Integrated Status and Effectiveness Monitoring Program (ISEMP) and Columbia Habitat Monitoring Program (CHaMP) 2016 Annual Combined Technical Report

BPA Projects ISEMP 2003-017-00 and CHaMP 2011-006-00

Report covers work performed under BPA contracts 46273, 71131, 71203, 71132, 73971, 71541, 71612, 71609, 71610, 71608

Report was completed under BPA contract # 73971

January 2016- December 2016

15th June, 2017

This report was funded by the Bonneville Power Administration (BPA), U.S. Department of Energy, as part of BPA's program to protect, mitigate, and enhance fish and wildlife affected by the development and operation of hydroelectric facilities on the Columbia River and its tributaries. The views in this report are the author's and do not necessarily represent the views of BPA.

This report should be cited as follows:

ISEMP/CHaMP. 2017. Integrated Status and Effectiveness Monitoring Program (ISEMP) and Columbia Habitat Monitoring Program (CHaMP) Annual Combined Technical Report, January – December 2016, BPA Projects 2003-017-00 and 2011-006-00, 93 Electronic Pages.

Table of Contents

List of Figures

List of Tables

Introduction

This report presents an update on the implementation of Bonneville Power Administration's (BPA) Integrated Status and Effectiveness Monitoring Program (ISEMP; BPA Project 2003-017-00) and the Columbia Habitat Monitoring Program (CHaMP; BPA Project 2011-006-00) during 2016. The work conducted under ISEMP and CHaMP covers key populations within the Upper Columbia River Spring-Run Chinook ESU and steelhead DPS, the Snake River Spring/Summer-Run Chinook ESU and steelhead DPS, and the Middle Columbia River Steelhead DPS.

Background

BPA is working with U.S. National Oceanic and Atmospheric Administration Fisheries (NOAA-Fisheries) and other regional fish management agencies to monitor status and trends of fish habitat for each major population group (MPG) in the Pacific Northwest identified through the Endangered Species Act (ESA) in support of habitat restoration, rehabilitation and conservation action performance assessments and adaptive management requirements of the 2008 Federal Columbia River Power System Biological Opinion (FCRPS BiOp). BPA created ISEMP and CHaMP as part of this effort.

Since its inception in FY2003, ISEMP has developed two monitoring and evaluation programs: (1) subbasin-scale status and trend monitoring efforts for anadromous salmonids and their habitat, and (2) effectiveness monitoring for suites of habitat restoration projects in selected watersheds. This work is critical for the development of a federal research, monitoring, and evaluation program for the Columbia River Basin, which is now a requirement under the NOAA-Fisheries 2000, 2004 and 2008 BiOps. In 2008, ISEMP became a component of BPA's BiOp Program to help ensure that provisions of the BiOp are satisfied.

CHaMP was first proposed in 2010 for implementation in 26 Columbia River Basin watersheds. Watersheds originally considered for CHaMP were prioritized due to several conditions, including (1) gaps in habitat monitoring for ESA-listed steelhead and/or Chinook populations; (2) offering maximum contrast in current habitat conditions; (3) provided reasonable opportunities for future restoration actions or monitoring current implementation activities; and (4) provided feasible opportunities for piloting the CHaMP protocol. The initial watersheds of interest included Wind River, Toppenish, Klickitat, Fifteen Mile, John Day (Lower Mainstem, North Fork, Upper Mainstem/South Fork, and Middle Fork John Day), Umatilla, Grande Ronde (Upper and Lower and Catherine Creek), Imnaha, Lolo, Lochsa, Tucannon, Asotin, South Fork Salmon, Big Creek, Lemhi, Pahsimeroi, Wenatchee, Entiat, Methow, and Okanogan. However, as a result of scientific and policy level reviews of the program by the Independent Science Review Panel (ISRP), Northwest Power and Conservation Council (NPCC), Bonneville Power Administration (BPA) and others, CHaMP was implemented in 2011 as a pilot project in eight Columbia River Basin watersheds (Methow, Entiat, Wenatchee, John Day, South Fork Salmon, Tucannon, Upper Grande Ronde and the Lemhi). Other agencies are also implementing CHaMP under different funding structures, including Oregon Department of Fish and Wildlife, California Department of Fish and Game, Confederated Tribes of the Umatilla Indian Reservation and the Columbia River Inter-Tribal Fish Commission.

ISEMP and CHaMP's Goal

Monitoring programs for salmonids and their habitat provide needed data to develop watershed assessments to inform management strategies. Other data from GIS and remote sensing can be also be used to complement these monitoring efforts to produce spatially explicit and continuous habitat information across the watershed (Wheaton et al. 2017). Useful watershed assessments quantify the long-term biological benefit stream habitat quality and quantity (HQQ) have on fish population processes (e.g., capacity, growth potential, survival). Current and potential or historic habitat condition bound the range of feasible habitat rehabilitation scenarios. The impacts these scenarios have on fish population processes and the resulting long-term effects on the population trajectory can be assessed through various quantitative frameworks (McHugh et al. 2017). Biological relevance should be determined by science-based fish-habitat relationships with a documented and understandable origin and specific to species, life-stage and season. The overall or relative importance of any ecological impairments identified in this process should be assessed through impacts to long-term (i.e., at least 50 years) population persistence (e.g., Pqe, VSP, PVA).

This is the type of watershed assessment that ISEMP and CHaMP was charged with developing to evaluate the potential effects of tributary habitat rehabilitation scenarios at the population scale, and with this goal we have developed several technical elements required for this approach (Table 1). These components are the tools by which watershed condition is assessed relative to its physical and biological potential, by which a suite of rehabilitation actions are evaluated as appropriate and their magnitude of habitat change estimated, and finally, by which a long-term population-level effect of an implementation strategy is projected (Table 1). In this report we provide a summary of (1) field data, both status and trend and effectiveness monitoring collected through 2016 under ISEMP and CHaMP; (2) the effectiveness of habitat restoration actions in three Intensively Monitored Watersheds (IMWs) in Washington, Idaho and Oregon; and (3) a review of tools and products built off the data collected under these monitoring efforts that are designed to support useful regional watershed assessments. Attachment A provides a compilation of quick facts to provide the reader with an overview of the scope of monitoring, tools and products developed by ISEMP and CHaMP.

Table 1. Summary of components and tools developed by ISEMP and CHaMP that make up a useful watershed assessment approach. VBET = Valley Bottom Extraction Tool; GNAT = Geomorphic Network and Analysis Toolbox; GUT= Geomorphic Unit Tool; RCAT = Riparian Condition Assessment Tool; RVD = Riparian Vegetation Departure; BRAT = Beaver Restoration Assessment Tool; WRAT = Wood Recruitment Assessment Tool; NRE = Net Rate of Energy Intake; QRF = Quantile Regression Forest; HSI = Habitat Suitability Model; FIS = Fuzzy Inference System; DEM = Digital Elevation Model; GPP = Gross Primary Production; BDA = Beaver Dam Analogs; PALS = Post-Assisted Log Structures; LCM = Life Cycle Model.

Goal	Component	Physical Processes	Technical Element
Describe the geomorphic context of each watershed to support the development of fish-habitat relationships and rehabilitation potential in a spatially explicit manner at the appropriate spatial and temporal grain (e.g., reach by season for fish-habitat relationships and habitat unit by season for rehabilitation actions).	Describe physical processes at multiple scales (reach to watershed) captured in a quantitative description of past and present HQQ.	Planform and longitudinal complexity	VBET, GNAT, GUT, Geomorphic Assessment Toolbar
		Sinuosity	GNAT, Geomorphic Assessment Toolbar
		Confinement and valley bottom or floodplain extent	VBET, Confinement Tool
		Stream power	GNAT, Geomorphic Assessment Toolbar
		Substrate composition	CHaMP samples and regional extrapolation
		Flow and water temperature	Hydraulic models, MODIS temperature models

Goal	Component	Physical Processes	Technical Element
		Reach type and condition variant	Geomorphic Assessment Toolbar
Describe the biological setting of each watershed to support the development of fish-habitat relationships and rehabilitation potential in a spatially explicit manner at the appropriate spatial and temporal grain (e.g., reach by season for fish-habitat relationships and habitat unit by season for rehabilitation actions).	Biological processes at reach to watershed scale captured in quantitative description of past and present HQQ	Gross Primary Productivity	MODIS temperature models, Conductivity Tool, Solar Exposure Tool
		Riparian vegetation	RCA, RVD
		Upland vegetation	BRAT, WRAT
Quantify the biological value of stream habitat features to rearing, holding and spawning juvenile and adult salmonids based on specific components of habitat quality and quantity. Estimate the reach-scale value of stream habitat for fish population processes (e.g., capacity, growth potential, movement) across river networks to predict the distribution of habitat conditions and to quantify the extent and location of habitat conditions based on rehabilitation action plans.	Fish-habitat relationships at reach to watershed scale captured in quantitative tool to project population benefit of current and modified HQQ based on rehabilitation action scenarios	Juvenile summer / fall capacity and growth potential	NREI
		Juvenile summer or winter rearing capacity, spawner capacity	QRF, HSI/FIS
		Extension of reach-scale fish-habitat relationships to watershed scale	Upscaling models
Quantify restoration modalities by the magnitude, extent and timing of change possible in stream habitat quality and quantity.	Restoration modalities at reach to watershed scale captured in quantitative tool to modify HQQ. Each restoration modality changes a different suite of habitat features that in turn impact the value of the habitat to fish.	In channel complexity – ELJ, wood and boulder placement, beaver dams	DEM to NREI, QRF, HSI, FIS

Goal	Component	Physical Processes	Technical Element
		Channel planform complexity – re-meander and side channel construction	DEM to NREI, QRF, HSI, FIS
		Riparian vegetation planting and management	RCA, RDV to GPP and NREI
		Access to previously blocked habitat	QRF, HSI, FIS
		BDA and PALS	BRAT and DEM to NREI, HSI FIS
		Temperature	NREI
Develop watershed-scale tributary habitat rehabilitation scenarios in a standardized manner to facilitate communication and to limit the range of options to be evaluated.	The standardization is achieved by using a suite of watershed assessment tools that identify the appropriate action types and locations. The magnitude of impact on the value of the habitat change to fish population processes is constrained by the application of a standardized set of "effect sizes" by rehabilitation action modality. The effect size of each rehabilitation action type specifies the magnitude of fish habitat value change and the expected time course and duration of change. Standardized approach to developing rehabilitation scenarios – spatially explicit actions, magnitude and timing of change in HQQ		LCM

Methods

ISEMP and CHaMP were designed to meet critically important features of a regional RME program: produce fish and habitat monitoring data with accompanying accuracy and precision assessments, implement all monitoring using sample designs constructed to meet explicit objectives, and manage, store and disseminate data through regional data management systems (PIT tag data in PTAGIS and habitat data in CHaMPmonitoring.org). In addition, we have developed tools to synthesize habitat metrics to describe habitat quality and quantity, contextualize this information across the watershed network, and ultimately describe the implications to salmon and steelhead populations to inform management strategies. We use probabilistic sampling designs such as the Generalized Random Tessellation Stratified (GRTS) sample selection algorithm (Stevens and Olsen 1999) for fish and habitat status and trend monitoring to limit spatial autocorrelation (all sites are considered independent samples and can be used to estimate metrics for entire watersheds), and Before-After-Control-Impact (BACI) experimental designs in a hierarchical framework (Walters et al. 1988, Loughin 2006, Loughin et al. 2007) for effectiveness monitoring. Past annual reports describe these methods in detail and are available for download from the ISEMP website (isemp.org).

Habitat Status and Trends Monitoring

The CHaMP protocol (https://isemp.egnyte.com/dl/OZyNX2p8LA) calls for field data collection during the low-flow period, typically from June through October. The basic CHaMP GRTS study design is a rotating panel design that consists of four panels: one panel of 15 sites that are visited annually, and three rotating panels of 10 sites each that are visited on a 3-year cycle. Sampling within this framework is implemented to achieve a balance between status estimation and trend detection, with a total of 45 sites visited in each subbasin every 3 years for a minimum of 9 years (3 cycles of rotating panels, as per Urquhart and Kincaid 1999) (https://isemp.egnyte.com/dl/qnlwI3S5PU). However, the design can and has been modified to address more specific questions for watershed if needed. For example, in the Tucannon, requests were made to evaluate the habitat response to targeted restoration. A hybrid design was created where treatment and control reaches in the mainstem were selected along with GRTS sites in the tributaries. The response design consists of topographic surveys which produce Digital Elevation Models (DEMs) of the channel morphology along with other important auxiliary fish habitat information that is collected.

Fish Status and Trends Monitoring

We track the status and trends of adult escapement and juvenile abundance under ISEMP in the Entiat, Secesh and Lemhi River subbasins using three methods: (1) adult PIT tagging and interrogation; (2) operation of rotary screw traps; and (3) remote-site fish surveys. Within the Entiat River status and trends frame, sites are randomly selected using a GRTS design (Stevens and Olsen 1999) across the entire anadromous range to enable the extrapolation of site-based surveys to the population scale. In the Lemhi River we use spatially continuous juvenile survey efforts to address patchy distribution and overall low abundance of target species and the physical difficulty of site-based surveys. These spatially continuous survey efforts still maintain individual GRTS sampling locations, enabling backwards compatibility with legacy sites. For both approaches, seasonal or annual surveys are employed to either recapture individuals to support growth rate analysis or passively “re-sight” PIT-tagged individuals (e.g., using floating or backpack PIT tag detectors) to establish seasonal habitat preferences. A broad network of instream PIT tag detection arrays passively monitor for movement of PIT-tagged fish year round throughout the Columbia Basin, with a concentration of ISEMP-operated arrays within target watersheds.

Adult Escapement

We have developed a hierarchical patch-occupancy model (State-Space Adult Dam Escapement [STADEM] and Dam Adult Branch Occupancy [DABOM] models) to estimate escapement to various populations in the Snake River basin for return years 2010 – 2016 using observations of returning adult spring/summer Chinook and steelhead that were PIT tagged at Lower Granite Dam. This model simultaneously estimates the probability of movement along the stream network and the probability of detection at various PIT tag observation sites. Estimates are adjusted post-hoc by applying a correction factor for differential trap rates throughout the run season and all estimates have a measure of precision (confidence intervals).

Juvenile Abundance

We use two methods for the purposes of assessing status and developing trends in juvenile abundance: rotary screw traps (RSTs) and remote-site juvenile surveys. RSTs are generally used to estimate the abundance of emigrating fry, parr, presmolt, and smolt at subpopulation and population spatial extents. We employ remote-site juvenile survey efforts as a means to estimate juvenile density, abundance, residence time, movement, growth, survival and habitat use prior to emigration, often in treatment versus reference locations. Either method can also be used to evaluate the effectiveness of habitat restoration actions at the subpopulation or population level. Estimates of abundance at the population scale are generated using a GRTS roll-up or a Darroch estimator for spatially continuous surveys.

Rotary Screw Traps

In the Entiat River subbasin, BPA funds the U.S. Fish and Wildlife Service (USFWS) through ISEMP to operate a RST March through November near the mouth of the Entiat River. The RST provides information about spring Chinook and steelhead production and life history characteristics. Measured or estimated parameters include outmigrant abundance, emigration timing, production (smolt/spawner), survival (parr to smolt and smolt to adult return rate [SAR]), genetic and age structure, length frequency distribution, and growth. Efficiency estimation and life-stage designations are standardized across the Upper Columbia and follow Washington Department of Fish and Wildlife (WFW) guidance.

In the Salmon River subbasin BPA fully funds the operation of a RST on the lower mainstem Secesh River and the lower mainstem Lemhi River. ISEMP collaborates with, and partially funds, the operation of RSTs in Hayden Creek and on the upper mainstem Lemhi River, upstream of the confluence with Hayden Creek with the Idaho Department of Fish and Game (IDFG) and Idaho Office of Species Conservation. These RSTs are strategically located to enable an assessment of freshwater productivity (juvenile production as a function of adult escapement) within treated and reference areas of the Lemhi River, and for the population as a whole via emigration estimates at the lower mainstem Lemhi River RST. Virtually all RSTs operated in the Snake River Basin use methods described in Steinhorst et al. (2004) to estimate RST efficiency. Fry, parr, presmolt, and smolt designations are likewise standardized by date-of-capture. We adopted these regional standards to maintain consistency with other RSTs operated in the Snake

River Basin in order to capitalize on those estimates as large-scale references for ISEMP estimates from the Secesh and Lemhi rivers.

Remote-Site Juvenile Surveys

Some remote-site juvenile surveys are randomly distributed at a broad geographic scale using GRTS to estimate overall population status and trends independent of restoration actions (Entiat), whereas other surveys are targeted at areas subject to restoration actions with control areas for comparison in evaluating response (Lemhi). A majority of remote fish survey sites are paired with CHaMP habitat surveys for the development of fish-habitat relationships and modeling of predicted fish response to restoration actions. We use a wide range of capture methodologies, including electrofishing, seining, nocturnal stalk netting, and trapping and angling to efficiently capture juvenile salmonids. We identify, enumerate, measure, and weigh all captured juveniles, and target species in good physical condition are typically fitted with PIT tags or marked by some other method, often as part of a discrete mark-recapture experiment but also to allow long-term tracking of movement, survival and growth. We may collect tissue samples for subsequent genetic analysis or aging.

Effectiveness Monitoring

ISEMP continued implementing an Intensively Monitored Watershed (IMW) in Bridge Creek in the John Day subbasin in Oregon, in the Entiat River in the Upper Columbia subbasin in Washington, and in the Lemhi in the Salmon River subbasin in Idaho in 2016. All three IMWs employ a BACI design in a hierarchical framework (Walters et al. 1988, Loughin 2006, Loughin et al. 2007), use the CHaMP protocol for habitat monitoring and PIT tagging for fish population monitoring. Detailed descriptions of the study designs and analyses for each IMW are available in ISEMP annual reports 2006 – 2013, ISEMP/CHaMP combined annual reports 2014 – 2015.

Bridge Creek IMW, OR

Bridge Creek is a 710 km^2 watershed that drains directly into the lower John Day River in the semi-arid region of the Columbia River Basin (Figure 1). Much of the lower valley of Bridge Creek suffers from channel incision, a common impairment among streams throughout the western United States, where much of the channel on Bridge Creek currently exists in a highly simplified and degraded state with a vastly reduced abundance and extent of riparian vegetation. The Bridge Creek watershed is used by a run of Mid-Columbia steelhead that are part of the ecologically distinct and threatened Lower John Day population, and is also used intermittently by Mid-Columbia Chinook salmon (Pollock et al. 2007). We hypothesized that encouraging beavers to build dams by installing structures upon which they could construct dams would have a host of physical effects, such as increasing sediment retention, aggrading the stream bottom causing a net aggradation effect and lowering summer water temperatures, that would result in increases in juvenile steelhead abundance, growth, survival and productivity.

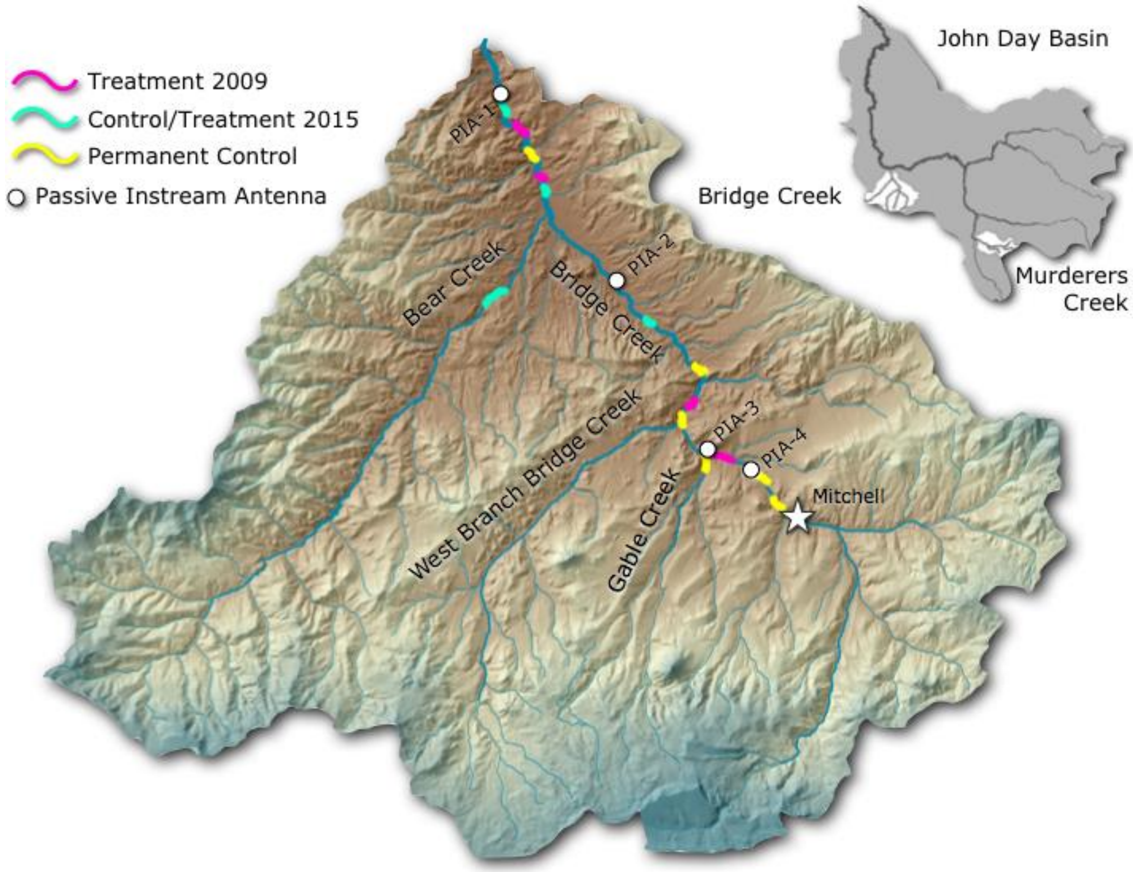

Figure 1. Location of the Bridge Creek Intensively Monitored Watershed and its external control, Murderers Creek, in the John Day subbasin, Oregon.

Lemhi IMW, ID

In the Lemhi IMW local co-managers recognized that insufficient instream flow, loss of access to historically important tributary habitat, and mainstem habitat simplification were primary limiting factors for Chinook and steelhead productivity. Tributary reconnections have been achieved through replacing tributary diversions with mainstem diversions, enabling the reconnection of tributaries, reducing total withdrawals, and allowing cooler tributary water to enter the mainstem Lemhi River. Simultaneously, restoration actions have addressed tributary passage impediments and improved habitat conditions within tributaries, providing access to relatively intact publicly held land. A number of mainstem habitat restoration actions were identified to improve habitat complexity and provide access to off-channel habitat. Beginning in 2009, ISEMP has been collecting monitoring data (Figure 2) to parameterize a life cycle model (LCM) to help guide future restoration actions.

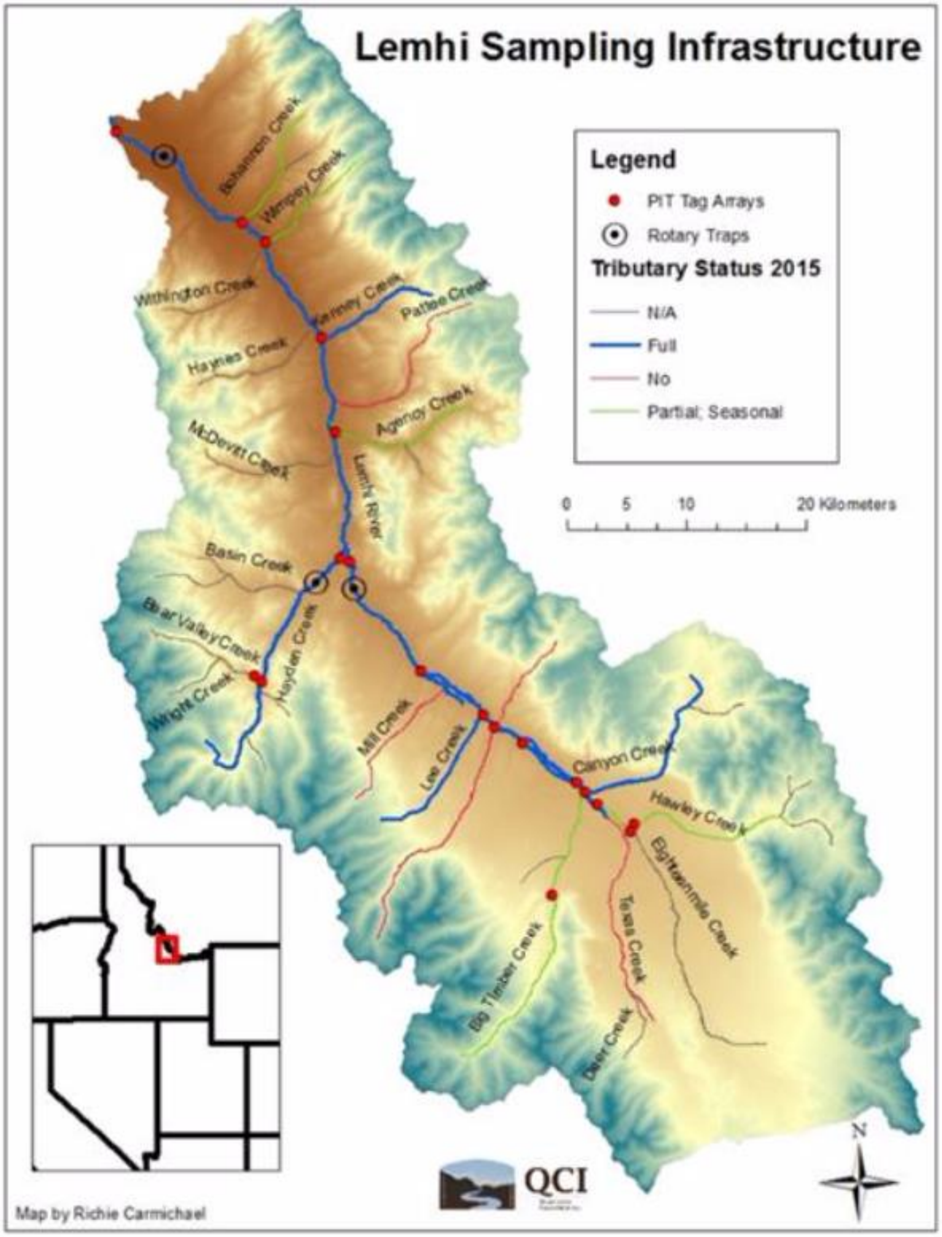

Figure 2. Lemhi River IMW sampling infrastructure.

ISEMP also monitors four individual site-based restoration sites within the Lemhi River basin: Amonson Creek, Lee Creek, Little Springs and Eagle Valley Ranch (Figure 3). The habitat monitoring has been conducted opportunistically so each site has a slightly different sample design. Habitat surveys follow the CHaMP protocol except that most surveys have site length determined by project size rather than the bankfull width per the CHaMP protocol. Analyses have included creating DEMs of difference to show changes in stream topography post-restoration and predicting changes in carrying capacity post-restoration using a Quantile Regression Forest (QRF) model.

Figure 3. Lemhi River Basin, Idaho. Red dots indicate the four sample restoration reaches.

Amonson Reach

In 2011 and 2014 IDFG worked with cooperative landowners to add complexity and restore a side channel to the Amonson Reach of the Lemhi River. ISEMP surveyed portions of the Amonson site and an untreated section of the mainstem Lemhi as representative of the reach pre-treatment in 2012 and 2016. CHaMP crews surveyed the entirety of the side channel in 2014 and will re-survey it in 2017.

Lee Creek

Prior to restoration, Lee Creek had been diverted into a ditch along the road, where it then crossed under the highway before flowing into the Lemhi River. In 2012, the Nature Conservancy and partners re-engineered the furthest downstream kilometer of Lee Creek in conjunction with restoring minimum flows year-round to ensure fish passage. A CHaMP survey was performed several days after the newly engineered project was completed in 2013, and again in 2016.

Little Springs Creek

In 2012, IDFG re-engineered Little Springs to set it back from the road so that it is no longer laterally confined and connected flow back to the Lemhi River. Restoration included adding meanders to increase sinuosity and large woody debris structures to increase cover and encourage scouring. CHaMP surveys were performed on the rehabilitated section of river in 2012, 2013, and 2016.

Eagle Valley Ranch

The goal of the Eagle Valley Ranch project is to encourage juvenile Chinook to rear in the Lemhi River for longer periods to increase downstream survival. Much of the lower Lemhi River lacks slow, backwater, and braided habitat types so a new side channel was constructed in 2016 along the Lemhi River, about 16 km upstream of its confluence with the Salmon River as part of a larger four phase project to add quality habitat to approximately 5 km of the lower Lemhi River. A CHaMP crew collected auxiliary data on the main channel in 2015 and we will leverage available engineering and LiDAR data sets as part of the effectiveness analysis.

Entiat IMW, WA

An engineered approach is being taken to address limited instream complexity for listed spring Chinook and steelhead on the mainstem of the Entiat River by adding rocks and wood to the river and breaching levees to reconnect the floodplain where possible. Two of four planned rounds of habitat actions have been implemented so far, affecting about 14% of the targeted stretch of river. Treatments are stratified by geomorphically distinct valley segments and geomorphic reaches. The Mad River is not targeted for habitat restoration and is the internal control (Figure 4). We have hypothesized that increasing instream complexity will result in increases in density, growth rates, survival and productivity of juvenile salmonids.

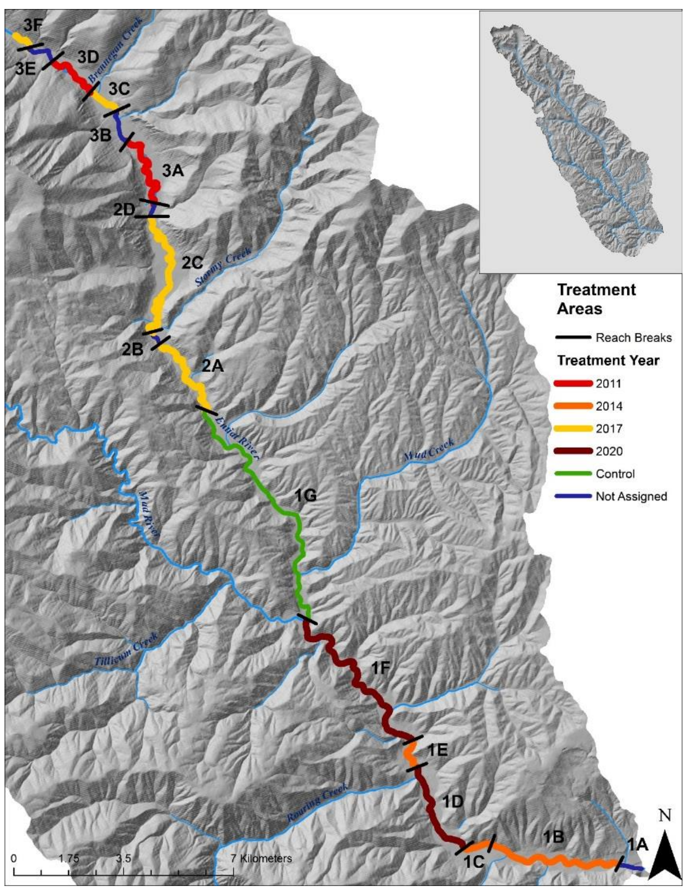

Figure 4. Location and timing of restoration actions in the Entiat River IMW.

We evaluated restoration effectiveness in the Entiat IMW using a BACI-style analyses with abundance, survival, growth, and fork length as restoration responses. We considered three phases of restoration: “Before” from 2010 – 2012, “Restoration 1” from 2013 – 2014, and “Restoration 2” from 2015 – 2016. All responses were evaluated at the valley segment and watershed level. Abundance was additionally evaluated at the reach level. Restoration was considered to affect the reach, valley segment, or watershed it took place in, depending on the analysis spatial scale. Abundance was scaled up to the valley segment and watershed level using GRTS design weights.

Ecohydraulic Models

The ecohydraulic models (i.e., QRF, Net Rate of Energy Intake [NREI], Habitat Suitability Index [HSI]) and Fuzzy Inference Systems [FIS]) transform field-based metrics into habitat capacity estimates that support restoration planning, project prioritization, and fuel realistic LCM input. We use these models to inform habitat condition maps, estimate juvenile carrying capacities, and evaluate alternative habitat scenarios (e.g., changes in temperature or topography). HSI and FIS models are used to estimate spawner capacity. For more details on the ecohydraulic models see ISEMP (2012, 2013, 2014), ISEMP/CHaMP (2015, 2016) and Attachment A.

HSI/FIS

The HSI model provides a spatially explicit depiction of the quality of spawning habitat within modeled reaches. Its primary inputs are depth, velocity and field measurements of substrate size (i.e., gravel, cobble, etc.). HSI models do not include variables such as temperature and food availability; however, spawning salmon do not feed while occupying redds, so HSI models can provide accurate predictions of potential redd locations. These data, in conjunction with the steelhead spawning habitat suitability criteria used by Maret et al. (2005) are used to compute a spawning HSI score for every 10-cm raster cell within a survey reach. Scores are then combined into a composite HSI score, and translated into a reach-scale estimate of available spawning habitat, weighted by its suitability. HSI models have also been criticized for being site-specific, making extrapolation to other locations unreliable; a more robust approach is to use more generalized habitat suitability curves using FIS. FIS are intuitive, flexible in adjusting model parameters and variables, are more robust with imprecise data, can incorporate expert knowledge, and can represent more complex multivariate relationships than traditional HSI models. When combined with high resolution hydraulic model outputs, FIS-based ecohydraulic models also provide a spatially explicit depiction of habitat suitability and an estimate of wetted usuable area (WUA), which can be used to estimate carrying capacity as described for the traditional HSI models. The current spawner FIS models use depth, velocity, substrate, and fish cover elements (e.g. LWD, undercuts, and deep pool distance) to make spatially explicit carrying capacity estimates throughout a reach.

NREI

The NREI model consists of two sub-models, a foraging model and a bioenergetics model which, given information about ambient food availability (i.e., invertebrate drift), water temperature, hydraulic conditions (depth and velocity, output from hydraulic model), and an average fish size, provide spatially explicit predictions of the energy costs (swimming costs) and benefits (gross energy intake) associated with occupying different locations in survey reaches. These predictions are then translated into an estimate of juvenile rearing capacity using a fish placement algorithm and a minimum NREI threshold (e.g., 40% of maximum consumption).

QRF

Quantile regression forest models have been developed to predict site-level carrying capacity based on empirical fish/habitat relationships. Using paired habitat and fish (parr or redd abundance) data, we have identified measured habitat characteristics (covariates) that are most highly associated with observed juvenile parr densities and observed redd abundance, and used them as covariates in a QRF model with fish densities as the response. Random forest models naturally incorporate interactions between correlated covariates, a common occurrence among stream habitat metrics, and accommodate potentially non-linear fish/habitat relationships. These QRF models predict a range of fish densities for any given CHaMP site, and we have chosen the 90th quantile of this range as a proxy for carrying capacity. We have developed QRF models to predict summer parr capacity and redd capacity for spring/summer Chinook salmon. We are working to develop similar QRF models for steelhead as well as winter parr capacity models for both species.

Upscaling Methodologies

Data collected from reach-scale surveys such as CHaMP are information rich with high resolution; however, with site lengths of 20 bankfull widths and 25 sites per year (49 within a full panel of sites), these surveys only cover a small percentage of the watershed. Site-level surveys need to be put into the context of the entire stream network of the watershed and this can be done using various techniques to upscale data from sites to reaches throughout the watershed. This process provides spatially explicit estimates of habitat metrics and capacity estimates that can be used in conjunction with watershed context detail to inform current and potential habitat, restoration potential, and appropriate restoration techniques.

Design-based watershed estimation

At the watershed level, or at spatial levels for which a reasonably high number of representative reach-level estimates exist (minimum 20+ sites), ISEMP is using design-based estimation tools to efficiently estimate population distributions for habitat and fish metrics. We use the GRTS-based sample design and the spsurvey package (in the R programming language) to make inference on CHaMP data at a watershed scale (Nahorniak et al. 2015). We roll up estimates of fish carrying capacity based on CHaMP reach-scale models (e.g., NREI and QRF) in the same fashion to derive watershed-level estimates of carrying capacity.

Empirical modeling-based estimation and network extrapolation

When spatial scales of interest consist of less than 20 or so directly measured sites, standard design-based estimators may not provide sufficient precision. ISEMP has developed empirically derived models to relate globally available information (e.g., geospatial attributes from GIS layers) to measured CHaMP metrics in a spatially explicit manner. These empirical models are used to relate measured CHaMP attributes to these globally available attributes (GAAs), and thus estimate CHaMP metrics at unmeasured reaches within watersheds, or into watersheds for which no habitat data exists. Inverse probability bootstrapping is used to properly account for sampling design while using model-based statistical techniques (Nahorniak et al. 2015). Caution must be exercised when extrapolating models into unsampled watersheds, as we are assuming that the empirical relationships observed are constant within and external to our CHaMP watersheds. In general, this assumption may not be true, but the more our empirical relationships describe spatially constant underlying physical laws, the less risk there is in this assumption.

We are also able to generate spatially continuous estimates from the empirical models by using the modeled estimates to fill in gaps between measured sites and create maps explicitly showing the estimated spatial distribution of CHaMP metrics. Note that the empirical models used for continuous estimation may be optimized differently than the empirical models used for watershed-level extrapolation, as they may be watershed specific, and may take advantage of spatial autocorrelation not present at the watershed level.

Processed-based ecohydraulic models

To identify habitat impairments and plan and test for appropriate restoration strategies to benefit populations, we believe estimates of carrying capacity at reach-scale resolution at the extent of the network will be most informative. While correlative approaches may have the ability to create continuous estimates of carrying capacity, the ability to manipulate covariate values to reflect changes due to restoration may not be possible, or at least uncertain in their prediction. To protect against spurious correlations, increase accuracy, and target variables subject to restoration, a processed-based understanding of how the network-scale variables influence carrying capacity is recommended. The ecohydraulic models described above (i.e., NREI, HSI, FIS) can help guide the development of network-level models that inform the relevant reach-scale metrics defining fish habitat.

For example, the general inputs for NREI can be summarized into inputs of food, temperature, and channel morphology and substrate as it pertains to hydraulics. The goal would be to create network extent of these metrics resolved at the reach scale (Figure 5). At CHaMP sites, temperature is collected from temperature loggers and summarized over various temporal scales (e.g., mean daily temperature). McNyset et al. (2015) demonstrated that MODUS satellite information of ground temperature taken daily is highly correlated with data logger temperature at CHaMP sites, and that this information can be extrapolated across the network with a high accuracy. Similarly, gross primary production (GPP, a surrogate for food resources for fish) estimated from dissolved oxygen sondes could be predicted by CHaMP-level metrics of solar input, conductivity, and temperature, and these same metrics derived by network-level models with similar accuracy and precision. Finally, our investigations with ecohydraulic models at CHaMP sites suggest that the hydraulic patterns such as shear zones, found near geomorphic unit transitions, are more important than geomorphic unit themselves in determining habitat quality. We are currently refining the relationship between hydraulic patterns, NREI values, and geomorphic unit assemblages to better capture this complexity rather than relying simply on metrics such as pool frequency or pool area.

We are using geomorphic assessments to describe what geomorphic reach type should be observed based on higher-level controls on stream behavior (e.g., valley bottom extent, valley confinement) and validating these reach types with habitat survey information. Geomorphic reach types can also be used to predict geomorphic unit assemblages, substrate and structural elements that are essential components of fish habitat. A tremendous amount of data is available in GIS and from remote sensing that describes the large-order controls that can predict expected reach types throughout the watershed network and provide a basis for upscaling reach-scale data. In addition, land use information, riparian vegetation changes, and evaluation of aerial photos and on-the-ground visits can describe the departure of the historic natural channel form and behavior from the current conditions. The discrepancy between historic and current, along with the ability to adjust, can describe restoration potential. Restoration potential can be used as a basis for watershed restoration planning (O'Brien et al. 2017). CHaMP surveys provide geomorphic unit assemblages that can be derived through topography and are consistent with the geomorphic assessment descriptions of geomorphic units (Wheaton et al. 2015). Because we can estimate the same general inputs to NREI models from bottom-up reach-level data and top-down "remote-sensed" data (Figure 5), we believe that a mechanistic-based empirical model will be able to predict NREI estimates of carrying capacity across the entire stream network for different restoration scenarios. For more information on this approach see Attachment B.

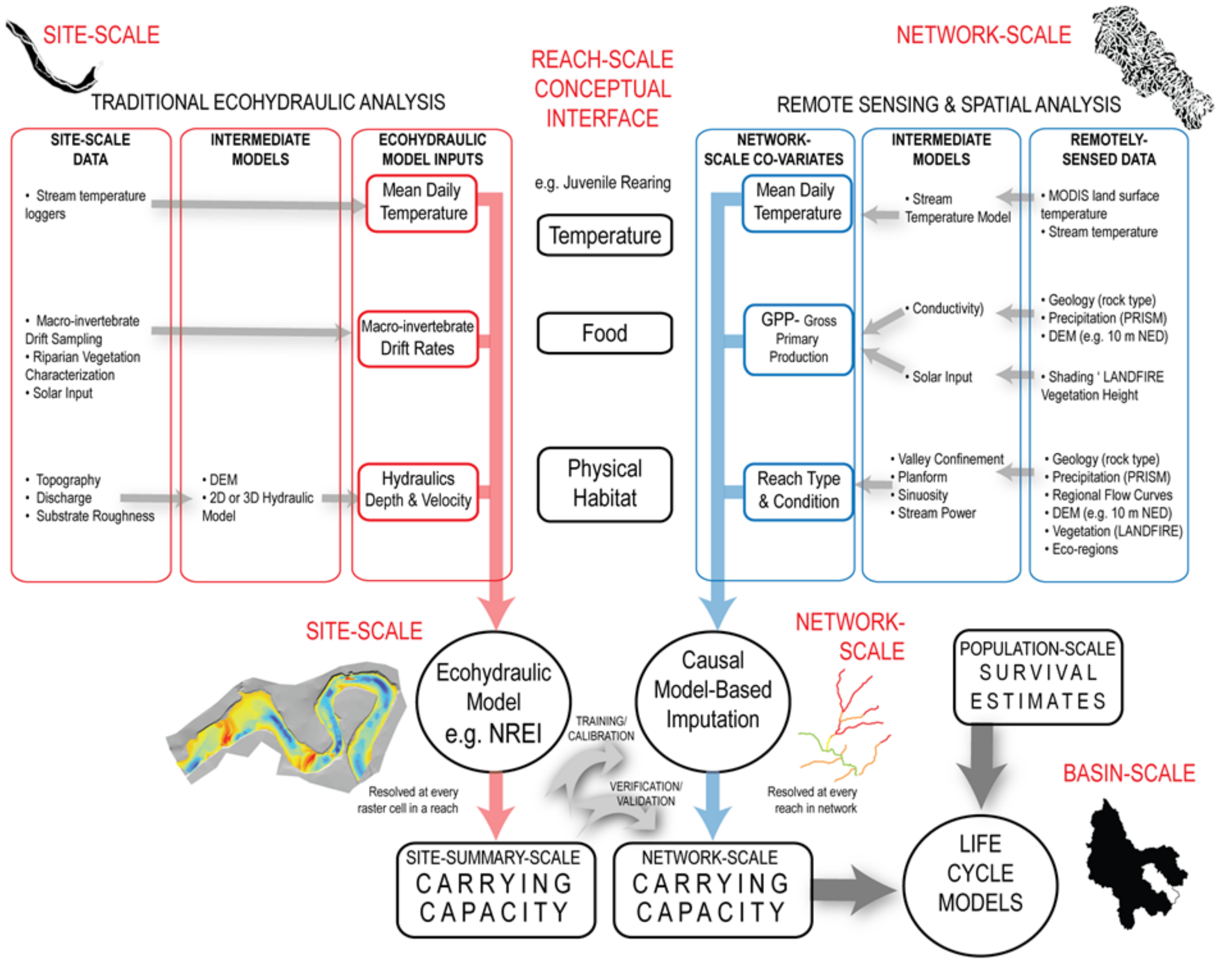

Figure 5. Example of how the components of the proposed framework can fit together for juvenile rearing using NREI model and causal-based imputation to produce robust carrying capacity estimates at the network and population scales to feed life-cycle models. Individual component pieces and concepts can be interchanged. The key attributes are the (a) conceptual alignment at the reach-scale between inputs used to drive the site-scale ecohydraulic models and the network co-variates; (b) the leveraging of readily available remotely sensed data to support network-scale modeling; and (c) use of traditional site-scale ecohydraulic analysis to train, calibrate causal model-based imputation and ultimately validate it. The framework aims to highlight the analytical tools and underlying theory necessary to transcend spatial scales in the riverscape of relevance to understanding fish population dynamics. From Wheaton et al. (2017).

Life Cycle Models

We have developed a LCM for salmon population dynamics to support salmonid management in the Columbia River basin. Three applications of the model have been developed as part of the Adaptive Management Implementation Plan (AMIP) Life-cycle Modeling Project, all focused on exploring the impacts of tributary restoration actions and providing an analytical framework for habitat action effectiveness monitoring.

The ISEMP LCM is implemented in the R programming language, an open source software package, and is freely available by download from the ISEMP website (www.isemp.org). This model is an improvement and enhancement of the Visual Basic QCI (2006) model "Salmon ISEMP Watershed Model Development: Adding Stochasticity to the Life History Model Structure" and the stage-based Yuen and Sharma (2005) model, and implements the Beverton-Holt spawner-recruit salmon population dynamics model (Beverton and Holt 1957). Many inputs are user specified, including inputs describing one or more sites within a watershed, initial salmonid populations and survival estimates by life stage, measures of uncertainty in parameter estimates, and estimates of natural parameter spatial, temporal, and pure variability. Hatchery fish introductions into a watershed, and parameters describing the relative robustness and fecundity of hatchery fish and their descendants, can also be user specified. The model calculates fish populations by life stage for each subsequent year up to a user-specified number of years.

The model also includes the option of user-specified levels of stochasticity for input parameters. This stochasticity serves two functions: (1) estimation of uncertainty of model results stemming from uncertainty of input parameters; and (2) estimation of temporal, spatial, and pure variability in the results stemming from temporal, spatial, and pure variability in the input parameters. Stochasticity is structured so as to give rise to natural correlations among input parameters. These correlation structures enable a stochastic model much more reflective of natural processes than could be achieved by assuming independence across all parameters.

Also included in the model is the ability to include time-based trends or step function changes for all user-specified parameters. Such changes may reflect, for example, changes in watershed management that lead to gradual increases in forested lands within a watershed, or discrete changes, such as a change in dam management, leading to a step function shift in seasonal water flows.

Multiple sites may be modeled simultaneously, where "sites" refer to a user-defined spatial scale over which the user wishes to define the input parameters. A site may be a reach within a tributary, a tributary within a watershed, a watershed within a subbasin, etc. The advantages of concurrent modeling of multiple sites, as opposed to modeling one site at a time, are three-fold: (1) the model accounts for lack of independence among sites within a watershed (e.g., a low water year for a single site is likely a low water year for all sites within a watershed); (2) modeling multiple sites concurrently allows for inclusion of cross-site migration, where fish at various life stages have some user-specified non-zero probability of migrating to a different site within a watershed; and (3) modeling multiple sites concurrently allows summarization of results at whatever spatial level chosen after the completion of the simulation (i.e., results may be summarized by site, stream, river, watershed, etc.).

The model has been primarily designed for stream-rearing Chinook and steelhead but is structured such that it is flexible enough to handle different species of salmonids (although they cannot be modeled simultaneously). The modeling environment does support sufficient life history variability to capture simultaneous and resident / anadromous forms, as would be necessary for a general *O. mykiss* population model. A stage-by-stage description of the model and the input and output files are provided in the ISEMP Watershed Model Version 3.0. User's Guide. User guide and software downloads are available from: www.isemp.org/products/tools.

Results

RME Habitat Status and Trends

The CHaMP sampling design is optimized for estimation of the mean and variance of CHaMP metrics at the watershed scale, or at least over a large portion of a watershed. This means that a wide variety of sites are sampled across a broad array of landscape characteristics. Analysis shows that CHaMP site characteristics represent the range of characteristics across the Columbia River basin for unsampled sites for both numeric and categorical GAA variables (e.g., Figure 6). Cross-validation at various spatial levels has shown that a subset of all GAAs have observable relationships to the CHaMP metrics (results available upon request).

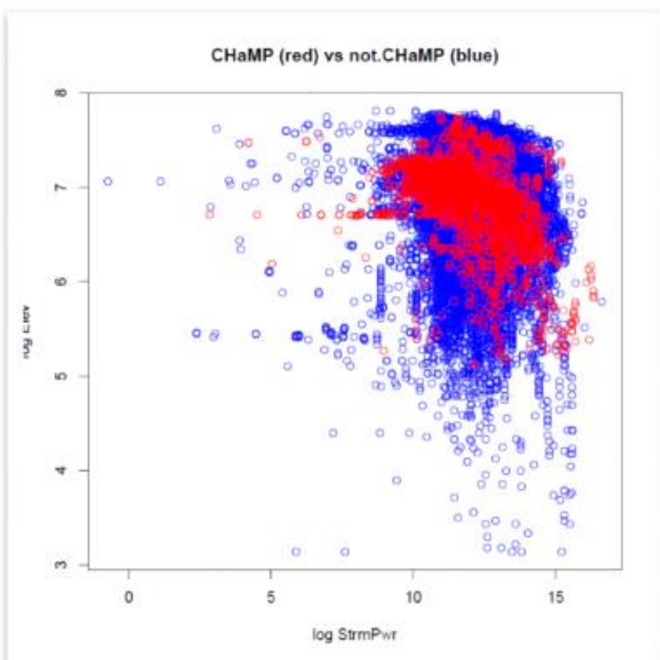

Figure 6. Example of CHaMP coverage of physical characteristics of the anadromous extent of the interior Columbia River Basin. In this case, CHaMP sampled streams (red dots) generally represent the core of the range of elevation (y axis) and stream power (x axis) found in the interior Columbia River basin (blue dots).

The CHaMP protocol generates a large amount of data that can be summarized into many different metrics. To focus the number of metrics that we report on here, we have provided summaries of metrics identified by the QRF model as important predictors of spring Chinook density by watershed and year. Annual assessment of sources of variance associated with those metrics shows that in general CHaMP protocols produce measurements with low noise compared to other sources of variability (Figure 7), and are standardized and repeatable across the Columbia River Basin. Figures 8 to 21 show annual watershed-level mean and variance estimates for these metrics.

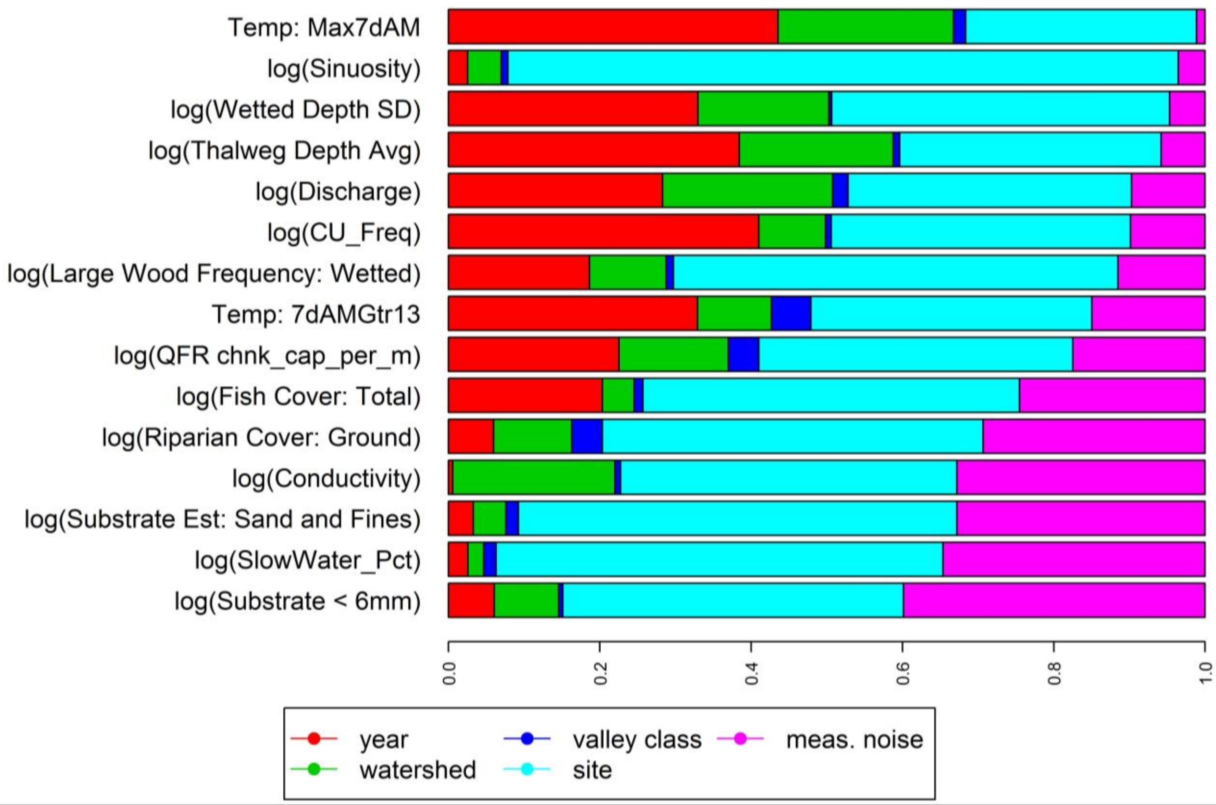

Figure 7. Estimated components of variance for metrics generated using the CHaMP protocol and identified as important predictors of spring Chinook density using the QRF model.

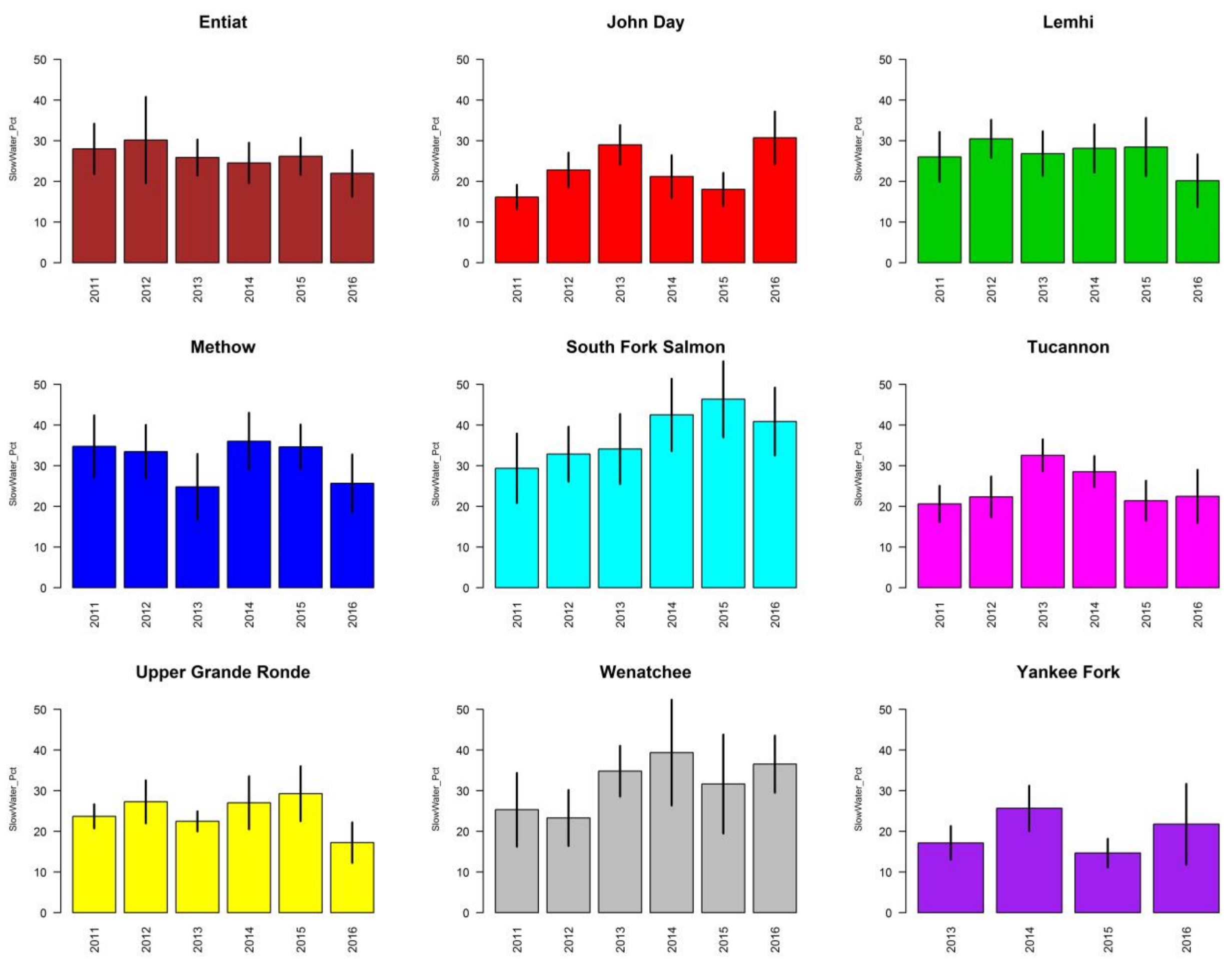

Figure 8. Status and trend of the percent slow water, a measure of pools and backwater habitat that provides high flow refugia for rearing parr, in streams across nine Columbia River Basin watersheds, 2011 – 2016.

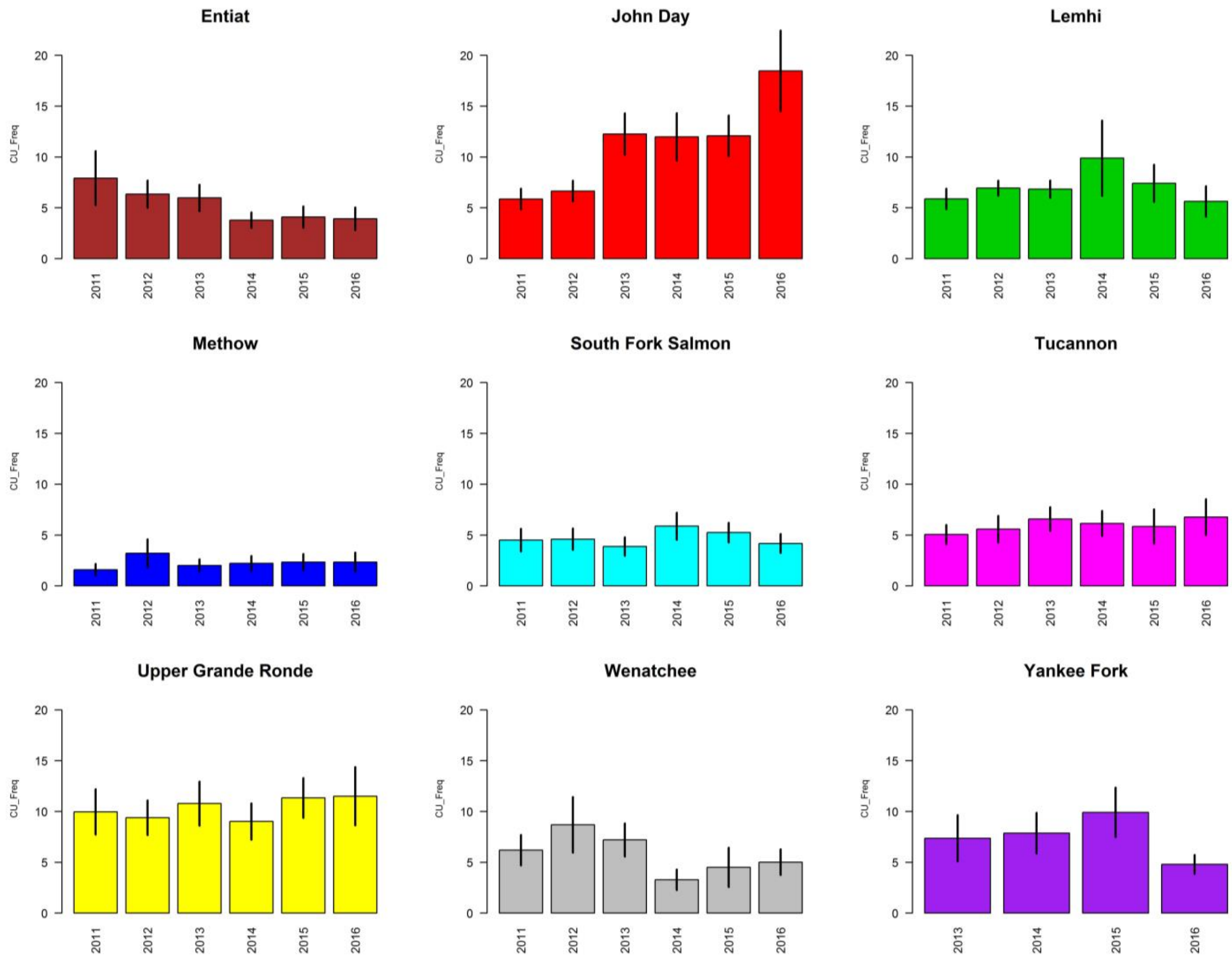

Figure 9. Status and trend of diversity of channel units, a measure of habitat complexity, in nine Columbia River Basin watersheds, 2011 – 2016.

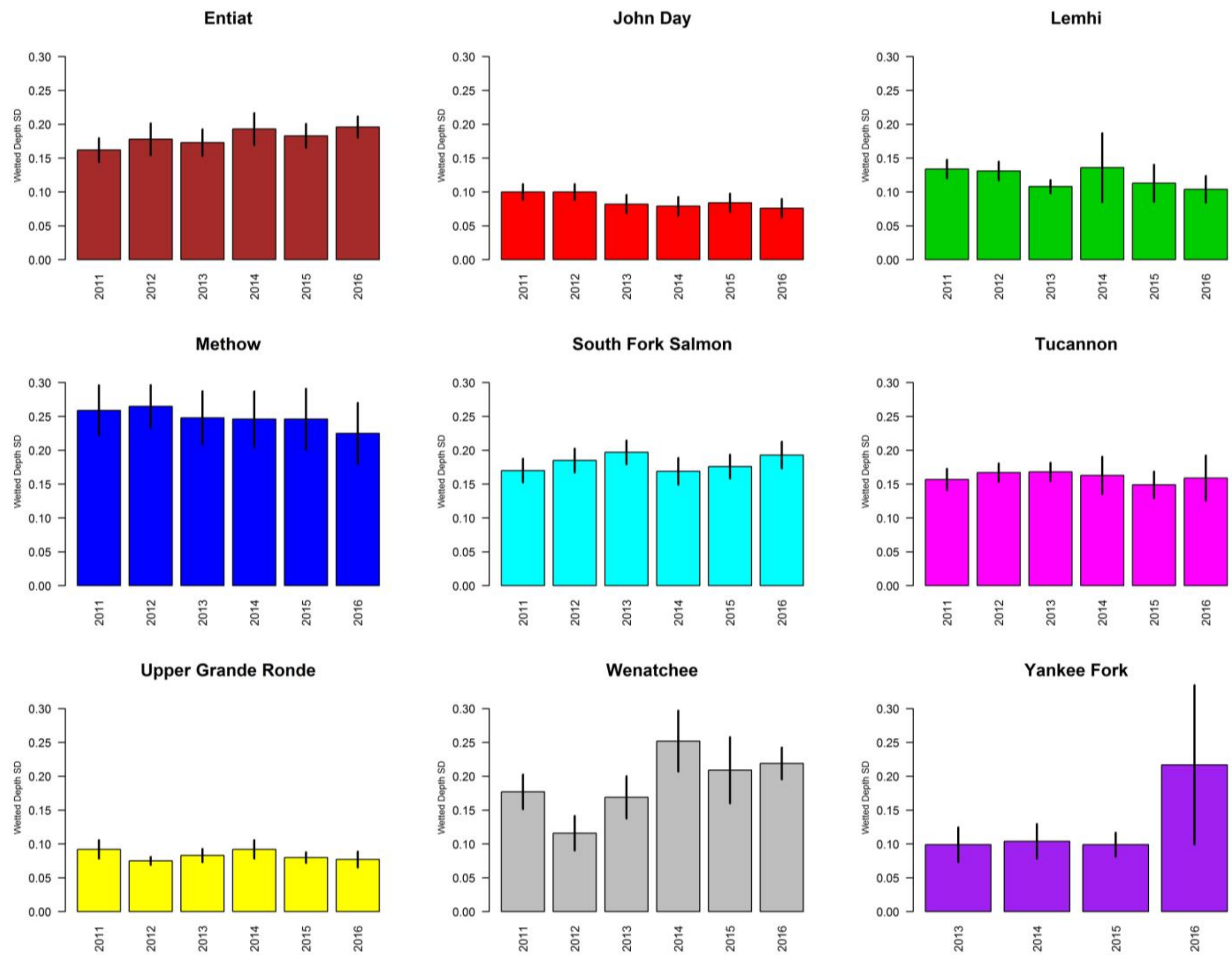

Figure 10. Status and trend of the standard deviation of average wetted depth, a measure of habitat complexity, in streams across nine Columbia River Basin watersheds, 2011 – 2016.

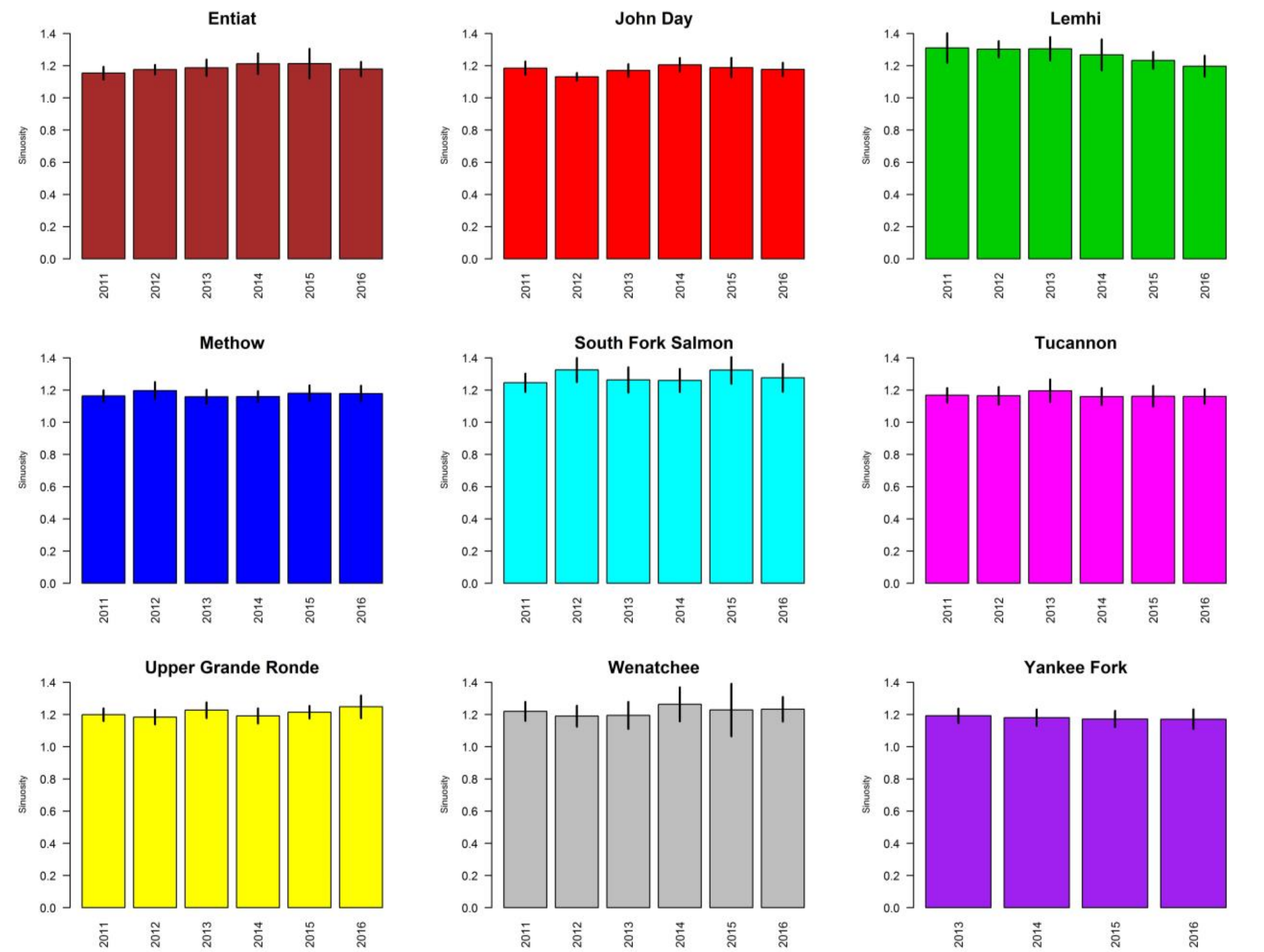

Figure 11. Status and trend of stream channel sinuosity, a measure of stream complexity, in nine Columbia River Basin watersheds, 2011 – 2016.

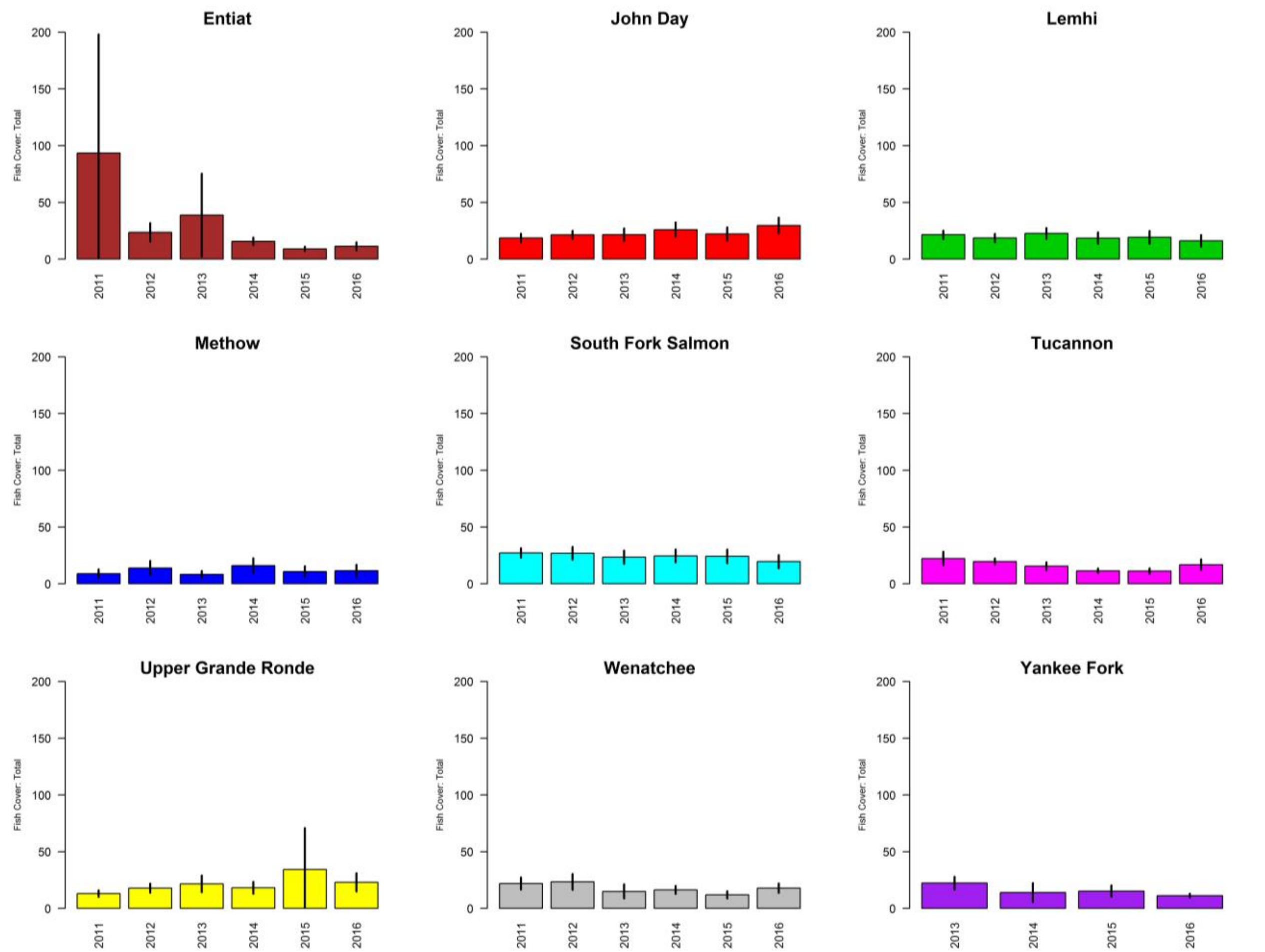

Figure 12. Status and trend of total fish cover in streams across nine Columbia River Basin watersheds, 2011 – 2016.

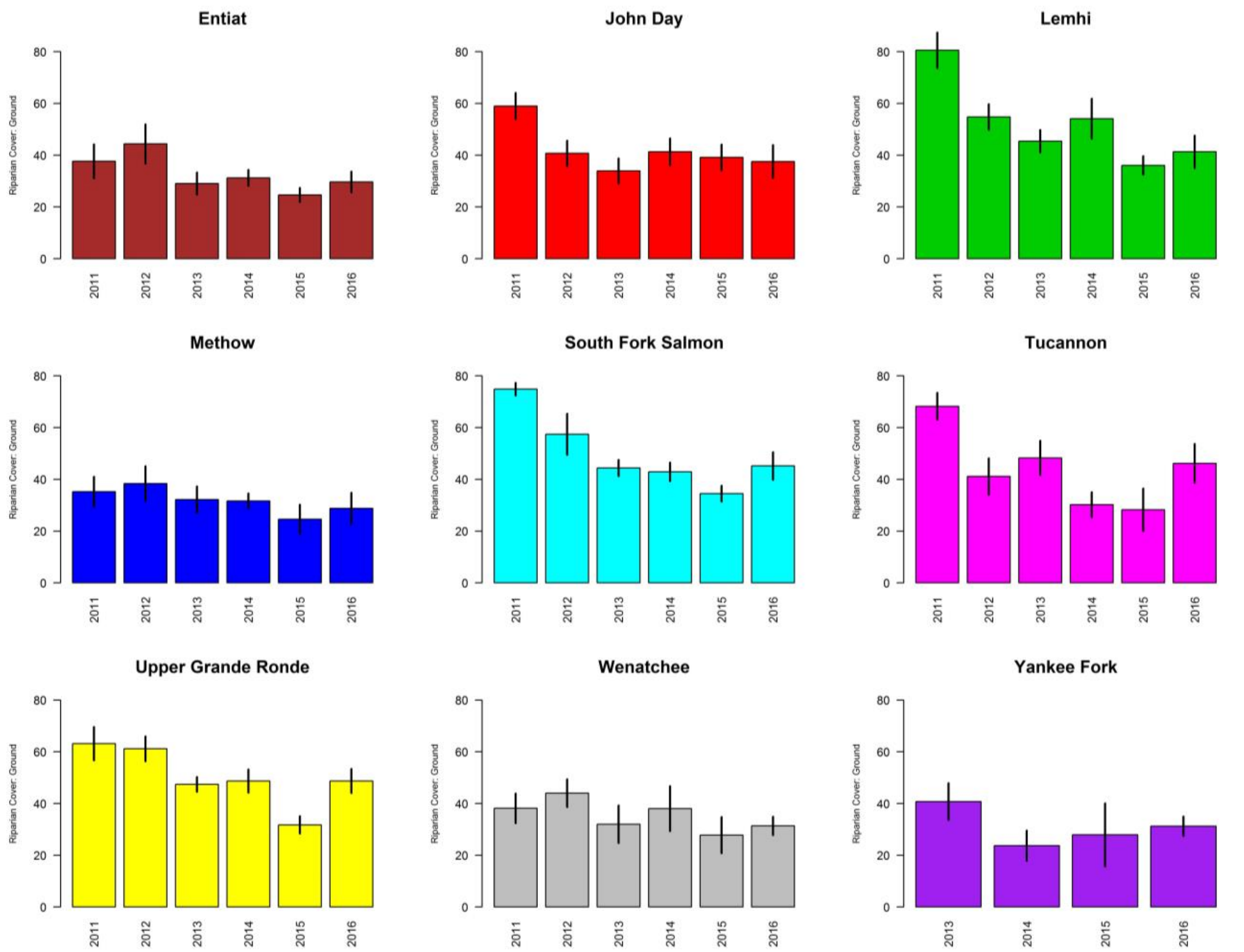

Figure 13. Status and trend of riparian ground cover, a proxy for the amount of streamside disturbance, along streams across nine Columbia River Basin watersheds, 2011 – 2016.

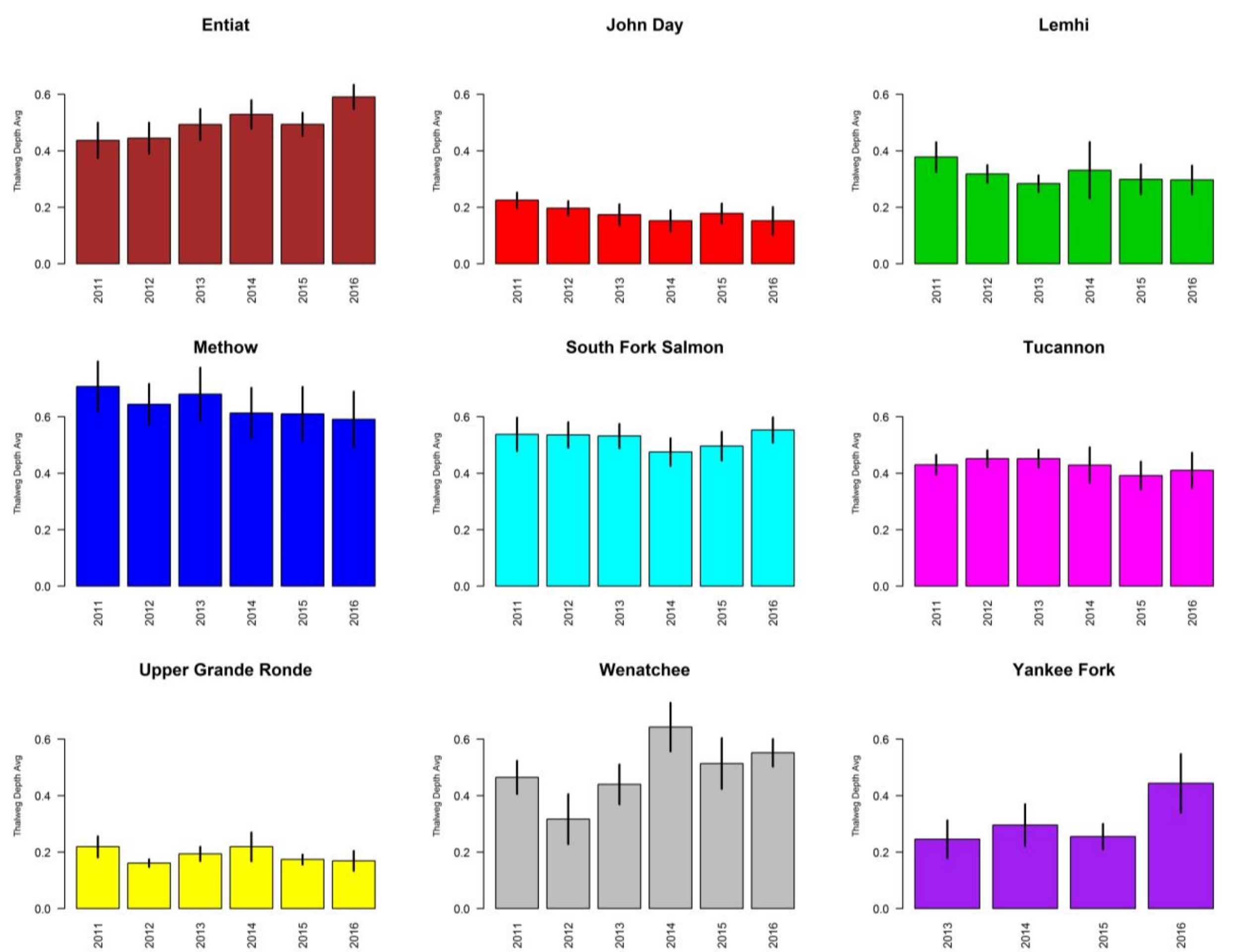

Figure 14. Status and trend of average thalweg depth in streams across nine Columbia River Basin watersheds, 2011 – 2016.

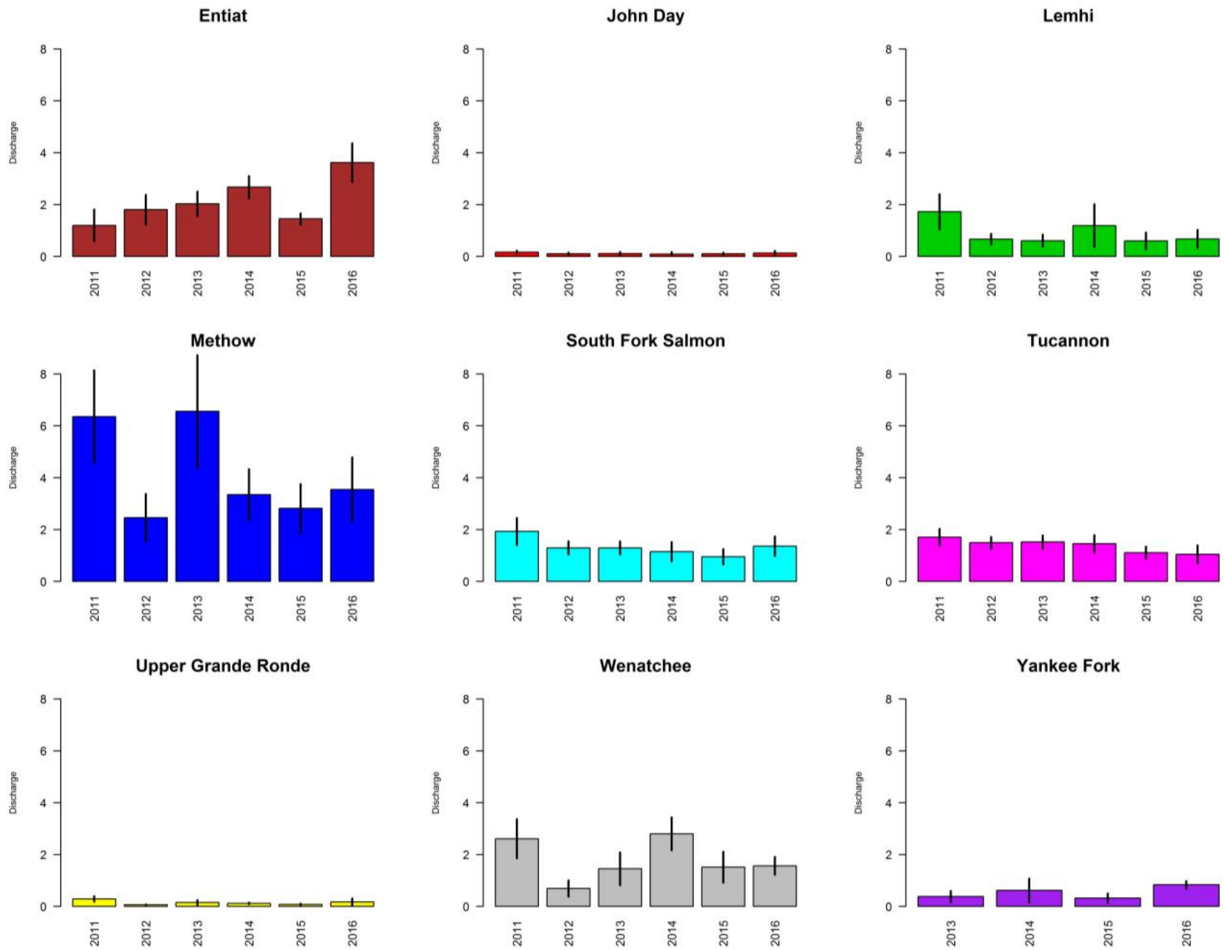

Figure 15. Status and trend of stream discharge across nine Columbia River Basin watersheds, 2011 – 2016.

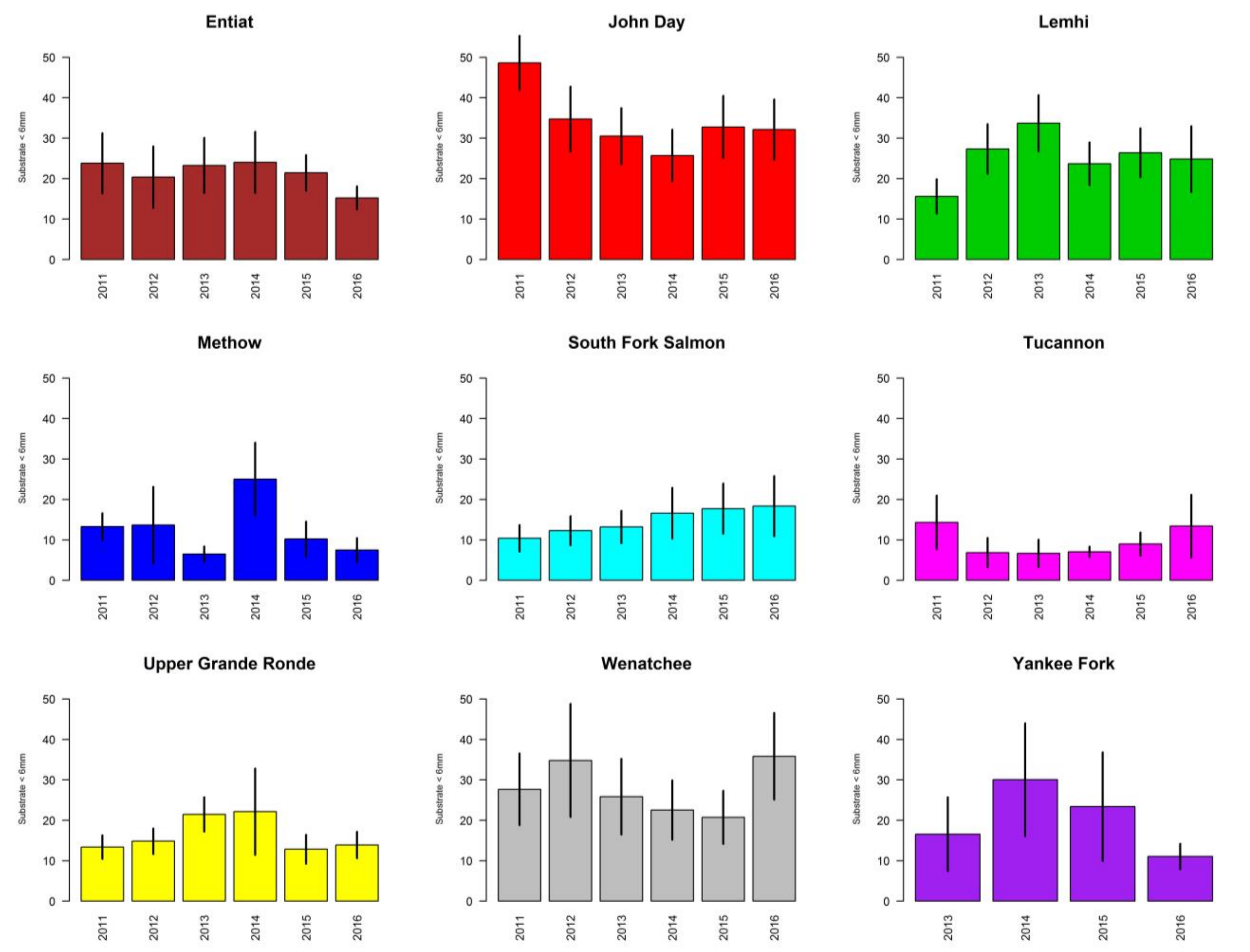

Figure 16. Status and trend of substrate less than 6 mm diameter, a measure of the amount of good spawning habitat, in nine Columbia River Basin watersheds, 2011 – 2016.

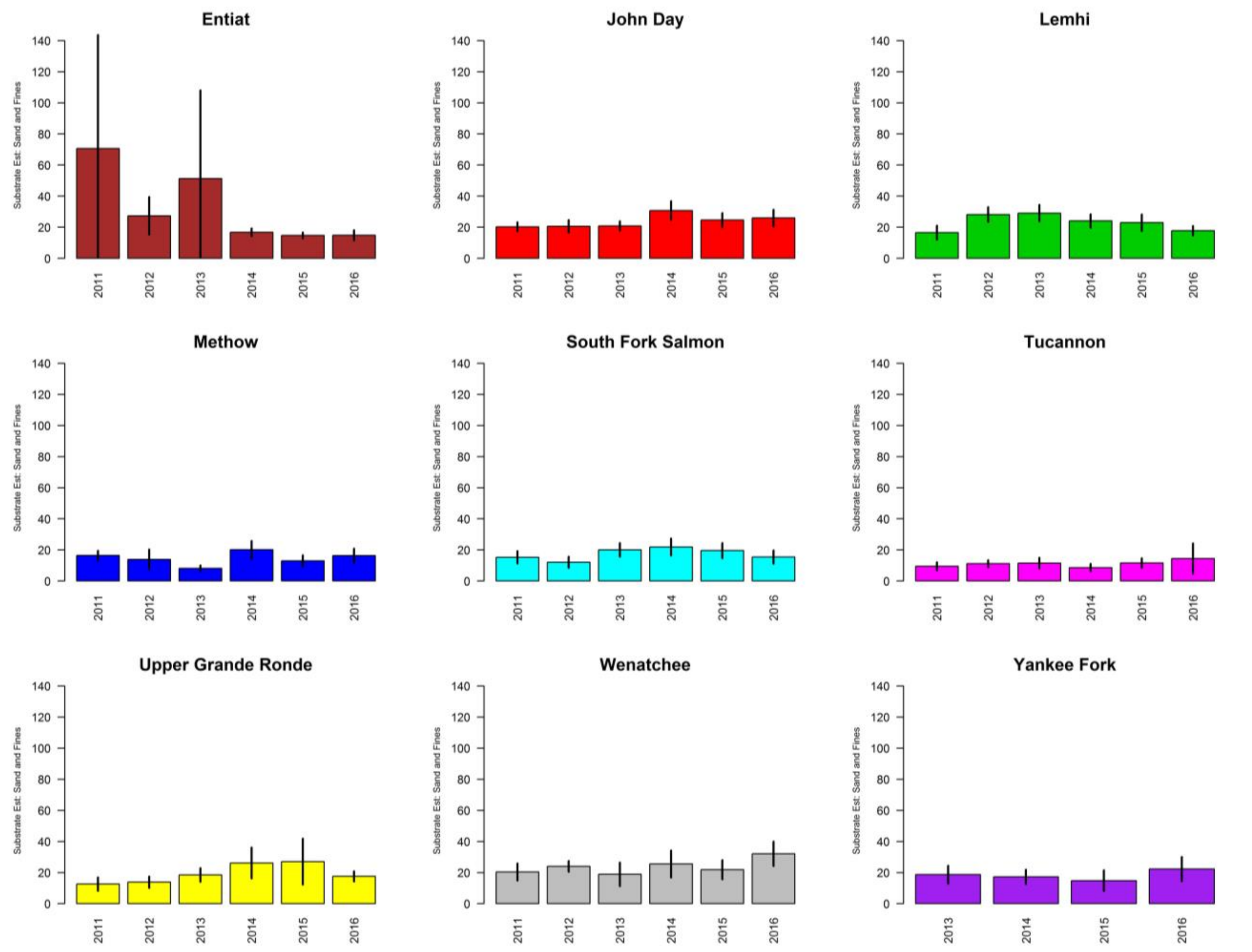

Figure 17. Status and trend of estimated sand and fines substrate, a measure of a factor that could lead to unsuitable spawning habitat, in nine Columbia River Basin watersheds, 2011 – 2016.

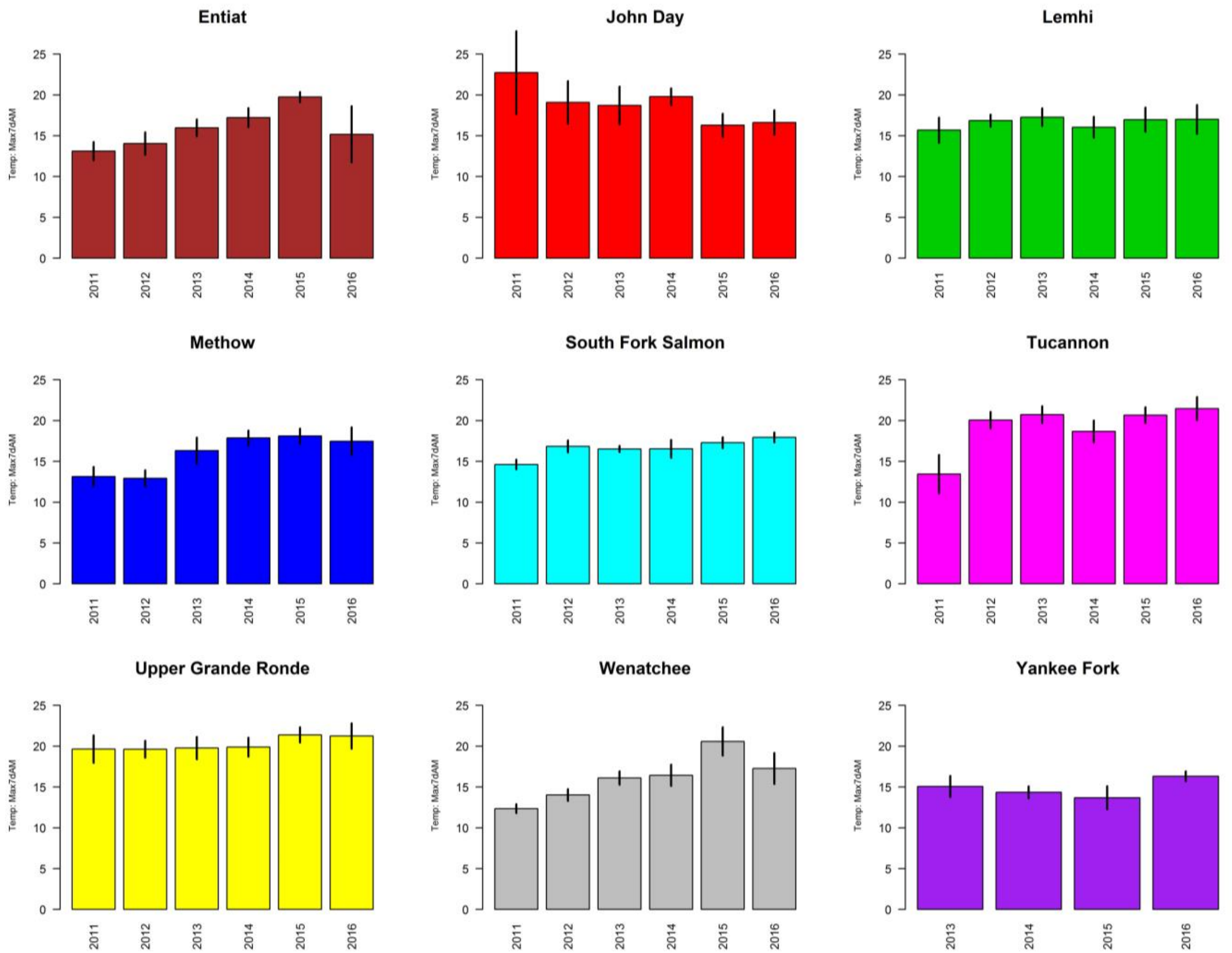

Figure 18. Status and trend of the maximum 7-day average temperature in nine Columbia River Basin watersheds, 2011 – 2016.

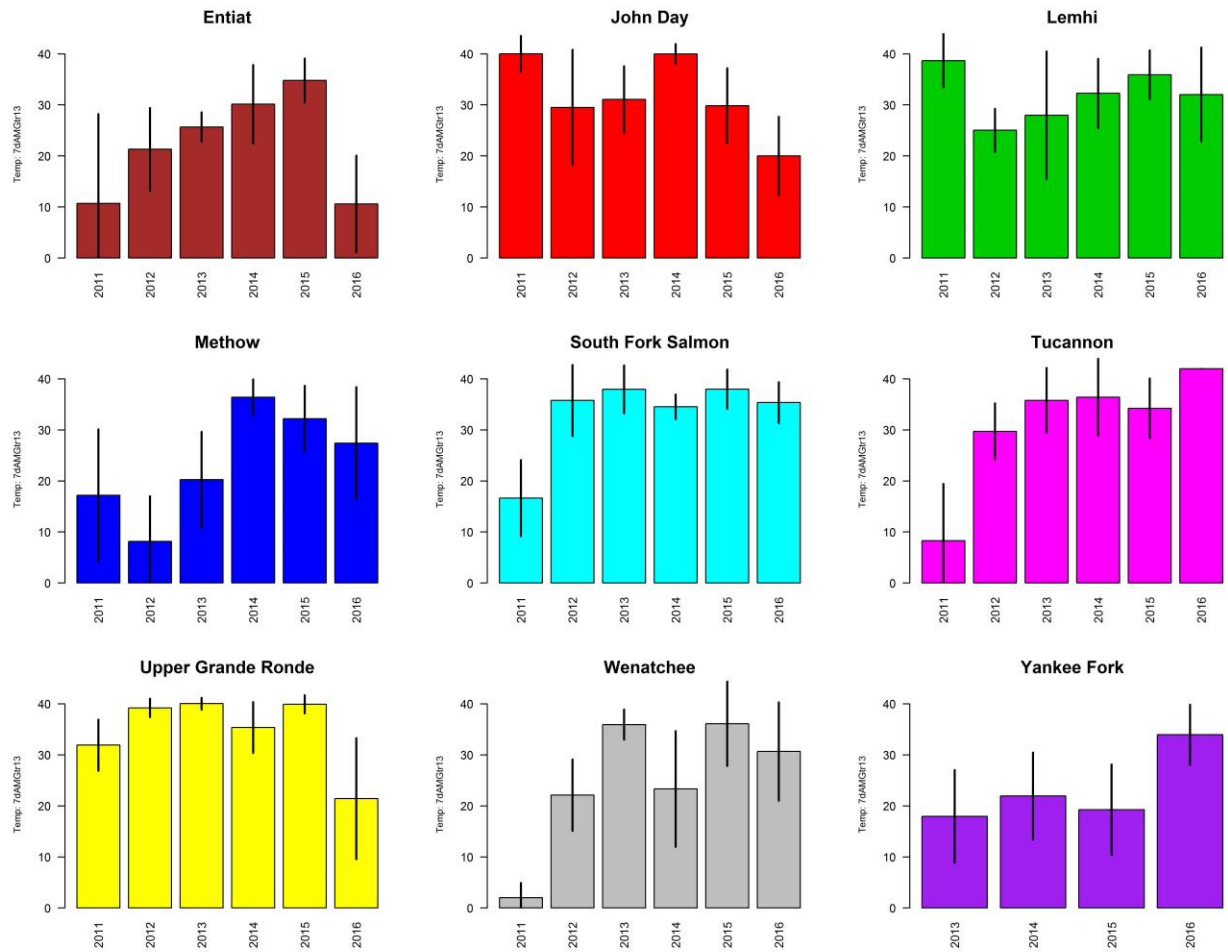

Figure 19. Status and trend of the number of 7-day average of daily maximum temperature values between peak salmon and steelhead spawning periods (July 15th to August 31st) that are greater than 13°C across nine Columbia River Basin watersheds, 2011 – 2016.

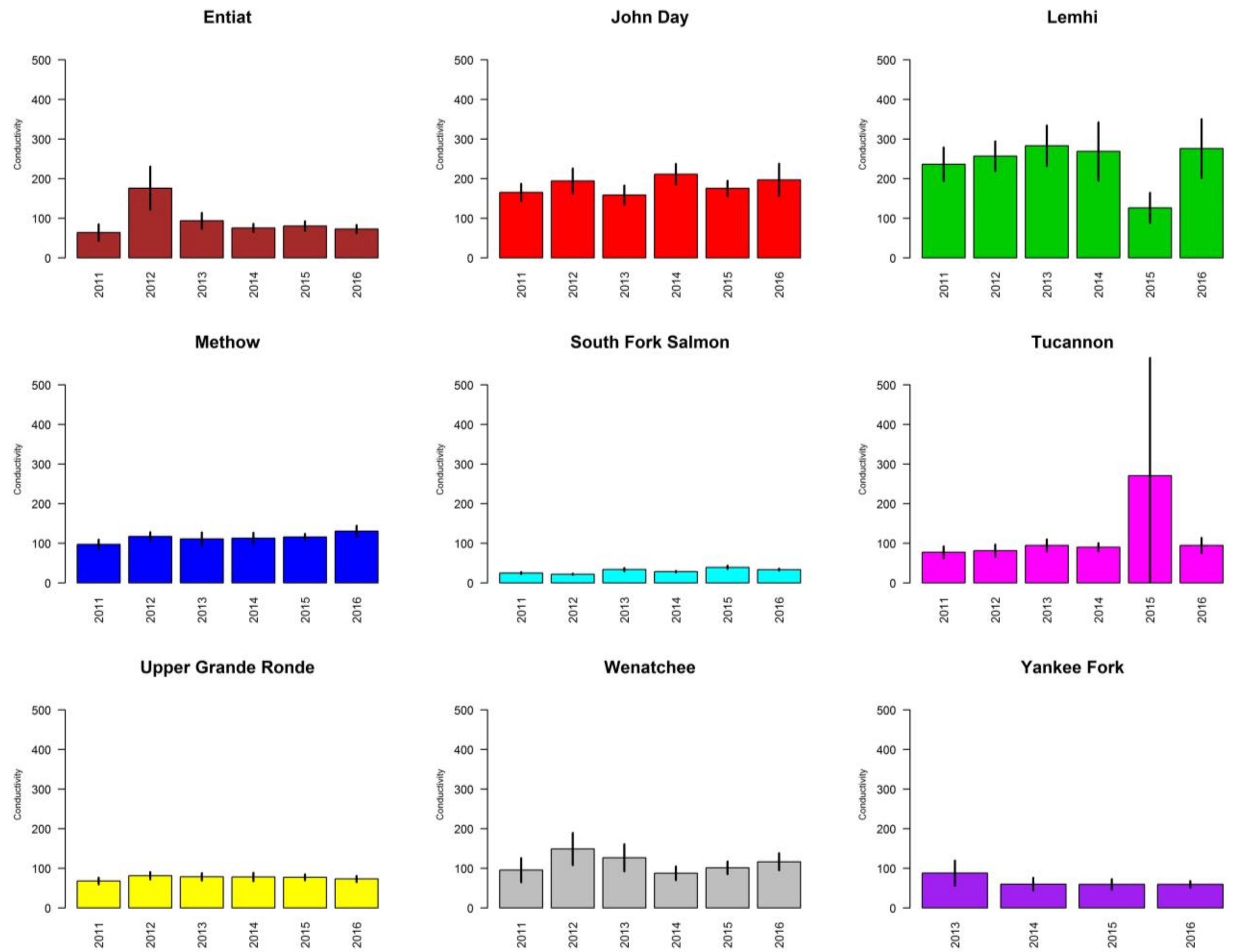

Figure 20. Status and trend of conductivity in streams, a proxy for food production, across nine Columbia River Basin watersheds, 2011 – 2016.

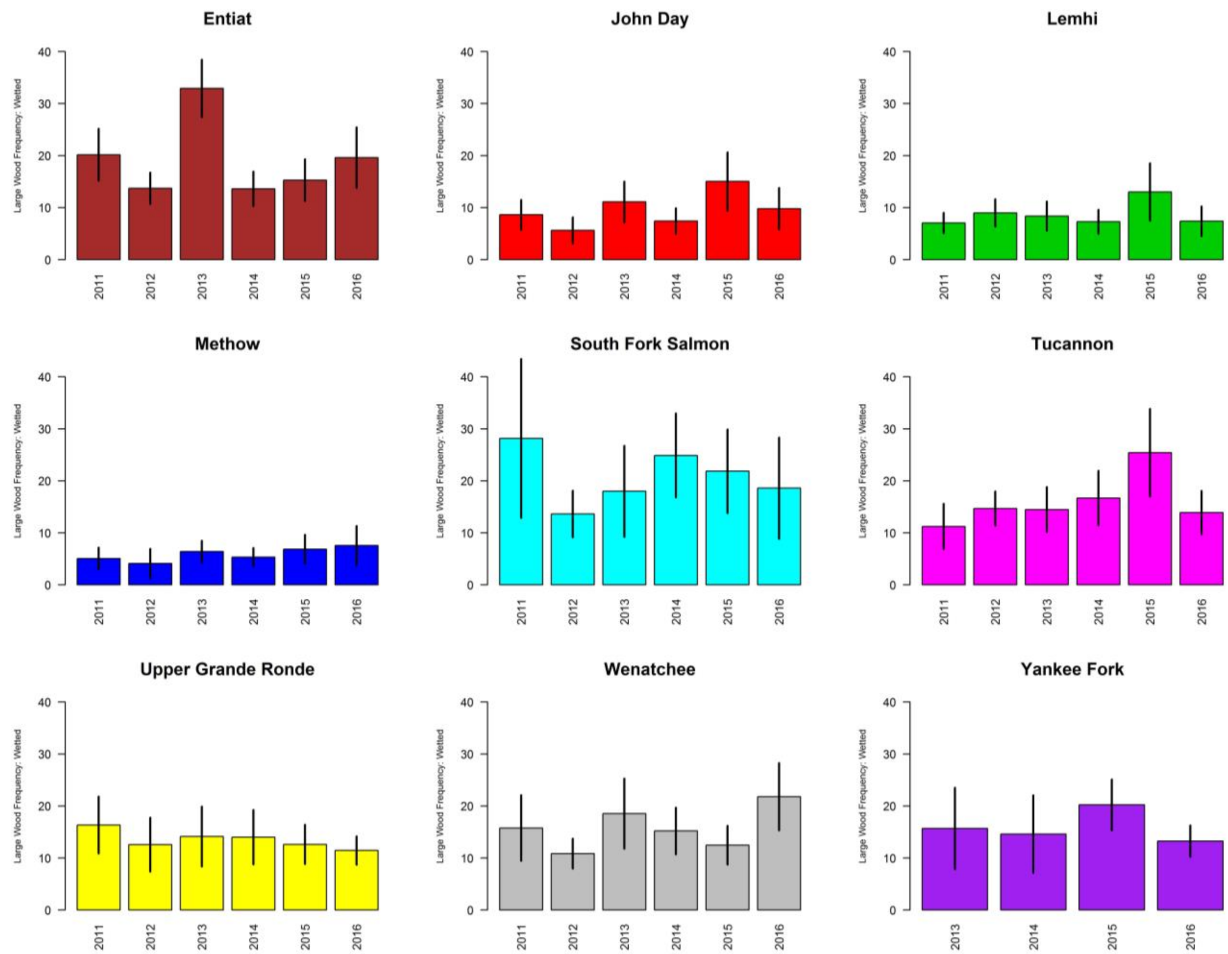

Figure 21. Status and trend of frequency of large wood within the wetted channel, a measure of habitat complexity, in nine Columbia River Basin watersheds, 2011 – 2016.

Fish Status and Trends

Adult Escapement

ISEMP is using the State-Space Adult Dam Escapement (STADEM) and Dam Adult Branch Occupancy (DABOM) models to generate annual estimates of adult escapement into tributaries in the Lemhi river subbasin. This approach has been adopted by the Washington Department of Fish and Wildlife (WDFW) to estimate escapement for the Wenatchee, Entiat, Methow, and Okanogan River subbasins.

Entiat River, WA

Estimates for adult steelhead escapement into the Entiat are available from WDFW (Figure 22).

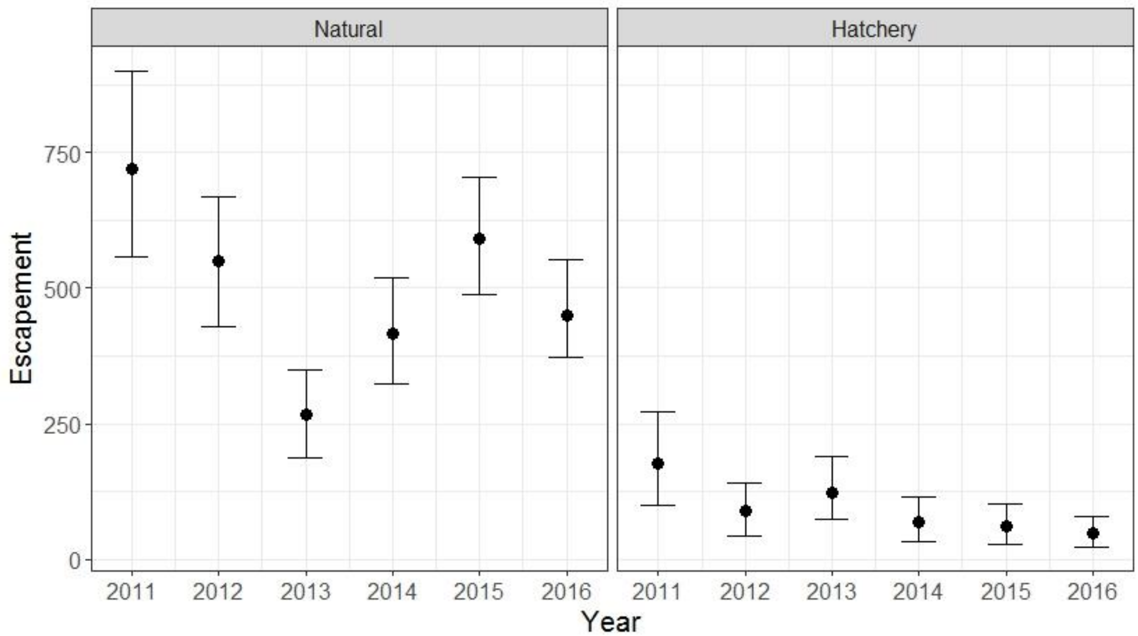

Figure 22. Population estimates and coefficient of variation for natural origin adult steelhead escapement into the Entiat River watershed estimated using PIT tag interrogations at IPTDS sites throughout the Columbia River. Data courtesy of Washington Department of Fish and Wildlife (Truscott et al. 2017).

Snake River, ID

Tables 2 and 3 provide summarized adult escapement data from IPTDS for Snake River TRT-identified populations of spring/summer Chinook salmon and steelhead. Currently, ISEMP is also collaborating with IDFG BPA projects 1991-073-00 and 2010-026-00 to produce:

- Age and sex structure accompanying escapement at the population and sub-population level (Figures 23 and 24).
- Length of returning adults at the population and sub-population level.
- Adult run-timing past LGR and into populations and sub-populations for natural origin adults.

- Fraction of the populations and sub-populations composed of repeat spawners.
- Rate of post-spawn survival to LGR (kelting rate).
- Fraction of A- and B-run populations composed of individuals that meet B-run classification criteria (e.g., 2-ocean age or greater or fork length [FL] greater than 77.5 cm).

Table 2**.** Population estimates and coefficient of variation for natural origin adult spring/summer Chinook salmon escapement estimated using PIT tag interrogations at IPTDS sites throughout the Snake River.

MPG	TRT Name	TRT	2010	2011	2012	2013	2014	2015
Grande Ronde / Imnaha	Catherine Creek	GRCAT	236 (0.26)	181 (0.15)	298 (0.2)	280 (0.13)	385 (0.12)	311 (0.17)
Grande Ronde / Imnaha	Lookingglass Creek	GRLOO	51 (0.38)	309 (0.13)	99 (0.26)	120 (0.16)	142 (0.18)	237 (0.15)
Grande Ronde / Imnaha	Lostine River	GRLOS	116 (0.34)	179 (0.17)	526 (0.15)	350 (0.16)	2131 (0.1)	2015 (0.08)
Grande Ronde / Imnaha	Big Sheep Creek	IRBSH	--	597 (0.13)	224 (0.25)	109 (0.29)	185 (0.24)	199 (0.25)
Grande Ronde / Imnaha	Imnaha River mainstem	IRMAI	474 (0.23)	2207 (0.08)	1215 (0.11)	748 (0.14)	1282 (0.12)	785 (0.13)
Lower Snake	Asotin Creek	SNASO	--	--	--	--	--	--
Middle Fork Salmon River	Big Creek	MFBIG	215 (0.38)	457 (0.15)	913 (0.13)	986 (0.09)	1285 (0.13)	1175 (0.1)
South Fork Salmon River	East Fork South Fork Salmon River	SFEFS	1088 (0.17)	718 (0.12)	1083 (0.13)	1255 (0.1)	1328 (0.11)	728 (0.15)
South Fork Salmon River	South Fork Salmon River mainstem	SFMAI	4620 (0.1)	3393 (0.07)	2389 (0.1)	1358 (0.09)	2110 (0.1)	959 (0.14)
South Fork Salmon River	Secesh River	SFSEC	1386 (0.16)	974 (0.12)	1396 (0.12)	1530 (0.09)	1668 (0.1)	659 (0.16)
Upper Salmon River	East Fork Salmon River	SREFS	343 (0.7)	161 (0.41)	283 (0.6)	307 (0.18)	320 (0.19)	--
Upper Salmon River	Lemhi River	SRLEM	156 (0.27)	267 (0.13)	83 (0.31)	393 (0.11)	464 (0.12)	718 (0.1)
Upper Salmon River	Salmon River lower mainstem below Redfish Lake	SRLMA	--	--	--	1271 (0.1)	1732 (0.1)	1614 (0.1)
Upper Salmon River	Salmon River upper mainstem above Redfish Lake	SRUMA	865 (0.64)	799 (0.31)	924 (0.53)	564 (0.14)	615 (0.15)	520 (0.16)
Upper Salmon River	Valley Creek	SRVAL	329 (0.73)	452 (0.33)	675 (0.53)	389 (0.15)	739 (0.13)	452 (0.18)
Upper Salmon River	Yankee Fork	SRYFS	--	--	307 (0.6)	343 (0.17)	213 (0.22)	127 (0.31)
Wet Clearwater	Lolo Creek	CRLOL	--	--	250 (0.2)	103 (0.16)	88 (0.2)	207 (0.16)
NA	NA	CRLAP	--	--	--	--	--	--
NA	NA	CRLOC	--	--	--	--	--	--
NA	NA	CRPOT	--	--	--	--	--	--
NA	NA	GRUMA	--	--	87 (0.32)	92 (0.54)	22 (0.47)	358 (0.21)
NA	NA	SCLAW_ SCUMA	61 (0.35)	94 (0.18)	795 (0.17)	380 (0.11)	442 (0.12)	660 (0.11)
NA	NA	SRLSR	55 (0.36)	93 (0.18)	10 (0.72)	7 (0.41)	--	--
NA	NA	SRPAH	184 (0.81)	34 (0.71)	205 (0.65)	344 (0.19)	320 (0.19)	320 (0.21)

Table 3**.** Population estimates and coefficient of variation for natural origin adult steelhead escapement estimated using PIT tag interrogations at IPTDS sites throughout the Snake River.

MPG	**TRT Name**	**TRT**	**Run Type**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**
Clearwater River	Fish Creek	CRLOC-s	B	102 (0.13)	421 (0.12)	186 (0.14)	92 (0.13)	84 (0.14)	420 (0.1)
Clearwater River	Lolo Creek	CRLOL-s	B	--	--	664 (0.09)	310 (0.09)	274 (0.1)	558 (0.1)
Clearwater River	South Fork Clearwater River	CRSFC-s	B	--	--	1188 (0.08)	664 (0.08)	528 (0.09)	922 (0.08)
Grande Ronde River	Joseph Creek	GRJOS-s	A	--	1581 (0.06)	1828 (0.06)	1564 (0.06)	1780 (0.07)	3028 (0.07)
Grande Ronde River	Grande Ronde River upper mainstem	GRUMA-s	A	293 (0.1)	368 (0.07)	668 (0.07)	1132 (0.06)	1185 (0.07)	2258 (0.07)
Grande Ronde River	Wallowa River	GRWAL-s	A	147 (0.12)	321 (0.09)	244 (0.12)	135 (0.1)	515 (0.1)	913 (0.08)
Imnaha River	Imnaha River	IRMAI-s	A	294 (0.1)	3045 (0.05)	2905 (0.06)	1333 (0.06)	2431 (0.07)	2343 (0.06)
Lower Snake	Asotin Creek	SNASO-s	A	1516 (0.07)	1156 (0.06)	1341 (0.06)	798 (0.06)	1032 (0.08)	1281 (0.07)
Lower Snake	Tucannon River	SNTUC-s	A	745 (0.08)	465 (0.08)	1023 (0.08)	339 (0.08)	475 (0.09)	867 (0.08)
Salmon River	Big Creek	MFBIG-s	B	1092 (0.19)	594 (0.08)	458 (0.14)	392 (0.08)	274 (0.13)	733 (0.14)
Salmon River	South Fork Salmon River	SFMAI-s	B	1175 (0.08)	2041 (0.07)	1112 (0.08)	645 (0.08)	743 (0.11)	1473 (0.08)
Salmon River	Secesh River	SFSEC-s	B	174 (0.23)	330 (0.16)	182 (0.23)	42 (0.4)	146 (0.24)	231 (0.21)
Salmon River	East Fork Salmon River	SREFS-s	A	--	--	32 (1.81)	35 (0.39)	--	54 (0.44)
Salmon River	Lemhi River	SRLEM-s	A	417 (0.1)	314 (0.09)	347 (0.11)	334 (0.08)	339 (0.09)	342 (0.1)
Salmon River	Pahsimeroi River	SRPAH-s	A	345 (0.71)	672 (0.39)	2133 (0.43)	469 (0.1)	665 (0.11)	653 (0.11)
Salmon River	Salmon River upper mainstem	SRUMA-s	A	389 (0.64)	375 (0.64)	652 (1.03)	192 (0.17)	183 (0.25)	347 (0.16)

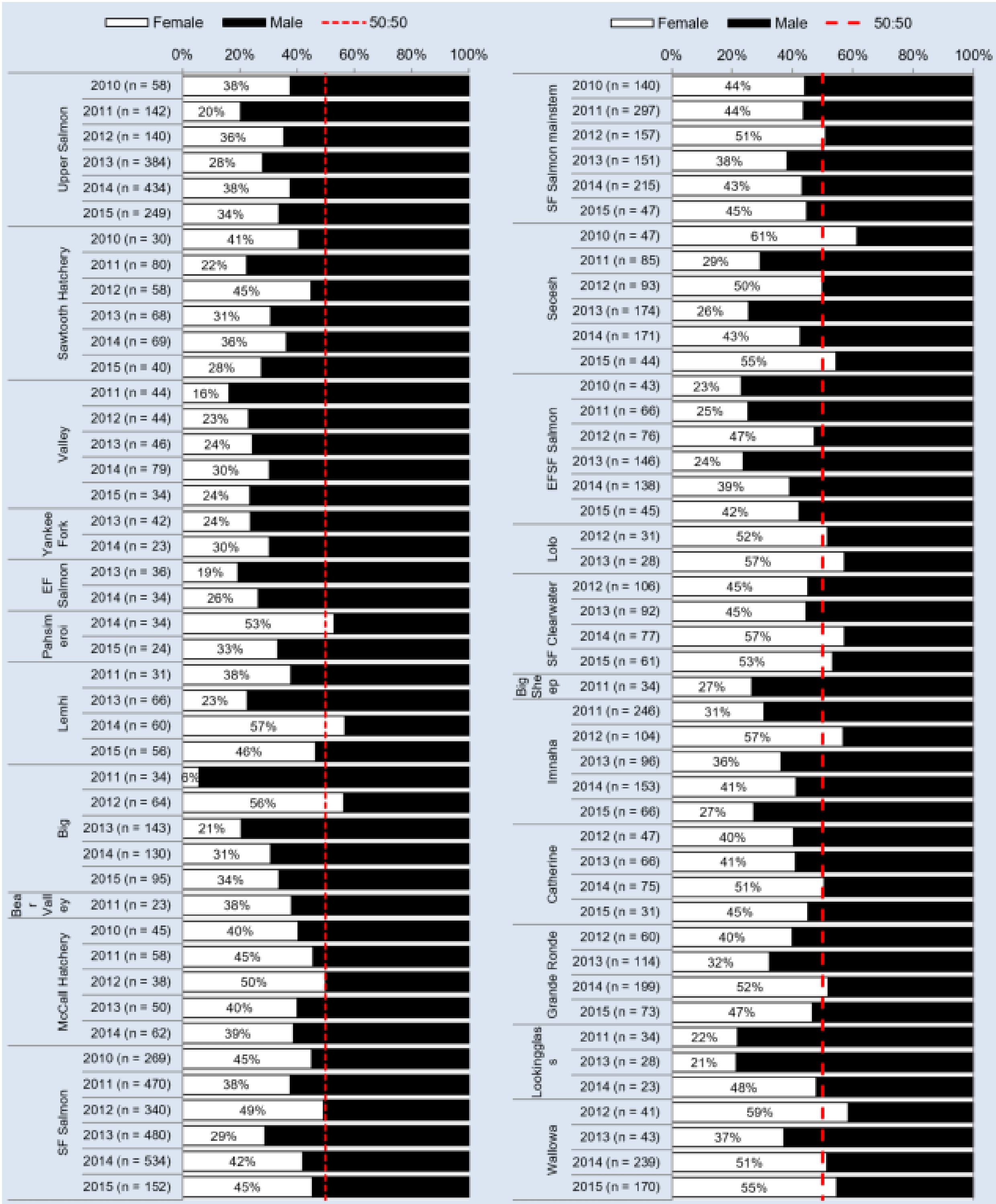

Figure 23**.** Estimates of sex ratios for natural-origin spring/summer Chinook salmon populations/subpopulations in the Snake River based on PIT-tagged individuals crossing IPTDS throughout the Snake River. Sex data provided by a sex-specific genetic assay. Figure provided by IDFG.

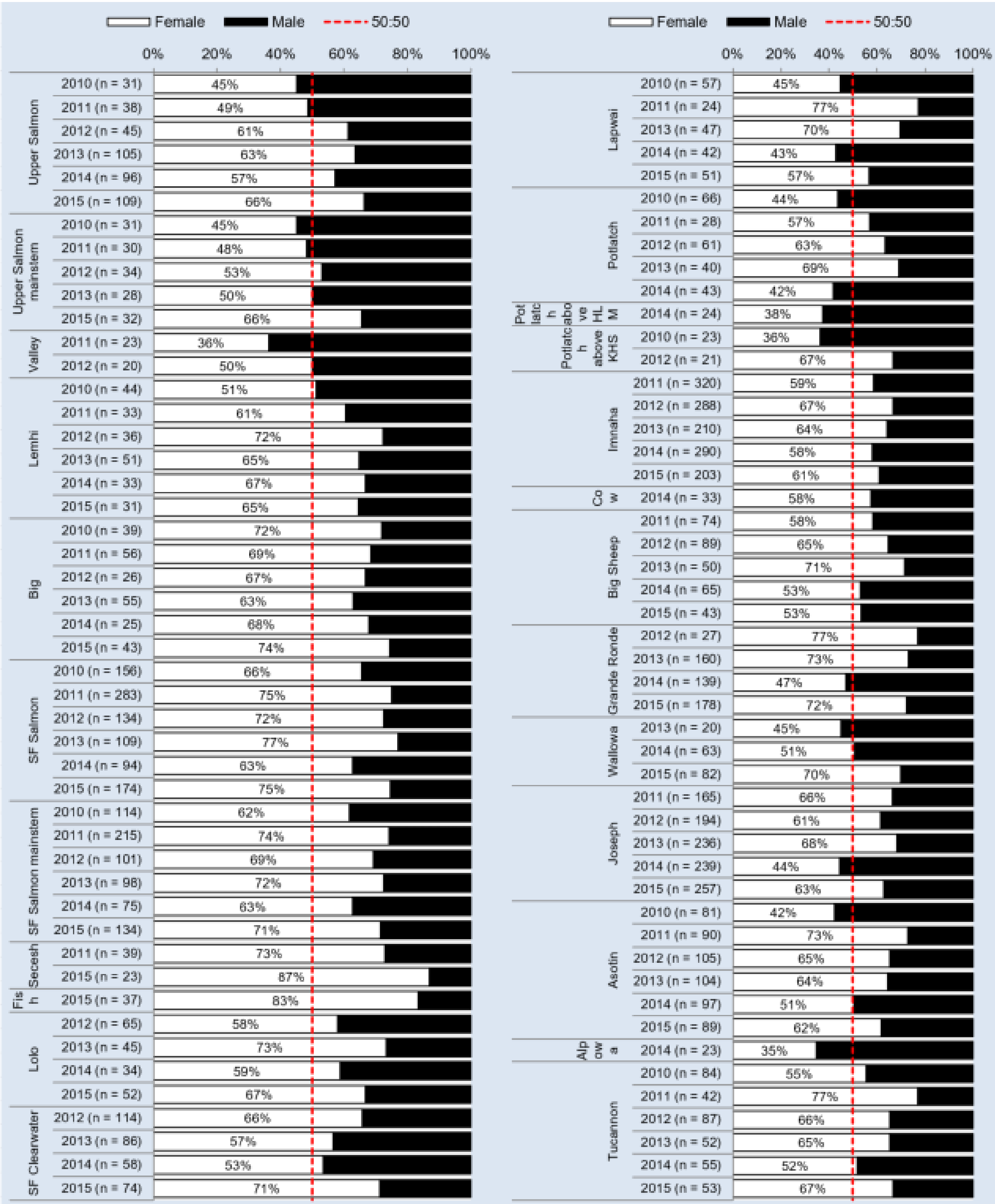

Figure 24. Estimates of sex ratios for natural-origin steelhead populations/subpopulations in the Snake River based on PIT-tagged individuals crossing IPTDS throughout the Snake River. Sex data provided by a sex-specific genetic assay. Figure provided by IDFG.

Juvenile Capacity

Chinook in CHaMP Watersheds

Estimates of juvenile summer rearing capacity for Chinook in nine watersheds where habitat data is collected using the CHaMP protocol are generated using QRF and are in number of juveniles per meter (Figure 25).

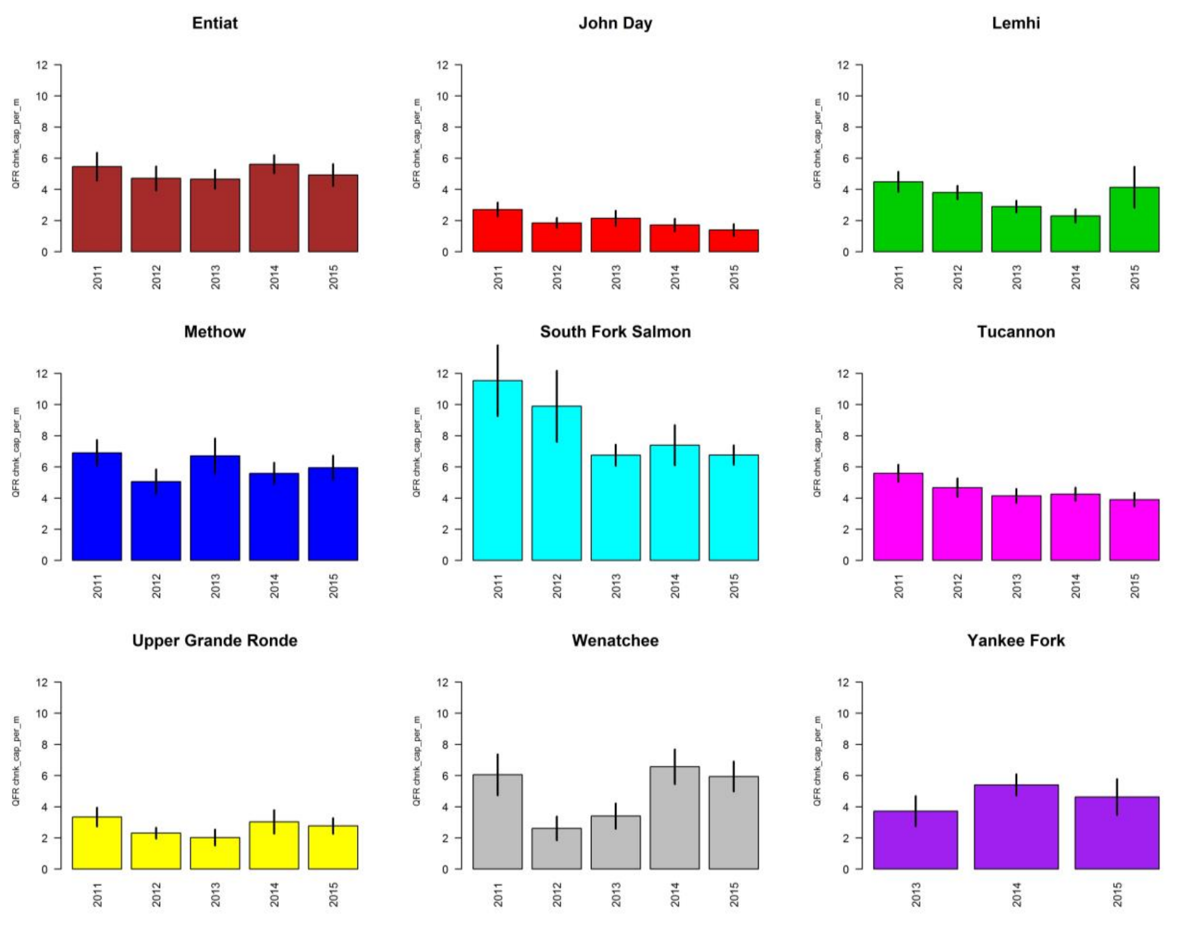

Figure 25. Estimated juvenile Chinook summer parr rearing capacity in nine Columbia River Basin watersheds from 2011 – 2015.

Juvenile Abundance

Entiat River, WA

Status and trend population estimates for Entiat River spring Chinook and steelhead summer standing crop populations are estimated using a GRTS-based roll-up (Figure 26).

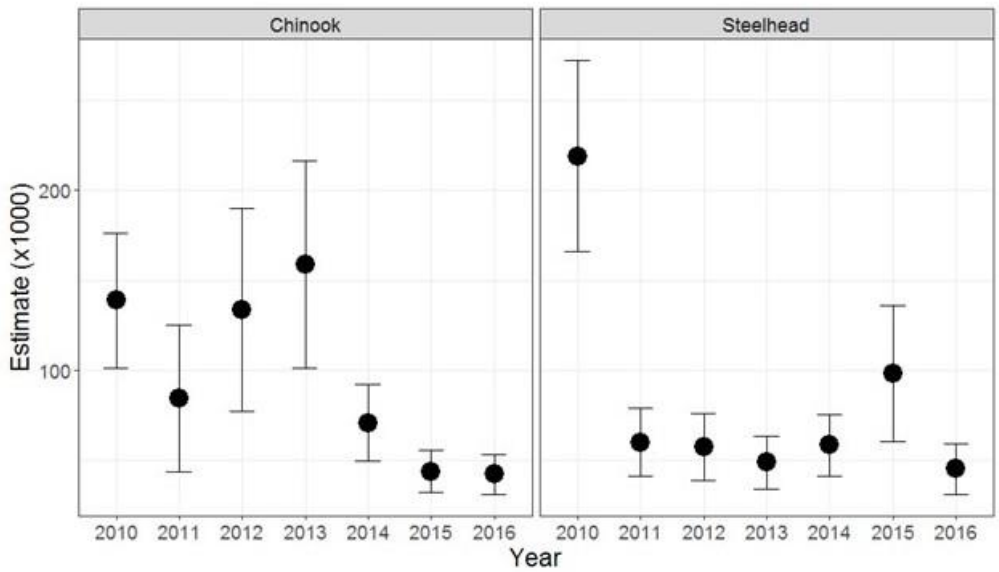

Figure 26. Spring Chinook and steelhead juvenile standing crop estimates for the Entiat River subbasin, 2010 – 2016.

Emigrants and Productivity

Entiat River, WA

During 2016, the USFWS Mid-Columbia Fish and Wildlife Conservation Office operated one RST on the Entiat River under the Entiat IMW. Trap operations were conducted 7 days a week between February and November when flow and water temperature permitted. A total of 10,088 fish were captured at the rotary screw trap and 7,184 salmonids were implanted with PIT tags. Natural origin juvenile spring Chinook and summer steelhead represent 26.7% and 7.0% of the total catch respectively. Point estimates of emigrant abundance (95% C.I.) for yearling and sub-yearling spring Chinook were 2,860 (± 1,628) and 22,346 (± 13,970), respectively. Summer steelhead emigrant abundance was estimated at 8,590 (± 1,426) (Figure 27) and the abundance of emigrating steelhead by age captured at the lower Entiat River rotary screw trap was estimated for 2010 – 2015 (Figure 28).

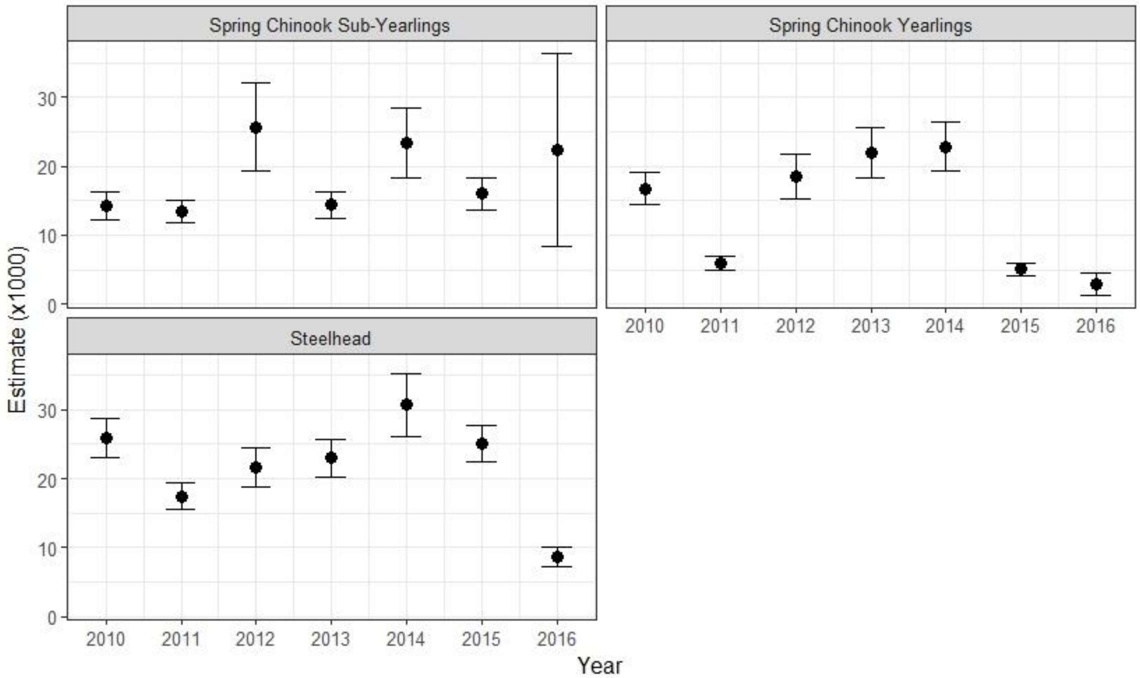

Figure 27. Estimates of emigrant abundance for spring sub-yearling and yearling Chinook and steelhead, 2010 – 2016. Data provided by USFWS Mid-Columbia Fishery Resource Office collected under BPA Project 2003-017-00.

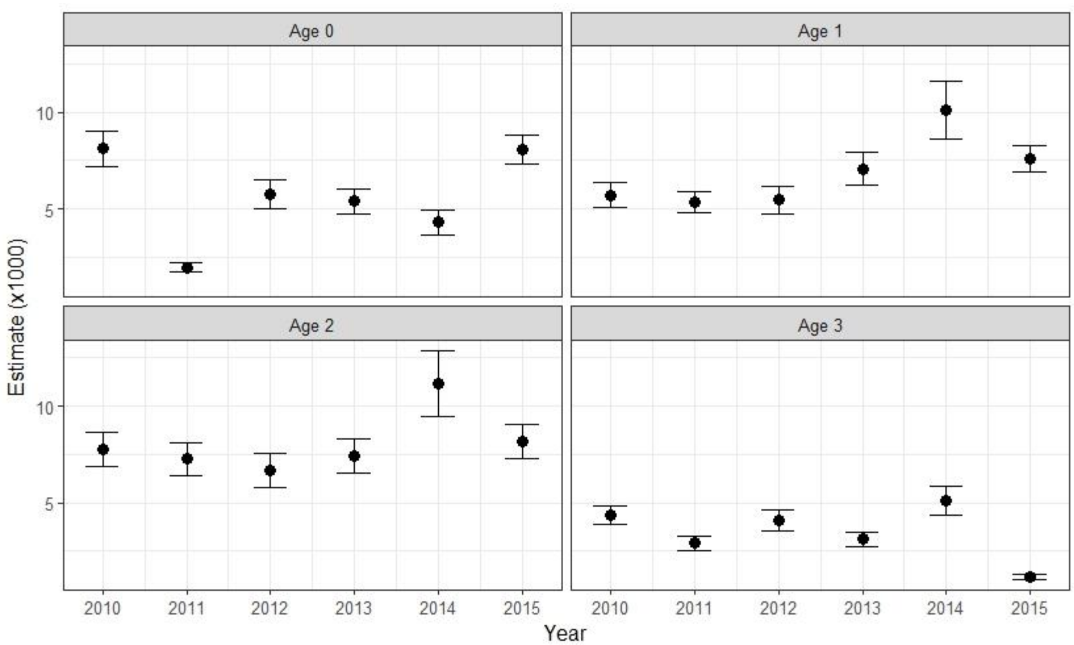

Figure 28. Calculated abundance of emigrating steelhead by age captured at the lower Entiat River rotary screw trap from 2010 – 2015. Data provided by USFWS Mid-Columbia Fishery Resource Office collected under BPA Project 2003-017-00.

Abundance estimates for yearling spring Chinook in 2016 allowed for the completion of 2014 brood year productivity estimates. Total egg deposition for 2014 brood year spring Chinook was estimated at 436,050 eggs. Deposition was based on 102 redds counted within the Entiat River basin (Fraser and Hamstreet 2015) multiplied by an estimated fecundity of 4,275 eggs. A total of 18,923 spring Chinook emigrants were estimated from the 2014 brood year. Egg-to-emigrant survival rate and emigrant-per-redd estimates were calculated at 4.34% and 186 fish, respectively for 2014 brood year for spring Chinook (Table 4). Mean fork length (±SD) of spring Chinook was 99.12 (±11.3) mm and 90.5 (±8.5) mm, for yearling and sub-yearling species respectively. Mean fork length (±SD) of summer steelhead was 134.7 (±35.8) mm. Spring Chinook smolt-to-adult return calculated for the 2010 brood year was estimated at 0.32% for sub-yearling, 0.76% for yearling out-migrants, and 0.49% for both juvenile life-histories combined.

Table 4. Estimated egg deposition (# of redds × estimated female fecundity), egg-to-emigrant survival rates, and emigrant per redd estimates for Entiat River wild spring Chinook juveniles, brood years 2002 – 2014. Data provided by USFWS Mid-Columbia Fishery Resource Office collected under BPA Project 2003-017-00.

Brood Year	Number of Redds	Estimated Egg Deposition	Estimated Number			Egg-to-Emigrant Survival (%)	Emigrant per Redd
			Sub-yearling	Yearling	Total		
2002	112	478,800	9,740	3,958	13,697	2.86% [a]	122 [a]
2003	108	461,700	9,123	5,349	14,472	3.13% [a]	134 [a]
2004	126	538,650	12,029	8,145	20,174	3.75% [a]	160 [a]
2005	148	632,700	13,386	9,090	22,477	3.55% [b]	152 [b]
2006	107	457,425	6,265	11,643	17,908	3.91%[c]	167[c]
2007	102	436,050	19,408	7,345	26,753	6.14%[c]	262[c]
2008	116	495,900	11,544	16,692	28,236	5.69%[c]	243[c]
2009	115	491,625	14,188	5,942	20,131	4.09%[c]	175[c]
2010	204	872,100	13,437	18,471	31,908	3.66%[c]	156[c]
2011	248	1,060,200	25,693	21,866	47,559	4.49%[c]	192[c]
2012	236	1,008,900	14,353	22,786	37,140	3.68%[c]	157[c]
2013	99	423,225	23,370	5,083	28,453	6.72%[c]	287[c]
2014	102	436,050	16,063	2,860[d]	18,923	4.34%[c]	186[c]

[a] Derived from upper trap (rkm 11.0) estimates.
[b] Derived from upper trap (rkm 11.0) sub-yearling and lower trap (rkm 2.0) yearling estimates.
[c] Derived from lower trap (rkm 2.0) estimates.
[d] Estimates derived from partial trapping season

Spring Chinook SAR estimates for all life-history groups (yearling, sub-yearling and combined) for the 2010 brood year were above or equal to the 9 year averages of 0.20%, 0.76%, and 0.46%, respectively. The 2010 brood year showed similar trends as previous brood years, with a higher SAR for spring Chinook tagged as yearlings as compared to those tagged as sub-yearlings (Table 5).

Table 5. Estimated smolt-to-adult return (SAR) for sub-yearling, yearling and adult wild spring Chinook in the Entiat River for brood years 2001 to 2010. Data provided by USFWS Mid-Columbia Fishery Resource Office under BPA Project 2003-017-00.

Brood Year	Total Observations			SAR		
	Sub-yearling	**Yearling**	**Adult Returns**	**Sub-yearling**	**Yearling**	**Combined life-histories**
2001	n/a	2	2	0.00%	0.51%	0.51%
2002	0	5	5	0.00%	0.70%	0.39%
2003	1	5	6	0.04%	0.38%	0.15%
2004	3	5	8	0.12%	0.25%	0.19%
2005	3	1	4	0.15%	0.16%	0.15%
2006	17	100	117	0.56%	1.51%	1.21%
2007	23	25	48	0.41%	1.09%	0.61%
2008	21	42	63	0.34%	0.94%	0.59%
2009	6	9	15	0.16%	1.34%	0.34%
2010	12	17	29	0.32%	0.76%	0.49%

Lemhi River, ID

To estimate steelhead productivity, we first estimated length-at-age relationships by fitting a linear model that predicts the fork length, y, based on the known age class, a to provide an estimate of the mean, μ_a, and standard deviation, σ_a, of fork lengths within each age class (a). We then fit a mixture model to the entirety of the length data for each year. The distribution of observed lengths is a combination of distinct log-normal distributions, one from each age class. We estimated the parameters of this mixed distribution of lengths, including the proportion of fish that make up each age class (π_a). We then estimated the probability that a fish of a given length is in each age class using Bayes' theorem.

$$P(X = a|Y = y) = \frac{f(y|X = a)P(X = a)}{f(y)}$$

The model distinguishes well between age-0 steelhead and other age classes, but performs only marginally better than random for distinguishing between older age classes (Figure 29). We assume the length-at-age relationship was constant through time because we only had one year of age-length data.

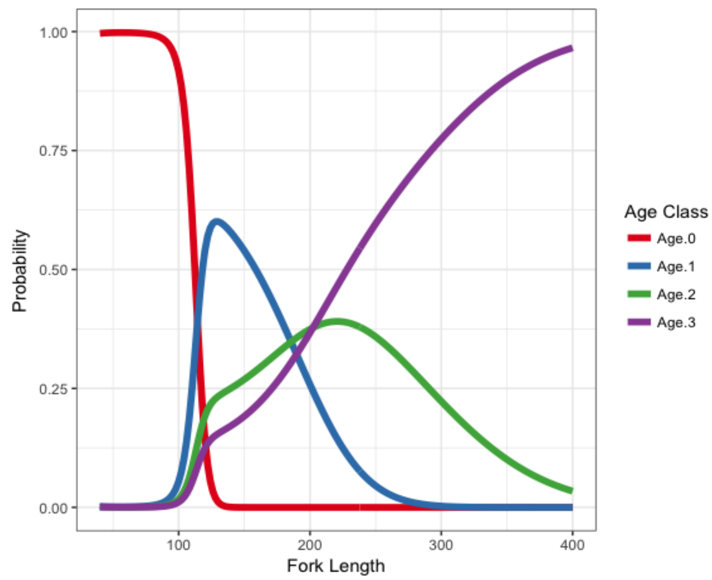

Figure 29. Probability that a fish of a given fork length is in each age class for steelhead in the Lemhi River. Every vertical slice sums to one.

Productivity estimates for both spring Chinook salmon and steelhead populations in the Lemhi River shows a lot of year-to-year variability, making it difficult to extract any sort of trend in productivity over this period (Tables 6 and 7, Figures 30 and 31). A longer time-series of productivity estimates will be necessary to filter out those year effects and estimate a true underlying trend.

Table 6. Estimates (CV) of adult escapement, emigrants, and productivity by brood year for natural origin spring/summer Chinook salmon in the Lemhi River.

Brood Year	**Adults**	**Emigrants**	**Productivity**
2011	267 (0.129)	16842.3 (0.032)	63.1 (0.134)
2012	83 (0.29)	21825.5 (0.035)	263 (0.309)
2013	393 (0.106)	31496 (0.032)	80.1 (0.111)
2014	464 (0.117)	79130 (0.024)	170.5 (0.121)
2015	718 (0.098)	53385 (0.035)	74.4 (0.105)

Table 7. Estimates (CV) of adult escapement, emigrants, and productivity by brood year for natural origin steelhead in the Lemhi River.

Brood Year	**Adults**	**Emigrants**	**Productivity**
2010	417 (0.095)	19477 (0.047)	46.7 (0.107)
2011	314 (0.089)	8770 (0.049)	27.9 (0.101)
2012	347 (0.104)	11469 (0.046)	33.1 (0.115)
2013	334 (0.081)	10861 (0.045)	32.5 (0.093)
2014	266 (0.106)	5311 (0.064)	20 (0.125)

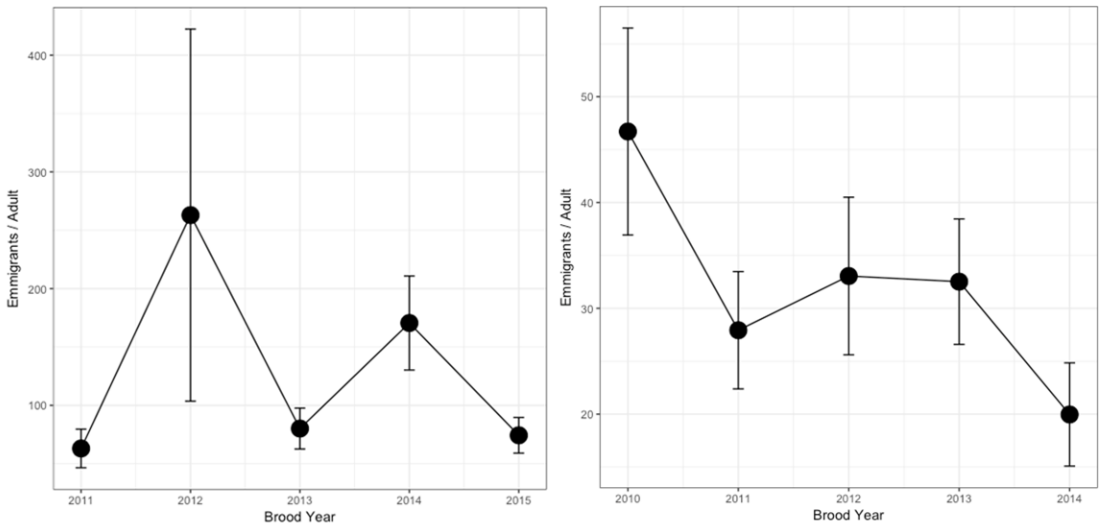

Figure 30. Estimates of productivity of natural origin spring/summer Chinook salmon (left panel) and natural origin steelhead (right panel) in the Lemhi River, plotted against brood year. Bars depict 95% confidence intervals.

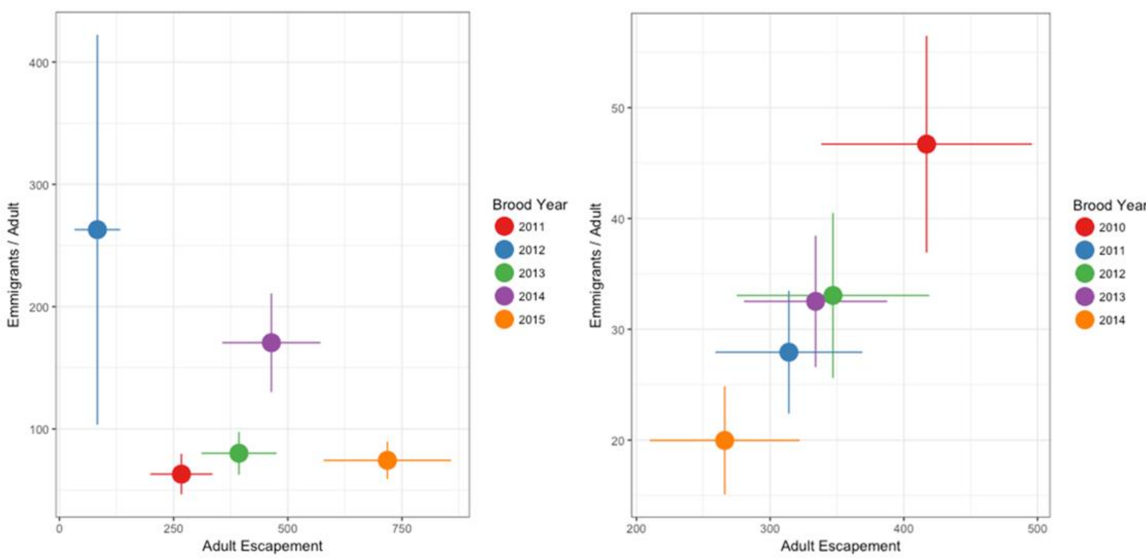

Figure 31. Estimates of productivity of natural origin spring/summer Chinook salmon (left panel) and natural origin steelhead (right panel) in the Lemhi River, plotted against estimates of escapement. Bars depict 95% confidence intervals.

Effectiveness Monitoring

Bridge Creek IMW, OR

The responses to the addition of beaver dam analogs (BDAs) for both steelhead and their habitat was dramatic as reported previously and in a recently published paper (Bouwes et al. 2016a, Attachment C). However, in 2016 densities dropped dramatically in Bridge Creek to pre-restoration densities (Figure 32). Although this is not enough to bring the post-restoration average down appreciably compared to what we published recently (Bouwes et al. 2016a), this does bring into question the long-term benefits of the large increase in beaver dam building activity.

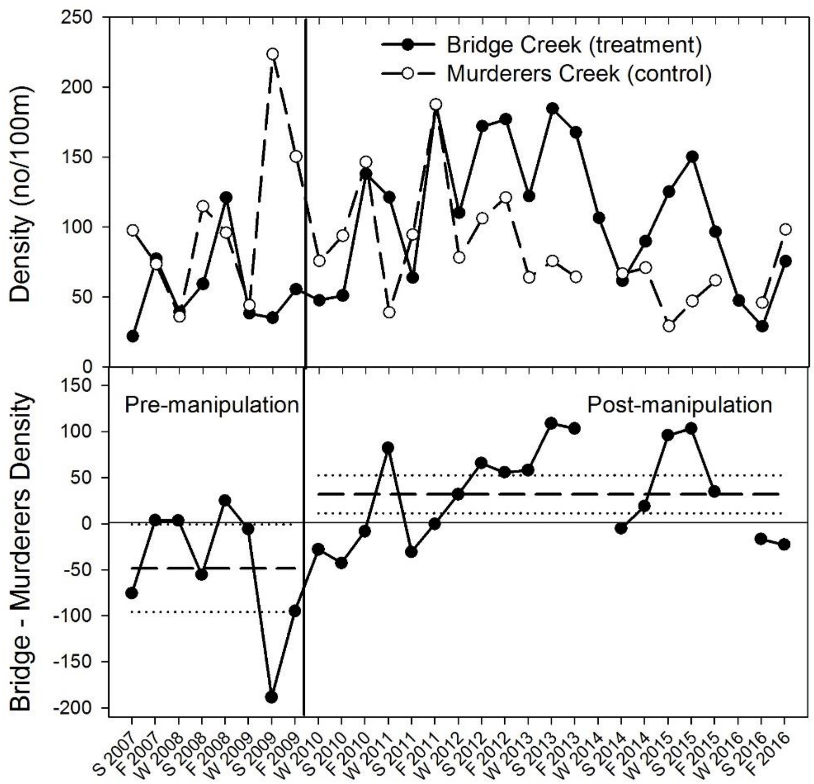

Figure 32. Time series of juvenile steelhead density (number/100m) estimates for the treatment Bridge Creek and control Murders Creek watersheds (upper panel) and difference between Bridge Creek and Murderers Creek densities (lower panel) for spring, fall, winter (S,F,W). Vertical line represents the date of the manipulation. In the lower pane, the dashed lines represent the mean value, and the dotted lines the upper and lower 90% confidence intervals, for pre- and post-manipulation.

The recent decline in juvenile densities might be explained by several environmental variables: beaver ponds go through an evolution from pond to wetland to meadow (Naiman et al. 1988) that could have large impacts on the temporal dynamics and benefits to fish habitat. Alternatively, or in addition to this evolution in habitat types, lower than normal discharge in 2014 and 2015 resulted in higher than normal stream temperatures, especially in 2015. During this time, discharge was almost at base flows during the upstream migration of steelhead spawners, rather than the high flows that are typical during the spawning season, and we observed very little upstream migration of adults in the spring of 2015. In addition, temperatures reached lethal levels in lower portions of the stream during the summer low flow period. In contrast, Murderers Creek is further upstream in the John Day drainage and does not reach these extreme temperatures. Differences between Bridge and Murderers Creek densities may be a result of the stream temperatures reaching lethal thresholds in Bridge Creek. However, it should be noted that beaver dams have been shown to have a cooling effect in Bridge Creek, particularly to summer daily maximum temperatures (Weber et al. 2017), so it is possible that without the large increase in beaver dams following the installation of beaver dam assist structures, the 2015 water year could have had a much larger negative impact on juvenile steelhead survival and therefore densities than we observed.

Entiat IMW, WA

To date we have 4 years of post-treatment data from the 2012 actions and 2 year of post-treatment data from the 2014 actions. Here we present results at the watershed scale (mainstem Entiat [treatment] compared with the Mad River [control]) and the valley segment scale (valley segments 1 and 3 are treatments, valley segment 2 and the Mad River are controls).

Chinook

Results from the Chinook analysis point to difficulties with small sample sizes and interactions between sampling design and Chinook life history patterns that limit analysis conclusions. At the watershed scale there was a significant decrease in Chinook abundances in restoration compared with control areas between Restoration 1 and Restoration 2 periods (Figure 33 right panel), but a similar pattern was not observed at the valley segment scale (Figure 33 left panel). Given that valley segment 2 is modeled as a control at the valley segment scale but is included in the restoration group of the Entiat River at the watershed scale, the different conclusions from analysis at these two spatial scales points to declines in valley segment 2 abundances. It is unlikely this decline in valley segment 2 is due to restoration activity but it does cause difficulties separating natural variation from restoration effects.

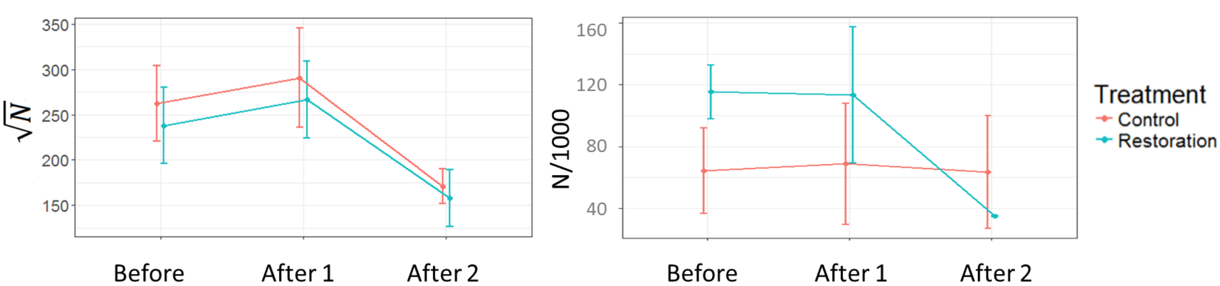

Figure 33. Comparison of Chinook abundance (transformed for normality) at the valley segment scale (left panel) and at that watershed scale (right panel) and standard errors in restoration and control areas.

Analysis for differences due to treatment of overwinter survival for Chinook at the watershed scale were unsuccessful since very low Chinook numbers in the Mad River frequently cause low capture success during sampling which increases uncertainty around survival calculations. Analysis for treatment effect at the valley segment scale showed a decreased survival in restoration areas compared with control areas between Restoration 1 and Restoration 2 periods ($p = 0.10$) (Figure 34).

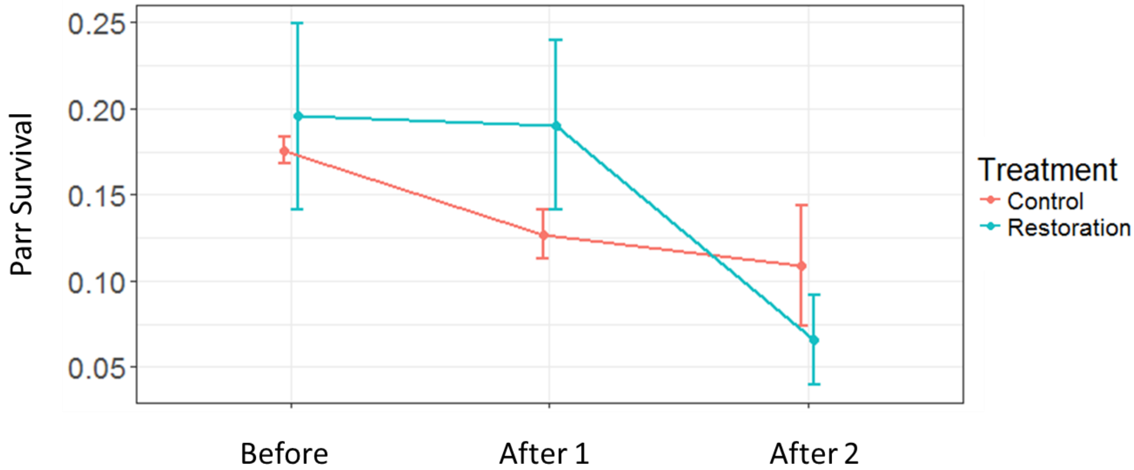

Figure 34. Comparison of Chinook over-winter survival and standard errors in restoration and control areas at the valley segment level in the Entiat IMW.

This trend may be explained by confounding interactions between Chinook life history and sampling design. Between Restoration 1 and Restoration 2 periods Chinook at control valley segments significantly increased in fork length compared with fish at restoration valley segments during the summer but not during the winter (Figure 35). The lack of significant difference in over-winter specific growth rate (Figure 36) implies that between summer and winter there was an immigration of larger fish into restoration valley segments. Valley Segment 1 is a restoration valley segment and is also at the mouth of the Entiat River near where it enters the Columbia River. It is likely that larger fish are entering Valley Segment 1 during the winter to prepare for emigration which occurs with spring flooding, typically in March. In 2016 spring flooding occurred earlier than usual and high flows delayed RST operations and it is possible more emigrating fish were missed than usual during this year which would result in depressed survival estimates.

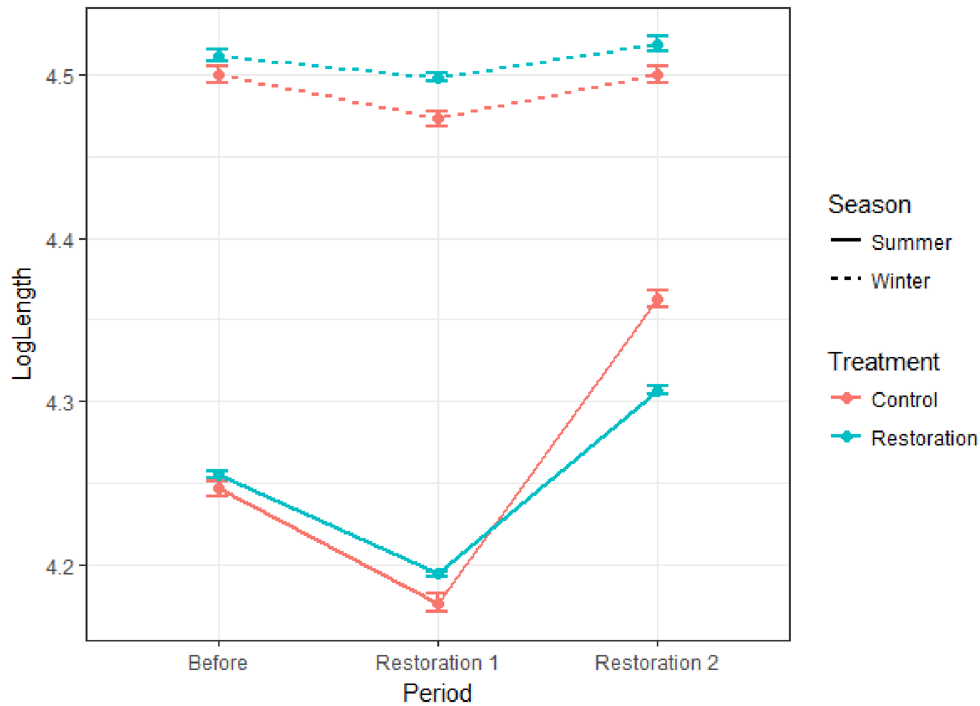

Figure 35. Comparison of Chinook fork length (mm, transformed for normality) and standard errors in restoration and control areas at the valley segment level in the Entiat IMW.

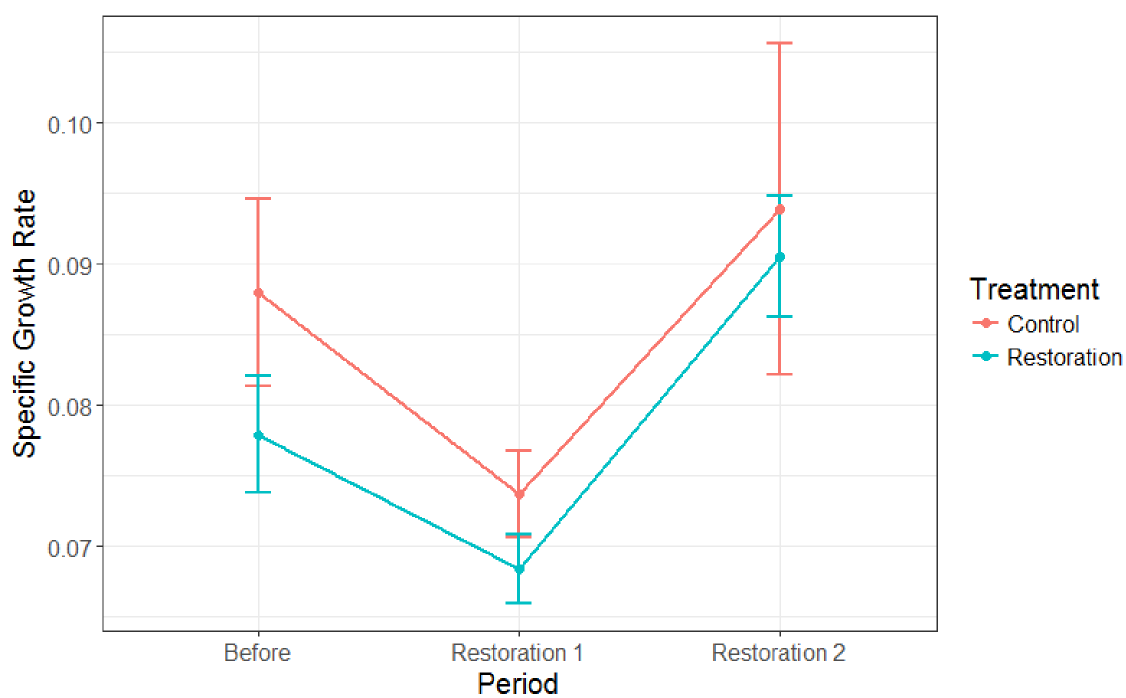

Figure 36. Comparison of Chinook specific growth rate (mm fork length per day) and standard error in restoration and control areas at the valley segment level in the Entiat IMW.

Steelhead

No significant response was detected for steelhead abundance at either the valley segment or watershed scales (Figure 37) but a significant increase in survival in post-restoration was observed for steelhead at the valley segment scale (Figure 38).

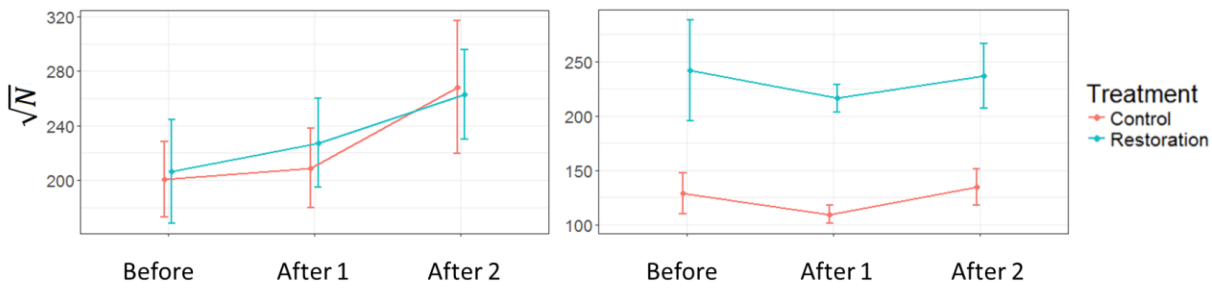

Figure 37. Comparison of steelhead abundance (transformed for normality) and standard errors in restoration and control areas at the valley segment scale (left panel) and watershed scale (right panel).

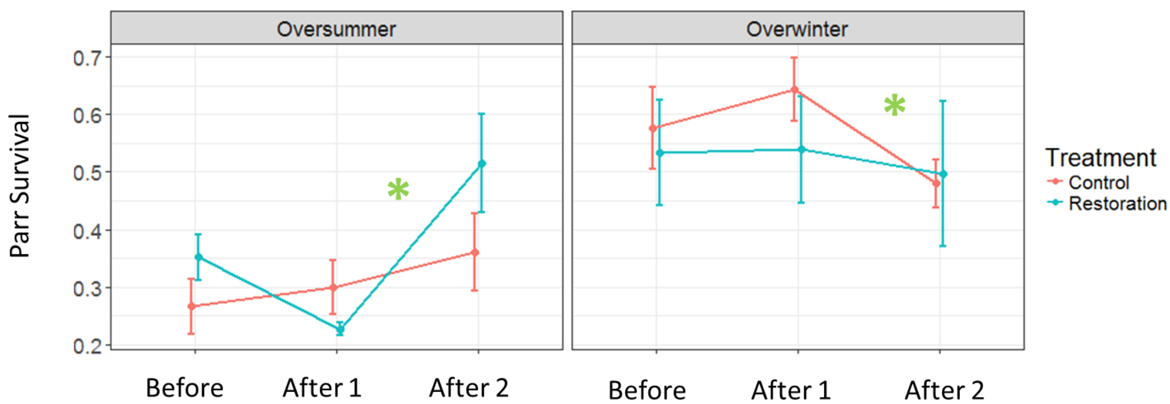

Figure 38. Comparison of steelhead oversummer and over-winter survival and standard errors in restoration and control areas at the valley segment level in the Entiat IMW.

At both the valley segment (Figure 39) and watershed (data not shown) scales between Restoration 1 and Restoration 2 periods steelhead size increased significantly at restoration sites compared with control sites, although this depended on age class and season. At the valley segment and watershed (data not shown) scale for all age classes and seasons specific growth rate decreased significantly ($p = 0.0012$) in restoration areas compared with control areas between Before and Restoration 1 periods (Figure 40). Between Restoration 1 and Restoration 2 periods there was a trend ($p = 0.065$) for decreases in specific growth rate in restoration compared with control areas. The decrease in growth rate is likely due to the increase in fish size as larger fish tend to have slower growth rates.

Figure 39. Comparison of steelhead fork length (mm) and standard error in restoration and control areas at the valley segment level by capture season and age class. * = significant (p <0.05) increase in restoration compared with control, * = significant (p <0.05) decrease in restoration compared with control.

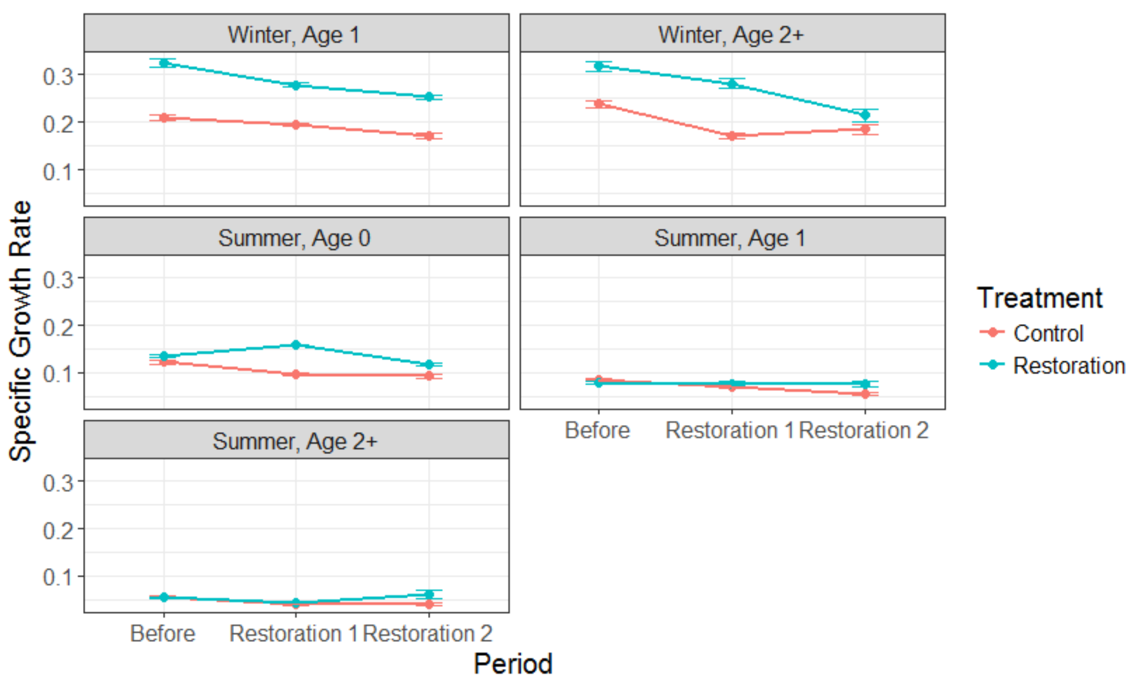

Figure 40**.** Comparison of steelhead specific growth rate (mm fork length per day) and standard error in restoration and control areas at the valley segment level by capture season and age class in the Entiat IMW.

Lemhi IMW, ID

Amonson Reach

Fundamental differences among surveys (i.e., site length, discharge) prevented us from determining changes in carrying capacity resulting from restoration actions (Figure 41). However, we did look at indicators likely correlated with carrying capacity and how those indicators changed through surveys (Table 8). Habitat in the main channel has improved post-restoration and the side channel adds additional rearing habitat that can be used as a thermal and velocity refuge with increased cover and complexity.

Figure 41. Pre- (right panel) and post-restoration side channel (middle panel) and main channel (left panel) habitat in the Amonson Reach.

Table 8. Habitat measurements showing differences in metrics among the four surveys completed at the Amonson Reach.

Habitat Metric	Before	2013 Main Channel	2016 Main Channel	Side Channel
Avg % of pool tail substrates comprised of fine sediment <2mm	65.00	5.33	30.37	55.89
Number of large wood pieces per 100 meters within bankfull channel	6.74	10.3	8.8	32.41
% of wetted area with fish cover	14.09	5.1	7.7	53.42
% of wetted area as slow water/pool channel units	22.18	55.8	31.52	59.35
Sinuosity	1.31	1.58	1.59	1.32

Lee Creek

Time constraints only allowed for surveying of a 450 m reach of Lee Creek, which we are assuming is representative of the entire site. Riparian vegetation establishment above the culvert, where the stream is approximately 130 m from the old channel, has been unsuccessful (Figure 42 and Table 9) and it may be that the banks are too high above the water table to support riparian vegetation. Conversely, below the culvert and outside of the survey area, where the stream bed is in a similar location as the old channel, vegetation is thriving. Based on a QRF model we predict that Chinook parr density at the site has increased from 0.29 fish/m in 2013 to 0.34 fish/m in 2016 (Table 8); however, total estimated Chinook parr capacity has decreased from 427 in 2013 to 377 in 2016.

Figure 42. The lowest extent of Lee Creek pre- (left panel) and post-restoration changes in vegetation (middle panel) and 10 cm DEM of the newly restored channel (right panel).

Table 9. Habitat measurements and predicted juvenile Chinook response pre-treatment (2013) and post-treatment (2016) at the Lee Creek restoration site.

Habitat Metric	2013	2016
Sinuosity	1.13	1.15
Bankfull Width to Depth Ratio Avg	32.26	15.04
Pool Residual Depth	0.40	0.24
Pool Frequency	1.61	5.74
Avg % of Fine Pool Tail Substrates <2mm	26.1	7.87
Total Fish Cover	5.90	18.98
Substrate D16	8.00	18.00
LWD Frequency – Bankfull	4.61	23.88
Predicted total Chinook parr summer rearing capacity	427.00	377.00
Predicted Chinook parr density (fish/m^2)	0.29	0.34

Little Springs Creek

Fish passage into Little Springs Creek is now open, creating an additional 6,962.5 m of stream habitat available to Chinook parr during summer months (Figure 43). Based on habitat measurements taken from CHaMP surveys in Little Springs Creek, we estimate that Little Springs Creek can now support up to 1.59 fish/m and has a carrying capacity of ~ 11,000 Chinook parr (Table 10).

Figure 43. Little Springs Creek site pre- (left panel) and post-restoration (middle panel). The right panel shows a CHaMP survey encompassing the entirety of the restoration site illustrated by a 10cm CHaMP DEM overlaid on satellite imagery.

Table 10. Habitat measurements pre-treatment (2013) and post-treatment (2016) at the Little Springs Creek restoration site.

Habitat Metric	2012	2013	2016
Estimated total Chinook parr capacity	1108	1008	899
Estimated Chinook parr density	0.37	0.33	0.29
Substrate D50	16	10	19
Substrate: % Cobble	4.31	6.73	18.44
Substrate >6mm	70.09	98.47	49.26
Gradient	0.66	0.64	0.63
Total Fish Cover	3.31	22.53	18.47
LWD Frequency – Bankfull	5.76	6.27	7.40
CV of Thalweg Depths	0.52	0.46	0.39
Bankfull Avg Width to Depth Ratio	12.50	14.77	70.37

Eagle Valley Ranch

To date we can characterize the habitat at the newly restored side channel at Eagle Valley Ranch, but more years of monitoring will be needed to determine the response to habitat actions (Figure 44, Table 11).

Figure 44. Eagle Valley Ranch site pre (left panel) and post-restoration (right panel).

Table 11. Habitat measurements and estimated Chinook parr densities from the 2016 survey of the newly restored side channel at the Eagle Valley Ranch site.

Habitat Metric	Value
Estimated maximum Chinook parr density	0.33
Estimated Chinook parr total capacity	1453
Bankfull Area (m^2)	5425
Bankfull Width to Max Depth Ratio Avg. CV	0.165
Percent Pools	44%
Substrate <6mm	14.6%
Fish Cover Total	21%
Gradient	0.45
Bankfull Width to Max Depth Ratio Avg	10.92
Sinuosity	1.308
Substrate: % Cobbles and Boulders	0.31
LWD Frequency	0.65

Lemhi IMW

Actions to date in the Lemhi have nearly doubled the total length of stream available to anadromous salmonids, and have resulted in a 22% increase in stream area and 19% increase in pool habitat available to anadromous salmonids. A QRF model was used to translate habitat availability by channel type (Table 12) into capacity estimates by channel type for each of the Lemhi tributaries considered for reconnection.

Table 12. Steelhead and spring/summer Chinook salmon capacity (fish/meter) by channel type for Lemhi River tributaries.

Species	Stream	Channel-Type							
		Cascade	**Confined**	**Island-Braided**	**Meandering**	**Plane-Bed**	**Pool-Riffle**	**Step-Pool**	**Straight**
Steelhead	Agency			3.09		1.90	1.97	1.92	
	Big Eightmile				1.66	1.96	2.02		
	Big Springs						1.33		
	Big Timber		2.12	1.86	1.20	1.95	1.98	2.02	2.66
	Bohannon	2.28		3.22		2.25	1.89	2.60	
	Canyon				1.37	1.90	2.00		
	Hawley				1.31	1.96	2.05	1.92	
	Hayden		1.79	1.98		2.13	2.08	2.18	2.88
	Kenney			2.69		1.97	2.00	2.08	
	Lee						1.80		
	Lemhi Mainstem		2.27	2.57	1.59		1.79		4.33
	Little Eightmile	1.92			1.64	2.03	1.89	2.10	
	Little Springs				0.65				
	Mill						1.66		
	Pattee	1.95		3.15		1.93	1.94	2.21	
	Texas				1.17				
	Wimpey	2.41		3.44		2.07	1.84	2.47	
Spring/summer Chinook Salmon	Agency			4.85		1.71	2.12	0.89	
	Big Eightmile				4.45	3.14	3.29		
	Big Springs						1.56		
	Big Timber		0.89	3.17	3.14	2.80	3.37	3.25	2.99
	Bohannon	0.00		4.92		1.15	2.14	0.01	
	Canyon				4.02	3.50	3.45		
	Hawley				3.13	3.46	3.96	2.74	
	Hayden		2.46	3.27		2.36	3.34	1.60	3.29
	Kenney			7.18		5.18	5.67	4.21	
	Lee						1.84		
	Lemhi Mainstem		4.01	5.67	4.14		3.61		4.30
	Little Eightmile	0.00			4.53	1.70	2.49	1.10	
	Little Springs				0.78				
	Mill						1.35		
	Pattee	0.00		4.88		1.88	2.11	0.69	
	Texas				3.44				
	Wimpey	0.00		2.91		1.21	1.77	0.25	

Survival and capacity estimates for pre-existing, reference, and reconnected habitat (inclusive of in-stream restoration actions) were used in the LCM to estimate the change in habitat availability, adult escapement, productivity (smolts per spawner), and total juvenile production for all actions completed through 2015 (Figure 45). Predicted changes in freshwater productivity can be viewed relative to the 3% and 7% improvements targeted in the 2008 BiOp for steelhead and spring/summer Chinook salmon, respectively. Model estimates suggest that tributary reconnection and habitat restoration actions in the Lemhi River will be sufficient to exceed survival improvements identified for steelhead (10% improvement relative to a 3% target), but are likely to fall short for spring/summer Chinook salmon (3% improvement relative to a 7% target).

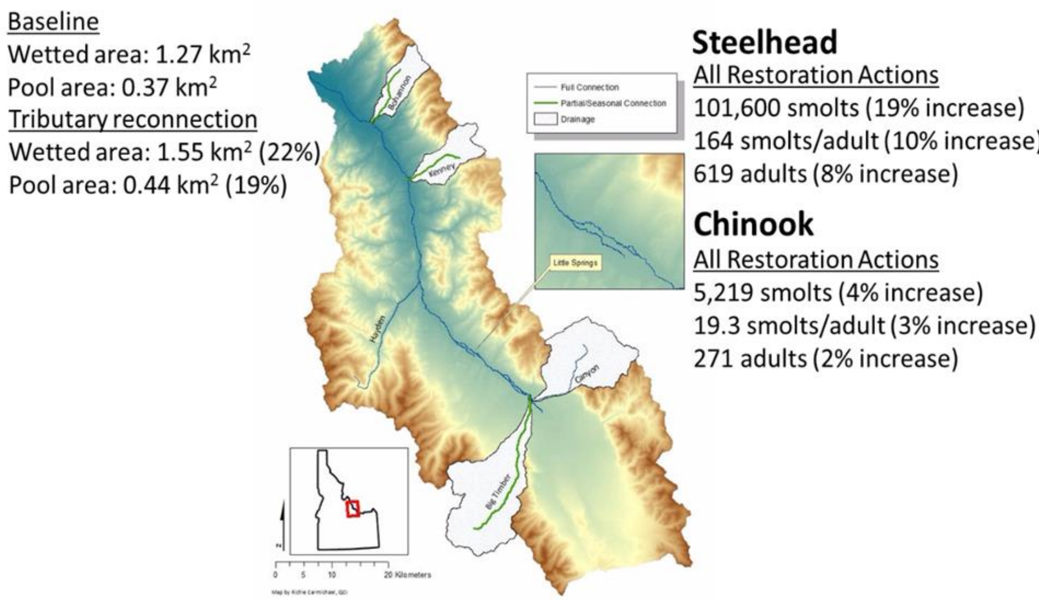

Figure 45. Estimated change in habitat availability (upper left panel) and fish production, freshwater productivity, and adult escapement from Lemhi River habitat restoration/reconnection actions completed through 2015.

Ecohydraulic Models

Capacity is difficult to validate for species that are rare, such as ESA-listed species, but if habitat capacity is related to habitat quality, then we may still find a relationship between observed versus predicted fish density, albeit lower than a 1:1 relationship. As reported previously, both QRF (ISEMP/CHaMP 2015, 2016) and NREI (Wall et al. 2015) models have shown a relationship between observed versus predicted fish density. All the ecohydraulic models are producing capacity estimates that can be used to guide restoration planning and as inputs into LCMs.

HSI/FIS

We have developed both HSI and FIS spawner models that are automated across all CHaMP site surveys. In addition, these models have been built for the CHaMP Workbench, allowing anyone to quickly and easily pull CHaMP data and run the models for project specific evaluations. These models are also being used to help identify configurations of geomorphic unit types that can be obtained from reach typing and therefore estimated across a stream network that best support redd construction. HSI and FIS models have proven useful for predicting spawning habitat capacity for adults (e.g., Figure 46).

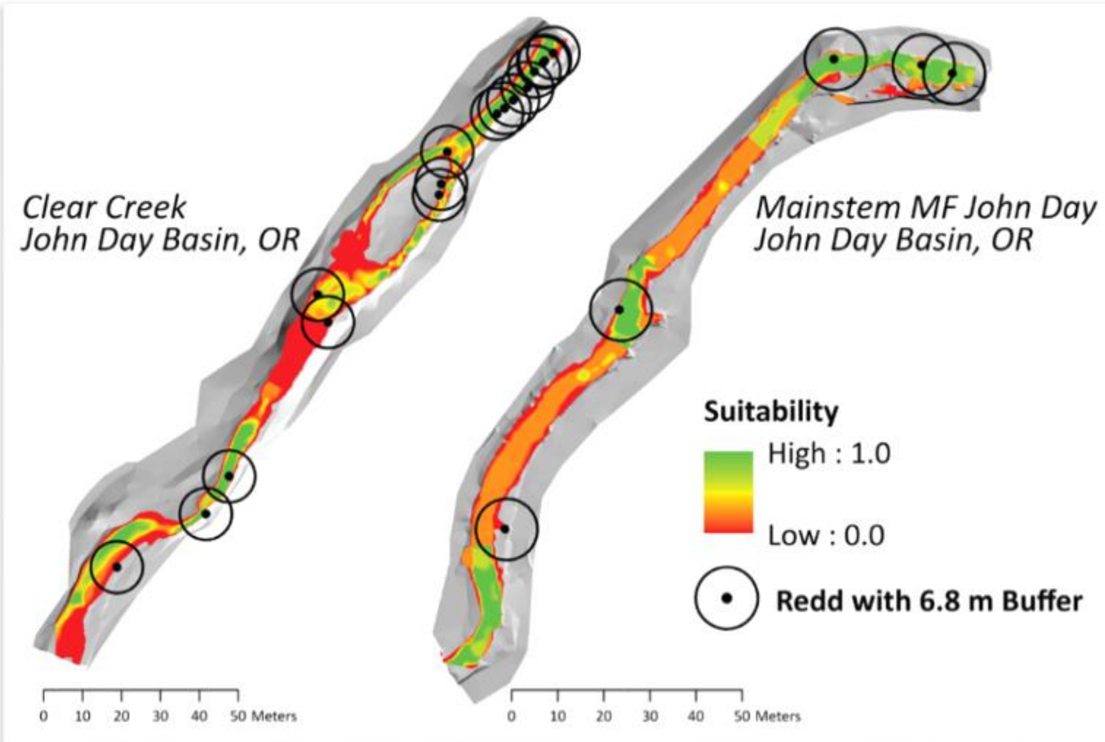

Figure 46. Comparison of FIS Chinook spawner habitat suitability predictions and observed habitat use (2013 and 2014 spawning events, combined) for two CHaMP sites within the John Day River Basin, Oregon (Redd locations from Bare et al. 2015).

QRF

A QRF model has been developed for Chinook based on select CHaMP metrics to estimate redd capacity (Table 13) and summer parr rearing capacity (Table 14). Work is ongoing to develop similar models for steelhead.

Table 13. Habitat covariates selected for the 2017 Chinook salmon redd capacity QRF model.

Metric	Description
MeanU	Average annual discharge (cfs). Mean daily flow, averaged over a year, calculated from the FLoWS model network. http://www.fs.fed.us/rm/boise/AWAE/projects/SpatialStreamNetworks.shtml
Elev_M	Elevation (m)
DistPrin1	Disturbance index including % urban, % agricultural, % impervious surface, and road density
Mx8dMean0813_0914	Maximum of 8-day mean temperatures for the period of 08/13 – 09/14. Averaged across 2011 – 2014
SubEstGrvl	Percent of coarse and fine gravel (2 – 64 mm) within the wetted site area

Table 14. CHaMP habitat metrics included in the summer Chinook parr rearing capacity QRF model.

Metric Category	Metric	Description
Channel Unit	Slow Water Frequency	Number of Slow Water/Pool channel units per 100 meters.
Complexity	Thalweg to Centerline Length Ratio	Ratio of the thalweg (Site Length Thalweg) and wetted centerline (Site Length Wetted) lengths.
Complexity	Wetted Width to Depth Ratio CV	Retired. Coefficient of Variation of wetted width to depth ratios derived from cross-sections.
Cover	Fish Cover: Total	Percent of wetted area with the following types of cover: aquatic vegetation, artificial, woody debris, and terrestrial vegetation.
Disturbance	Disturbance Index	Disturbance index that includes measures of % urban, % agricultural, % impervious surface, and road density (Whittier et al. 2011)
Riparian	Riparian Cover: Big Tree	Percent aerial coverage from big trees (> 0.3 m DBH) in the canopy.
Size	Bankfull Width to Depth Ratio Avg	Average width to depth ratios of the bankfull channel measured from cross-sections. Depths represent an average of 10 depths along each cross-section.
Size	Discharge	The sum of station discharge across all stations. Station discharge is calculated as depth x velocity x station increment for all stations except first and last. Station discharge for first and last station is 0.5 x station width x depth x velocity.
Substrate	Substrate: D16	Diameter of the 16^{th} percentile particle derived from pebble counts.
Substrate	Substrate < 6mm	Average percentage of pool tail substrates comprised of sediment < 6mm.
Temperature	7dAMGtr18	Number of 7-day average of daily maximum (7dAM) values between July 15^{th} and August 21^{st} that are greater than 18° C. Relates to salmon and trout rearing and migration.
Temperature	SummerHourlyAverageTemp	Average of all hourly temperature measurements collected July 15^{th} – August 31^{st}.
WaterQuality	Conductivity	Measure of concentration of ionized materials in water, or the ability of water to conduct electrical current.
Wood	Large Wood Frequency: Wetted	Number of large wood pieces per 100 meters within the wetted channel.

We have predicted capacity at all CHaMP sites (e.g., Lemhi River Chinook summer parr and Chinook redd capacity, Figure 47). For those CHaMP sites that have been sampled in multiple years, we calculated the mean of the habitat metrics among years to make predictions. We used the 90^{th} quantile of predicted fish density as a proxy for carrying capacity and for redd density each prediction is for the 1 rkm surrounding the x-site for each of 116 CHaMP sites. We can also fit an extrapolation model using GAAs from the list of master sample sites that CHaMP sites were originally selected from to provide continuous estimates of capacity across the network (e.g., Figure 48).

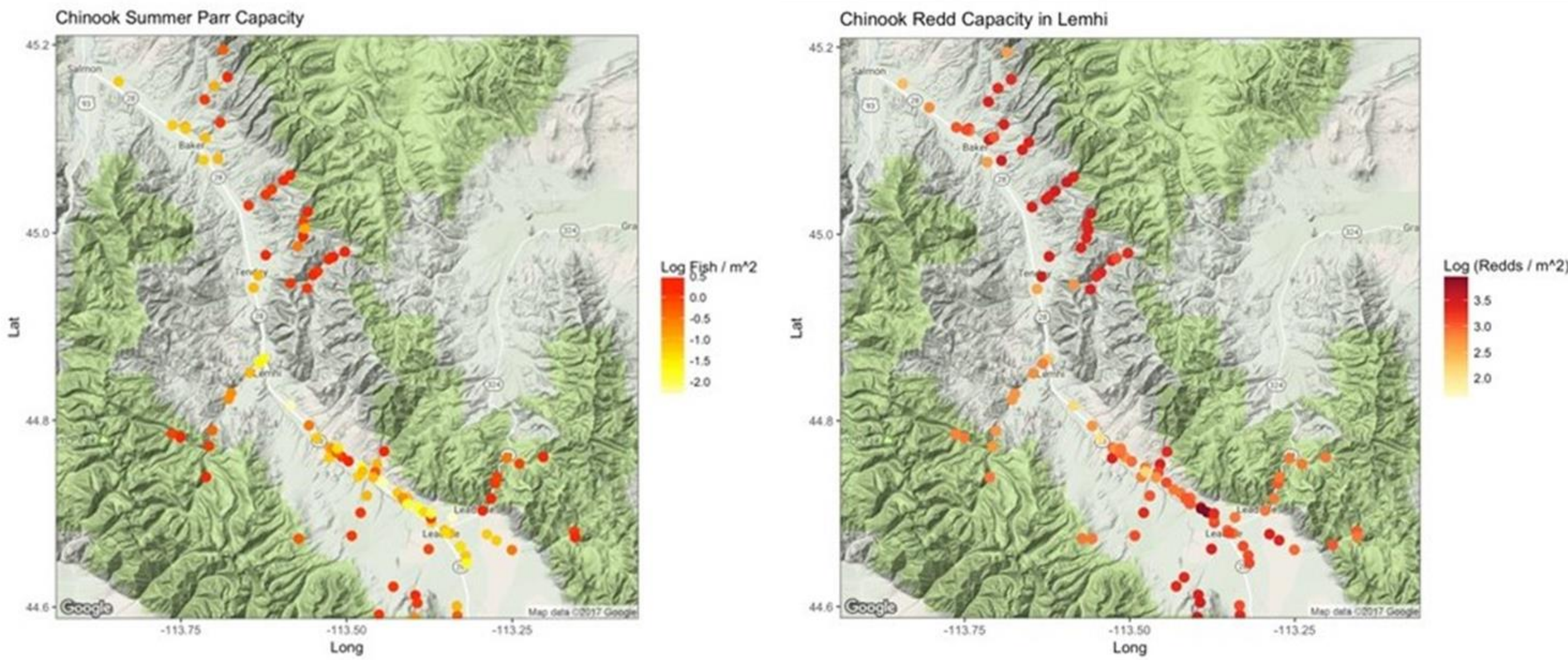

Figure 47. Predictions of carrying capacity (fish/m^2) and predicted redd capacities at all CHaMP sites in the Lemhi River.

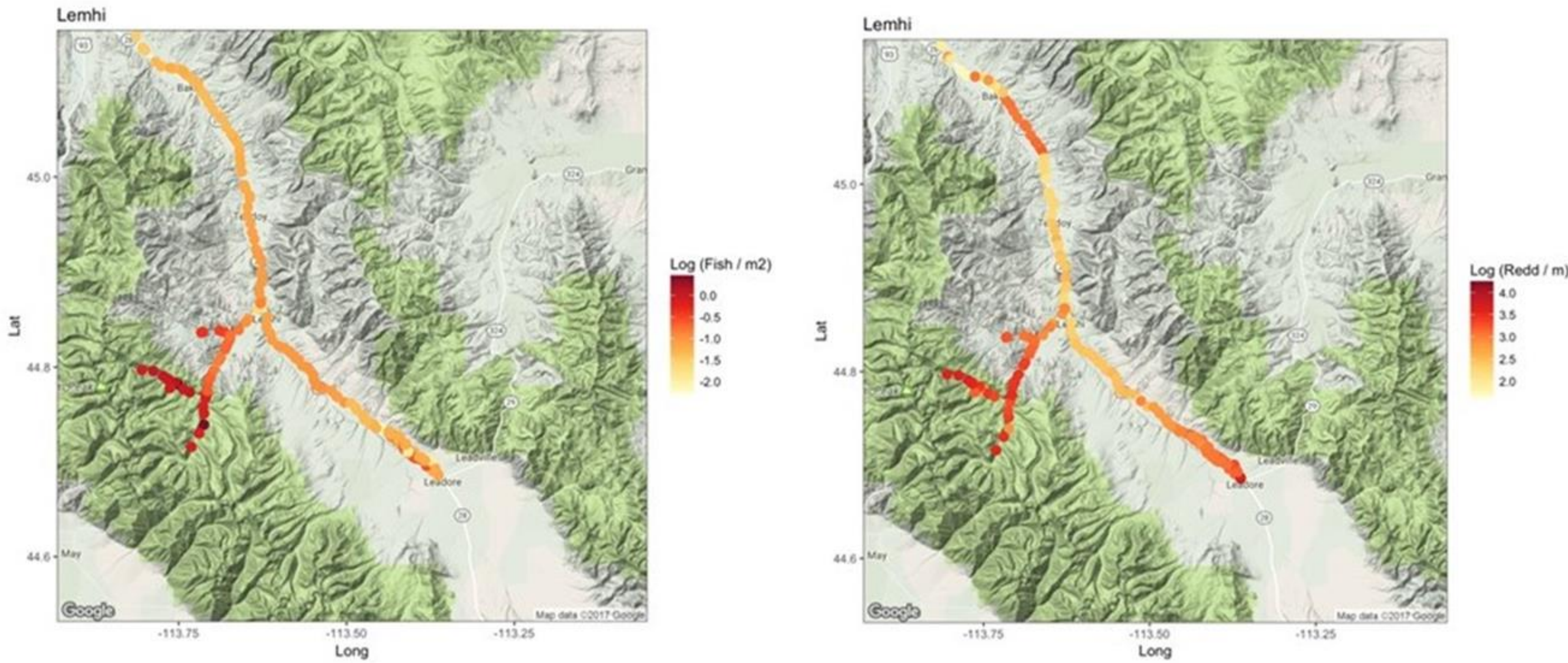

Figure 48. Continuous estimates of parr summer capacity (parr/m^2) (left panel) and redd capacity (redds/m) (right panel) for the Lemhi River based on extrapolating QRF site-level estimates into areas within the current spring/summer Chinook range.

NREI

The NREI model has streamlined code and can be run by experienced analysts with key points of user input/QAQC. We have used the model to simulate NREI and carrying capacity for 570 visits throughout all CHaMP watersheds, with at least one simulation for each CHaMP site that has a hydraulic model solution. Reach/site-level carrying capacities have been estimated for spring Chinook and/or steelhead (e.g., Figure 49). The NREI model currently produces these estimates over the observed range throughout the CRB of drift, temperature and fish sizes so that lookup tables can be used for various combinations of these input variables to increase flexibility of user options.

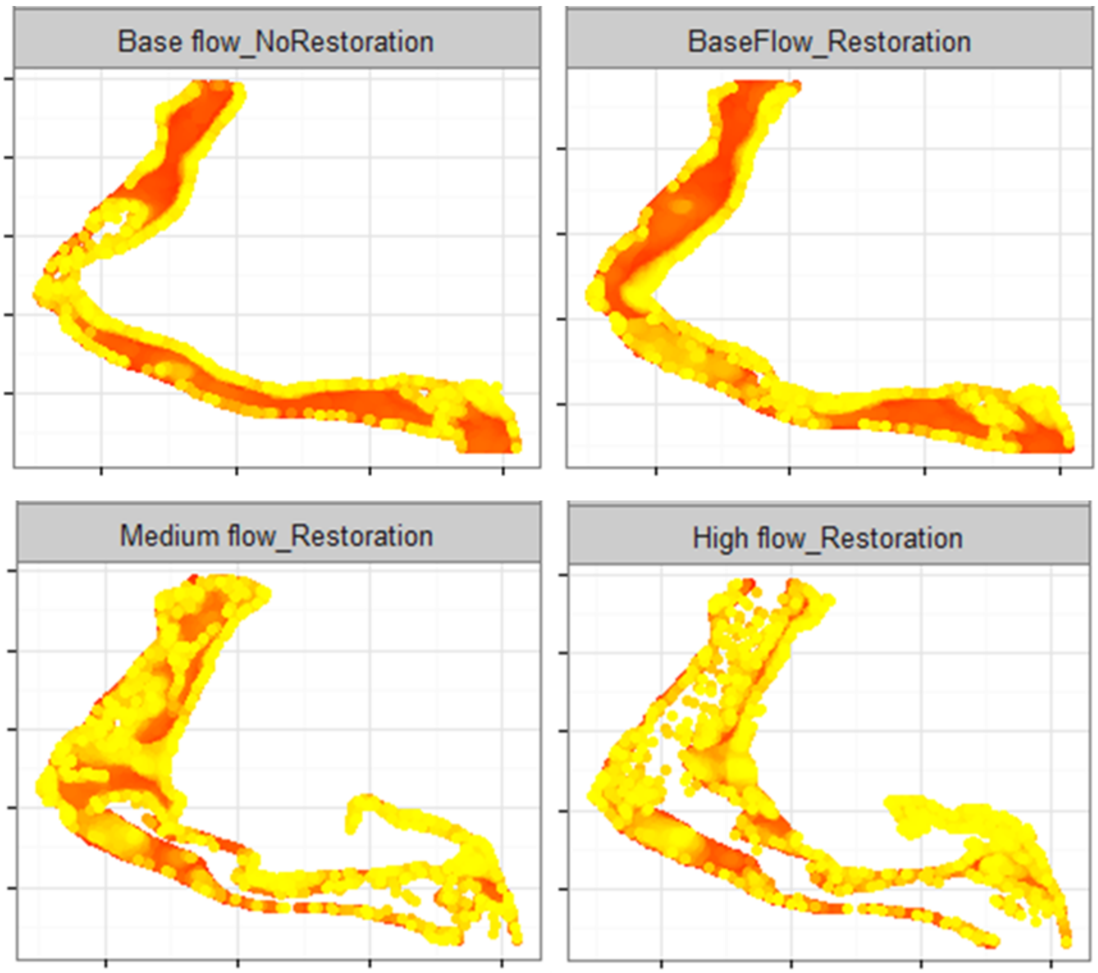

Figure 49. Example output map showing the spatial distribution of NREI estimates for spring Chinook at two CHaMP sites on the Entiat River at base flow before and after restoration actions were implemented (top panels) and at medium and high flows without restoration actions (bottom panels).

We have also used these NREI models to evaluate expected and observed benefits of stream restoration. For example, we use the model to predict the change in habitat condition and carrying capacity of a reach after the placement of post-assisted log structures, by manipulating the current DEM of a reach derived from a CHaMP survey to match the design hypotheses of the restoration (Wall et al. 2016). We also used the model to evaluate the actual changes one year after the restoration. We observe consistent patterns between expected and observed changes that suggest that we have a good understanding of both the physical and biological responses to this restoration approach (Figure 50).

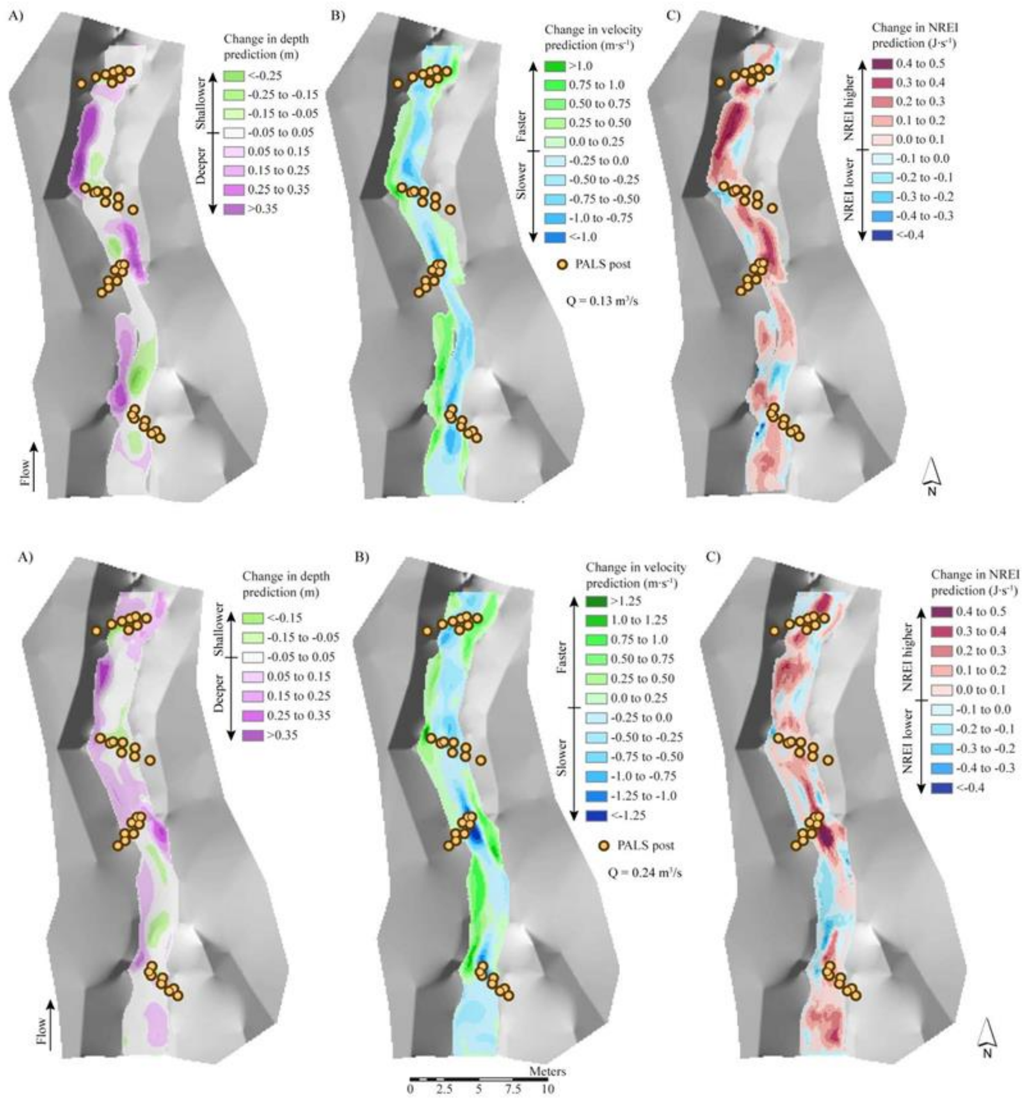

Figure 50. Spatial arrangement and magnitude of changes (post –pre restoration) to depth (A), velocity (B), and NREI (C) in expected (top panel) vs. observed (lower panel) topography.

Upscaling Methodologies

Design-based watershed estimation

Using a design-based, that is, a GRTS roll-up, approach is a statistically robust way to make capacity estimates for a spatial region such as a watershed or valley segment within a watershed (e.g., Figure 51).

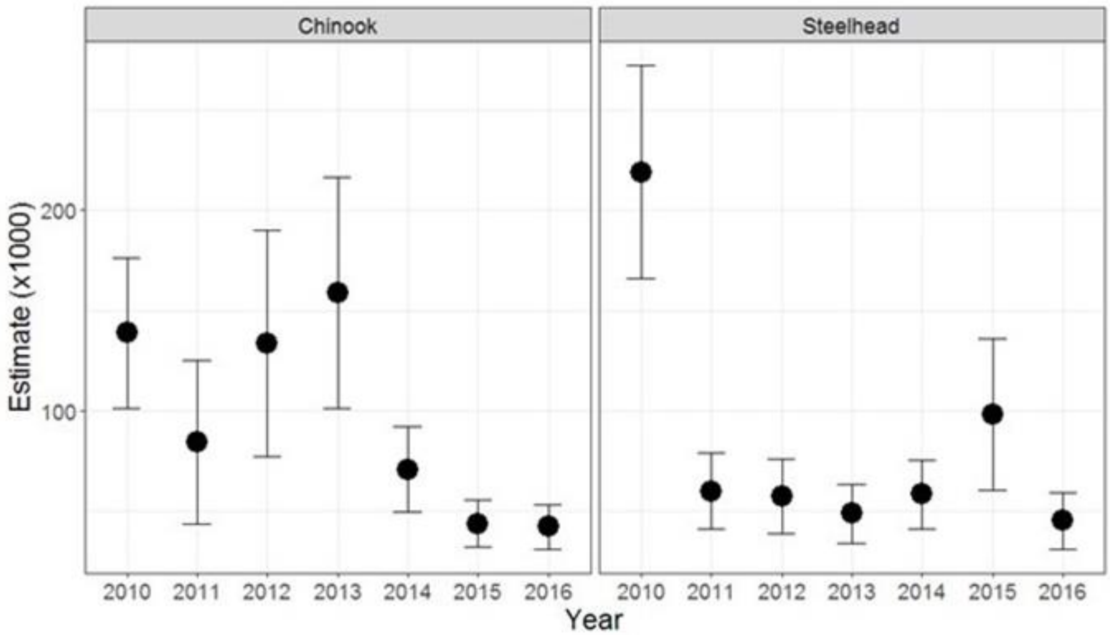

Figure 51. Watershed-scale estimates of summer Chinook and steelhead parr abundance in the Entiat River, WA using design-based upscaling methodology.

Empirical modeling based estimation and network extrapolation

We have developed regression models that use landscape characteristics to extrapolate in-stream, site-based metrics across the entire network, expanding CHaMP and ISEMP information to areas not sampled. Site-based CHaMP metrics are used to generate spatially continuous metric estimates at all points along stream networks throughout the interior Columbia River basin. The network estimates are model-based and are being made both within current CHaMP watersheds, as well for watersheds of management interest where CHaMP data are not being collected (e.g., the South Fork Clearwater, Lower Clearwater, Lolo, Lochsa, and Upper Salmon River tributaries above Redfish Lake).

For example, we can upscale point-based Chinook carrying capacity predictions from a QRF model made at all CHaMP sample sites using a model-based regression approach that uses a variety of GAAs as covariates. Predictions were made for sites spaced at ~1km throughout the interior Columbia River Basin; these sites are called the master sample. The master sample sites were clipped to only include those within the Chinook domain, as defined by StreamNet. Figure 52 shows the estimates at the Columbia River Basin scale, and summaries can also be made for individual subbasins (e.g., Figures 53 – 55).

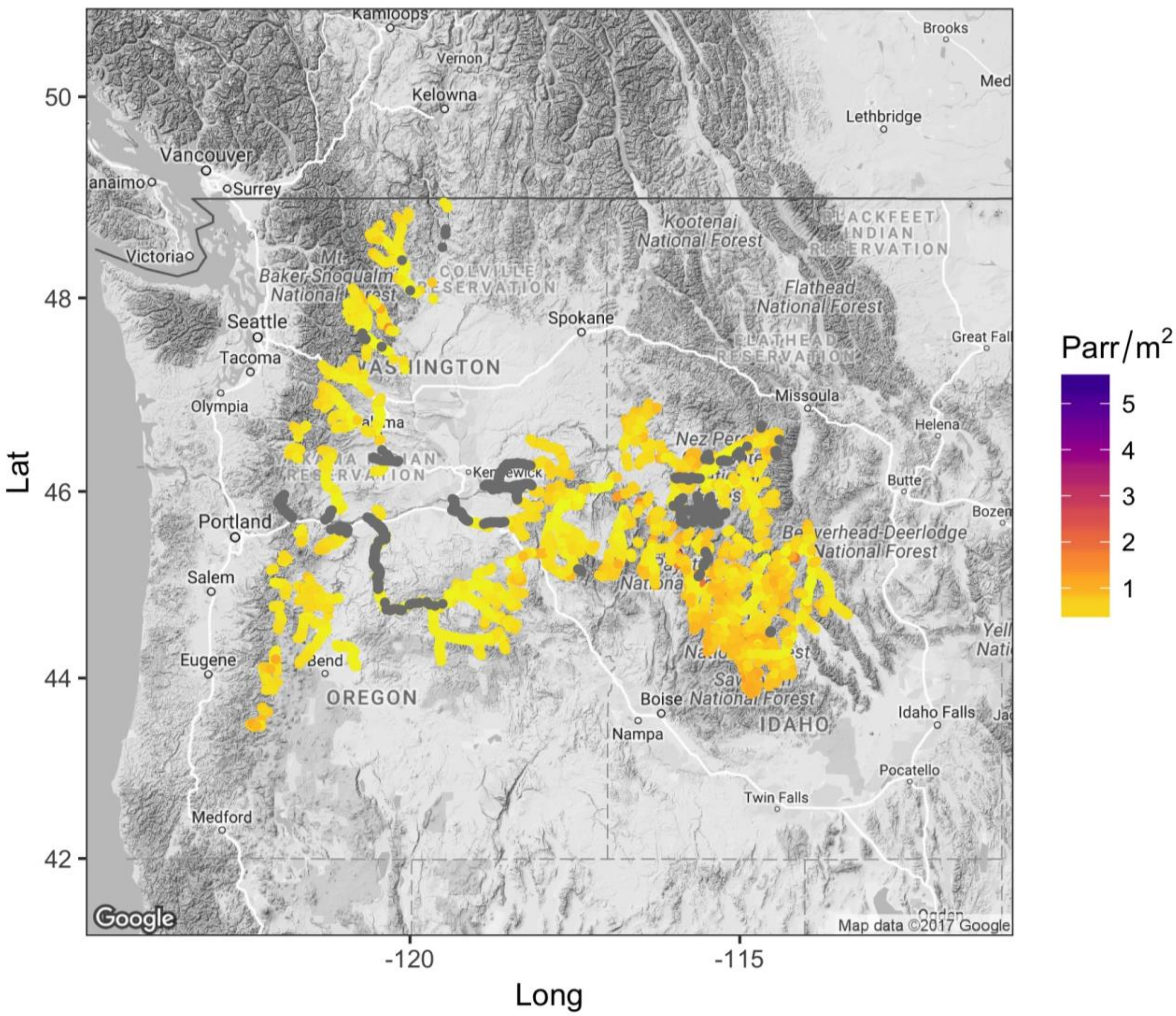

Figure 52. Chinook parr carrying capacity estimates for the interior Columbia River Basin within the Chinook domain defined by StreamNet. Gray points are areas where the estimates of capacity were beyond reasonable biological values, usually due to outlier values for one or more of the extrapolation covariates, or missing values.

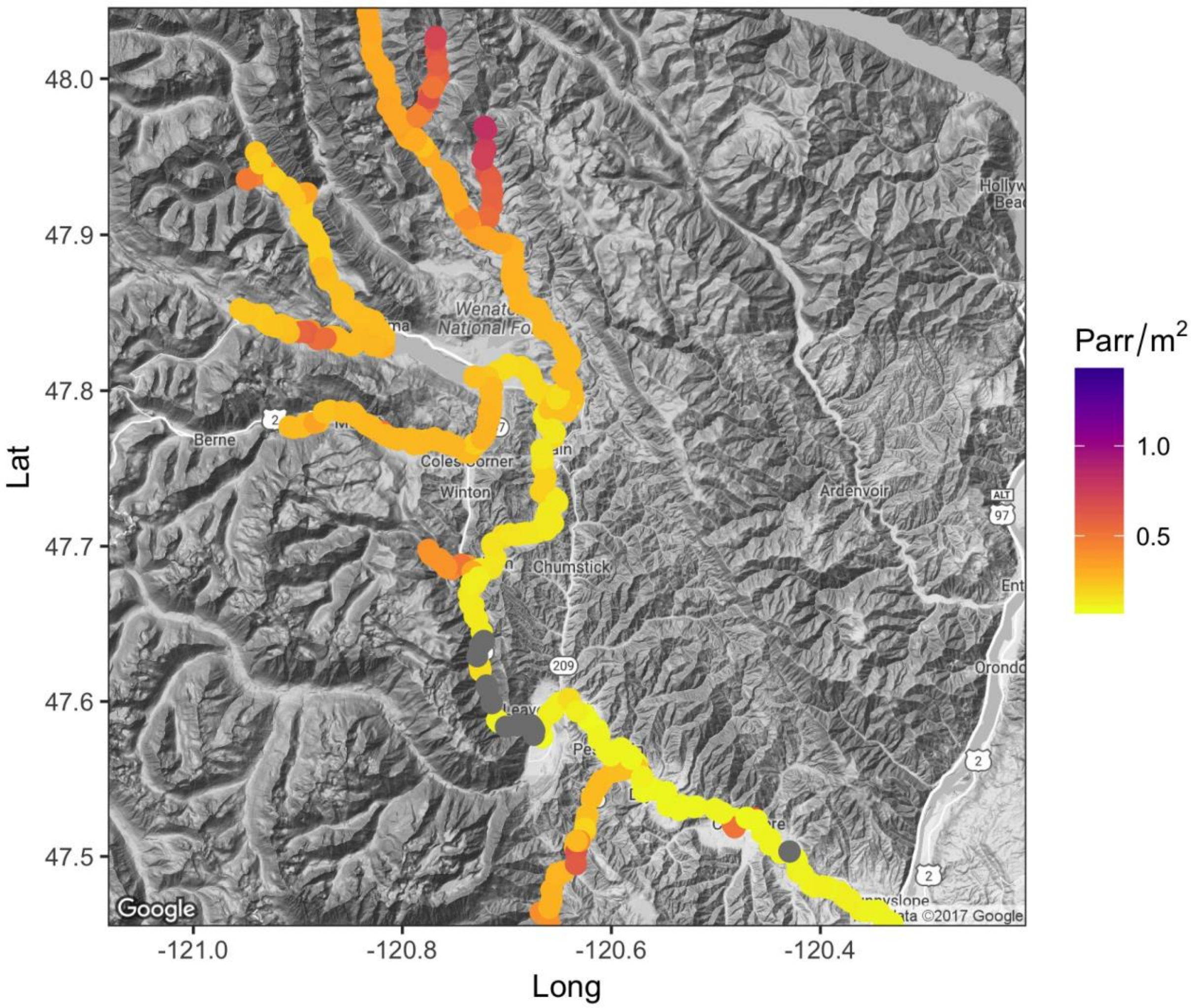

Figure 53. Chinook parr carrying capacity estimates for the Wenatchee River subbasin, Washington, within the Chinook domain defined by StreamNet. Gray points are areas where the estimates of capacity were beyond reasonable biological values, usually due to outlier values for one or more of the extrapolation covariates, or missing values.

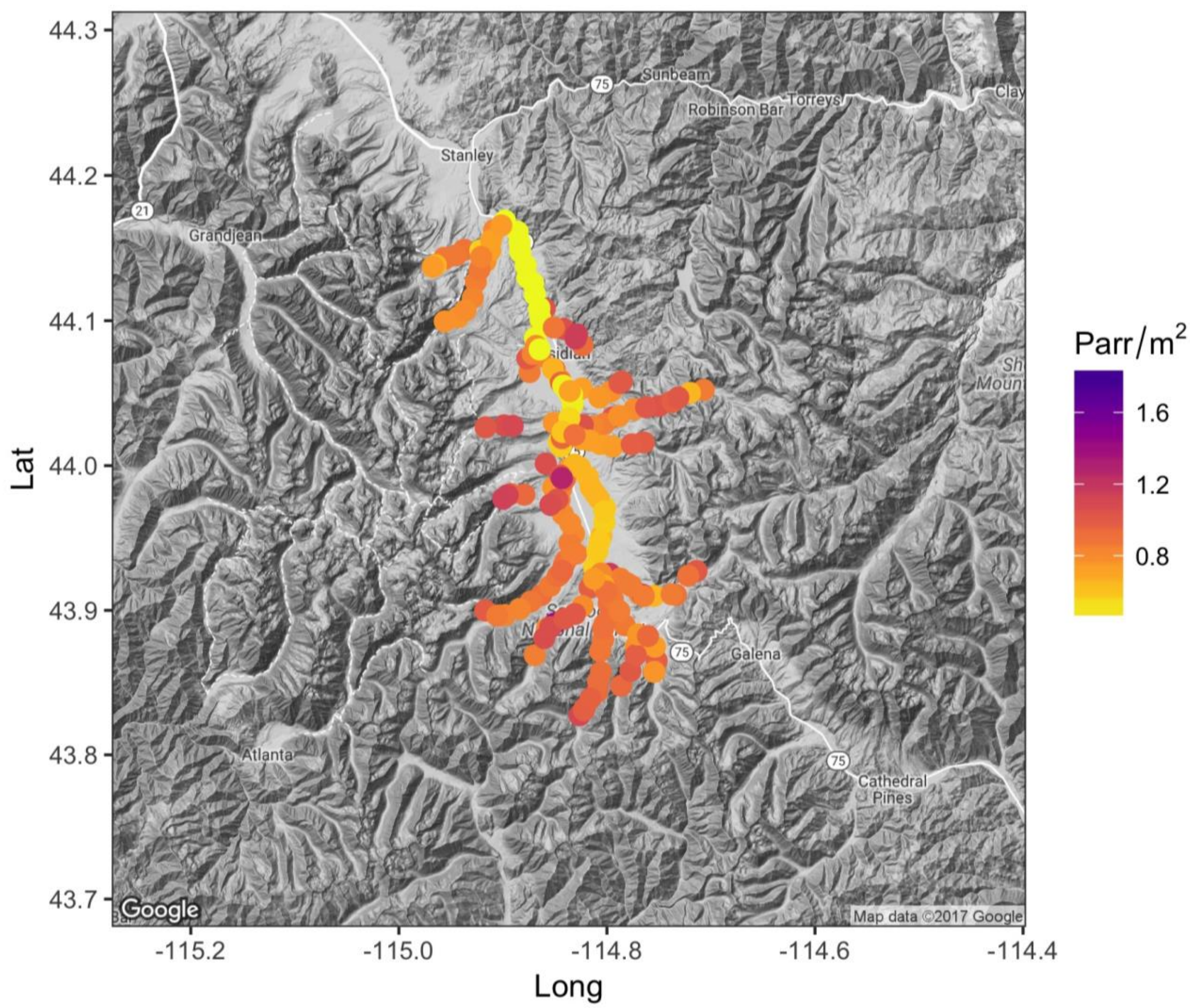

Figure 54. Chinook parr carrying capacity estimates for the Upper Salmon River subbasin, Idaho, within the Chinook domain defined by StreamNet.

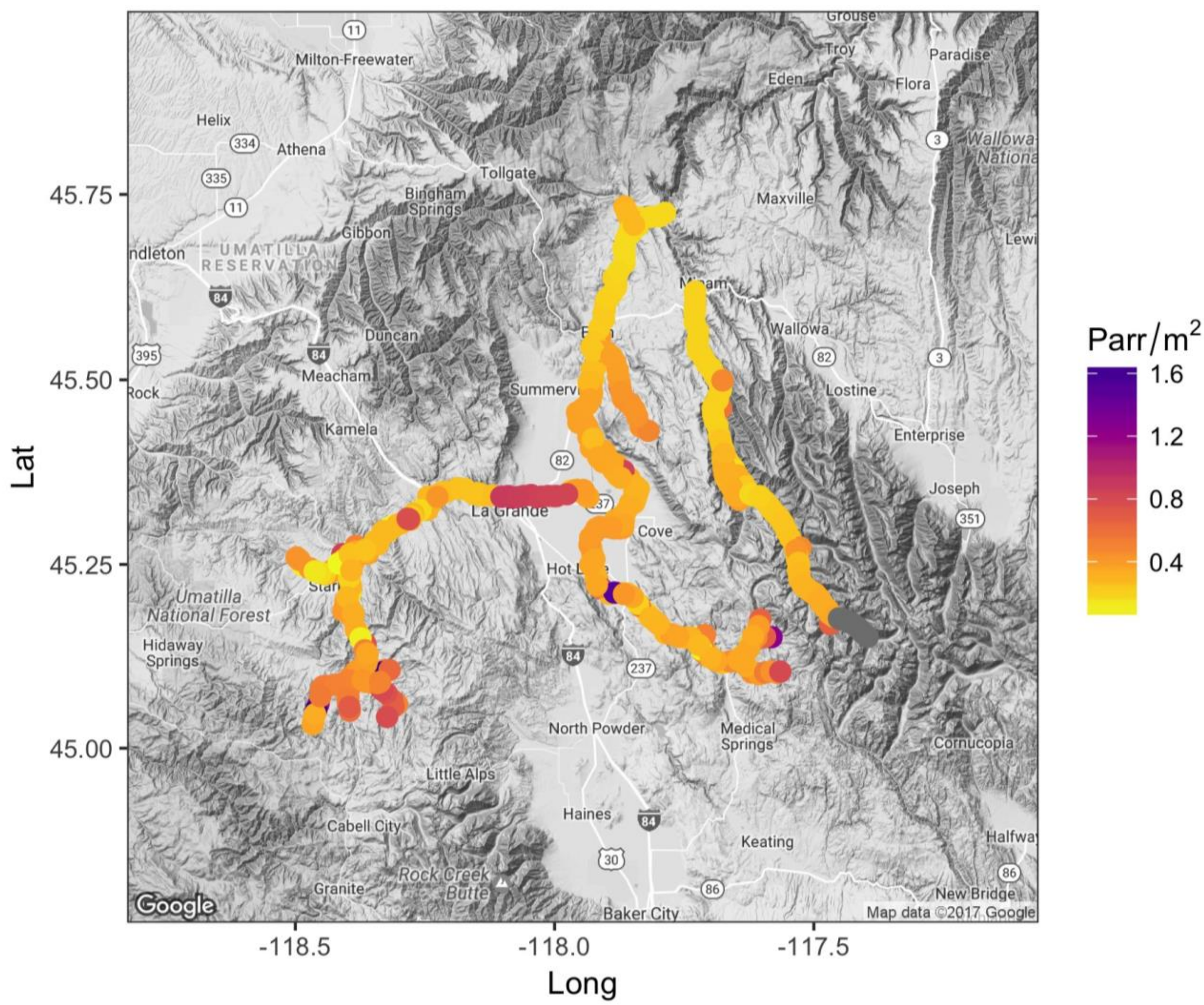

Figure 55. Chinook parr carrying capacity estimates for the Upper Grande Ronde and Minam River subbasin, Oregon, within the Chinook domain defined by StreamNet.

Processed-based habitat models

Upscaling reach-scale ecohydraulic data and models to fish populations provides a process-based methodology for informing fish habitat status, trends and context for identifying restoration opportunities. In this method, reach-based estimates of fish habitat capacity are linked to the local geomorphic setting, production, and temperature, which is then assessed for the entire watershed. In 2016, we focused on processing consistent geospatial layers to quantitatively inform reach assessments. We have produced tools that quickly produce the continuous information (including channel sinuosity, valley bottom sinuosity and Strahler order, gradient, valley bottom width, channel width, and braidedness, among others) that can describe reach type, geomorphic unit assemblage, substrate, and structural elements (e.g., wood).

Geomorphic assessment have been completed in the Middle Fork John Day, Wenatchee, Entiat and Methow River subbasins. Maps of reach types, condition (e.g., Figure 56) and recovery potential (e.g., Figure 57) are available upon request. We are completing geomorphic assessments in three other basins, with characterizations of channel types, geomorphic condition, recovery potential and example management plans (Table 15). The 2017 focus will be on the refinement of models to link reach scale capacities and channel unit configurations to geomorphic setting, which will allow upscaling of reach capacities to fish populations.

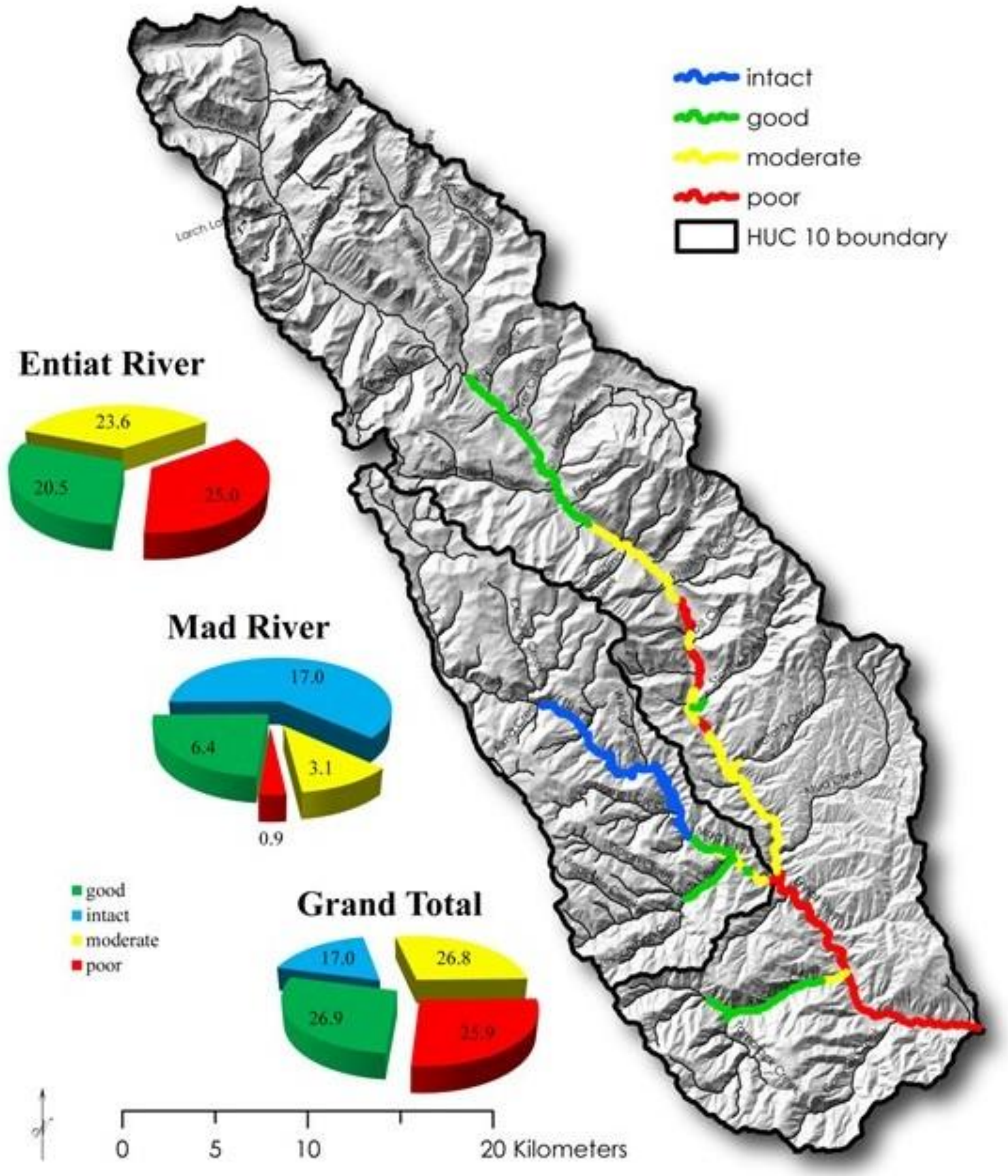

Figure 56. Geomorphic condition of stream reaches located in the portion of the stream network that is accessible to anadromous steelhead in the Entiat River watershed. The geomorphic condition results are summarized for each HUC10 as well as for the entire steelhead domain. Map projection is UTM Zone 10 N and horizontal datum is GCS North American 1983.

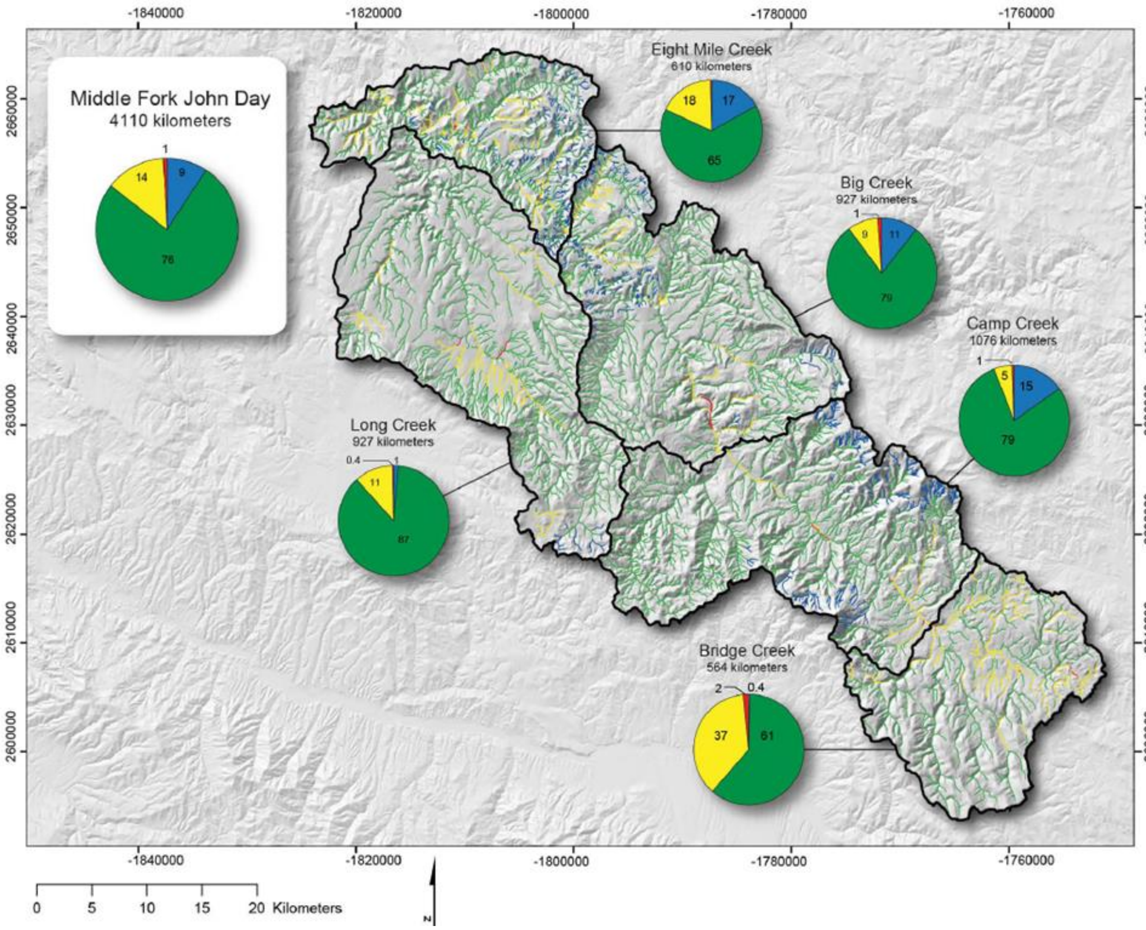

Figure 57. Recovery potential by reach for the Middle Fork John Day network.

Table 15. Status of ongoing geomorphic assessment efforts in the Columbia River Basin through ISEMP and CHaMP.

Watershed	Completed	In Progress	Report	River Classification And Geomorphic Condition	Recovery Potential	Management Plan
Middle Fork John Day	X		X	X	X	X
Lemhi		X		X		
Upper Salmon		X		X		
Yankee Fork		X		X		
Grande Ronde		X				
Wenatchee	X		X	X		
Entiat	X		X	X		
Methow	X		X	X		
Asotin	X		X	X	X	
Tucannon	X		X	X		
Pine Creek	X		X	X	X	

We have created several GIS network tools to streamline use of GIS data. The Geomorphic Network and Analysis Toolbox (GNAT) is a geospatial set of tools designed to review and update network integrity, segment streams, and model geomorphic attributes. These tools are available for use on any hydrography network, but are particularly useful for the high resolution 1:24k NHD hydrography, which has variable integrity across the Pacific Northwest, and needs some pre-processing to ensure network integrity prior to use in analyses. Other network tools are available to efficiently summarize higher order controls, such as the valley bottom extent (VBET; Gilbert et al. 2016), valley confinement, and riparian condition assessment tools (RCAT; Macfarlane et al. 2016a). Other information such as wood loading potential (WRAT; Hough-Snee et al. 2015), and the beaver restoration assessment tool (BRAT; Macfarlane et al. 2016b) aid in assessing restoration strategies. Attachment A provides high-level summaries of these tools.

Life Cycle Models

LCMs have been parameterized and are operational for the Lemhi, Middle Fork John Day (MFJD) and Entiat spring Chinook populations and are underway for the Entiat steelhead population. Models have been validated against empirical data (e.g., Figure 58) and are producing biologically reasonable estimates.

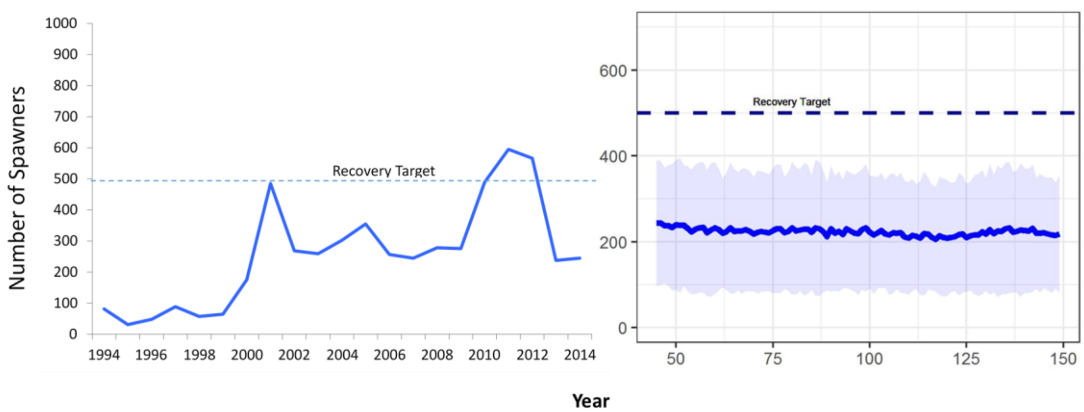

Figure 58. Time series of observed Entiat River spring Chinook salmon spawner abundance (left panel. Data from Hamstreet [2012] and Fraser and Hamstreet [2015]) and Chinook spawner abundance predicted by the LCM in the Entiat watershed under baseline conditions (right panel).

Alternative management scenarios across the watershed are being compared to baseline conditions to evaluate the impacts of reasonable restoration plans to the population status. The LCMs have been built on three components: (1) reach-scale hydraulic and ecohydraulic models that inform capacity input needs; (2) published or empirical demographic parameter estimates (i.e., stage-specific survival, fecundity, emigration/maturation probabilities); and (3) the LCM for simulating population dynamics using these data. More detailed descriptions of the LCM is available in the Adaptive Management Implementation Plan 2016 Report to the ISAB.

MFJD Steelhead LCM

For the MFJD, scenario testing for the effectiveness of adding wood to the stream channel or reducing stream temperature through increasing riparian vegetation were simulated. Simulations predicted a much larger response in capacity and productivity associated with thermal restoration that benefitted the abundance, productivity, and viability of MFJD steelhead (T1 and T2, Figures 59 and 60), whereas even relatively large additions of wood (approximately a third of the modeled domain) had minimal effects on the abundance and productivity of the MFJD steelhead population (W1, Figure 59). At best, a spawner abundance increase on the order of 7% may be feasible, but that did translate into reduced quasi-extinction risk, below the 'vulnerable' benchmark of 10%, for the model population.

Extinction risk declined under each of the restoration scenarios that either reduced high stream temperatures (T1 and T2) or increased instream habitat complexity (W1) (Figure 60), with the greatest reduction in extinction risk occurring under the maximum vegetation restoration scenario (T1). However, these inferences are based on the assumption that the status quo and restoration scenarios modeled here represent truth, but in reality they fail to consider many other potential threats (e.g., non-native species, climate change, etc.). Therefore, our estimates of extinction risks should be considered conservative (see McHugh et al. 2017, Attachment D).These results suggest that warm summer water temperatures are a primary limiting factor for steelhead in the MFJD, and that restoration of instream habitat via the addition of woody structures would likely need to occur at much higher densities that currently implemented to have meaningful impacts on steelhead populations.

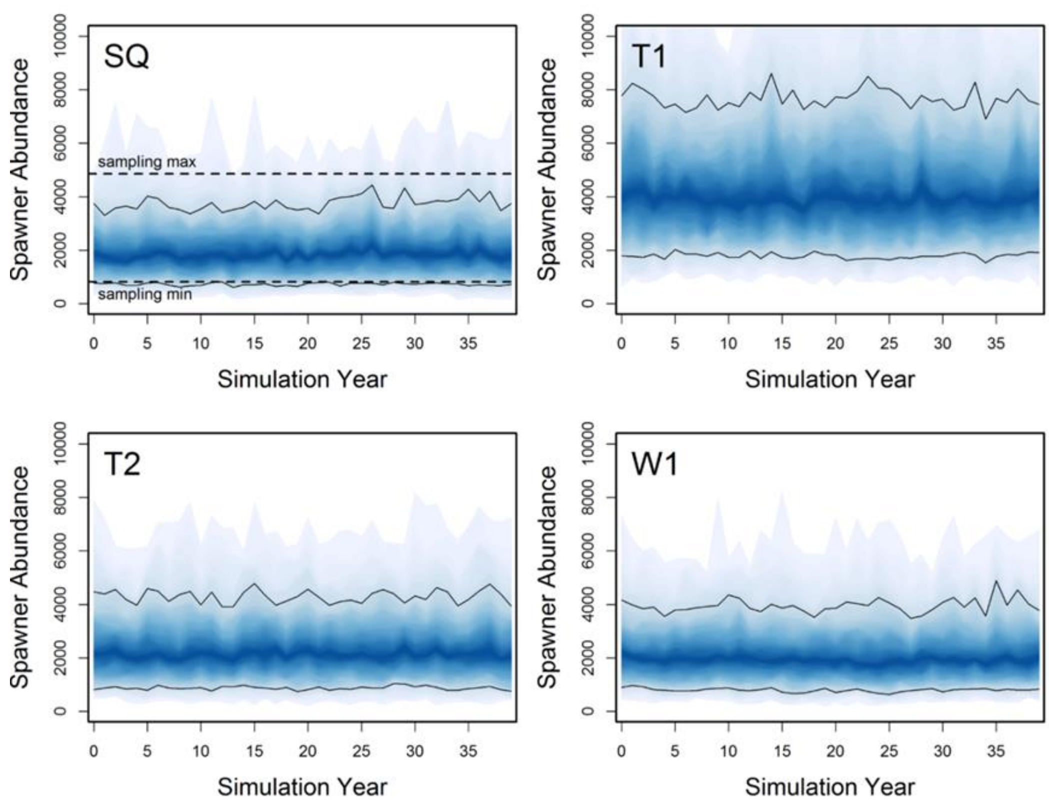

Figure 59. Time series of spawner abundance for MFJD scenarios: (SQ) Baseline current conditions scenario; (T1) Best-case thermal restoration scenario; (T2) Thermal restoration given that all currently existing riparian restoration projects reach maturity; and (W1) Large woody debris additions. Note, in panel SQ the solid horizontal line and upper/lower dashed lines correspond to recent average abundance observations and min/max, respectively.

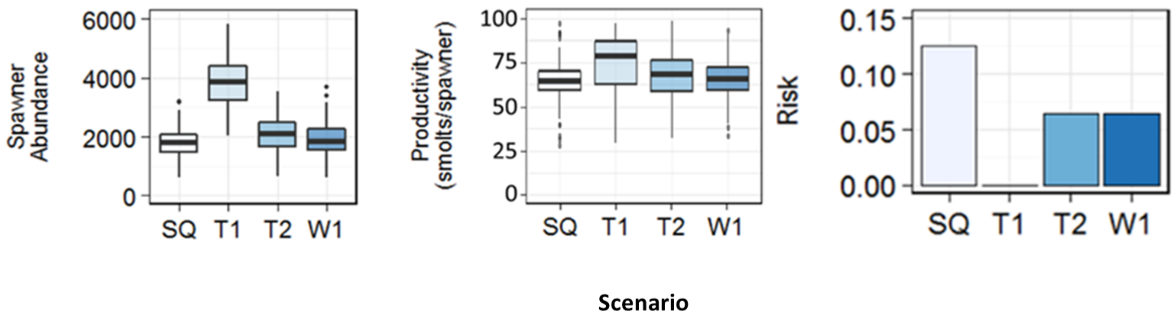

Figure 60. Population performance metrics MFJD (a) Abundance (geometric mean escapement), (b) Productivity (smolts per spawner) across scenarios, and (c) Quasi-extinction risk.

Entiat Spring Chinook LCM

The monitoring carried out under the Entiat IMW provides significant data on abundance, survival and growth that can be used in species-specific LCMs. The Entiat River Chinook LCM predicts population trends for juvenile abundance and adult returns over time as a function of habitat capacity, and the population's long-term response to habitat actions implemented to date. We developed model scenarios to evaluate the effects of a subset of actions that have been implemented, and the effect of those actions plus a 2% increase in Chinook survival on the abundance, productivity, and viability of the Entiat River Chinook population.

The limited amount of habitat improvement actions available for this simulation resulted in a relatively modest increase in juvenile rearing capacity (estimated with NREI) within treated reaches, averaging 7% overall across the sites and ranging from 0% to a 35% increase. Extrapolating these results to the watershed scale resulted in an increase in the watershed carrying capacity of less than 1%, which translated to a small increase in the number of Chinook spawners predicted to return to the Entiat River (middle panel, Figure 61). Improved carrying capacity plus a 2% increase in survival resulted in a greater increase in the number of spawners (right panel, Figure 61), although neither scenario resulted in spawner numbers meeting or exceeding the recovery target (UCSRB 2007).

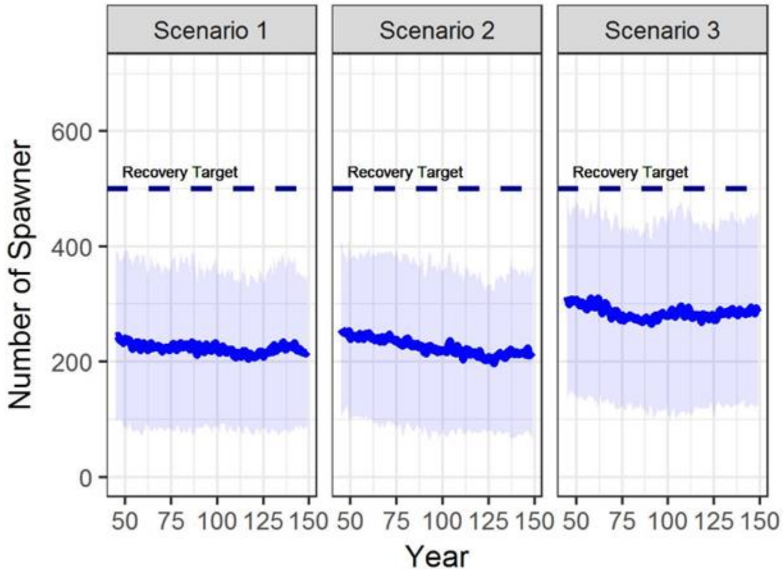

Figure 61. The number of Chinook spawners predicted by the Entiat Chinook LCM for (1) baseline conditions; (2) effect of increased juvenile rearing capacity only; and (3) effect of increased juvenile rearing capacity plus increased juvenile survival probability by 2%.

We also calculated the VSP (abundance and productivity) score as measure of risk (probability of extinction) using a probability of quasi-extinction threshold P(QET) following McElhany et al. (2000) for each scenario (Figure 62). A score of 0 indicates a population is either extinct or at a very high risk of extinction, and 4 indicates a population has very low risk of extinction in 100 years. Only scenario 3, habitat restoration plus an increase in survival, reduced the population’s risk of extinction.

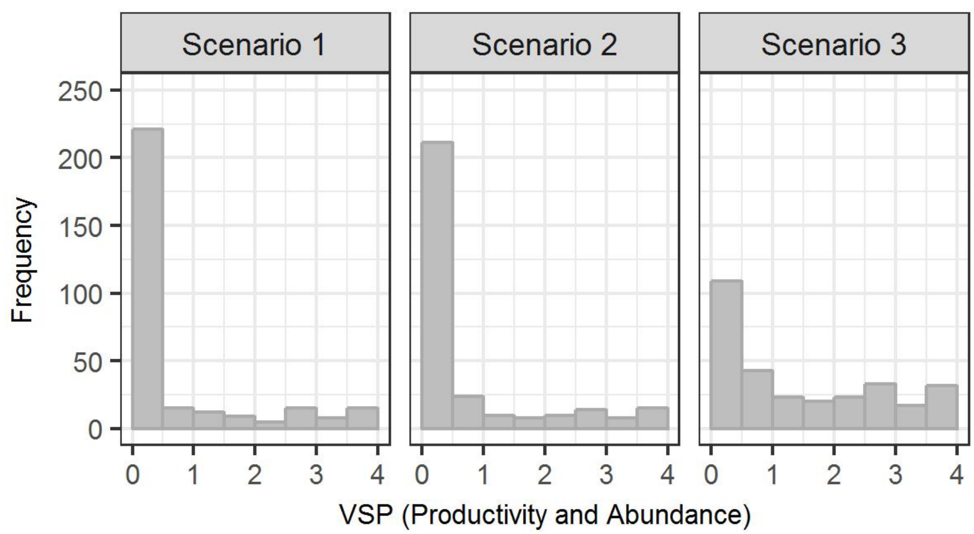

Figure 62. VSP scores for productivity and abundance for three scenarios in the Entiat IMW : (1) baseline conditions; (2) effect of increased juvenile rearing capacity only; and (3) effect of increased juvenile rearing capacity plus increased juvenile survival probability by 2%.

Lemhi Spring/summer Chinook LCM

We have developed a simple model of the freshwater portion of the life cycle for spring/summer Chinook salmon and parameterized the model using data from the Lemhi River basin. This is a minimal, empirical model, including only life stages whose abundance or survival can be directly observed, that is, spawners, parr (juveniles rearing in their natal basin during the first summer of life), and smolts (operationally defined as juvenile emigrants passing Lower Granite Dam [LGR]). We have simulated the effects of past or future tributary reconnection projects using habitat-based information to constrain key stage-specific parameters.

Between 2009 and 2012, several tributaries previously inaccessible to Chinook, due mainly to seasonal dewatering in the lower reaches, were reconnected to the main channel. Juvenile Chinook have not yet been observed using these tributaries, so any increase in rearing capacity is not reflected in the data used to fit the model. To assess the potential effect of these restoration actions on overall freshwater productivity, we replaced the empirical posterior distribution of total parr capacity by a lognormal distribution with the same CV but a log-mean based on QRF predictions that included the reconnected tributaries. This assumes that juveniles will eventually occupy all accessible areas and that intrinsic productivity does not change. Results showed stage-specific intrinsic productivity and capacity estimates that appear biologically reasonable (Figure 63).

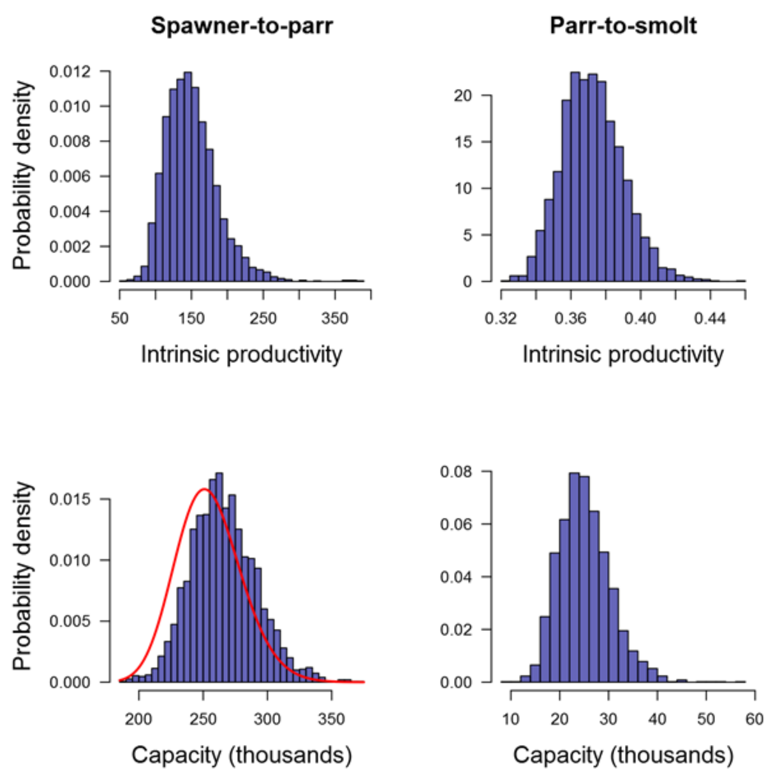

Figure 63. Posterior distributions of intrinsic productivity and capacity in the spawner-to-parr and parr-to-smolt Beverton-Holt transition functions for Lemhi Chinook. Priors were uniform over the range of the posterior except in the case of parr capacity (lower left), where the informative prior based on QRF predictions is shown in red.

Comparing observed and fitted values of parr abundance demonstrates the importance of observation uncertainty (Figure 64). The model attributes three exceptionally high observed values to measurement noise based on the associated standard errors, resulting in a more conservative estimate of the slope of the spawner-to-parr relationship at low spawner abundance (i.e., intrinsic productivity). The prior on parr capacity is informative since the model infers that none of the observed escapements have come close to saturating the system with parr (Figure 65). In contrast to the spawner-to-parr relationship, there is not much evidence of density dependence in the parr-to-smolt transition based on the raw data. After shrinkage of the measurement errors, the estimated intrinsic productivity (i.e., maximum parr-to-smolt survival) is around 0.37.

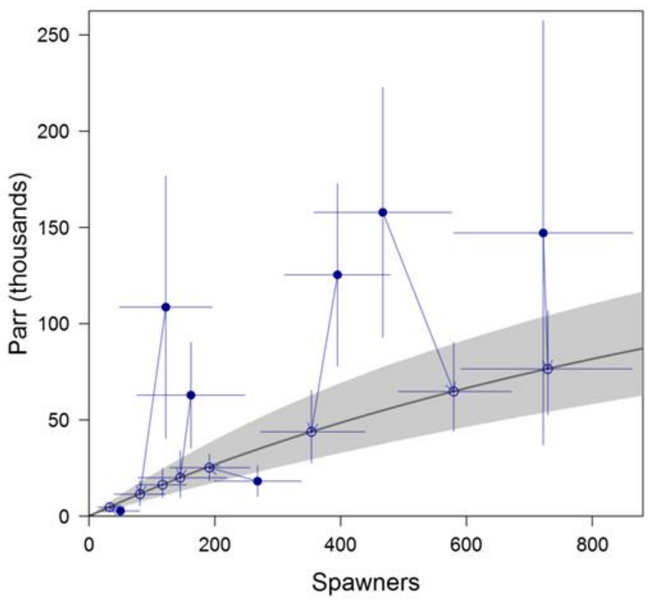

Figure 64. Estimated spawner-to-parr Beverton-Holt function for Lemhi Chinook (black line: posterior mean, gray envelope: 95% credible interval). The observed data (solid points, with error bars indicating observation SEs) are connected by arrows to the corresponding fitted values (open circles, with error bars indicating 95% credible intervals).

The model predicts a fairly modest increase in population-scale smolt production due to tributary reconnection, both in absolute terms (an average 10% increase in smolts per spawner) and relative to uncertainty (Figure 66). It is possible that this simple scenario analysis underestimates the true improvement; for example, if the reconnected tributaries have higher intrinsic productivity than the previously accessible subwatersheds, then the increase would be apparent at lower spawner abundance. Even in this case, however, the asymptotic difference between baseline and reconnection scenarios (i.e., at high spawner abundance) would remain the same.

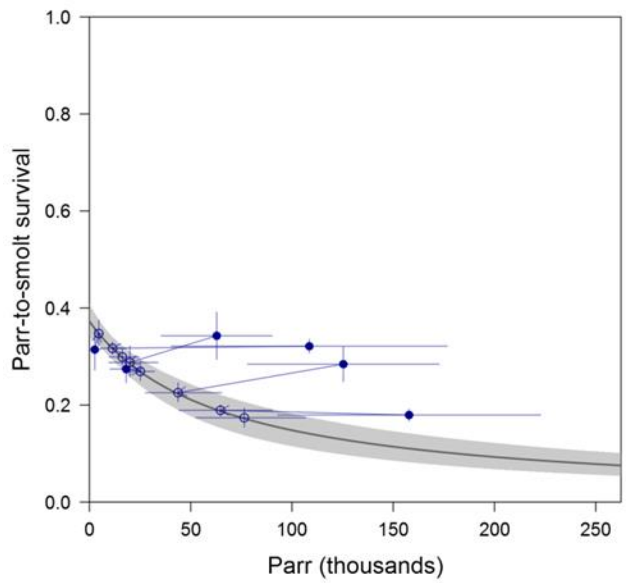

Figure 65. Estimated parr-to-smolt Beverton-Holt function for Lemhi Chinook expressed as a relationship between abundance and survival (black line: posterior mean, gray envelope: 95% credible interval). The observed data (solid points, with error bars indicating observation SEs) are connected by arrows to the corresponding fitted values (open circles, with error bars indicating 95% credible intervals).

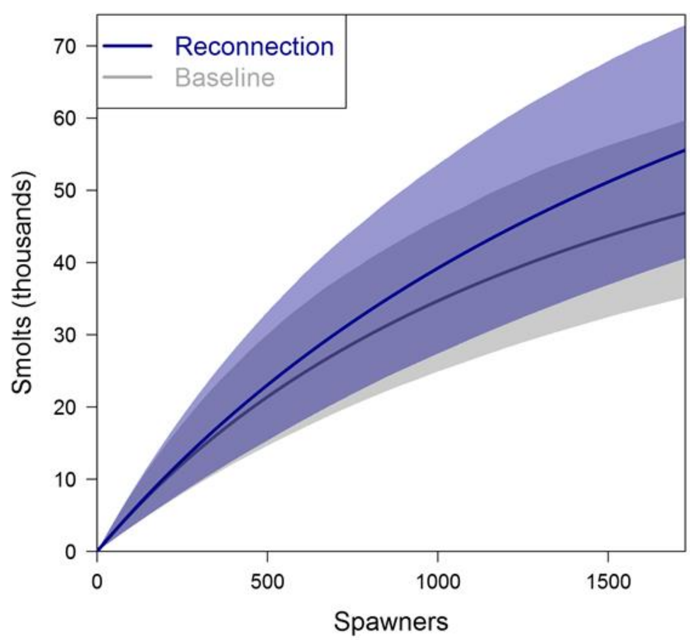

Figure 66. Composite Beverton-Holt curves for Lemhi Chinook spawner-to-smolt production, under baseline conditions and after tributary reconnection. Lines show posterior means and shading indicates 95% credible intervals.

Our analysis shows that it is possible to recover biologically plausible estimates of stage-specific transition functions (e.g., Beverton-Holt parameters) from sparse, noisy data, but in some cases auxiliary information (in this case, parr capacity predicted by QRF) is needed to constrain the estimates. This simple two-stage model of freshwater juvenile production suggests there is moderate density dependence in the spawner-to-parr transition and weak density dependence in parr-to-smolt survival. The former incorporates any habitat constraints on egg deposition, as well as habitat effects on fry and summer parr rearing, while the latter includes the effect of overwintering habitat.

This analysis also illustrates how habitat-derived metrics such as QRF capacity predictions, when used as prior information in a Bayesian statistical framework, provide a mechanism to simulate habitat restoration actions. In this case, we simulated an increase in total parr rearing habitat due to tributary reconnection by increasing the prior median on parr capacity in accordance with QRF estimates. The estimated population-level freshwater productivity (smolts per spawner) increased roughly 10% on average, but this effect was largely obscured by parameter uncertainty. This result emphasizes the importance of a formal accounting of uncertainty in model outputs used to provide management advice (Harwood and Stokes 2003). Similarly, Roni et al. (2010) showed that given typical levels of habitat restoration, the signal (population response of juvenile salmonids) is often undetectable given the noise.

Discussion

To adequately plan for tributary habitat action implementation, the region needs periodic watershed (population-scale) assessment of stream habitat condition (HQQ) at a spatial grain consistent with rehabilitation actions and at a content grain consistent with species by life-stage, season, and population-process constraints. These assessments can then be used to develop quantifiable estimates of long-term biological benefits of watershed rehabilitation action plans. These estimates of benefit form testable, measurable hypotheses that can drive an adaptively managed implementation and evaluation process linking rehabilitation action planning, watershed monitoring, effectiveness evaluations and subsequent action plan development.

Periodic assessments on the range of every 3 to 5 years will allow the appropriate temporal interval for change to accrue since tributary stream HQQ does not change rapidly, either naturally or due to human actions. Longer intervals would allow “easier” change detection, but would lack the ability to track fine-scale temporal events and would be too long between check-ins when implemented in an adaptive management/decision support system context since there is a need to catch necessary course corrections before the program has gone too far off track. Shorter intervals between assessments are likely not necessary or cost-effective given the rate of change of stream habitat features.

Implementing watershed restoration actions in an adaptive management context would ensure adequate accountability and needed course corrections. Large-scale natural resource decision making involves large extents (space and time), large investments of resources (money and managed resources such as land, water or target species), public trust and considerable risk. Risk-averse strategies may seem most appropriate given the potential to waste public resources or trust, but risk-averse strategies are not constructed from risk-averse actions. Rather, risk-averse actions involve cautious planning, cautious actions and redundant, countervailing actions to hedge outcomes. None of these tactics are part of a truly risk-averse strategy since cautious design principles will result in outcomes or impacts that are not large, and thus while the risk of any particular action not having an adverse outcome is minimized, the risk to the natural resource will have been increased as time and money are wasted on actions with little or no potential benefit (Rist et al. 2013). To properly manage the risk to the resource, a system that allows large-effect, "risky" actions must be employed, but in a manner that the risk of actions "failing" is managed through a controlled set of steps based on prediction, monitoring, assessment, evaluation and reaction (Bouwes et al. 2016b). Adaptive management implementation schemes provide not only the necessary accountability, but more importantly, the absorption of risk.

Predicted outcomes with quantifiable performance metrics (interim and final) are the basis of adaptive management; they essentially form the hypotheses that are tested through action, monitoring and evaluation, and necessitate the use of models for the implementation of an adaptive management framework. In the case of a watershed-scale habitat restoration implementation scheme, the models would predict the current versus potential stream habitat condition, the magnitude of change in habitat due to a specific suite of actions, and finally, the population-level effect or benefit. The population-level effect may not be detectable on the evaluation time-frame (3 – 5 years); nonetheless, the estimated benefit would be projected forward in time to allow distinguishing between multiple potential action scenarios. Thus, the key model (or model outcome or product) that would drive a watershed-scale tributary habitat rehabilitation adaptive management schema would be a spatially explicit, dynamic (on the 3 – 5 year period) HQQ evaluation (by species where appropriate) in the currency of stream habitat features that both determine fish population processes and are altered by standard approaches to stream rehabilitation.

Given this, life cycle models are obviously a key component to the evaluation of tributary habitat rehabilitation program design and implementation and the method that should be used to evaluate the population-level effect of a watershed restoration action plan. Indeed, projecting a population forward for 50 years or more and evaluating the change in extinction risk or persistence is the standard assessment method in conservation biology worldwide (Morris and Doak 2002). The specific model form or performance metrics are less important than the consistent application of a population projection tool that integrates the change in physical habitat across all life-stages impacted and that allows the population-level benefit, if any, to accrue. Stage-specific by population-process specific descriptions of biological limiting factors are not sufficient for this task, even if they are spatially explicit, such as a reach-scale estimation of capacity for a single life-stage. All populations change in a bounded, regulated manner and we assume that natural populations at risk are "over-regulated", that is, an unnatural limitation on population growth has been imposed by human activities and that relaxing this limitation will "fix" the population problem. However, the idea that a single rate-limiting factor exists or can be identified is a dangerously naïve framework on which to base a large-scale natural resource management process; indeed, the silver bullet of single-factor population restrictions are rare in at-risk species. Rather, the habitat impairments that adversely impact population processes are best thought of as a series of rate limitations, each only being apparent when the preceding one is relaxed. For example, relaxing a juvenile habitat capacity limitation by opening access to large numbers of blocked stream segments may only result in a small change in population size due to an adult spawning capacity limitation or a juvenile survival limitation that was not previously a regulating factor. Thus, having a dynamic, multifaceted, process-based population projection framework allows more robust and realistic assessment of the effects of rehabilitation actions by integrating across life-stages (juvenile and adult), population processes (capacity, survival and movement) and space (headwaters, mainstem and ocean).

Obviously there is considerable uncertainty in projecting the impacts of tributary rehabilitation actions on at-risk populations of salmonids. The uncertainty arises from the long-term, large-scale nature of the situation, where multiple interacting biological and physical processes determine the fate of an individual or life-stage. While we currently possess considerable understanding of these processes and how they act as determinants of population behavior, each in isolation is still known with error, and when combined across an entire life-cycle, the uncertainty can be compounding. However, the uncertainty is bounded and manageable given the extensive life-stage abundance and survival data on these populations, and the understanding of fish-habitat relationships developed from field and laboratory experiments. The uncertainty is bounded since population projections must be consistent with many independent metrics, such as stage-specific abundance and age or size structure that are used for calibration and validation. The uncertainty is manageable due to an explicit accounting of sources of uncertainty. Natural variation in physical and biological processes is quantifiable and thus can be incorporated into analytical models of population processes and rehabilitation action outcomes. In a decision-support context, uncertainty is managed through the application of sensitivity analysis to understand the role uncertainty plays in each component of the quantitative tools, and thus how it affects the ability to differentiate between alternative actions.

Summary

In 2016 ISEMP and CHaMP continued to make progress on a number of key issues, including new results from effectiveness monitoring in the Entiat, Bridge Creek and Lemhi IMWs. In additions we have made advancements in upscaling reach-level habitat and fish abundance estimates to the network scale, and rolling out models useful to biologists, managers and policy makers for the adaptive management of interior Columbia River Basin listed salmonids and their tributary environment.

ISEMP and CHaMP have continued to produce publically available fish and habitat status and trends data of known accuracy and precision, and to leverage that data in powerful habitat and life cycle models that have been shown to produce output reflecting reality. In addition, the use of these tools by co-managers across the Columbia River Basin is spreading: Oregon Department of Fish and Wildlife, Columbia River Inter Tribal Fish Commission, and the Asotin IMW are all using the ISEMP LCM to help guide restoration planning; the Asotin IMW and Geomorphic Assessment Restoration Plan use all of the ISEMP/CHaMP network tools (e.g., VBET, RCAT etc.) as well as the NREI habitat model; the Shoshone-Bannock tribe has incorporated the 3-D Delft hydraulic model, FIS, CHaMP topographic and auxiliary metrics and the network models into its fisheries program; and the QRF model is being used by the Bureau of Reclamation in the Upper Salmon to prioritize habitat restoration actions, to name a few.

Results from the IMWs continue to add to our knowledge about the effectiveness of tributary habitat restoration actions on target fish populations, especially with respect to the necessity for long-term monitoring. In a comprehensive review of IMWs across the Columbia River basin Bennett et al. (2016) concluded that while IMWs face implementation challenges they are still the most reliable way of assessing the effectiveness of watershed restoration and should be conducted for at least 10 years. Recent data from the Bridge Creek IMW underlines this conclusion: initial findings from Bridge Creek showed that the installation of beaver dam analogs resulted in a significant increase in the number of beaver dams which resulted in increased juvenile steelhead density, survival, and production; however, we are now seeing a decline to pre-restoration densities, possibly as a result of drought conditions and the natural evolution of habitat over time.

Large variability in environmental factors that can control fish population dynamics is expected but unpredictable. It is important to understand if actions still provide benefit under extreme conditions over longer time periods, especially since those conditions are likely to become more common in interior Columbia River tributaries as a result of climate change. Simulations by Mantua et al. (2010) predicted that rising water temperatures will thermally stress salmon throughout Washington's watersheds, and that by the 2080s there will be a complete loss of snowmelt dominant basins, and that the few transient runoff watersheds left will experience more rainfall dominated behavior, more severe summer low flow periods, and more intense winter flooding, likely resulting in reduced reproductive success. Similarly, Crozier et al. (2008) reported that climate change will cause clear hydrologic changes across western North America, including milder winters, more rain, less snow, more severe drought in the summer, and more intense precipitation and flooding in the winter. IMWs are designed as long-term experiments to not only observe changes in population dynamics of a relatively long-lived species (e.g., >4 years), but also to increase the probability that the responses to restoration are observed not only under average environmental conditions but across the range of possible conditions, including drought and high water years. It is important to maintain monitoring over an appropriate time scale so that we can assess the longer-term benefits of habitat restoration actions in increasing fish population resiliency to extreme conditions. For example, it may be that the restoration actions in Bridge Creek mitigated against more drastic impacts on the steelhead population of a low flow, high water temperature year since beaver dams help lower water temperatures, allowing the steelhead population to survive an otherwise catastrophic event.

ISEMP and CHaMP have produced robust methods for upscaling site- or reach-level habitat and fish data to the watershed scale that are operational and available for use by biologists and managers alike. We have created a set of tools that are ready to be integrated into the fish biologist community that in combination limit bias and improve precision in planning for habitat restoration actions. The different approaches (i.e., design-based, empirical modeling, and processed-based) can be combined and tailored to meet each watershed's specific needs. For example, the design-based approach produces a statistically robust average response but information about sites of high fish production, or hot spots, is lost, while the processed-based approach retains information about hot spots. A combination of both approaches provides a rigorous estimate of watershed-scale responses while also retaining important spatial information.

As of 2016 ISEMP and CHaMP have successfully implemented a LCM for steelhead in the Middle Fork of the John Day, spring Chinook in the Entiat, and spring/summer Chinook in the Lemhi. The model is currently being reviewed by the Independent Scientific Advisory Board as part of the AMIP process. AMIP is recommending that LCMs be an integral part of an adaptive management strategy to design and assess alternative suites of habitat restoration actions and make testable, quantitative predictions (Zabel et al. 2017), a function that ISEMP/CHaMP LCM are already being used for by a number of collaborators and within the ISEMP subbasins.

References

Bare, C.M., K.B.S. DeHart, J L. Latshaw, J.J. Rogers, I.A. Tattam, J.R. Ruzycki, and R.W. Carmichael. 2015. Escapement and productivity monitoring of summer steelhead and spring Chinook salmon in the John Day River. 2014 Annual Report BPA Contract # 1998-016-00:89 pp.

Bennett, S., G. Pess, N. Bouwes, P. Roni, R.E. Bilby, S. Gallagher, J. Ruzycki, T. Buehrens, K. Krueger, W. Ehinger, J. Anderson, C. Jordan, B. Bowersox, and C. Greene. 2016. Progress and Challenges of Testing the Effectiveness of Stream Restoration in the Pacific Northwest Using Intensively Monitored Watersheds. Fisheries 41:92-103.

Beverton, R.J.H., and S.J. Holt. 1957. On the dynamics of exploited fish populations. Fisheries Investigation Series 2, volume 19, UK Ministry of Agriculture Fisheries, and Food, London, UK.

Bouwes, N., N. Weber, C.E. Jordan, M.M. Pollock, W.C. Saunders, I.A. Tattam, C. Volk, and J.M. Wheaton. 2016. Ecosystem experiment reveals benefits of natural and simulated beaver dams to a threatened population of steelhead (*Oncorhynchus mykiss*). Scientific. Reports. 6, 28581; doi: 10.1038/srep28581.

Bouwes, N., S. Bennett, and J. Wheaton. 2016b. Adapting Adaptive Management for Testing the Effectiveness of Stream Restoration: An Intensively Monitored Watershed Example. Fisheries 41:84-91.

Crozier, L.G., A.P. Hendry, P.W. Lawson, T.P. Quinn, N.J. Mantua, J. Battin, R.G. Shaw, R.B. Huey. 2008. Potential responses to climate change in organisms with complex life histories: evolution and plasticity in Pacific salmon. Evolutionary Applications 1(2):252-270.

Fraser, G.S. and C.O. Hamstreet. 2015. Chinook salmon spawning ground surveys on the Entiat River, 2014. U. S. Fish and Wildlife Service, Leavenworth, Washington.

Gilbert, J.T., W.W. Macfarlane, and J.M. Wheaton. 2016. The Valley Bottom Extraction Tool (V-BET): A GIS tool for delineating valley bottoms across entire drainage networks. Computers & Geosciences 97:1-14.

Hamstreet, C.O. 2012. Spring and summer Chinook salmon spawning ground surveys on the Entiat River, 2011. U. S. Fish and Wildlife Service, Leavenworth Washington.

Harwood, J., and K. Stokes. 2003. Coping with uncertainty in ecological advice: lessons from fisheries. Trends in Ecology & Evolution 18:617-622.

Hough-Snee, N., A. Kasprak, R.K. Rossi, N. Bouwes, B.B. Roper, and J.M. Wheaton. 2015. Hydrogeomorphic and Biotic Drivers of Instream Wood Differ across Sub-Basins of the Columbia River Basin, USA. River Research and Applications 32:1302-1315. 10.1002/rra.2968. https://isemp.egnyte.com/dl/d1TKjb4i5d.

ISEMP/CHaMP. 2016. Integrated Status and Effectiveness Monitoring Program (ISEMP) and Columbia Habitat Monitoring Program (CHaMP) Annual Combined Technical Report, January 2015 – December 2015, BPA Projects 2003-017-00 and 2011-006-00, 39 Electronic Pages.

ISEMP/CHaMP. 2015. 2014 Combined Annual Technical Report for the Integrated Status and Effectiveness Monitoring Program and Columbia Habitat Monitoring Program. P144403, Bonneville Power Administration. 10.13140/RG.2.1.1695.5920

ISEMP. 2014. The Integrated Status and Effectiveness Monitoring Program: Annual Report for Fiscal Year 2013. P139896, Bonneville Power Administration, Portland, OR. 10.13140/RG.2.1.4480.8724.

ISEMP. 2013. ISEMP Synthesis Report 2012: Results from Effectiveness Monitoring, Modeling, and Status and Trend Monitoring. P134453, Bonneville Power Administration. 10.13140/RG.2.1.2875.2409.

ISEMP. 2012. ISEMP Lessons Learned Synthesis Report 2003-2011. P127284, Bonneville Power Administration. 10.13140/RG.2.1.4382.5681.

ISEMP. 2010. Integrated Status and Effectiveness Monitoring Program: 2009 Annual Report. Terraqua. 10.13140/RG.2.1.2723.6881.

ISEMP. 2009. Annual Report of the Integrated Status and Effectiveness Monitoring Program: Fiscal Year 2008. P112576, Terraqua. 10.13140/RG.2.1.3772.2646.

ISEMP. 2008. Annual Report of the Integrated Status and Effectiveness Monitoring Program: Fiscal Year 2007. 10.13140/RG.2.1.2936.6803

ISEMP. 2007. A Review of the Integrated Status and Effectiveness Monitoring Program: 2003-2006. 00027480-1, Bonneville Power Administration. 10.13140/RG.2.1.4558.6964.

Loughin, T.M. 2006. Improved experimental design and analysis for long-term experiments. Crop Science 46(6):2492-2502.

Loughin, T.M., M.P. Roediger, G.A. Milliken, and J.P. Schmidt. 2007. On the analysis of long-term experiments. Journal of the Royal Statistical Society: Series A (Statistics in Society) 170(1):29–42.

Macfarlane, W.W., J.T. Gilbert, M.L. Jensen, J.D. Gilbert, N. Hough-Snee, P.A. McHugh, J.M. Wheaton, and S.N. Bennett. 2016a. Riparian vegetation as an indicator of riparian condition: detecting departures from historic condition across the North American West. Journal of Environmental Management. Volume & Issue TBD.

Macfarlane, W.W., J.M. Wheaton, N. Bouwes, M.L. Jensen, J.T. Gilbert, N. Hough-Snee, and J.A. Shivik. 2016b. Modeling the capacity of riverscapes to support beaver dams. Geomorphology. Volume & Issue TBD.

Mantua, N., I. Tohver, and A. Hamlet. 2010. Climate change impacts on streamflow extremes and summertime stream temperature and their possible consequences for freshwater salmon habitat in Washington State. Climate Change 102:187. Doi:10.1007/s10584-010-9845-2

Maret, T.R., J.E. Hortness, and D.S. Ott. 2005. Instream flow characterization of upper Salmon River basin streams, central Idaho, 2004. 2328-0328.

McElhany, P., M. Ruckelshaus, M. Ford, T. Wainwright, and E. Bjorkstedt. 2000. Viable salmonid populations and the recovery of evolutionarily significant units. US Dept. Commerce NOAA Tech. Memo. NMFS-NWFSC 42:156.

McHugh, P.A., W.C. Saunders, N. Bouwes, C.E. Wall, S. Bangen, J.M. Wheaton, M. Nahorniak, J.R. Ruzycki, I.A. Tattam, and C.E. Jordan. 2017. Linking models across scales to assess the viability and restoration potential of a threatened population of steelhead (*Oncorhynchus mykiss*) in the Middle Fork John Day River, Oregon, USA. Ecological Modelling 355:24-38.

McNyset, K., C. Volk, and C. Jordan. 2015. Developing an Effective Model for Predicting Spatially and Temporally Continuous Stream Temperatures from Remotely Sensed Land Surface Temperatures. Water 7:6827-6846. 10.3390/w7126660. https://isemp.egnyte.com/dl/p6U7sqJ7G8.

Morris, W.F. and D.F. Doak. 2002. Quantitative Conservation Biology. Sinauer Associates Inc, Sunderland MA, 480pp

Nahorniak, M., D.P. Larsen, C. Volk, and C.E. Jordan. 2015. Using Inverse Probability Bootstrap Sampling to Eliminate Sample Induced Bias in Model Based Analysis of Unequal Probability Samples. PloS one 10:e0131765.

Naiman, R., C. Johnston, and J. Kelley. 1988. Alteration of North American Streams by Beaver. BioScience, 38(11), 753-762. doi:10.2307/1310784

O'Brien, G.R., J. Wheaton, K. Fryirs, P. McHugh, N. Bouwes, G. Brierley, and C. Jordan. 2017. A geomorphic assessment to inform strategic stream restoration planning in the Middle Fork John Day Watershed, Oregon, USA. Journal of Maps 13:369-381.

Pollock, M.M., T.J. Beechie, and C.E. Jordan. 2007. Geomorphic changes upstream of beaver dams in Bridge Creek, an incised steam channel in the interior Columbia River basin, eastern Oregon. Earth Surface Processes and Landforms 32:1174-1185.

Quantitative Consultants, Inc.(QCI). 2006. Salmon ISEMP watershed model development: Adding stochasticity to the Life History Model Structure. Available at: http://www.nwfsc.noaa.gov/research/divisions/cbd/mathbio/isemp/docs_salmon.cfm

Rist, L., A. Felton, L. Samuelsson, C. Sandström, and O. Rosvall. 2013. A new paradigm for adaptive management. Ecology and Society 18(4): 63. http://dx.doi.org/10.5751/ES-06183-180463

Roni, P., G. Pess, T. Beechie, and S. Morley. 2010. Estimating changes in coho salmon and steelhead abundance from watershed restoration: how much restoration is needed to measurably increase smolt production? North American Journal of Fisheries Management 30:1469-1484.

Steinhorst, K., Y. Wu, B. Dennis and P. Kline. 2004. Confidence intervals for fish out migration estimates using stratified trap efficiency methods. Journal of Agricultural, Biological, and Environmental Statistics 9: 284-299.

Stevens, D.L., Jr. and A.R. Olsen. 1999. Spatially restricted surveys over time for aquatic resources. Journal of Agricultural, Biological, and Environmental Statistics. 4:415-28.

Truscott, B.L., J.M. Cram, A.R. Murdoch and K. See. 2017. Upper Columbia Spring Chinook and Steelhead Juvenile and Adult Abundance, Productivity and Spatial Structure Monitoring. Project # 2010-034-00, Contract #72749, Bonneville Power Administration, Portland, Oregon.

Upper Columbia Spring Chinook Salmon and Steelhead Recovery Plan. 2007 (accessed at http://www.westcoast.fisheries.noaa.gov/publications/recovery_planning/salmon_steelhead/domains/interior_columbia/upper_columbia/uc_plan.pdf

Urquhart, N.S. and T.M. Kincaid. 1999. Designs for detecting trend from repeated surveys of ecological resources. Journal of Agricultural, Biological and Environmental Statistics 4: 404-414.

Wall, C.E., N. Bouwes, J.M. Wheaton, W.C. Saunders, and S.N. Bennett. 2015. Net rate of energy intake predicts reach-level steelhead (*Oncorhynchus mykiss*) densities in diverse basins from a large monitoring program. Canadian Journal of Fisheries and Aquatic Sciences 73:1081-1091.

Wall, C. E., N. Bouwes, J.M. Wheaton, S.N. Bennett, W.C. Saunders, P.A. McHugh, and C.E. Jordan. 2016. Design and monitoring of woody structures and their benefits to juvenile steelhead trout (*Oncorhynchus mykiss*) using a net rate of energy intake model. Canadian Journal of Fisheries and Aquatic Sciences 74:727-738.

Walters, C.J., J.S. Collie, and T. Webb. 1988. Experimental designs for estimating transient responses to management disturbances. Canadian Journal of Fisheries and Aquatic Sciences 45:530-538.

Weber, N., N. Bouwes, M.M. Pollock, C. Volk, J.M. Wheaton, G. Wathen, J. Wirtz, and C.E. Jordan. 2017. Alteration of stream temperature by natural and artificial beaver dams. Plos One 12:e0176313

Wheaton, J. M., K.A. Fryirs, G. Brierley, S.G. Bangen, N. Bouwes, and G. O'Brien. 2015. Geomorphic Mapping and Taxonomy of Fluvial Landforms. Geomorphology 248:273-295. 10.1016/j.geomorph.2015.07.010. https://isemp.egnyte.com/dl/oS9HyjwkZz.

Wheaton, J.M., N. Bouwes, P.A. McHugh, C. Saunders, S. Bangen, P. Bailey, M. Nahorniak, C.E. Wall, C. Jordan. 2017. Upscaling Site-Scale Ecohydraulic Models to Inform Salmonid Population-Level Life Cycle Modelling and Restoration Actions – Lessons from the Columbia River Basin. Earth Surface Processes and Landforms.

Yuen, H. and R. Sharma. 2005. Using simulation techniques to estimate management parameters on Snake River steelhead: Declines in productivity make rebuilding difficult. North American Journal of Fish Management 25:446-463.

Zabel, R., and multiple authors. 2017. Life cycle models of salmonid populations in the interior Columbia River Basin. Report submitted to the ISAB for review.