture techniques, increases the number of PIT tags implanted in fish, and provides much finer resolution for species distributions compared to site-based sampling. Mark-recapture electrofishing surveys were conducted in each tributary 2013-2016, except in Hayden Creek where a preliminary shift to mark-recapture surveys occurred in 2011 (Appendix B). During the years of site-based sampling, the proportion of stream length sampled within each tributary ranged from 1.0% - 7.6%, but when the mark-recapture sampling framework was implemented, the proportion of stream length sampled within each tributary increased considerably, and ranged from 29.5% - 99.4% (Appendix B).

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 3m. 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 PIT-tagged salmonids at a rate of 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 the low numbers and limited distribution of Cutthroat Trout in sampled areas. Prior to 2013, fish locations were associated with an individual site, but in 2013 within the mark-recapture framework, 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. Therefore, fish distributions have much finer resolution during 2013-2016. We stored survey data in the IDFG Stream 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 undoubtedly 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. Although Bear Valley Creek (a tributary of Hayden Creek) was sampled in 2009 and 2011-2013, we limited the Hayden Creek standing stock estimates to main-stem Hayden Creek because densities could not be extrapolated in Bear Valley Creek during the non-sampled years. We did not sample Bear Valley Creek after 2013 because we increased sampling effort within main-stem Hayden Creek where the majority of Chinook Salmon and steelhead production occurs. However, we included the Bear Valley Creek data in the distribution assessment.

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. 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 main-stem interrogation query revealed that only one individual from the Lemhi basin larger than this threshold (390 mm) has ever been detected on a main-stem Columbia River interrogation site. Thus, this maximum size threshold is sufficient for including the majority of the probable anadromous component of the *O. mykiss* population encountered during electrofishing surveys.

Movements—Interrogations at tributary PIT-tag antenna arrays were used to assess movements into and out of tributaries to understand how reconnection efforts affected the ability of fish to move freely in the basin. To investigate fish moving out of tributaries, the PIT-tag codes of fish tagged in electrofishing surveys were queried on PTAGIS to determine if they were detected on the array at the mouth of the tributaries they were tagged in. We included all tags implanted during electrofishing surveys, not just those implanted during the mark-recapture sampling events that were used to estimate abundance in the tributaries. Conversely, to assess movements into tributaries, the PIT-tag arrays at the mouths of priority tributaries were queried for all detections generated by fish not tagged in 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). For these queries we included fish marked during electrofishing surveys and at rotary screw traps. For the investigation of fish entering tributaries, all arrays on Big Timber Creek were included in the query to understand if fish had the ability to migrate freely through all sections of the stream. We only considered the first interrogation record of each fish at an array. For example, if a fish was detected immigrating into a tributary on October 1 and then detected emigrating on October 15, we only included the October 1 record in our summary.

Juvenile Chinook Salmon and Steelhead Emigration—Emigration from the key areas was estimated from data collected with the three RSTs. Rotary screw traps operated according to protocols established for anadromous emigrant monitoring by IDFG (Apperson et al. 2016). All traps were checked daily when in operation. We anesthetized captured fish with MS-222 or Aqui-S and scanned for PIT tags, measured weights to the nearest 0.1 g, and measured fork lengths to the nearest 1 mm. A subsample of fish were implanted with PIT tags. Chinook Salmon and steelhead 60-69mm were tagged with 9-mm PIT tags and Chinook Salmon and steelhead ≥70mm were tagged with 12-mm PIT tags. We released up to 50 PIT-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 JTRAP database and all PIT-tag records were uploaded to PTAGIS.

Abundance estimates of juvenile Chinook Salmon and steelhead emigrating past rotary screw traps 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 2,000 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 time 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). From July 1 through August 31, the catch during the summer time period 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.

Life stages were summed into cohorts by brood year, which is the year that the fish were spawned. For example, the total abundance estimate for brood year 2014 is calculated as the sum of age-0 fry caught in the spring period during 2015, age-0 parr caught during the summer of 2015, age-0 parr caught during the fall of 2015, and age-1 smolts captured during the spring of 2016. Although screw traps captured sufficient numbers of fry in some years to estimate abundance, a few years had zero fry captured, or no recaptured fry. Therefore, trend investigations for Chinook Salmon emigrant abundance exclude the fry life stage from brood year totals. For steelhead abundance estimation, the trapping season was divided into two strata: 1) the start of trapping through May 31; 2) June 1 through the end of trapping season. The abundance estimates from the two trapping periods were summed into cohorts by trapping year.

Juvenile Chinook Salmon Survival—Survival rates were estimated based on detections of PIT-tagged fish. Survival to Lower Granite Dam of Chinook Salmon from the RSTs by fall parr and age-1 smolts were estimated following the protocol used by Apperson et al. (2016). Chinook Salmon survival rates to LGR were estimated 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 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 the exhibits resident or fluvial life history strategies.

Watershed-specific and seasonal survival rates in the Lemhi River basin 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 rotary screw traps and roving electrofishing surveys were included in the analyses, as were recapture events at rotary screw traps and live-resights at PIT-tag arrays (ISEMP and CHaMP 2017a).

Survival rates were estimated for multiple combinations of seasons and watershed reaches to identify potential limiting factors temporally and spatially. 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. The Lemhi River population was partitioned into the upper Lemhi River and Hayden Creek subpopulations for survival analyses because few redds are observed in the lower Lemhi River. To estimate cohort-based survival, the Lemhi River basin was divided into three strata: 1) the main-stem upper Lemhi River above the LemhiW RST and LRW array; 2) the lower Lemhi River between the LemhiW RST/LRW array and the L3AO RST/LLR array; and 3) Hayden Creek above the HAYDNC RST and HYC PIT-tag array (ISEMP and CHaMP 2017a). Survival rates within the

reaches above the LemhiW RST and HAYDNC RST were based on the fish releases during summer electrofishing surveys. Therefore, 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 releases at the screw traps 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 either the upper Lemhi River stratum or Hayden Creek stratum before December 31, or were released at the LemhiW RST or HAYDNC RST in the fall.

Data Analysis

Habitat Capacity—The life cycle model developed for the Lemhi River basin incorporates fish and habitat data collected from the ISEMP/CHaMP and IMW survey efforts. The model relies on a combination of empirical and literature based estimates to parametrize survival and capacity values for egg, fry, summer parr, fall parr, smolt, adult, and spawner life stages (ISEMP and CHaMP 2015). Parameter estimates for egg, fry, and adult life stages were derived from the literature (see ISEMP and CHaMP 2015 for detailed description of sources). Estimates for the summer parr life stage were measured empirically from summer electrofishing surveys. Estimates for fall parr and smolt life stages were measured empirically at RSTs. Primary objectives of the life cycle modeling include identifying life-stage-specific limiting factors, understanding effectiveness of habitat restoration on fish survival and productivity, and predicting if the level of restoration efforts will be sufficient to achieve recovery goals (ISEMP and CHaMP 2016b). Summer parr capacity values were estimated using Quantile Regression Forest (QRF) models based on fish abundance estimates at CHaMP survey sites and twelve corresponding habitat metrics to estimate capacity at the reach scale (ISEMP and CHaMP 2016b). The QRF models the 90th quantile of fish abundance at 116 individual CHaMP sites to serve as a proxy for carrying capacity (ISEMP and CHaMP 2017b). To estimate summer rearing capacities of each priority tributary, we multiplied the channel-type capacity estimate from the QRF model (see ISEMP and CHaMP 2017b) by the cumulative length of each channel unit type within tributaries based on the sampling frame hydrography layer in the Lemhi CHaMP Geodatabase File (available at http://isemp.org/products/explore-data/spatial).

Chinook Salmon Productivity—The restoration efforts in upper Lemhi River tributaries (Big Timber, Hawley, Little Springs, and Canyon creeks) should provide important summer 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 smolts emigrating past the LemhiW RST was compared to productivity measured at 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. Although the spawning ground transects in the upper Lemhi River do not cover the entire area above the RST, they encompass the majority of all spawning activity, so negative bias associated with redd abundance should be minimal.

We set the before and after periods for the analysis as follows. Although initial reconnection efforts opened up some summer rearing habitat as early as 2009 (i.e., Big Timber

Creek), electrofishing surveys in tributaries documented the presence of Chinook Salmon parr using newly available habitats in 2011, with more consistent use after 2012. Therefore, 2011 was chosen as the cutoff year for the "post-treatment" period. As such, productivity comparisons were made with brood year 2010 because those individuals would have been using the reconnected habitats as parr in 2011.

We conducted the statistical comparison in a linear model framework using the equation:

$$PL = PH + T,$$

where *PL* is productivity of the upper Lemhi River subpopulation, *PH* is productivity of the Hayden Creek subpopulation and *T* is time 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 the time period variable had a p-value less 0.05.

RESULTS

Habitat Monitoring

Watershed-level habitat was relatively stable, but a few metrics showed statistically significant variation (QCI, unpublished data; Figure 1.4). The proportion of total stream area with fish cover remained relatively stable throughout the study period, ranging from 18.8% to 23.8%. Frequency of large woody debris was fairly constant during 2011-2014, but there was a considerable increase in 2015. The average sinuosity of the Lemhi River did not change from 2011 through 2015, but the frequency of slow water habitat in the basin almost doubled from 2011 to 2014. Although the total percent of slow water habitat in the basin did not change significantly from 2011 through 2015, the frequency of slow water habitat was significantly higher in 2015 than in 2011. The 7-day average of daily maximum temperature was lowest in 2012 and highest in 2015, but it only ranged from 16.0°C – 17.1°C. Average substrate embeddedness in the Lemhi River basin was highest in 2013 at 11.6% and lowest in 2015 at 4.9%. The embeddedness was significantly lower in 2015 than it was in 2012 and 2013. At the largest scale, aquatic habitat quality within the Lemhi River basin did not change much during 2011-2015. However, the cumulative restoration actions completed through 2016 represent a 22% increase in wetted stream area and a 19% increase in pool habitat compared to pre-restoration conditions (ISEMP and CHaMP 2017b).

Fish Monitoring

Adult Chinook Salmon and Steelhead Escapement—Chinook Salmon escapement estimates based on PIT-tag detections varied by almost an order of magnitude (QCI, unpublished data; Figure 1.5). The upper Lemhi River spawning aggregate had higher estimates than Hayden Creek 2010-2014; but, in 2015, Hayden Creek had an estimated 351 adults (SE = 54.5) return compared to the 332 adults (SE = 52.3) in the upper Lemhi River (QCI, unpublished data; Figure 1.6). The proportion of total escapement to the Lemhi River basin that migrated into Hayden Creek ranged from 15.2% in 2014 to 48.6% in 2015, with an average proportion of 29.7% from 2010 through 2015. Although the majority of fish spawn in Hayden Creek or the main-stem Lemhi River above the Hayden Creek confluence, some Chinook Salmon spawn downstream of this area. The proportion of total escaping Chinook Salmon that spawned in the Lemhi River downstream of the

Hayden Creek confluence ranged from 0% in 2010 and 2014 to 19.3% in 2012, with a 2010-2015 average of 6.9%.

Steelhead returns were less variable through time than Chinook Salmon (Figure 1.5). Escapement estimates ranged from 268 adults (SE = 28.4) in 2014 to 419 adults (SE = 40) in 2010. No clear trend was observed from 2010-2016. Tributary specific escapement estimates revealed most steelhead spawned in Hayden Creek, as well as tributaries below the mouth of Hayden Creek (QCI, unpublished data; Figure 1.7). Only a small proportion of total steelhead escaping to the basin spawned in the Lemhi River above the LRW PIT-tag array. The proportion of total escapement that spawned in the basin above LRW ranged from 6.4% in 2010 to 23.9% in 2011, with a 2010-2016 average of 13.9%.

Spawning Surveys—Chinook Salmon redd counts in the upper Lemhi River and Hayden Creek varied considerably from 2001 through 2016 (Figure 1.8). Spawning activity in Hayden Creek was substantial, accounting for 18.7%-50.6% of the total redds in the Lemhi River basin. For the years that PIT-tag based escapement estimates were available (2010-2015), the redd counts were very consistent with escapement estimates.

Redd counts provide a longer time series of spawning activity and show a similar trend during the years that PIT-tag based escapement estimates are available (Figure 1.8). Since 2011, 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 in 2004 with 40 redds. In general, the number of redds declined from 2001 to 2007 and then increased through 2015 when 310 redds were observed. Redd counts were much lower in 2016 with 166 redds, but still above the 2001-2015 average of 151 redds.

Extent of salmon spawning locations did not increase substantially through time. In years with more spawning activity, the density of redds increased within certain areas of the Lemhi River (Figure 1.9) and Hayden Creek (Figure 1.10), rather than spawning occurring in newly occupied reaches. No Chinook Salmon redds have been documented in any of the priority tributaries. Other than Hayden Creek, the only tributary where Chinook Salmon redds have been documented is Big Springs Creek (less than 10 redds; IDFG, unpublished data).

Steelhead spawning ground surveys did not reveal any clear trends in adult abundance in the priority tributaries or Hayden Creek. We did not observe any redds in Bohannon Creek during the first year of redd counts in 2013 (Figure 1.11). Redd counts were highest in 2014 with 33 redds and continued to decline through the 2017 spawning season to 2 redds. We did not survey Kenney Creek consistently through the years, but available redd counts do not indicate a trend except that redd counts were lower after 2008 (Figure 1.12). We documented the highest spawning activity in 2008 with 22 redds and the second highest redd count in 2014 with 10 redds. Similar to Bohannon Creek, the spawning activity decreased after 2014 and we documented one redd in 2016 and no redds in 2017. No steelhead redds were observed in Big Timber or Canyon creeks during any years. One steelhead redd was observed in Little Springs Creek during 2017. Steelhead redds were most broadly distributed in Hayden Creek, with the upstream extent of spawning occurring higher than within Bohannon or Kenney creeks (Figure 1.13). Redd distribution in Hayden Creek illustrates what might be expected of a fully utilized tributary.

Adult Chinook Salmon Migration Characteristics—Upon entry to the Lemhi River, fish made extensive daily movements until early to mid-July (Figure 1.14), at which point they reached their summer holding locations. Holding areas were in Hayden Creek, the upper Lemhi River (above the Hayden Creek confluence), and in the Lemhi River within 12 km of the Hayden Creek

confluence. No radio-tagged fish were detected holding in locations of the Lemhi River downstream of the Agency Creek confluence. Fish generally occupied holding locations that were near spawning locations and made limited movements toward the end of summer. However, a few fish in some years made daily movements of up to 3 km/d between 15 August and 1 September. All salmon that used holding areas in Hayden Creek stayed there to spawn, whereas salmon that used holding areas in the Lemhi River either spawned in Hayden Creek or the upper Lemhi River. These results indicate that no passage barriers were present in the main-stem Lemhi River or Hayden Creek.

Summer Juvenile Salmonid Standing Stock and Distribution—Juvenile steelhead were the most abundant and widely distributed species in all tributaries. Standing stock was variable across years within streams, such that few overall trends were apparent (Figure 1.15). Hawley Big Timber, and Canyon creeks exhibited increasing trends over the entire study period. However, sampling designs and effort were much different during 2013-2016 (Appendix B), so these trends should be viewed with caution. When considering this period separate from 2009-2012, Canyon Creek and Hawley creeks still exhibit positive trends, but an overall trend in Big Timber creek becomes less apparent. Standing stock was generally greatest in Hayden Creek (Figure 1.16), but Big Timber Creek had similar levels of standing stock in most years (Figure 1.15, top left panel). Distributions of steelhead within streams remained fairly consistent across years (Appendix D). Canyon Creek exemplifies the steelhead distribution pattern we observed in all priority tributaries, with juvenile steelhead encountered in the majority of sampled areas from 2009 through 2016 (Figure 1.17). This pattern was consistent with annual distributions observed within Hayden Creek (Figure 1.18).

Juvenile Chinook Salmon were not observed in any of the priority tributaries until 2011 when they were captured in Canyon Creek (Figure 1.19). By 2013, Chinook Salmon were present in all of the priority tributaries in the upper Lemhi River except Hawley Creek (Appendix D). Chinook Salmon were not observed in lower Lemhi River tributaries until 2014 in Bohannon Creek and 2016 in Kenney Creek. Standing stock in Hayden Creek was an order of magnitude higher than all the priority tributaries (Figure 1.20). Although standing stock exhibited considerable inter-annual variability, upper Lemhi River tributaries had consistently higher standing stocks than lower Lemhi River tributaries. In priority tributaries where Chinook Salmon were present, as standing stock increased through time, so did the upstream extent of distribution (as illustrated in Canyon Creek, Figure 1.21). However, Chinook Salmon were not as broadly distributed in the priority tributaries as in Hayden Creek (Figure 1.22).

Bull trout were observed in all priority tributaries and were most abundant in Big Timber, Bohannon, Hawley, and Kenney creeks (Figure 1.23). Standing stock appeared to be increasing in Big Timber Creek, decreasing in Bohannon and Canyon creeks, and generally stable in Hawley and Kenney creeks. However, Bull Trout data were sparser than steelhead data in years prior to 2013, so trends should be viewed with caution. Standing stock was generally highest in Hayden Creek (Figure 1.24), but standing stock in Big Timber Creek was equal to or exceeded Hayden Creek in some years. Bull trout distributions were similar across years within the tributaries, and did not exhibit much expansion or contraction. In most tributaries, Bull Trout were primarily distributed through the upper sites (Appendix D), except in Kenney Creek, where Bull Trout were located throughout sites from the mouth to the upstream survey extent (Figure 1.25). We observed similar distribution patterns in Hayden Creek (Figure 1.26).

Movements—In tributaries where substantial numbers of Chinook Salmon were PIT-tagged, the rate at which individuals resided until the following year varied considerably (Appendix E). Little Springs Creek had a consistently high proportion of Chinook Salmon that did not

emigrate until the following year after tagging (average = 17.6%, SD = 16.2%), suggesting that those fish were using the tributary for winter habitat. Canyon Creek also had brood years of Chinook Salmon that had a high proportion migrate out the following year, with average proportion of 26.0% (SD = 18.2%), whereas Big Timber Creek had no Chinook Salmon that were detected migrating past the array in the following year. Proportions of Chinook Salmon detected leaving Hayden Creek in the following year were never more than 7.0%, suggesting that most fish were emigrating out of Hayden Creek as summer or fall parr before winter (Table 1.2).

Steelhead were detected emigrating from all of the tributaries, but the proportions of tagged individuals emigrating were generally highest in the tributaries that have been functionally reconnected (Appendix E). Little Springs Creek had a consistently high proportion of steelhead detected emigrating each year, with an average of 35.0% (SD = 8.7%) of a tagging cohort detected on LLS. Kenney Creek also had a relatively high proportion of steelhead emigrating, with all but three tagging cohorts emigrating at rates greater than 24.1% and an average of 18.6% (SD = 12.6%). The pattern of steelhead emigration out of Hayden Creek was similar to that of Little Springs and Kenney creeks with an average of 18.9% (SD = 8.4%, Table 1.2). Hawley Creek had the lowest proportion of tagged steelhead that were detected leaving, with an average of 0.7% (SD = 0.7%, Appendix E). The proportion of PIT-tagged steelhead detected emigrating from Big Timber Creek decreased precipitously after 2012 and the average proportion was 9.3% (SD = 9.9%). Proportions of steelhead detected emigrating from Canyon Creek were less than 6.9% for all tagging cohorts except in 2012, which had 24.2% of the tagging cohort detected emigrating and the average proportion was 6.2% (SD = 7.0%).

Fluvial Bull Trout have only been confirmed in the Hayden Creek drainage before this study, but observations of PIT-tagged individuals emigrating from other tributaries suggest the potential re-establishment of a fluvial component from elsewhere in the Lemhi River basin. Hayden Creek had the largest proportion of Bull Trout detected leaving the tributary and it has generally been increasing over the past few years with an average of 14.4% (SD = 8.0%, Table 1.2). Of the priority tributaries, Kenney Creek had the highest proportion of Bull Trout leaving, with an average of 4.5% (SD = 2.8%, Appendix E). Bull trout were first PIT-tagged in Kenney Creek in 2011, but not 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, Big Timber, Little Springs, and Hawley creeks, none were detected on the PIT-tag arrays at the mouths of those tributaries. Given the small number of PIT-tagged Bull Trout in Little Springs, Canyon, and Hawley creeks, PIT arrays may have failed to detect emigrating fish.

The number of fish detected moving from the Lemhi River into tributaries was highly variable (Appendix E). All tributaries had PIT-tagged fish detected moving in, but Chinook Salmon and Bull Trout were not detected moving into every tributary. Hawley Creek was the only tributary where Chinook Salmon were never detected immigrating. Canyon Creek and Hawley Creek were the only tributaries where Bull Trout were never detected immigrating. Chinook Salmon were first detected moving into Little Springs Creek in 2012, but were not detected moving into Big Timber or Canyon creeks until 2015. In the lower Lemhi River watershed, Chinook Salmon were not detected moving into Bohannon or Kenney creeks until 2015. In all years that the PIT-tag array has been operating in Hayden Creek, Chinook Salmon, steelhead, and Bull Trout were detected immigrating (Table 1.3). Little Springs Creek had the most steelhead detected (Appendix E). Since 2011, 178 steelhead tagged in the Lemhi River basin have been detected moving into Little Springs Creek.

We determined that many of the immigrants into Big Timber Creek remained within the lower reaches of the tributary based on detections at the additional PIT-tag arrays (Appendix E). Although the upper array has been operating since 2014, no fish tagged outside of Big Timber Creek were detected moving past this array until 2016. The period of operation was much shorter for the middle array, but the first detection of a fish PIT-tagged outside of Big Timber Creek also occurred in 2016. The number of immigrants migrating past all three arrays was low, but the minimal detections provide confirmation that fish were able to migrate from lower Big Timber Creek upstream past the BTU array.

Juvenile Chinook Salmon and Steelhead Emigration—Annual trends in abundance of juvenile Chinook Salmon migrating past the rotary screw traps were generally similar among the three traps (Figure 1.27). Total brood year abundance estimated at the Upper Lemhi RST ranged from 4,349 fish (SE = 406.0) for brood year 2007 to 58,415 fish (SE = 2,889.4) for brood year 2014. Brood year abundance in Hayden Creek ranged from 3,369 fish (SE = 125.0) for brood year 2005 to 32,095 fish (SE = 2,679.8) for brood year 2007. Total emigrants from the basin estimated at the Lower Lemhi River RST ranged from 7,679 fish (SE = 745.5) for brood year 2006 to 74,168 fish (SE = 1710.8) for brood year 2014. Given that some Chinook Salmon production occurs in the Lemhi River below the upper Lemhi RST, the estimates of emigrant abundance at the lower Lemhi River RST are combinations of the production that occurs in the lower Lemhi River and the sum of Hayden Creek and upper Lemhi River emigrants that survive to the lower Lemhi RST. Brood year abundance estimates were least variable through time for Hayden Creek. At the upper and lower Lemhi River RSTs brood year estimates demonstrated an overall positive trend from brood year 2005 to brood year 2014. However, this was strongly influenced by the exceptionally high abundances of the 2009 and 2014 brood years.

Abundance estimates of juvenile steelhead migrating past the upper Lemhi River and Hayden Creek rotary screw traps were less variable across years than at the lower Lemhi River trap (Figure 1.27). Steelhead abundance at the upper Lemhi RST ranged from 9,722 fish (SE = 798.5) in 2011 to 35,715 fish (SE = 2,482.2) in 2008. Abundance at the Hayden Creek RST was lowest in 2006 with 469 steelhead (SE = 169.8), but that was the first year of trap operations, so it only includes the fall migration period. Considering years with full operation, abundance estimates at the Hayden Creek RST ranged from 3,472 fish (SE = 405.0) in 2010 to 18,311 fish (SE = 971.0) in 2013. Basin-wide abundance estimates of steelhead migrating past the lower Lemhi River RST ranged from 5,501 fish (SE = 741.4) in 2014 to 47,485 fish (SE = 8,829.7) in 2008. Although no clear trend existed in abundance estimates at the upper Lemhi River and Hayden Creek rotary screw trap, a negative trend was observed at the lower Lemhi River RST.

Juvenile Chinook Salmon Survival—Survival rates within the Lemhi River basin varied among watersheds and seasons (QCI, unpublished data; Figure 1.28). Summer survival rates of parr were much higher than winter survival rates of fall parr in all reaches. Summer survival rates were generally higher in Hayden Creek than in the Lemhi River with mean survival in Hayden Creek of 0.60 (SE = 0.12) and mean survival in the Lemhi River of 0.32 (SE = 0.06). In all years, winter survival rates of Hayden Creek fall parr wintering in the lower Lemhi River were higher than upper Lemhi River fall parr wintering in the same reach. Additionally, upper Lemhi River fish wintering in the upper Lemhi River had higher survival than those that wintered in the lower Lemhi River. Hayden Creek fall parr had an average survival rate in the lower Lemhi River of 0.20 (SE = 0.05) and upper Lemhi River fall parr had an average winter survival rate of 0.06 (SE = 0.02). The average winter survival rate of upper Lemhi River salmon in the upper Lemhi River was 0.13 (SE = 0.05).

Survival to Lower Granite Dam 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 1.29). The difference in survival between fall parr and smolts ranged from 0.1 to 0.52 (Mean = 0.31) for the upper Lemhi River sub-population 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.

Habitat Capacity—Tributary restoration efforts have significantly improved the total rearing capacity of the Lemhi River basin. Prior to tributary reconnection efforts, summer rearing capacities within the main-stem Lemhi River, Big Springs Creek, and Hayden Creek were estimated as 255,364 steelhead and 519,545 Chinook Salmon. The reconnection of Canyon Creek in 2011 opened up an additional 21.0 km of summer rearing habitat, representing an estimated increase in rearing capacity of 41,412 steelhead and 72,953 Chinook Salmon (Table 1.4). Reconnection of Little Springs Creek opened up an additional 6.96 km of rearing habitat in 2011, which translated to an estimated increase in rearing capacity of 4,526 steelhead and 5,431 Chinook Salmon. The reconnection of Kenney Creek added 8.4 km of additional habitat and increased rearing capacity by 17,039 steelhead and 42,908 Chinook Salmon. Surveys in the other priority tributaries predict that functional reconnection of Bohannon, Big Timber, and Hawley creeks will increase rearing capacity by 140,066 steelhead and 190,208 Chinook Salmon.

The Lemhi River basin model predicts that restoration actions completed to date, including the partial reconnection of Big Timber Creek, are sufficient to meet the desired survival improvements for steelhead, but not for Chinook Salmon (ISEMP and CHaMP 2017b). The predicted response in productivity of steelhead is 164 smolts/adult, which represents a 10% increase from pre-restoration conditions (ISEMP and CHaMP 2017b). The predicted response in productivity of Chinook Salmon is 19.3 smolts/adult, which represents a 3% increase from pre-restoration conditions (ISEMP and CHaMP 2017b).

Chinook Salmon Productivity—Productivity varied considerably across years within the upper Lemhi River and Hayden Creek as well between spawning areas within years. Productivity in the upper Lemhi River ranged from 6 smolts/redd for brood year 2006 to 72 smolts/redd for brood year 2012. Productivity in Hayden Creek ranged from 10 smolts/redd from brood year 2011 to 130 smolts/redd from brood year 2008. When comparing productivity as the combined number of fall parr and smolts per redd, there was not a significant treatment effect (time period effect $p = 0.61$, Table 1.5). However, there was a significant treatment effect (time period effect $p = 0.01$) 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 = 57 smolts/redd) compared to years prior to restoration efforts (Mean = 23 smolts/redd, Figure 1.30).

DISCUSSION

Declines in anadromous fish abundance have prompted significant investment in tributary habitat restoration as a method to support fish recovery efforts in the Columbia River basin. Considerable effort has been devoted towards implementing restoration actions and conducting effectiveness monitoring in the Lemhi River basin. Results from the Lemhi River IMW show that restoration actions have elicited detectable responses from fish. Responses vary among species and life stages, but indicate early signs of success from initial restoration actions, and highlight

the potential effects that may result from completion of additional actions identified in the SRBA Lemhi River Basin Habitat Action Framework. Here, we discuss the key results of the Lemhi River IMW project and relate them to restoration actions that have been completed through 2016. Furthermore, we demonstrate how IMW results were used to adapt monitoring and restoration strategies in the Lemhi River basin.

The most noteworthy responses to restoration actions have been exhibited 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 (see Appendix A). As of 2016, we have documented juvenile Chinook Salmon during summer electrofishing surveys in all of the priority tributaries except for Hawley Creek. Following barrier removal projects and re-watering dry stream segments from 2009-2011, juvenile Chinook Salmon were first captured in priority tributaries in 2012. They have been observed in all subsequent years, indicating that juveniles produced in the upper main-stem river are accessing newly connected tributary habitats for rearing opportunities. Canyon Creek surveys documented the first occurrence of Chinook Salmon in 2011, which was the same year that a water conservation project established a year-round surface water connection with the Lemhi River.

Movement patterns in Little Springs Creek illustrate the overall trend in upper Lemhi River tributaries where restoration efforts are occurring. In addition to juvenile Chinook Salmon immigrating into tributaries when surface water connections are re-established and barriers are removed, they are also expanding their upstream extent in subsequent years following project implementation. This pattern suggests juvenile salmon are using the tributaries more to their full potential as time progresses. Chinook Salmon have been observed in Kenney and Bohannon creeks, but not at the level observed in upper Lemhi River tributaries. Very little spawning occurs in the Lemhi River below the Hayden Creek confluence, so those tributaries are a considerable distance from areas where Chinook Salmon are produced. As such, the individuals we documented in the summer months are probably using coldwater tributaries as thermal refugia during their downstream migration.

Responses by adult salmonids were most apparent at the tributary scale. Steelhead spawning has occurred in reconnected tributaries, but Chinook Salmon spawning activity has not been observed. In Kenney Creek, steelhead spawning occurred in the lower 1.75 km below the LKC-02 diversion prior to reconnection. We continue to document redds in this reach after reconnection occurred. Redds have also been documented above the LKC-02 diversion after it was modified to accommodate year-round fish passage. In recent years, steelhead redd counts have not been conducted above this location, so the current upstream extent of spawning is unknown. However, these results confirm that successful spawning has occurred following removal of a known passage barrier, but the limited amount of upstream expansion may be due to the overall low abundance of spawners using this tributary. Furthermore, we documented adult Bull Trout migrating between the Salmon River and Kenney Creek from 2012 through 2014, suggesting that a fluvial life history component was established there following reconnection in 2011. Following reconnection of Little Springs Creek in 2011, adult steelhead have been detected on the PIT-tag array each year since 2013, but no confirmed spawning activity occurred until one redd was observed in 2017.

Steelhead have also spawned in priority tributaries not fully reconnected to the main-stem river. Bohannon Creek is not considered functionally connected because of a dewatered stream reach below the BHC-3 diversion, but completed restoration projects have made substantial improvements to streamflow and fish passage. Prior to any reconnection efforts, steelhead spawning occurred in the lower 6km below the East Bohannon Creek confluence. In 2014, a water

conservation agreement spilled water through a seasonally dewatered transect below BHC-3 from April 1-June 30, and since then redds have been documented through that reach each year. Adult steelhead have also been detected on the Big Timber and Canyon creek arrays, but we did not observe any steelhead redds during spawning ground surveys. Although total escapement of steelhead to the basin has not exhibited a detectable increase and no significant changes in tributary-specific escapement have occurred, adult steelhead have undoubtedly responded to specific restoration actions within tributaries.

The response of adult Chinook Salmon has not followed the same trajectory as that of adult steelhead. Prior to implementation of restoration actions, and throughout the IMW monitoring period, Chinook Salmon spawning has only occurred in Hayden Creek, the main-stem Lemhi River, and Big Springs Creek. Although habitat size 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 facilitated 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).

The amount of suitable spawning habitat in the occupied portions of the Lemhi River and Hayden Creek exceed the amount needed to accommodate the current level of escapement to the basin. Total redd capacity in the areas that have remained accessible to Chinook Salmon since major anthropogenic disturbance in the basin (main-stem Lemhi River, Hayden Creek, and Big Springs Creek) was estimated at 2,031 redds in 2016 (USSIRA 2017). Given that the maximum combined redd count during the course of recent monitoring was 426 redds in 2001, we would not expect much competition for spawning areas. Therefore, we may not see spawning occurring in reconnected areas until escapement approaches or exceeds current redd capacity and there is a greater impetus for colonizing new habitats.

Salmon recolonization has most commonly been documented as adults moving into new habitat, but juvenile dispersal has also been shown as a possible mechanism for recolonization (Anderson et al. 2008; Hamann and Kennedy 2012). As more juvenile Chinook Salmon use the tributaries for rearing opportunities, we may expect some of those fish returning as adults to spawn in the tributary if they imprint on rearing areas rather than natal areas (Dittman and Quinn 1996). In the Cedar River, Washington, removal of a dam on the main-stem river opened up 33 km of main-stem and tributary habitats above the dam in September 2003, with immediate Chinook Salmon spawning above that location that year (Burton et al. 2013). In the subsequent years, redds were only observed in the main-stem habitat, and it was not until 2007 when the highest abundance occurred above the dam that spawning was documented in tributary habitats (Burton et al. 2013). The first documented spawning in tributaries coincided with the highest escapement above the dam, and when the proportion of total basin redds occurring above the dam was highest. This initial spawning may have been the result of increased competition for available habitat leading to adults pioneering, or the adults may have used those tributaries for rearing opportunities when they were juveniles. If the latter, we may expect some of the juveniles we observed in the reconnected tributaries to return to those areas as adults to spawn. The detections of three adult Chinook Salmon in Little Springs Creek suggest we are documenting the initial stages of recolonization.

Results from electrofishing surveys and PIT-tag array interrogations suggest that connectivity between tributaries and main-stem habitats are important to salmonid life history expression within the Lemhi River basin. We observed high rates of movement between tributaries and the main-stem Lemhi River, which is consistent with literature for anadromous and potamodromous salmonids showing these movement patterns are common responses to shifting environmental conditions and life-history events (Bramblett et al. 2002; Anderson et al. 2008; Uthe et al. 2016). Moreover, the recent observations of PIT-tagged Bull Trout emigrating from Bohannon Creek may suggest the early stages of re-establishment of a fluvial life history form in that tributary. The presence of juvenile Chinook Salmon within tributaries demonstrates that summer rearing opportunities may be important for juvenile salmonid production. For example, juveniles may move into non-natal tributary habitats during the late-spring and early-summer to avoid deleterious effects of sediment loads during main-stem high flows (Scrivener et al. 1994) or migrate into those areas during autumn to seek over-winter habitat (Swales et al. 1986; Bradford et al. 2001).

In the Lemhi River basin, water temperatures sometimes exceed optimum thresholds during the summer (USSIRA 2017) and winters can be prolonged with adverse ice conditions. Tributary habitats therefore provide important refugia throughout the entire year. Autumn migrations by sub-yearling trout and salmon are common in the Lemhi River basin (Bjornn 1971); the immigration of juveniles into Little Springs Creek exhibits a peak in October and November suggesting that fish are seeking wintering areas in this groundwater-influenced tributary. Additionally, a high proportion of fish tagged during summer electrofishing surveys did not emigrate from Little Springs Creek until the following year, suggesting they wintered within the tributary. This pattern 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).

In addition to providing non-natal rearing opportunities for salmonids, tributaries have the potential to yield a substantial proportion of basin-wide production. The number of juvenile Chinook Salmon and steelhead that emigrate from Hayden Creek is similar to, and sometimes exceeds, the abundance from the entire upper Lemhi River basin. This observation provides insight into the historical importance of the tributaries, and therefore, justification for tributary reconnections as a major focus of the restoration strategy. Hayden Creek and upper Lemhi River Chinook Salmon smolts had very similar survival during their migration to Lower Granite Dam, indicating that production within tributary habitats can contribute fish with comparable fitness to the main-stem production area.

Interconnected habitats are also important to seasonal survival, as demonstrated by the variation in survival rates among rearing areas. Parr survival in Hayden Creek during the summer and fall was higher than parr survival in the upper Lemhi River, which may be the result of consistently cold water in this tributary. However, the time period of investigation for parr survival is different between the two subpopulations, with the upper Lemhi River timeframe encompassing June-December, while the Hayden Creek timeframe only includes September-December. Therefore, it is unclear whether there is a true survival difference or our inference is an artifact of sampling and analytical methods. Further investigation is needed to isolate summer survival rates and make them more comparable between the subpopulations and rearing areas.

Winter survival rates not only varied among reaches, but also between subpopulations within watersheds. Hayden Creek parr that emigrated from Hayden Creek in the fall and wintered in the lower Lemhi River had higher survival rates than upper Lemhi River fall parr that wintered in the same area. Given that sheltering during winter is strongly density-dependent in salmonids

(Armstrong and Griffiths 2001), this may be the result of Hayden Creek fall parr gaining occupancy to the best available winter locations in the lower Lemhi River. Futhermore, the average size of Hayden Creek fall parr is up to 20 mm smaller than the average size of upper Lemhi River fall parr (IDFG, unpublished data), which may enable Hayden Creek parr to more readily obtain adequate winter locations because sheltering requirments can be easier for small fish when availability of suitable shelter is reduced (Finstad et al. 2007). Our winter 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). Interestingly, the lack of consistency among winter studies supports 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). Although our results demonstrate low winter survival throughout the entire basin compared to Chinook Salmon in the Yukon River drainage (0.22, Bradford et al. 2001) and Coho Salmon *O. kisutch* in the Hood Canal, WA drainage (0.25-0.46, Quinn and Peterson 1996), the contrast between upper Lemhi River fall parr wintering in the upper Lemhi River compared to the lower Lemhi River emphasizes that suitable winter habitat is a major limiting factor in the lower portion of the basin.

Chinook Salmon survival rates through the migration corridor to LGR provide valuable information about life history diversity and potential 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 parr from the same production areas. This pattern is consistent with results from previous studies in the Lemhi River basin as well as other drainages in Idaho (Copeland et al. 2014). The survival rate of spring emigrants is a true measure of survival from rearing areas in the spring to LGR, whereas survival of fall emigrants is a composite estimate of winter survival downstream of natal areas and the survival during the final stage of migration to LGR in the following spring. Our watershed-specific survival results indicate that winter survival in the basin is low, but what remains unknown is how that compares to winter survival within the Salmon River for those fall parr that winter within that watershed. Partitioning estimates of fall parr survival to LGR into a winter 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. Notwithstanding, increasing winter survival rates within the Lemhi River should translate to an increase in emigrants arriving at LGR.

The primary restoration strategy of reconnecting tributaries and improving habitat conditions in the main-stem Lemhi River has resulted in increased Chinook Salmon production. Our results suggest that productivity of Chinook Salmon age-1 smolts increased throughout the monitoring period. We hypothesized that reconnection and flow conservation efforts in the upper Lemhi River basin would increase productivity of Chinook Salmon. Interestingly, we detected a significant treatment effect when considering only age-1 smolts per redd, but not when considering total emigrants per redd. This observation 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 need to investigate other metrics to develop a mechanistic basis for this relationship. We expect that fish using these 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 numbers of tagged juvenile Chinook Salmon immigrating into tributaries 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.

Additional tributary flow in the Lemhi River likely 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). In a drainage like the Lemhi River, where much of the lower river is at a minimum flow agreement in most years, the cold-water inputs should have a positive benefit as thermal refuge through this problematic reach where temperatures often exceed optimal threshold levels. With the exception of Little Springs Creek, the other five priority tributaries drain high-elevation, mountainous areas, which contribute consistently cold water throughout the entire year. Given the already low flows associated with anthropogenic influences in the basin and the anticipated changes in flow regime as a result of climate change (Isaak et al. 2012), the benefits of these coldwater inputs are likely to be amplified in the future. Currently, we do not have a full understanding of the indirect benefits of tributary reconnections, but consideration of climate change in restoration planning suggests that even if benefits are not evident now, they may become important in the future (Beechie et al. 2013), thus bolstering the current positive effects of projects. Current thermal regimes in the Lemhi River frequently exceed optimal for summer parr rearing and adult spawning thermal thresholds (USSIRA 2017). Modeling of future climate scenarios with a 3°C increase in water temperatures resulted in more frequent exceedance of optimum temperatures for spawning and rearing, but also caused temperatures to exceed maximum thresholds for approximately 25% of the timeframes (USSIRA 2017), underscoring the future benefits of increased flow and coldwater inputs associated with current restoration and reconnection efforts.

Overall, the predicted changes in capacity suggest that restoration actions completed to date are sufficient to meet productivity objectives for steelhead, but not for Chinook Salmon (ISEMP and ChaMP 2017b). The restoration actions completed through 2016 did not result in significant change to basin-level habitat metrics, but they increased wetted stream area by 22% and pool habitat by 19% (ISEMP and CHaMP 2017b). The IMW monitoring results indicate that juvenile fish are occupying and increasing their upstream extent of use in newly connected tributary habitats, but the current standing stocks are well below the predicted capacity based on life cycle models. Furthermore, total escapement of steelhead and Chinook Salmon to the Lemhi River remained well below the minimum abundance threshold for population viability identified by NMFS. Even with the slight positive trend in Chinook Salmon escapement over the monitoring period, the peak of 683 adults in 2015 is well below the viability objective of 2,000 spawning adults (NOAA 2016). The early signs of success we documented during the course of the IMW project (i.e. juvenile salmonids using reconnected habitats) are encouraging, but it may take more time until the benefits of these current actions fully manifest themselves at the population level. As such, recent focus on increasing habitat quality (i.e., restoration in the lower Lemhi River, Appendix C) to address winter survival limitations, may result in a more rapid response than we have seen so far from increasing habitat quantity (i.e., tributary reconnections).

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 USBWP Tech Team meetings to influence restoration implementation and provide up-to-date information for project ranking and planning efforts. We also convened annual coordination meetings among IMW and ISEMP personnel to assess how well sampling protocols were meeting objectives and develop sampling modifications to address shortfalls. Overall, communication and collaboration have been essential to adaptively managing the restoration and monitoring programs in the Lemhi River basin.

We discuss changes to habitat and fish monitoring first. Habitat monitoring used the PIBO protocol from 2009-2010, but was switched to a more fish-centric habitat monitoring protocol under CHaMP in 2011 to better meet the needs of the project objectives (ISEMP and CHaMP 2016a). The most significant change to the fish monitoring program was the shift from site-based electrofishing surveys to a spatially continuous survey design. A post-hoc analysis of electrofishing results from 2009-2012 identified that the survey effort using the GRTS design would need to be increased by 300% to achieve the desired precision of less than 15% CV for abundance estimates (ISEMP and CHaMP 2017a). Movement data revealed fish were moving into and out of GRTS sites between mark and recapture sampling events (ISEMP and CHaMP 2017a). To address these two issues, the spatially continuous fish sampling was implemented in 2013, which maintained compatibility with the CHaMP habitat GRTS sites to continue the development of fish habitat relationship models. Results from 2013-2016 indicate the current level of sampling effort is adequate for the desired precision goals and addressing the objectives of the monitoring program.

The flexibility of the Lemhi River IMW project, particularly the spatial hierarchy with effectiveness monitoring occurring at different scales, enables us to add new project components as results highlight novel or previously unknown issues. Not only are results providing relevant information to help restoration planning, but as new restoration projects and actions are implemented, we have been diligent in responding with changes in our monitoring framework. With the large-scale Eagle Valley Ranch restoration project occurring in the lower Lemhi River, the previous sample design for electrofishing in the lower river was modified to enable effectiveness monitoring with a BACI study design (Appendix C), but also overlap CHaMP habitat sites and provide sufficient data for ISEMP modeling objectives.

The Lemhi River IMW results have been critical to the restoration planning, prioritization, and implementation process in the basin. Furthermore, the knowledge gained in the Lemhi River basin has recently been applied to restoration planning in the Pahsimeroi River and upper Salmon River drainages (USSIRA 2017). Early planning documents viewed the lower Lemhi River as a travel corridor, and specific actions focused on increasing flows and removing barriers to facilitate upstream passage by adults and downstream passage by juveniles. Recent survival results indicate that winter survival in the lower Lemhi River is a limiting factor for Chinook Salmon (ISEMP and CHaMP 2015). Tributary reconnections and improving summer rearing habitat remains a major focus of restoration actions, but recent projects are targeting areas in the lower main-stem Lemhi River and incorporating specific treatment types expected to provide ideal wintering habitats. Given that tributary reconnections are increasing habitat quantity and fish have to use that new habitat to receive the benefits of restoration, the more recent focus on improving quality of currently-occupied habitats should provide a more immediate benefit to the fish, and thus a more immediate response. The Lemhi River IMW identifies limiting factors with spatial context, and those results are integrated into the planning process through the use of literature to help guide specific project actions. Furthermore, we modify our monitoring framework and increase sampling coverage to assess the effectiveness of those projects and treatment types.

The next 10 to 15 years will be imperative to the success of the Lemhi River IMW. Restoration planned within the next five years should result in functional reconnection of the remaining three priority tributaries. Furthermore, additional tributaries not identified for reconnection in the SRBA Lemhi River Basin Habitat Action Framework (e.g., Pratt Creek) will likely be functionally reconnected within this timeframe. The remaining three phases of the Eagle Valley Ranch restoration project are in various stages of planning and implementation, so project completion may not occur for at least five years. Post-treatment monitoring should occur for at least five years to fully understand fish population and habitat response to restoration actions.

Therefore, we need to continue our existing monitoring effort for another 10 to 15 years to enable sufficient 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, but have not resulted in a basin-scale response. The results suggest that restoration has caused an increase in summer rearing capacity of Chinook Salmon (USSIRA 2017). This effort has also highlighted the need for large-scale projects in the lower Lemhi River that incorporate specific restoration actions designed to increase winter survival. The indication that 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 into the future. The initial responses to restoration that we documented are encouraging, but full understanding of fish population and habitat responses in the Lemhi River will require monitoring for an additional 10 to 15 years.

PART 1 TABLES

Table 1.1. Site metadata for PIT-tag arrays in the Lemhi River basin.

Stream	Site Location	Installation Date	PTAGIS Code	Latitude	Longitude
Lemhi River	Lower Lemhi River	08/18/2009	LLR	45.176	-113.885
Lemhi River	Lemhi River Weir	08/18/2009	LRW	44.866	-113.625
Hayden Creek	Mouth	08/19/2009	HYC	44.861	-113.632
Big Timber Creek[a]	Upstream of Hwy 28	02/25/2010	BTC	44.688	-113.370
Kenney Creek[b]	Mouth	06/01/2010	KEN	45.027	-113.658
Canyon Creek	Mouth	11/12/2010	CAC	44.692	-113.355
Little Springs Creek	Mouth	06/14/2011	LLS	44.781	-113.545
Bohannon Creek	Mouth	12/06/2011	BHC	45.112	-113.747
Wimpy Creek	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[c]	Hawley/Eighteenmile Confluence	10/21/2013	HEC	44.669	-113.311
Lee Creek	Mouth	12/21/2013	LCL	44.747	-113.475
Hayden Creek	Hayden/Bear Valley Confluence	09/15/2014	HYB	44.772	-113.708
Big Eightmile Creek	Mouth	09/15/2014	LB8	44.738	-113.463
Big Springs Creek	Mouth	09/15/2014	LBS	44.727	-113.433
Big Timber Creek	Upper section of creek	09/15/2014	BTU	44.614	-113.397
Eighteenmile Creek	Upstream of Hwy 29	09/15/2014	18M	44.683	-113.353
Big Timber Creek	Middle section of creek	05/15/2015	BTM	44.661	-113.378

[a] Discontinued 10/15/2013.
[b] Inoperable since 11/01/2015.
[c] Discontinued 06/01/2017.

Table 1.2. Number of steelhead, Chinook Salmon, and Bull Trout PIT-tagged in Hayden Creek that were subsequently detected on the PIT-tag array near the mouth of Hayden Creek (HYC). Proportion of tagging cohort is shown in parentheses.

Tag year	Number tagged	Number detected by year								Total
		2009	2010	2011	2012	2013	2014	2015	2016	
					Steelhead					
2009	532	53 (10.0)	16 (3.0)	9 (1.7)	1 (0.2)	0	0	0	0	79 (14.8)
2010	45		7 (15.6)	4 (8.9)	1 (2.2)	0	1 (2.2)	0	0	13 (28.9)
2011	121			20 (16.5)	10 (8.3)	2 (1.7)	1 (0.8)	0	0	33 (27.3)
2012	92				17 (18.5)	8 (8.7)	1 (1.1)	0	0	26 (28.3)
2013	1170					89 (7.6)	69 (5.9)	23 (2.0)	2 (0.2)	183 (15.6)
2014	525						52 (9.9)	48 (9.1)	2 (0.4)	102 (19.4)
2015	817							68 (8.3)	45 (5.5)	113 (13.8)
2016	561								17 (3.0)	17 (3.0)
					Chinook Salmon					
2009	55	23 (17.3)	4 (3.0)	0	0	0	0	0	0	27 (20.3)
2010	13		38 (26.4)	2 (1.4)	0	0	0	0	0	40 (27.8)
2011	88			70 (20.7)	2 (0.6)	0	0	0	0	72 (21.6)
2012	191				94 (38.1)	2 (0.8)	0	0	0	96 (38.9)
2013	144					135 (36.6)	26 (7.0)	0	1(0.3)	161 (43.6)
2014	50						55 (20.3)	9 (3.3)	0	64 (23.6)
2015	57							204 (35.1)	26 (4.5)	230 (39.5)
2016	56								149 (17.5)	149 (17.5)
					Bull Trout					
2009	55	6 (10.9)	1 (1.8)	0	0	0	0	0	0	7 (12.7)
2010	13		1 (7.7)	0	0	0	0	0	0	1 (7.7)
2011	88			4 (4.5)	2 (2.3)	0	2 (2.3)	0	0	8 (9.1)
2012	191				7 (3.7)	15 (7.9)	7 (3.7)	1 (0.5)	0	30 (15.7)
2013	144					2 (1.4)	9 (6.3)	2 (1.4)	0	13 (9.0)
2014	50						9 (18.0)	2 (4.0)	0	11 (22.0)
2015	57							15 (26.3)	3 (5.3)	18 (31.6)
2016	56								4 (7.1)	4 (7.1)

Table 1.3. Number of steelhead, Chinook Salmon, and Bull Trout PIT-tagged in the Lemhi River basin outside of Hayden Creek that were subsequently detected on the PIT-tag array near the mouth of Hayden Creek (HYC).

Species	**Number detected by year**								**Total**
	2009	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	
Steelhead	12	18	19	20	15	23	38	23	168
Chinook Salmon	3	14	15	16	10	8	23	19	108
Bull Trout	1	4	3	11	9	15	8	14	65

Table 1.4. Summer rearing capacity estimates (fish/m) of steelhead and Chinook Salmon by channel type in priority tributaries and the reference tributary. Cumulative stream length (km) by channel type is shown in parentheses. For a detailed description of basin-wide capacity estimates by channel type see ISEMP and CHaMP (2016).

	Channel Type								Capacity
Stream	**Cascade**	**Confined**	**Island-Braided**	**Meandering**	**Plane-Bed**	**Pool-Riffle**	**Step-Pool**	**Straight**	**(total fish)**
				Steelhead					
Big Timber Creek		2.12 (0.46)	1.86 (8.00)	1.20 (1.50)	1.95 (9.12)	1.98 (8.97)	2.02 (0.26)	2.66 (2.24)	59,627
Bohannon Creek	2.28 (1.24)		3.22 (0.07)		2.25 (7.29)	1.89 (1.09)	2.60 (5.59)		36,069
Canyon Creek				1.37 (0.20)	1.90 (5.34)	2.00 (15.49)			41,412
Hawley Creek				1.31 (3.74)	1.96 (6.92)	2.05 (12.37)	1.92 (0.29)		44,370
Hayden Creek		1.79 (2.00)	1.98 (6.76)		2.13 (3.81)	2.08 (0.95)	2.18 (0.17)	2.88 (5.04)	41,961
Kenney Creek			2.69 (0.24)		1.97 (4.90)	2.00 (1.33)	2.08 (1.96)		17,039
Little Springs Creek				0.65 (6.96)					4,526
				Chinook Salmon					
Big Timber Creek		0.89 (0.46)	3.17(8.00)	3.14(1.50)	2.80(9.12)	3.37 (8.97)	3.25(0.26)	2.99 (2.24)	93,655
Bohannon Creek	0.00 (1.24)		4.92(0.07)		1.15(7.29)	2.14 (1.09)	0.01(5.59)		11,142
Canyon Creek				4.02(0.20)	3.50(5.34)	3.45 (15.49)			72,953
Hawley Creek				3.13(3.74)	3.46(6.92)	3.96 (12.37)	2.74(0.29)		85,412
Hayden Creek		2.46 (2.00)	3.27(6.76)		2.36(3.81)	3.34 (0.95)	1.60(0.17)	3.29 (5.04)	56,070
Kenney Creek			7.18(0.24)		5.18(4.90)	5.67 (1.33)	4.21(1.96)		42,908
Little Springs Creek				0.78(6.96)					5,431

Table 1.5. Analysis of productivity in the upper Lemhi River as a 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	252.603	86.332	7	2.926	0.022
Slope	-0.085	0.178	7	-0.479	0.646
Time period	61.625	116.396	7	0.529	0.613
		Smolts			
Intercept	51.001	9.087	7	5.613	0.001
Slope	0.215	0.179	7	1.198	0.270
Time period	-39.544	12.032	7	-3.287	0.013

PART 1 FIGURES

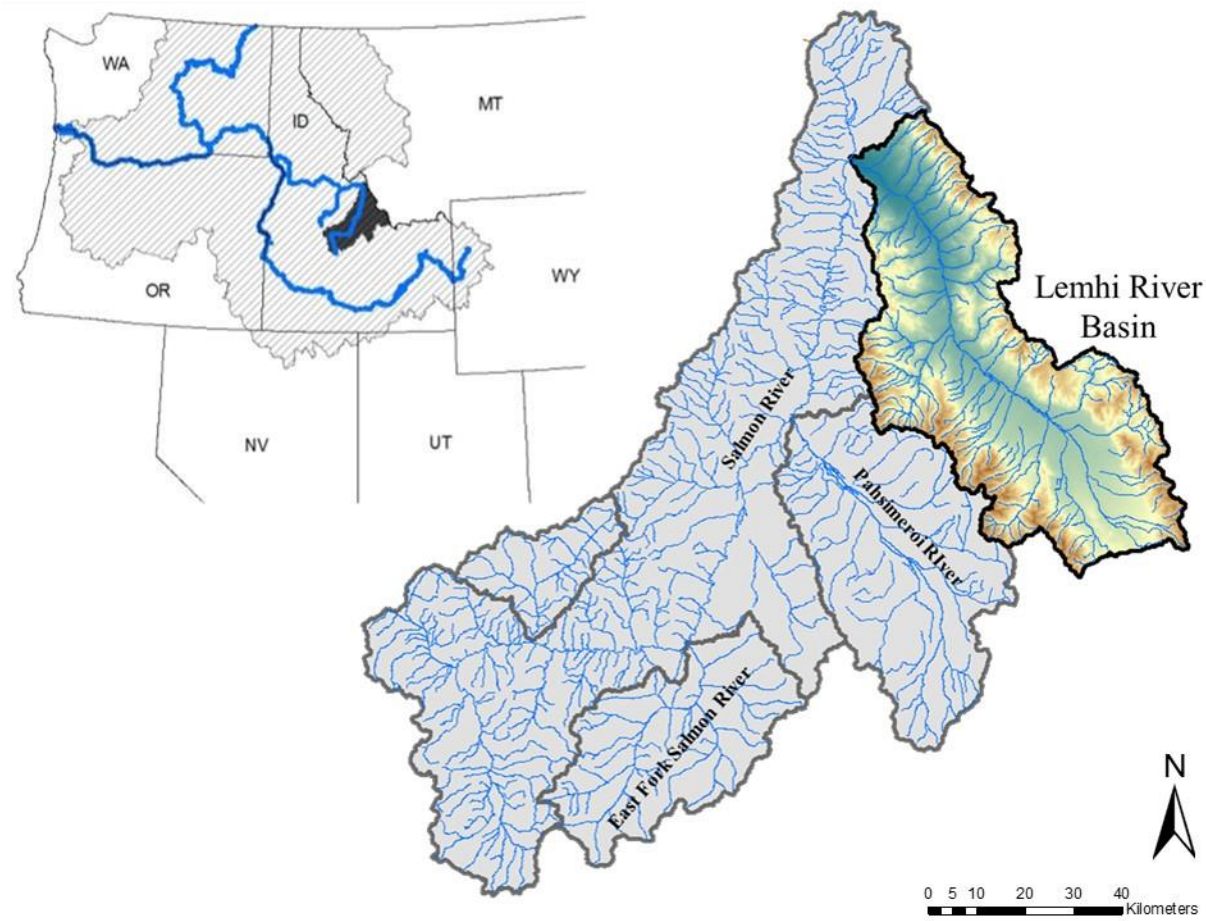

Figure 1.1. Location of the Lemhi River basin in the upper Salmon River drainage, Idaho.

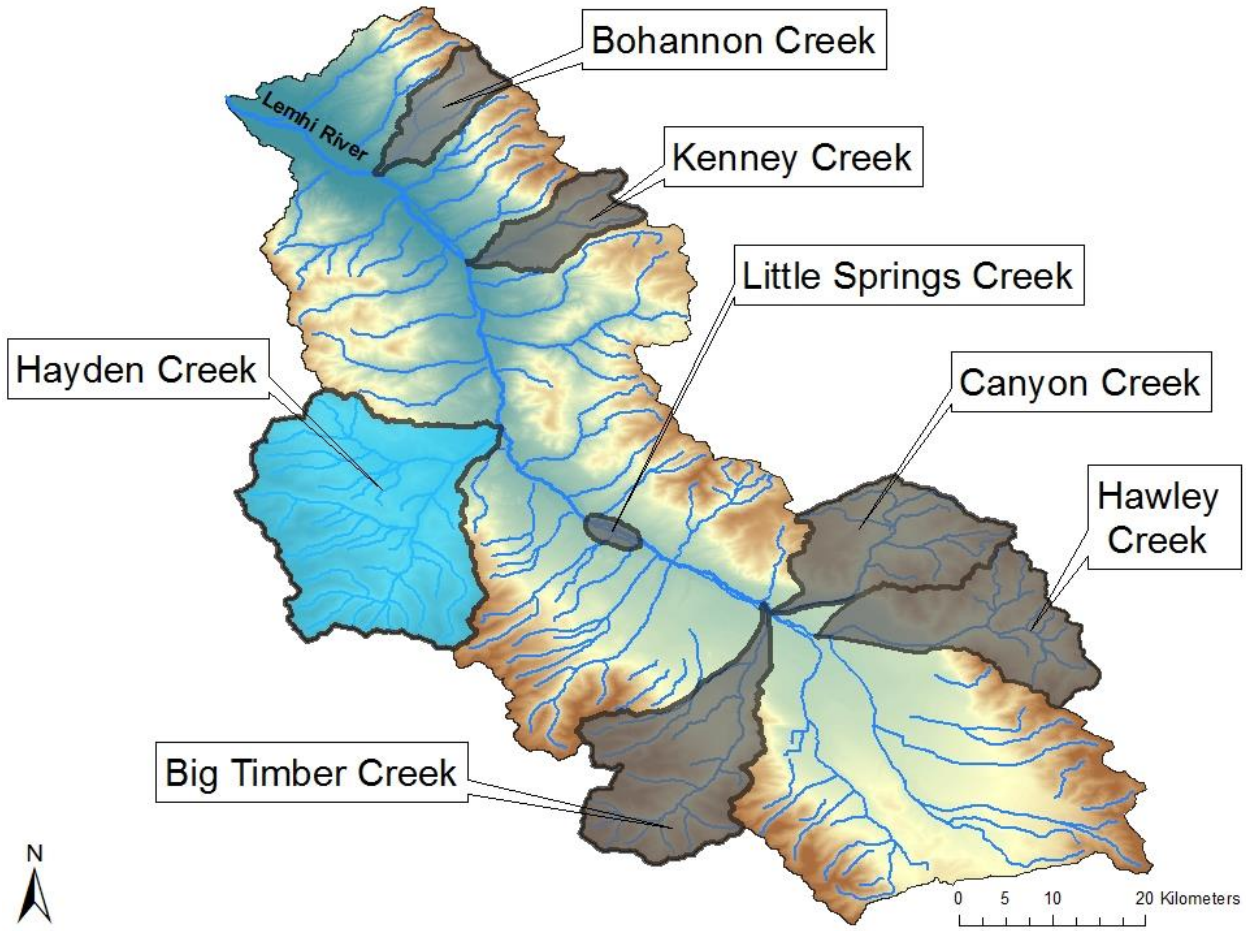

Figure 1.2. Priority candidate tributaries for reconnection (grey) and the reference tributary, Hayden Creek (blue).

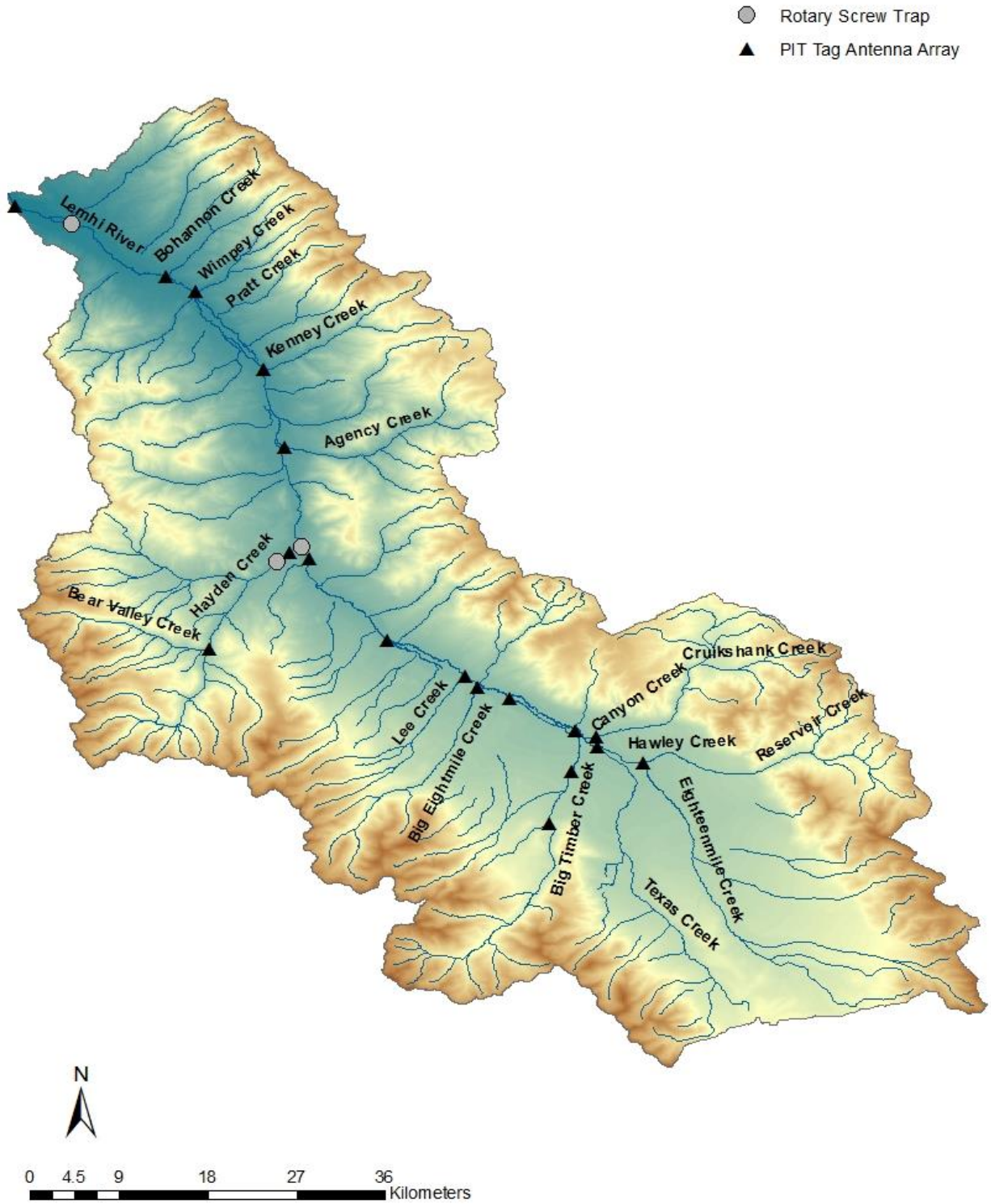

Figure 1.1. Locations of PIT-tag antenna arrays (triangles) and rotary screw traps (circles) installed in the Lemhi River basin.

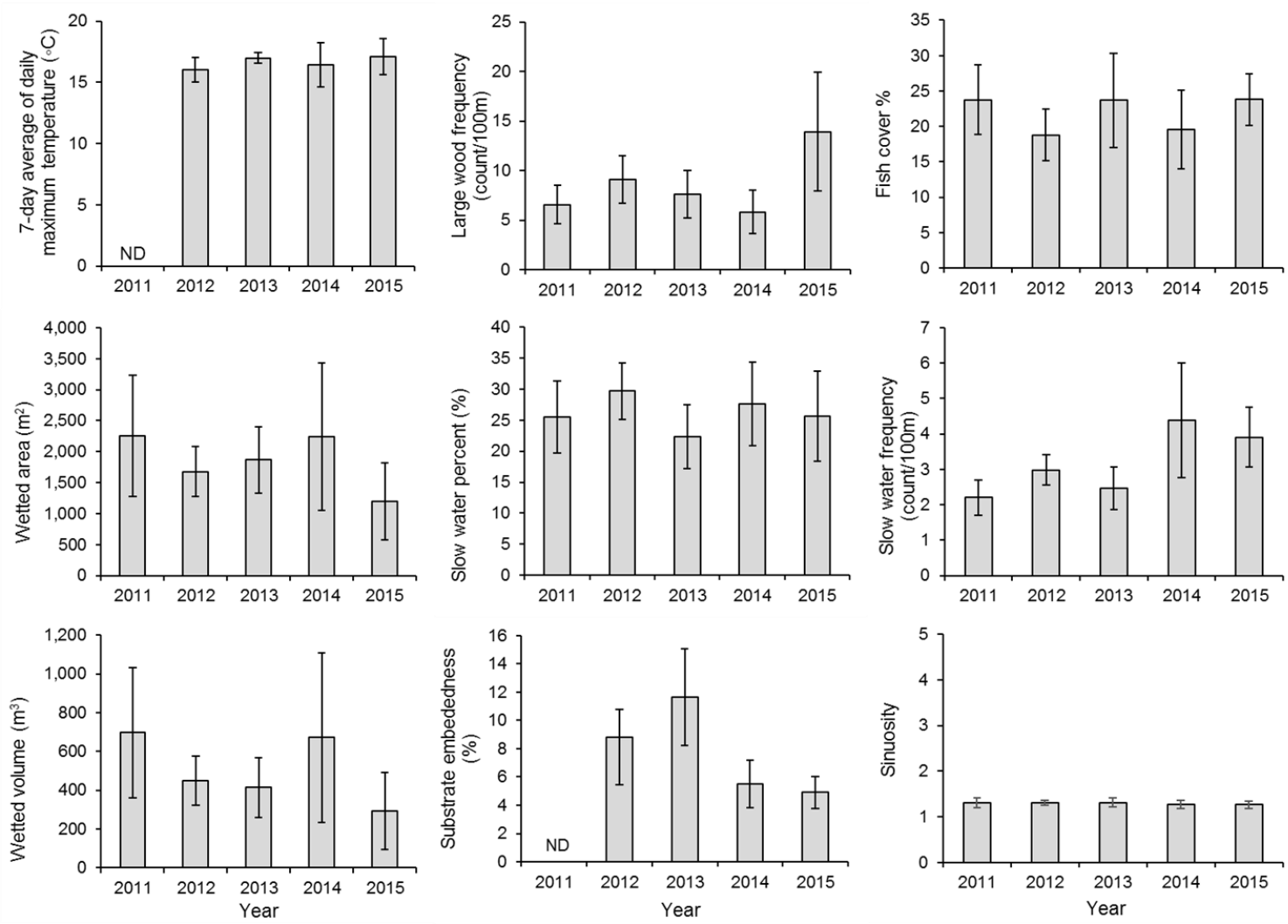

Figure 1.4. Annual trends in habitat metrics measured from habitat surveys in the Lemhi River basin, 2011-2015 (QCI, unpublished data). Estimates are shown with 95% confidence intervals. ND = no data available. See ISEMP and CHamP (2016b) for additional habitat results and a detailed description of metrics.

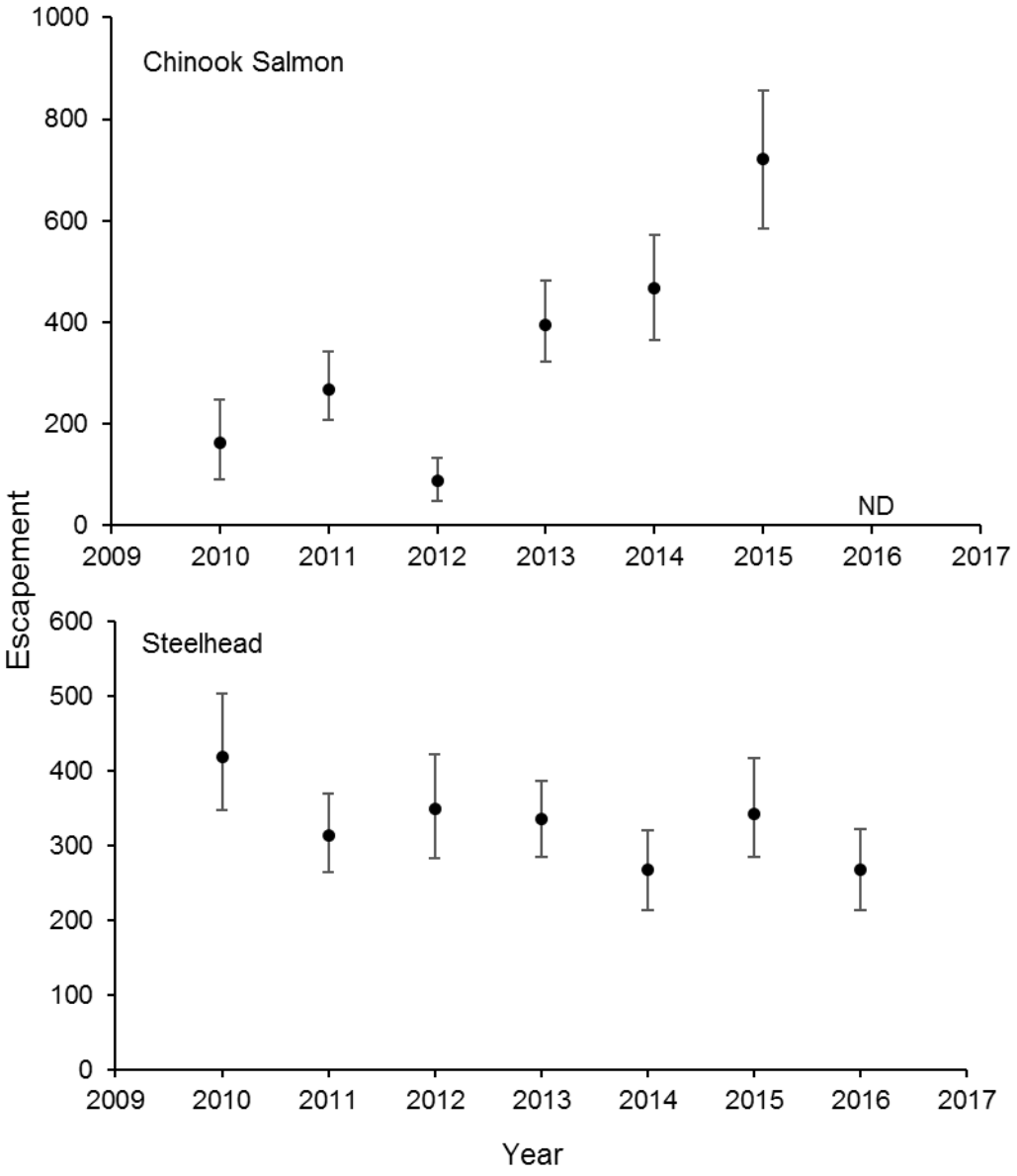

Figure 1.5. PIT-tag based estimates of escapement to the Lemhi River of adult Chinook Salmon (top) and steelhead (bottom). Estimates are shown with 95% confidence intervals (QCI, unpublished data). ND = no data available.

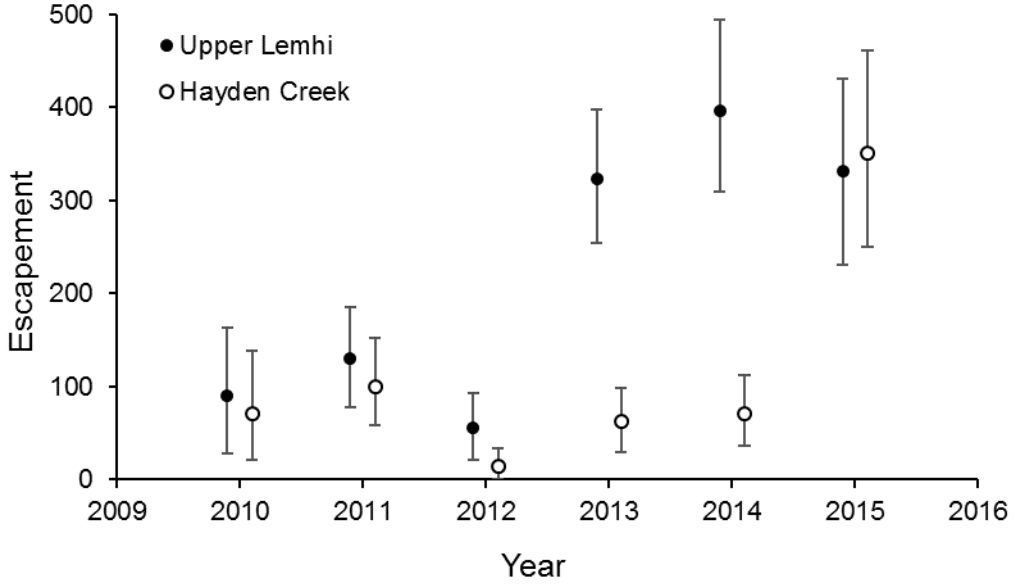

Figure 1.6. PIT-tag based estimates of Chinook Salmon escapement to the upper Lemhi River (closed) and Hayden Creek (open). Estimates shown with 95% confidence intervals (QCI, unpublished data).

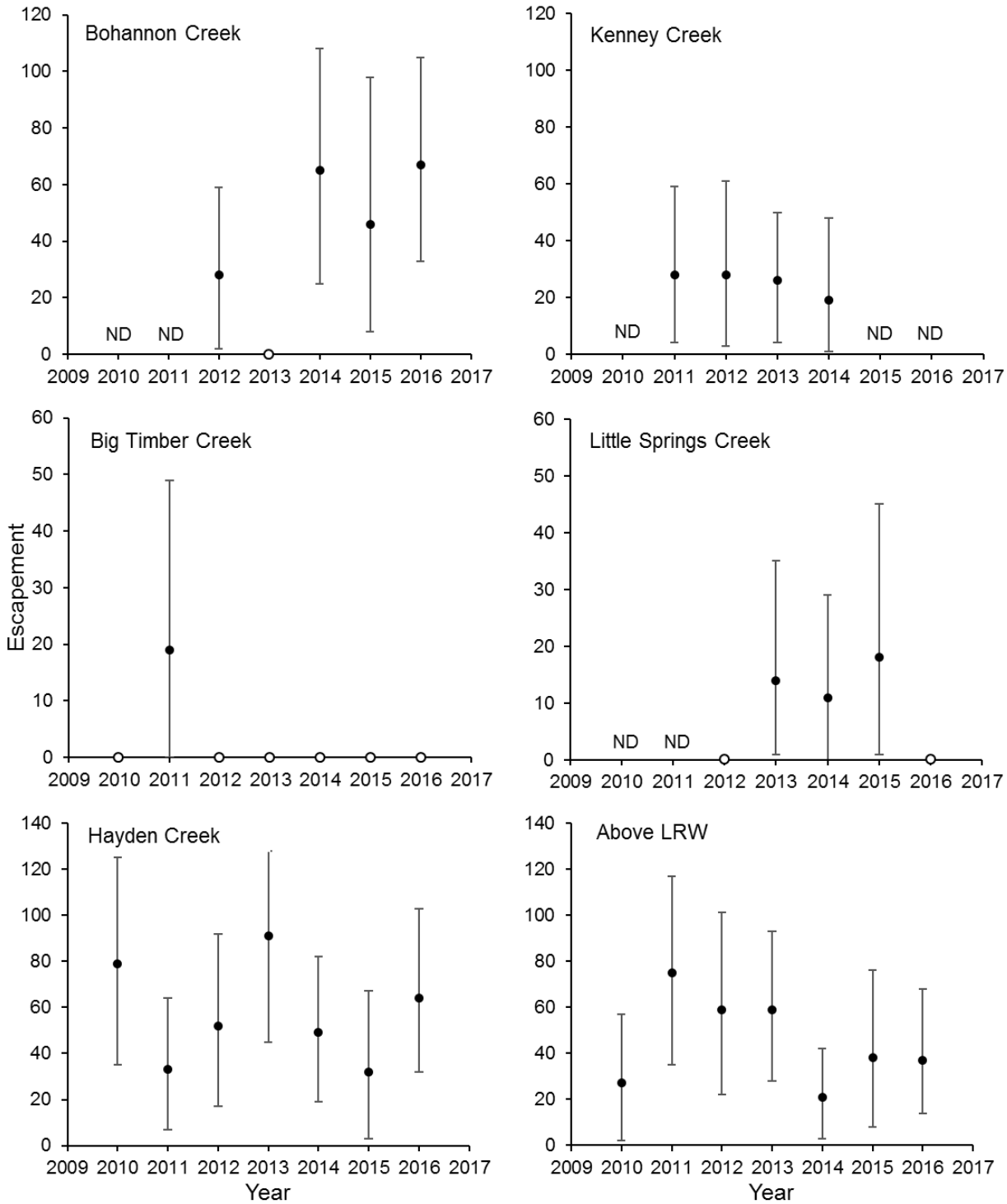

Figure 1.7. PIT-tag based estimates of steelhead escapement in priority tributaries and the reference tributary, Hayden Creek (QCI, unpublished data). Estimates shown with 95% confidence intervals. Hawley and Canyon creeks are omitted because no adult steelhead were detected on those PIT-tag arrays. ND = no data available. Above LRW = main-stem Lemhi River upstream of Hayden Creek.

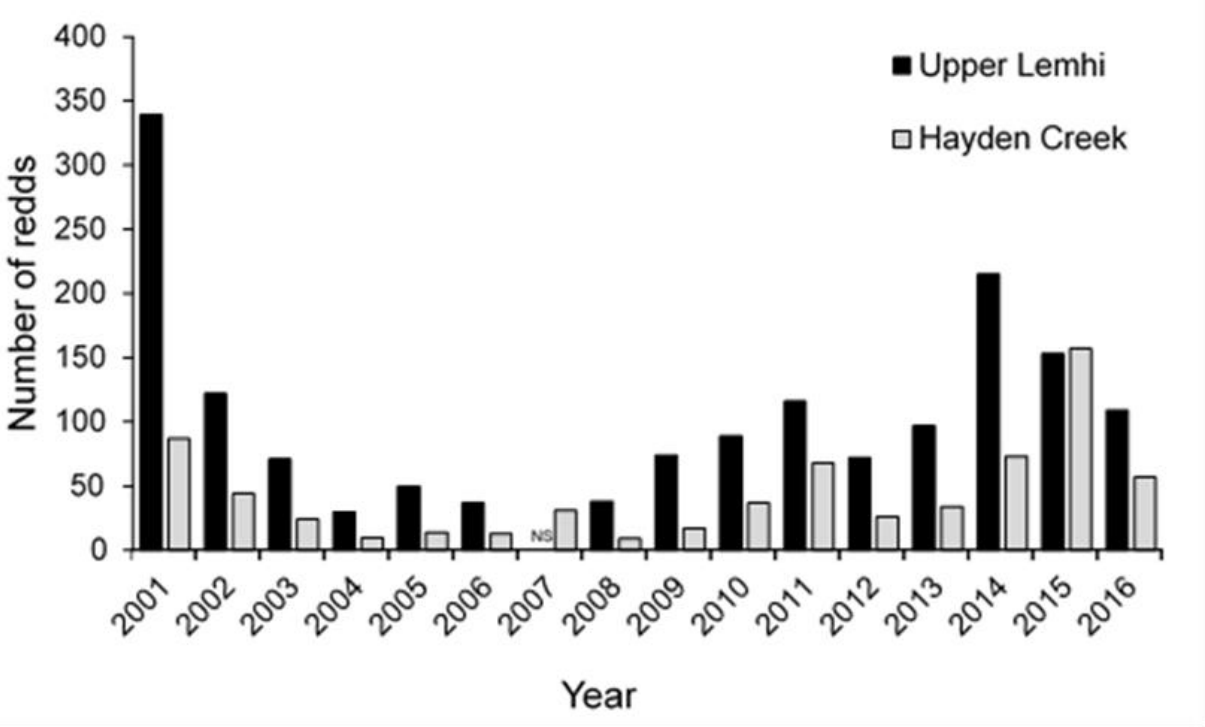

Figure 1.8. Redd counts from annual Chinook Salmon spawning ground surveys in the upper Lemhi River (black) and Hayden Creek (grey). Surveys were not conducted in the Lemhi River in 2007. NS = not sampled.

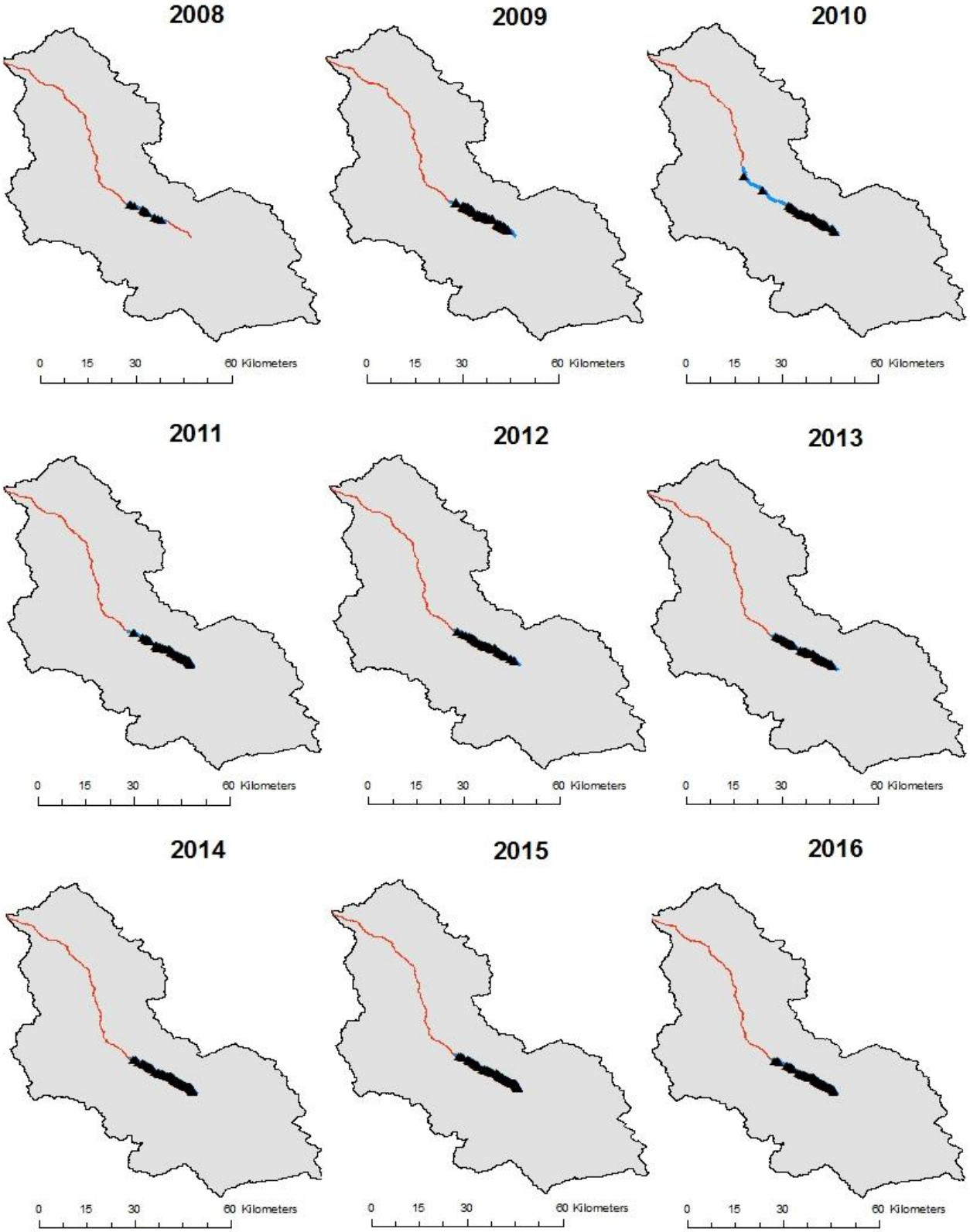

Figure 1.9. Locations of Chinook Salmon redds observed during annual spawning ground surveys in the upper Lemhi River, 2008-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

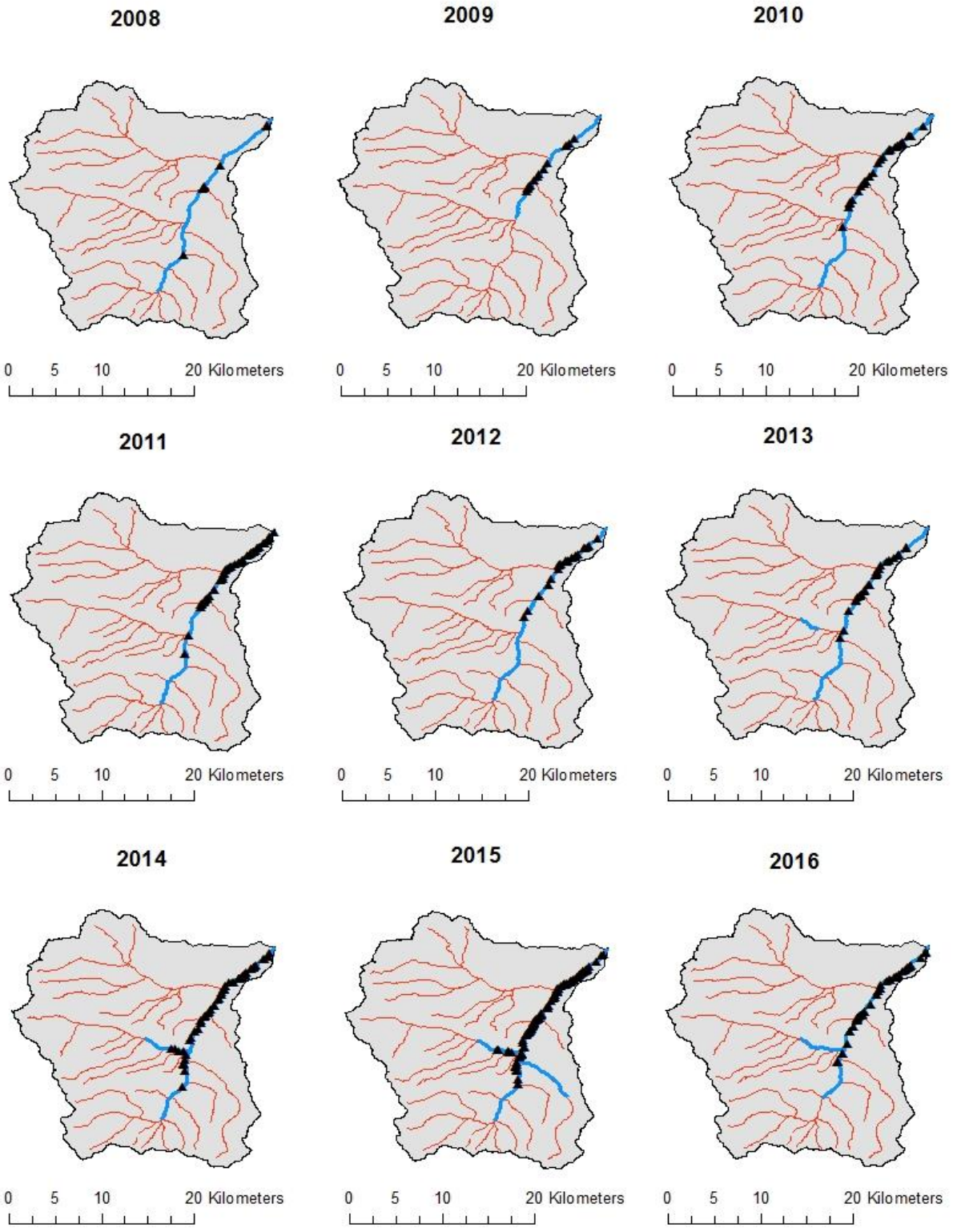

Figure 1.10. Locations of Chinook Salmon redds observed during annual spawning ground surveys in Hayden Creek, 2008-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

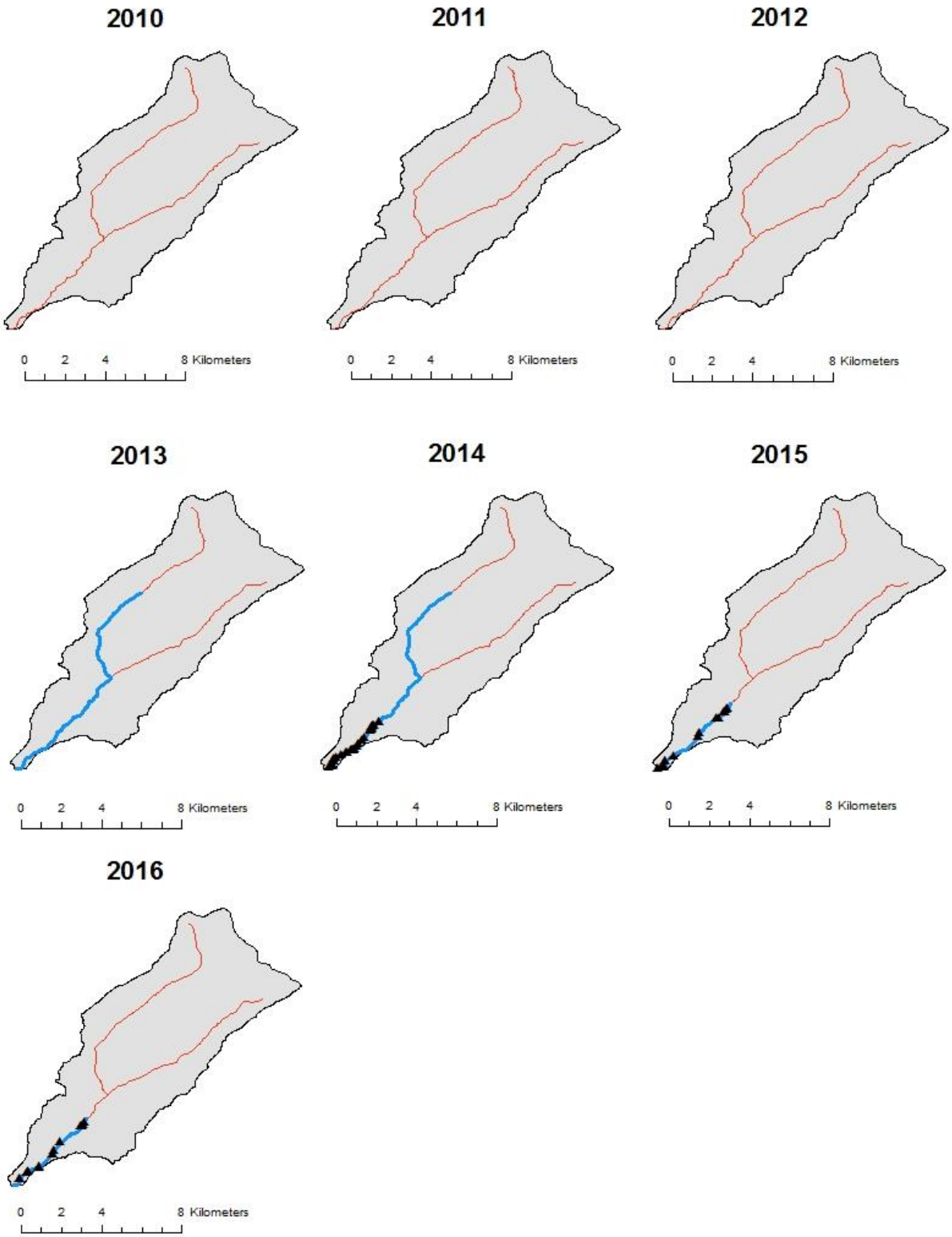

Figure 1.11. Locations of steelhead redds observed during annual spawning ground surveys in Bohannon Creek, 2010-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

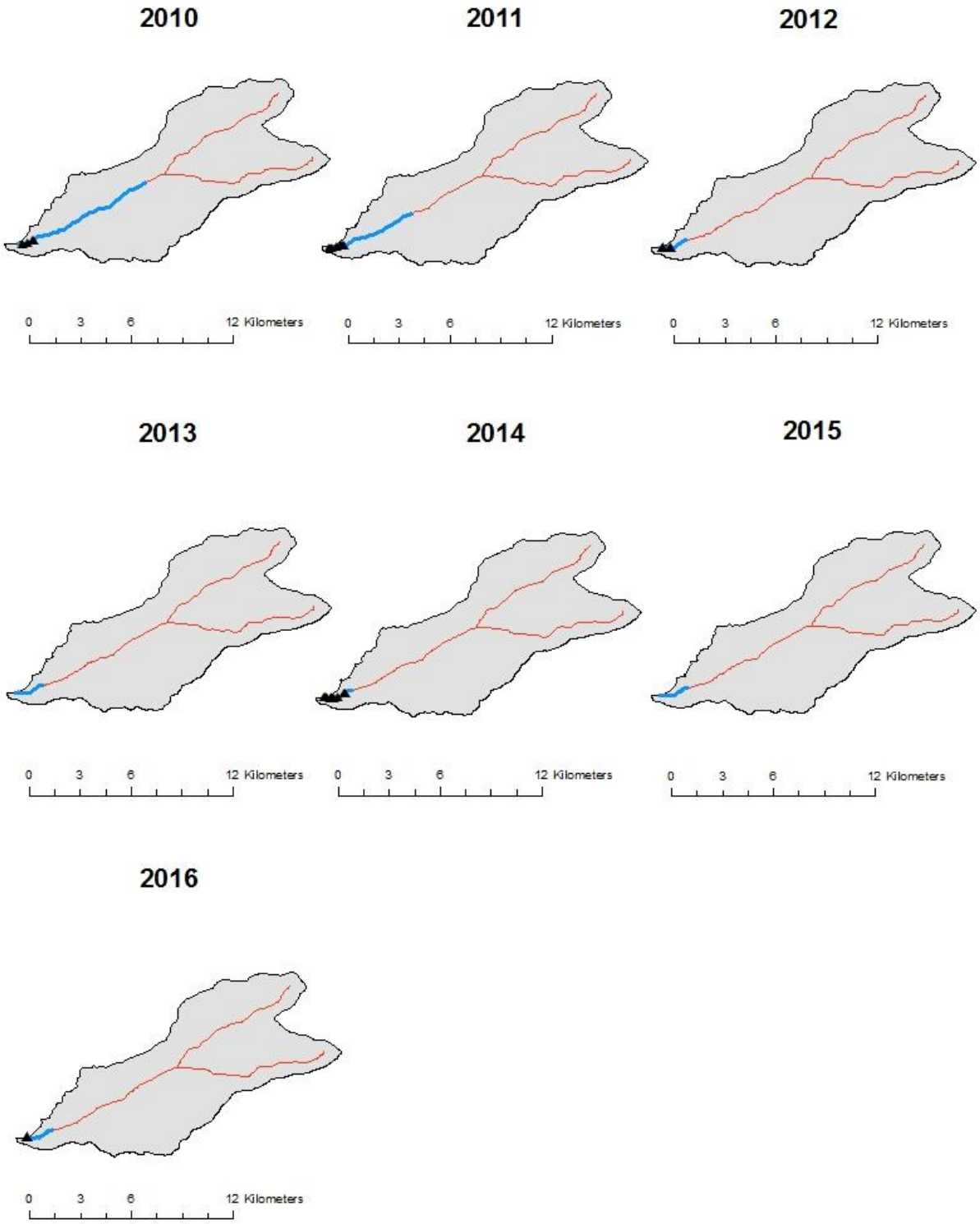

Figure 1.12. Locations of steelhead redds observed during annual spawning ground surveys in Kenney Creek, 2010-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

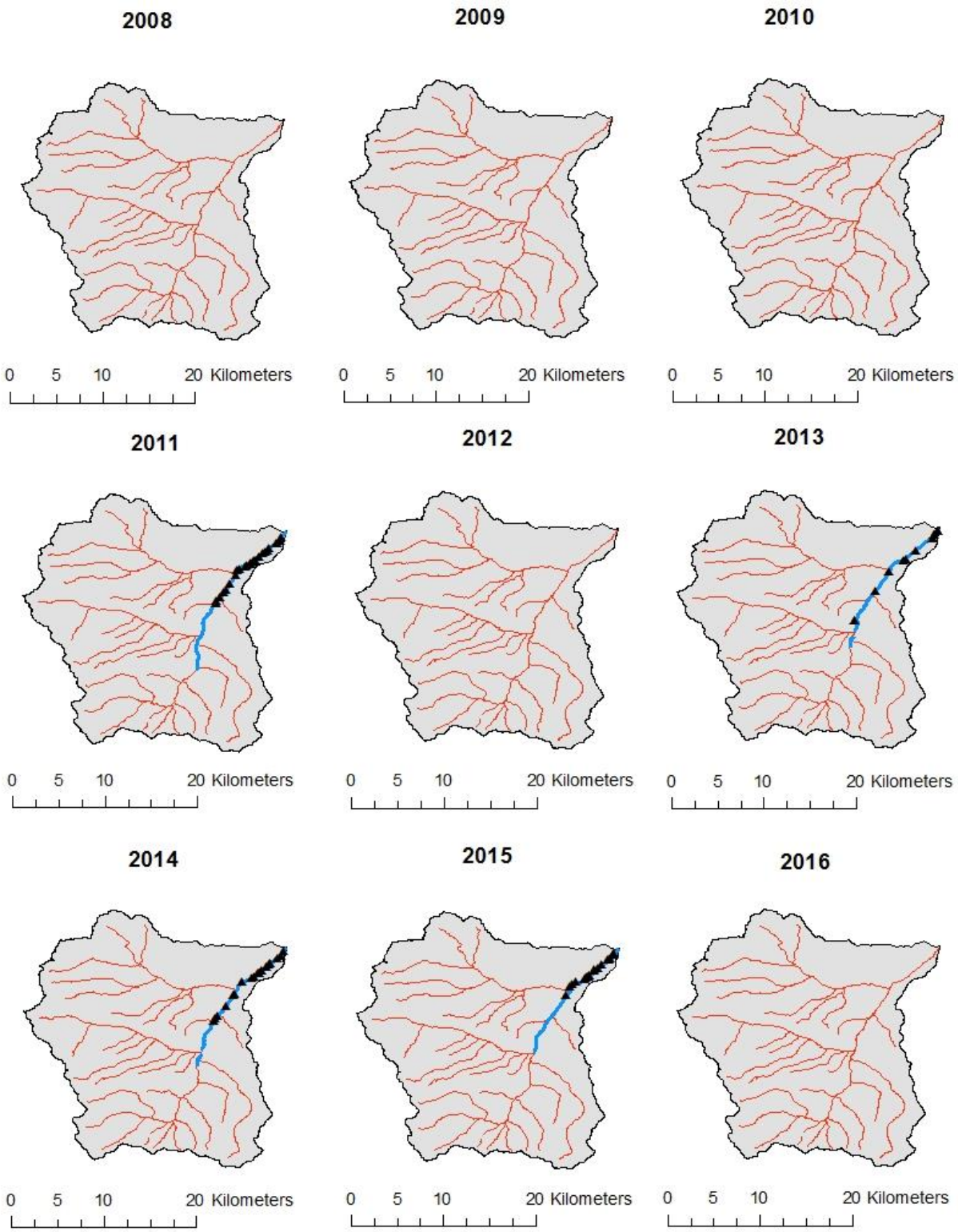

Figure 1.13. Locations of steelhead redds observed during annual spawning ground surveys in Hayden Creek, 2008-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

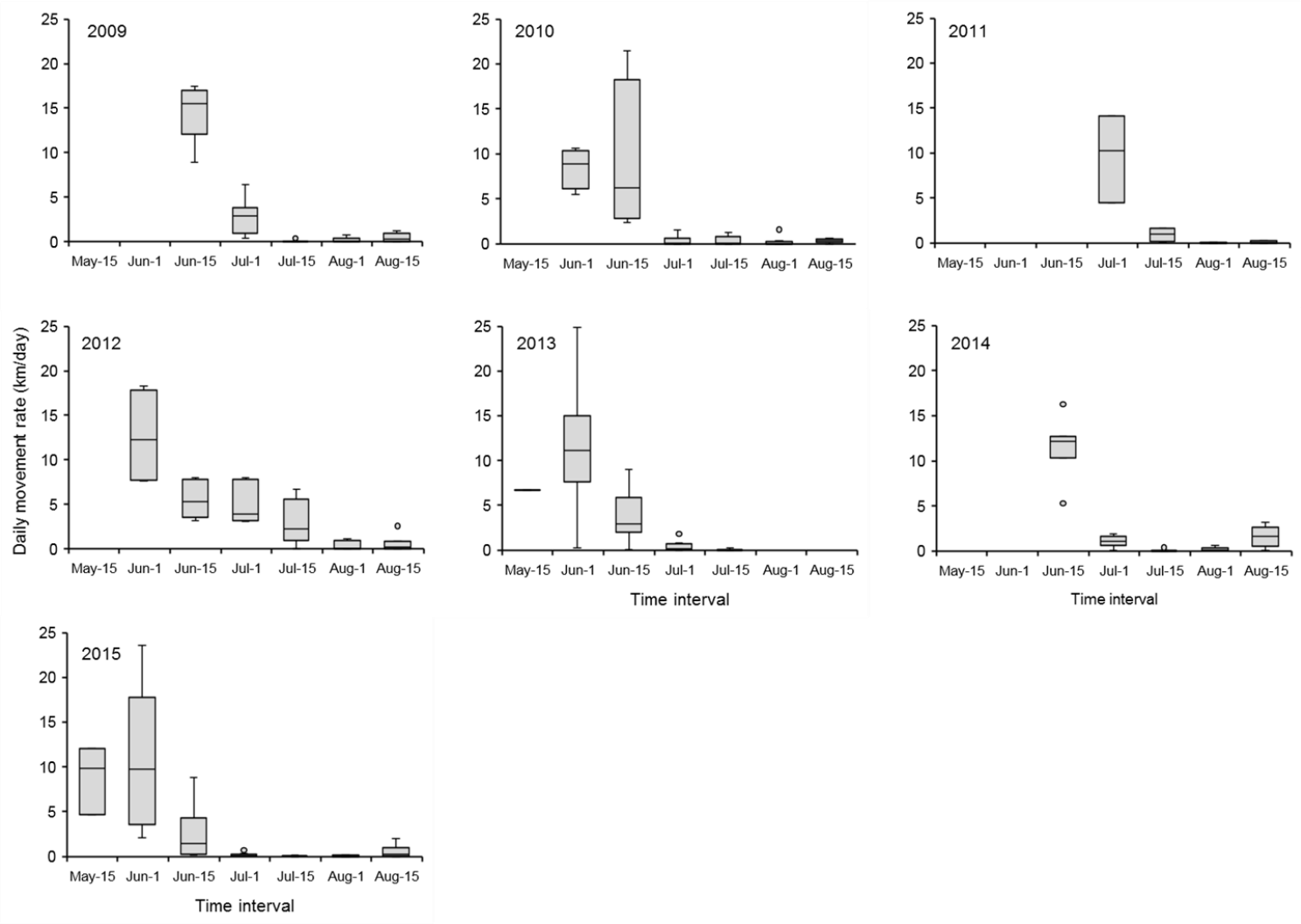

Figure 1.14. Mean daily movement rates of radio tagged adult Chinook Salmon during bi-weekly intervals, 2009-2015. No radio tagged individuals were detected with telemetry equipment in 2016.

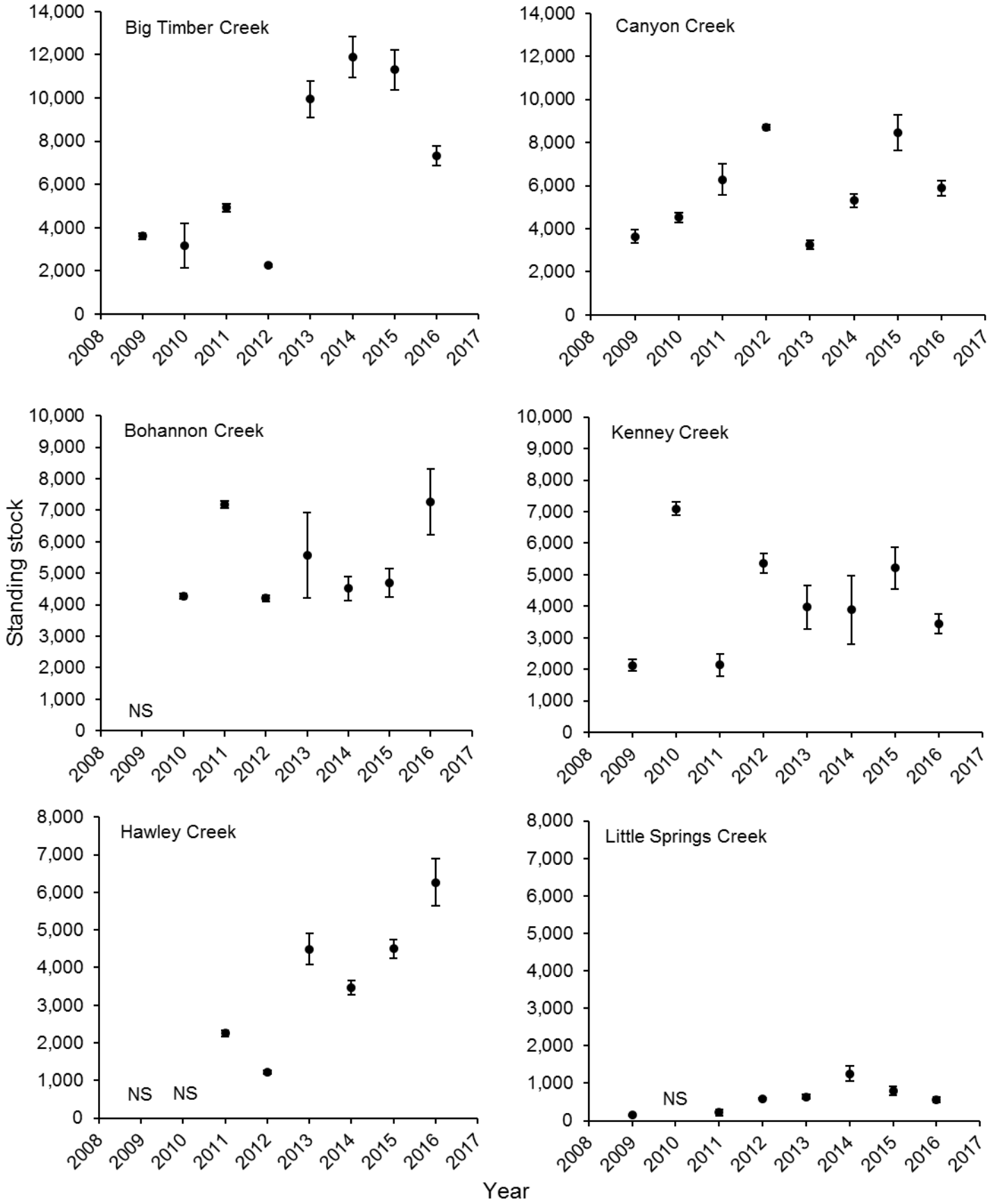
Big Timber Creek
Canyon Creek
Bohannon Creek
Kenney Creek
Hawley Creek
Little Springs Creek
NS
NS
NS
NS
Standing stock
Year
0
2,000
4,000
6,000
8,000
10,000
12,000
14,000
1,000
3,000
5,000
7,000
9,000
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017

Figure 1.15. Estimates of juvenile steelhead standing stock in the six priority tributaries. Estimates are shown with standard errors. NS = not sampled.

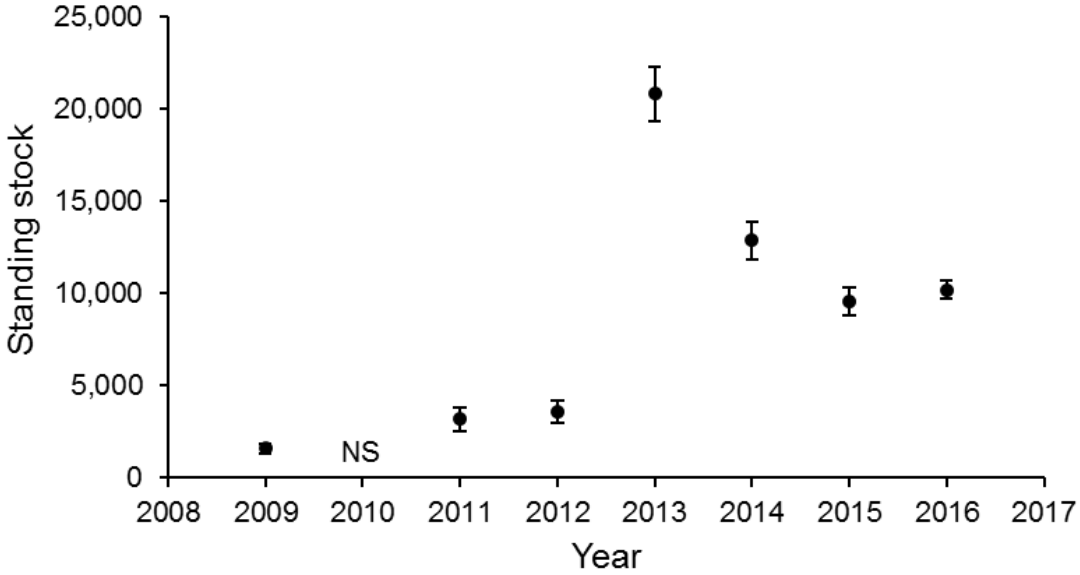

Figure 1.16. Estimates of juvenile steelhead standing stock in the reference tributary, Hayden Creek. Estimates are shown with standard errors. NS = not sampled.

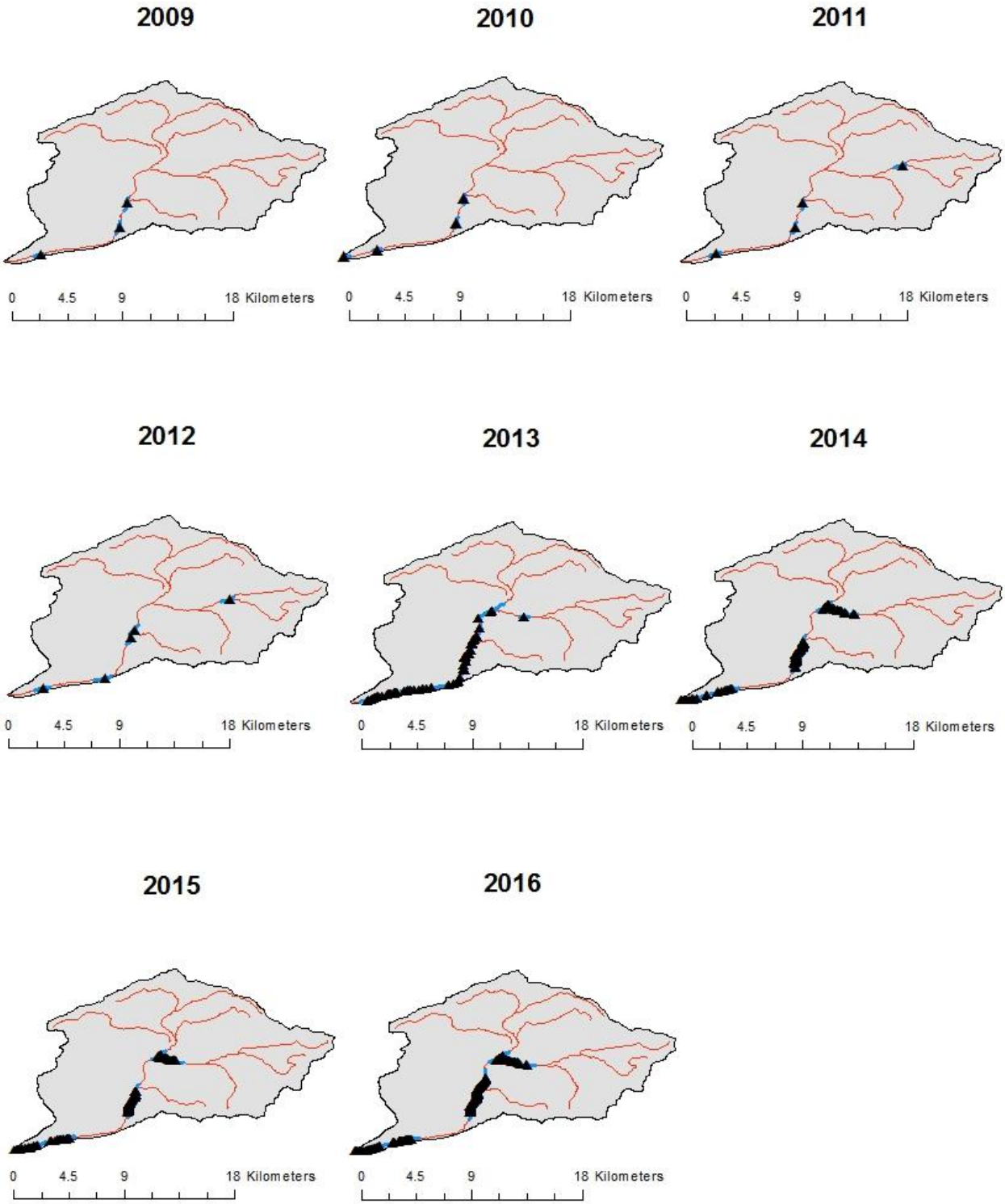

Figure 1.17. Distribution of steelhead encountered during annual summer electrofishing surveys in Canyon Creek, 2009-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

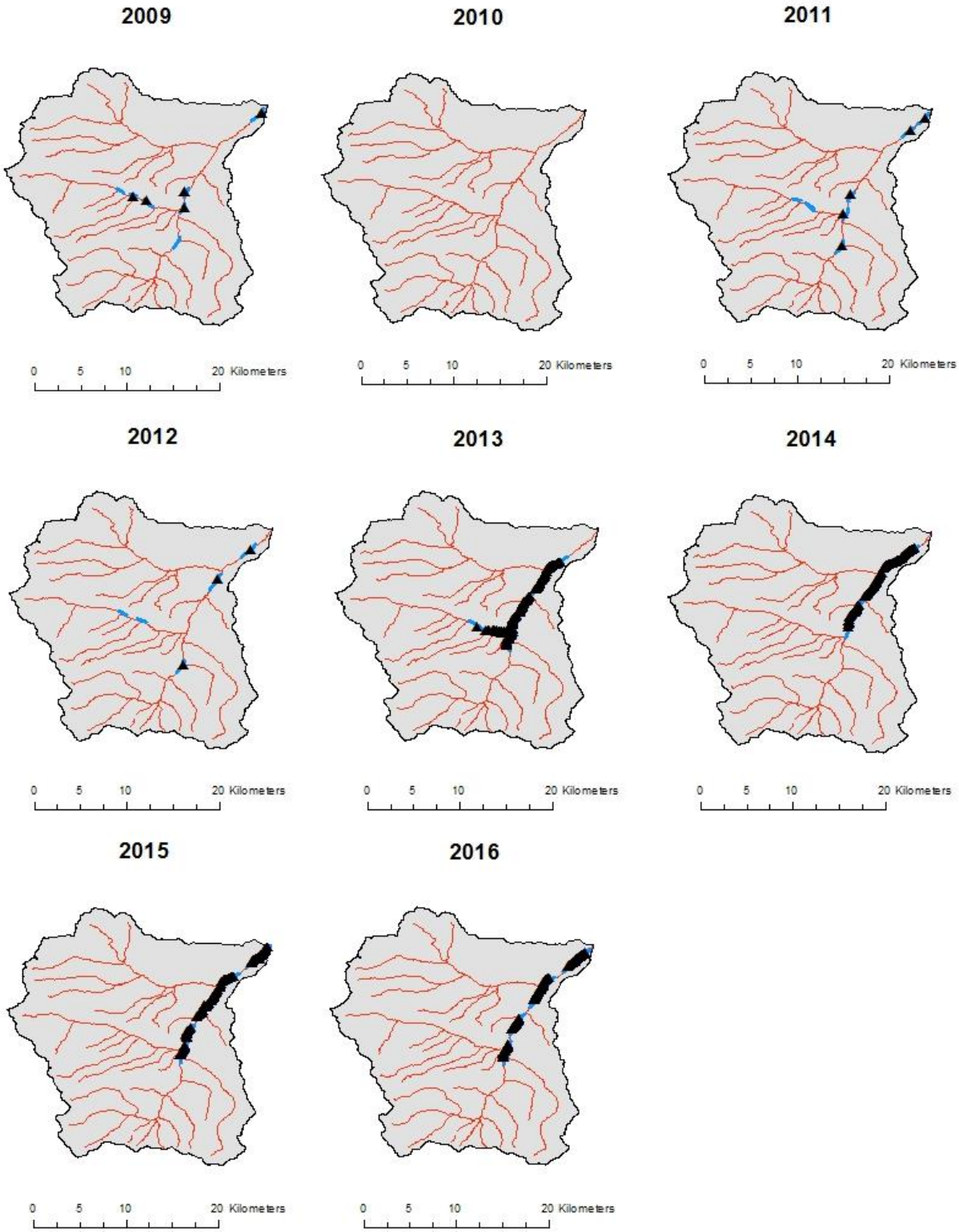

Figure 1.18. Distribution of steelhead encountered during annual summer electrofishing surveys in Hayden Creek, 2009-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

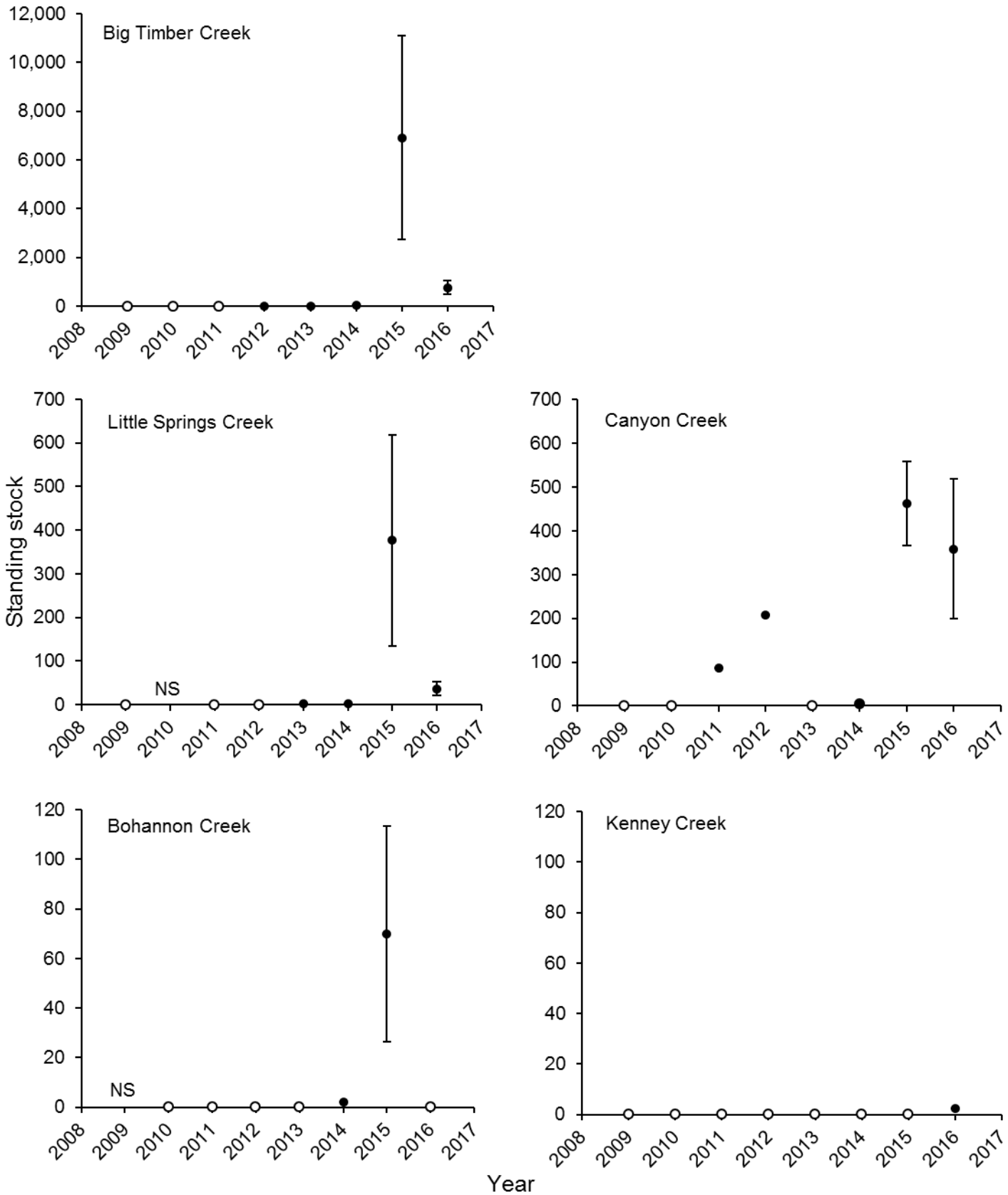

Figure 1.19. Estimates of juvenile Chinook Salmon standing stock in the priority tributaries.

Hawley Creek is omitted because juvenile Chinook have not been encountered during any surveys. Open circles represent estimates equal to zero. Estimates are shown with standard errors. NS = not sampled.

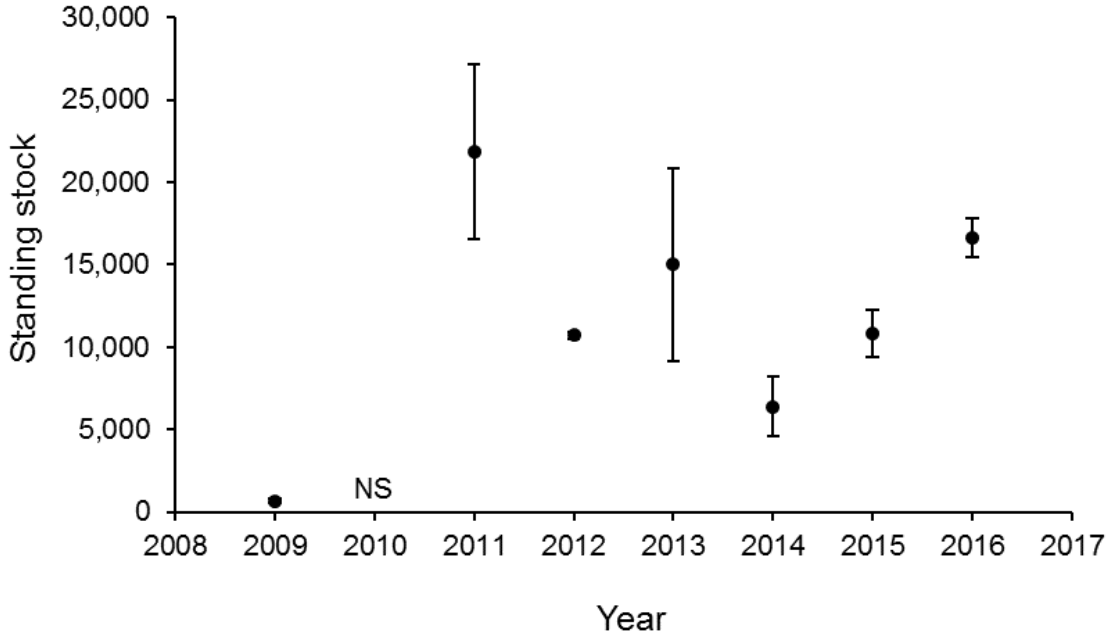

Figure 1.20. Estimates of juvenile Chinook Salmon standing stock in the reference tributary, Hayden Creek. Estimates are shown with standard errors. NS = not sampled.

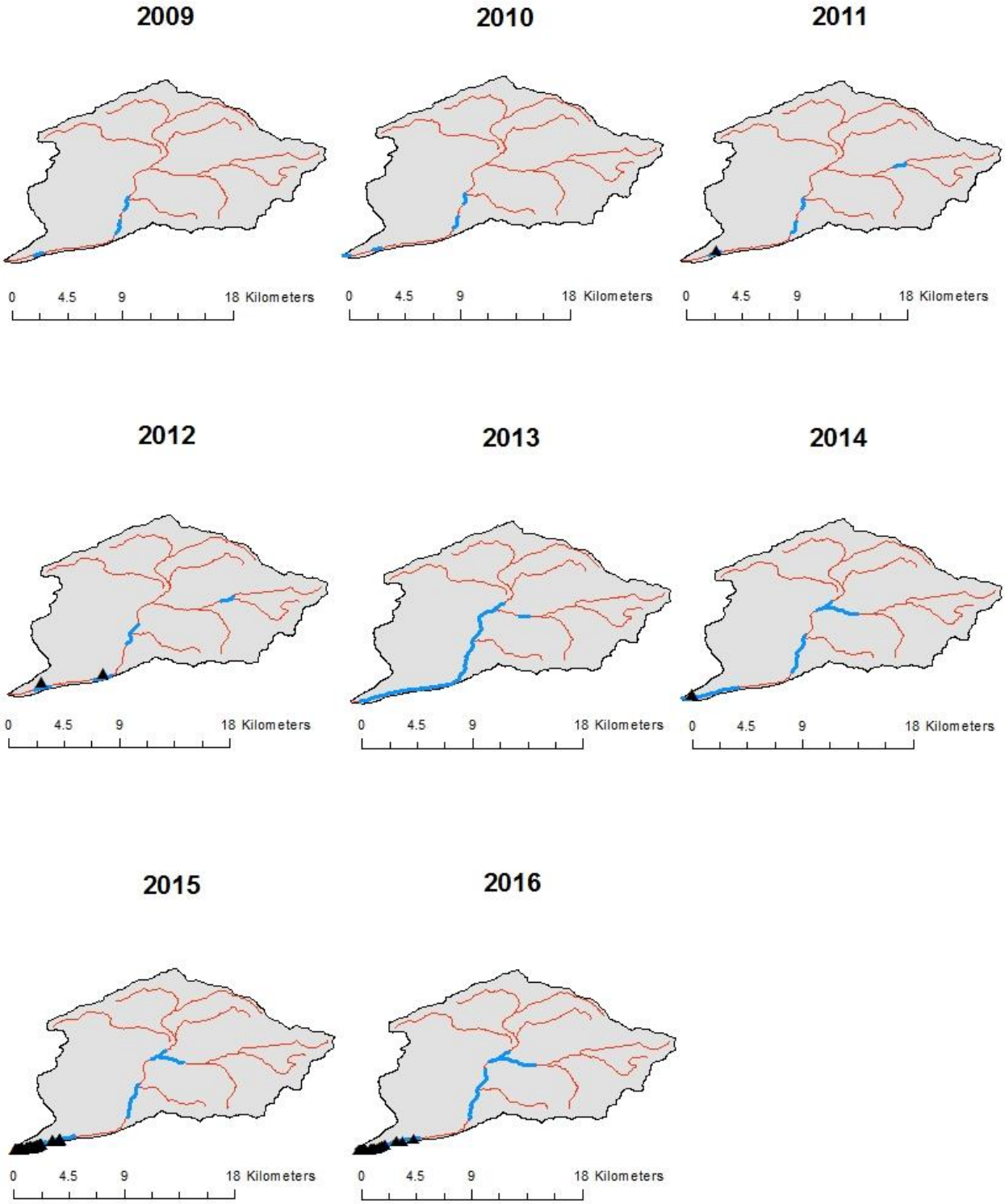

Figure 1.21. Distribution of juvenile Chinook Salmon encountered during annual summer electrofishing surveys in Canyon Creek, 2009-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

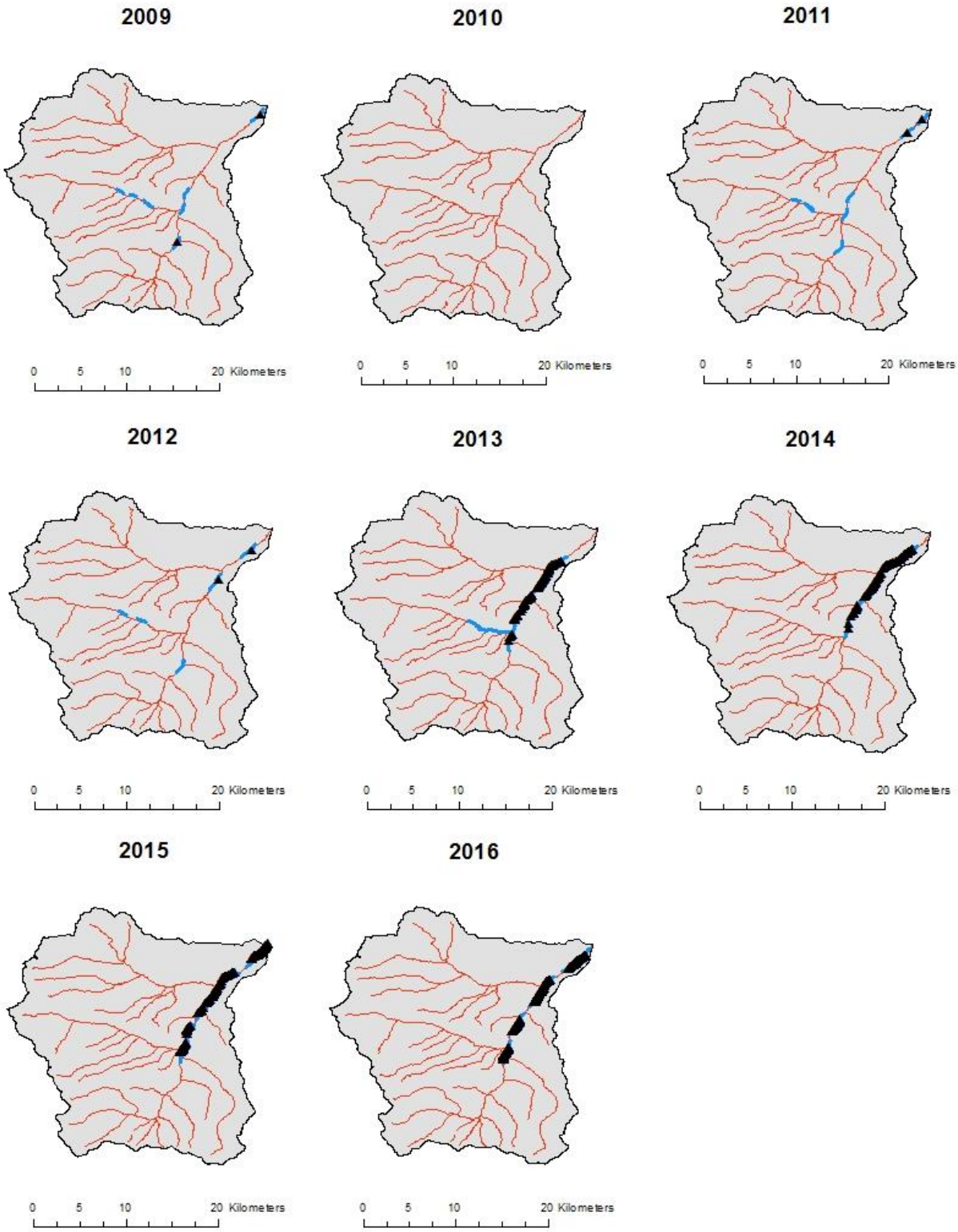

Figure 1.22. Distribution of juvenile Chinook Salmon encountered during annual summer electrofishing surveys in Hayden Creek, 2009-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

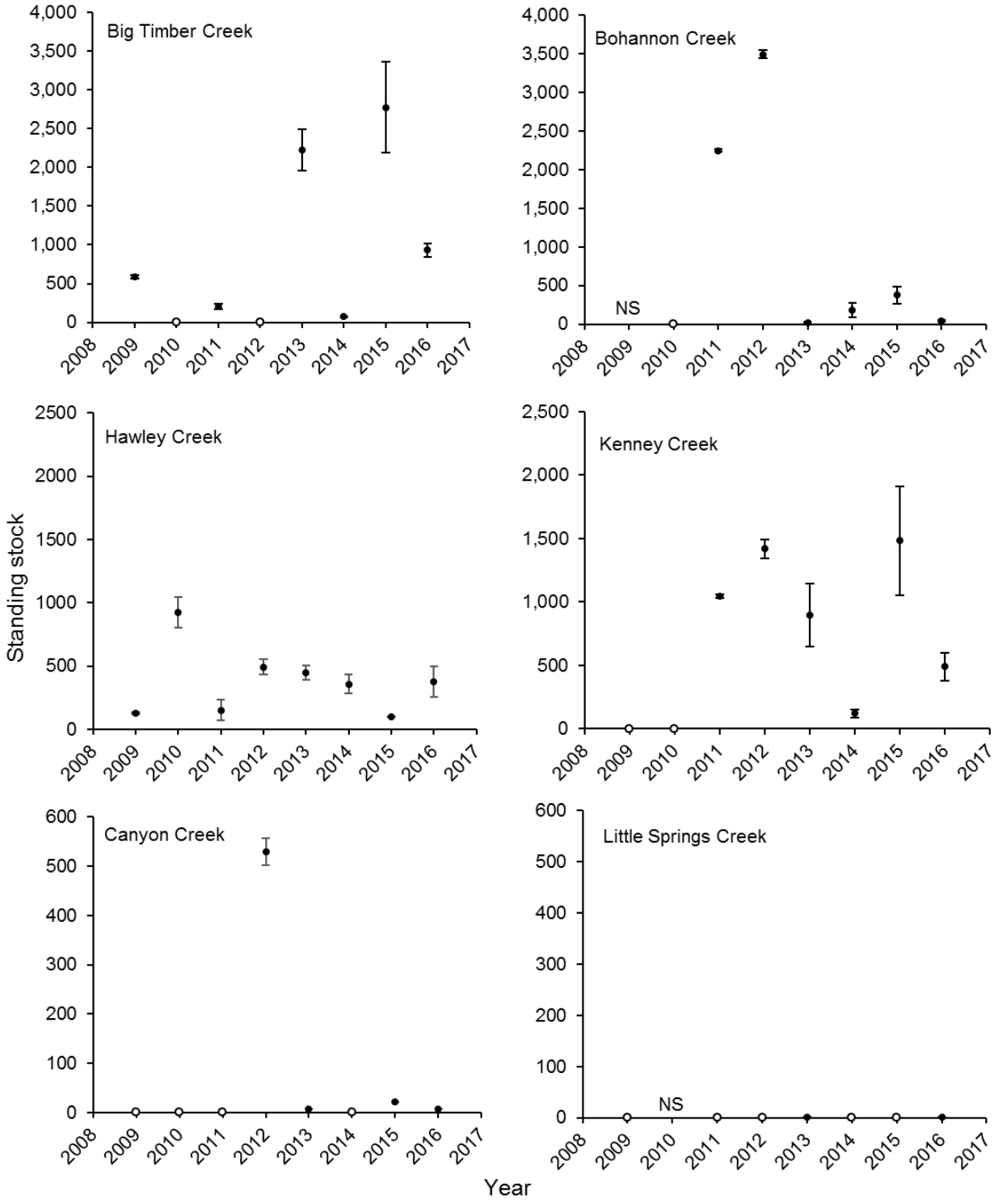

Figure 1.23. Estimates of Bull Trout standing stock in the six priority tributaries. Estimates are shown with standard errors. Open circles represent estimates equal to zero. NS = not sampled.

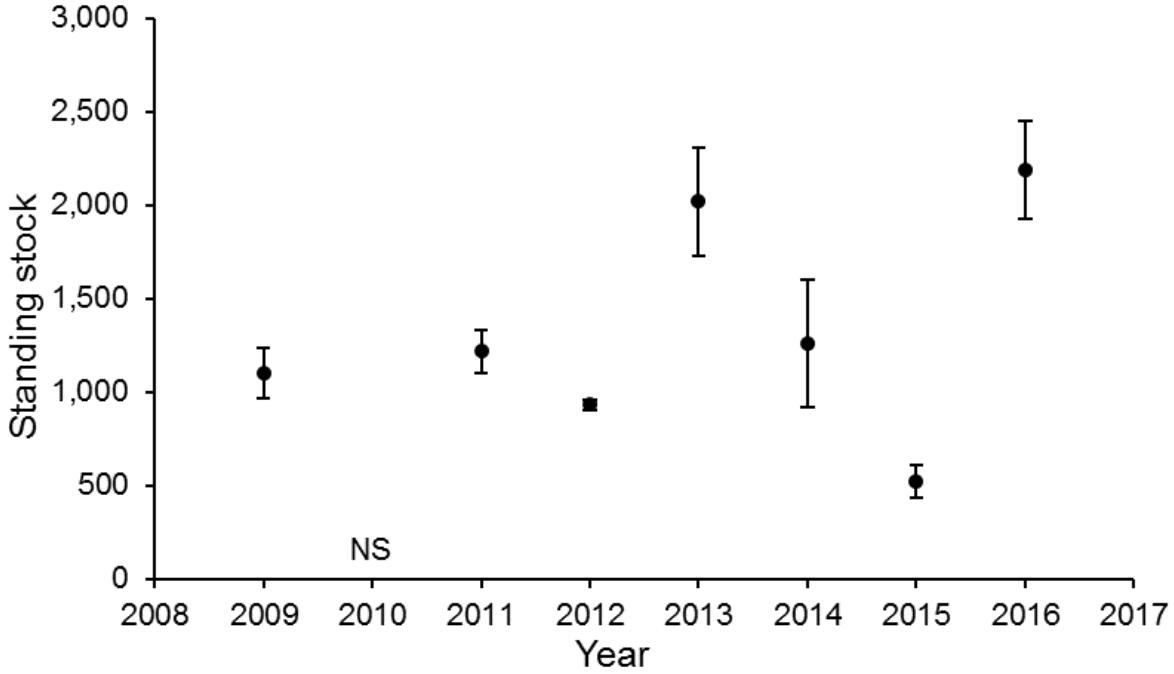

Figure 1.24. Estimates of Bull Trout standing stock in the reference tributary, Hayden Creek. Estimates are shown with standard errors. NS = not sampled.

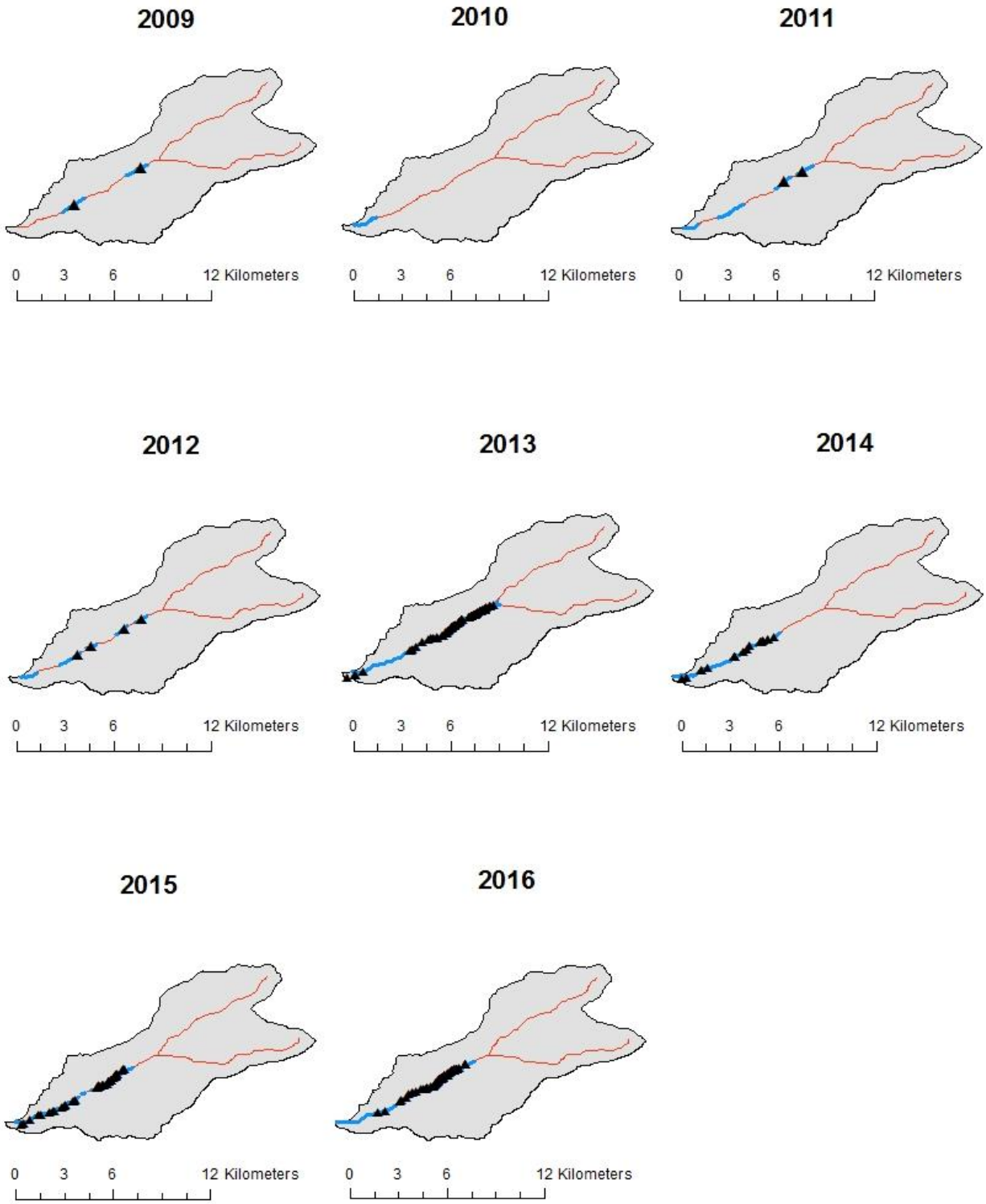

Figure 1.25. Distribution of Bull Trout encountered during annual summer electrofishing surveys in Kenney Creek, 2009-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

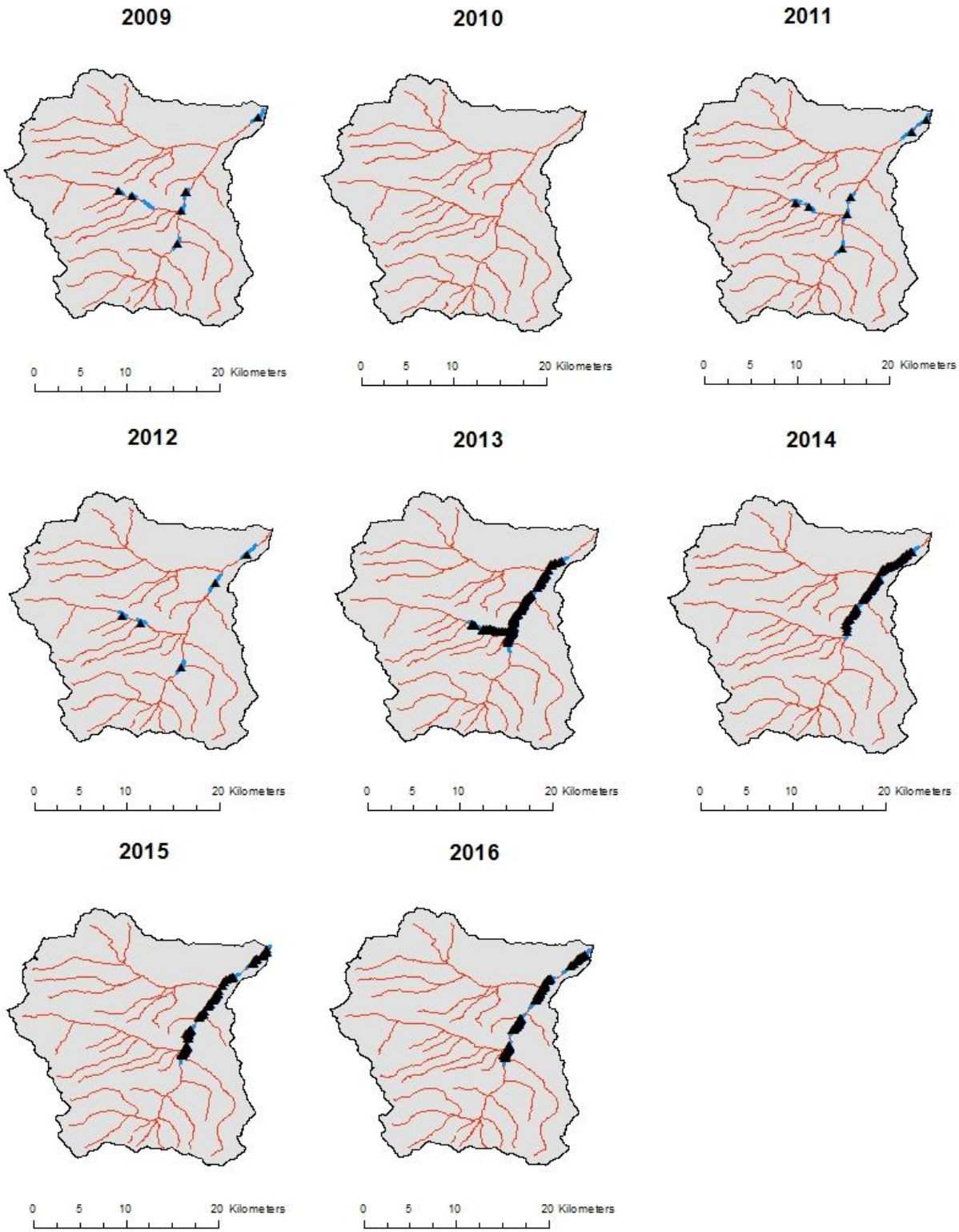

Figure 1.26. Distribution of Bull Trout encountered during annual summer electrofishing surveys in Hayden Creek, 2009-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

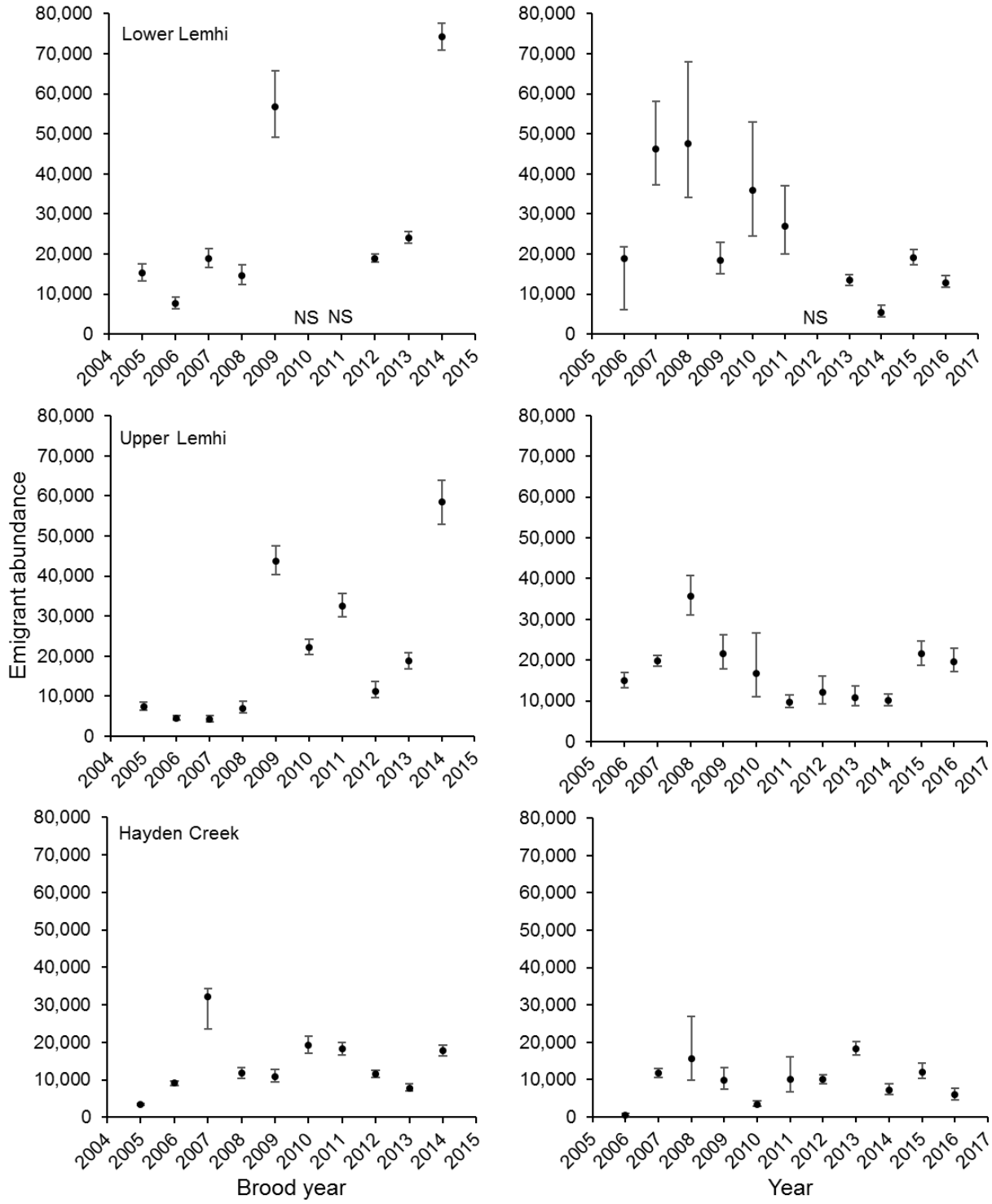

Figure 1.27. Estimates of emigrant abundance at rotary screw traps for Chinook Salmon (left panel) and steelhead (right panel) migrating from the lower Lemhi River (top row), upper Lemhi River (middle row), and Hayden Creek (bottom row). Estimates are shown with 95% confidence intervals. NS = not sampled.

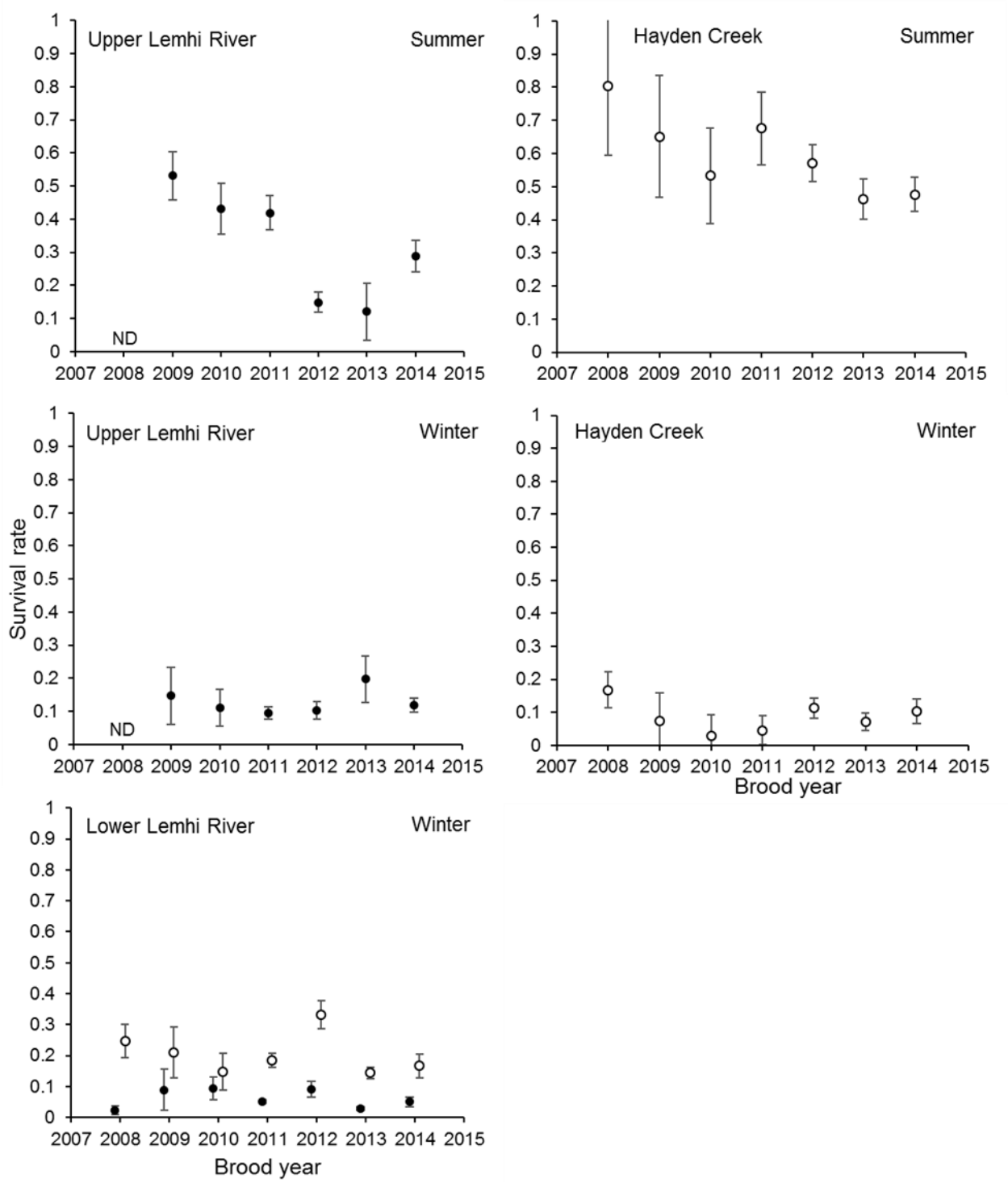

Figure 1.28. Summer/fall and winter survival rates of upper Lemhi River (closed) and Hayden Creek (open) sub-yearling Chinook Salmon in reaches of the Lemhi River basin for brood years 2008-2014 (QCI, unpublished data). ND = no data available.

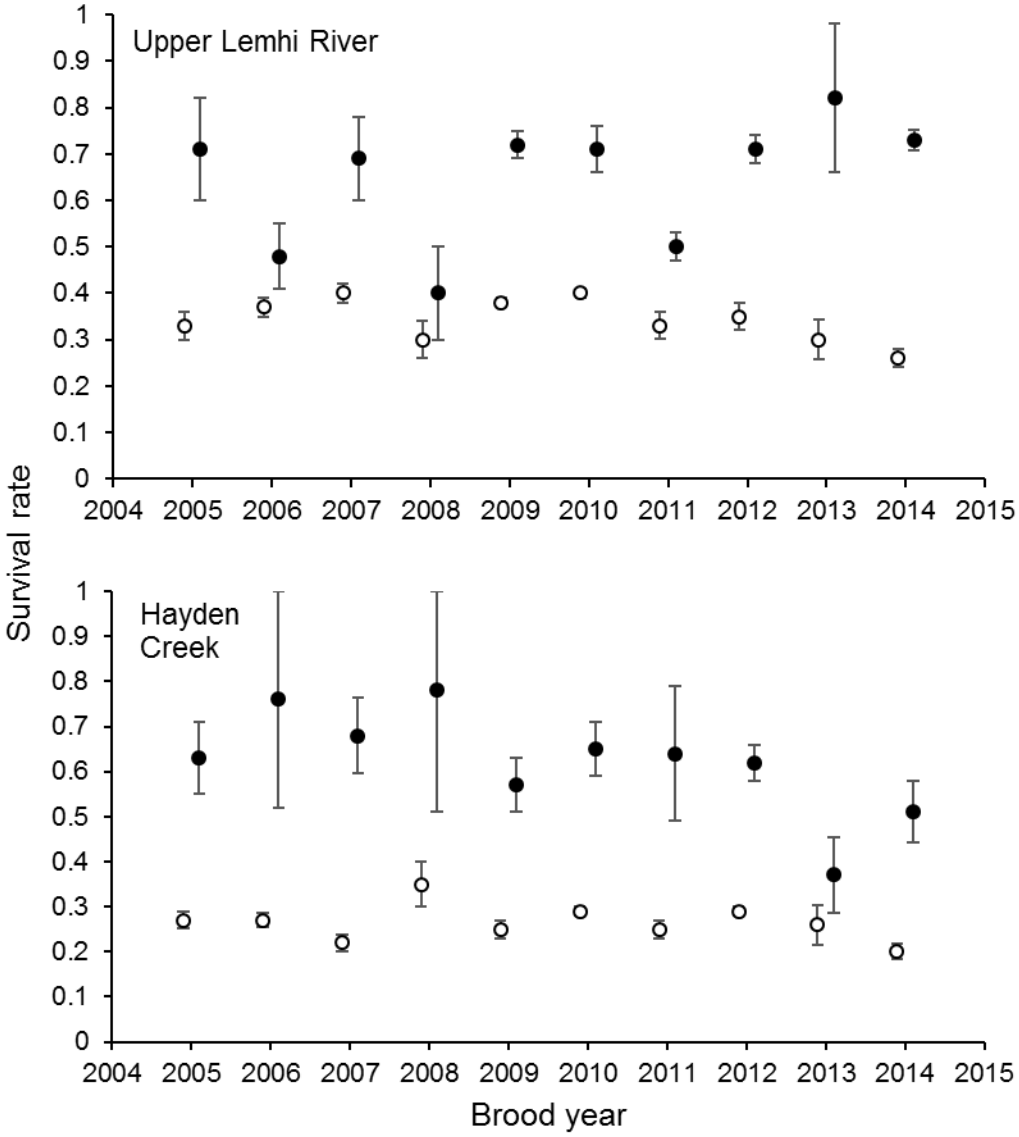

Figure 1.29. Fall parr (open) and age-1 smolt (closed) survival rates to Lower Granite Dam estimated from the upper Lemhi RST (top) and Hayden Creek RST (bottom) for brood years 2005-2014. Estimates are shown with standard errors.

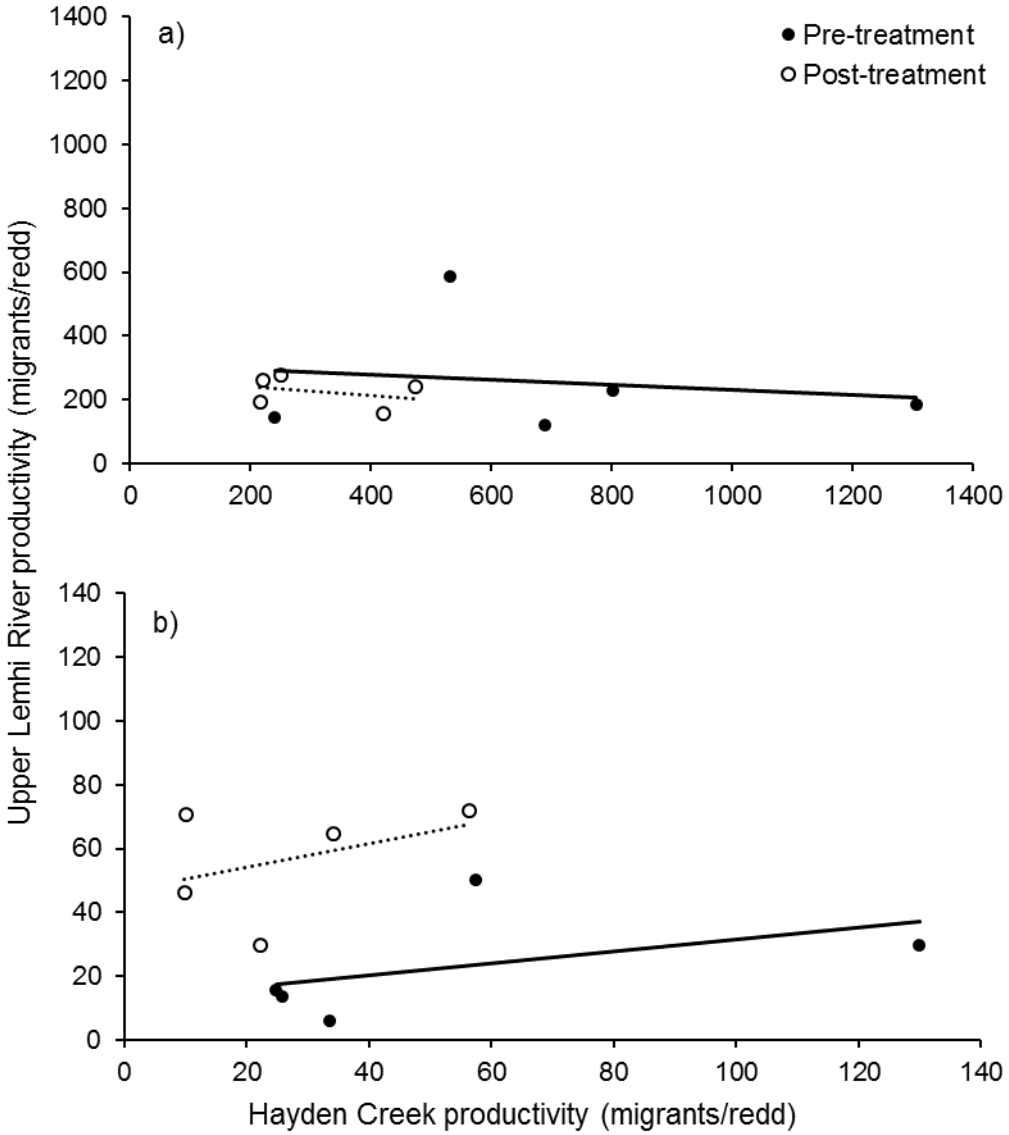

Figure 1.30. Relationships between upper Lemhi River productivity and Hayden Creek productivity before and after reconnection efforts commenced. Brood years 2005-2009 are considered pretreatment and brood years 2010-2014 are considered post-treatment. Productivity is measured as total fall parr and age-1 smolts per redd (panel a) and age-1 smolts per redd (panel b).

PART 2. THE POTLATCH RIVER INTENSIVELY MONITORED WATERSHED

INTRODUCTION

Significant restoration efforts are underway to enhance the production and productivity of wild steelhead trout *Oncorhynchus mykiss* (hereafter steelhead) within the Potlatch River basin of northern Idaho. The Potlatch River Intensively Monitored Watershed (IMW) study is designed to rigorously test the effectiveness of stream restoration efforts aimed at increasing freshwater production of steelhead. Intensive monitoring efforts in the Potlatch basin contribute valuable knowledge on fish-habitat relationships and wild steelhead life history, which better enable managers to improve natural spawning populations of salmonids in the Clearwater basin, throughout Idaho, and the Pacific Northwest.

Potlatch River Basin Overview

The Potlatch River basin lies over a large, complex landscape in northern Idaho. The area encompasses 152,622 hectares (377,776 acres) (USDA SCS 1994) and the principle communities within the watershed are Bovill, Deary, Troy, Kendrick, and Juliaetta (Figure 2.1). The Potlatch River main-stem is approximately 89.4 km (55.6 miles) long and is the largest tributary to the lower Clearwater River. It enters the Clearwater River 20 km east of Lewiston, Idaho. Elevations range from 243.8 m (800 feet) at the mouth to 999.8 m (3,280 feet) above sea level in the headwaters. Land use within the basin is principally cropland, pastureland, and rangeland (42%) and forestland (57%, USDA SCS 1994).

The Potlatch River basin is comprised of two distinct parts 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). Important tributaries include Big Bear Creek (BBC), Little Bear Creek (LBC), West Fork Little Bear Creek (WFLBC), and Pine Creek (PNC). The majority of land in the watershed is privately owned and the principle land use is agriculture.

In contrast, 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, dense canopy cover, meadow connectivity, stable banks, small substrate composition, and cooler water temperatures (Bowersox and Brindza 2006). Important tributaries include the East Fork Potlatch River (EFPR) and the West Fork Potlatch River (WFPR). The majority of the land in the watershed is public with large tracts of private timberlands and the principle land use is logging.

Fish Community

The Potlatch River fish community is comprised of anadromous, non-anadromous, native, and non-native species. Steelhead are the only anadromous fish of endemic stock. There is also a resident *O. mykiss* component found in the watershed (Bowersox et al. 2016). Historically, the Potlatch River supported populations of Chinook Salmon *O. tshawytscha* and Pacific Lamprey

Entosphenus tridentatus (Johnston 1993). However, Chinook Salmon, while observed intermittently, have not re-established a self-sustaining population and lamprey have not been documented in the basin over the past 20 years (Brett Bowersox, IDFG, personal communication). Other common species include Redside Shiner *Richardsonius balteatus*, suckers *Catostomus* spp., dace *Rhinichthys* spp., sculpins *Cottus* spp., and Brook Trout *Salvelinus fontinalis*. Less-abundant species include members of the Centrarchidae, Percidae, and Cyprinidae families.

Steelhead in the Clearwater River basin were listed as threatened under the Endangered Species Act in 1997 as part of the Snake River Distinct Population Segment (DPS). Lower Clearwater Mainstem population of steelhead is genetically distinct from other wild Clearwater River steelhead groups (Nielsen et al. 2009; Ackerman et al. 2016) and comprise the only "large" independent population in the Clearwater River major population group (MPG; ICBTRT 2003). As such, the Lower Mainstem Clearwater River steelhead population must achieve viability in order for the Clearwater MPG and the Snake River DPS to become viable (NOAA 2016). The Potlatch River basin supports the largest spawning area of wild steelhead in the Lower Clearwater Mainstem population (ICBTRT 2003; Bowersox et al. 2009). There are no releases of hatchery steelhead in the Potlatch River and hatchery stray rates into the basin are minimal, ranging from 0 – 3% annually (Banks and Bowersox 2015). Even as the basin has been significantly altered over the past 150 years, a wild run of steelhead has persisted; therefore, restoration projects have great potential to enhance Potlatch River steelhead and recovery of the Lower Clearwater Mainstem population.

The first extensive fish and habitat surveys conducted in the Potlatch River basin began in the 1980s. Johnson (1985) documented the extent of anadromous salmonid production and identified key spawning areas in the main-stem river upstream of Cedar Creek. Schriever and Nelson (1999) found juvenile *O. mykiss* distributed throughout the basin, with the highest densities in high-gradient canyon reaches (Schriever and Nelson 1999). Bowersox and Brindza (2006) replicated that study with similar results and found strong correlations between *O. mykiss* parr abundance and large woody debris (LWD) in the upper watershed and riffle/pocket water habitats in the lower watershed. These studies established the groundwork for current monitoring efforts by documenting steelhead abundance and distribution, identifying primary limiting factors, and prioritizing tributaries for future restoration.

Limiting Factors

Land use practices, primarily agriculture and timber harvest, have significantly altered the aquatic habitat and hydrograph in the Potlatch River basin. Limiting factors include extreme flow variation, high summer water temperatures, high sediment loads, and lack of riparian and instream habitat complexity (Johnson 1985; Schriever and Nelson 1999; Bowersox and Brindza 2006). Limiting factors are addressed for lower and upper watersheds separately because of significant differences in stream morphology, hydrology, and land use.

In the lower Potlatch River watershed, the primary limiting factors are low summer base flows and tributary blockages (Johnson 1985; Bowersox and Brindza 2006). The Potlatch River basin receives the bulk (95%) of its annual precipitation from December to June, with little input during the summer months (USDA SCS 1994). Thus, there is a natural pattern of high flow periods in the late winter/early spring followed by decreasing flows through the summer. However, conversion of timbered and meadow terrain into cropland in the uplands and headwaters of lower watershed tributaries has greatly reduced the water storage capacity in the basin, exaggerating spring peaks and extending low flow periods. Low flow conditions are pervasive throughout the watershed and most major tributaries experience flows <0.5 cfs during the summer (Banks and

Bowersox 2015). Tributary blockages are the other major factor limiting steelhead rearing habitat in the lower watershed. Barriers exist on nearly every major tributary, most of which are road culverts upstream of canyon reaches. Approximately 60-100 km of stream in the lower watershed are blocked to some life stage of steelhead (Tiege Ulschmid, IDFG, personal communication).

The primary limiting factor in the upper watershed is a lack of pools and instream cover (Johnson 1985; Schriever and Nelson 1999; Bowersox and Brindza 2006). At the turn of the century, commercial logging operations began in the upper Potlatch River watershed. Logging infrastructure, including rail lines and roads, were built directly up the stream channels. As a result, streams were often straightened or re-located and riparian vegetation and instream woody debris was removed. Presently, streams in the upper watershed lack LWD and other complex habitats, and the riparian communities are not mature enough to actively recruit materials into the streams.

Potlatch River Basin Restoration Overview

Habitat restoration work in the basin is guided by the Potlatch River Watershed Management Plan (Resource Planning Unlimited 2007). The Potlatch River Technical Advisory Group assisted the Latah County Soil and Water Conservation District (LCSWD) in developing the plan using fish, habitat, and water quality information obtained by local, state, and federal investigations in the basin. A prioritization of limiting factors and restoration strategies for key tributaries in the basin were incorporated into the plan.

Restoration strategies address the key limiting factors unique to each watershed. In the lower Potlatch River watershed, the primary restoration strategies are to expand juvenile steelhead rearing habitat and to increase base-flow conditions by removing barriers and supplementing summer stream flows. In the upper watershed, the primary restoration strategies are to improve riparian function and to increase instream habitat complexity by re-meandering stream channels, installing log structures at key locations, planting riparian areas, and excluding cattle with fencing.

A collaborative partnership among private landowners, local communities, and governmental agencies plan and conduct restoration activities in the Potlatch River basin. The primary agencies involved include the LCSWD, Idaho Department of Fish and Game (IDFG), Idaho Office of Species Conservation, Natural Resources Conservation Service (NRCS), and the U.S. Forest Service (USFS). Restoration implementation funding is provided by many sources including: USFS, NRCS, Bonneville Power Administration, and Pacific Coastal Salmon Recovery Funds. Initial restoration efforts were conducted opportunistically with willing landowners throughout the basin; however, since 2008 efforts have been made to direct restoration activities within two index watersheds: BBC in the lower watershed and EFPR in the upper watershed.

Restoration efforts in the BBC watershed have involved primarily barrier removals, riparian plantings, and a flow supplementation project (Figure 2.2). To date, 10 barriers have been removed, >55,000 trees/shrubs were planted, and approximately 16 km of stream were treated with flow supplementation (Appendix F). The flow supplementation project was conducted in 2015 and 2016 as a pilot project. All completed projects have occurred in the upper reaches of treatment tributaries where steelhead are either currently blocked from accessing or do not sustain perennial flows. Although these projects provide positive downstream benefits, their full impact will not be achieved until either passage or flow is restored to these systems. For example, projects completed in BBC are located upstream of Big Bear Falls, a partial barrier to steelhead migration (Bowersox et al. 2016). Therefore, several high priority, large-scale barrier removal and flow supplementation projects are being planned that will provide access to these upstream

projects by opening up approximately 30-40 km of additional spawning and rearing habitat in the watershed (Table 2.1).

More restoration work has been completed in the EFPR watershed, such as LWD/riparian treatments, riparian fencing and plantings, and projects associated with road best management practices (i.e., road rocking or decommissioning for sediment reduction; Figure 2.3). To date, approximately 4.0 km of stream have been treated with LWD/riparian treatments (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 (Appendix F). To date, the majority of projects have been road best management practices and culvert removals in tributaries, although the LWD/riparian treatments have been in the EFPR itself. In the EFPR watershed, juvenile and adult steelhead are distributed primarily in the EFPR rather than its tributaries (Knoth et al., in prep., see spawning distribution results below); consequently, the immediate benefits of most of these projects are unclear. Road treatments and riparian plantings will likely have longer-term benefits to steelhead, while the instream LWD/riparian treatments provide a more immediate impact. There are multiple LWD treatment/riparian restoration projects planned over the next five years that will treat approximately 10-15 km of the EFPR (Table 2.1).

Intensively Monitored Watershed Study

The Potlatch River basin was chosen for intensive habitat effectiveness monitoring because of the robust habitat restoration program and long-term monitoring there. In 2005, monitoring began in the lower Potlatch River watershed using Pacific Coast Salmon Recovery Fund monies. In 2008, monitoring began in the upper Potlatch River watershed using NOAA Fisheries IMW funds. In the first five years, Bowersox and Biggs (2012) established baseline levels of steelhead production and productivity metrics and documented steelhead life history parameters in the index watersheds. In 2013, monitoring expanded to include additional metrics of fish response (juvenile abundance, survival, and growth) to better identify the causal mechanisms of a response (Bennett et al. 2016).

The goal of the Potlatch River IMW study is to evaluate fish and habitat responses to habitat restoration projects in the Potlatch River basin. The study is designed to assess responses 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). The study design allows managers to better understand the relationship between a habitat action and fish response and how localized responses to restoration propagate up to a higher, management-scale level. To implement this design, the specific monitoring objectives were to:

1. Assess steelhead/resident *O. mykiss* production and productivity within the index watersheds (Big Bear Creek and the East Fork Potlatch River) in the Potlatch River basin.
    a. Determine abundance of juvenile steelhead emigrants from the index watersheds.
    b. Estimate adult steelhead escapement into the index watersheds.
    c. Examine adult steelhead spawning distribution and habitat use within the index watersheds.

2. Monitor juvenile steelhead density, survival, and growth in control and treatment tributaries within the upper and lower Potlatch River watersheds.
3. Monitor change in habitat variables associated with habitat restoration projects in the control and treatment tributaries within the upper and lower Potlatch River watersheds.

Given the limiting factors and restoration projects being implemented, we have 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). In the long-term, 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 drainage.
3. LWD treatments are intended to increase the quantity of instream rearing habitat (e.g., 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 tributaries compared to control tributaries (Solazzi et al. 2000). Other potential responses include changes in emigrant age structure and/or length-at-age (Hunt 1988, as cited in Roni 2005).

In this report, we synthesize data collected as part of the Potlatch IMW project and conduct a preliminary evaluation of restoration effectiveness. Report sections are organized by monitoring level (index-watershed, tributary-scale, and reach-scale) to provide a holistic view of restoration and monitoring efforts. We focus the fish and habitat results in the context of restoration projects completed in the two index drainages. In addition, we model anticipated fish responses to future restoration projects with a life cycle model. Lastly, we make recommendations about future directions for the Potlatch River IMW and discuss how results can inform adaptive management of restoration projects within and outside the watershed.

METHODS

Study Design

We used a hierarchical study design with three monitoring levels (index-watershed, tributary-scale, and reach-scale) to evaluate fish and habitat responses to habitat restoration in the basin. The broadest monitoring level (index-watershed monitoring) was intended to measure the total benefits of restoration (sum of all projects) for steelhead in two index watersheds, which have different limiting factors. Multiple life stages were monitored to allow assessment of size- or age- specific responses. The finer, tributary-scale monitoring was intended to isolate responses

by restoration type. Reach-scale monitoring of individual projects assisted with isolating responses by restoration type and provided more specific guidance to the restoration program.

At the broadest scale, the study was a before/after comparison of steelhead production and productivity in two index watersheds (BBC and EFPR, Figure 2.4). The primary response metrics included adult escapement, spawning distribution, juvenile emigration, and smolt-per-female productivity. We used weirs or PIT-tag detections to estimate adult escapement. We assessed spawning distribution using radiotelemetry and summer parr abundance via snorkeling. Juvenile emigration was estimated using mark-recapture techniques at screw traps in each watershed. Age (adults and juveniles) and sex (adults) data were collected annually and used to attribute juvenile production back to the appropriate adult brood year for smolt per female productivity analysis. We used a life-cycle model parameterized with data collected to make predictions of the potential responses to the restoration program. Details on each data collection method are found in the Fish and Habitat Monitoring section below.

Monitoring infrastructure associated with index-watershed monitoring efforts included picket weirs, a resistance board floating weir, PIT-tag detector antenna arrays, and rotary screw traps. For BBC, two picket weirs were located on BBC and LBC approximately 300 m upstream of their confluence and the PIT-tag array and rotary screw trap were located on BBC approximately 400 m upstream of the confluence with the main-stem Potlatch River. For the EFPR, a resistance board floating weir and juvenile screw trap were located on the EFPR about 400 m upstream of the confluence with the main-stem Potlatch River.

At the tributary scale, we conducted an assessment of specific restoration effects on juvenile steelhead production and habitat conditions using a Before/After/Control/Impact (BACI) study design (Roni 2005). The primary response metrics included juvenile steelhead density, survival, and growth which are necessary for identifying the causal mechanisms of a response (Bennett et al. 2016). We used single-pass electrofishing methods to assess juvenile steelhead density in individual tributaries. In addition, roving electrofishing surveys were conducted to PIT tag juvenile steelhead to monitor parr-to-smolt survival and summer-fall growth. We conducted habitat surveys to monitor the key habitat parameters related to the primary limiting factors in each watershed. The BACI design incorporated suitable pretreatment fish and habitat data collected prior to the implementation of the IMW. Data collection details are found in the Fish and Habitat Monitoring sections below.

The treatment tributaries in the lower watershed included BBC, LBC, and WFLBC and the control tributary was Pine Creek (PNC), the adjacent tributary entering the main-stem Potlatch River just upstream of the BBC watershed (Figure 2.4). In the upper watershed, the treatment tributary was the EFPR and the control tributary was the WFPR. We defined the WFPR as the area upstream of the confluence with the EFPR, including portions of the main-stem Potlatch River as well as the major tributaries Cougar, Feather, and Moose creeks.

We monitored several high priority projects at the reach scale to determine the localized response of steelhead (adults and/or juveniles) and habitat conditions to the treatment. We customized monitoring plans to each individual project, but generally either BA or BACI designs (Roni 2005) were utilized. We used the following techniques to monitor fish and habitat responses to individual treatments: radiotelemetry, electroshocking, snorkeling, and remote, continuous sensing of water-quality indicators. Project-specific monitoring details are discussed in the Fish and Habitat Monitoring sections below.

We selected three projects to include in this report because each represents one of the primary restoration treatments applied in the basin (barrier removal, flow supplementation, and LWD installation). The Dutch Flat Dam barrier removal project was completed in WFLBC in the lower watershed in 2013 (Figure 2.2). The Spring Valley Creek (SVC) flow supplementation project was conducted in SVC and LBC in the lower watershed in 2015 and 2016. The Bloom Meadows LWD installation was completed in the EFPR in 2009 (Figure 2.3).

We are currently at different stages of the early treatment phase in the index watersheds; therefore, it is appropriate to present and discuss the watershed-scale and tributary-scale monitoring results differently for each drainage. In the BBC watershed, projects completed to date are not expected to have a large effect due to minimal implementation, so results are presented in terms of long-term trends, rather than pre- and posttreatment terms. Conversely, there has been a greater restoration effort in the EFPR watershed. We expect the completed projects to have a measureable effect so we present the results in pretreatment (2008-2009) and treatment (2010-current) terms. Watershed-scale and tributary-scale results are preliminary and unless noted otherwise, we used graphical comparisons for inference.

Index-Watershed Monitoring

Adult Steelhead Escapement and Diversity—Ultimately, we expect the accumulated benefits of restoration will result in an increase in adult production in the index watersheds. As such, we monitored adult steelhead escapement into the BBC watershed using picket weirs at BBC and LBC to capture migrating adult steelhead from 2005 until 2012. Beginning in 2013, we monitored adult steelhead escapement into the BBC watershed using an instream PIT-tag array. Adult escapement into EFPR was monitored using a resistance-board weir. We installed weirs as early in the spring as possible, typically mid-February (BBC) or March (EFPR), and continued operations until the kelt (post-spawn fish) outmigration was complete. Detailed information on annual operations for both the weirs and PIT-tag array can be found in IDFG annual project reports (Banks and Bowersox 2015; Bowersox 2008; Bowersox et al. 2007; Bowersox et al. 2009, 2011, 2012; Knoth et al. in prep). Sex, weight (g), fork length (FL; mm), and the presence of any marks or tags were recorded for each individual upstream migrating steelhead captured at the weir.

We calculated adult escapement above weirs using software developed by Steinhorst et al. (2004). Within the software, a Bailey's modification of the Lincoln-Peterson estimator was used as follows:

$$N = c(m+1)/(r+1),$$

where *N* is adult abundance, *c* is the number of kelts captured, *m* is the number of adults passed upstream, and *r* is number of marked adults recaptured as kelts. The estimate was computed using an iterative maximization of the log likelihood, assuming fish are captured independently with probability p (equivalent to weir efficiency) and tagged fish mix thoroughly with untagged fish. The 95% confidence intervals were computed with the bootstrap option (2,000 iterations). Model assumptions are that marked and unmarked adults had the same survival during spawning and individual fish were recaptured independently with equal probability.

After 2012, adult escapement in BBC watershed was estimated by expanding PIT-tag detections of adult fish tagged by the Integrated Status and Effectiveness Monitoring Project (BPA Project Number 2003-00-017) at Lower Granite Dam (LGR). Array efficiency was calculated using the USER software program 4.7.0 (Lady and Skalski 2009) with the number of unique upstream,

unique downstream, and shared upstream and downstream detections as inputs. We estimated adult escapement following Connolly et al. (2005, 2008):

$$N = \frac{\left(\frac{d}{s}\right)}{e},$$

where N is adult escapement, d is the number of adult steelhead detected at the array, s is the LGR sampling rate, and e is the detection efficiency of the array.

We measured characteristics of the spawning population relevant to population dynamics. When the weirs were operated, age composition was based on scale samples and sex was determined from physical appearance. When the BBC PIT-tag array was operated (2013-2016) scale samples were collected from PIT-tagged adults at LGR. All scale samples were sent to the Nampa Research Aging Laboratory in Nampa, Idaho for aging (Wright et al. 2015). The sex of adult steelhead was determined from genetic samples of adults PIT tagged at LGR that subsequently migrated to the basin (Powell et al. 2017).

Spawning Distribution—We conducted radio-telemetry studies during two time periods (2010-2011 and 2015-2016) to identify spawning reaches, evaluate potential passage barriers, and assess barrier-removal projects in the two index watersheds. In 2010 and 2011, we gastrically tagged 16 and 12 upstream migrating female steelhead at the BBC and LBC weirs, respectively. In 2015 and 2016, 22 and 19 BBC-origin adults (determined from PIT-tags placed in juveniles and detected in the hydrosystem as adults) were tagged respectively at the LGR fish trap and released upstream. In addition, we gastrically tagged 16 and 19 upstream-migrating adults at the EFPR weir in 2015 and 2016, respectively.

We tracked tagged fish within the Potlatch River basin using a Lotek SRX 600 telemetry receiver by truck, ATV, foot, or fixed-wing aircraft on a weekly or bi-weekly basis during the spawning migration. When tags were located, the date/time, strength of signal, GPS position, and whether the fish was visually observed was recorded. A fixed telemetry site at the mouth of the Potlatch River indicated when a radio-tagged steelhead entered the system. We used the furthest upstream detection of the radio-tagged fish as the proxy spawning location of individual fish. For more detailed information on tagging and tracking protocols see Bowersox et al. (2011, 2012) and Knoth (2016).

Fry and Parr Trend Monitoring—We conducted snorkel surveys to monitor steelhead fry and parr densities to determine whether juvenile production increased after restoration treatments within the BBC watershed. Fry and parr trend monitoring was not conducted in the EFPR watershed. In the BBC watershed, surveys were conducted during early to mid-June during 2008-2016. We selected sample sites using the generalized random-tessellation stratified (GRTS) methodology (Stevens and Olsen 2004) to provide a spatially-balanced panel of survey sites (Figure 6). In most years, a minimum of 20 sites were completed in the BBC watershed to provide enough statistical power to track annual variation in steelhead parr density over time at the watershed scale (Bowersox and Biggs 2012).

Snorkeling methods were consistent with protocols outlined in Apperson et al. (2015). Trout fry <75 mm were not identified to species but were considered *O. mykiss* because there are no other spring-spawning salmonids present in the watershed (Johnson 1985, Schriever and Nelson 1999, Bowersox and Brindza 2006). Non-steelhead fish presence and density was recorded during surveys but is not included in this report. Data from all species and habitat

characteristics were entered into the IDFG Stream Survey database (http://fishandgame.idaho.gov/ifwis/StreamReports). Steelhead fry and parr densities (number/100 m^2) were averaged across all sites in the watershed to produce a single watershed-wide estimate for each year. Annual estimates were plotted to visually assess long-term trends in pretreatment data and will be compared to post-treatment estimates once restoration treatments are completed in the watershed.

Juvenile Steelhead Emigration, Diversity, and Survival—We operated rotary screw traps to monitor juvenile emigrant production and life history diversity in response to habitat treatments in BBC and the EFPR. We began trapping as early in winter or spring as possible, typically late January (BBC) or February (EFPR) and continued until early June when low flows prevented screw trap operation. Trapping was resumed in the fall at both sites when sufficient flows and funding allowed (2005, 2006, 2008-2010, 2012, 2013, and 2016 in BBC; 2008, 2009, 2011-2013, and 2016 in EFPR). Detailed information on annual operational periods for the BBC and EFPR rotary screw traps can be found in Banks and Bowersox (2015), Bowersox (2008), Bowersox et al. (2007), Bowersox et al. (2009, 2011, 2012), and Apperson et al. (2016).

We calculated emigration estimates from trap operations with the stratified Lincoln-Petersen estimator with Bailey's modification as previously explained in the adult escapement section. The trapping season at BBC was divided into periods based upon trapping efficiency. A running average of weekly trapping efficiency was plotted to group periods of similar recapture probability. A single, average trapping efficiency was calculated for the EFPR trap due to low trap efficiency across the entire season.

Scales were used to establish age composition of juvenile emigrants and assign age proportions to juvenile emigration estimates. Scales were collected from every fifth steelhead to achieve a representative sample over the entire juvenile emigration period. Scales were sampled posterior to the dorsal fin above the lateral line and stored on water-resistant paper inside scale envelopes. Scale samples were sent to the Nampa Research Aging Laboratory in Nampa, Idaho to be aged (Wright et al. 2015).

We estimated apparent survival from rotary screw traps to LGR to assess trends in smolt productivity. We annually queried PTAGIS for detections of juveniles tagged at Potlatch River screw traps during spring or the previous fall. Potential interrogation sites in the hydrosystem were Lower Granite, Little Goose, Lower Monumental, McNary, John Day, and Bonneville dams and the estuary towed array. Juvenile emigrant survival indices to LGR were estimated using PitPro 4.19 (Westhagen and Skalski 2009). The PitPro algorithm combines capture-recapture PIT-tag interrogation data from instream arrays and hydrosystem passage facilities into a Cormack-Jolly-Seber model to estimate survival and capture probabilities to tributary and hydrosystem arrays. Apparent survival rates are biased low since some individuals will not emigrate until subsequent years or may residualize within the Potlatch River basin but we consider them to be a useful index until a better analytical model can be configured.

Productivity Estimates—We computed freshwater productivity estimates for juvenile recruits/female spawner in each of the index watersheds to measure population-level responses to restoration projects. We are still compiling the pretreatment productivity dataset and will evaluate the full before/after design once planned restoration projects are completed in the index watersheds. Annual abundances of female spawners were calculated by applying the observed sex ratio at the weir or array to the total adult escapement estimate. Juvenile production associated with that brood year of female spawners was estimated by applying age class proportions to subsequent years' juvenile emigration estimates (i.e., brood year 2011 females

produce age-1 juveniles in 2012, age-2 juveniles in 2013, etc.). The total juvenile outmigration produced by that brood year was divided by the number of female spawners estimated in each brood year to estimate juvenile recruits/female spawner.

Life Cycle Modeling—We used a life cycle model to examine watershed-scale responses in juvenile steelhead production following the implementation of restoration projects in each index watershed. In the BBC modeling scenario, we modeled the potential increase in juvenile steelhead production (number of smolts) following the completion of three large-scale restoration projects in the BBC watershed. The planned barrier-removal and flow-supplementation projects have the potential to restore access to approximately 30 km of additional rearing habitat in the BBC watershed, nearly doubling the amount of rearing habitat currently available. In the EFPR modeling scenario, we modeled the potential increase in juvenile steelhead production (number of smolts) after treating an additional 15 km of rearing habitat with LWD treatments in the watershed. Current steelhead densities in the EFPR are <8 parr/100m^2 (Knoth et al., in prep.) and we modeled the potential response in juvenile production if the planned LWD treatments result in a commensurate increase in steelhead densities closer to the fair (10 parr/100 m^2), good (14 parr/100 m^2), and excellent (20 parr/100 m^2) ratings developed by Northwest Power Planning Council (Petrosky and Holubetz 1988).

To assess these scenarios, we developed an individual-based simulation model (Grimm and Railsback 2005) to forecast potential responses in juvenile steelhead production in each index watershed. A general diagram of the modeling process is displayed in Figure 2.5. Separate models were fit for the EFPR and BBC waatersheds. To populate the model, productivity of each study watershed, in number of out-migrants, was based on the Ricker function where the parameters were estimated using data from 2005 to 2013 for the BBC study area and from 2008 to 2013 for the EFPR study area. Recruits in the Ricker model were defined as out-migrants. In addition, average age of smolts, as well as the average amount of years each fish spent in the ocean was estimated using data collected from 2006 to 2016 for the BBC study area and 2008 to 2016 for the EFPR study area.

For both study watersheds, the individual-based model was initialized (i.e., year t) with 16 spawners in the watershed and 16 fish in the ocean. The model progressed using annual time steps for 100 years. The number of offspring produced in year t+1 was simulated using the Ricker function based on the number of spawners in year t (i.e., 16 fish in year one) with normal error with mean zero and standard deviation of 150 for BBC and 600 for the EFPR. The age of outmigration of each offspring was drawn from the multinomial distribution with parameters based on the average age of outmigration in each watershed. The number of out-migrants was saved as an output in each time step. Out-migrant-to-adult survival rates were “tuned” in the model such that simulated spawner-recruit data mimicked the range observed in the actual subpopulations (i.e., 1.75% for BBC and 0.2% for the EFPR). In addition, the survival rate was temporally autocorrelated at the 10-year scale to mimic the decadal oscillation in the Pacific Ocean. The number of years that each out-migrant spent in the ocean before returning to spawn was also drawn from the multinomial distribution with parameters based on the estimated average ocean age for each subpopulation. The number of spawners was also saved as an output for each time step. The spawners then produced offspring as described above and the model progressed for 100 time steps. The BBC study area was modeled on a per-km scale whereas the EFPR study area was modeled as a discrete unit.

Because the BBC study watershed was modeled on a per-km scale, estimates of out-migrants and spawners under the status quo (i.e., before restoration) were multiplied by 39.79 (i.e., the current amount of habitat available in BBC) to estimate the total number of out-migrants

and spawners in the subpopulation. To evaluate the BBC scenario outlined above, we multiplied the number of spawners and out-migrants by 75.02, the amount of linear habitat proposed to be available after large-scale projects are completed. We assumed that the potential additional habitat made available would be of the same quality as the current habitat.

Given the amount of proposed habitat restoration for the EFPR scenario and the expected increase in abundance, we expect that rearing densities in the watershed would increase by an average of 10% if habitat is restored to fair condition, 23% if it restored to good condition, and 42% if it is restored to excellent condition. It is uncertain whether the modeled increase will result in higher productivity of juvenile steelhead and no additional production of out-migrants (i.e., will not affect carrying capacity) or will increase carrying capacity. Thus, in addition to the baseline simulation (i.e., no habitat restoration), scenarios were simulated where the intrinsic productivity (i.e., the alpha parameter in the Ricker model) of the stock increased by 10, 23, and 42%, as well as scenarios were the number of spawners that produce peak recruits (i.e., carrying capacity or the beta parameter in the Ricker model) increased by 10, 23, and 42%.

Tributary-Scale Monitoring

Habitat Surveys—Bowersox et al. (2009) developed a Low-Water Habitat Availability Protocol (LWHAP) to estimate late-summer steelhead rearing habitat present within treatment and control tributaries in the lower watershed. Sample sites were selected using a GRTS design and tributaries were stratified into upland and canyon reaches to disperse sites throughout each tributary. We surveyed two, 500 m sites within each reach (upland and canyon), resulting in four sites per tributary and a total of 16 sites in the watershed (Figure 2.6).

We conducted LWHAP surveys the first week of August each year to provide temporal consistency. We recorded the length (m) of wetted habitat and number of pools at each site. Wetted habitat was defined as any area where there was standing water. Pools were delineated as any wetted stream with a maximum depth exceeding 30 cm. We calculated the average proportion of wetted habitat (linear %) and pool density (number/100 m) for each tributary annually. Annual estimates were plotted to examine before treatment trends in habitat conditions in treatment and control tributaries in the watershed.

Habitat surveys in the upper watershed were conducted before and after treatments began with different amounts of effort across time, but the effort was always balanced between treatment and control tributaries within each time frame. Pre-IMW sample sites (2003 and 2004) were included in the analysis to bolster pretreatment data and were selected using a stratified random sampling technique (Schriever and Nelson 1999). Current sample sites (2008 and 2013-2016) were selected using a GRTS design (Figure 2.7). Between 19-21 sites, 10-11 sites, and 10-14 sites were sampled in treatment and control tributaries in 2003–2004, 2008, and 2013-2016, respectively. The spatial coverage and number of sample sites within a given year is adequate to characterize the habitat conditions for each tributary; and hence, variability among sites is small enough that systemic changes to habitat within a stream should be detectable.

We conducted habitat surveys in the upper watershed to monitor key variables associated with the primary limiting factors (deficiency of riparian habitat and in-stream habitat complexity). The main response variables included LWD quantity, pool density, and canopy cover. We modified protocols by Moore et al. (2006) and Bouwes et al. (2011) to focus on the primary limiting factors. Briefly, we enumerated all LWD pieces (≥10 cm in diameter or 1 m in length) within the wetted channel. We defined pools 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. Canopy cover was visually estimated within 5-10 m of bankfull during the 2003-2004 and 2008 surveys. In 2013-2016, we measured canopy cover with a densiometer at four points along 10 sub-transects equally distributed throughout each 100-m site, for a total of 40 measurements per 100-m site. Each technique generated a percentage.

We visually compared trends in habitat variables collected in the treatment and control tributaries during pretreatment and treatment periods. For each variable, we averaged data for all sites within a tributary for a given year to generate an annual estimate for comparison. In addition, we also analyzed each habitat variable as a ratio (treatment data:control data in a given year) to better illustrate the relative change between treatment and control tributaries over time. Within this analysis, a value of 1 would indicate equal quantities or proportions between the treatment and control tributaries, values > 1 would indicate greater treatment quantities/proportions relative to the control tributary, and values < 1 would indicate greater control quantities/proportions relative to the treatment tributary. Data from 2003-2004 and 2008 were considered pretreatment data and data from 2013-2016 were considered treatment data.

Juvenile Steelhead Density, Parr-to-Smolt Survival, and Growth—We monitored juvenile density as the primary response metric to habitat treatments at the tributary scale. As such, we conducted single-pass electrofishing surveys during the late spring/early summer to estimate juvenile steelhead densities in treatment and control tributaries. Data from pre-IMW surveys (1995-1996 and 2003-2004) were incorporated into the analyses to increase pretreatment information, though not all tributaries were sampled in all periods. Specifically, the treatment tributary in the upper watershed (EFPR) was not sampled in 1995-1996. Pre-IMW samples sites (1995-1996 and 2003-2004) and current sample sites (2013-2016) were co-located with habitat sites. In the lower watershed, 9-25 sites were sampled within each treatment and control tributary annually (Figure 2.8). In the upper watershed, 7-13 sites were sampled within the treatment and control tributaries annually (Figure 2.9).

Electrofishing survey site boundaries were established at distinct habitat breaks 80-120 m apart, with the downstream boundary being close to the identified site location. Crews began at the lower boundary of each site and electrofished upstream in a single pass. Fish captured during the surveys were identified to species and enumerated. All steelhead ≥ 80 mm were weighed (g), measured (FL, in mm), and scanned for the presence of PIT-tags. Juvenile steelhead (≥ 80 mm) not previously tagged were anesthetized using MS-222 solution and tagged in the abdomen with a PIT-tag. Site length and five widths were measured at each location to estimate area sampled. Mean density estimates (fish/100 m^2) were calculated for each tributary or reach.

In addition to standardized electrofishing surveys, we also conducted roving electrofishing surveys to PIT tag juvenile steelhead in treatment and control tributaries to estimate parr-to-smolt survival. Roving electrofishing was conducted during June-July each year. Juvenile steelhead tagged during summer roving PIT-tagging were expected to emigrate out of the Potlatch River through the hydrosystem during the following spring. All captured juvenile steelhead ≥ 80 mm were anesthetized in MS-222, measured (FL; mm), weighed (g), and PIT tagged. Tagging was consistent with NOAA electrofishing criteria (NOAA 2000). PIT-tag data was uploaded to the PTAGIS database daily. Numbers of tags placed by year and tributary are in Appendix G.

We estimated apparent survival for fish tagged in tributaries to LGR the following spring as a proxy for parr-to-smolt survival. Procedures were similar to estimating apparent survival for smolts from tributary screw traps as described in the watershed-level monitoring section above. We assumed emigration rate to be standard across roving tag groups. Trends in apparent survival

for tag groups in treatment and control tributaries were visually compared across the years (BBC) and pretreatment /treatment periods (EFPR).

Understanding the relationship between tributary survival and base flow conditions (a primary limiting factor) will give insights into the mechanisms behind future responses. Therefore, we calculated BBC watershed-wide survival estimates (BBC, LBC, and WFLBC tag groups combined) to the BBC tributary array in 2010-2016 (2011 was not estimated because of low detections at the array). We computed a Pearson product-moment correlation coefficient to assess the relationship between base flow conditions and watershed-wide survival estimates. We used main-stem Potlatch River flow measurements (August base flow conditions at USGS gauge # 13341570) as a proxy for tributary flow conditions.

Beginning in 2014, we conducted additional electrofishing surveys during the fall months to recapture previously PIT-tagged juvenile steelhead to monitor juvenile summer-fall growth as a response to restoration treatments within select treatment and control tributaries. Surveys were conducted during late October/early November in order to standardize the estimated growth period among years. We only sampled the WFLBC, LBC and PNC because we could consistently recapture previously tagged steelhead. We calculated growth (mm/d) as the change in fork length between time of tagging and time of recapture for each recaptured PIT-tagged fish. Means and standard deviations were calculated for each tributary to compare growth rates between treatment and control tributaries across time.

Power Analyses—The primary restoration techniques are all expected to positively increase juvenile steelhead densities in the treatment tributaries. As such, we conducted power analyses on electrofishing data to determine the number of sample sites and years needed to detect a statistically significant difference in parr densities in treatment and control tributaries at a rate of change that varied from 10% to 60%. Separate power analyses were conducted for tributaries in the EFPR and BBC watersheds. The parameters of interest are statistical power (1-beta), number of sites, and number of years for each level of expected increase. A 10 - 60% increase represented a reasonable range of rates considering the restoration projects being implemented.

The power analyses were conducted using simulation techniques. Expected density at each site in the control streams was simulated using the mean density and standard deviation (among sites) for each stream using electrofishing data collected during 2013 - 2016. The number of sites simulated varied from 8 to 30 by units of two, and the number of years of data that were simulated was 10, 20, or 30. Simulated abundance at the control streams was constant. Densities in the treatment streams were simulated using similar methods except that year one was simulated based on the mean and standard deviation observed and each year thereafter was simulated with an annual increase in density that varied from 10% to 60% by units of 10%. For example, when simulated density increased by 10%, it increased by 10% every year from the previous year (i.e., annually, lambda = 1.1). When it increased by 20% it increased by 20% every year (i.e., lambda = 1.2), and so on up to 60% increase every year (in 10% increments). This tactic allowed us to evaluate how the number of sites, the number of years, and the change in density influenced statistical power.

We evaluated simulation results using linear models (EFPR watershed) and linear mixed effects models (BBC watershed). A mixed effect model was used for the BBC watershed because there are multiple treatment streams in the drainage. In both models, treatment type (i.e., control or treatment) and year were treated as fixed effects. In the lower watershed model, stream was treated as a random effect. We based inference on the year*treatment interaction term (Smith

2002). The statistical significance of this term was assessed using a Wald test with $\alpha = 0.05$. The process of simulating data, fitting the models and assessing the statistical significance was repeated 5,000 times for each combination of sample size (i.e., number of sites), number of years, and effect size (i.e., the relative rate of change in abundance between treatment and control reaches). Statistical power was then estimated by dividing the number of times the interaction term was significant by the number of simulation iterations. For instance, if the interaction term was significant in 2,500 of 5,000 simulation iterations, then statistical power was estimated at 50%. In this scenario, we would have a 50% chance of rejecting the null hypothesis of no difference in density between treatment and control streams, given the observed variation in abundance among sites.

Reach-Scale Monitoring

East Fork Potlatch River LWD Treatments—The EFPR Bloom Meadows LWD project was completed in 2009 and 44 LWD structures were installed for fish habitat and bank stability in a 1.6 km reach (Figure 2.3). We predicted the structures would provide summer and winter rearing habitat and the density of juvenile steelhead in the project area would vary seasonally. From 2010-2016, the LWD structures were electrofished during the summer to determine after treatment steelhead density in the project area. The electrofishing surveys included twenty structures combined into five separate groups (four structures/site). Crews began at the downstream end of each site and electrofished upstream in a single pass. Sites ranged between 48-250 m in length. All juvenile steelhead captured were enumerated. Mean (SE) juvenile density was estimated across all five groups of structures for each year and standardized for parr/100 m^2. Annual trends in juvenile densities in the project area were assessed in relation to trends observed at the tributary scale.

Additionally, we examined seasonal changes in juvenile densities within the project area. Each group of LWD structures was electrofished once in July, September, and December from 2012-2014 to monitor changes in the seasonal density of juvenile steelhead in the project area. To facilitate seasonal comparisons, the mean and standard deviation of juvenile density (fish/100 m^2) were estimated across all five groups of structures for each of three seasons (summer, fall, winter). We assumed catchability of juvenile steelhead was equal across all seasons.

Spring Valley Creek Flow Supplementation—We conducted a two-phase pilot study to test the feasibility of using a headwater reservoir to supplement late summer flows in Spring Valley Creek (SVC) and LBC in the lower Potlatch River watershed (Figure 2.2). In 2015, releases began after the downstream reach became intermittent in August to determine the amount of flow needed to re-charge the system and achieve perennial flow. In 2016, releases began during June before the test reach went intermittent to identify the minimum flow needed to maintain perennial flows downstream.

We expected flow supplementation would increase the quantity of juvenile rearing habitat (available wetted habitat and pool abundance) and improve the quality of existing rearing habitat (improved temperature and dissolved oxygen). We evaluated the response of habitat conditions to flow supplementation with a BACI design. The treatment reach encompassed SVC and LBC and covered 16 km downstream of the reservoir (Figure 2.10). Control reaches were located in WFLBC in 2015 (one control site) and WFLBC and BBC in 2016 (four control sites). Parameters of interest included wetted habitat, pool density, water temperature, and dissolved oxygen. We assessed wetted habitat and pool density during habitat surveys in the treatment reach prior to the flow supplementation and throughout the treatment period. Water temperature and dissolved oxygen were monitored via remote, continuous data loggers at stations located every 2 km

throughout treatment reach. Treatment data were further stratified by flume, meadow, and canyon reaches. Detailed methodology can be found in Appendix H. We used graphical comparisons for inference.

Dutch Flat Dam Removal—In 2013, Dutch Flat Dam on the WFLBC (Figure 2.2) was removed which restored access to >10 km of potential spawning and rearing habitat. Prior to the dam removal in 2011, we conducted an electrofishing survey to estimate juvenile steelhead densities at several locations downstream (5 sites) and upstream (4 sites) of the dam site. Seven sites were selected using GRTS and two sites were selected based on professional judgement to represent a large reach of habitat that did not contain a GRTS site. Surveys followed establish electrofishing protocols (see Tributary-Scale Monitoring section above for details).

We expected improved passage would result in the expansion of adult spawning and juvenile rearing distribution. In 2015 and 2016, we evaluated success with a subset of the results from the radiotelemetry study (see Spawning Distribution section above for details). As part of this effort, radio-tagged adult steelhead were tracked during their spawning migration in the WFLBC. We supplemented these observations with spawning surveys directly upstream of the Dutch Flat Dam project site in the spring of 2015 and 2016. In 2015, the spawning survey was conducted in 1.2 km reach directly upstream of the project site and ended downstream of the first of two culverts suspected of being partial-migration barriers. The two culverts were replaced in the fall of 2015. In 2016, the spawning survey was conducted in a 2.2 km reach directly upstream of the dam site and ended upstream of the replaced culverts. We recorded the number and GPS location of completed redds, live adults, and carcasses during each survey.

In addition, we conducted post-project monitoring of juvenile steelhead density upstream and downstream the project site to compare against pretreatment data. As part of annual snorkeling efforts, we sampled three sites (1 downstream and 2 upstream) around the Dutch Flat Dam project site in 2015 and 2016. The sites were located approximately 0.1 km and 2.2 km above and 1.5 km below the dam site and standard snorkeling protocols were followed (Apperson et al. 2015).

RESULTS

Index-Watershed Monitoring

Big Bear Creek—Adult escapement into BBC watershed varied six-fold across the years, with the largest change from 2011 to 2012 (Figure 2.11). Estimates ranged from 50 adults (95% CI 19-94) in 2006 to 317 adults (95% CI 220-363) in 2012. The precision of the mark/recapture estimates (2005-2012) and PIT-tag expansion estimates (2013-2016) were similar. There was an oscillating pattern in the escapement estimates during the past 8 years.

The sex ratio and age composition of BBC adult steelhead varied annually. On average, the sex ratio of BBC adults was skewed towards females (56%, range 35% - 75%) (Figure 2.12). The age-at-smolt and time in the ocean of returning adults each ranged from one to three years (Figure 2.13). Total age at spawn time ranged from 3-7 years. The most frequent life history was 2:2 for females and 2:1 for males (freshwater:ocean age).

Spawning distribution expanded from 2010-2011 to 2015-2016. In 2010-2011, the farthest documented location of an adult fish was 16.8 km upstream of the LBC weir in the WFLBC, directly downstream of Dutch Flat Dam (Figure 2.14). In addition, we detected two adult fish in BBC

directly downstream of Big Bear Falls, a partial-migration barrier (Bowersox et al. 2016). No fish were detected in LBC upstream of WFLBC confluence. In 2015, we tracked 14 adult steelhead radio-tagged at LGR into the BBC watershed. Nine adults migrated to the WFLBC, three of which traveled upstream of the Dutch Flat Dam project site (barrier was removed in 2013). In 2016, 15 adult steelhead radio-tagged at LGR were tracked throughout the spring migration. Of these fish, four migrated to WFLBC (one fish was located upstream of Dutch Flat Dam project site), four migrated to LBC (two fish were located upstream of WFLBC confluence), four to BBC (downstream of Big Bear Falls), and three remained in the main-stem river below BBC. We observed the greatest expansion of spawning in the WFLBC and LBC between the two periods.

There was greater annual variability in observed *O. mykiss* fry densities compared to parr densities (Figure 2.15). Fry densities ranged from 0.83-25.06 fish/100 m^2 (mean = 8.08 fish/100 m^2); whereas, parr densities ranged from 0.74-11.98 fish/100 m^2 (mean = 4.05 fish/100 m^2) across the years. In addition, we observed more variance in estimates among years than among sites within years. There was no trend either fry or parr estimates over time.

Spring emigration from the BBC watershed varied five-fold across the years, with the largest change from 2013-2014 (Figure 2.16). The mean annual emigration estimate was 9,389 juveniles (range 3,837-22,649 juveniles). Juvenile emigration estimates peaked in 2013. There were no fall emigration estimates for six years because either flow conditions or personnel shortages did not allow trapping (2007, 2011, 2014, and 2015) or not enough juveniles were trapped to generate an estimate (2006 and 2013). Fall estimates ranged from 91 fish (95% CI 14-84) to 2,032 fish (95% CI 1,056-4,096) and comprised 6.4% of the following years' spring emigration estimate on average.

Emigrant age composition was consistently dominated by age-2 emigrants (Figure 2.17). On average, age-2 fish comprised 63% of the emigrants annually (range 47%-87%). Emigrant length-at-age was relatively consistent across the years. Mean FL of age-1, age-2, and age-3 juveniles captured at BBC screw trap was 136 mm (range 80 mm-202 mm), 170 mm (range 106 mm-256 mm), and 184 mm (range 141 mm-256 mm), respectively from 2008-2016 (Figure 2.18). Young-of-year steelhead were captured periodically at BBC trap in very low numbers and were not included in the age composition or length-at-age analyses.

Productivity estimates for BBC display a strong density-dependent relationship (Figure 2.19). Juveniles/spawner estimates ranged from 53 emigrants/female in brood year 2010 to 277 emigrants/female in brood year 2006 (Table 2.2). In general, escapement greater than 70 females produced less than 100 emigrants/female.

Apparent survival for emigrants from the BBC trap to LGR peaked at three-year intervals. Mean apparent survival was 49% and ranged between 27% - 81% (Table 2.3). Peaks in apparent survival tended to coincide with years of low juvenile emigration and high emigrant length at age. For example, the two years with the highest apparent survival (2008 and 2011) also had the lowest emigration estimates and highest age-1 length at age (Table 2.3, Figures 2.16 and 2.18). This pattern may be an indicator of density dependent effects in the BBC watershed.

East Fork Potlatch River—Adult escapement into the EFPR watershed was relatively consistent across the years (Figure 2.20). We could not produce an expanded mark-recapture estimate for 2011 because high flows disrupted weir operations; therefore, 33 fish represented a minimum escapement estimate. Excluding 2011, estimates ranged from 72 adults (95% CI 41-113) in 2010 to 140 adults (95% CI 33-232) in 2008. The 2013 estimate is considered a census because all recaptured kelts were previously captured as prespawn adults.

The sex ratio and age composition of EFPR adult steelhead varied annually. On average, the sex ratio of adult steelhead was skewed towards females (mean = 55%, range 33%-76%) (Figure 2.21). Age at smolt ranged from one to four years and time in the ocean ranged from one to three years (Figure 2.22). Total age at spawn time ranged from 3-7 years. The most prominent life history was 2:2 for females and 2:1 for males.

Spawning distribution in the EFPR watershed was concentrated primarily in the EFPR. In 2015, 16 radio-tagged adults ended their upstream spawning migration in the EFPR and one adult migrated approximately 300 m into Ruby Creek (Figure 2.23). The farthest documented upstream spawning migration in 2015 was 22 km upstream from the weir site and 13% of tagged adults (2 of 16 fish) ended their migration upstream of the confluence with Jackson Creek. In 2016, 15 radio-tagged fish ended their upstream migration in the EFPR. Interestingly, we tracked one radio-tagged fish into the WFPR approximately 15 km upstream of the confluence with the EFPR. The farthest documented upstream spawning migration in 2016 was approximately 28 km upstream from the weir site and 47% of the tagged adults (7 of 15 fish) ended their migration upstream of the confluence with Jackson Creek. Migration flow conditions during March and April were higher in 2016 compared to 2015 and likely influenced spawning distribution. In both years combined, 94% (30 of 32) radio-tagged fish ended their spawning migration in the EFPR rather than in its tributaries.

Spring juvenile emigration from the EFPR watershed was highlighted by two peak estimates in 2010 and 2013 (Figure 2.24). Peak estimates ranged between 30,733 to 40,224 fish. Pretreatment estimates (2008-2009) averaged 10,587 fish (range 10,106 fish-11,069 fish) and treatment period estimates (2010-current) averaged 18,703 fish (range 7,965 fish-40,224 fish). There were no fall emigration estimates generated for five years either because flow conditions or personnel shortages did not allow trapping (2011, 2014, and 2015) or not enough juveniles were trapped to generate an estimate (2009 and 2013). Fall estimates ranged from 1,296 fish (95% CI 780-2,128) to 1,866 fish (95% CI 1,261-3,056) and on average were 10.7% of the following years' spring estimate.

Emigrant age composition and length at age have displayed shifts during the most recent treatment years. Age-1 fish comprised on average 77% of the emigrant population from 2010-2013 (range 74%-84%), but only 58% from 2014–2016 (range 54%-62%; Figure 2.25). Conversely, the average proportion of age-2 fish increased from 21% (range 15%-26%) to 38% (range 32%-43%) over the same timeframe. The mean FL of age-1 and age-2 emigrants also increased during the two most recent treatment years (2015 and 2016) (Figure 2.26). From 2010-2014, the mean FL of age-1 and age-2 emigrants was 89 mm and 136 mm respectively. In comparison, the mean FL of age-1 and age-2 emigrants in 2015-2016 was 98 mm (range 67 mm-142 mm) and 145 mm (range 102 mm-183 mm), respectively. The mean FL of age-3 emigrants was consistent across time.

Productivity estimates tended to decrease as female escapement increased (Figure 2.19). Brood year productivity estimates ranged from 234.2 emigrants/spawner in 2013 up to 752.4 emigrants/spawner in 2012 (Table 2.4). Brood year 2011 was excluded because the parent estimate is biased low, which would in turn bias the productivity estimate high.

Mean apparent survival for emigrants from the EFPR trap to LGR (index of smolt survival) has increased during the most recent treatment years (Table 2.5). From 2010-2012, mean apparent survival was 7.0% (range 6.0%-8.0%), slightly lower than the pretreatment average (10%). In comparison, mean apparent survival from 2013-2016 was 14% (range 9.0%-17.0%).

Life Cycle Modeling—In the BBC modeling scenario, we predicted increased rearing habitat quantity via large barrier removal and flow supplementation projects would increase smolt production significantly. Simulated values for the current scenario fell within the range of empirical observations, but with smaller overall variation, indicating the model performed well (Appendix I). Under this modeling scenario, the model predicted average smolt production would nearly double (from 10,714 to 20,596 fish) following the implementation of planned projects in the BBC watershed (Figure 2.27).

In the EFPR modeling scenario, we predicted improving stream complexity via LWD treatments in 15 km of stream would produce a detectable response in smolt production. The model performed reasonably well as the majority of simulated values fell within the range of empirical observations for the current scenario (Appendix I). In the EFPR scenario, predicted changes in smolt production did not differ substantially between model simulations that increased intrinsic productivity and simulations that increased carrying capacity (Figures 2.28 and 2.29). The predicted increase in production ranged from approximately 1,900 to 8,000 smolts (10%-43% increase) while juvenile densities increased from 10 fish/100 m^2 to 20 fish/100 m^2.

Tributary-Scale Monitoring

Lower Potlatch River Watershed—The amount of wetted habitat and pool density varied among tributaries and across years (Figure 2.30). The amount of wetted habitat ranged from 48.5%-100% and pool density ranged from 0.55-5.4 pools/200 m. Wetted habitat and pool density were generally higher in treatment tributaries (BBC, LBC, WFLBC) compared to the control tributary (PNC). On average, WFLBC had the greatest amount of wetted habitat, while LBC and BBC had the highest pool density across the years. In the treatment tributaries, the two years with the lowest amount of wetted habitat (2007 and 2015) coincided with the two years of lowest flows in August at the Potlatch River flow gauge. This observation suggests the main-stem flow gauge is a good predictor of base-flow conditions in the treatment tributaries.

Juvenile steelhead density estimates in treatment and control tributaries displayed similar trends over time, suggesting the tributaries experienced similar environmental phenomena (Figure 2.31). Among tributaries, either the WFLBC or LBC had the highest densities each year. For all tributaries (treatment and control), estimates peaked in 2013 and declined through 2015.

Survival of juveniles was variable with respect to several factors. Apparent survival from tributary to LGR the following spring varied among years and tributaries. Estimates decreased substantially in 2014 and remained low through 2016 in treatment and control tributaries (Figure 2.33, top panel). In 2017, apparent survival for LBC, WFLBC, and PNC rebounded to levels prior to 2013, with the greatest increase in LBC. Survival from BBC could not be generated for six years because of low detections in the hydrosystem. Watershed-wide survival to the tributary array varied three-fold over time. Estimates ranged four and a half fold and were the highest in 2013 (Figure 2.33, bottom panel). Big Bear Creek watershed survival was positively correlated (r = 0.662) with mean August flows at the Potlatch River gauge, suggesting base flows are a key factor influencing juvenile survival in the drainage.

Juvenile growth rates varied among tributaries and across years (Table 2.6). In general, growth rates were the highest in 2014 and lowest in 2016. In 2015, LBC growth rates were 0.124 mm/day; however, the estimate was based on a small sample size (n = 2). Juveniles in the WFLBC had the highest variability in growth across the years, whereas juveniles in PNC had the lowest.

Upper Potlatch River Watershed—The three habitat metrics measured here fluctuated annually (Figure 2.33). Percent canopy cover was consistently higher in the treatment tributary compared to the control tributary. Canopy cover ranged between 29%-47% in the EFPR and 16%-36% in the WFPR. Pool density was the highest in 2014 and 2015 for both tributaries, but returned to pretreatment levels in 2016. Large woody debris data was not collected in 2015. Large woody debris density was consistently higher in the EFPR compared to the WFPR. There were increases in two of the three ratio metrics during treatment years in the EFPR. The canopy cover ratio showed an increasing trend since 2014. In 2016, canopy cover was nearly two times greater in the EFPR relative to the WFPR. Likewise, the pool density ratio displayed a positive increase since 2013. In 2016, the EFPR had approximately five times the number of pools relative to the WFPR. The LWD ratio varied over pretreatment (2003-2004 and 2008) and treatment (2013-2016) periods with no discernable trend.

East Fork Potlatch River juvenile density estimates were more widely variable over time compared to WFPR (Figure 2.34). Estimates in the EFPR ranged from 0.5 - 2.4 fish/100 m^2 and were consistently higher than in the WFPR. West Fork Potlatch River estimates were consistently <0.5 fish/100 m^2 across the years. Estimates in the EFPR and WFPR showed similar trends over the pretreatment (1995-1996 and 2003-2004) and treatment (2013-2016) periods.

Apparent survival from EFPR to LGR the following spring varied among years (Figure 2.35). Survival of EFPR tag groups averaged 7.5% (range 7.0%-8.0%) during pretreatment years and 11% (range 5%-15%) during treatment years. Rates could not be estimated for tag groups in 2010, 2014, and 2015 due to low detections of these fish at downstream arrays the subsequent year.

Power Analyses—In general, there was a greater probability to detect a given increase in juvenile steelhead density in the BBC watershed compared to EFPR watershed, given the current effort. In the BBC watershed, we could detect a 50% increase in parr density, with a power of at least 0.80, within approximately 15 years at our current sampling rate (10 sites/treatment and control tributary; Figure 2.36). In comparison, it would take 20-30 years of post-monitoring to detect a similar level of increase, with the same power, at our current sampling rate (~15 sites/treatment and control tributary) in the EFPR watershed (Figure 2.37).

Increasing sampling effort would have a different level of impact on our power to detect changes in parr density for the two watersheds. In the BBC watershed, increasing the number of sample sites from 10 to 15 would reduce the time needed to detect a 50% increase (at a power of 0.80) to <10 years (Figure 2.36). Conversely, it would still take nearly 20 years of post-monitoring in the EFPR to detect a 50% increase in parr density (at a power of 0.80) even if the number of sample sites doubled to 30 sites/tributary (Figure 2.37).

Reach-Scale Monitoring

East Fork Potlatch River LWD Treatments—Juvenile steelhead density estimates at LWD structures were comparable to tributary densities and varied seasonally (Figure 2.38). Mean juvenile density within the project area, during the summer, was 0.82 fish/100 m^2 (range 0.12-2.0 fish/100 m^2). Mean juvenile densities in the project area were consistently lower than tributary densities, but showed similar trends over time. In 2012, juvenile density within the project area increased notably from summer to winter. We were unable to sample during the winter 2013 and fall 2014, but it appears that LWD structures are being used as winter cover based on 2012 data.

Spring Valley Flow Supplementation—Flow supplementation re-established or maintained complete connectivity and increased pool habitat in the treatment reach (Figure 2.39). Prior to flow supplementation in 2015, the treatment reach downstream of the reservoir went intermittent (29% and 72% wetted habitat in meadow and canyon reaches, respectively). Flow supplementation increased the amount of wetted habitat 71% and 28% in the meadow and canyon reaches respectively, and re-established 100% connectivity in the entire 16-km treatment reach. Supplementation re-watered approximately 8 km of additional rearing habitat that were dry prior to treatment. In 2016, flow supplementation began before the treatment reach became intermittent and maintained 100% wetted habitat throughout the treatment reach. In 2015, flow supplementation increased pool density by 117% and 86% in the meadow and canyon reaches, respectively. Total pool density (meadow and canyon reaches combined) was higher in 2016 when a perennial flow was maintained throughout the summer.

Flow supplementation also had positive impacts on water quality in the treatment reach. Mean water temperatures in the meadow and canyon reaches were 16.5°C and 14.1°C respectively during flow supplementation in 2015, while the control reach went dry until September (Figure 2.40). During the flow supplementation period in 2016, the meadow and canyon reaches were on average 3.7°C and 2.7°C cooler respectively than the BBC control, and 2.0°C and 1.0°C cooler respectively than the WFLBC control. Water temperatures in the treatment and control reaches converged as fall rains began in September. In 2015, DO levels increased by 34% in the treatment reach during flow supplementation, while DO levels at the control reach could not be measured because it went dry (Figure 2.41). During flow supplementation in 2016, DO levels averaged 7.92 mg/L and 6.70 mg/L at 2 km and 16 km respectively in the treatment reach. In comparison, DO levels at control sites in WFLBC and BBC were 8.37 mg/L and 0.68 mg/L respectively over the same time span.

Dutch Flat Dam Removal—We observed an upstream expansion in steelhead spawning distribution following the removal of Dutch Flat Dam. In 2015 and 2016, four individual radio-tagged adults were tracked upstream of the project site during the spring spawning migration (Figure 2.14, see Spawning Distribution section). During redd surveys in 2015, we documented six steelhead redds and four possible redds in a 1.2 km reach that started directly upstream of the project site and ended below the first of two culverts suspected of being partial migration barriers. The two culverts were replaced in the fall 2015. During surveys in 2016, we documented five steelhead redds, five live adults, and one carcass in a 2.2 km reach that started directly upstream of the project site and ended just upstream of the replaced culverts.

During the pre-project electrofishing survey, mean juvenile steelhead densities (106.2 fish/100 m^2, all size classes combined) downstream from the dam were sixty-fold higher than densities above the dam (1.7 fish/100 m^2), which indicated the dam was a significant fish barrier. During post-project snorkel surveys we documented *O. mykiss* fry and parr above the project site. In 2015, average fry and parr densities were 8.4 fish/100 m^2 and 1.5 fish/100 m^2 respectively above the project site. In 2016, average fry and parr densities were 31.0 fish/100 m^2 and 1.7 fish/100 m^2 above the project site.

DISCUSSION

Results from the Potlatch River IMW study provide the necessary baseline to evaluate restoration actions. These results underscore the need for habitat restoration in the basin, validate restoration approaches being applied, and inform restoration implementation and management decisions within and outside the basin. Baseline stock-recruit productivity curves imply habitat

capacity is limited within Potlatch River index watersheds. To date, approximately 25% of planned restoration treatments have been completed in the index watersheds. Funding, personnel, and private-land limitations have impacted the pace of project implementation. Thus, restoration efforts have not been to scale to generate a definitive response of steelhead in the index watersheds, but we have documented smaller-scale fish and habitat responses that illustrate the potential for future changes in the study populations. Further, recent shifts in the EFPR emigrant age composition and length-at-age suggest an initial response by the study population. Life cycle-modeling exercises indicate planned restoration projects have the potential to generate detectable responses within the index watersheds. Project data have been widely utilized to direct prioritization of restoration needs and locations in the Potlatch River basin. Over time, restoration and monitoring efforts have adapted to become more cohesive, leading to a more efficient and productive restoration program.

We begin the discussion by highlighting key findings and management implications of the effectiveness monitoring in the IMW. Next, we cover the challenges and lessons learned in the planning, restoration, and monitoring components of the study, including adaptive management of the restoration and monitoring programs. We end with a discussion of the wider implications of the study.

Restoration Effectiveness

Rearing capacity is limited within index watersheds, resulting in strong density effects on emigrants-per-female productivity. Growth, mortality, and emigration are commonly observed to change with density in salmonid populations (Grant and Kramer 1990), though the contribution of each rate to population regulation is unclear. Keeley (2001) documented increased mortality, decreased growth, and increased variance in size distribution of *O. mykiss* within a closed population contained in an artificial stream channel; in contrast, emigration of fish out of the artificial study area decreased the magnitude of density-dependent effects. Within the Potlatch River basin, density dependence is most obvious in the BBC watershed. Rearing habitat is severely limited by tributary blockages and low flow, and movement of rearing steelhead within the watershed is difficult by mid- to late summer. In Lapwai Creek, Idaho, growth, survival, and movement of juvenile steelhead varied in response to a complex combination of intraspecific competition and reduced flows (Hartson and Kennedy 2015; Myrvold and Kennedy 2015). Further, we found low flow conditions are an important driver of parr-smolt survival in the BBC watershed. Consequently, barrier removal and flow supplementation are plausible techniques for reducing density-dependent effects and increasing population productivity.

Reach-scale fish and habitat responses illustrate the potential response as additional areas and stream kilometers are treated. Improved passage resulting from barrier removals should result in the expansion of adult spawning and juvenile rearing distributions (Anderson et al. 2008). In our monitoring, we documented immediate colonization and utilization of reconnected spawning and rearing habitat following the first major barrier removal in the BBC watershed. Grantham et al. (2012) found that juvenile steelhead survival declined in relation to the magnitude and duration of low flow conditions in a coastal California watershed; therefore, one might expect a positive response in survival to higher, less-severe flow conditions. Similarly, we observed higher juvenile survival in years with higher base flows in the BBC watershed. In LBC, the flow supplementation project resulted in substantial and immediate improvements in rearing conditions. Flow supplementation doubled the amount of potential rearing habitat (an additional 8 km increase in wetted habitat), provided 100% connectivity, increased pool habitat density, and improved water quality (in terms of water temperature and dissolved oxygen) in the treatment reach. Improvements to flow conditions should allow for greater dispersal of juvenile cohorts,

thereby reducing density effects on growth and survival (Hartson and Kennedy 2015; Myrvold and Kennedy 2015). In the EFPR watershed, we documented an increase in the relative abundance of juvenile steelhead in the fall and winter at LWD project sites, which suggests habitat treatments are providing winter rearing habitat. Similarly, other studies have documented the benefits of LWD for wintering juvenile salmonids (Johnson et al. 1986; Solazzi et al. 2000). These reach-scale responses are encouraging and as additional restoration work is completed, our monitoring design will allow us to understand how localized responses scale up to the population level.

Planned restoration projects have the potential to generate detectable responses by steelhead in the index watersheds, though the magnitude and timeframe of response differed among watersheds. In the BBC watershed, modeling simulations indicated that implementation of three large restoration projects could potentially increase mean smolt production by 92%. The increase in smolt production is directly proportional to increase in rearing habitat. These three projects would reconnect nearly 35 km of additional rearing habitat, effectively doubling the linear amount of rearing habitat currently available in the drainage. We assumed the reconnected habitat would be of the same quality as the current habitat in the drainage based on habitat surveys in the restoration areas (Tiege Ulschmid, IDFG, personal communication). It is important to note that restoring access or flow is only the first step in the restoration process. Once access and/or flow is restored, additional instream and riparian restoration work will help to fully maximize the intrinsic potential of the watershed. Given the positive fish and habitat responses to barrier removal and flow supplementation projects and the extensive size of the planned projects, we feel the model's predictions of potential responses are realistic.

In contrast, multiple small projects are needed to generate a detectable response by steelhead in the EFPR watershed. Modeling indicated that planned LWD/riparian projects over the next five years could potentially increase smolt production between 10-40%, depending on the level of increase in rearing density achieved (i.e., an estimated 43% increase in smolt production if parr densities are 20 fish/100 m^2). Roni et al. (2010) found 20% or more of a watershed needs to be restored in order to achieve a 25% increase in smolts in Puget Sound watersheds, but that increase would be difficult to detect because of inherent variance. We will have treated approximately 35% of the EFPR after the planned projects are implemented, though the power analysis indicated it would take >20 years to detect a 40-60% increase in parr density under our current tributary-level monitoring effort. Previous work has documented a significant, positive correlation (R^2 = 0.7048) between steelhead abundance and LWD density in the EFPR (Bowersox and Brindza 2005). Based on this relationship, LWD densities of around 500 pieces/100 m^2 would correlate to parr densities of 20 fish/100m^2. Currently, LWD densities are approximately 180 wood pieces/100 m^2 in the EFPR watershed; therefore, planned LWD/riparian projects would need to increase LWD densities by 170% to achieve target densities (500 pieces/100 m^2). In comparison, other studies have documented salmonid population responses to LWD treatments when LWD abundance was increased by 150-790% (Cederholm et al. 1997; Roni and Quinn 2001). Overall, the modeling results appear reasonable and indicate a greater restoration effort is needed in the EFPR watershed, relative to BBC, to generate a detectable response, but the EFPR monitoring approach needs to be reconsidered in order to guide restoration and more effectively detect effects.

The watershed-level evaluation will be completed using a before-after comparison. Before-after study designs will be limited by a lack of pretreatment data to inform analysis. A key component of successful project evaluation is a time series baseline prior to treatment (Kondolf 1995). We have accumulated a minimum of eight years pretreatment population production/productivity data for the BBC watershed and two years for the EFPR watershed. In addition, we have garnered an extensive understanding of life-history characteristics of the

steelhead within each index watershed. These observations are important because habitat restoration activities may not only impact steelhead productivity but may also lead to changes in age structure (Hunt 1988, as cited in Roni 2005) and size of individuals (Roni and Quinn 2001). Restoration efforts began in the EFPR and BBC watersheds in 2008 and 2013, respectively. We estimate it will take at least 5 years to implement enough projects to begin to generate population-level shifts in smolt production in the index watersheds. Because of the complex and protracted life history of steelhead (i.e., total age at spawn time ranged from 3-7 years), population-level effects will not be instantaneous and we anticipate it will take approximately 10 years of post-treatment monitoring to evaluate the full before-after study design.

Restoration efforts in the EFPR watershed are focused on increasing habitat complexity (more pool habitat and instream cover) through LWD treatments and riparian enhancement. The placement of instream structures is a common restoration practice in rivers (Roni et al. 2008), though such treatments are usually temporary until riparian restoration results in natural LWD recruitment (Palmer et al. 2014). Instream LWD is valuable to fish in creating pools, increasing habitat complexity, trapping spawning gravels, and providing food for aquatic invertebrates (Bisson et al. 1987). In the EFPR watershed, instream LWD and/or riparian treatments have been concentrated primarily in the EFPR, where our monitoring data shows the bulk of steelhead spawning and rearing occurs (Knoth et al., in preparation). We have observed recent increases in canopy cover and pool density in the treatment tributary relative to the control tributary. In addition, we documented juvenile steelhead using LWD structures during the summer and winter. These are positive signs that the restoration treatments are working as intended and lend support to the restoration approaches being implemented.

Although only a minor fraction of available rearing habitat in the EFPR watershed has been treated, other processes are contributing to changes in habitat complexity. Natural beaver colonization is occurring throughout the EFPR and we estimate approximately 2.5-3.0 km of the EFPR has current beaver activity (Tiege Ulschmid, IDFG, personal communication). Beavers can increase habitat complexity spatially and temporally (Schlosser and Kallemeyn 2000) similar to LWD treatments, and recent work from the Bridge Creek IMW has demonstrated the importance of beaver in creating habitat for steelhead and Chinook Salmon in interior Columbia River tributaries (Pollock et al. 2007, 2012). Additionally, two prominent landowners (Idaho Department of Lands and Potlatch Corporation) recently canceled grazing leases in the EFPR watershed, which will assist in the recovery of riparian conditions. Taken as a whole, these activities are likely contributing to the observed changes in habitat conditions in the EFK; and in future years, we plan to monitor the expansion/contraction of beaver activity in the drainage to better account for their effects in future evaluations.

Steelhead life history metrics, such as emigrant age composition and size structure, are also important factors to evaluate the effects of restoration activities. A lack of stream complexity limits summer and winter rearing habitat in the EFPR, but restoration efforts coupled with other processes (e.g., beaver colonization) appear to be increasing habitat complexity in the watershed. For partially migrating species, freshwater growing conditions influence the duration of freshwater rearing (Olsson et al. 2006). We hypothesized that improvements to habitat conditions in the EFPR may cause juvenile steelhead to rear an additional year there rather than migrating downstream to rear elsewhere in the basin. Though not conclusive, we have documented recent increases in emigrant age structure and length-at-age in the EFPR. Furthermore, improvements to rearing habitat conditions that result in a larger and older emigrant may potentially increase their survival to adulthood. Size is a determining factor in salmonid smolt survival during outmigration, with larger individuals experiencing greater survival (Zabel and Williams 2002). In addition, there is evidence of a positive relationship between steelhead smolt size and marine

survival (Beakes et al. 2010), though the strength of this relationship remains weak. Consequently, recent shifts in emigrant age composition and length-at-age may indicate initial population-level shifts in the EFPR watershed and offer insights into the future response of production and productivity metrics.

In summary, we have documented several key accomplishments over the first 10 years of the study. Although not enough restoration actions have been completed to fully evaluate the program, preliminary results have demonstrated the value of the restoration approaches being implemented as well as the monitoring design. In the following section, we discuss higher-level challenges and lessons learned thus far. In addition, we discuss the adaptive management approaches we have applied.

Challenges and Lessons Learned

Implementing restoration actions at proper temporal and spatial scales is a common challenge among IMW's (Bennett et al. 2016). Within the Potlatch River IMW, the timeline and amount of habitat restoration has not met goals set forth in management plans because of funding, personnel, and private-land limitations. Commonly, habitat restoration funds are divided between competing entities, which usually limits the size of restoration projects. Furthermore, large projects, such as those planned for the BBC watershed, are expensive and require coordinated funding efforts because funding a single, large project impacts funding for multiple smaller projects. There are two primary entities conducting restoration work in the Potlatch River basin: LSWCD and the IDFG. These entities have fiscal and administrative constraints that restrict the number of projects they can implement annually. Private ownership is another major factor affecting project implementation in the basin. Restoration partners spend a considerable amount of time and effort building relationships with private landowners. Although these efforts are valuable in generating interest and investment in the project, they limit the pace of project implementation in the short term.

Restoration projects put in place within the index watersheds are not yet to a scale projected to produce a detectable population-level response. The majority of projects completed thus far have been either one or more of the following: 1) relatively small in scale, 2) occurred outside IMW index watersheds, 3) provide more long-term, indirect benefits (i.e., sediment reduction via road decommissioning). Our modeling exercises suggested that a small number of large projects have the potential to produce a detectable response by steelhead in the BBC watershed; however, funding large projects is a challenge. The high cost (>$2,000,000 combined) has forced practitioners to either proceed with lower-cost, and possibly less effective, alternatives (i.e., installing baffles in a culvert vs. replacing culvert completely) or find additional funding opportunities to reduce the burden to any one source. Further, some of these projects fall outside the traditional approach to habitat restoration (e.g., restoring/enhancing steelhead passage at Big Bear Falls, a natural barrier altered by degraded flow conditions [Bowersox et al. 2016]) and need additional policy-level support to proceed. Once these issues are resolved, we are optimistic these large projects will produce a detectable response.

We have used an adaptive management approach to better align restoration actions with monitoring priorities in the IMW. Initially, restoration planning and implementation was guided by the Potlatch River Watershed Management Plan (Resource Planning Unlimited 2007). This document provided a broad road map for restoration actions and prioritization. Early restoration projects were implemented opportunistically throughout the basin, and it was not until the current IMW framework was established in 2008 that priorities were put forth. Now that additional monitoring data have been collected, a more refined restoration-planning document needs to be

developed. In 2016, the Idaho Office of Species Conservation formed the Potlatch River Implementation Group, comprised of restoration partners and state and federal management agencies, to better coordinate and plan restoration activities within the basin. A primary goal of the group is to develop an updated project implementation and prioritization framework for habitat restoration in the Potlatch River basin, specifically within the index watersheds. To facilitate project prioritization in the basin, life-cycle model outputs of smolt production can be combined with project costs to conduct cost-benefit analyses in terms of cost per smolt for individual projects. We provide a brief example in Appendix J. This example demonstrates how data collected through the IMW study will be vital in developing more efficient and strategic project implementation within the Potlatch River habitat restoration program.

Changes to the Potlatch River IMW monitoring design have strengthened our ability to detect a population-level fish response and understand the mechanisms responsible for generating the response. Initial monitoring efforts focused on intensive monitoring of steelhead production and productivity at the basin scale and juvenile density at the tributary level. These data established baseline levels of emigrant/spawner productivity and life history diversity in the study populations and pretreatment levels of juvenile density in treatment and control tributaries. In 2013, we leveraged our pretreatment data set and incorporated a BACI design to evaluate changes in juvenile density, survival, and growth to restoration projects at the tributary scale. These efforts, coupled with reach-scale monitoring efforts, provided us greater ability to isolate responses from multiple restoration types and better understand the causes of a response. In this report, we developed a life cycle model to estimate potential responses in juvenile production to restoration actions in the watersheds. The model provides a mechanism to establish informed restoration goals or benchmarks and prioritize restoration actions in the index watersheds. Overall, these modifications allow us to better meet management objectives.

Additional improvements to our monitoring design will help us better evaluate future restoration efforts. A small increase in the number of electrofishing sites in the lower watershed will reduce the post-treatment monitoring time to reliably detect a significant increase in juvenile density to <10 years. We will add three to five additional electrofishing sites per tributary in the lower watershed. In contrast, more substantial changes are needed to the upper watershed tributary-level monitoring to reduce the post-treatment monitoring time. Under the current BACI framework, it is estimated to take 20-25 years to detect a significant increase in juvenile density. We are exploring options to reduce the timeframe of post-treatment monitoring, including using a staircase monitoring design (Walters et al. 1988) in place of the current BACI design. One advantage to using a staircase design is that by staggering projects within the EFPR, we can use treatment sections as controls until they are treated, thus preventing the loss of other control areas (Roni 2005). Under this design, we would incorporate an additional control reach within the upper EFPR watershed, since all completed and planned instream treatments are concentrated in the lower 22 km of the drainage (below Pivash Creek). We will re-run the power analysis with this design to determine the sampling effort needed to reliably detect a significant response. In addition, we are making changes to PIT-tag array infrastructure and seasonal sampling priorities to better assess juvenile survival and growth in the watershed. These changes, coupled with ongoing watershed-scale monitoring, will increase our power to detect a response and subsequently reduce the timeframe of post-treatment monitoring.

The next 10-15 years will be critical to achieve the goals of the Potlatch River IMW study. Within the next year, we aim to complete the updated restoration planning and implementation document and make improvements to our tributary-level monitoring design. We are also working towards developing a main-stem Potlatch River PIT-tag array to improve the monitoring of life history strategies and juvenile survival within index watersheds as well as to assess annual adult

escapement into the entire watershed. The array will be valuable to restoration effectiveness monitoring in the watershed by partitioning mortality downstream of the index watersheds, but will have additional value in regards to status monitoring of the Lower Clearwater Mainstem steelhead population. Restoration implementation and effectiveness monitoring will be ongoing during this time, and we anticipate the bulk of planned restoration projects will be completed within the index watersheds during the next 5 to 7 years. Once planned restoration projects are implemented, we estimate a minimum 5 to 10 year post-treatment monitoring period is needed to gather data through multiple steelhead life cycles and ensure an adequate data set to detect changes in production/productivity related to restoration efforts. During a five year post-treatment monitoring period, we could only generate two complete brood year juvenile recruit/spawner productivity estimates for evaluation. In contrast, a 10 year post-monitoring period would allow us to generate seven complete brood year productivity estimates and provide an adequate timeframe to assess tributary-level changes in juvenile production metrics.

In addition to restoration evaluation, data from the Potlatch River IMW have been used to inform status assessment and recovery efforts of the Lower Clearwater Main-stem steelhead population and inform regional management decisions. The Potlatch River spawning area is critical to the recovery of the Clearwater River MPG. As such, the Potlatch River dataset provides a valuable monitoring component for the Lower Clearwater Main-stem steelhead population (NOAA 2016). Age structure and length-at-age data of Potlatch River steelhead (juveniles and adults) has been integrated into the Federal Columbia River Power System's Biological Opinion and NOAA's draft recovery plan (NMFS 2008; NOAA 2016). In addition, Potlatch River juvenile steelhead distribution data has been used extensively to comment on in-water work windows, USFS Forest Plans, and fill-removal applications within the study area. We regularly communicate project results at local, state, and regional venues. Collectively, these efforts result in greater protection of wild steelhead in the lower Clearwater River drainage and contribute towards the recovery of this ESA-listed population.

CONCLUSION

Initial efforts have been made towards evaluation of the Potlatch River restoration program, but much more work is needed. A moderate amount of restoration work has been completed in the index watersheds, including barrier removals to provide access to additional spawning and rearing habitat, LWD and/or riparian treatments to increase instream habitat complexity, and flow supplementation to enhance base flow conditions. Preliminary fish and habitat responses are promising and provide validation to these approaches. Current planning efforts in the basin led by the Idaho Office of Species Conservation have improved communication and coordination between restoration and monitoring components, resulting in more efficient project implementation. Several high-priority projects are in the planning and development phase in the BBC watershed; however, funding and institutional issues will take considerable time and effort to resolve before these projects can be fully implemented. The monitoring framework in the lower Potlatch River watershed is robust, and relatively minor changes are needed to improve future evaluations. In contrast, a greater restoration and monitoring effort is needed in the upper watershed, including changes to the monitoring design to increase the power to detect significant changes and reduce the timeframe of post-treatment monitoring. Improvements to our tributary-level monitoring efforts will help us better isolate responses to specific restoration projects and provide an improved framework to understand how these responses propagate up to the watershed-scale. An additional 10 to 15 year funding commitment is needed to allow time to complete project implementation and fully evaluate the population's response to these efforts. We

have built a solid foundation of restoration and monitoring over the first 10 years of this study and we will continue these efforts into the future to aid in the recovery of Potlatch River wild steelhead.

PART 2 TABLES

Table 2.2. Inventory of planned restoration projects in the Potlatch River index watersheds (Big Bear Creek=lower; East Fork Potlatch River=upper) during 2017 - 2022. LWD = large woody debris, PALS = post-assisted log structure.

Index watershed	Stream	Treatment	Project year	Anticipated stream treated (km)
Lower	Big Bear Creek	Barrier Removal	2021-2022	17.6
Lower	Spring Valley Creek/Little Bear Creek	Flow Supplementation	2018-2022	8.0
Lower	Big Meadow Creek/West Fork Little Bear Creek	Barrier Removal	2017	9.6
Lower	Big Meadow Creek/West Fork Little Bear Creek	LWD/PALS/Riparian	2019	1.8
Lower	Big Meadow Creek/West Fork Little Bear Creek	LWD/PALS/Riparian	2018	0.3
Lower	Nora Creek/ Little Bear Creek	Riparian/Meadow restoration	2017	2.4
Lower	Big Bear Creek	Riparian/Meadow restoration	2019	2.5
Lower	Big Bear Creek	Riparian/Meadow restoration	2019	0.9
Lower	Big Bear Creek	Riparian/Meadow restoration	2020-2022	3.2
Lower	Big Bear Creek	Riparian/Meadow restoration	2018	0.4
Lower	Big Bear Creek	Riparian/Meadow restoration	2020	0.4
Lower	Big Bear Creek	Riparian/Meadow restoration	2023	0.7
			Watershed Total	47.8
Upper	East Fork Potlatch River	LWD/PALS/Riparian	2018	1.0
Upper	East Fork Potlatch River	LWD/PALS/Riparian	2019	1.5
Upper	East Fork Potlatch River	LWD/PALS/Riparian	2017	0.3
Upper	East Fork Potlatch River	LWD/PALS/Riparian	2019	5.0
Upper	East Fork Potlatch River	LWD/PALS/Riparian	2020-2021	2.0
Upper	East Fork Potlatch River	Riparian/Meadow restoration	2018	3.5
Upper	East Fork Potlatch River	Barrier Removal	2017	1.3
			Watershed Total	14.6

Table 2.2. Productivity in juvenile steelhead emigrants per female spawner in Big Bear Creek by brood years 2005 - 2016.

Brood Year	Adult escapement	Proportion female	Female spawners	Emigrant age				Total Production	Juveniles/Female
				0	1	2	3		
2005	202	0.72	145	0	3,091	5,542	72	8,705	59.9
2006	50	0.40	20	3	2,366	2,298	880	5,547	277.3
2007	100	0.69	69	0	2,489	4,334	241	7,065	102.4
2008	127	0.35	44	40	1,238	6,067	168	7,513	169.0
2009	135	0.45	61	0	3,516	3,332	277	7,125	117.3
2010	251	0.60	151	13	337	6,538	1,049	7,937	52.7
2011	124	0.53	66	0	4,121	11,106	523	15,750	239.7
2012	317	0.68	216	30	10,494	7,011	932	18,467	85.7
2013	122	0.70	85	0	942	4,880	192	6,014	70.4
2014	249	0.50	125	0	2,742	3,216	-	5,958	47.9
2015	109	0.44	48	0	2,446	-	-	2,446	51.0
2016	254	0.68	173	-	-	-	-	0	0.0

Table 2.3. Apparent survival to Lower Granite Dam and associated detection probabilities for smolts tagged at the juvenile screw trap in Big Bear Creek during 2005-2016. SE=standard error.

Year	*n*	Apparent Survival	SE	Detection probability	SE
2005	2278	0.67	na	na	na
2006	537	0.38	na	0.35	na
2007	1264	0.38	0.03	0.23	na
2008	836	0.76	0.07	0.49	na
2009	926	0.38	0.02	0.43	0.03
2010	1827	0.54	0.04	0.22	0.02
2011	569	0.81	0.04	0.37	0.03
2012	1857	0.32	0.02	0.37	0.02
2013	1830	0.44	0.03	0.29	0.02
2014	2330	0.60	0.04	0.25	0.02
2015	2916	0.27	0.05	0.12	0.02
2016	1167	0.35	0.02	0.38	0.03

Table 2.4. Productivity in juvenile steelhead emigrants per female spawner in the East Fork Potlatch River by brood years 2008 - 2016.

Brood Year	Adult escapement	Proportion female	Female spawners	Emigrant age 0	1	2	3	Total BY Production	Juveniles/Spawner
2008	140	0.33	46	205	9,528	7,092	77	16,902	368.0
2009	92	0.46	42	20	23,143	3,682	602	27,447	652.7
2010	72	0.76	55	790	10,662	2,408	656	14,516	266.3
2011	33	0.59	19	0	8,547	5,902	385	14,835	764.3
2012	101	0.52	53	430	33,666	4,497	1,008	39,601	752.4
2013	81	0.58	47	49	5,653	4,947	359	11,008	234.2
2014	96	0.47	45	0	9,527	3,086	-	12,613	279.2
2015	105	0.56	59	0	4,521	-	-	4,521	76.2
2016	89	0.61	54	-	-	-	-	0	0.0

Table 2.5. Apparent survival to Lower Granite Dam and detection probabilities for smolts tagged at the juvenile screw trap in the East Fork Potlatch River during 2008-2016. SE = standard error.

Year	*n*	Apparent Survival	SE	Detection probability	SE
2008	272	0.15	0.03	0.35	0.08
2009	1244	0.05	0.01	0.29	0.08
2010	2110	0.08	0.01	0.29	0.06
2011	1283	0.07	0.01	0.43	0.06
2012	894	0.06	0.01	0.56	0.07
2013	1413	0.17	0.13	0.13	0.10
2014	722	0.13	0.02	0.33	0.07
2015	814	0.17	0.09	0.14	0.08
2016	544	0.09	0.02	0.41	0.09

Table 2.6. Summer-fall growth rates (mm/d) of juvenile steelhead (≥80 mm) in select tributaries (West Fork Little Bear Creek [WFLBC], Little Bear Creek [LBC], and Pine Creek [PNC]) in the lower watershed of the Potlatch River, Idaho from 2014-2016.

Year	Tributary	Designation	Growth (mm/d)	SE	*n*	Days at large
2014	WFLBC	Treatment	0.059	0.012	26	123
	PNC	Control	0.060	0.010	28	147
2015	LBC	Treatment	0.124	0.032	2	138
	WFLBC	Treatment	0.028	0.010	10	132
	PNC	Control	0.068	0.022	16	141
2016	LBC	Treatment	0.025	0.004	77	127
	WFLBC	Treatment	0.006	0.006	43	117
	PNC	Control	0.043	0.008	64	130

PART 2 FIGURES

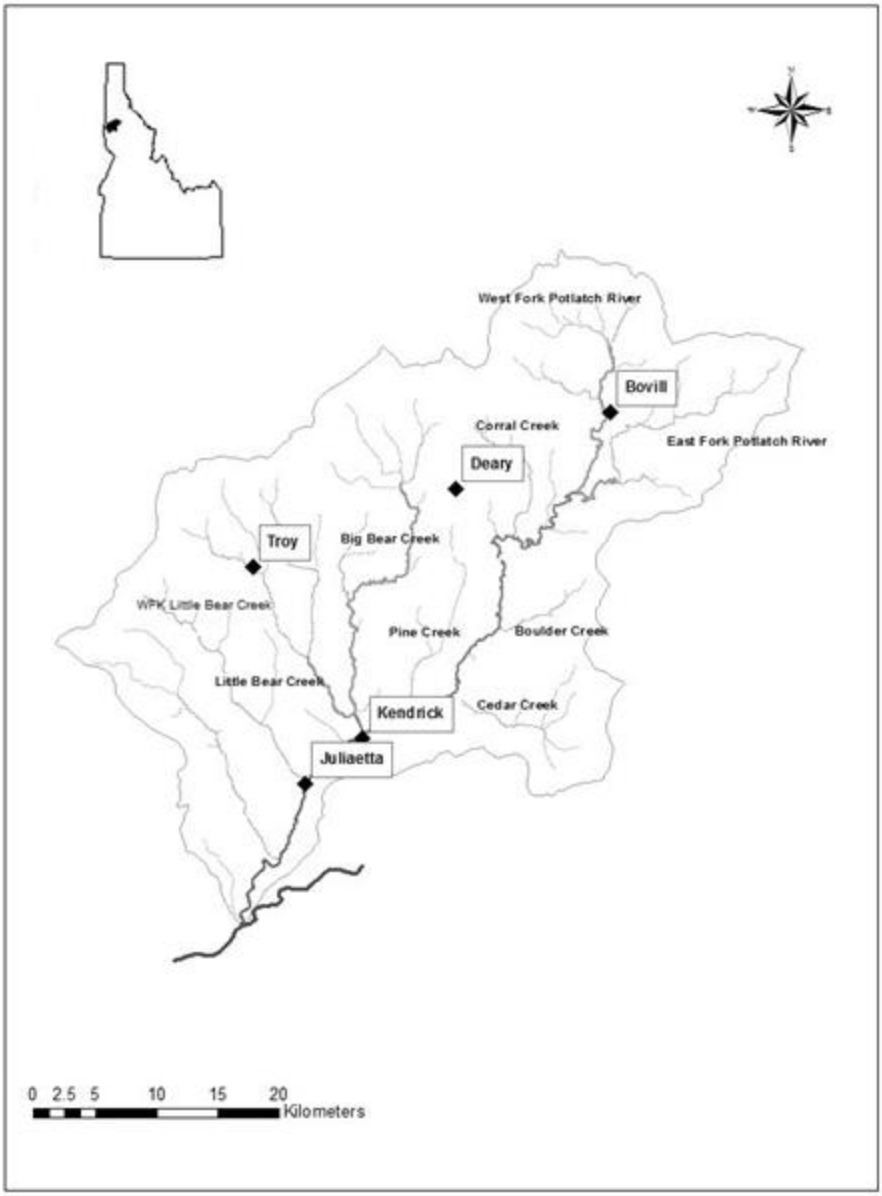

Figure 2.1. Location and key features of the Potlatch River basin in northern Idaho.

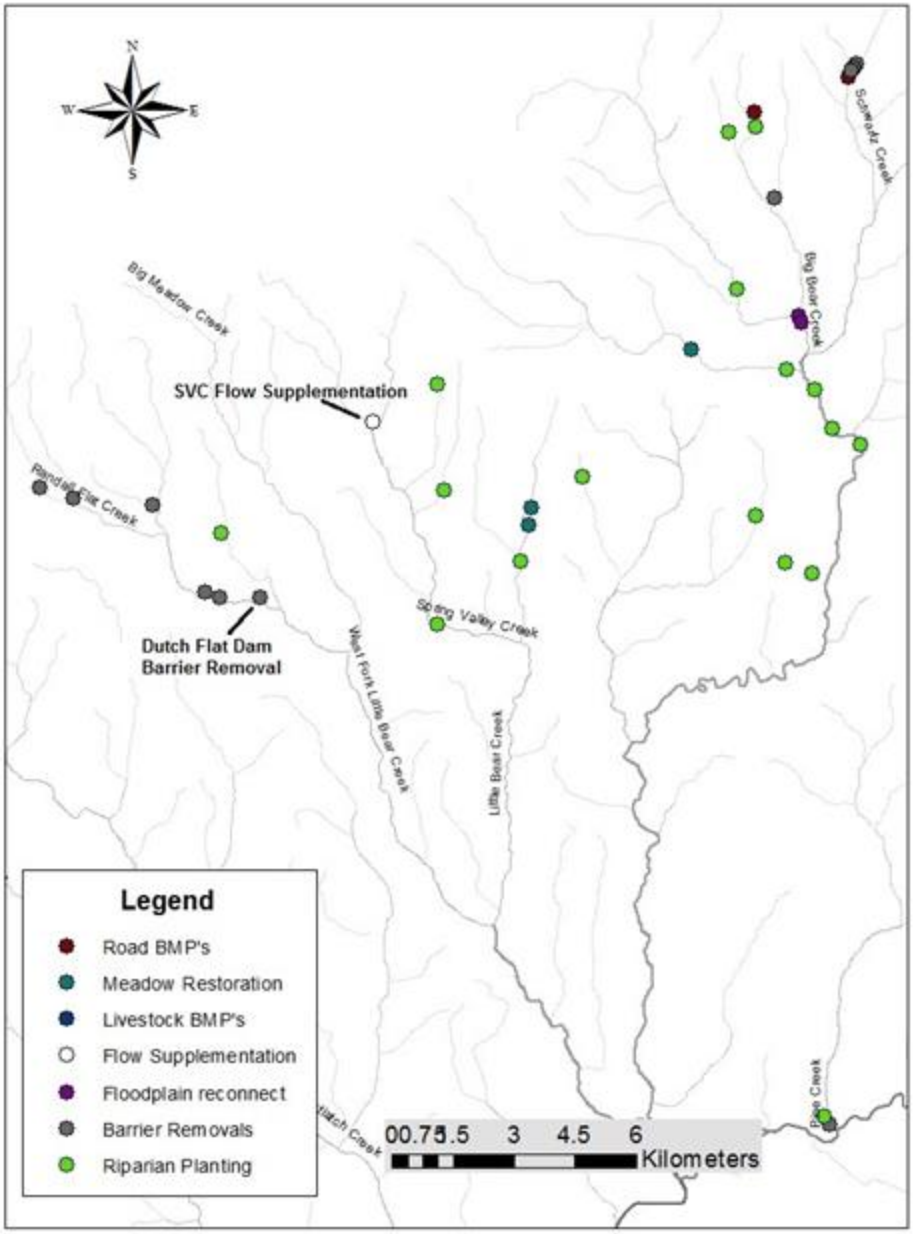

Figure 2.2. Locations of completed restoration projects in the Big Bear Creek watershed by type. Best management practices is abbreviated as BMP. The Spring Valley Creek (SVC) flow supplementation and Dutch Flat Dam removal projects are indicated.

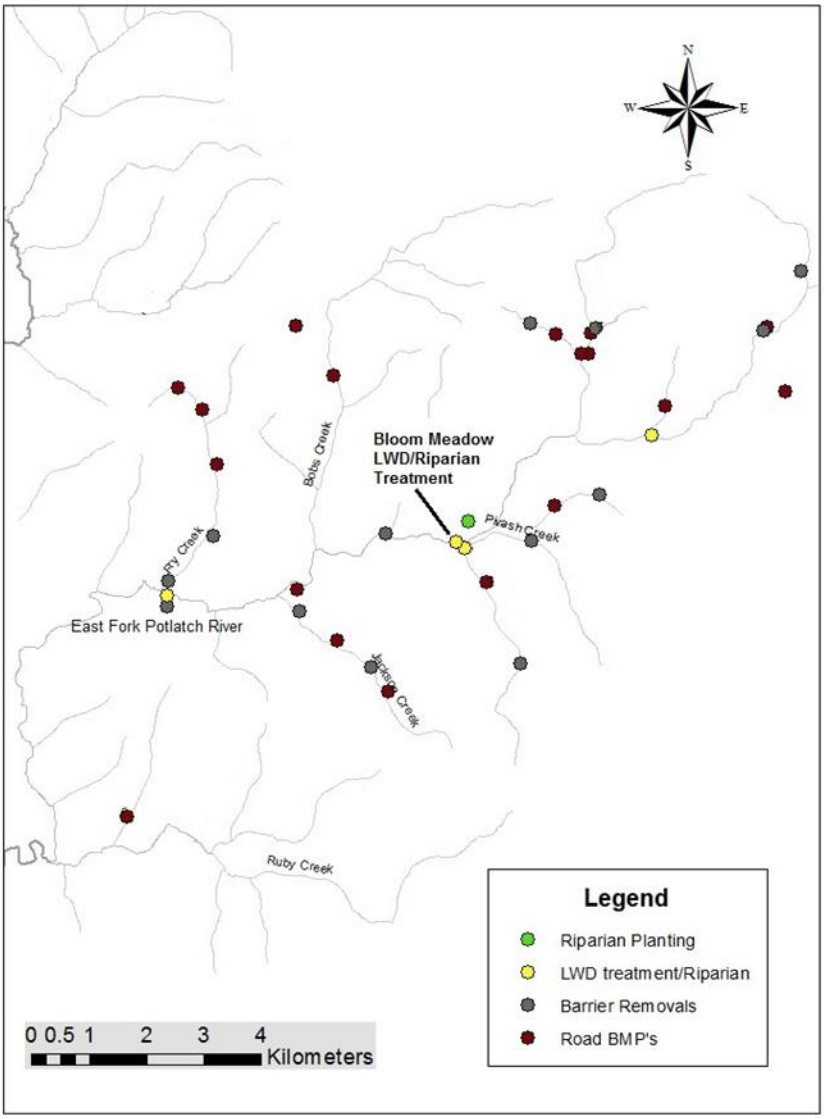

Figure 2.3. Locations of completed restoration projects by type in the East Fork Potlatch River watershed. LWD = Large woody debris, BMP = best management practices. The Bloom Meadow project is indicated.

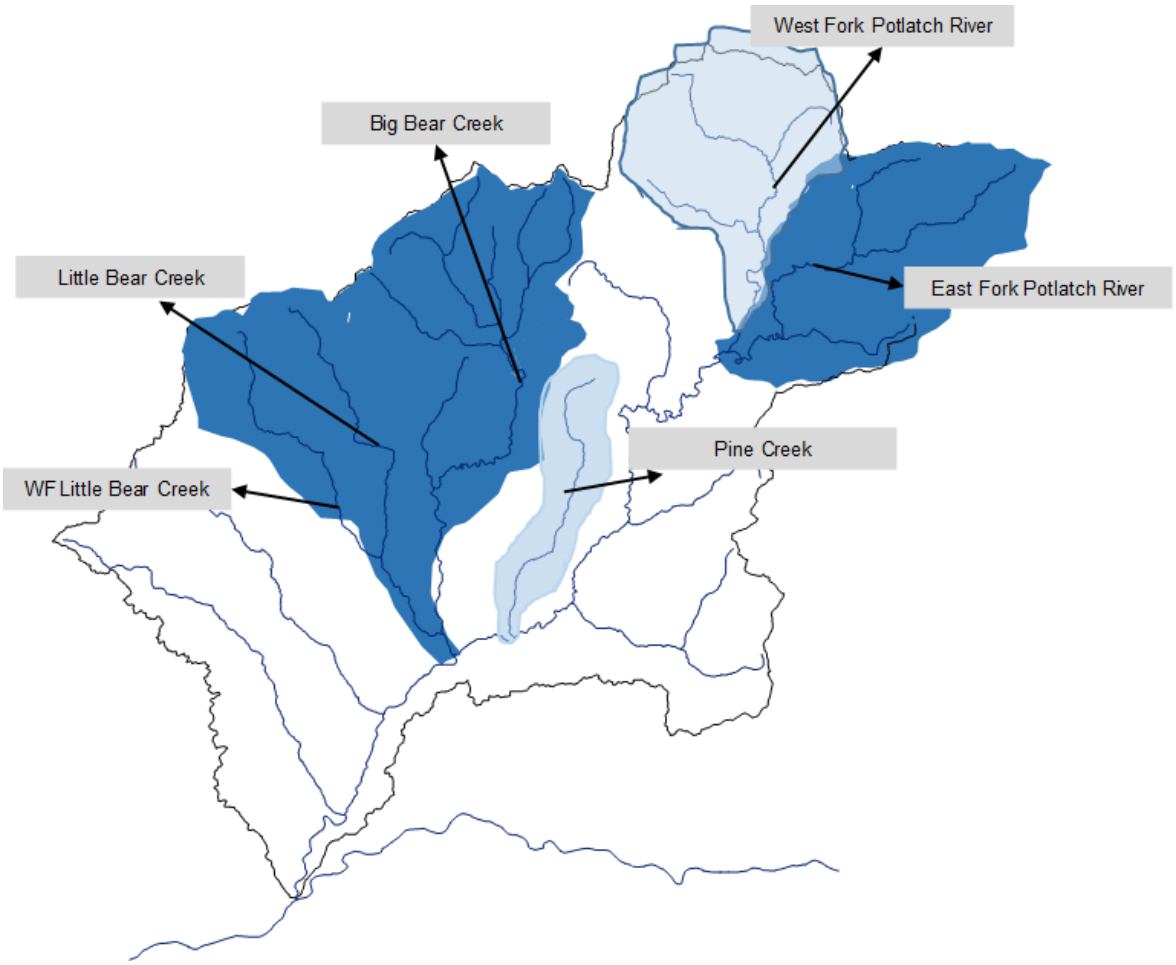

Figure 2.4. Treatment and control streams within the tributary-level study design in the Potlatch River watershed. Dark blue areas indicate treatment tributaries and light blue areas indicate control tributaries within each watershed.

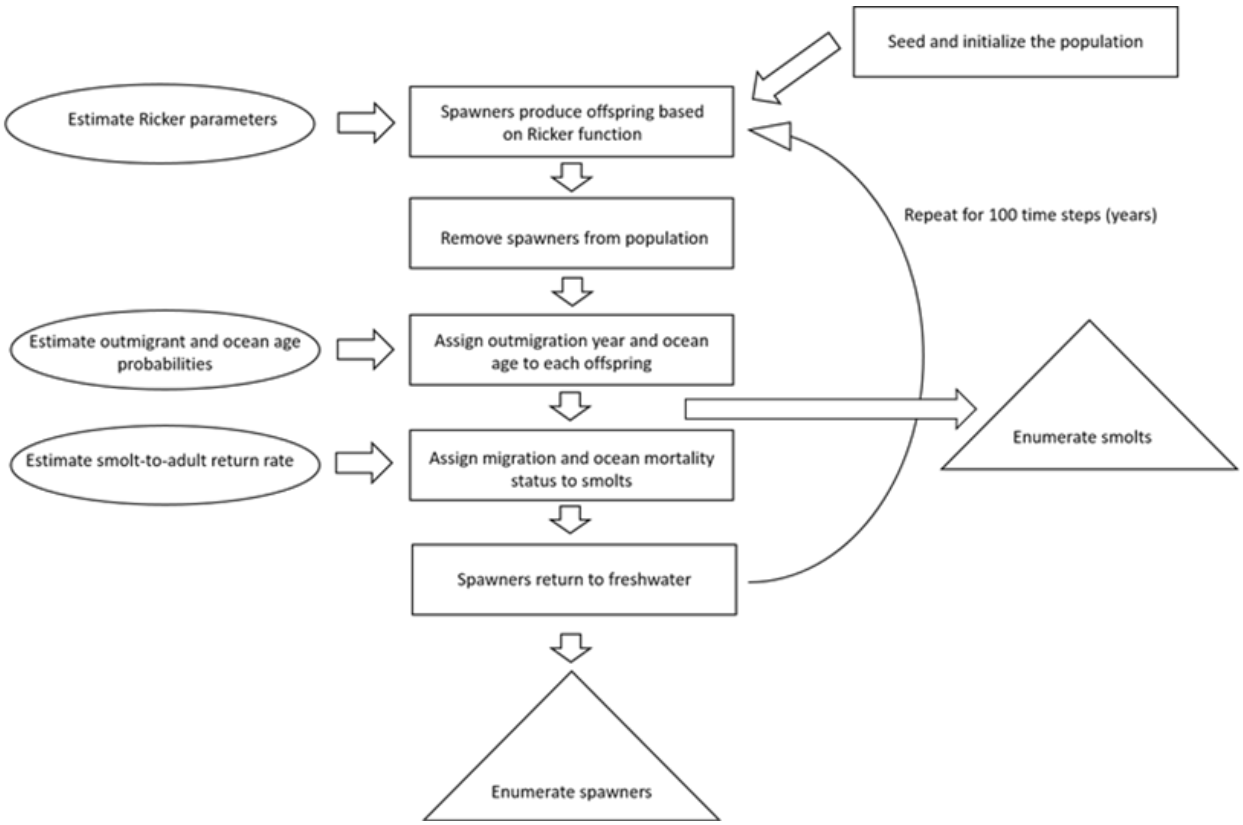

Figure 2.5. Flowchart of the individual-based simulation model. Ovals represent estimated parameters used to inform the model, and triangles represent outputs from the model.

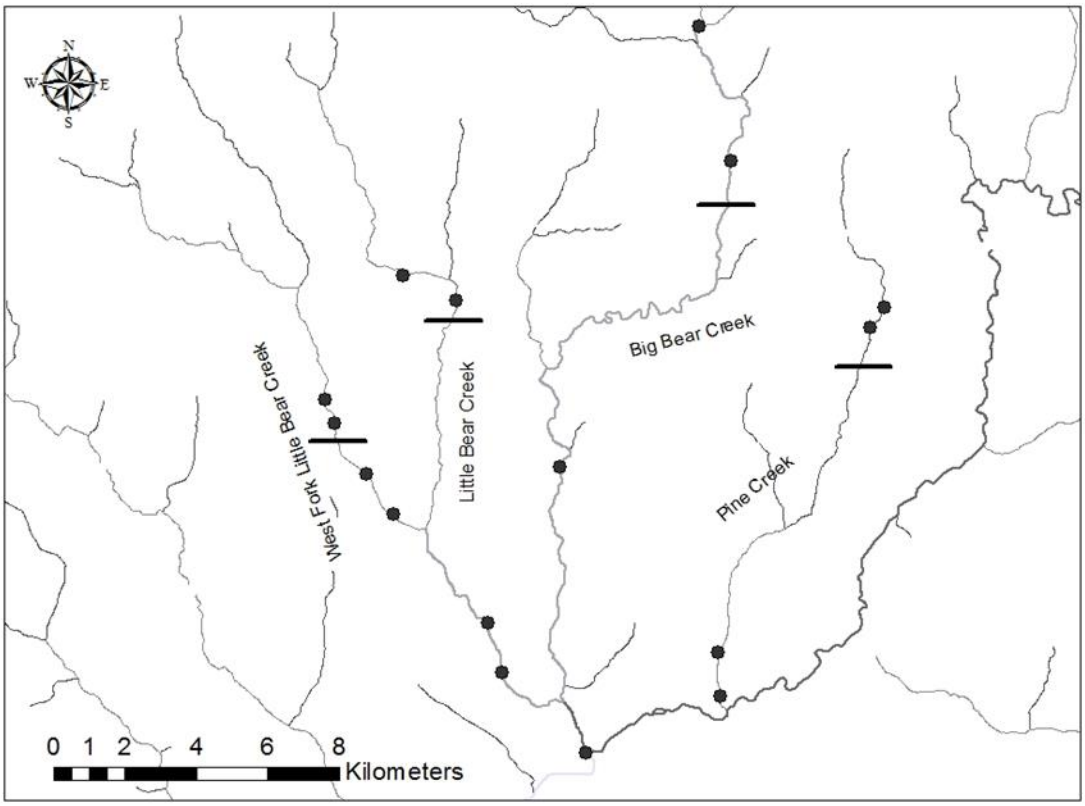

Figure 2.6. Locations of sites sampled using the low-water habitat availability protocol in the lower Potlatch River watershed. Treatment tributaries are West Fork Little Bear Creek, Little Bear Creek, and Big Bear Creek and the control tributary is Pine Creek. Vertical black lines indicate break between upland (upstream) and canyon reaches (downstream).

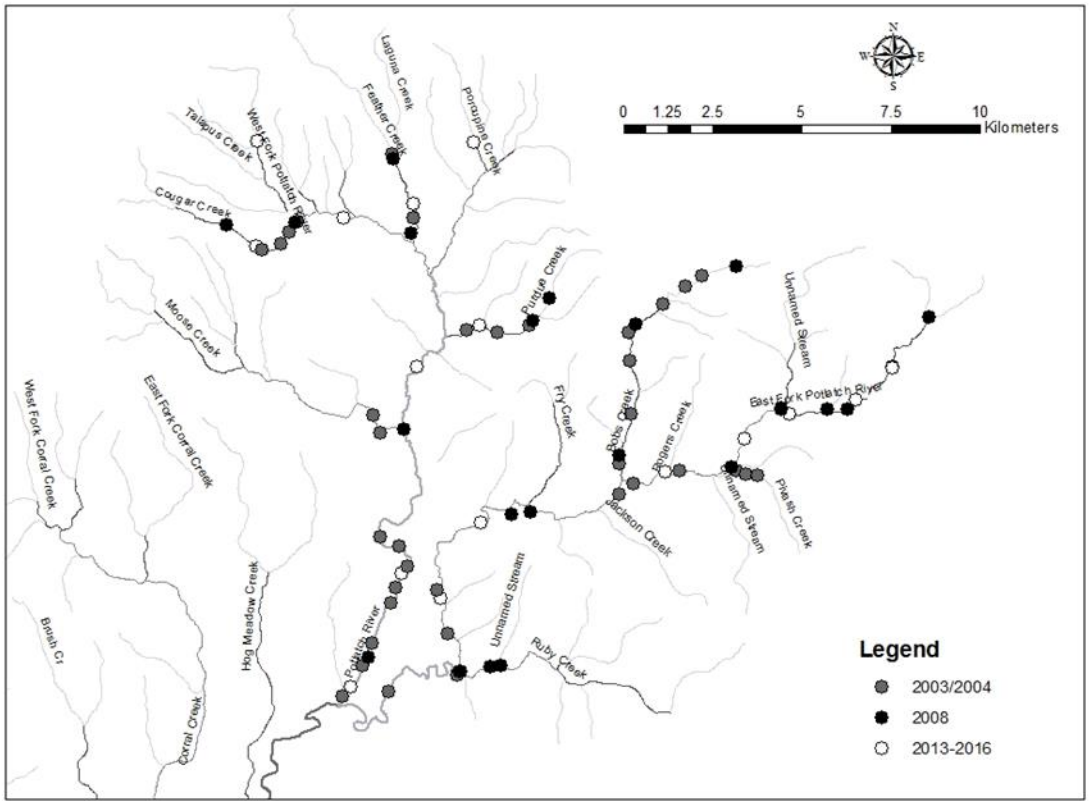

Figure 2.7. Locations of habitat survey sites within treatment and control streams in the upper Potlatch River watershed. Sites were surveyed during three time periods. The treatment stream is the East Fork Potlatch River and its tributaries and the control stream is the West Fork Potlatch River and its tributaries.

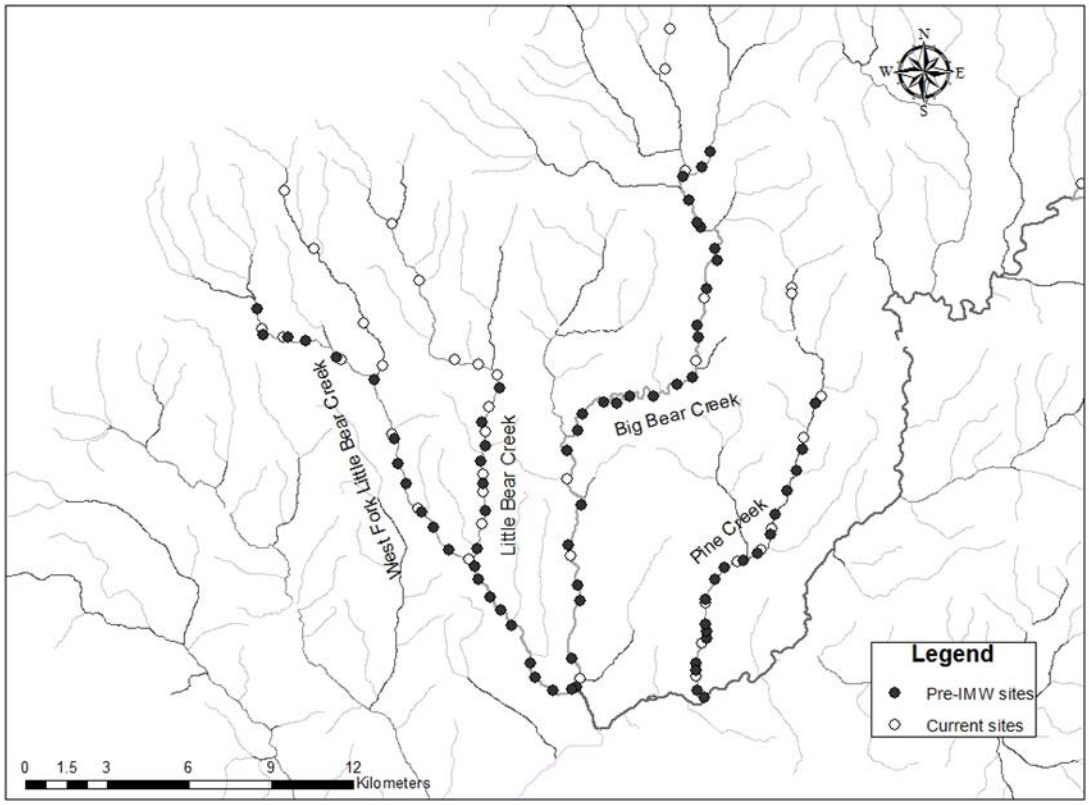

Figure 2.8. Locations of electrofishing sites within treatment and control tributaries in the lower Potlatch River watershed. Treatment tributaries are West Fork Little Bear Creek, Little Bear Creek, and Big Bear Creek and the control tributary is Pine Creek. Sites surveyed before the current study are distinguished from currently sampled sites.

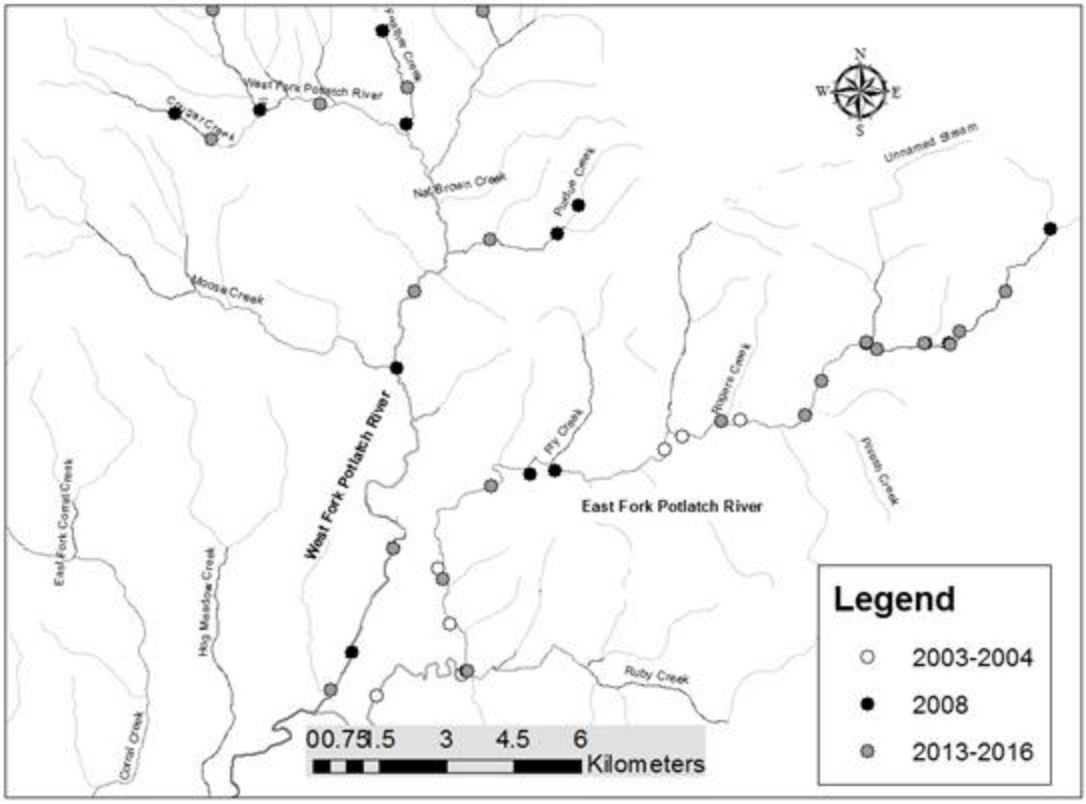

Figure 2.9. Locations of electrofishing sites in treatment and control tributaries within the upper Potlatch River watershed. Sites were surveyed during three time periods. The East Fork Potlatch River is the treatment stream and the West Fork Potlatch River is the control stream.

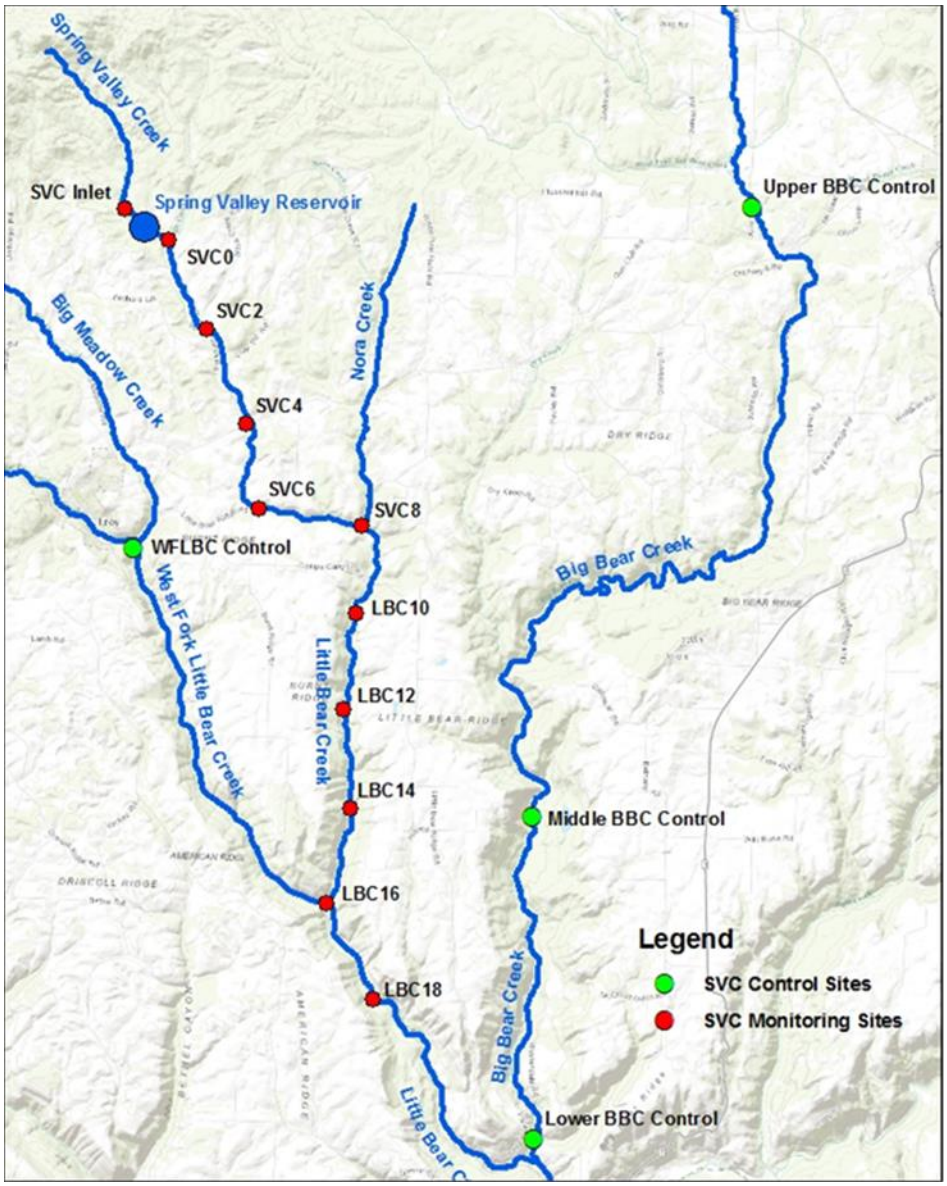

Figure 2.10. The Spring Valley Creek (SVC) flow supplementation project area. Control sites were located in West Fork Little Bear Creek (WFLBC) in 2015 and WFLBC and Big Bear Creek (BBC) in 2016.

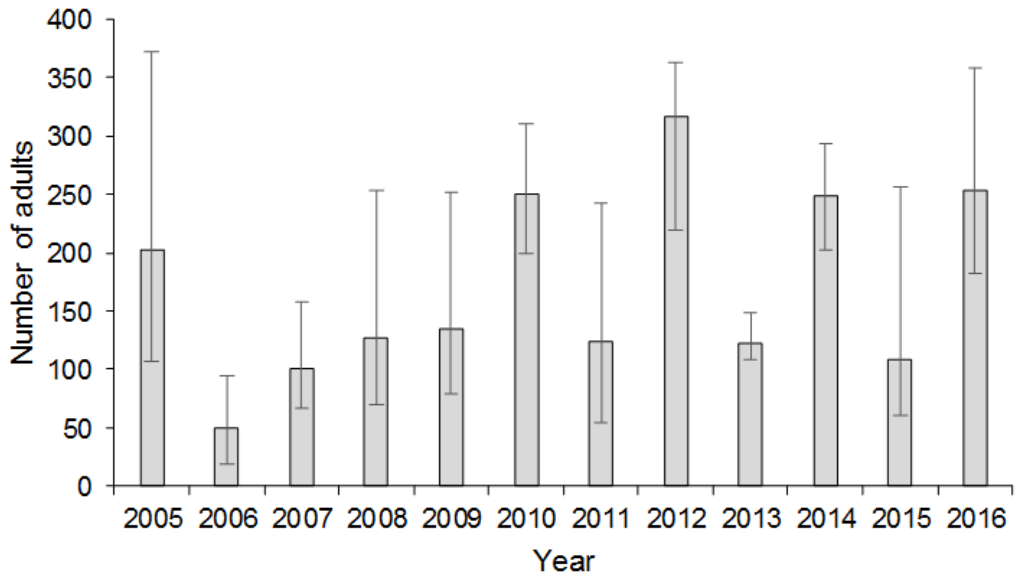

Figure 2.11. Adult steelhead escapement into the Big Bear Creek watershed during 2005-2016 with 95% confidence intervals.

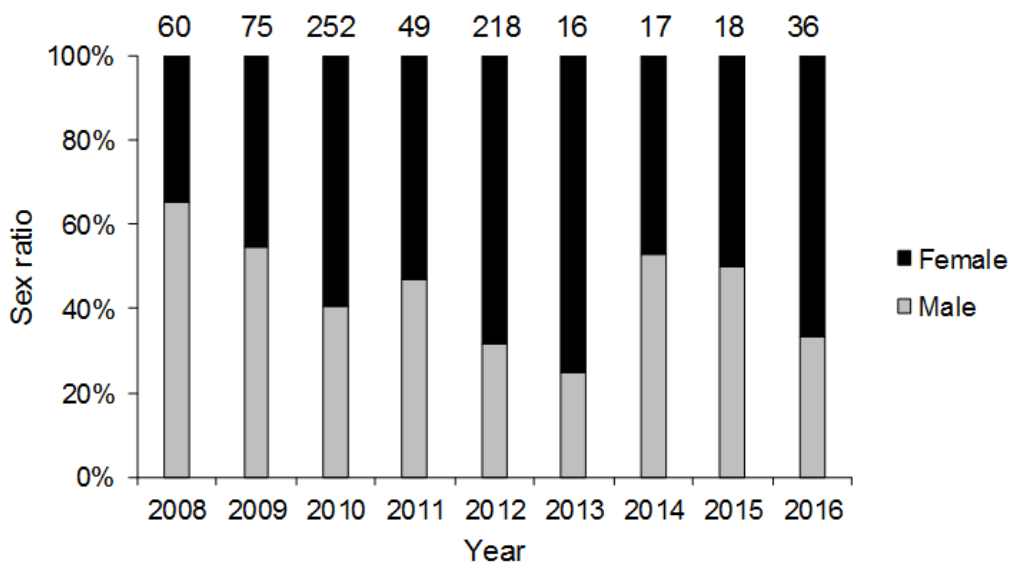

Figure 2.12. Sex ratio of adult steelhead in the Big Bear Creek watershed during 2008-2016. Numbers above bars indicate numbers of fish sexed.

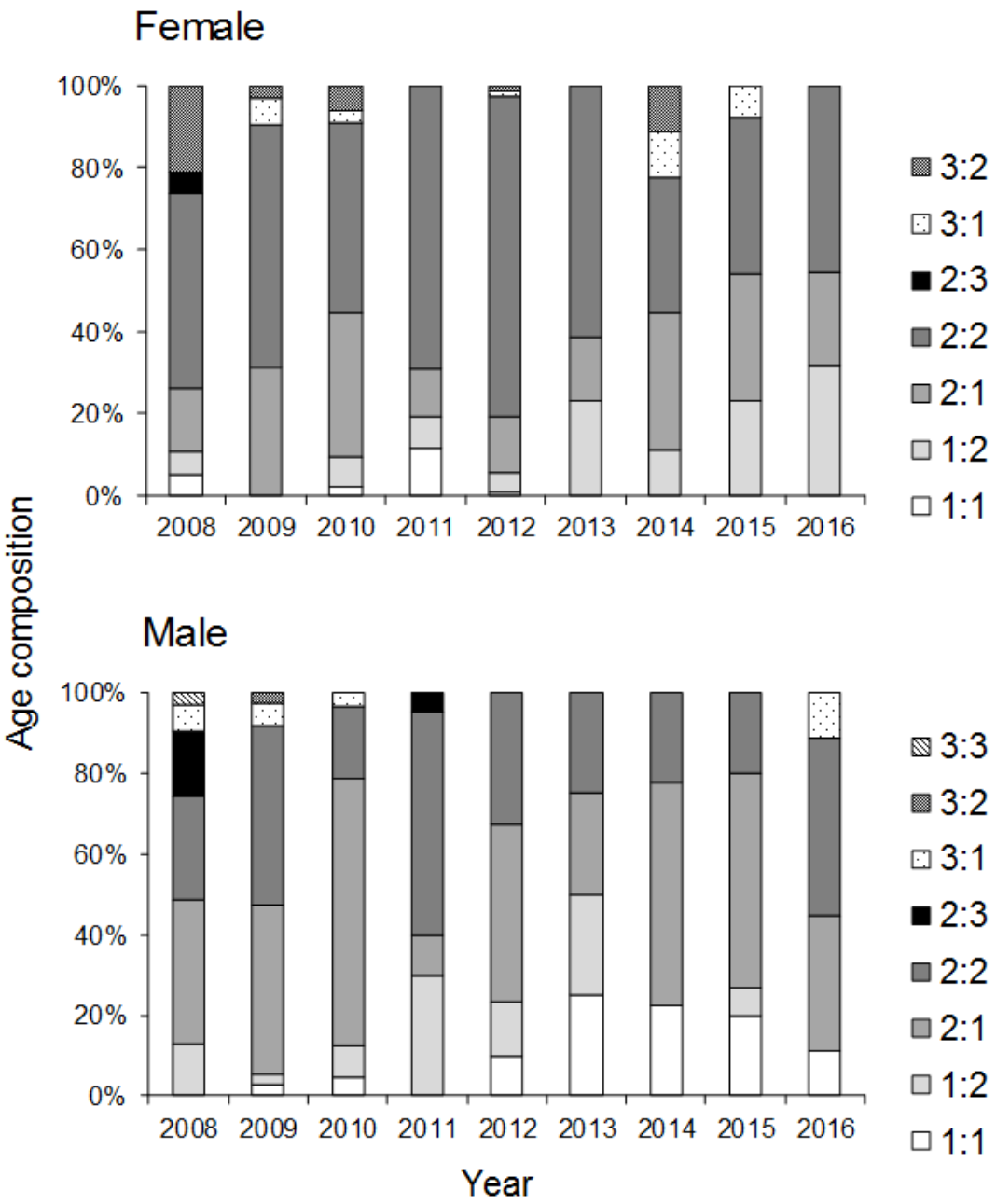

Figure 2.13. Age composition of adult steelhead in Big Bear Creek watershed during 2008-2016. Age designations show years in freshwater separated from years in the ocean by a colon.

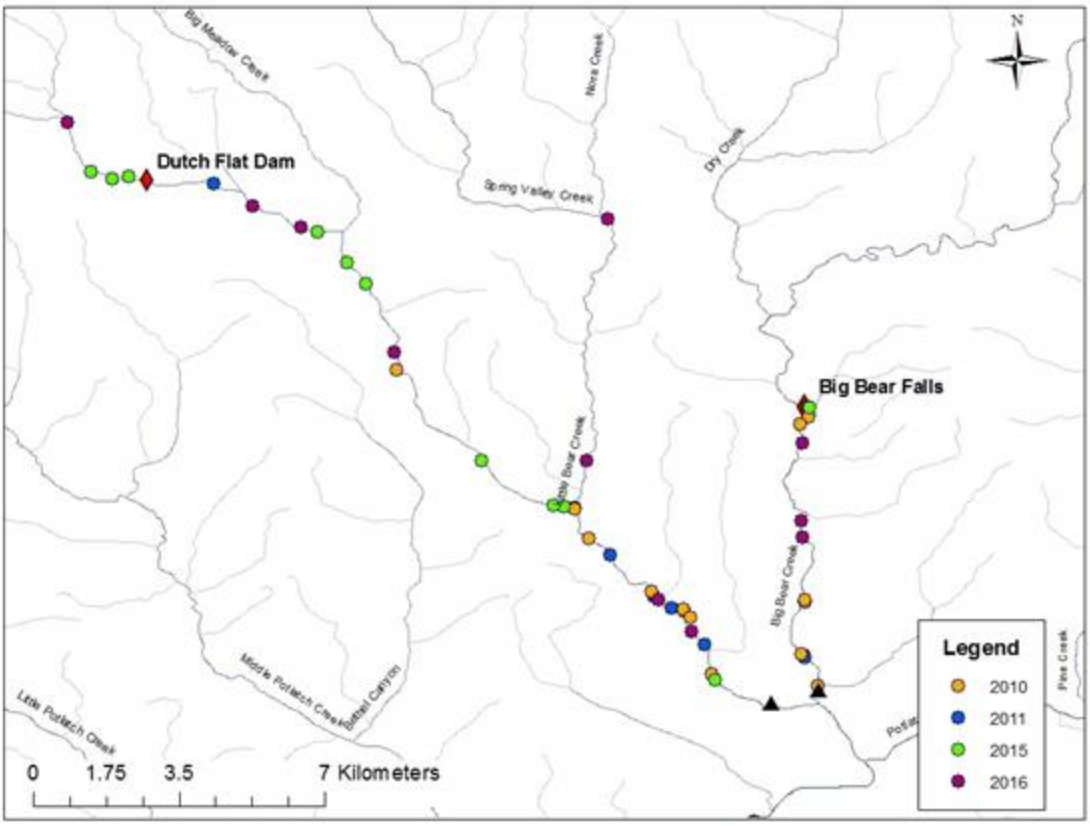

Figure 2.14. Spawning locations of radio-tagged adult steelhead in Big Bear Creek watershed by year. Points indicate furthest upstream detection recorded during tracking surveys. Black triangles indicate location of weirs where fish were radio tagged in 2010 and 2011. Fish were radio tagged at Lower Granite Dam trapping facility in 2015 and 2016.

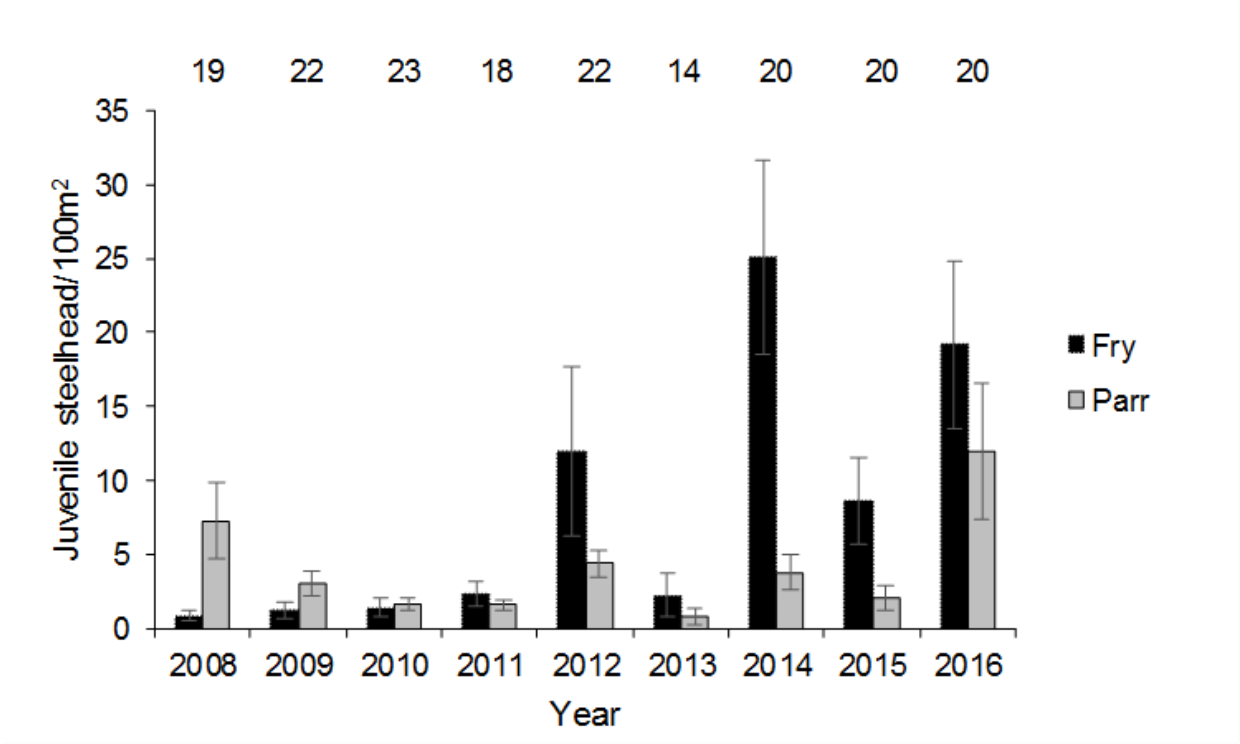

Figure 2.15. *Oncorhynchus mykiss* fry and parr densities from snorkel surveys within the Big Bear Creek watershed during 2008-2016 with standard deviations. Numbers indicate sites surveyed annually.

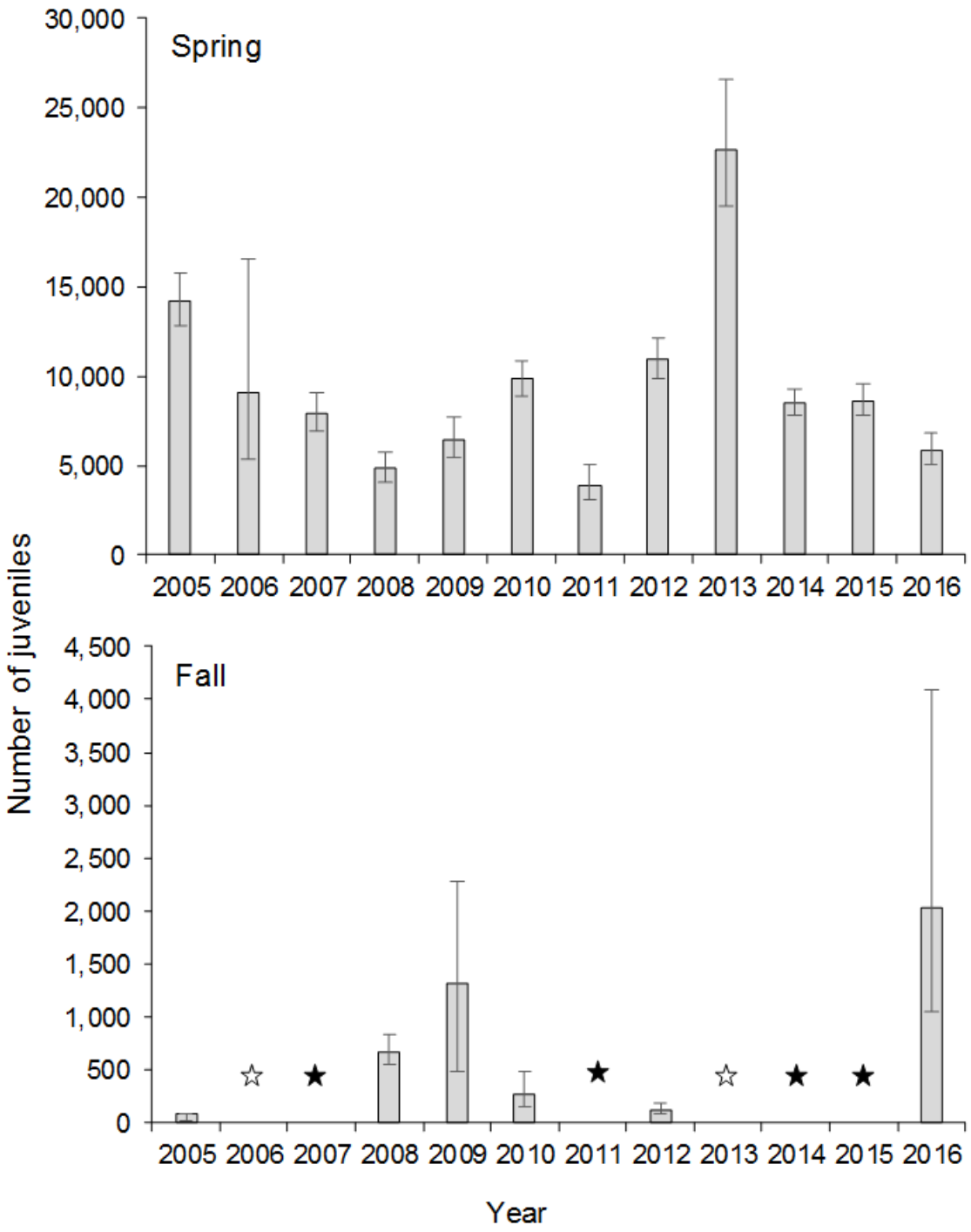

Figure 2.16. Numbers of juvenile steelhead annually emigrating from the Big Bear Creek watershed with 95% confidence intervals. Spring estimates are in the top panel and fall estimates are in the bottom panel. White stars indicate years when too few juveniles were captured to generate an estimate and black stars indicate years when the trap was not operated.

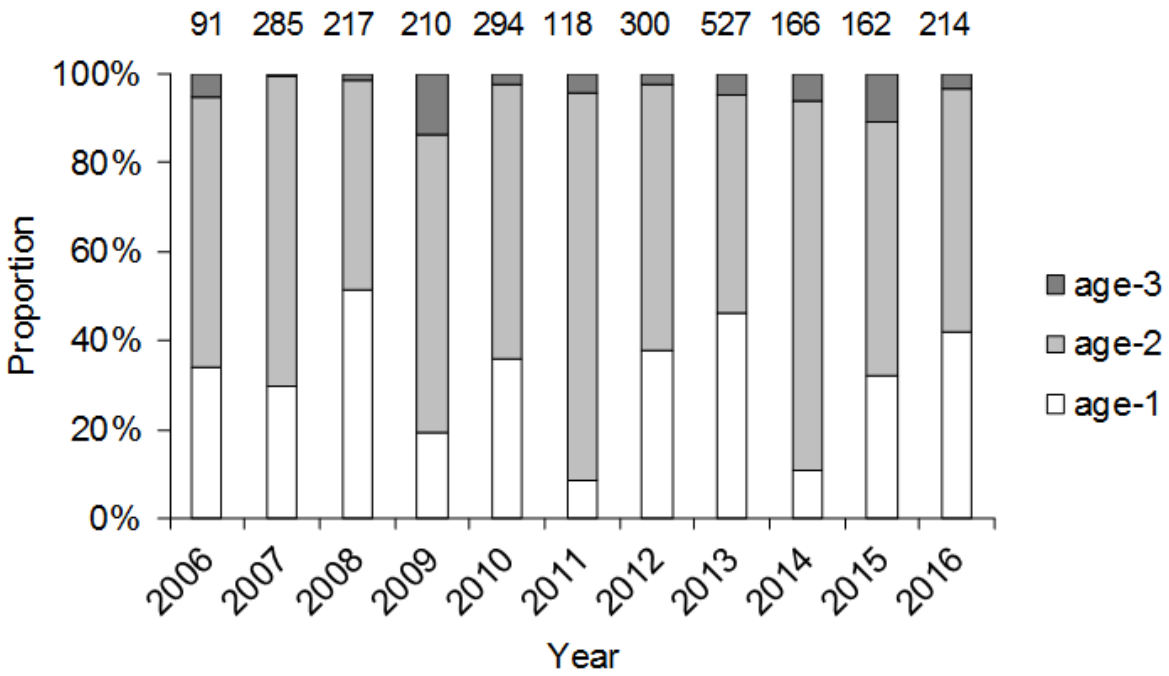

Figure 2.17. Age composition of juvenile steelhead emigrants from the Big Bear Creek watershed during 2006–2016. Numbers above bars indicate numbers of fish aged.

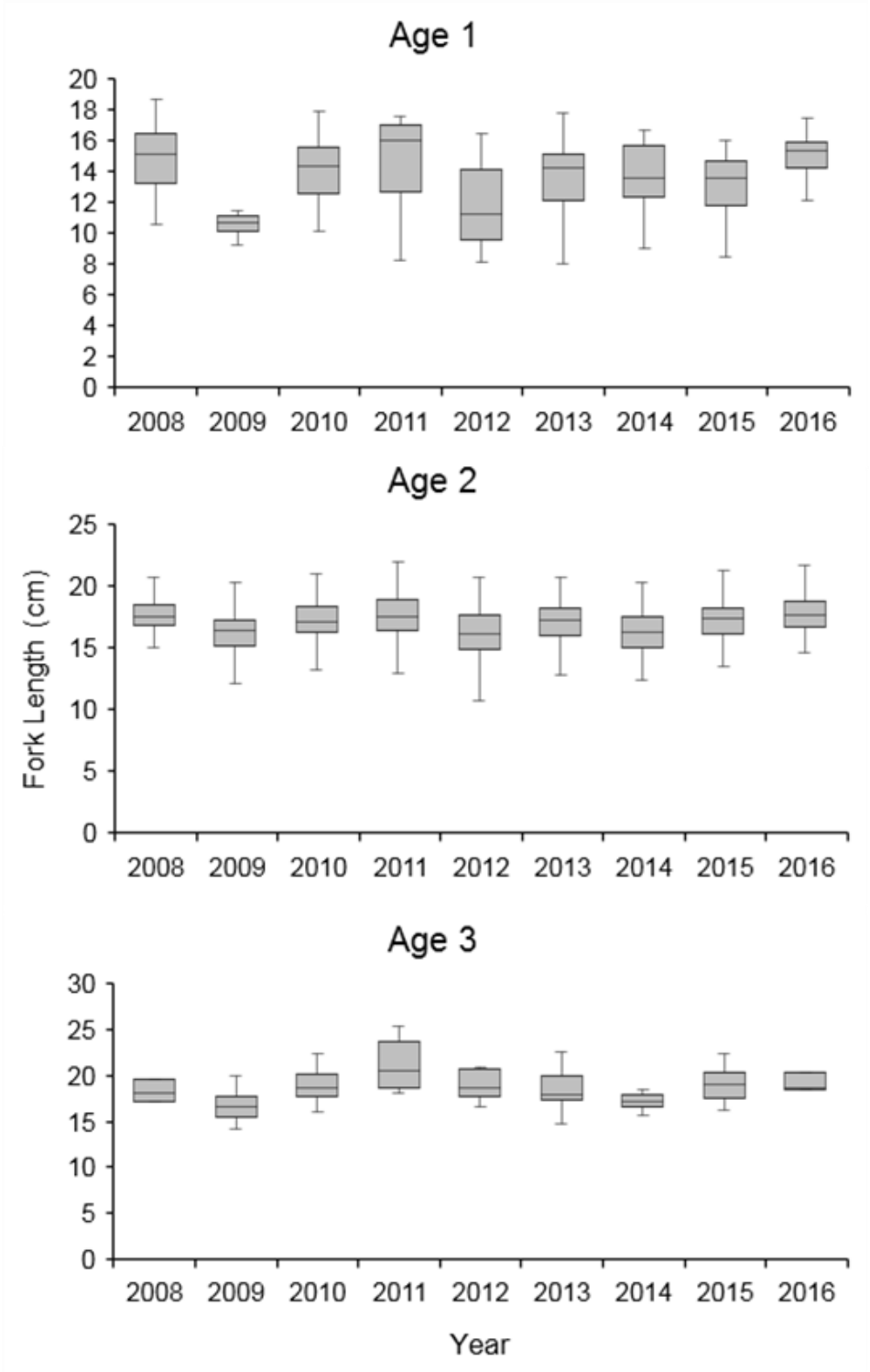

Figure 2.18. Boxplots of length-at-age of juvenile steelhead emigrants from the Big Bear Creek watershed during 2008-2016. The bottom and top of each box represents the first and third quartiles. The band inside the box represents the median and the whiskers indicate the minimum and maximum values.

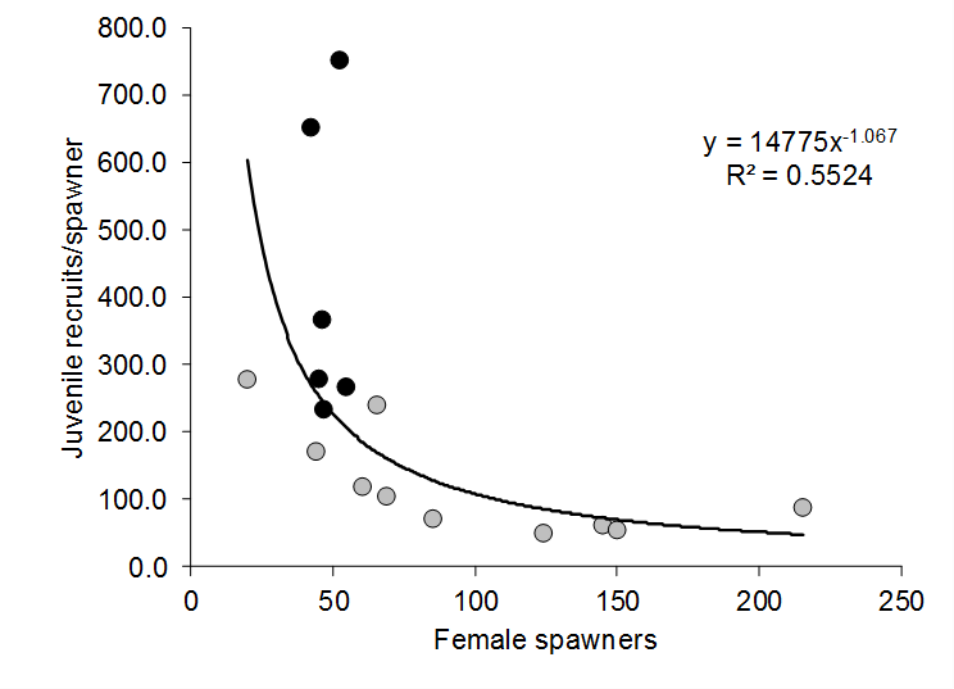

Figure 2.19. Juvenile productivity (juveniles/female) versus number of female spawners for the Big Bear Creek (grey dots) and East Fork Potlatch River (black dots) watersheds. Data for Big Bear Creek are from brood years 2005-2014 and the East Fork Potlatch River from brood years 2008-2014. Brood year 2011 for the East Fork Potlatch River was excluded because the parent estimate is incomplete.

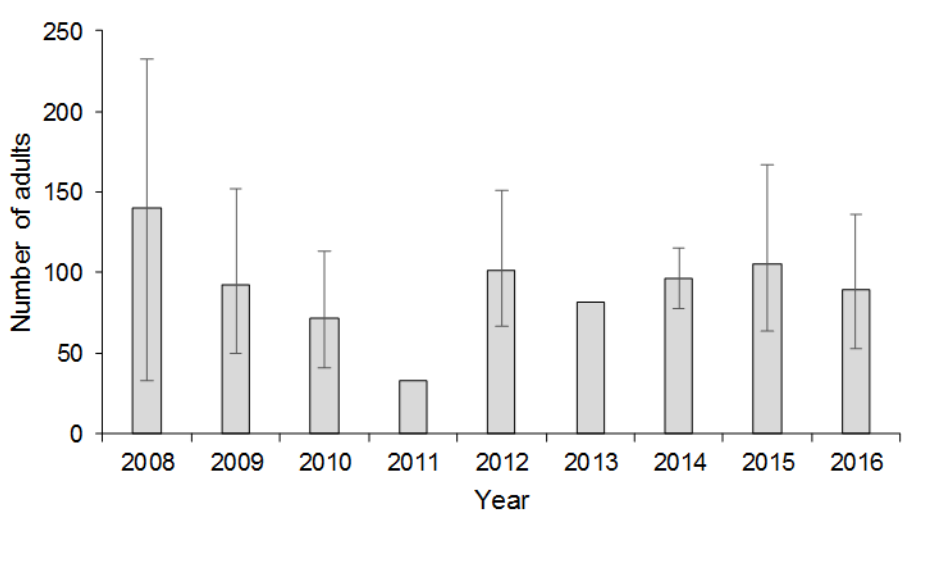

Figure 2.20. Adult escapement into the East Fork Potlatch River watershed during 2008-2016 with 95% confidence intervals. The 2011 estimate is incomplete because of high flows and the 2013 value is a census.

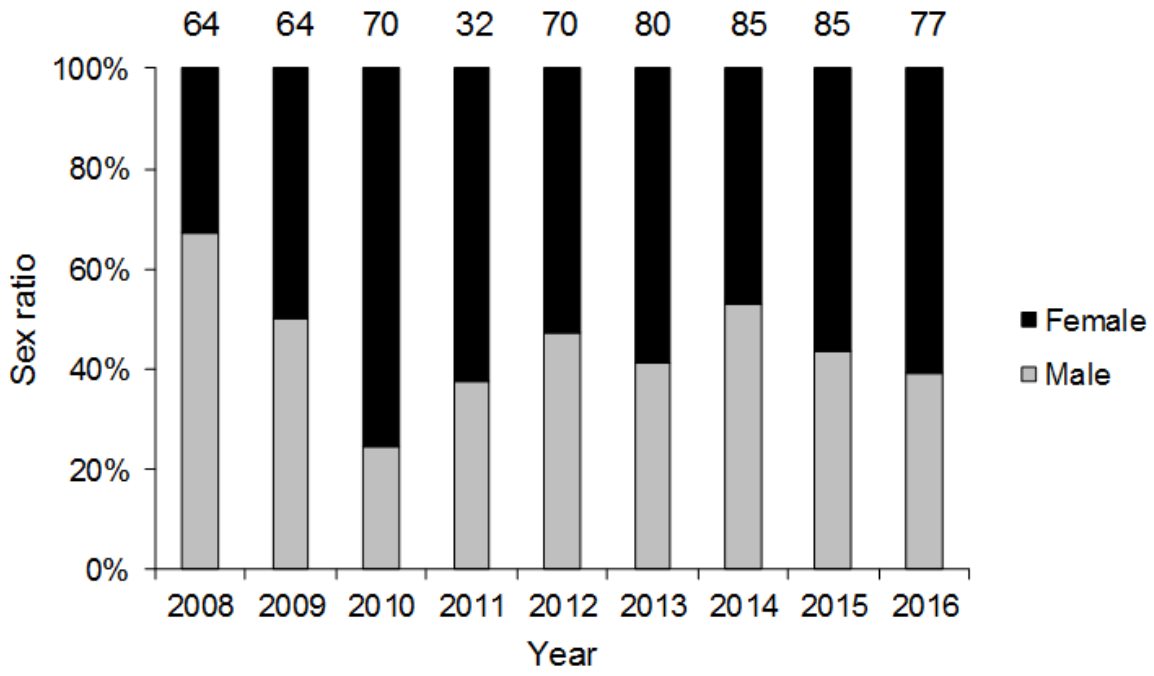

Figure 2.21. Sex ratio of adult steelhead in the East Fork Potlatch River watershed during 2008-2016. Numbers above bars indicate fish sexed.

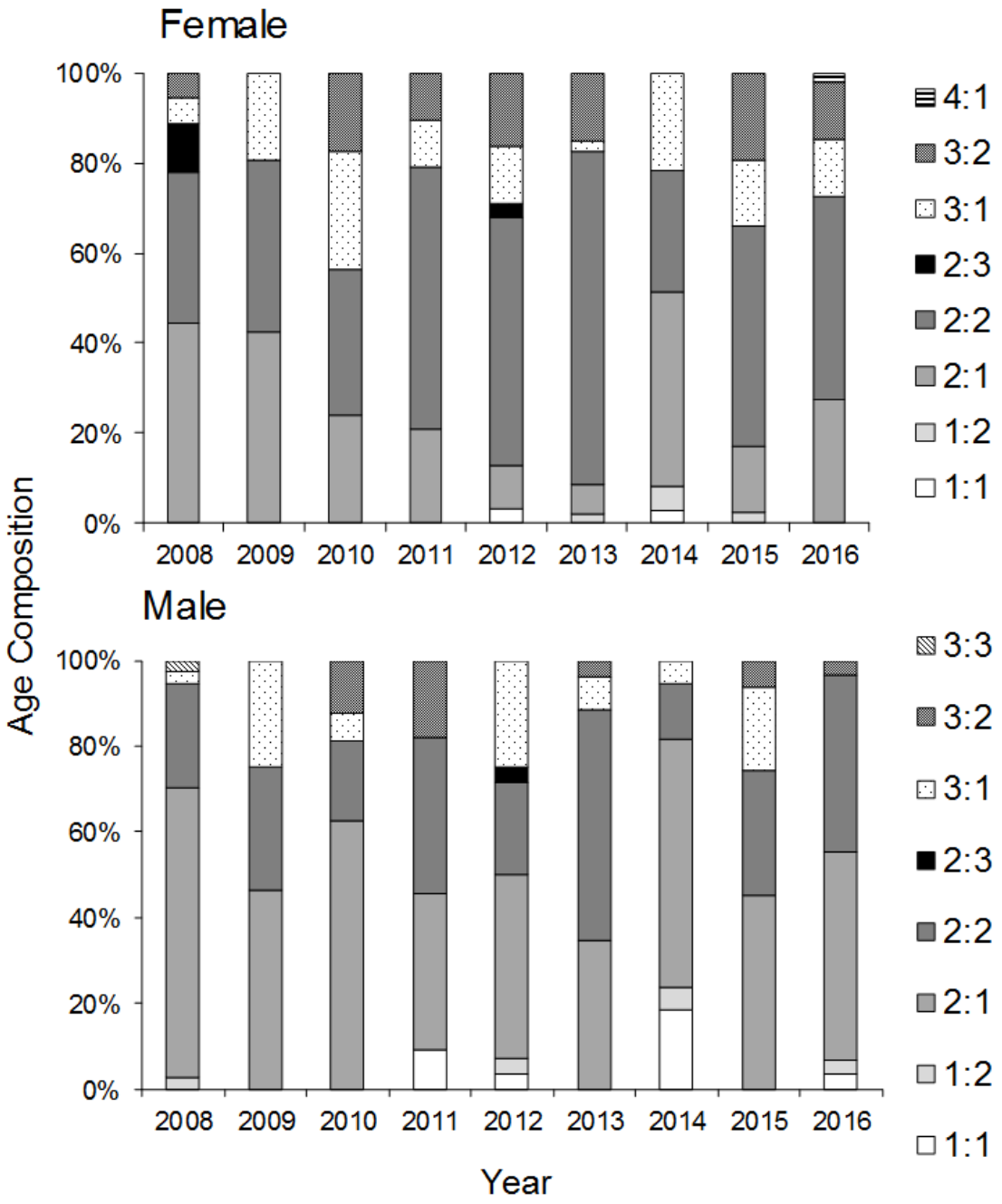

Figure 2.22. Age composition of adult steelhead in the East Fork Potlatch River watershed during 2008-2016. Age designations show years in freshwater separated from years in the ocean by a colon.

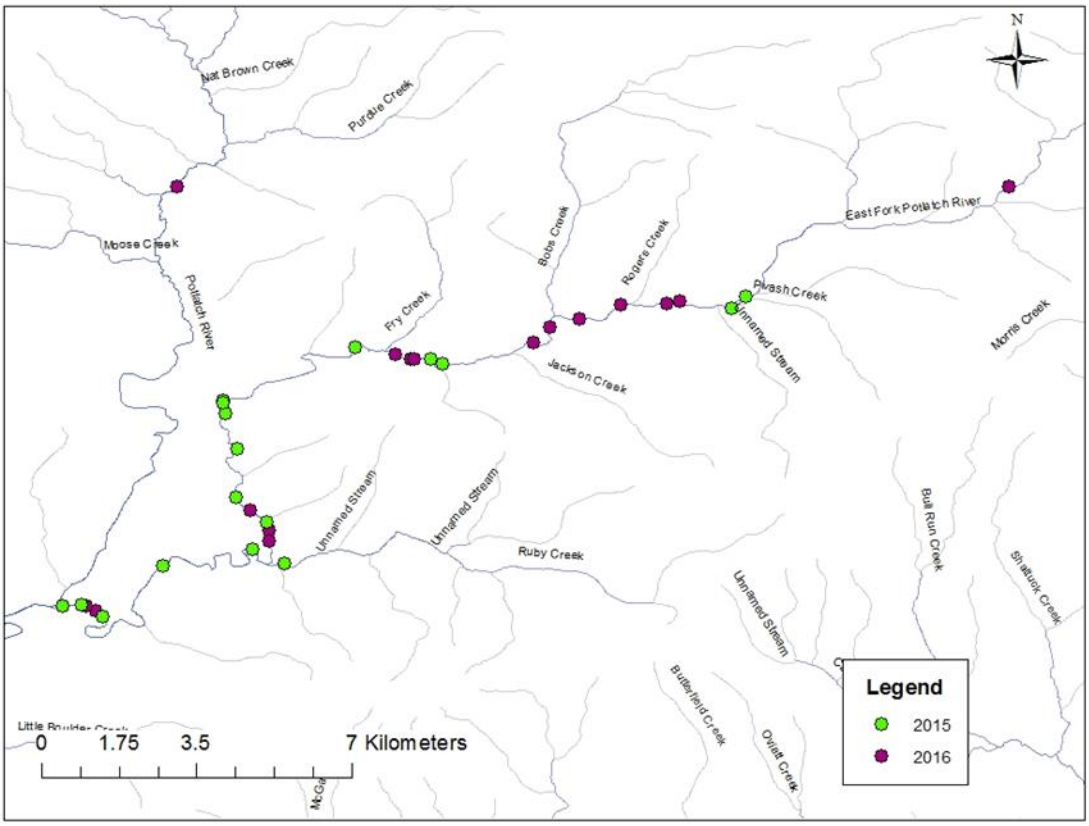

Figure 2.23. Spawning locations of radio-tagged adult steelhead in the East Fork Potlatch River watershed in 2015 and 2016. Points indicate furthest upstream detection recorded during tracking surveys.

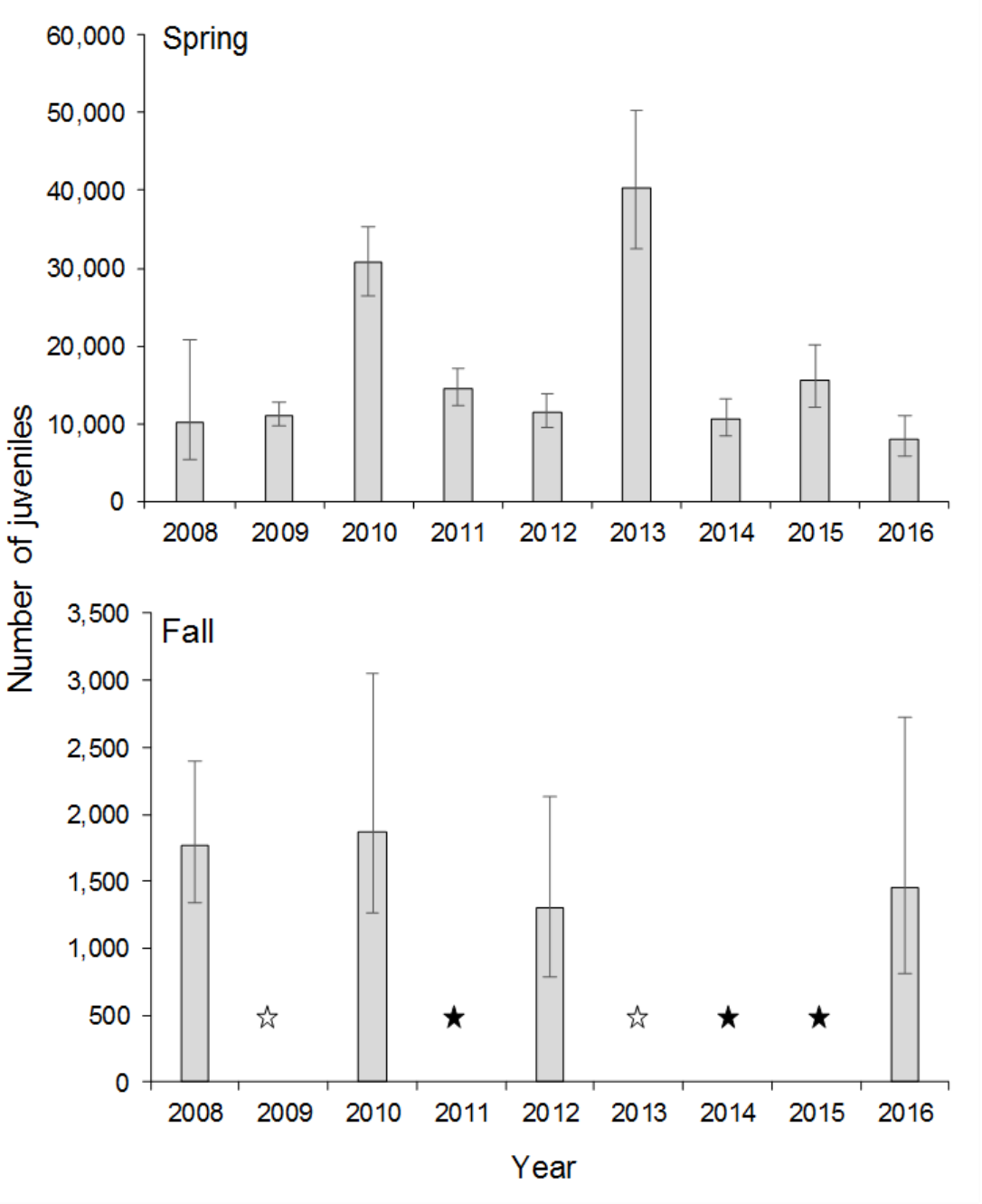

Figure 2.24. Numbers of juvenile steelhead annually emigrating from the East Fork Potlatch River watershed during 2008-2016 with 95% confidence intervals. Spring estimates are in the top panel and fall estimates are in the bottom panel. White stars indicate years when too few juveniles were captured to generate an estimate and black stars indicate years when the trap was not operated.

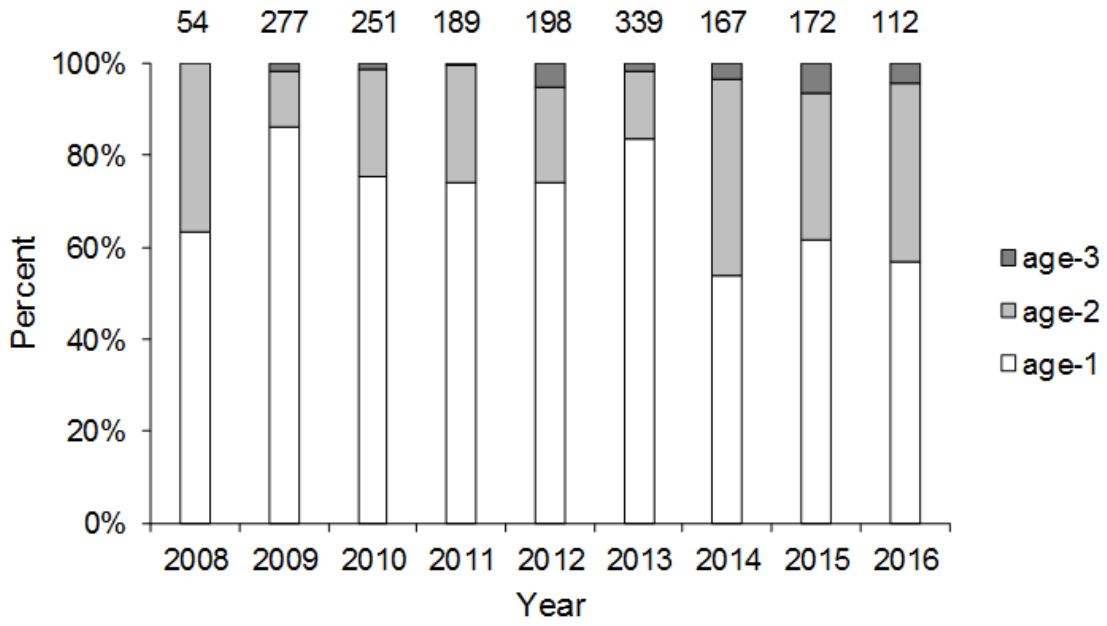

Figure 2.25. Age composition of juvenile steelhead emigrating from the East Fork Potlatch River watershed during 2008-2016. Numbers above bars are fish aged.

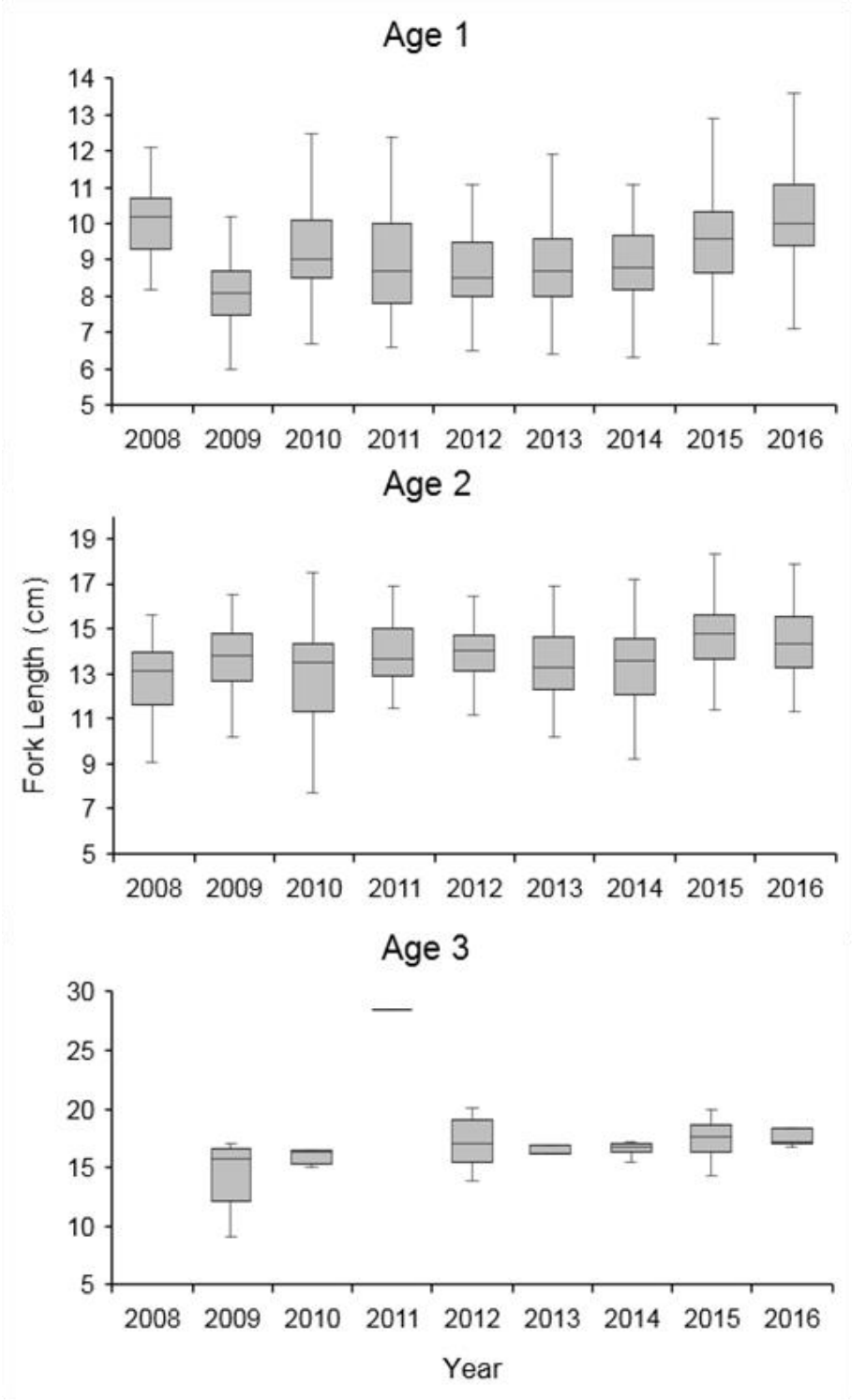

Figure 2.26. Boxplots of length-at-age of juvenile steelhead emigrating from the East Fork Potlatch River watershed during 2008-2016. The bottom and top of each box represents the first and third quartiles. The band inside the box represents the median and the whiskers indicate the minimum and maximum values.

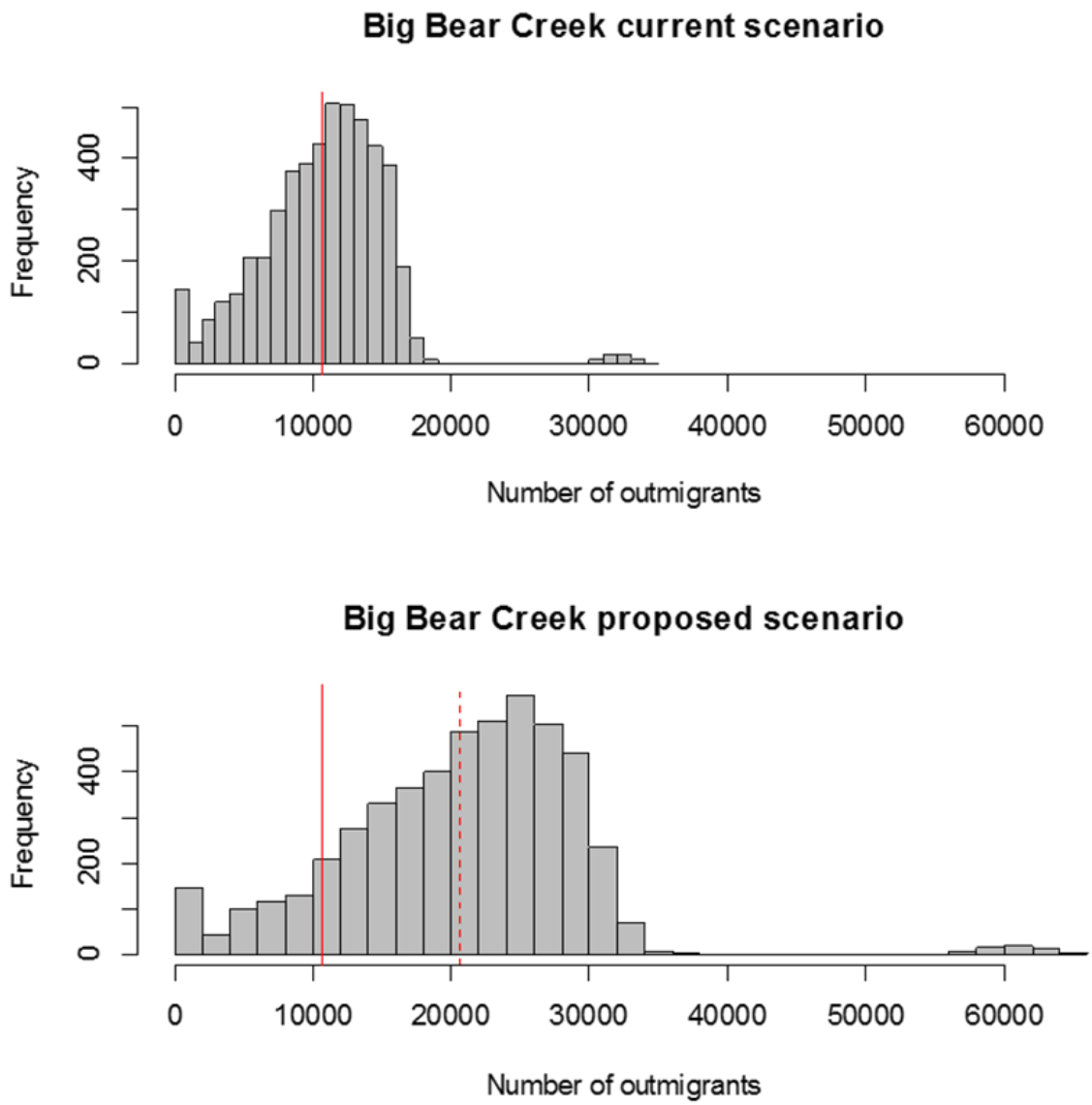

Figure 2.27. Simulated distribution of the number of steelhead emigrants produced from Big Bear Creek watershed under the current scenario (top panel) and under the proposed restoration scenario (bottom panel). The solid vertical red line represents the mean number of out-migrants produced under the current scenario in both panels and the dashed vertical red line represents the mean number of out-migrants produced under the proposed restoration scenario.

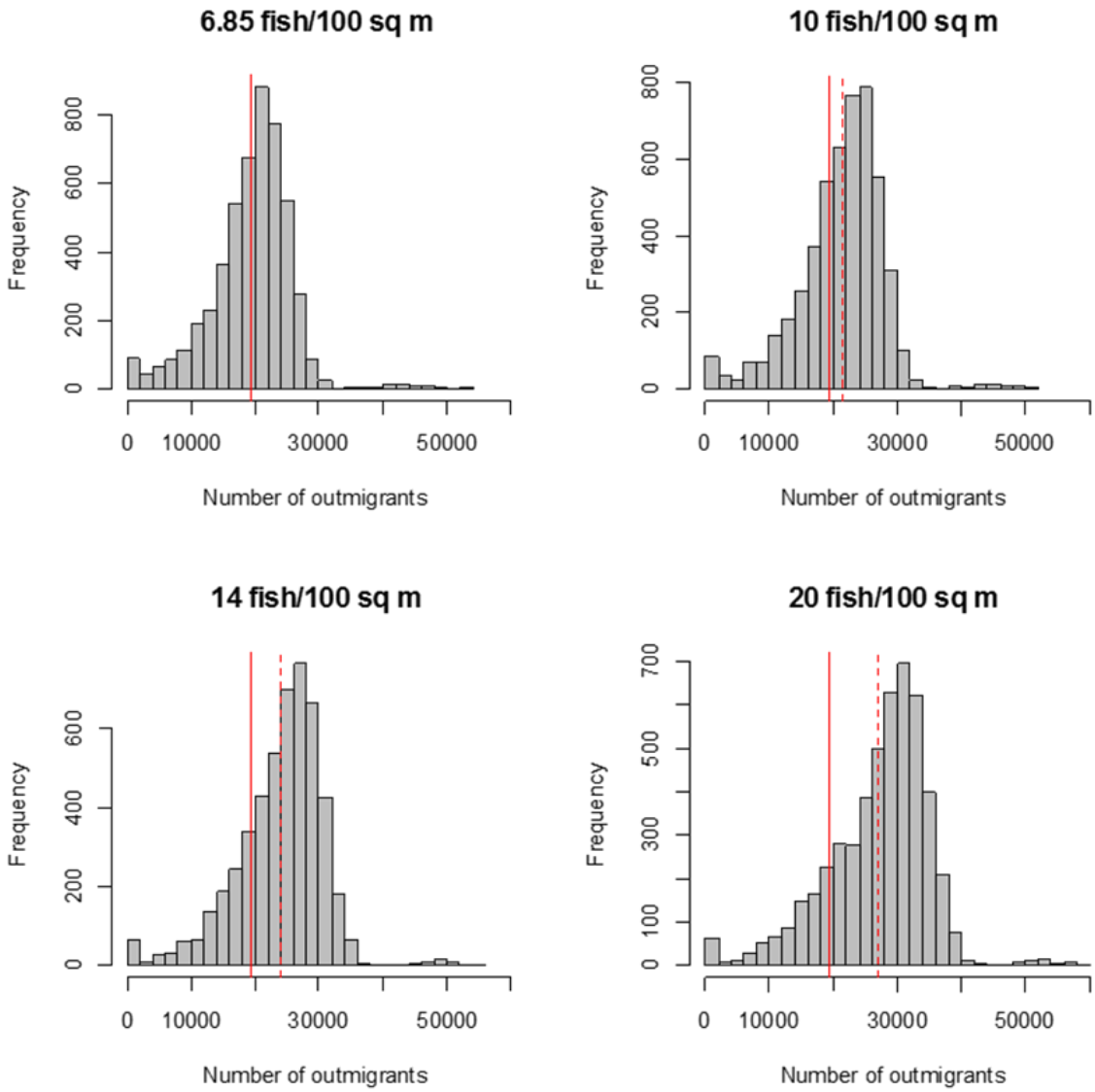

Figure 2.28. Simulated distribution of the number of steelhead emigrants produced in the East Fork Potlatch River with parr densities typical of the current scenario (top left panel) and under three proposed restoration scenarios. The solid vertical red line represents the mean number of out-migrants produced under the current scenario in all panels and the dashed vertical red lines represent the mean number of out-migrants produced under the respective proposed restoration scenarios. For these simulations we assumed that improvement in habitat only affected stock productivity.

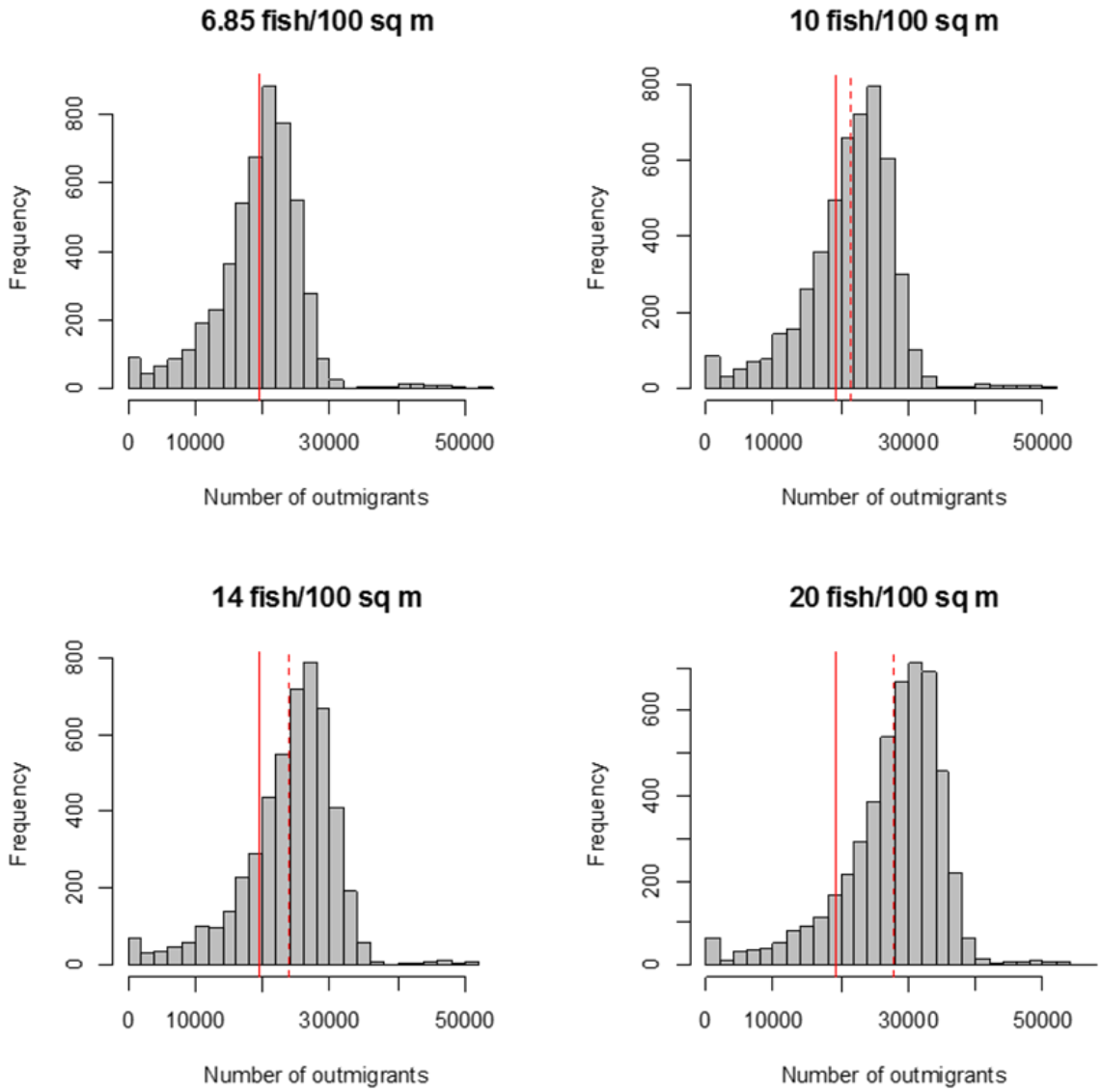

Figure 2.29. Simulated distribution of the number of steelhead out-migrants produced in the East Fork Potlatch River with parr densities typical of the current scenario (top left panel) and under three proposed restoration scenarios. The solid vertical red line represents the mean number of out-migrants produced under the current scenario in all panels and the dashed vertical red lines represent the mean number of out-migrants produced under the respective proposed restoration scenarios. For these simulations we assumed that improvement in habitat only affected carrying capacity.

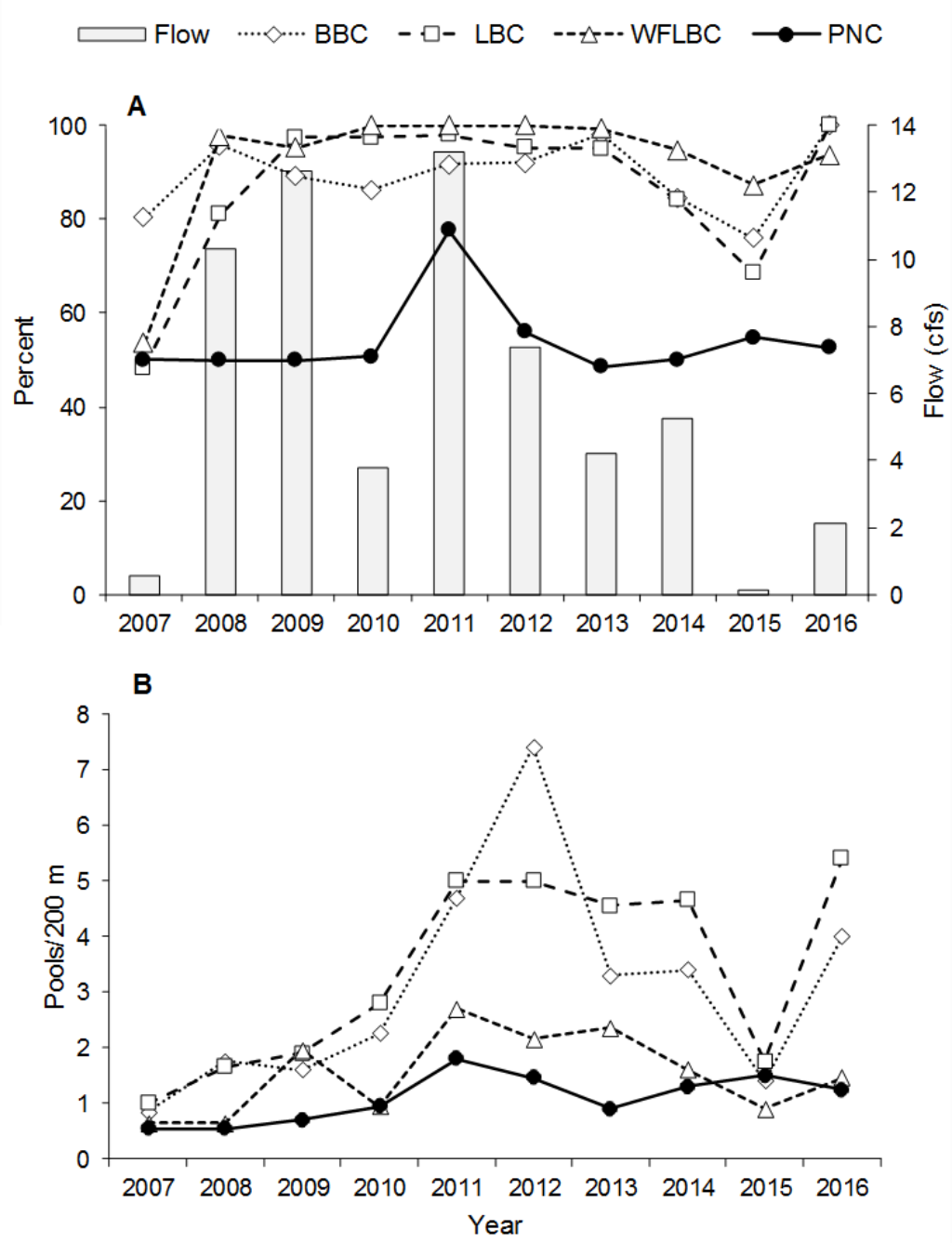

Figure 2.30. The amount of wetted habitat (panel A) and pool density (panel B) in treatment and control tributaries in lower Potlatch River watershed during 2007-2016. The columns in panel A indicate mean flow (cfs) during August for each year at USGS stream gauge located at the mouth of Potlatch River.

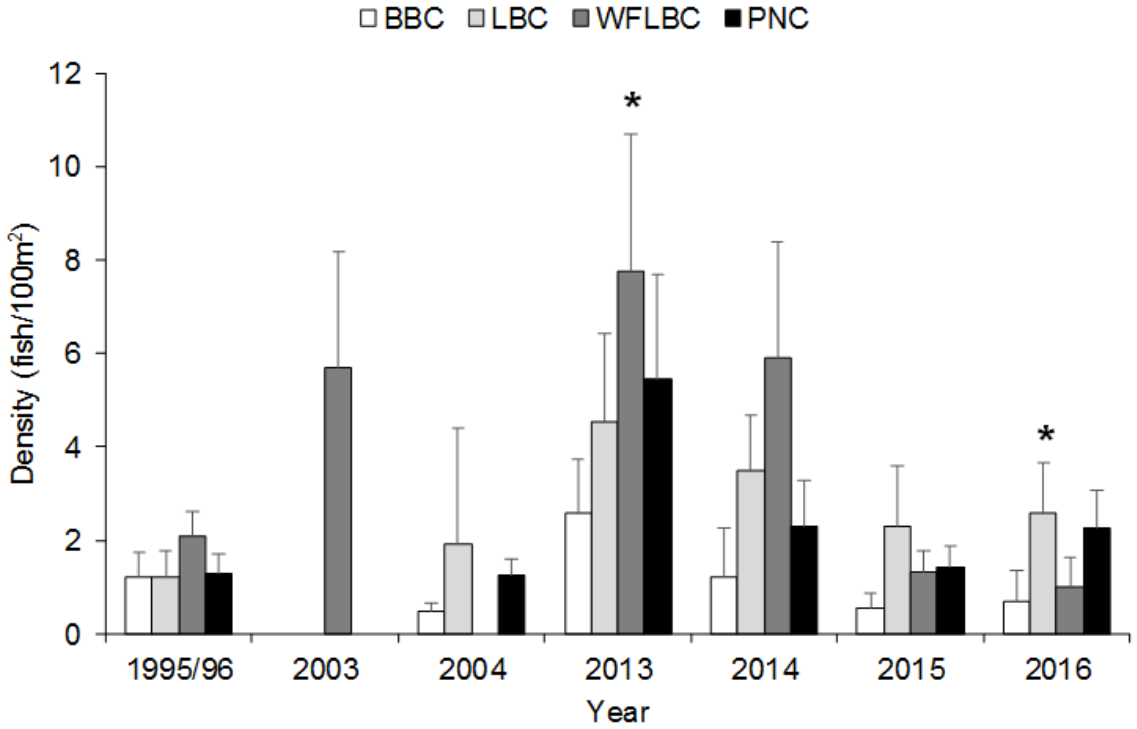

Figure 2.31. Density of juvenile steelhead ≥80 mm in treatment (Big Bear Creek-BBC; Little Bear Creek-LBC; West Fork Little Bear Creek-LBC) and control (Pine Creek-PNC) tributaries in lower Potlatch River watershed during 1995-2016. Asterisks above columns indicate the year when the first restoration treatment was completed in the tributary.

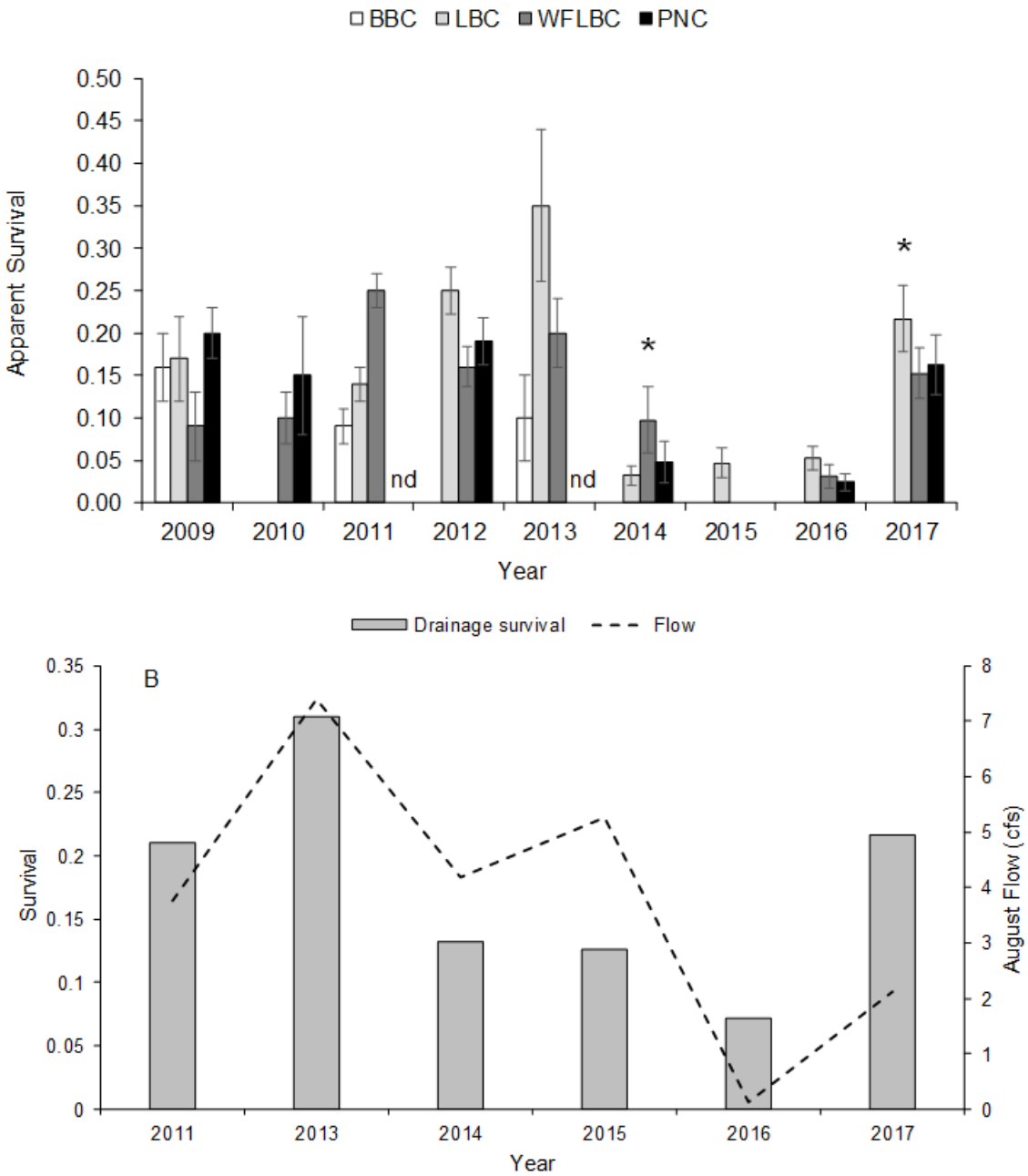

Figure 2.32. Survival of steelhead juveniles from the lower Potlatch watershed during 2008-2015. Panel A shows apparent survival to Lower Granite Dam from treatment (Big Bear Creek-BBC; Little Bear Creek-LBC; West Fork Little Bear Creek-LBC) and control streams (Pine Creek-PNC). No data=nd. Asterisks above columns indicate the year when the first restoration treatment was completed in the stream. Panel B shows survival from tagging to the Big Bear Creek PIT-tag antenna array Dashed line represents mean August flow at the main-stem Potlatch River gauge during each year.

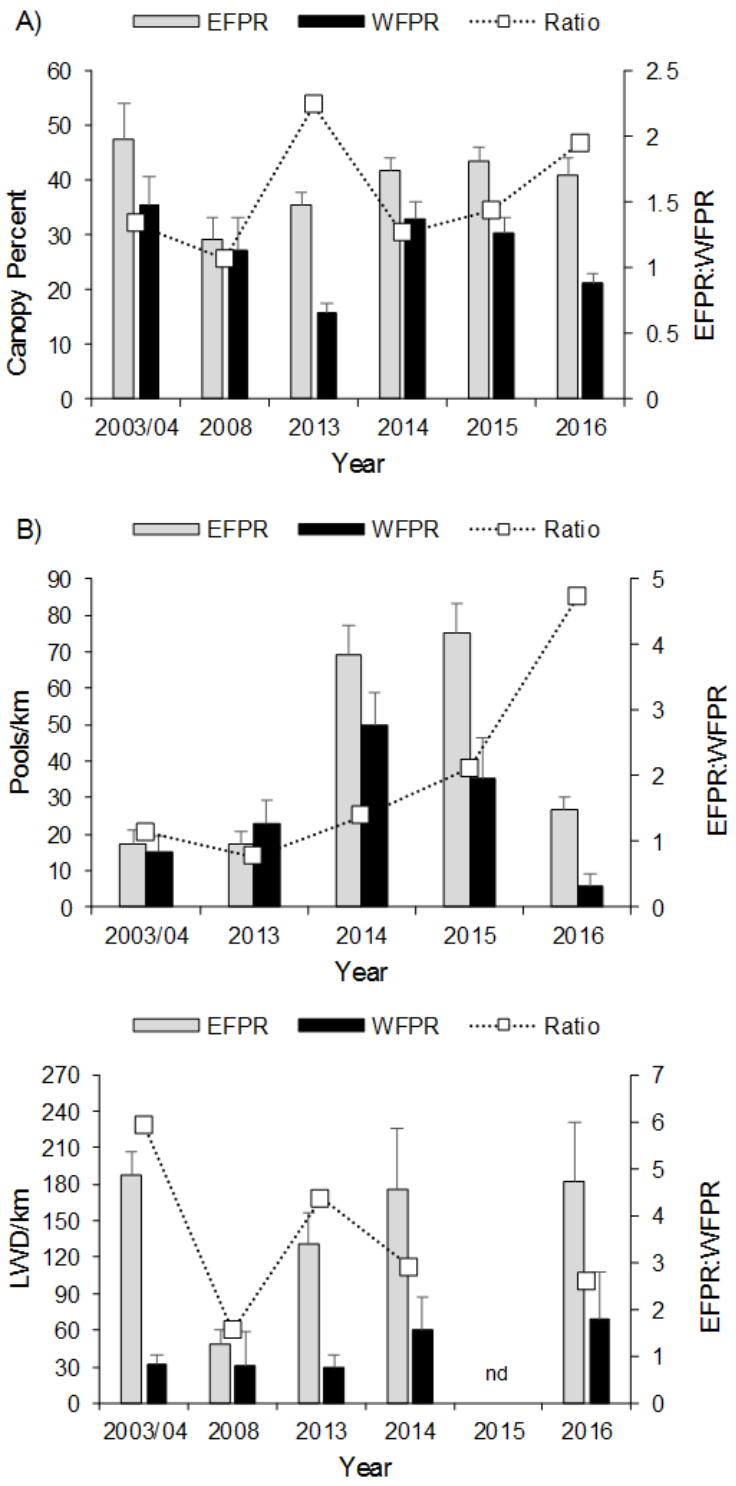

Figure 2.33. Canopy cover (panel A), pool density (panel B), and large woody debris (LWD, panel C) within the treatment (East Fork Potlatch River-EFPR) and control (West Fork Potlatch River-WFPR) streams in the upper Potlatch River watershed during 2003–2016. Dotted line on the secondary axis indicates the EFPR:WFPR ratio value for each habitat metric in each year.

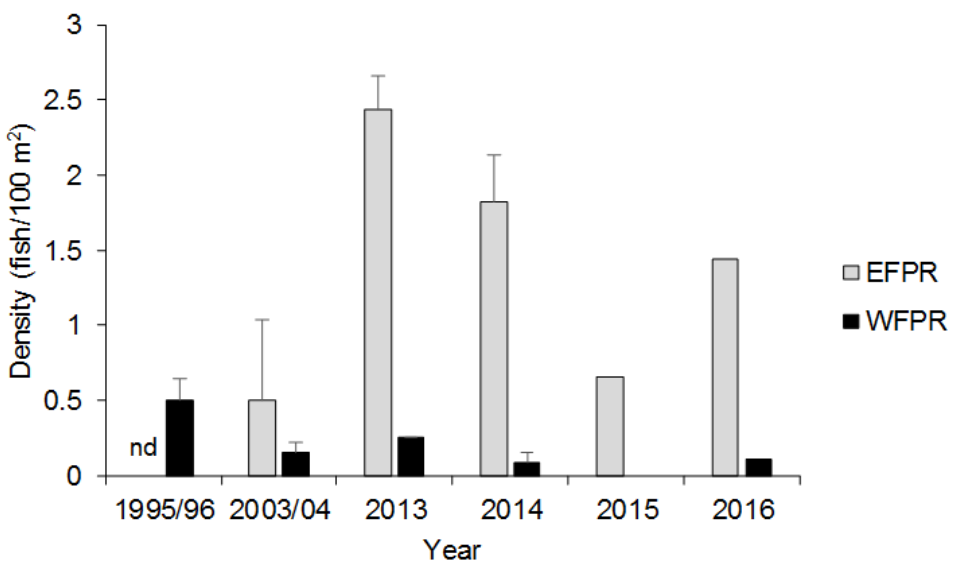

Figure 2.34. Density of juvenile steelhead ≥80 mm based on single-pass electrofishing surveys in treatment (EFPR) and control (WFPR) tributaries in upper Potlatch River watershed during 1995–2016.

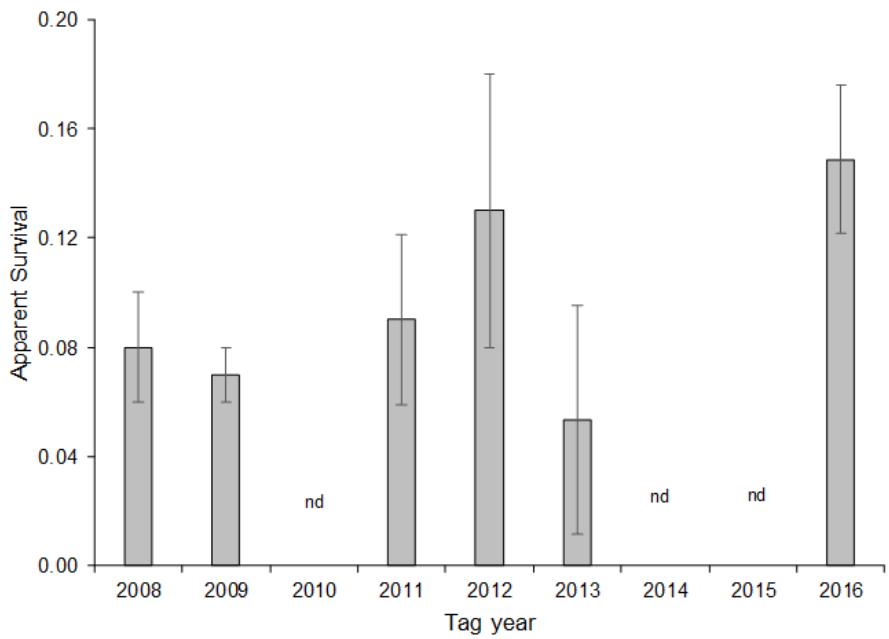

Figure 2.35. Apparent survival to Lower Granite Dam of juvenile steelhead tagged upstream from the East Fork Potlatch River screw trap during 2008-2015. No data = nd.

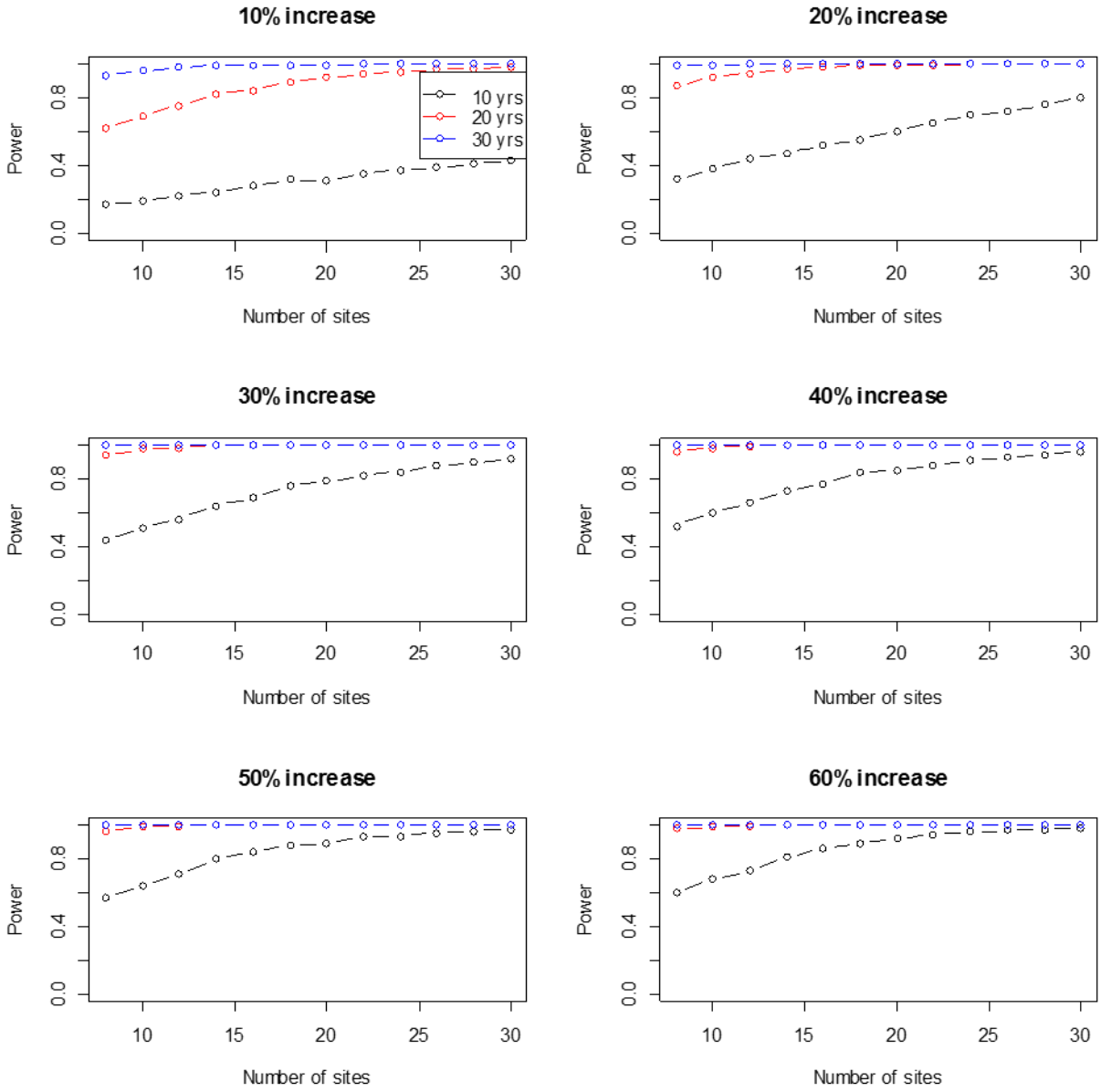

Figure 2.36. Power to reject the null hypothesis given a realized relative rate of change in abundance of steelhead parr in the treatment and control tributaries in the Big Bear Creek watershed over periods of 10, 20, and 30 years. Differences in the relative rate of change between the treatment and control varied from 10% to 60%. The number of sites surveyed per stream in the treatment (three streams) and control (one stream) varied from two to 30.

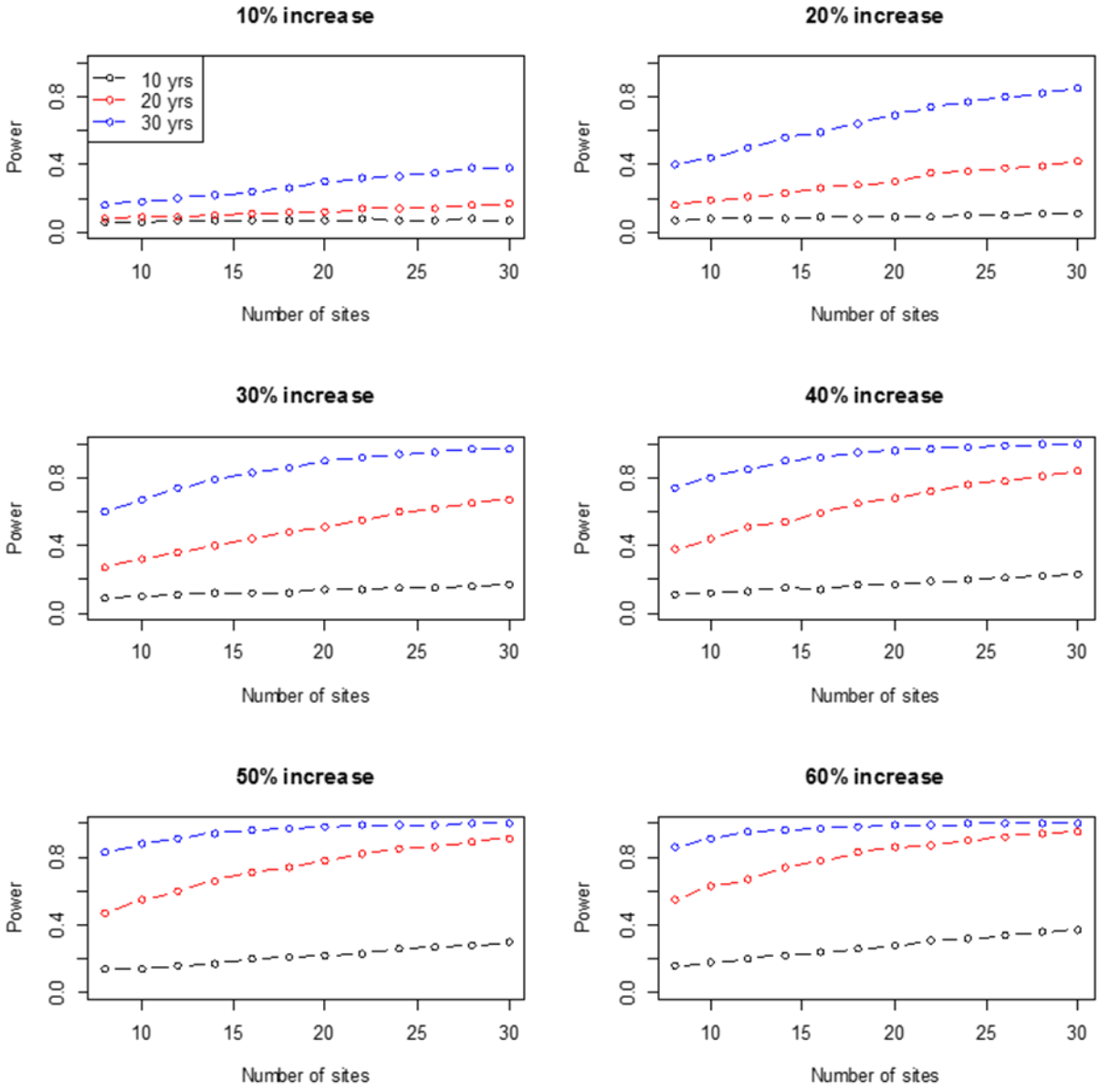

Figure 2.37. Power to reject the null hypotheses given a realized relative rate of change in abundance of steelhead parr in the treatment and control tributary in the East Fork Potlatch River watershed over periods of 10, 20, and 30 years. Differences in the relative rate of change in the treatment and control tributaries varied from 10% to 60% and the number of sites surveyed in the treatment and control varied from 2 to 30.

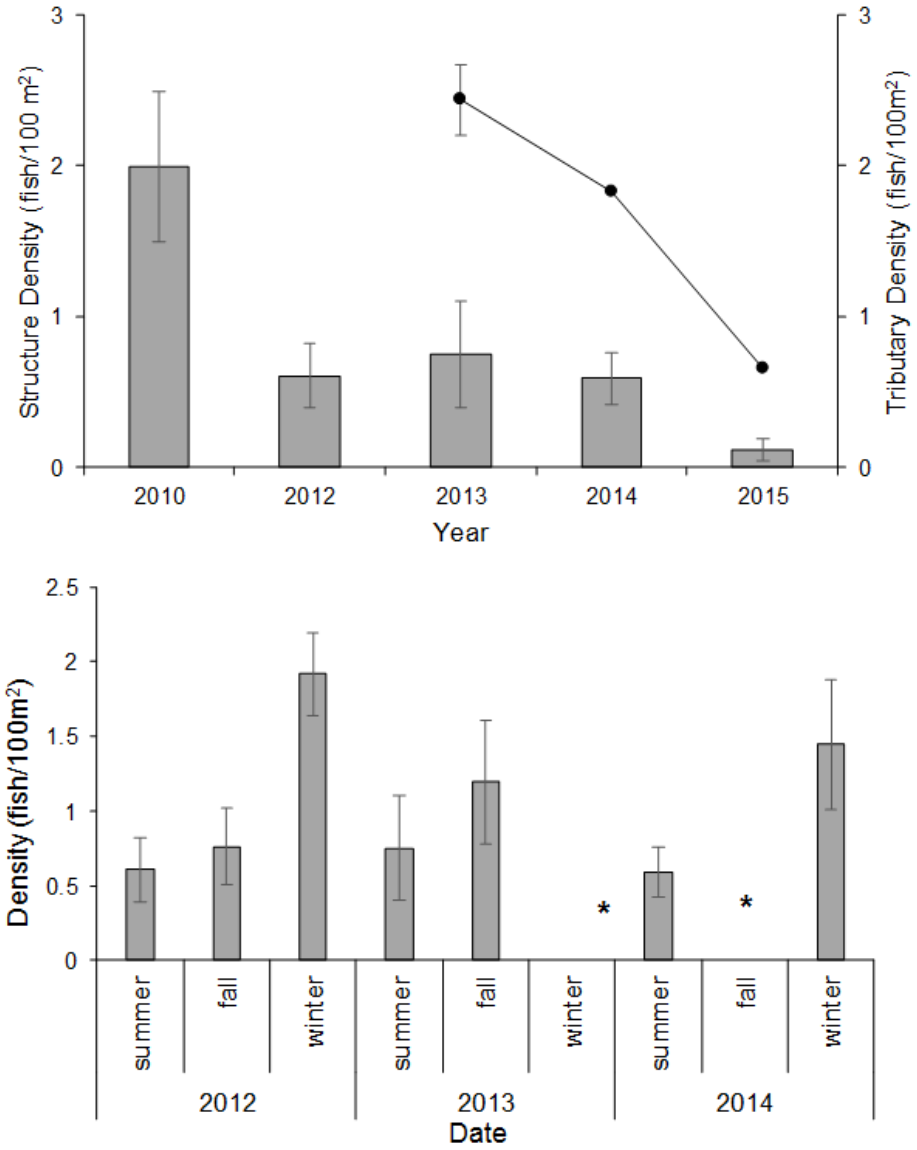

Figure 2.38. Density of juvenile steelhead ≥80 mm at large woody debris structures in the East Fork Potlatch River 2010–2015. In panel A, the solid trend line indicates density throughout the river. In panel B, density is displayed by season. Bars indicate a standard deviation. Asterisks represent no data.

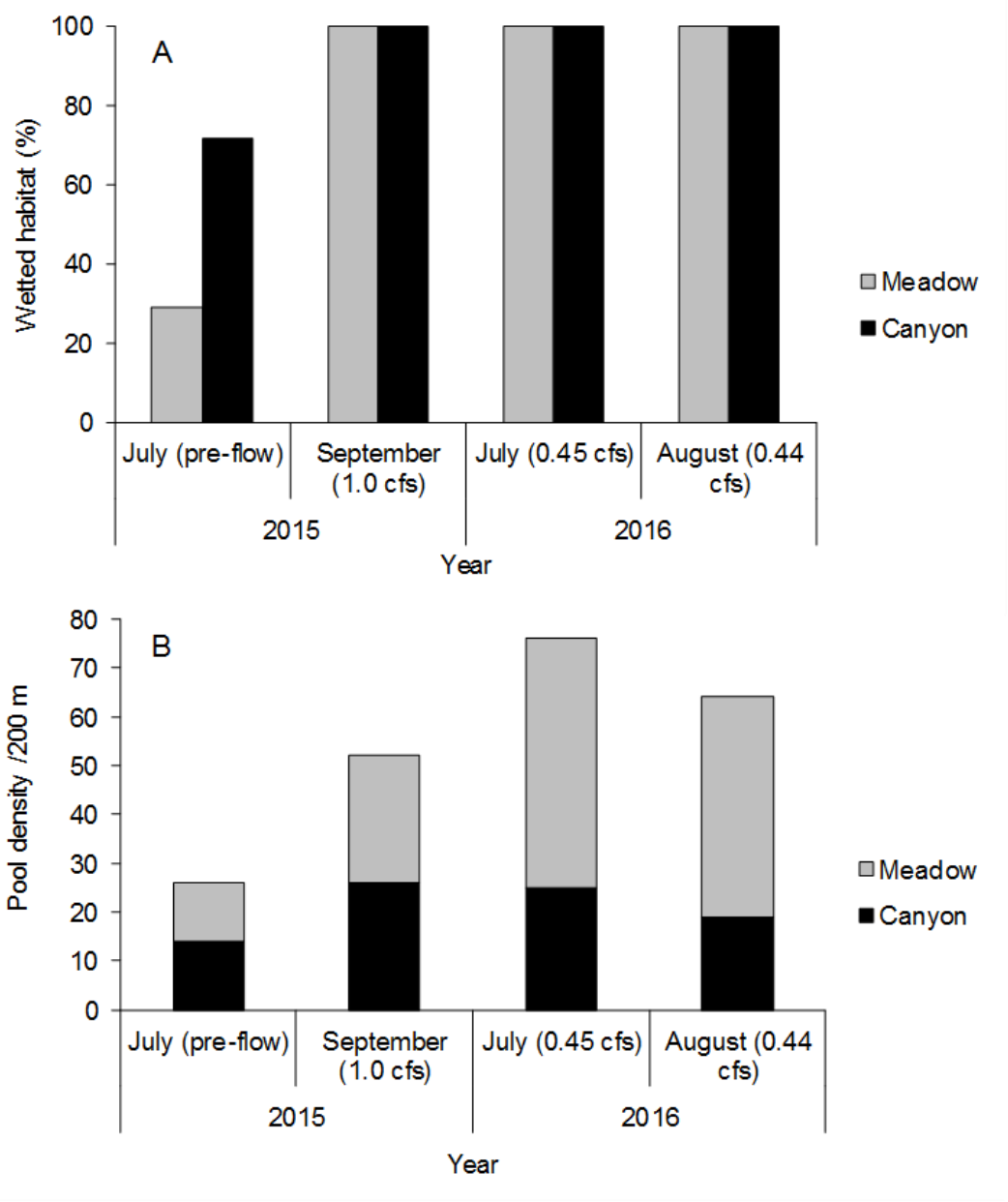

Figure 2.39. Wetted habitat (%, panel A) and pool density (panel B) in the treatment reach downstream of Spring Valley Reservoir prior to and during different flow supplementation strategies in 2015 and 2016. In 2015, the test reach went intermittent prior to flow supplementation. In 2016, flow supplementation maintained perennial flows throughout the study period. The treatment reach was stratified into meadow and canyon sections.

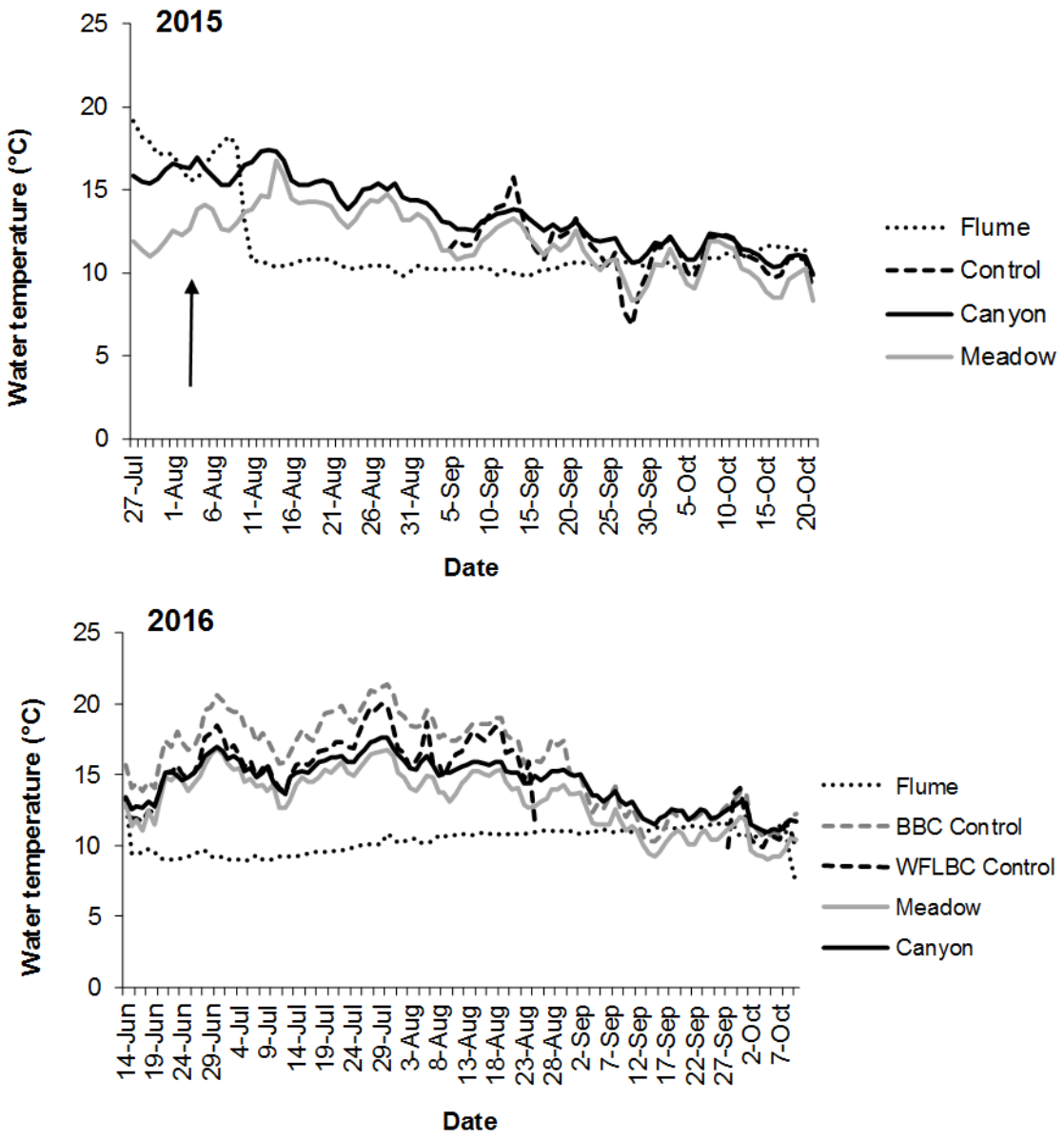

Figure 2.40. Temperature profile of treatment reaches (meadow and canyon) downstream of Spring Valley Reservoir and control reaches (BBC and WFLBC) during flow supplementation in 2015 and 2016. The dotted flume line is the water temperature leaving the reservoir. In the top panel the arrow indicates the date water releases began. In 2015, the control site went dry until early September.

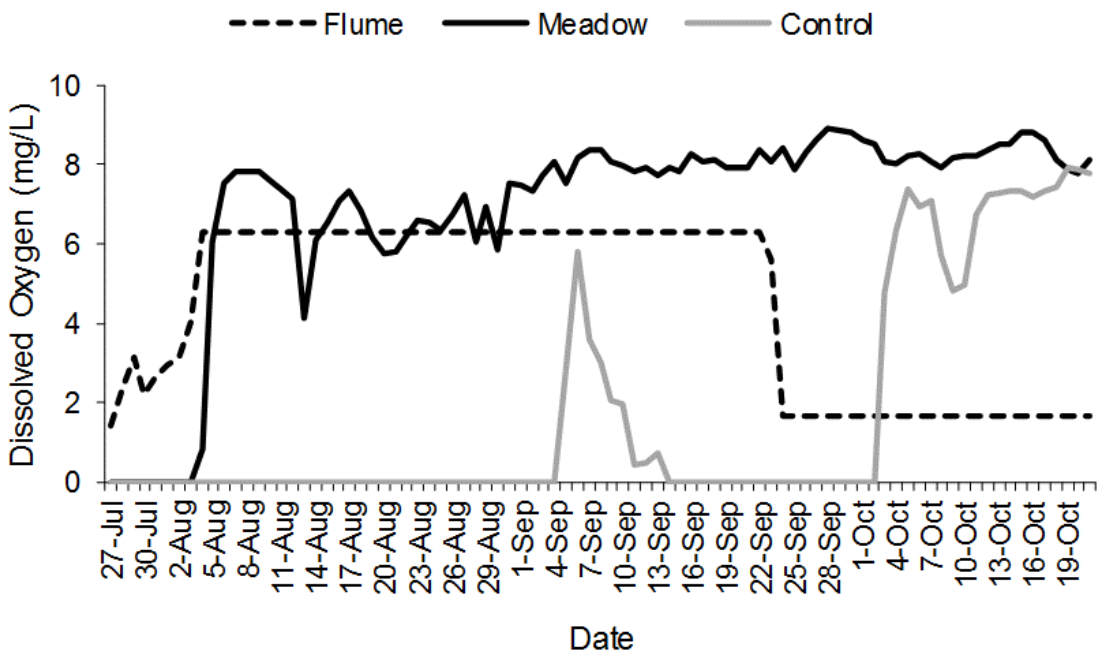

Figure 2.41. Dissolved oxygen profile of treatment (meadow) and control (BBC) reaches downstream of Spring Valley Reservoir during flow supplementation in 2015. The dashed flume line is the dissolved oxygen levels leaving the reservoir. The control site went dry and only recorded dissolved oxygen values after rain events.

SYNTHESIS

This report contains information from two large monitoring projects in disparate parts of Idaho. It is worth considering commonalities and differences in the results and experiences in order to draw lessons to apply to the habitat restoration program in other parts of Idaho and the Pacific Northwest. In this Synthesis section, we compare and contrast the Lemhi and Potlatch IMW projects in terms of their results, inherent challenges, and adaptive management. These observations should be instructive for anyone managing, implementing, or funding a stream habitat restoration program.

Responses to Habitat Manipulations

We documented common responses with regard to three classes of habitat manipulations: actions affecting the configuration of habitat within a stream reach, actions affecting connectivity among reaches, and actions affecting connectivity between the stream channel and the surrounding landscape. These categories roughly mirror the three broad strategies for river restoration proposed by Palmer et al. (2014): i.e., restoration as channel design, restoration for ecological function, and restoration beyond the stream channel.

Responses to Channel Restoration

Channel design is an important consideration in both restoration programs but is especially emphasized in the plan for the East Fork Potlatch River. These actions are intended to establish more natural meander patterns and increase the complexity of instream habitat (e.g., with engineered log structures). The primary goal is to increase the quality of existing habitat. In Idaho, winter conditions can impose a demographic constrain on salmonid populations (Smith and Griffith 1994; Kiefer and Lockhart 1999; Mitro and Zale 2002; Walters et al. 2013); hence, winter habitat is important.

The results from the two IMWs suggest initial responses to in-channel manipulations have taken place. Parr densities increased in the fall and winter at LWD projects in the East Fork Potlatch River. Presumably fish move because previously-occupied habitat was deficient in some respect (Ogston et al. 2015) and movements to LWD projects should eventually increase stream-wide density. We hypothesize that the observed increase in the average age composition of EFPR emigrants may be driven by increased retention through the winter, although there has not yet been a detectable increase in stream wide density. In the Lemhi River, several metrics indicated low winter survival in the lower Lemhi River, which has poor habitat; therefore, a large project is being implemented at Eagle Valley Ranch to address winter survival using meanders, activation of historic channels, creation of multiple side channels, and engineered logjams.

Although one might reasonably expect in-stream structures to have beneficial effects, some evaluations of in-stream structures have had ambiguous results (Stewart et al. 2009; Nilsson et al. 2017) although others that brought large amounts of data to bear did find benefits (Roni et al. 2008; Whiteway et al. 2010). Previous work in Idaho found that trout densities are positively correlated to LWD and other measures of cover, with important fishery effects (Thurow 1990). Further, complex habitat structures increased capacity for age-1 and older steelhead parr in streams with low habitat complexity but quantification was difficult (Rich and Petrosky 1994; Kiefer and Lockhart 1999). The EFPR power analysis corroborates this latter point well; it may take many projects and a long timeframe to discern a detectable numerical response within the study population. Several studies have documented winter survival improvements following addition of LWD (Johnson et al. 2005, Jones et al. 2014, Ogston et al. 2015); hence, other metrics

may be more sensitive to the effects of in-stream structures than abundance. The Eagle Valley Ranch project evaluation in the Lemhi has been designed with this in mind.

Responses to Connectivity Restoration

Actions to enhance longitudinal connectivity are main features of the restoration programs in the Lemhi River and Big Bear Creek. These actions allow greater movement of fish; affect the movement of water, sediment, wood, and nutrients within the stream network (Beechie et al. 2010); and tend to be more focused on ecological processes than channel manipulations (Palmer et al. 2014). Many stream systems in Idaho are typified by low summer flows; thus, increases in flow are accompanied by increases in habitat volume and moderated water temperatures. Therefore, flow enhancements are often planned to allow fish passage, although not all fish barriers addressed by the Potlatch IMW inhibited stream flow.

Manipulations to improve connectivity have induced changes in fish distribution within a short amount of time in both IMWs. In the Lemhi, movements of juveniles into formerly blocked tributaries are clearly documented. Adult steelhead have made limited spawning movements above former barriers in both IMWs (e.g., the Dutch Flat Dam in the Potlatch River basin and the Kenney Creek diversion in the Lemhi River basin). Flow supplementation in the Big Bear Creek watershed resulted in immediate increases in pool habitats, lower temperatures, and increased dissolved oxygen. Further, there are indications of improved juvenile survival from reconnections in the upper Lemhi River, although the exact mechanism is unclear.

Enhancements to connectivity, e.g., removal of passage barriers, may have large effects on fish populations (Roni et al. 2008). Previous evaluations of habitat projects in Idaho found that barrier removals have had the largest effect on increasing anadromous fish production (Rich and Petrosky 1994). The BBC power analysis confirms this potential in the Potlatch River IMW. Potential for increased production of steelhead also exists in the Lemhi River (ISEMP and ChaMP 2017b).

Response to Off-Channel Restoration

The common thread that binds this diverse category of restoration techniques is lateral connectivity of the stream channel to off-channel habitat, the riparian zone, and eventually upland areas. Lateral processes affect the delivery of key habitat elements to the stream channel (water, sediment, wood, and food), whereas longitudinal processes affect how elements move in the channel (chiefly downstream). Activation of the floodplain and creation of lateral habitats (e.g., side channels and nooks) provide important refugia and increased growth opportunity. Many of these habitats may be important seasonally but temporary residence in them may significantly improve fish performance in later life stages. Furthermore, restoration of the riparian zone addresses the long-term health of the entire stream network, allowing improvement of habitat beyond the typical scope of channel-design projects.

Off-channel restoration is an important component for the Lemhi River and East Fork Potlatch River programs. Riparian protection measures in the EFPR have shown positive results already in terms of canopy cover and pool density, although LWD abundance has not yet changed appreciably. These habitat metrics may be greatly increased by beaver colonization and grazing retirements. In the Lemhi River, fish monitoring has just been instituted to address implementation of floodplain activation and lateral habitats. Habitat monitoring has not found significant changes at the scale of the Lemhi River basin yet. Previous evaluations of habitat projects in Idaho found that off-channel developments may yield high densities of fish but opportunities to implement them

may be limited, and that riparian revegetation to reduce fine sediment inputs take years to achieve benefits and are affected by several confounding factors (Rich and Petrosky 1994).

Meeting Inherent Challenges

Many contextual factors can impinge on the ability to evaluate and guide a habitat restoration program. The challenge is to assess effects of multiple projects and important confounding factors in a way that allows effects by action type and reach to be separated from the cumulative watershed response (Roni et al. 2015). It is easy to see how real-world conditions complicate accomplishing and evaluating stream habitat restoration.

Both Idaho IMWs are larger than other IMWs in the Pacific Northwest (see Bennett et al. 2016). They are at the scale at which IDFG fisheries management programs operate (see IDFG 2013); hence, experiences should be directly applicable. However, there is a tradeoff between size of the watershed and the tractability of technical problems. It is harder to manipulate the large areas or the long reaches needed to see a detectable response at larger scales. Finite monitoring resources get stretched thin such that response estimates may become less precise. More potential confounding factors come into play (e.g., different hydrological regimes in the Potlatch River basin). Even at basin scales, the full scope of freshwater rearing of anadromous salmonids in Idaho may not be completely covered (Copeland et al. 2014).

Changes on the landscape prior to and during the IMW projects must be properly accounted for. In the Potlatch River, retirement of grazing rights and beaver colonization were not planned restoration actions, but will have effects on the habitat and focal species. In the Lemhi River, there was a history of actions taken to conserve fish (e.g., the installation of screens at irrigation diversions), potentially causing the baseline period to be non-stationary. Both foregoing examples will likely lead to positive responses by fish populations but other factors such as increasing urbanization in both watersheds will likely not. All potential factors must be considered in the evaluation, even if not incorporated into the formal analytical framework.

Social factors within the community in which restoration programs are implemented also have large effects on success. In Idaho, private landowners control many important stream reaches, as is the case in both IMWs. It takes time to build landowner relationships and support for restoration projects, even if funding and resources are available. Project implementation almost always lags behind expectations because of factors such as weather, permitting processes, etc. When resources are limited, priorities must be assigned and such consensus is not always easy to achieve. The Lemhi River basin has a history of restoration with a functional group to guide it, i.e., the social process is mature. In contrast, the restoration community in the Potlatch is more recent, initial restoration efforts were opportunistic, and thus continued efforts are needed to prioritize implementation in the index watersheds. Hence, social factors may slow or prevent implementation of restoration needed to elicit a detectable response. These procedural issues can derail even the best-designed monitoring project (Roni et al. 2015).

Effects of social factors underscore the importance of an objective document to guide restoration planning and establish priorities. Ideally, this document would address potential limiting factors in a unified framework to address the populations of concern. There needs to be a clearly articulated goal and statements of restoration objectives relative to limiting factors with criteria for success (Palmer et al. 2005). These considerations are vital to efficient and effective restoration projects and to designing an effective monitoring program to evaluate the restoration program. With careful thought to the criteria for success and key uncertainties, the monitoring program provides information to manage the portfolio of restoration projects within the local

program and allows coordination among project implementers. With clear priorities, restoration program managers can use monitoring information to demonstrate relevance to landowners and develop credibility with them (e.g., Pierce et al. 2013). Experience in both IMWs shows that building such relationships can take time and relationships change at different rates (for good or ill). Clear priorities allow program implementers to strike a balance between strategy and opportunity in a fluctuating social environment. Such flexibility can be important to long-term success.

Expectations for Responses to Habitat Restoration by Fish Populations

Expectations for habitat restoration programs are important to their implementation and management. Responses to habitat manipulations are seldom instantaneous; hence, it is important to identify realistic time frames for the expected responses to ensure reasonable expectations of the restoration program and associated monitoring by implementers, funders, and managers. We follow with observations on response timeframes and magnitude of necessary manipulations.

Timeframe may be in reference to the effect restoration has on stream habitat or to the sensitivity of the response parameter. Certain actions can have detectable effects in a relatively short time, such as barrier removals or increases in flow may increase amount of accessible habitat almost upon implementation. Likewise, fish move, grow, and survive over periods less than a year. Conversely, riparian restoration may take decades to yield trees with large enough limbs to form pools when they fall into a stream channel. Full occupancy of newly-accessible habitat may take generations of fish. This is not a comprehensive list but makes the point that the ultimate desired responses take time and evaluation should consider leading indicators that can show progress towards the ultimate response. However, implementers and managers should also remember that shorter-term solutions will need to be maintained until longer-term processes provide broader and more permanent benefits (e.g., LWD placements versus a mature riparian zone).

The magnitude of planned restoration projects may be insufficient to elicit the desired fish response or the response may be affected by natural variability such that it is difficult to detect. Most efforts at habitat restoration are at site or reach scales, while effects of past and current disturbances are often at larger scales (Bond and Lake 2003; Bernhardt and Palmer 2011). Restoration portfolios are often composed of small, incremental projects that are easy (but not necessarily cheap) to implement and therefore may not address critical needs effectively (Kondolf et al. 2008). On the other hand, large projects may be more likely to be successful than smaller ones (Lake et al. 2007). For example, Hood (2007) showed that more benefit should be derived from restoring a single 100-ha channel in the Skagit River estuary than 10 1-ha channels. Roni et al. (2010) found 20% or more of the riparian and in-channel habitat in a watershed needs to be restored in order to achieve a 25% increase in smolts from Puget Sound watersheds, but that increase would be difficult to detect because of inherent variance. Shape of the response (step vs linear vs non-linear) is also important to detecting it within a reasonable time (Lake et al. 2007); a sharp step increase to a stable state by a response parameter would be ideal but seldom happens.

Desired restoration effects are often expressed as an increase in the number of adult fish. Numerical responses by fish populations usually take time to be achieved, are influenced by a plethora of factors, often vary annually, and are thus difficult to detect. Louhi et al. (2016) found few effects of restoration on trout densities when comparing data from three years before and after restoration but such effects were detectable after monitoring for 12 years post-treatment.

Low densities of fish in reference reaches make detection of effects difficult (O'Neal et al. 2016), as does lack of spawning adults (Johnson et al. 2005). Both IMWs have these problems. Productivity is also an important population response but is measured over generational scales. For example, it will take at least five years for the Potlatch River IMW to get two complete spawner-to-juvenile-emigrant cycles, which is insufficient for estimation of post-treatment effects, even assuming restoration is immediately effective.

Restoration programs often focus on reconnecting the upper parts of watersheds to main stems but full use of these habitats often takes time. Location of a source population for newly-accessible habitat is important because distance and ease of movement affect dispersal (Bond and Lake 2003). Philopatric fishes such as salmon may take longer than expected to use new habitat. .Demonstration of movement shows the scope for potential benefit but it will take time for the fish to fully occupy newly available habitat. However, headwater production areas are often important to many high profile trout fisheries (e.g., Silver Creek Rainbow Trout [Thurow 1978]; St Joe River Westslope Cutthroat Trout [Thurow and Bjornn 1978]; Blackfoot River Yellowstone Cutthroat Trout [Thurow 1981]; Middle Fork Salmon River Westslope Cutthroat Trout [Mallet 2013), and hence restoration of these areas is important to fisheries managers.

Typically, it takes time for a restoration program to get projects funded and implemented. As described in the previous section, social factors often slow or inhibit progress towards restoration objectives. Furthermore, large projects are often needed to address root problems (Bernhardt and Palmer 2011) but are harder to propose, get funded, and implement. For example, the Big Bear Falls fish passage project has an initial design but funding for implementation still not identified 4 years later, despite its likely benefit. To effectively set priorities for a restoration program, regular evaluation of the costs and benefits of potential projects is necessary. Restoration evaluations typically focus on cumulative effects, which take time to assess and feed back into program prioritization (Kondolf et al. 2008). All of these factors point towards the need to adaptively manage a restoration program.

Adaptive Management

Adaptive management is a key part of any restoration program because no plan survives unaltered after contact with reality. Monitoring is typically less expensive than restoration actions (Neville et al. 2016) and should produce timely information to guide restoration in an efficient manner. For example, both IMW projects show how monitoring information can highlight needs not anticipated, thus focus restoration projects towards program goals more efficiently. As discussed above, clear guidance documents and transparency can foster effective working relationship and make adjustments such as these easier to implement.

The monitoring programs themselves also require adaptive management. Both projects have had to reconsider study designs necessary to evaluate restoration objectives. This experience highlights the need for review, feedback, and adaptation in monitoring design (Lovett et al. 2007). During design or re-assessment periods, one should consider metrics that are likely to be sensitive to the restoration measures and that will help explain changes in the desired response variable (e.g., as growth and survival are linked to abundance). Such a review can catch errors or notice trends that may not have been apparent at the outset. Thus, strong monitoring designs can accommodate change while protecting the quality and consistency of the data (Lovett et al. 2007). In the case of a habitat restoration program, these measures also allow reach-level effects to be scaled up to the management level, providing relevant evaluation and guidance. A hierarchical monitoring structure is especially helpful in this regard (e.g., monitoring at key juvenile stages will help understand response in adult abundance).

CONCLUSION

Experience from 10 years of monitoring in the Lemhi and Potlatch watersheds has imparted valuable lessons for habitat restoration in Idaho. These two case studies cover many of the likely categories of limiting habitat factors in Idaho. The focal species range from obligate anadromous (Chinook Salmon) to facultative anadromous (steelhead) to potamodromous (Bull Trout). Anadromous and fluvial salmonids are the foci of high-profile fisheries in Idaho, so results are important to fisheries managers. One might also assume that what is good for steelhead should be good for Idaho's fluvial trout in general (Rainbow Trout, Cutthroat Trout, and Bull Trout); that is, good growth, survival and access to main-stem habitats where they can get bigger.

The distinguishing characteristic of these projects is the goal to evaluate response to habitat actions in terms of causation rather than to measure cumulative effects only. Essentially, this goal allows determination of why certain actions work or do not work. With this deeper understanding, results can be applied more generally to shape restoration to suit other scenarios, setting the scope for expectations of restoration effectiveness. We offer five pieces of advice to fisheries managers regarding restoration programs. First, think about the processes that create habitat and how they apply to scenario of interest in short and long terms. Second, how restoration strategies should elicit meaningful change in fish population dynamics. These two form an initial restoration plan. Third, have explicit and realistic expectations; communicate these to all stakeholders with a transparent guiding document. Fourth, collect appropriate information to guide and evaluate the restoration program, including leading indicators with regard to fish movement, growth, and survival (e.g., age composition). Fifth, re-visit the plan on a periodic basis to adaptively manage the program, including the evaluation portion. These lessons learned will help Idaho's habitat restoration program to be more efficient and strategic, building credibility with landowners and cooperating agencies.

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APPENDICES

Appendix A. Lemhi River tributary characteristics and restoration projects inventory

Big Timber Creek

Big Timber Creek is one of the largest tributaries of the Lemhi River, encompassing approximately 224 km^2 of federal, state, and private lands. Although no quantitative evidence exists documenting spawning by Chinook Salmon and steelhead, the basin is thought to be a historically important spawning area for both species (Murphy and Horsmon 2004). Prior to irrigation impacts in the drainage, Big Timber Creek had an average base flow of 45 cfs (BLM 1998). The decreed water rights in the drainage claimed 102.54 cfs across 34 water rights, which resulted in dewatering of the lower portion of Big Timber Creek during irrigation season (BLM 1998). In addition to dewatering and passage issues from diversion dams in the lower reaches of Big Timber Creek, the largest diversion on the stream, the Carey Act Dam (BTC-12), blocked access to the highest quality spawning areas above the canyon (BLM 1998).

Baseline fish surveys, prior to the bulk of restoration actions, documented the presence of Mountain Whitefish, steelhead, Westslope Cutthroat Trout, and Bull Trout (Murphy and Horsmon 2004). It was unknown if the Bull Trout and Westslope Trout encountered in surveys exhibited fluvial or resident life history strategies. A weir was operated about 1.8 km upstream from the mouth during spring 2012 and 2013; a single fluvial Rainbow Trout was captured in each year (Messner et al., in review).

Considerable progress has been made towards reconnecting Big Timber Creek, but it is not currently considered functionally reconnected. Although a year-round surface water connection with the Lemhi River now exists, seasonal passage barriers still persist in the form of irrigation diversions. The following restoration actions have occurred:

1) Big Timber Creek Culvert Replacement (OSC 038 05 SA) – Removed the culvert at the junction of Hwy 28 and Big Timber Creek that served as a seasonal passage barrier. An opened bottom box culvert was installed to provide fish passage during all seasons. Project completed in 2009.

2) L63/Big Timber Creek Siphon (OSC 035 05 SA) – A siphon was installed to remove the old L63 ditch intercept that crossed the Big Timber Creek channel and created impassible barrier during the irrigation season. Project completed in 2010.

3) Big Timber Reconnect and Upper Lemhi Flow Improvement (OSC 007 06 SA) – A seasonally dewatered reach, which blocked fish passage, was eliminated by moving the point of diversion. As a result of moving the point of diversion, 4.5 cfs was spilled through the previously dewater reach. Project completed in 2010.

4) Improve Flow in Big Timber Creek 1.4 (OSC 005 07 SA) – A point of diversion was moved to a pump station on the Lemhi River, which provided an additional 1.4 cfs of flow to lower Big Timber Creek, bringing the total flow to approximately 6 cfs. Project completed in 2011.

Canyon Creek

Canyon Creek drains approximately 151 km^2 of primarily public (BLM, Forest Service, and State) lands with only 2.5% of the watershed in private ownership (BLM 1988). Typical historical flows in Canyon Creek ranged from 9 cfs to 26 cfs. The eight water rights in Canyon Creek have a cumulative permitted allocation of 13.52 cfs (BLM 1998). As with Big Timber Creek, this tributary

was thought to have been an important spawning area for Chinook Salmon and steelhead in the upper Lemhi River basin. Prior to restoration efforts, the species assemblage was limited to steelhead, Westslope Cutthroat Trout, Bull Trout, and Brook Trout (BLM 1988). A weir was operated about 1.0 km upstream from the mouth during spring 2012 and 2013; two fluvial Rainbow Trout and four Brook Trout were captured in 2012 and no fish were captured in 2013 (Messner et al., in review).

Canyon Creek was considered functionally connected following the flow enhancement project completed in 2011. The complete suite of restoration action completed to date include:

1) Big Timber Reconnection and Upper Lemhi Flow Improvement (OSC 007 06 SA) – Eliminated a diversion blocking fish passage to Canyon Creek. Project completed in 2007.

2) Canyon Creek Riparian Restoration (OSC 003 07 SA) – Streambanks were modified to re-establish floodplain connection in the lower 0.63 miles of Canyon Creek. Willows were planted along this section of stream to improve riparian condition and fencing was installed to exclude cattle. Project completed in 2011.

3) Canyon Creek Reconnect (OSC 003 08 SA) – This project occurred concurrently with Big Timber Reconnect because the actions to increase flow in the two tributaries resulted in a common pump station on the Lemhi River. The combined actions with project 005 07 SA resulted in a cumulative water savings of 4 cfs, which provided base flows of 5-7 cfs in the lower reaches of Canyon Creek. This established a year round surface connection with the Lemhi River. Project completed in 2011.

4) Canyon Creek Access and Flow Enhancement CC-03 (OSC 018 13 SA) – Replaced an open ditch irrigation system with 0.5 miles of pipeline resulting in a water savings of 4 cfs. The conserved water remains in the lower 3 miles of Canyon Creek until it enters the Lemhi River. A concurrent project will remove the old diversion, which was a seasonal passage barrier, and replace with a passable structure and a new fish screen. Project is currently in the process of implementation.

Hawley Creek

Hawley Creek is a tributary of Eighteenmile Creek, which forms the Lemhi River at its confluence with Texas Creek. Hawley Creek flows into Eighteenmile Creek approximately 5 km upstream of the formation of the Lemhi River and drains 168 km^2. Two diversions in the lower reaches that deliver senior water rights used to dewater the lower 9.7 km of the stream (Warren and Bliss 2006). The LHaC-03 diversion (1883 priority date) at the mouth of Hawley Creek Canyon diverted the entirety of Hawley Creek and returned some water directly upstream of LHaC-02 (1879 priority date), which is 3.2km upstream from the Eighteenmile Creek confluence (Warren and Bliss 2006). This diversion then diverted all of the flow, dewatering the Hawley Creek channel downstream. The irrigation practices caused Hawley Creek to dewater at least seasonally, for over a century, which resulted in channel incision, poor riparian cover, and sedimentation issues. In 2005, water was returned to the historic channel during the non-irrigation season when LHaC-03 was modified to remove an earthen dam (Warren and Bliss 2006).

As with the other upper Lemhi River tributaries, Hawley Creek is thought to have supported extensive Chinook salmon and steelhead spawning activity (BLM 1998). Baseline fish surveys conducted by the IDFG Screen Program in 2005 documented the presence of steelhead, Bull Trout, and Westslope Cutthroat Trout (Warren and Bliss 2006).

A lot of restoration work has occurred in Hawley Creek in recent years and more projects are planned for the near future. However, Hawley Creek remains functionally disconnected because existing diversions serve as seasonal passage barriers. The following projects have been completed:

1) Big Timber Reconnection and Upper Lemhi Flow Improvement (OCS 007 06 SA) – Re-established the historic Hawley/Eighteenmile Creek channel and removed a seasonal passage barrier. Completed in 2008.

2) Hawley Creek BLM Culvert Replacement (OSC 016 09 SA) – An undersized perched culvert that created a seasonal fish passage barrier was replaced with 24' x 45' bridge that opened up year-round passage for all life stages of fish. Project completed in April 2013.

3) Hawley Creek Private Culvert Replacement (OSC 066 08 SA) – Removed two side-by-side culverts that were seasonal passage barriers and replaced with a 24' x 35' bridge that opened up year round passage for all life stages of fish. Project completed in April 2013.

4) Upper Hawley Creek Irrigation Improvement (OSC 009 12 SA) – Removed a headgate that served as a passage barrier and converted 2.1 miles of open irrigation ditch to a screened pipeline. The barrier removal opened up 1.4 miles of habitat upstream of the old headgate. The irrigation improvements resulted in an increase of 5.3 cfs, which improved flow through approximately 7.1 miles of stream. A fish screen was installed at the new point of diversion. Project completed in 2015.

5) Eighteenmile Creek Railroad Grade Culvert to Bridge (OSC 016 11 SA) – Two side-by-side culverts on Eighteenmile Creek were replaced with a 45' x 16' bridge. Project completed in 2015.

6) Hawley-Eighteenmile Creek Intercept Irrigation Project (BPA 2010-072-00) – Removed a culvert that was a passage barrier and moved the point of diversion downstream to use with a pump and pod system, which conserved 0.7 cfs. Completed in 2015.

7) Hawley-Eighteenmile Creek Fish Screen Project (BPA 2007-399-00) – A drum screen was installed at the new point of diversion on the Hawley-Eighteenmile Creek Intercept project. Screen installed in 2015.

8) Hawley Creek Access and Flow Enhancement LHaC-01 (OSC 019 13 SA) – Replaced an open ditch irrigation system with a pipeline that conserved 1.5 cfs, leading to a flow increase in 8.5 miles of the stream. A component of the irrigation improvement project includes the removal of a diversion structure that was a seasonal passage barrier. Additionally, five beaver dam analog structures were installed to create fish habitat, inundate the historic floodplain, and promote re-establishment of riparian vegetation. Completed in 2017.

9) Hawley Creek Access and Flow Enhancement LHaC-02 (OSC 020 13 SA) - Replaced an open irrigation ditch with 2 miles of pipeline that resulted in a water savings of 2.5 cfs. This increased flow in approximately 8.5 miles of stream length. Completed in 2017.

Little Springs Creek

Little Springs Creek is a relatively small, spring-fed creek with a total stream length of approximately 6.7 km. Multiple diversions on this stream served as passage barriers and completely dewatered portions of the stream channel (SRBA 2005). Steelhead, Westslope Cutthroat Trout, and Bull Trout are present. Little Springs Creek is an important rearing area for juvenile Chinook Salmon and steelhead, but it was historically an important spawning area for anadromous salmonids as well.

Little Springs Creek is the most comprehensively restored tributary in the basin. It was functionally reconnected in 2011. The actions completed to date resulted in the removal of every barrier and protection of over 90% of the riparian habitat. The stream received extensive rehabilitation through the following projects:

1) Little Springs Creek Riparian Fencing (OSC 034 05 SA) – A 4-pole jack fence was installed along 0.54 miles of stream length to exclude cattle and protect intact stream habitat and riparian areas. Project completed in 2005.

2) Amonson Little Springs Creek (OSC 010 06 SA) – Stabilized 2.8 miles of streambank to prevent erosion and sedimentation issues. Instream habitat actions included 10 rootwads, 31 large willow clumps, 95 pools with point bars, and installation of spawning and riffle substrate. A 1.3-mile jack fence was installed to exclude cattle grazing and protect riparian areas. Project completed in 2009.

3) Little Springs Creek Pond Restoration (OSC 009 06 SA) – A 2,970 ft spring-creek channel was constructed to direct water around an artificial pond to prevent artificially high temperatures within Little Springs Creek. The newly constructed channel contained a functional floodplain and instream habitat to promote rearing and spawning. Project completed in 2010.

4) Lemhi River L-52 Removal (OSC 011 09 SA) – The L-52 canal, which diverted water from Little Springs Creek, was removed to save water in the creek and reduce summer water temperatures. Three fish passage barriers were also removed as a component of the project. This project also connected Mill Creek with Little Springs creek, increasing rearing habitat and flow input to Little Springs Creek. Water savings attributed to this project were 2.0 cfs in Little Springs Creek and 1.0 cfs in Mill Creek. Project completed in 2012.

5) Cottom Little Springs Creek (OSC 005 10 SA) – An undersized road culvert that served as a seasonal passage barrier was replaced with a 5’ x 24’ squashed culvert to enable passage by all life stages of fish during all times of the year. A 0.38-mile stream reach was narrowed from 12 feet to 4 feet to restore channel function and decrease summer water temperatures. Willow plantings occurred on 1.5 acres of riparian habitat. An unused irrigation ditch was plugged, resulting in the reconnection of a small tributary (Walters Creek) to Little Springs Creek. This added 1.85 cfs of base flow to Little Springs Creek. Project completed in 2012.

6) LSC/Lemhi River Restoration (OSC 006 10 SA) – The channelized 0.35 miles of lower Little Springs Creek was reconstructed to create a natural functioning channel and floodplain with complex rearing and spawning habitat. Eighteen logjams and two rootwads were installed throughout the restored reach to create habitat and encourage pool

scouring. The regraded floodplain was planted with native willows and grasses. Projected completed in 2012.

7) L50 Habitat Restoration and LSC-3 Diversion Removal (015 10 SA) – A 1.8 cfs Lemhi River water right was comingled with Little Springs Creek and subsequently diverted at the LSC-3 diversion, which was a fish passage barrier. The water right point of diversion was transferred to L46a, which enabled the removal of LSC-3, thereby opening up fish passage and increasing flow in Little Springs Creek. Project completed in 2012.

Kenney Creek

Kenney Creek drains approximately 62.8km^2 and enters the Lemhi River about 29 km upstream of the confluence with the Salmon River. Historical base flows in Kenney Creek prior to irrigation impacts were between 10 and 15 cfs (BLM 1998). Historically, three diversions operated on Kenney Creek, with a total water right claim of 6.31 cfs (Murphy and Horsmon 2003). The LKC-02 diversion is approximately 1.75 km upstream of the mouth, which used to dewater the lower reaches of Kenney Creek during low-water years and functioned as an impassable barrier during low flow conditions (Murphy and Horsmon 2003).

The importance of Kenney Creek as a spawning tributary for Chinook Salmon is unknown, but it was thought to provide summer rearing habitat to juveniles, particularly during the warm summer months when the lower Lemhi River exceeded optimal thermal criteria. Steelhead spawning is well documented in Kenney Creek. Baseline fish surveys conducted by the IDFG Screen Program in 2003 did not observe any juvenile Chinook Salmon, but did observe steelhead, Westslope Cutthroat Trout, Bull Trout, and Brook Trout (Murphy and Horsmon 2003). They suggested that Bull Trout probably exhibited resident life histories because of the passage barrier at LKC-02. A weir was operated about 360 m upstream from the mouth during spring 2012 and 2013; five steelhead and four fluvial Rainbow Trout were captured in 2012 and four steelhead and two fluvial Rainbow Trout were captured in 2012 (Messner et al., in review).

Kenney Creek was functionally reconnected in 2011. Multiple restoration projects have occurred in this tributary and the lower portion of the stream is located on the Kenney Creek Ranch, which now has a conservation easement. The following projects were completed:

1) Kenney Creek Diversion Closure/Sprinkler (OSC 047 04 SA) – A diversion that served as a seasonal passage barrier was removed and a sprinkler was installed to return 0.4 cfs of flow to Kenney Creek. Project completed in 2007.

2) LKC-02 and LKC-03 (BPA 18384) – Modified the existing diversions that were seasonal passage barriers and installed new headgates and fish screens to improve fish passage. Project actions completed in 2005.

3) Upper Kenney Creek Riparian Fence (BPA 44042) – Installed 4-pole jack fence along 1.2 miles of stream to exclude cattle from 50 acres of riparian habitat. Project completed in 2010.

4) Kenney Creek Ranch Acquisition (OSC 063 05 SA) – The 520 acre Kenney Creek Ranch and three adjacent parcels totaling 79 acres were secured in 2010, with formal conservation easement terms solidified in 2011. The easement secured control of all Kenney Creek diversions, obligating the landowner to abide by the water management

agreement. The agreement specifies a minimum flow of 4 cfs, which results in permanent surface water connection of Kenney Creek to the Lemhi River.

5) Kenney Creek Culvert Replacement Project (BPA 54777) – The Old Lemhi Road culvert on Kenney Creek was a seasonal barrier to juvenile fish and was replaced with a free spanning bridge to allow natural channel migration and fish passage. Project completed in 2012.

Bohannon Creek

Bohannon Creek drains approximately 57 km^2 and flows into the Lemhi River about 16 km upstream of the confluence with the Salmon River. Historically, 13 diversions existed on Bohannon Creek that served 25 waters rights with a cumulative volume of 40.82 cfs. These withdraws were typically significant enough to dewater portions of the creek in most years (BLM 1998). In addition to irrigation development on the creek, historical mining practices have left a significant legacy, with large portions of the channel dredged and tailings placed in large portions of the riparian area (BLM 1998).

Steelhead spawning was documented nearly up to the East Fork Bohannon Creek confluence (6 km upstream from the mouth), but the dewatering that occurred below the LBC-03 diversion may have affected spawning distributions in some years (Murphy and Yanke 2003). Surveys by the IDFG Screen Program in 2003 documented the presence of steelhead, Bull Trout, and Brook Trout. Interestingly, they only found Cutthroat Trout in Bohannon Lake, which were mixed origins of Westslope and Yellowstone Cutthroat Trout brood stock (Murphy and Yanke 2003). A weir was operated about 75 m upstream from the mouth during spring 2012 and 2013; 15 steelhead and 8 fluvial Rainbow Trout were captured in 2012 and 5 steelhead and 3 fluvial Rainbow Trout were captured in 2012 (Messner et al., in review).

Bohannon Creek is not considered functionally connected because of seasonal passage issues below the BHC-3 diversion. A project planned for implementation in 2017-2018 will address this last remaining component of reconnection. The following projects are completed:

1) Remove/Install Diversion (BPA 19794) – The LBC-05 diversion was modified to accommodate a pipe and sprinkler system, which resulted in a water savings of 1.5 cfs. Project completed in 2005.

2) LBC-03, LBC-04, LBC-05, and LBC-06 Diversion Improvements (BPA 23364) - Modified existing diversions and installed screens to improve fish passage. Project actions completed in 2006.

3) Bohannon Creek Culvert Replacement Project (BPA 54777) – The Old Lemhi Road culvert on Bohannon Creek was a seasonal passage barrier to juvenile fish and was replaced with a properly-sized culvert to allow channel function and accommodate year-round passage of all life stages of fish. Project completed in 2012.

4) Bohannon: Diversion Removal and Consolidation and Construction of New Diversion (BPA 58410) – Installed a more efficient irrigation system and consolidated four upper Bohannon Creek irrigation diversions into two (1 new and 1 modified). A minimum flow agreement was established in a seasonally dewatered reach. The water conservation measures resulted in a 5-8 cfs increase in Bohannon Creek discharge during the irrigation season. Project completed in 2012.

5) Bohannon: Add Structures to Increase Habitat Complexity (BPA 58410) – Installation of habitat structures improved a simplified section of Bohannon Creek to increase complexity. Project completed in 2012.

6) 429c-14, 429c-15, 429c-16, 429c-17 Bohannon Creek (BPA 69884) – A minimum flow agreement of 2 cfs between April 1 and June 30 was implemented at the BHC-3 diversion to prevent the downstream reach from dewatering and drying up steelhead redds. Project was first implemented in 2014 and was renewed every year since then.

7) Lower Bohannon Creek Private Culvert Replacement (OSC 017 11 SA) – Two side-by-side undersized culverts created a seasonal juvenile fish passage barrier and was replaced with a 40' x 16' full span bridge. Project completed in 2015.

8) IDFG: Upper Bohannon Creek Culvert Replacement (BPA 67754) – A culvert that was a seasonal passage barrier was replaced with a bridge to enable year-round passage by all life stages of fish. Project completed in 2015.

Hayden Creek

Hayden Creek is the largest tributary to the Lemhi River and drains 392.7 km^2. It is one of only two tributaries (Big Springs Creek) that maintained perennial connection to the Lemhi River following agricultural development. The drainage contains many sub-drainages that provide considerable flow to Hayden Creek and contain high quality spawning and rearing habitat. Hayden Creek has continued to support populations of anadromous Chinook Salmon and steelhead, as well as fluvial and resident life history forms of Bull Trout and Westslope Cutthroat Trout. As such, it provides insight into the historical importance of the other tributaries in the basin. It also serves as a reference tributary within the IMW study design. A weir was operated about 1 km upstream from the mouth during spring 2012 and 2013; 26 steelhead, 2 fluvial Rainbow Trout, a Bull Trout, and a Mountain Whitefish were captured in 2012 and 7 steelhead and 2 fluvial Rainbow Trout were captured in 2012 (Messner et al., in review).

1) Kenney Creek Ranch Acquisition (OSC 063 05 SA) – As part of the Kenney Creek ranch project, the title of the Kenney Creek property was transferred to the owner of the Hayden Creek Ranch in return for a conservation easement on the Hayden Creek Ranch. This project protected the 1,000-acre Hayden Creek ranch and contains provisions for maintaining the high quality condition of that reach of Hayden Creek. Project completed in 2011.

Appendix B: Electrofishing sampling effort in Lemhi River tributaries.

Table B1. 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)	Standing stock 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	8.71	23.28	37.42
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
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
Hawley Creek				
2009	Depletion	0.60	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
Hayden Creek				
2009	Depletion	0.75	18.76	4.01
2010[a]	--	--	--	--
2011	Mark Recapture	1.00	18.76	5.34
2012	Depletion	0.59	18.76	3.17
2013	Mark Recapture	9.82	18.76	52.34

2014	Mark Recapture	10.60	18.76	56.50
2015	Mark Recapture	13.30	18.76	70.87
2016	Mark Recapture	8.71	18.76	46.41
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
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

[a] Not sampled

Appendix C: Eagle Valley Ranch Restoration Reach Evaluation

The Eagle Valley Ranch (EVR) restoration project is a large-scale, multi-year project in the lower Lemhi River basin. This project includes floodplain reconnection to the Lemhi River, re-meandering of the main channel, activation of historic channels, side channel construction, and placement of large woody debris. Due to the scale of the project, conducting a rigorous project-specific evaluation is a priority. Specific responses being evaluated include in-stream habitat conditions, adult and juvenile anadromous and resident/fluvial salmonid standing stock, and side channel use by juvenile anadromous and resident/fluvial salmonids. Pretreatment electrofishing surveys were conducted in the restoration reach in 2014. Starting with the 2015 field season, specific transects were chosen for sampling based on physical characteristics to evaluate the project within a BACI study design. The BACI design includes the EVR treatment reach, two control reaches downstream of Hayden Creek that represent the channelized and simplified habitat associated with much of the degraded lower Lemhi River, and a reference reach downstream of EVR that maintains the historic channel complexity and intact riparian forest that was prevalent throughout the Lemhi in a pre-degraded state (Figure 1).

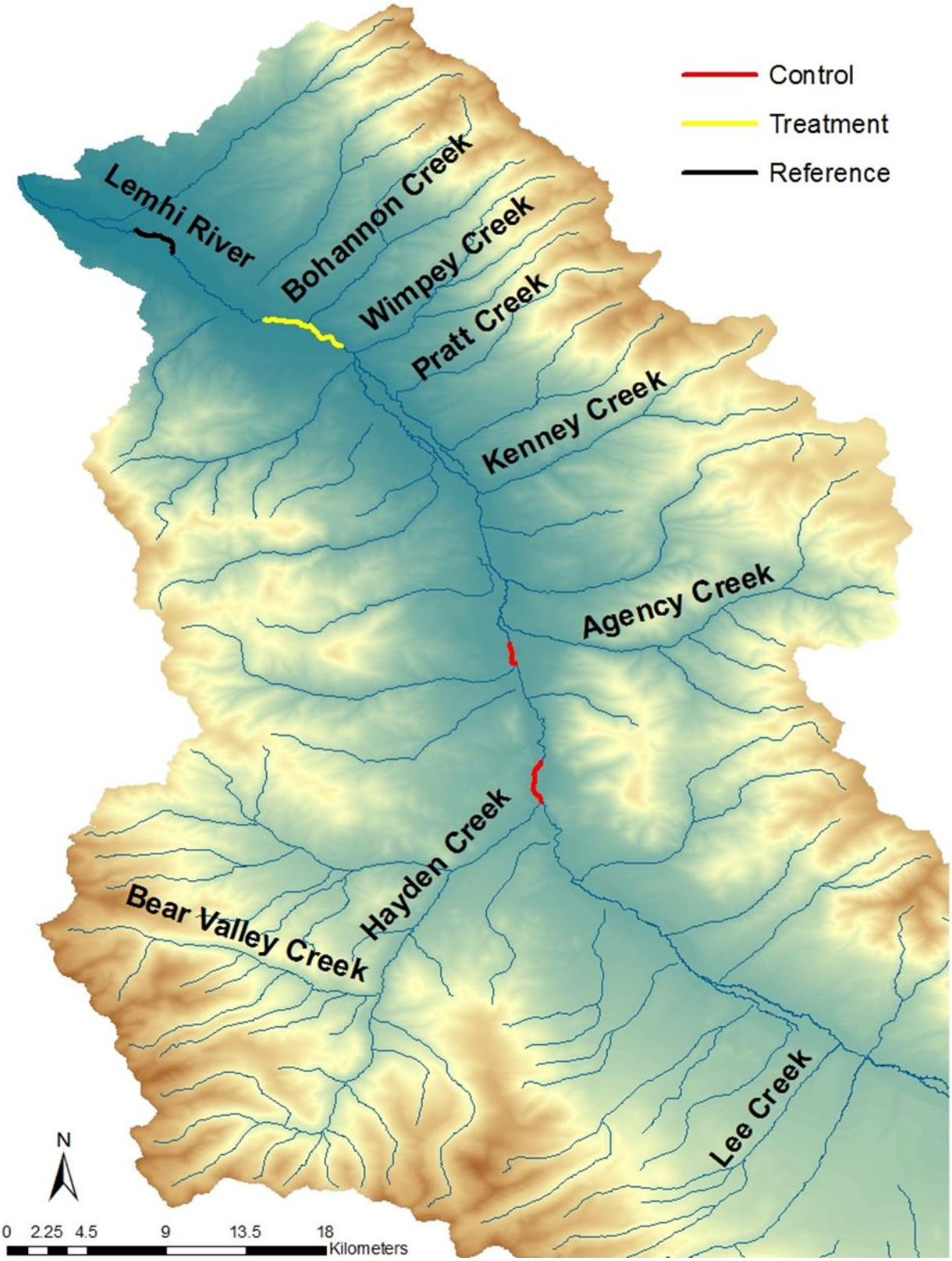

Figure C1. Control (red), treatment (yellow), and reference (black) reaches surveyed in the lower Lemhi River as part of the BACI study design for project-level monitoring.

Appendix D. Focal Species Distributions in Priority Tributaries in the Lemhi River Basin.

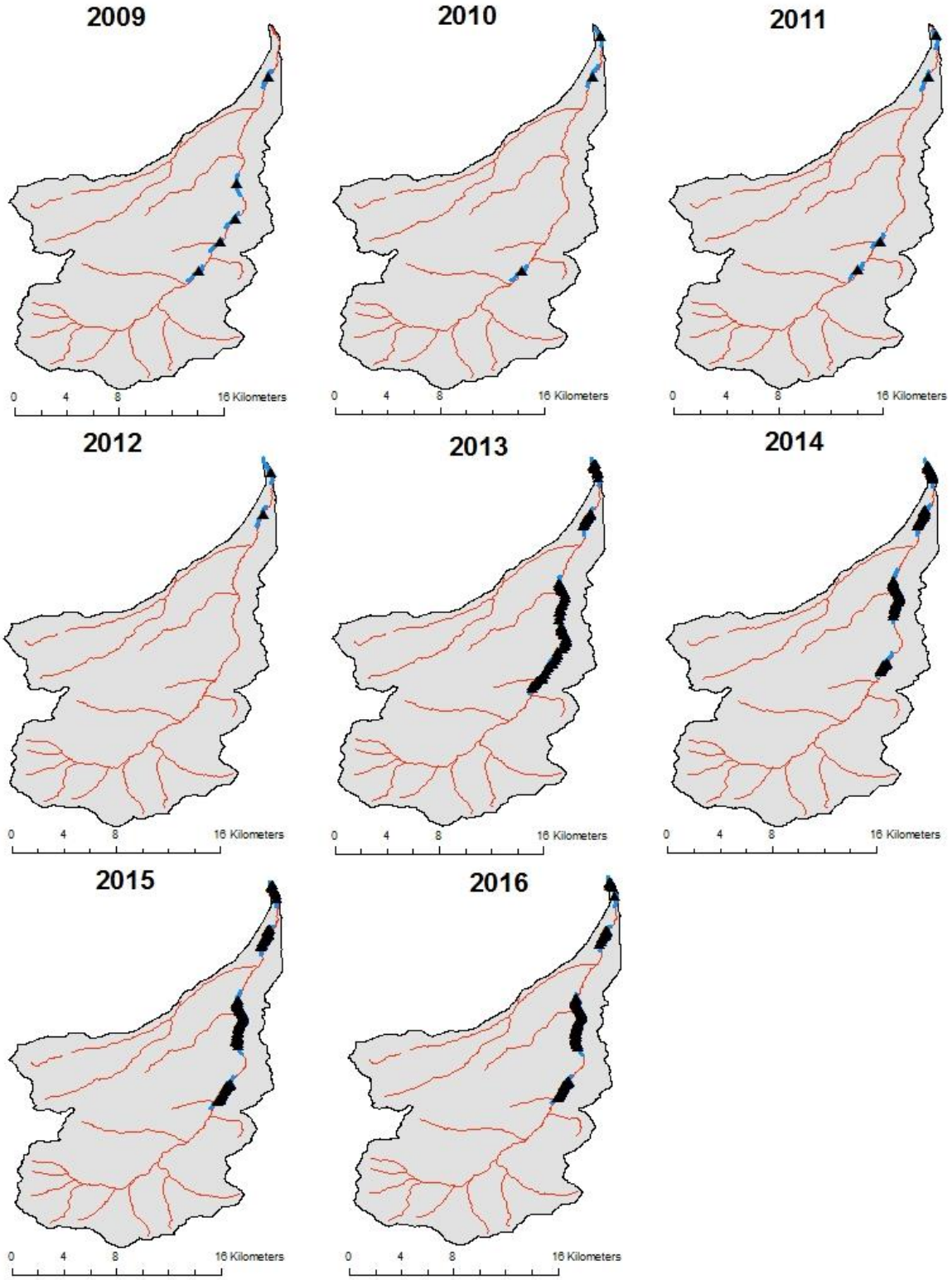

Figure D1. Distribution of steelhead encountered during annual summer electrofishing surveys in Big Timber Creek, 2009-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

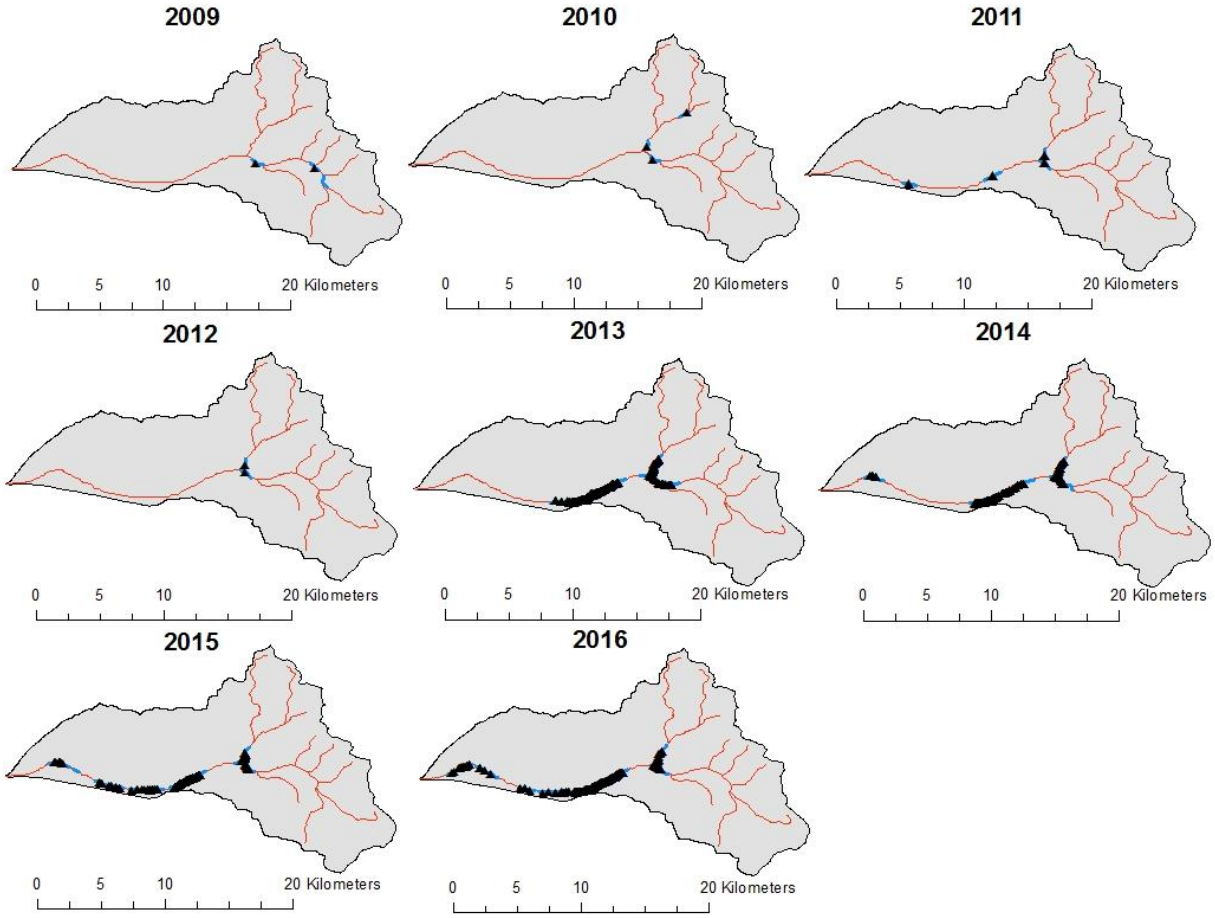

Figure D2. Distribution of steelhead encountered during annual summer electrofishing surveys in Hawley Creek, 2009-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

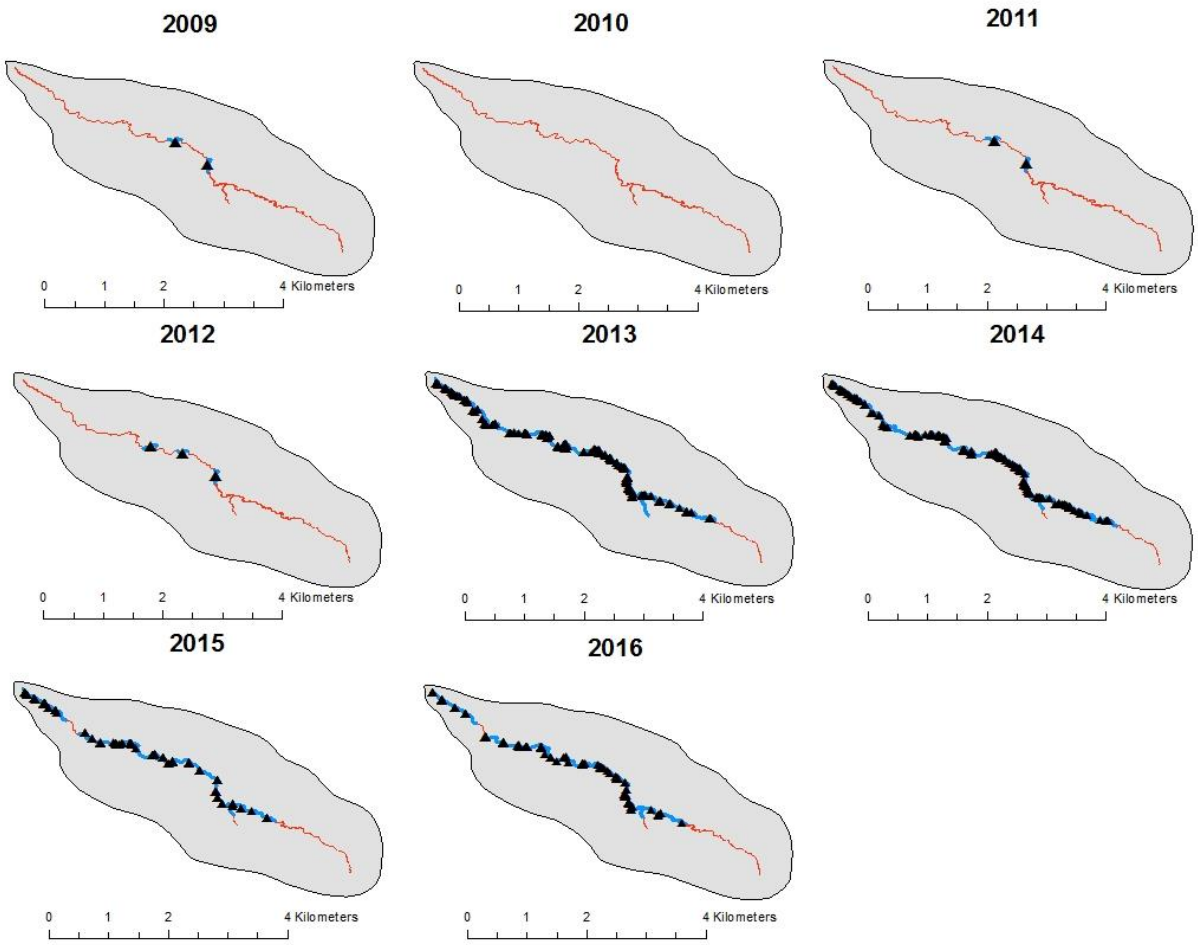

Figure D3. Distribution of steelhead encountered during annual summer electrofishing surveys in Little Springs Creek, 2009-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

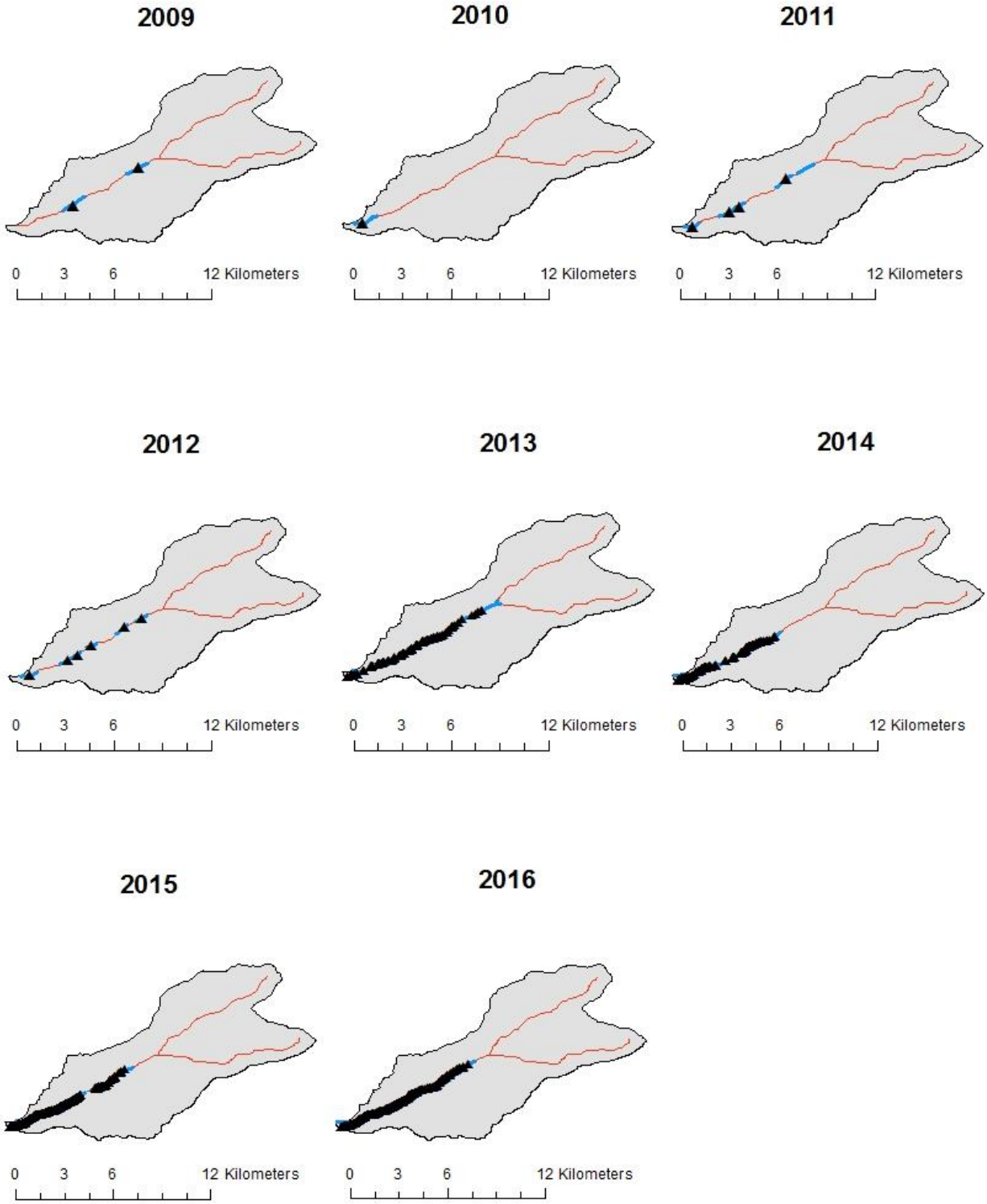

Figure D4. Distribution of steelhead encountered during annual summer electrofishing surveys in Kenney Creek, 2009-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

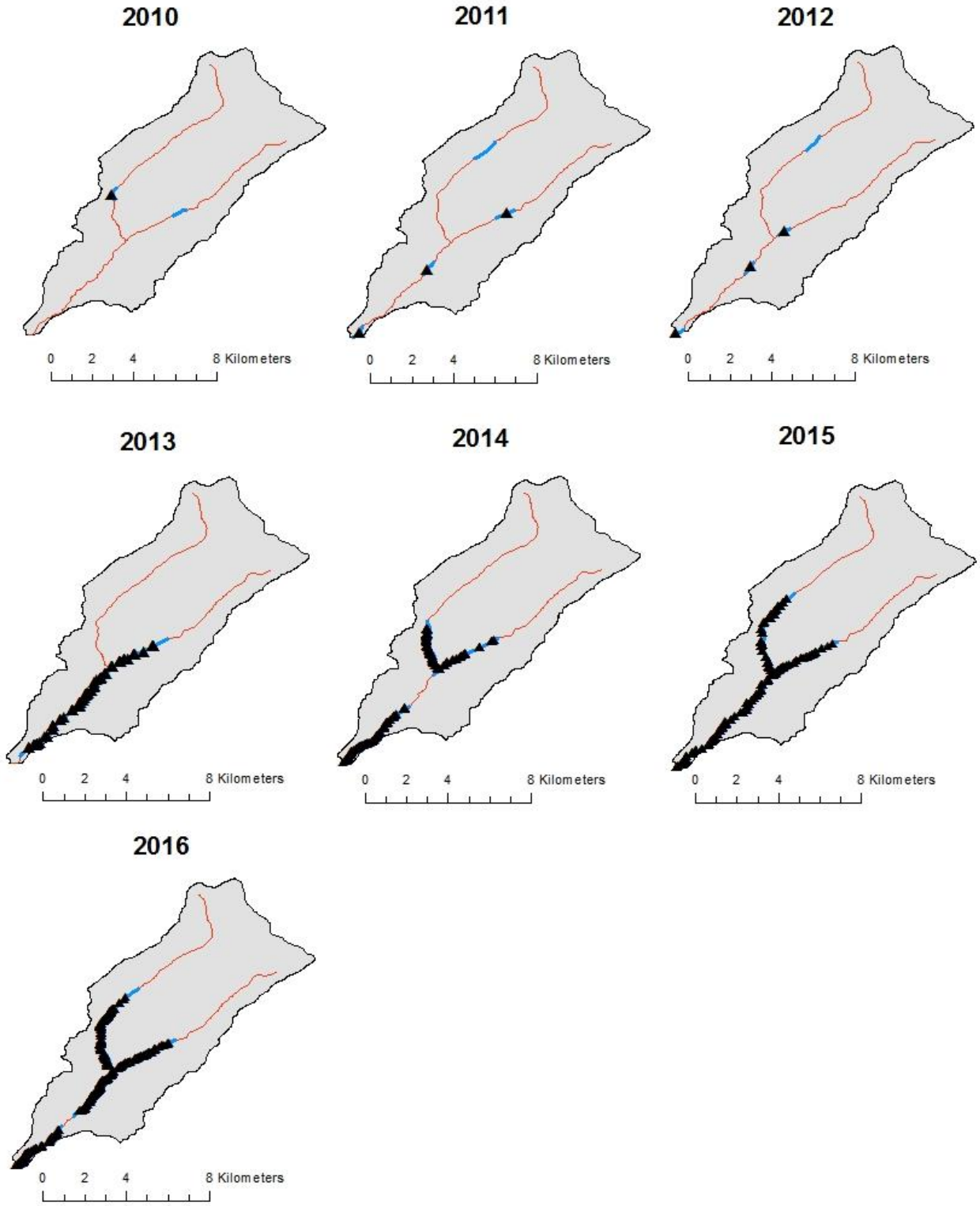

Figure D5. Distribution of steelhead encountered during annual summer electrofishing surveys in Bohannon Creek, 2010-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

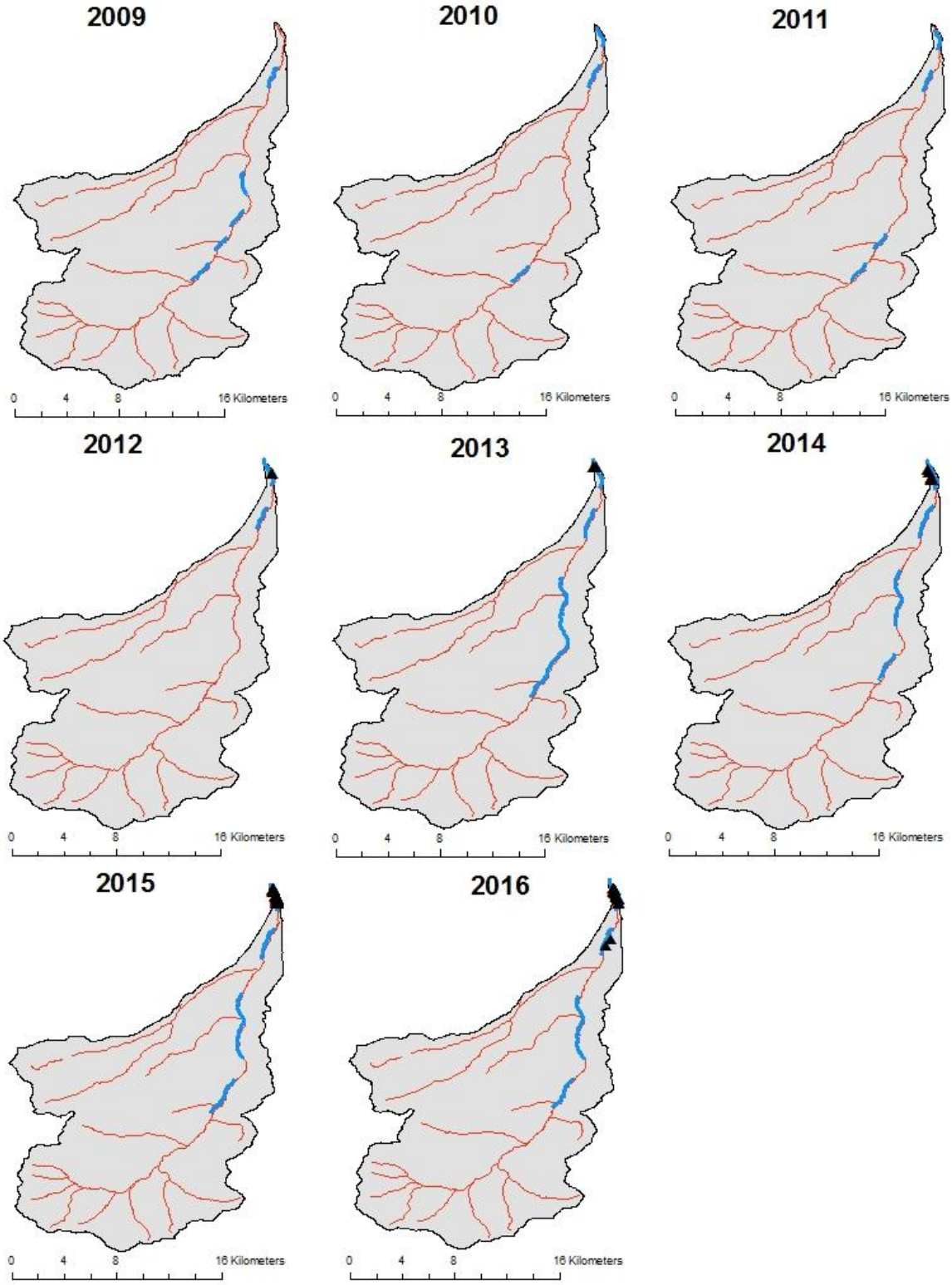

Figure D6. Distribution of juvenile Chinook Salmon encountered during annual summer electrofishing surveys in Big Timber Creek, 2009-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

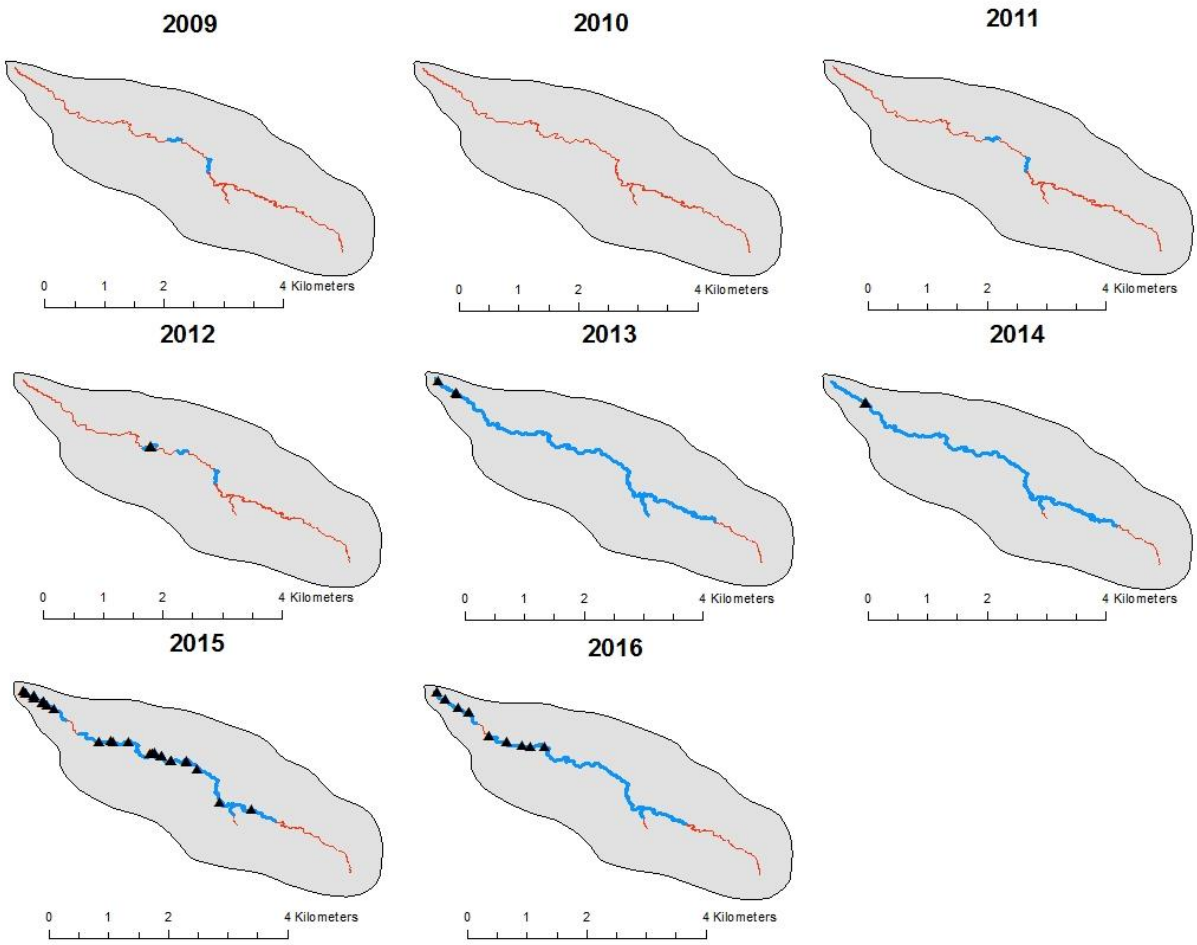

Figure D7. Distribution of juvenile Chinook Salmon encountered during annual summer electrofishing surveys in Little Springs Creek, 2009-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

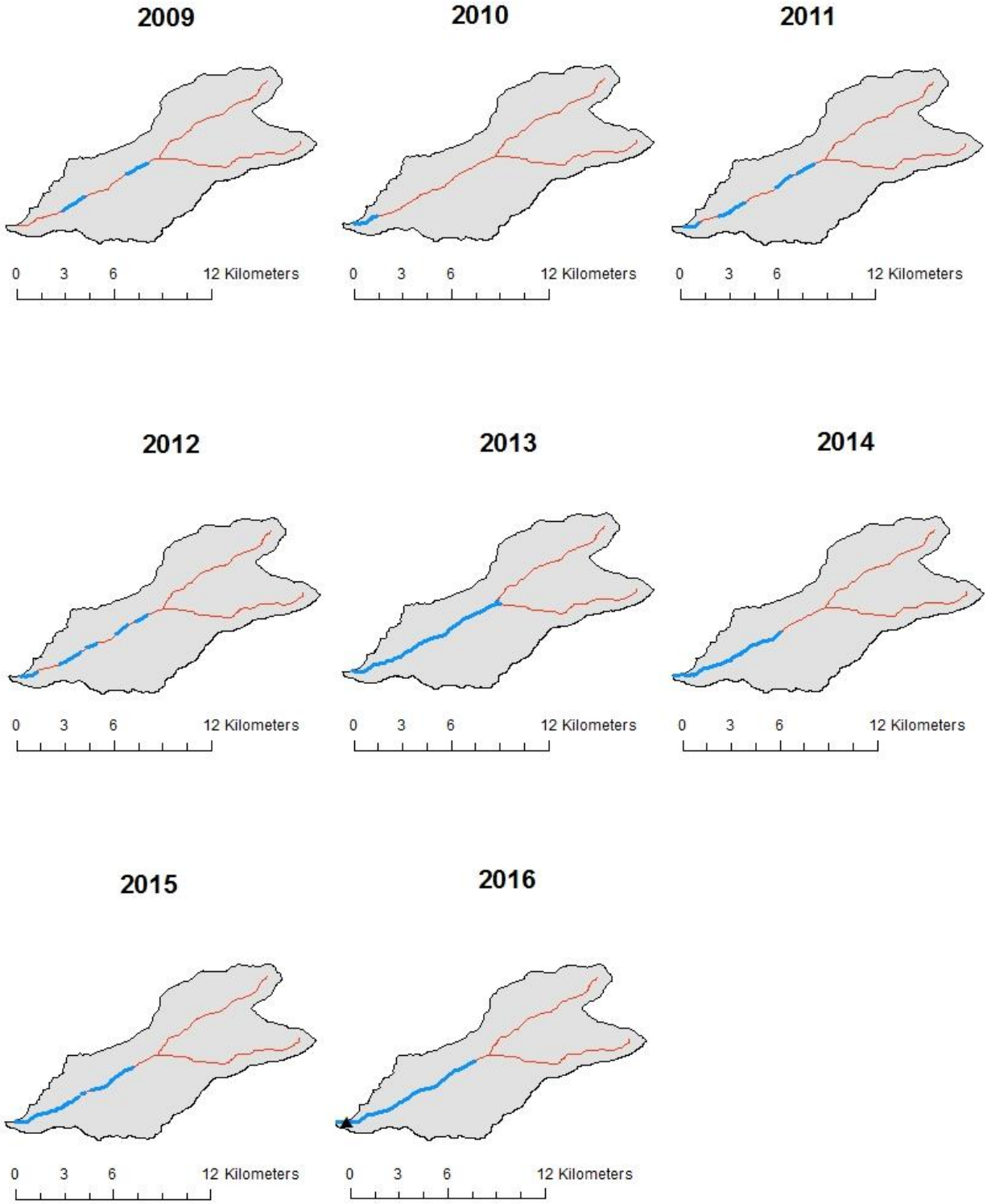

Figure D8. Distribution of juvenile Chinook Salmon encountered during annual summer electrofishing surveys in Kenney Creek, 2009-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

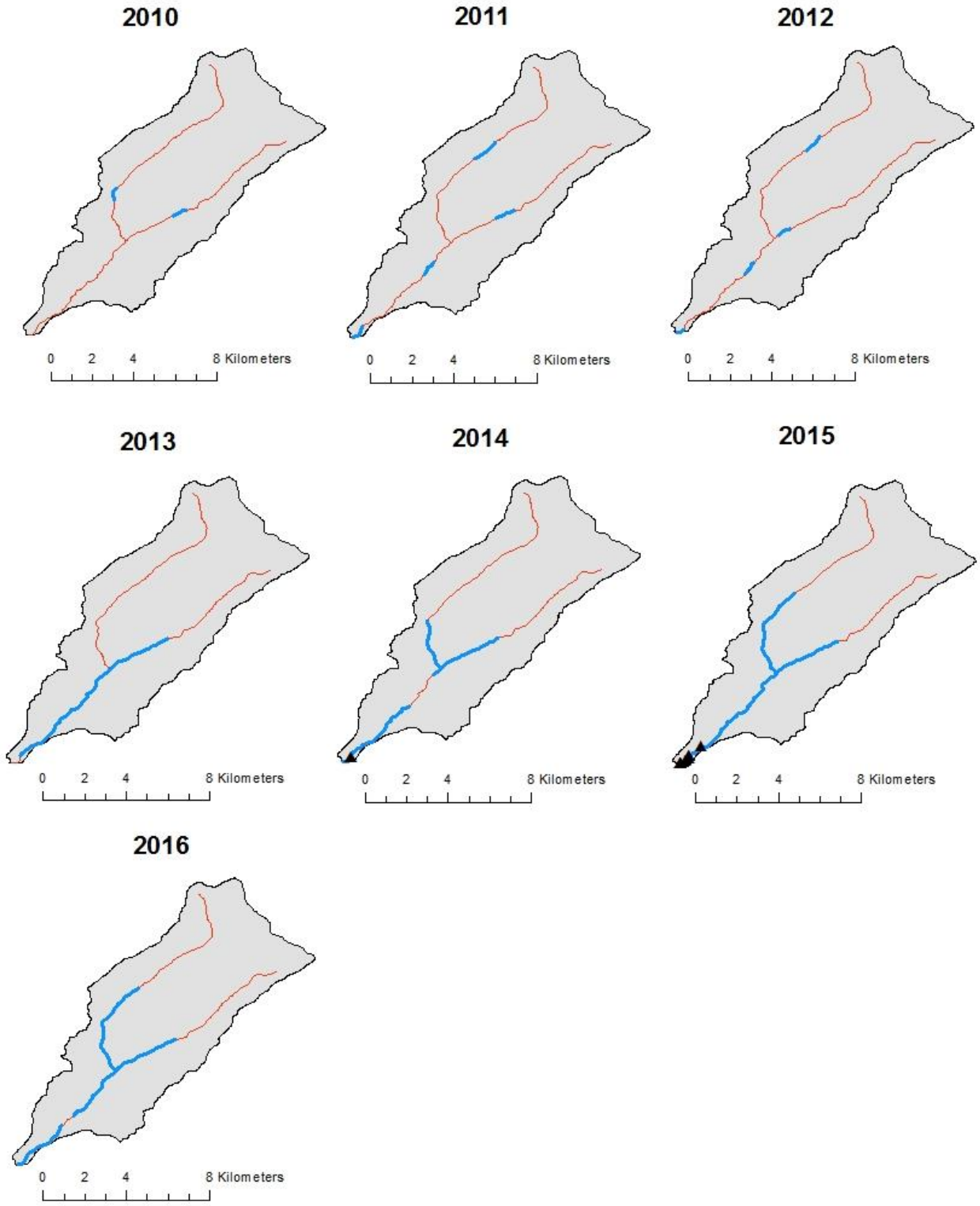

Figure D9. Distribution of juvenile Chinook Salmon encountered during annual summer electrofishing surveys in Bohannon Creek, 2010-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

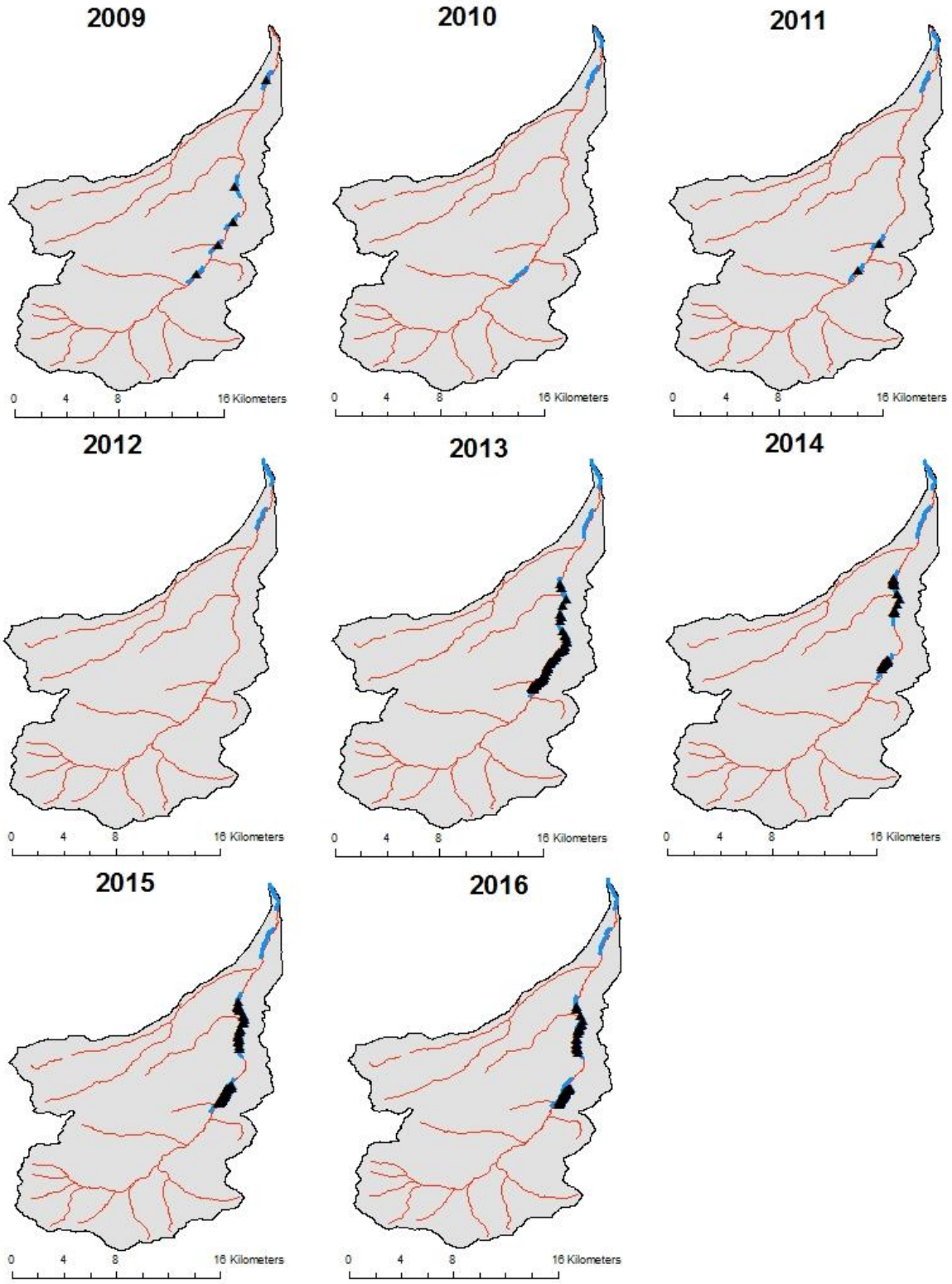

Figure D10. Distribution of Bull Trout encountered during annual summer electrofishing surveys in Big Timber Creek, 2009-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

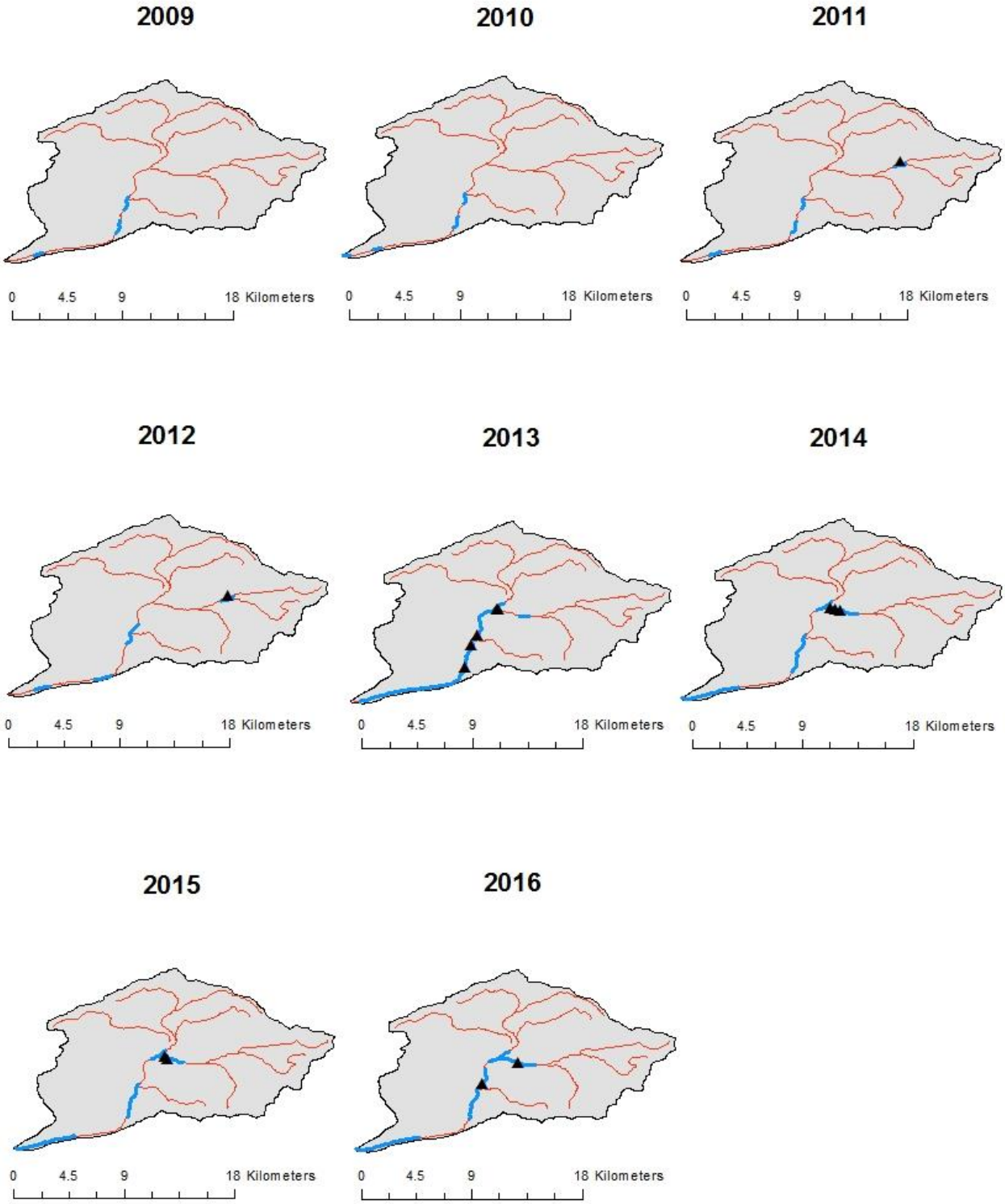

Figure D11. Distribution of Bull Trout encountered during annual summer electrofishing surveys in Canyon Creek, 2009-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

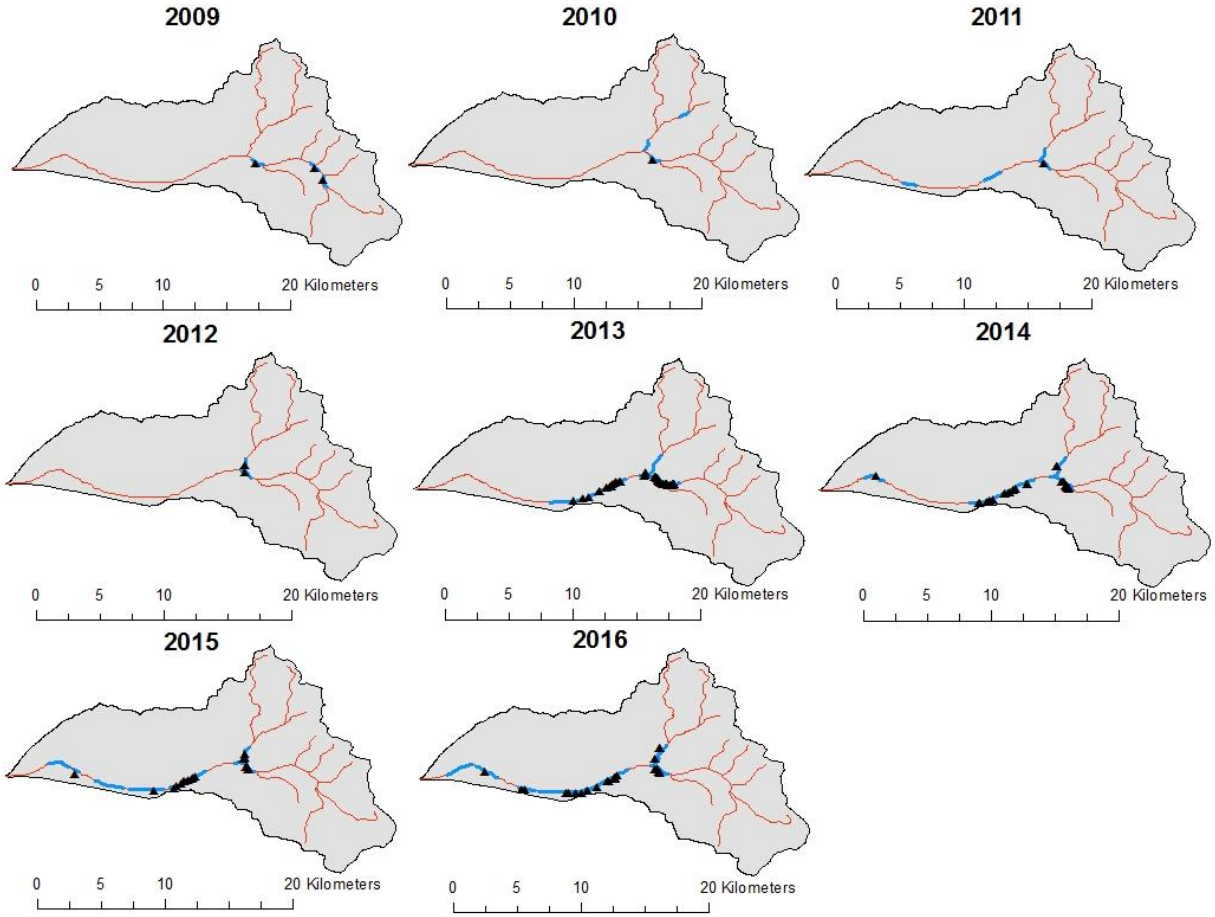

Figure D12. Distribution of Bull Trout encountered during annual summer electrofishing surveys in Hawley Creek, 2009-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

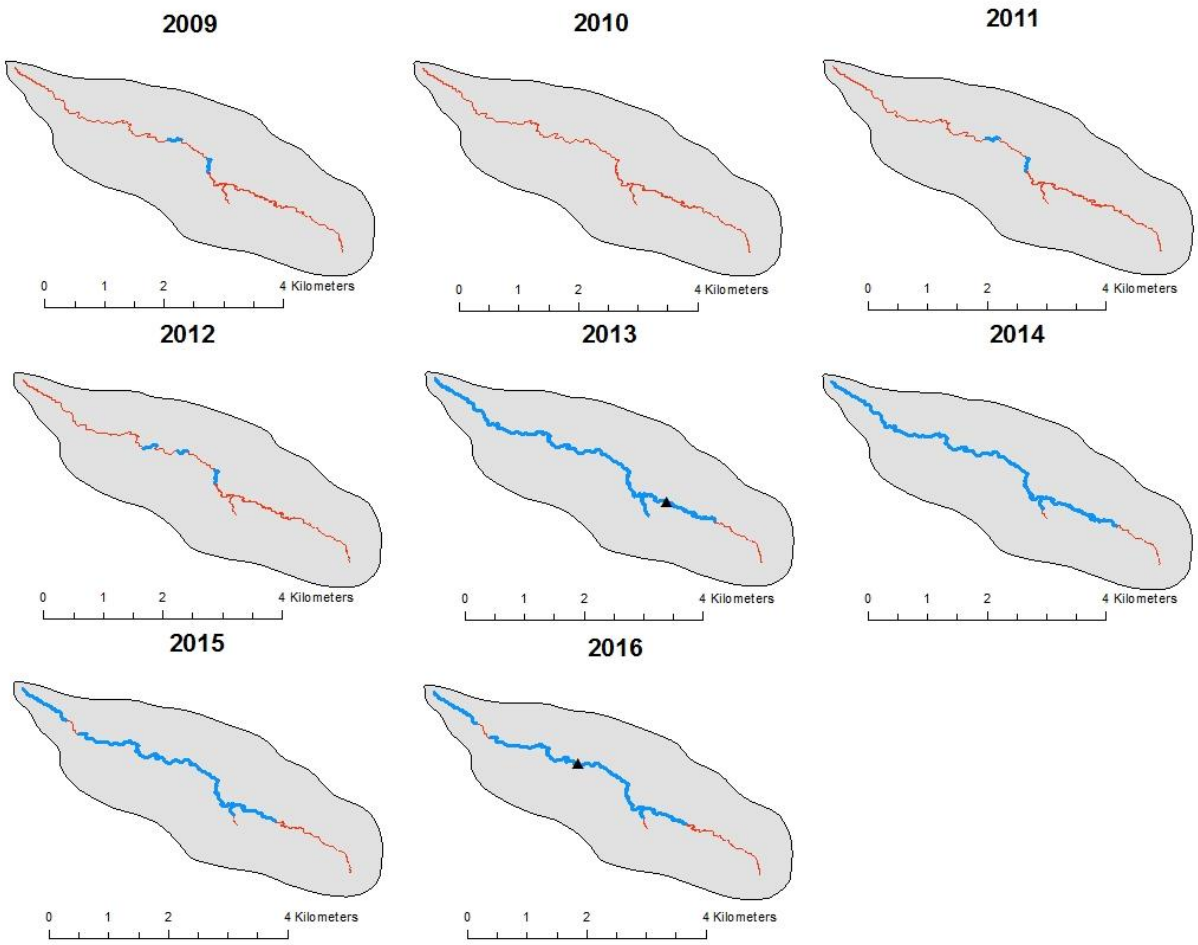

Figure D13. Distribution of Bull Trout encountered during annual summer electrofishing surveys in Little Springs Creek, 2009-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

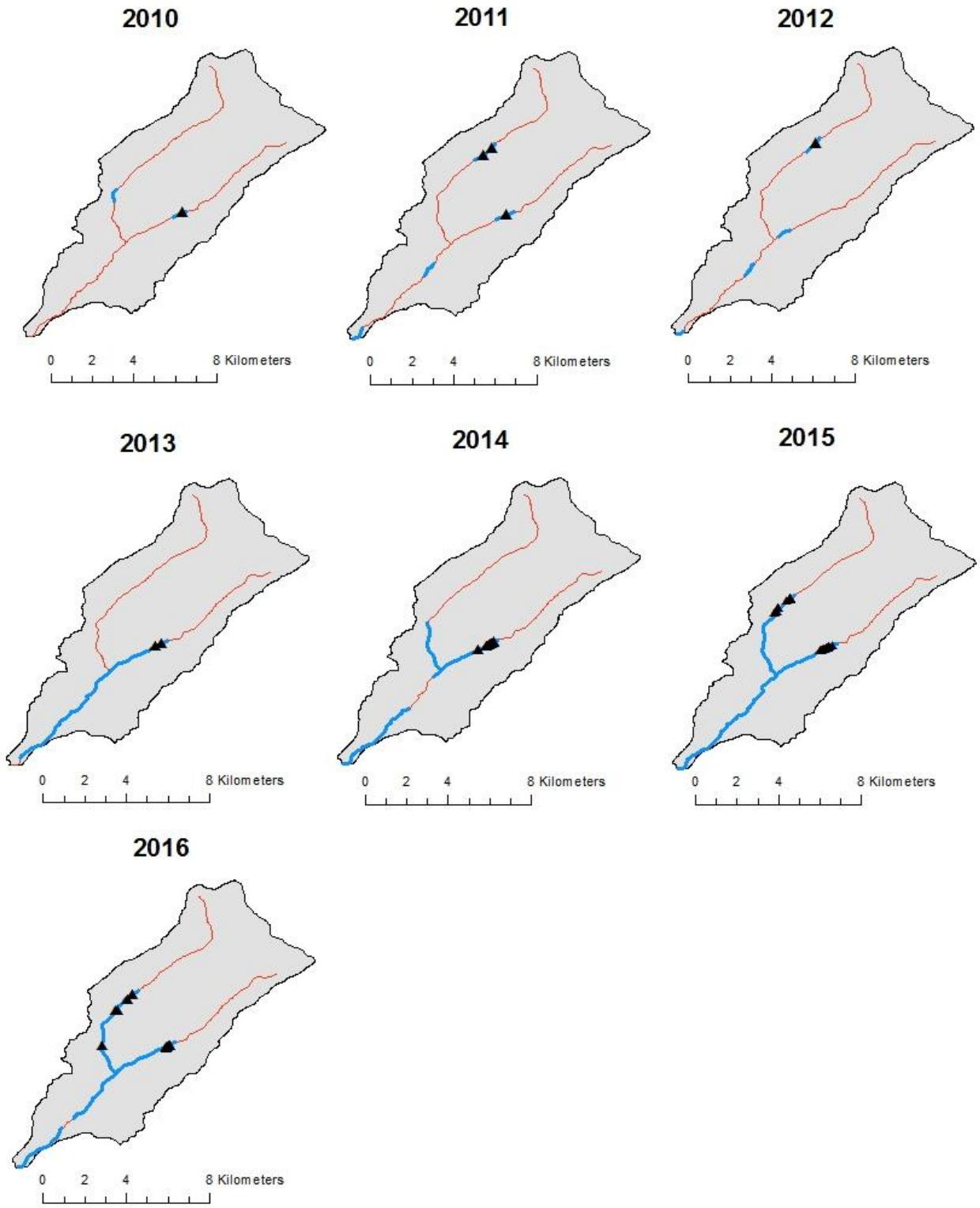

Figure D14. Distribution of Bull Trout encountered during annual summer electrofishing surveys in Bohannon Creek, 2010-2016. Blue lines indicate sampled areas and red lines represent non-sampled reaches.

Appendix E. Lemhi River Priority Tributary PIT-tag Array Interrogation Tables

Table E1. Number of steelhead and Chinook Salmon PIT-tagged in Big Timber Creek that were subsequently detected on the PIT-tag arrays near the mouth (BTC and BTL). Proportion of tagging cohort is shown in parentheses.

Tag year	Number tagged	Number detected by year							Total
		2010	2011	2012	2013	2014	2015	2016	
				Steelhead					
2009	162	0	0	0	0	0	0	0	0
2010	56	9 (16.1)	2 (3.6)	1 (1.8)	0	0	0	0	12 (21.4)
2011	176		22 (12.5)	5 (2.8)	2 (1.1)	0	0	0	29 (16.5)
2012	152			21 (13.8)	12 (7.9)	3 (2.0)	0	0	36 (23.7)
2013	779				8 (1.0)	1 (0.1)	1 (0.1)	0	10 (1.3)
2014	438					5 (1.1)	3 (0.7)	0	8 (1.8)
2015	560						3 (0.5)	0	3 (0.5)
2016	626							0	0
				Chinook Salmon					
2009	0	--	--	--	--	--	--	--	--
2010	0	--	--	--	--	--	--	--	--
2011	0		--	--	--	--	--	--	--
2012	7			2 (28.6)	0	0	0	0	2 (28.6)
2013	2				0	0	0	0	0
2014	4					0	1 (25.0)	0	1 (25.0)
2015	43						2 (4.7)	0	2 (4.7)
2016	22							0	0

Table E2. Number of steelhead, Chinook Salmon, and Bull Trout PIT-tagged in the Lemhi River basin outside of Big Timber Creek that were subsequently detected on the lower (BTC and BTL), middle (BTM), and upper (BTU) PIT-tag arrays.

Species	Number detected by year							Total
	2010	2011	2012	2013	2014	2015	2016	
				Lower				
Steelhead	0	1	5	15	18	11	0	50
Chinook Salmon	0	0	0	0	0	31	0	31
Bull Trout	0	0	0	1	0	1	0	2
				Middle				
Steelhead	--	--	--	--	--	0	3	3
Chinook Salmon	--	--	--	--	--	0	0	0
Bull Trout								
				Upper				
Steelhead	--	--	--	--	0	0	2	2
Chinook Salmon	--	--	--	--	0	0	1	1
Bull Trout	--	--	--	--	0	0	0	0

Table E3. Number of steelhead and Chinook Salmon PIT-tagged in Canyon Creek that were subsequently detected on the PIT-tag array near the mouth (CAC). Proportion of tagging cohort is shown in parentheses.

Tag year	Number tagged	Number detected by year							Total
		2010	2011	2012	2013	2014	2015	2016	
					Steelhead				
2009	112	1 (0.9)	0	0	0	0	0	0	1 (0.9)
2010	109		3 (2.8)	1 (0.9)	0	0	0	0	4 (3.7)
2011	191		2 (1.0)	4 (2.1)	2 (1.0)	0	0	0	8 (4.2)
2012	355			21 (5.9)	61 (17.2)	1 (0.3)	3 (0.8)	0	86 (24.2)
2013	441				8 (1.8)	4 (0.9)	1 (0.2)	0	13 (2.9)
2014	378					22 (5.8)	4 (1.1)	0	26 (6.9)
2015	285						7 (2.5)	4 (1.4)	11 (3.9)
2016	768							25 (3.3)	25 (3.3)
					Chinook Salmon				
2009	0	--	--	--	--	--	--	--	--
2010	0	--	--	--	--	--	--	--	--
2011	2			1 (50.0)	0	0	0	0	1 (50.0)
2012	23			12 (52.2)	5 (21.7)	0	0	0	17 (73.9)
2013	0				--	--	--	--	--
2014	0					--	--	--	--
2015	49						6 (12.2)	3 (6.1)	9 (18.4)
2016	32							15 (46.9)	15 (46.9)

Table E4. Number of steelhead, Chinook Salmon, and Bull Trout PIT-tagged in the Lemhi River basin outside of Canyon Creek that were subsequently detected on the PIT-tag array near the mouth of Canyon Creek (CAC).

Species	Number detected by year							Total
	2010	2011	2012	2013	2014	2015	2016	
Steelhead	0	1	4	3	2	3	7	20
Chinook Salmon	0	0	0	0	0	6	13	19
Bull Trout	0	0	0	0	0	0	0	0

Table E5. Number of steelhead PIT-tagged in Hawley Creek that were subsequently detected on the PIT-tag arrays near the mouth of Hawley Creek and Eighteenmile Creek (HEC and 18M). Proportion of tagging cohort is shown in parentheses.

Tag Year	Number Tagged	Number detected by year				
		2013	2014	2015	2016	Total
2013	422	1(0.2)	6(1.4)	0	0	7(1.7)
2014	161		1(0.6)	1(0.6)	0	2(1.2)
2015	253			0	0	0
2016	541				0	0

Table E6. Number of steelhead, Chinook Salmon, and Bull Trout PIT-tagged in the Lemhi River basin outside of Hawley Creek that were subsequently detected on the PIT-tag array near the mouth of Hawley Creek and Eighteenmile Creek (HEC and 18M).

	Number detected by year				
Species	2013	2014	2015	2016	Total
Steelhead	0	3	1	0	4
Chinook Salmon	0	0	0	0	0
Bull Trout	0	0	0	0	0

Table E7. Number of steelhead and Chinook Salmon PIT-tagged in Little Springs Creek that were subsequently detected on the PIT-tag array near the mouth (LLS). Proportion of tagging cohort is shown in parentheses.

Tag year	Number tagged	Number detected by year						Total
		2011	2012	2013	2014	2015	2016	
				Steelhead				
2011	25	2 (8.0)	8 (32.0)	2 (8.0)	0	0	0	12 (48.0)
2012	173		4 (2.3)	64 (37)	3 (1.7)	0	0	71 (41.0)
2013	501			72 (14.4)	55 (11.0)	5 (1.0)	0	132 (26.3)
2014	174				44 (25.3)	8 (4.6)	0	52 (29.9)
2015	177					55 (31.1)	12 (6.8)	67 (37.9)
2016	184						50 (27.2)	50 (27.2)
				Chinook Salmon				
2011	0	--	--	--	--	--	--	--
2012	7		1 (14.3)	3 (42.9)	0	0	0	4 (57.1)
2013	10			6 (60.0)	2 (20.0)	0	0	8 (80.0)
2014	1				0	0	0	0
2015	26					11 (42.3)	2 (7.7)	13 (50.0)
2016	13						5 (38.5)	5 (38.5)

Table E8. Number of steelhead, Chinook Salmon, and Bull Trout PIT-tagged in the Lemhi River basin outside of Little Springs Creek that were subsequently detected on the PIT-tag array near the mouth of Little Springs Creek (LLS).

Species	Number detected by year						Total
	2011	2012	2013	2014	2015	2016	
Steelhead	13	17	50	21	30	47	178
Chinook Salmon	0	2	2	2	9	6	21
Bull Trout	0	0	3	1	1	2	7

Table E9. Number of steelhead and Bull Trout PIT-tagged in Kenney Creek that were subsequently detected on the PIT-tag array near the mouth of Kenney Creek (KEN). Proportion of tagging cohort is shown in parentheses.

Tag year	Number tagged	Number detected by year						Total
		2010	2011	2012	2013	2014	2015	
				Steelhead				
2009	56	0	2 (3.6)	0	0	0	0	2 (3.6)
2010	216	1 (0.5)	39 (18.1)	9 (4.2)	1 (0.5)	2 (0.9)	0	52 (24.1)
2011	139		8 (5.8)	14 (10.1)	5 (3.6)	7 (5.0)	0	34 (24.5)
2012	557			50 (9.0)	143 (25.7)	24 (4.3)	4 (0.7)	221 (39.7)
2013	461				14 (3.0)	90 (19.5)	9 (2.0)	113 (24.5)
2014	280					11 (3.9)	24 (8.6)	35 (12.5)
2015	365						4 (1.1)	4 (1.1)
				Bull Trout				
2009	0	--	--	--	--	--	--	--
2010	0	--	--	--	--	--	--	--
2011	33			1 (3.0)	0	0	0	1 (3.0)
2012	82			3 (3.7)	0	2 (2.4)	0	5 (6.1)
2013	87				2 (2.3)	4 (4.6)	1 (1.1)	7 (8.0)
2014	13					0	0	0
2015	39						2 (5.1)	2 (5.1)

Table E10. Number of steelhead, Chinook Salmon, and Bull Trout PIT-tagged in the Lemhi River basin outside of Kenney Creek that were subsequently detected on the PIT-tag array near the mouth of Kenney Creek (KEN).

Species	Number detected by year						Total
	2010	2011	2012	2013	2014	2015	
Steelhead	0	6	3	10	7	1	27
Chinook Salmon	0	0	0	0	0	1	1
Bull Trout	0	0	2	7	2	0	11

Table E11. Number of steelhead and Bull Trout PIT-tagged in Bohannon Creek that were subsequently detected on the PIT-tag array near the mouth of Bohannon Creek (BHC). Proportion of tagging cohort is shown in parentheses.

Tag year	Number tagged	Number detected by year					Total
		2012	2013	2014	2015	2016	
			Steelhead				
2010	51	0	0	0	0	0	0
2011	169	19 (11.2)	4 (2.4)	0	0	0	23 (13.6)
2012	131	0	5 (3.8)	3 (2.3)	0	0	8 (6.1)
2013	923		20 (2.2)	23 (2.5)	0	0	53 (5.7)
2014	337			21 (6.2)	21 (6.2)	3 (0.9)	45 (13.4)
2015	481				35 (7.3)	39 (8.1)	74 (15.4)
2016	1,307					50 (3.8)	50 (3.8)
			Bull Trout				
2010	1	0	0	0	0	0	0
2011	63	0	0	0	0	0	0
2012	38	0	0	0	0	0	0
2013	10		0	0	0	0	0
2014	12			0	0	0	0
2015	30				0	0	0
2016	17					1 (5.9)	1 (5.9)

Table E12. Number of steelhead, Chinook Salmon, and Bull Trout PIT-tagged in the Lemhi River basin outside of Bohannon Creek that were subsequently detected on the PIT-tag array near the mouth of Bohannon Creek (BHC).

Species	Number detected by year					Total
	2012	2013	2014	2015	2016	
Steelhead	0	1	7	7	10	25
Chinook Salmon	0	0	0	3	3	6
Bull Trout	0	0	0	1	1	2

Appendix F. Details of completed habitat restoration actions in the index drainages of the Potlatch River.

Table F1. Project type, location, completion year, and measure of implementation extent in the Big Bear Creek and East Fork Potlatch River index watersheds.

Restoration technique	Subwatershed	Tributary	Project end date	Stream treated (km)	Road treated (km)	Number of plants	LWD structures	Fencing (feet)
Barrier removal	Little Bear	WF Little Bear	2013	--	--	7,213	--	--
Barrier removal	Big Bear	Unnamed	2015	--	--	--	--	--
Barrier removal	Little Bear	WF Little Bear	2015	--	--	--	--	--
Barrier removal	Little Bear	WF Little Bear	2015	--	--	--	--	--
Barrier removal	Little Bear	WF Little Bear	2016	--	--	--	--	--
Barrier removal	Little Bear	WF Little Bear	2016	--	--	--	--	--
Barrier removal	Little Bear	WF Little Bear	2016	--	--	--	--	--
Barrier removal	EF Big Bear	Schwartz Creek	2013	--	--	--	--	--
Barrier removal	EF Big Bear	Schwartz Creek	2014	--	--	--	--	--
Barrier removal	EF Big Bear	Schwartz Creek	2013	--	--	--	--	--
Floodplain Connect	Big Bear Creek	MF Big Bear Cr	2016	--	--	--	--	--
Floodplain Connect	Big Bear Creek	MF Big Bear Cr	2016	--	--	11,416	--	--
Flow Supplementation	Little Bear	Spring Valley Cr	2015, 2016	16.00	--	--	--	--
Livestock BMPs	Big Bear	Big Bear Creek	2014	--	--	--	--	1,561
Meadow Restoration	Big Bear	WF Big Bear	2017	0.58	--	4,403	--	--
Meadow Restoration	Little Bear	Nora Creek	ongoing	--	--	--	--	--
Meadow Restoration	Little Bear	Nora Creek	ongoing	--	--	--	--	--
Riparian Plantings	Little Bear	Nora Creek	2009	--	--	1,420	--	--
Riparian Plantings	Dry Creek	Unnamed	2007	--	--	381	--	--
Riparian Plantings	Spring Valley	Unnamed	2007	--	--	158	--	--
Riparian Plantings	Dry Creek	Unnamed	2015	--	--	5,673	--	--
Riparian Plantings	Spring Valley	Unnamed	2012	--	--	458	--	--
Riparian Plantings	Big Bear	WF Big Bear Cr	2009	--	--	1,821	--	--
Riparian Plantings	Big Bear	Big Bear Creek	2009	--	--	598	--	--
Riparian Plantings	Big Bear	Unnamed	2009	--	--	2,360	--	--
Riparian Plantings	Spring Valley	Spring Valley Cr	2009	--	--	1,107	--	--
Riparian Plantings	Big Bear	Big Bear Creek	2015	--	--	2,400	--	--
Riparian Plantings	Dry Creek	Unnamed	2008	--	--	3,381	--	--

Riparian Plantings	Big Bear	Big Bear Creek	2014	--	--	1,198	--	--
Riparian Plantings	WF Little Bear	Unnamed	2013	--	--	2,466	--	--
Riparian Plantings	Big Bear	Unnamed	2014	--	--	6,127	--	--
Riparian Plantings	Big Bear	Unnamed	2013	--	--	1,077	--	--
Riparian Plantings	Big Bear	MF Big Bear Cr	2011	--	--	1,716	--	--
Riparian Plantings	Big Bear	Unnamed	2009	--	--	257	--	--
Road BMPs	EF Big Bear	Schwartz Creek	2016	--	1.93	--	--	--
Road BMPs	Big Bear Creek	Big Bear Creek	2010	--	1.61	--	--	--
	BBC Drainage Totals			***16.58***	***3.54***	***55,630***	***0***	***1,561***
Barrier removal	EF Potlatch	Bob's Creek	2015	--	--	--	--	--
Barrier removal	EF Potlatch	EF Potlatch R	2016	--	--	--	--	--
Barrier removal	EF Potlatch	EF Potlatch R	2015	--	--	--	--	--
Barrier removal	EF Potlatch	Jackson Creek	2014	--	--	--	--	--
Barrier removal	EF Potlatch	Jackson Creek	2014	--	--	--	--	--
Barrier removal	EF Potlatch	Pivash Creek	2015	--	--	--	--	--
Barrier removal	EF Potlatch	Purdue Creek	2016	--	--	--	--	--
Barrier removal	EF Potlatch	Purdue Creek	2016	--	--	--	--	--
Barrier removal	EF Potlatch	Rogers Creek	2015	--	--	--	--	--
Barrier removal	EF Potlatch	Unnamed	2016	--	--	--	--	--
Barrier removal	EF Potlatch	Bloom Creek	2015	--	--	--	--	--
Barrier removal	EF Potlatch	Fry Creek	2014	--	--	--	--	--
Barrier removal	EF Potlatch	Jackson Creek	2016	--	--	450	--	--
Barrier removal	EF Potlatch	Mallory Creek	2015	--	--	--	--	--
Barrier removal	EF Potlatch	Mallory Creek	2014	--	--	--	--	--
Barrier removal	EF Potlatch	Pivash Creek	2014	--	--	--	--	--
LWD/PALS/Riparian	EF Potlatch	EF Potlatch R	2012	1.33	--	3,800	150	5,000
LWD/PALS/Riparian	EF Potlatch	Bloom Creek	2015	0.75	--	600	24	--
LWD/PALS/Riparian	EF Potlatch	EF Potlatch R	2009	1.90	--	4,800	44	11,616
Riparian Plantings	EF Potlatch	Rogers Creek	2015	--	--	1,612	--	--
Road BMPs	EF Potlatch	Bloom Creek	2016	--	1.50	--	--	--
Road BMPs	EF Potlatch	Unnamed	2015	--	0.64	--	--	--
Road BMPs	EF Potlatch	Fry Creek	2015	--	1.30	--	--	--
Road BMPs	EF Potlatch	Fry Creek	2014	--	0.00	--	--	--
Road BMPs	EF Potlatch	Fry Creek	2015	--	2.57	--	--	--
Road BMPs	EF Potlatch	Jackson Creek	2014	--	0.40	--	--	--

Road BMPs	EF Potlatch	Jackson Creek	2016	--	1.61	--	--	--
Road BMPs	EF Potlatch	Purdue Creek	2015	--	0.53	--	--	--
Road BMPs	EF Potlatch	Purdue Creek	2015	--	0.48	--	--	--
Road BMPs	EF Potlatch	Purdue Creek	2015	--	2.57	--	--	--
Road BMPs	EF Potlatch	Ruby Creek	2015	--	1.29	--	--	--
Road BMPs	EF Potlatch	Bob's Creek	2014	--	1.37	--	--	--
Road BMPs	EF Potlatch	Bob's Creek	2014	--	3.01	--	--	--
Road BMPs	EF Potlatch	Unnamed	2014	--	2.41	--	--	--
Road BMPs	EF Potlatch	Jackson Creek	2016	--	1.61	--	--	--
Road BMPs	EF Potlatch	Jones Creek	2015	--	1.42	--	--	--
Road BMPs	EF Potlatch	Mallory Creek	2015	--	0.97	--	--	--
Road BMPs	EF Potlatch	Mallory Creek	2014	--	0.80	--	--	--
Road BMPs	EF Potlatch	Mallory Creek	2014	--	3.22	--	--	--
Road BMPs	EF Potlatch	Mallory Creek	ongoing	--	1.61	--	--	--
Road BMPs	EF Potlatch	Pivash Creek	2010	--	2.66	--	--	--
Road BMPs	EF Potlatch	Rogers Creek	2010	--	5.31	--	--	--
	East Fork Potlatch River Watershed Totals			***3.98***	***37.28***	***11,262***	***218***	***16,616***

Appendix G. Tagging for tributary survival and growth studies in Potlatch River index watersheds.

Table G1. The number of juvenile steelhead PIT tagged during roving tagging surveys in select tributaries in Potlatch River basin, 2008-2016.

	Tag Year								
Tributary	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**
Big Bear Creek	123	189	252	25	201	157	47	39	23
Little Bear Creek	113	341	298	383	408	219	229	311	446
WF Little Bear Creek	113	499	526	380	302	247	385	160	385
Pine Creek	285	613	-	410	-	259	203	242	308
EF Potlatch River	293	212	151	430	66	337	432	120	380

Appendix H. Detailed methodology of monitoring activities associated with the Spring Valley Creek flow supplementation study in 2015-2016.

Study Area

The treatment reach encompassed 8 km of Spring Valley Creek and 10 km of Little Bear Creek downstream of Spring Valley Reservoir (SVR; Figure 2.12). In 2015, 10 monitoring sites were established every 2 km downstream of SVR. Sites were divided into meadow and canyon reaches (SVC0-SVC8 and LBC10-LBC16, respectively). A reference site was established in an adjacent tributary, the West Fork of Little Bear Creek (WFLBC Control). In 2016, three additional reference sites (Upper, Middle, and Lower BBC Control) and two additional monitoring sites (SVC Inlet and LBC18) were added.

Water Releases

There were distinct release strategies conducted in 2015 and 2016 to determine the optimum flow release to supplement downstream flows while at the same time minimizing impacts to the reservoir. In 2015, releases occurred from August 3 to October 21 and did not begin until the downstream reach became intermittent. In 2015, water releases were 0.5 cfs for 3 weeks, increased to 1.0 cfs for 2-3 weeks, and reduced to 0.5 cfs for the remainder of the study period. We modified the release plan in 2016 to identify the minimum amount of water needed to provide perennial flows downstream of the reservoir. Water releases occurred from June 6 to October 10 and began before the test reach went intermittent. The onset of water releases in 2016 was determined by in-stream flows at two downstream monitoring sites (LBC8 and LBC10). From June 6 to August 1, flows were held at 0.2 cfs at LBC8 and LBC10 and from August 1-October 10 and then flows were dropped to maintain 0.1 cfs at LBC8 and LBC10.

Monitoring Sites

Stream Flow—At each site, a Decagon CTD-10 pressure sensor and Decagon EM50 data logger were installed to record change in water depth every 60 minutes. Each pressure sensor was placed in a perforated PVC protective housing and deployed into deepest portion of a pool. We downloaded the pressure sensors every few weeks using Decagon ECH2O software version 1.74 loaded onto a field laptop. The water depth was paired with site specific stream discharge data for a given day to build a rating curve to estimate flows at various water depths. A Marsh-McBirney model 2000 portable flow meter was used to measure and calculate stream velocities and discharge at each site. Horizontal distance, stream depth and water velocities were taken at a minimum 15 locations across the stream channel. The flow sensor was set at 60% total depth for thirty seconds and a velocity was recorded. Total discharge for each site was calculated in cubic feet per second (cfs) based on site area and velocity. Mean flows were estimated for each site for the duration of the project.

Temperature—We used the Decagon CTD-10 pressure sensor and Decagon EM50 data loggers to record stream temperature at all sites every 60 minutes. Each pressure sensor was placed in a perforated PVC protective housing and deployed into deepest portion of a pool. A temperature profile was built for each site showing change in water temperature over the duration of the project. The daily mean, minimum and maximum temperatures were computed for each site.

Dissolved Oxygen—Onset HOBO U-26 Dissolved Oxygen Loggers were placed at six locations and were set to record measurements every 30 minutes. In 2015, they were placed at

Spring Valley Reservoir (SVR), and at stream kilometers 2, 6, 8, 10 and the WFLBC control site. In 2016, we placed monitors at stream kilometers 2, 10, and 18 in the treatment reach and the one at the WFLBC control site and BBC control site. The loggers were downloaded every few weeks using Onset Hoboware pro software version 3.5.0 loaded onto a field laptop. Each logger was attached to a cement block and placed at the deepest portion of a pool. The logger's sensor was oriented with the flow and was cleaned weekly to avoid fouling.

Habitat Surveys

Habitat surveys were conducted from July 22 to September 23 and from July 18 to August 30 in 2015 and 2016, respectively. Habitat surveys were conducted at all sites during the same general time period when flows were stabilized. We conducted the surveys at various water releases to provide a wide variety of conditions to evaluate each release volume. All survey sites, except at LBC16, were 200 m in length. Survey site LBC16 was 120 m to minimize influence of hydrologic inputs from the WFLBC. Each site was divided into a 100-m section above and below the pressure sensor. We surveyed 1,520 m in 2015 and 1,720 m in 2016, roughly 10% of our study reach. Habitat surveys were also conducted at our reference sites in both years.

Habitat surveys consisted of measurements of total wetted length, large woody debris abundance, pool abundance and pool quality. We used measuring tapes to determine the total length of wetted stream sections within each transect. Large woody debris (LWD) was quantified using a standard stream assessment protocol (see habitat monitoring section). Pools were identified using the LWAP protocol (Bowersox et al. 2009, see main text).

Appendix I. Supplementary life cycle modeling results for the Big Bear Creek and East Fork Potlatch River modeling scenarios.

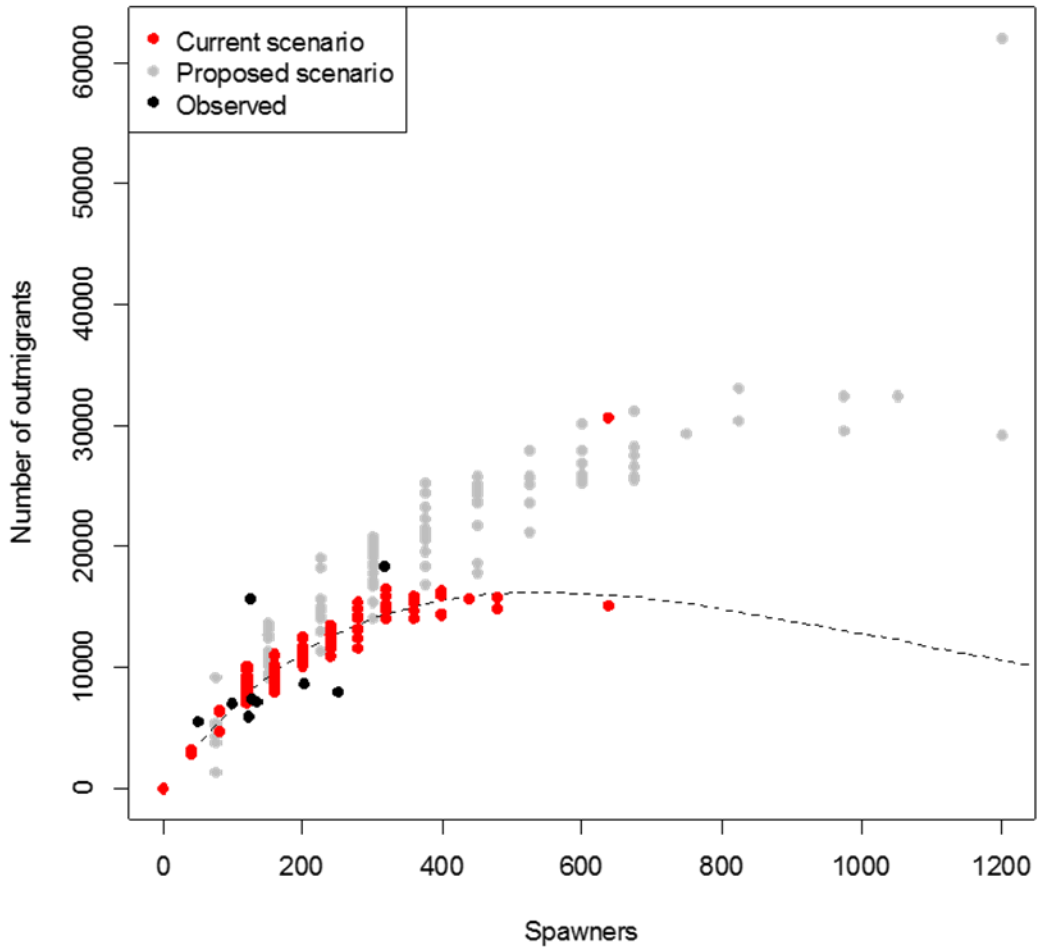

Figure I1. Observed and predicted (black points and dashed black line) spawner-recruit relationship for steelhead in Big Bear Creek watershed as well as the simulated relationship under the current status (red points) and the proposed restoration scenario (grey points).

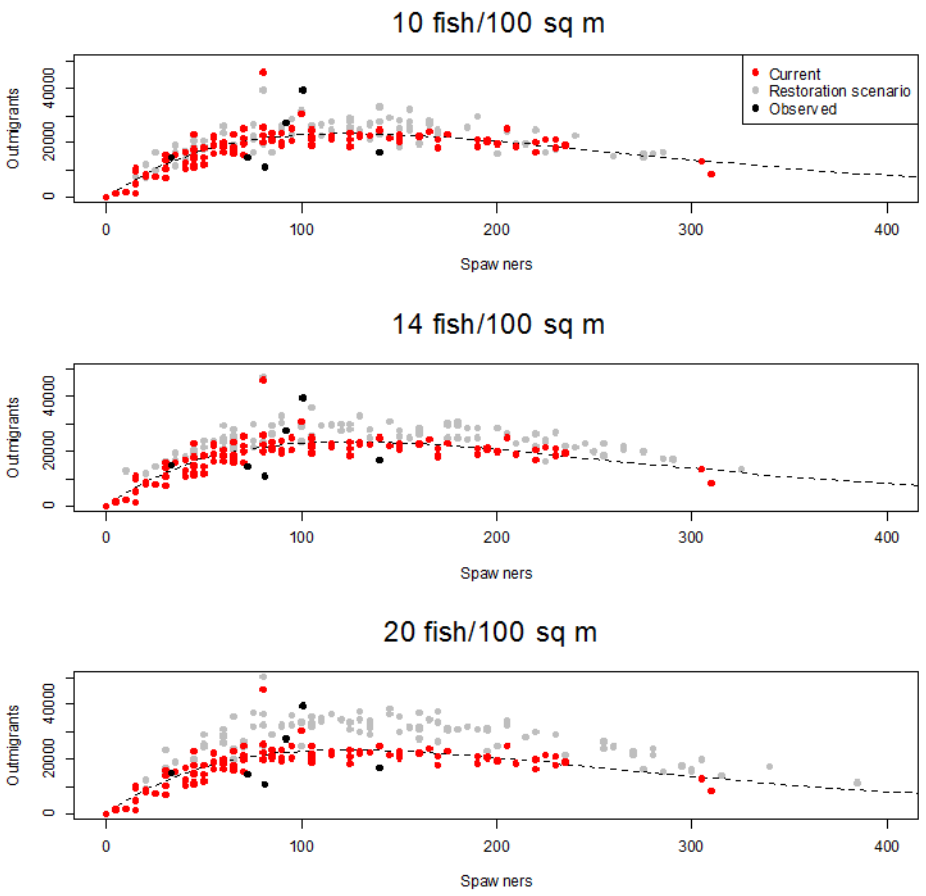

Figure I2. Observed and predicted (black points and dashed black line) spawner-recruit relationship for steelhead in the East Fork Potlatch River watershed as well as the simulated relationship under the current status (red points; all three panels) and under three proposed restoration scenario (grey points) when improvement in habitat only effects productivity of the stocks.

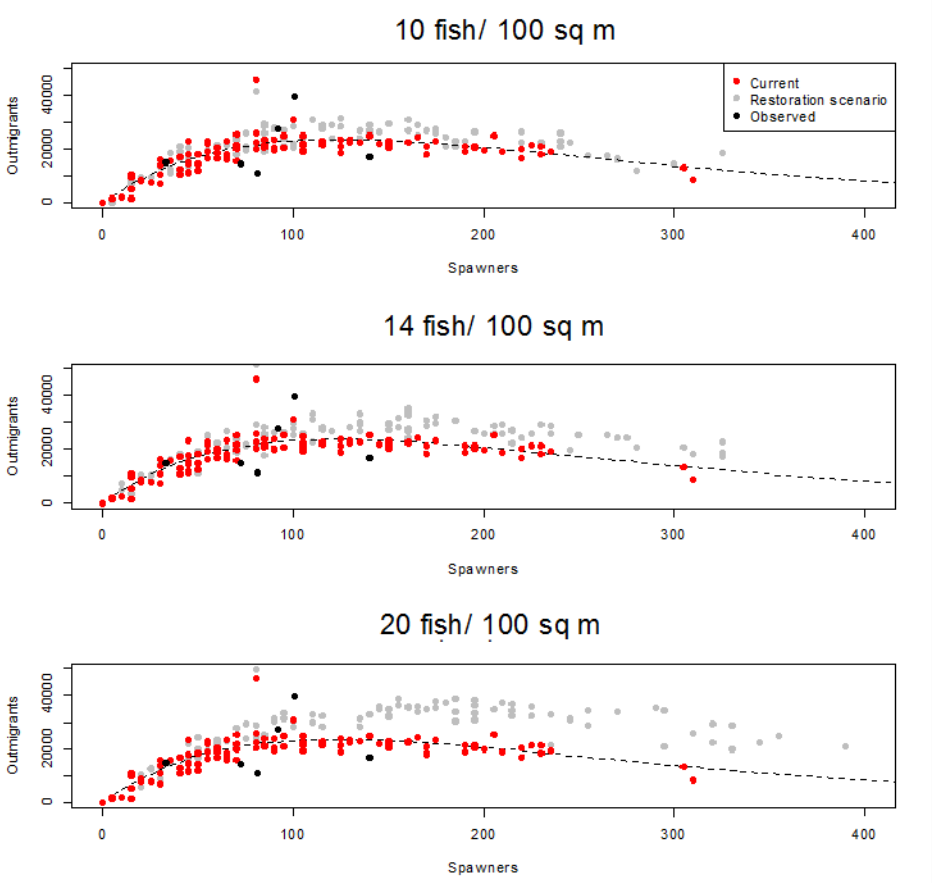

Figure I3. Observed and predicted (black points and dashed black line) spawner-recruit relationship for steelhead in the East Fork Potlatch River watershed as well as the simulated relationship under the current status (red points; all three panels) and under three proposed restoration scenario (grey points) when improvement in habitat only effects carrying capacity of the stocks.

Appendix J. Cost-benefit analysis for future restoration projects in the Potlatch River basin.

There is a need to develop a project prioritization framework for habitat restoration in the Potlatch River basin. Therefore, we developed a cost-benefit framework to evaluate future restoration projects in the basin and used it on a small, representative set of projects (Table J1). In the following example, we compared four projects from Table 2.2: 1) Big Bear Creek Falls barrier modification, 2) Spring Valley Creek flow supplementation, 3) Big Meadow Creek culvert barrier modification, and 4) East Fork Potlatch River LWD treatment. First, we used the life-cycle model results (see p. 87) to estimate the potential benefit in terms of additional smolts produced for each project. The East Fork Potlatch River LWD project was a single project treating 1 km of stream. The Big Bear Falls barrier modification project has the greatest potential impact and could increase smolt production >5,000 fish. In comparison, the East Fork Potlatch River LWD project would have the least potential impact and could potentially increase smolt production by 500 fish. Next, we compared the estimated cost of each of the projects. The Big Bear Falls project is the most expensive project at $1,000,000 and the Big Meadow Creek culvert project is the least expensive at $130,000. Finally, we calculated the cost per smolt for each project by dividing project cost by potential benefit. The Big Meadow Creek culvert project would be the most efficient project at $58.43 per additional smolt. The cost/benefit of the Big Bear Falls and Spring Valley Creek projects would be approximately $200.00 per smolt each. The East Fork Potlatch River LWD project would be the least efficient project at $566.00 per smolt. In terms of project prioritization, the Big Meadow Creek culvert project would generate the highest return on investment.

Table J3. Potential smolt increase, project cost, and estimated cost per smolt of four future restoration projects in the Potlatch River basin. BBC Falls- Big Bear Creek Falls barrier modification, SVC Flow- Spring Valley Creek flow supplementation, BMC Culvert- Big Meadow Creek barrier modification, EFPR LWD- East Fork Potlatch River LWD treatment.

Project	Potential smolt increase	Project Cost	Cost per smolt
BBC Falls	5,238	$1,000,000	$190.91
SVC Flow	2,779	$600,000	$215.91
BMC Culvert	2,225	$130,000	$58.43
EFPR LWD	507	$287,175	$566.64

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