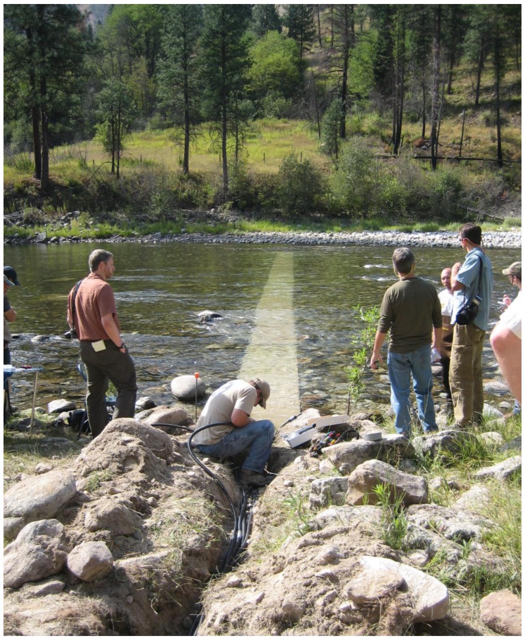

CALENDAR YEAR 2014

Combined Annual Technical Report

for the

Integrated Status and Effectiveness Monitoring Program

and

Columbia Habitat Monitoring Program

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Bonneville Power Administration's

Combined
Integrated Status and Effectiveness Monitoring Program
and
Columbia Habitat Monitoring Program
Calendar Year 2014 Annual Report

BPA Project Nos. ISEMP 2003-017-00 and CHaMP 2011-006-00

June 22, 2015

Prepared by:

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and

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EXECUTIVE SUMMARY

This combined report for Bonneville Power Administration (BPA) Integrated Status and Effectiveness Monitoring Program (ISEMP; BPA Project 2003-017) and the Columbia Habitat Monitoring Program (CHaMP; BPA Project 2011-006) covers Calendar Year (CY) 2014. Here we give an update on ISEMP's progress and lessons learned as we work toward the end of the 2008 Biological Opinion (BiOp) on the Federal Columbia River Power System (FCRPS; NMFS 2008) in 2018, the results and lessons learned from 4 years of implementing CHaMP, a summary of the 2015 work plan for both programs, and a collation of our responses over the years to Independent Science Advisory Board (ISAB), Independent Science Review Panel (ISRP), and Northwest Power and Conservation Council (Council) reviews and questions.

A shared goal of ISEMP and CHaMP is to develop and export standardized monitoring and analytical approaches to help answer many of the questions posed in the 2008 BiOp related to the recovery of spring Chinook (*Oncorhynchus tshawytscha*) and steelhead (*O. mykiss*), including three management questions identified by BPA (see box). To date, ISEMP has launched two successful, large-scale monitoring programs - CHaMP, and the Lower Granite Dam (LGR) run reconstruction program. In 2014 we continued to make advancements in many areas, for example, defining and presenting fish-habitat relationships, extrapolating site-based data to larger spatial scales, and developing and exporting tools to improve fisheries managers' ability to estimate various parameters for the populations under their care.

In 2014 CHaMP improved standardization of its salmonid habitat monitoring protocol and advanced development of powerful and innovative approaches to analyzing CHaMP metrics and topographic survey data. These include Geomorphic Change Detection (GCD) software, the River Bathymetry Toolkit (RBT), extrapolation frameworks, and integration with other programs (i.e., U.S. Forest Service PACFISH/INFISH Biological Opinion (PIBO), and BPA's Action Effectiveness Monitoring (AEM) program).

Each product is an integral piece of ISEMP and CHaMP's shared mission and collectively they represent substantial progress toward the development of standardized approaches to inform and evaluate the status and trends of fish and their habitat within the context of freshwater habitat limiting factors and restoration effectiveness.

Management Use of ISEMP and CHaMP Products

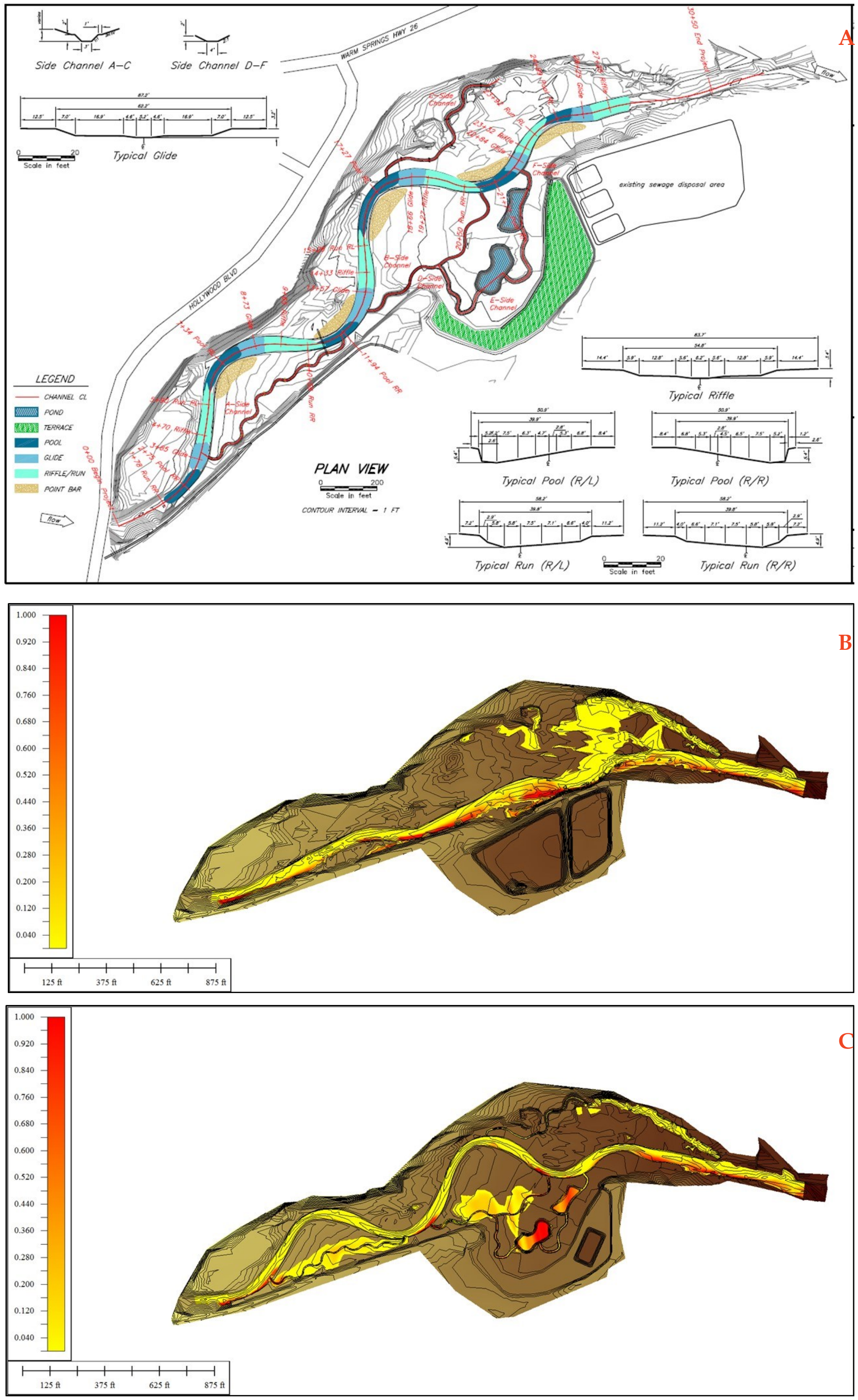
Many of the tools and products being developed by ISEMP and CHaMP are either already being used by managers or are beginning to gain traction, which is encouraging as new ideas and approaches often take time to gain acceptance. For example, the California Department of Fish and Wildlife Coastal Watershed Planning and Assessment Program, assisted through the Pacific States Marine Fisheries Commission, has adopted the CHaMP protocol for restoration monitoring in the Big Navarro-Garcia watersheds of coastal California; a PIT tag-based run decomposition/adult escapement approach developed in the Lemhi is now being used in the Upper Columbia by Washington Department of Fish and Wildlife, and the Upper Columbia Salmon Recovery Board is using ISEMP's Habitat Model to guide restoration planning in the Upper Columbia.

Some tools have been incorporated into how BPA itself does business. For example, BPA engineer Sean Welch is using ISEMP's Habitat Model in the Grande Ronde to provide a quantitative methodology to assess the potential habitat improvement for specific restoration actions, and the Upper Columbia Habitat Program is working with a cooperating partner to provide impartial evaluation of proposed projects using the Habitat Model.

Mr. Welch has found that the model is directly applicable within project-scale analysis conducted on stream restoration projects. In particular, evaluating the existing stream reach in the project area for the presence/absence of suitable habitat (e.g., for spawning, rearing, or holding) allows a baseline condition to be established and provides the foundation for alternatives assessment during the restoration design process. The ability to efficiently assess "habitat gain" using a quantitative methodology provides a very clear "moment" in the project design cycle where the potential benefit to fish can be assessed. The model provides equal or greater weighting than the design-scale geomorphic and engineering analyses, or at a minimum, helps ensure hydrologic, hydraulic and habitat outcomes are better integrated through alternatives development. Keeping the model's program on an open, non-ESRI based architecture allows for a much broader distribution and user group without being tied to a third party software vendor, a "good government" practice.

"The Habitat Model is a direct and tangible benefit to BPA, its cooperating partners and most importantly, the resource we are all working towards improving".

Sean Welch, PE, BPA



This is an example of a project where BPA engineer Sean Welch used the Habitat Model to guide restoration actions to improve Chinook juvenile rearing in Shitike Creek in the Warm Springs Reservation, Or. Panel A shows the stream reach targeted for restoration actions, and Panel B shows existing suitable habitat for juvenile rearing, where yellow is poor rearing habitat and red is good habitat. Panel C shows the extent and location of rearing habitat predicted by the Habitat Model after restoration actions have been implemented. Applying the Habitat Model to the data provides a good visual of the suitability difference between the two conditions. Mr. Welch is currently preparing an assessment of existing habitat conditions using the Habitat Model for the 3 mile Birdtrack Springs reach, and plans to use this report format on future projects.

KMQ1: What are the tributary habitat limiting factors preventing the achievement of desired objectives?

KMQ2: What are the relationships between tributary habitat actions and fish survival or productivity improvements, and what actions are potentially most cost effective?

KMQ3: Are tributary actions achieving the expected level of habitat improvement, and associated biological response?

Lessons Learned in 2014

Fish and Habitat Responses to Restoration Actions

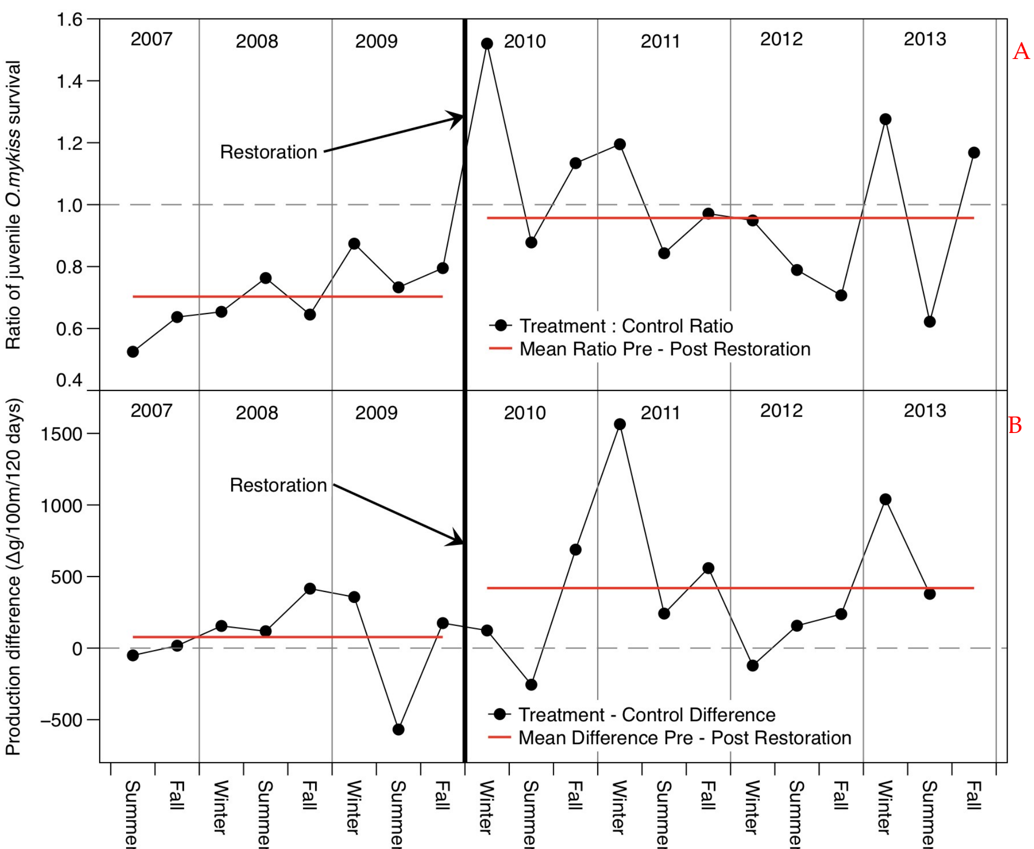
Detecting changes in freshwater productivity in response to habitat restoration over the timeframe of the BiOp requires intense sampling and Intensively Monitored Watersheds (IMWs) are an effective approach to achieve this. In CY2014 ISEMP's IMWs: Bridge Creek IMW in the John Day subbasin in Oregon, the Entiat River IMW in the Upper Columbia subbasin in Washington, and the Lemhi IMW in the Salmon River subbasin in Idaho, provided valuable information for restoration approaches that can be used within other impaired watersheds.

Bridge Creek IMW

The first stage of restoration in Bridge Creek, a deeply incised stream, was in 2009 with the installation of over 100 structures to encourage beavers to build dams and thereby reduce stream incision. Since then, beavers have built and maintained dams on approximately half of those structures, and the number of natural beaver dams (not built on the structures) has also increased by 300 percent within the study area. Monitoring results indicate that the dams have significantly reduced incision of the stream channel and increased the number and size of pool habitat. Owing partly to the high sediment loads in Bridge Creek, the geomorphic response has been rapid, with some degree of floodplain reconnection taking place in all of the treatment reaches. We recorded a reduction in maximum daily water temperatures by 1 to 2 degrees Celsius over control sites, and an increase of 0.17 meters per year in the water table elevation. This increase in water table elevation is expected to promote expansion of riparian vegetation and floodplain resources that provide important functions to salmon and steelhead, including shading and surface water temperature regulation. Most importantly, these documented changes in habitat have resulted in documented improvements in fish survival, abundance, and productivity.

Entiat IMW

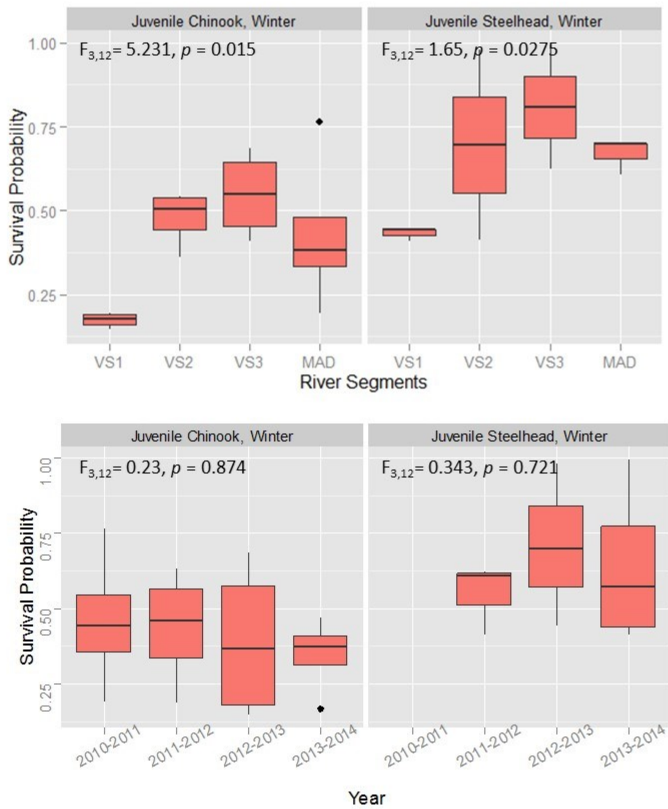
Instream complexity is limited in the Entiat River and an engineered approach is being taken to stream restoration, including adding rocks and wood to the river and reconnecting the floodplain by breaching levees where possible. Two of four rounds of habitat actions have been implemented so far and monitoring results to date show encouraging habitat and fish responses. Pool frequency and depth, and the amount of large wood in the river was significantly greater post-restoration in the area of the river treated, but no significant increase was detected in the amount of habitat complexity. We were not able to detect a change in juvenile Chinook or steelhead abundance in treated reaches., but we were encouraged to find that estimates of over-winter survival probabilities for juvenile steelhead and Chinook showed a significant increase in survival post-restoration in the area of the river



Restoration actions designed to improve habitat by encouraging beaver to build dams in Bridge Creek resulted in increased seasonal probability of survival for juvenile O. mykiss (Panel A). The difference between O. mykiss survival in Bridge and Murderer's Creek (control) increased post-restoration (black line), as did the average difference between Bridge and Murderer's before and after treatment (red line).

Seasonal production of juvenile O. mykiss also increased post-restoration on Bridge Creek (treatment) compared to Murderer's Creek (control) (Panel B). We found a seasonal difference in production (black line) and average difference pre- and post-restoration (red line) between treatment and control watersheds.

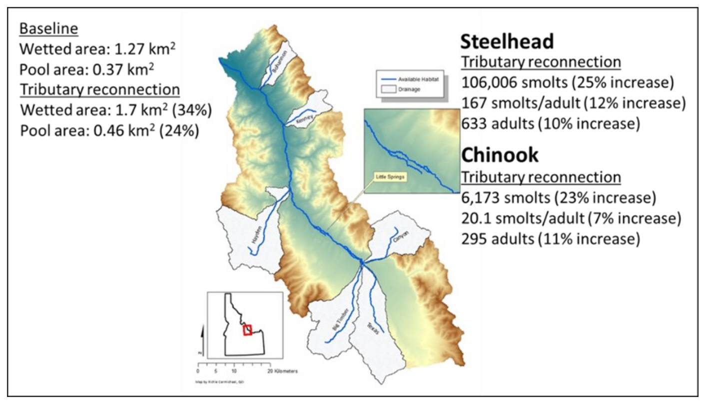
receiving restoration actions in 2012, although this did not translate into a significant increase in the population annual survival rate.



We found a higher probability of over-winter survival for juvenile Chinook and steelhead in the upper section of the mainstem Entiat River that received restoration actions designed to increase in-stream habitat complexity and side channel habitat were implemented in 2012 (VS3, top panel), but this did not result in increases in the probability of over-winter survival at the population scale (bottom panel).

Lemhi IMW

In the Lemhi IMW restoration actions are mostly focused on reconnecting dewatered tributaries. Our results from the last 6 years of monitoring have shown important changes in the Lemhi populations due to restoration actions: the Little Springs Restoration Project increased juvenile Chinook survival from an estimated 29 percent to 80 percent and steelhead from 110 individuals to 1,297 fish over 3 years. Using data from our intensive monitoring effort, an ISEMP-developed life cycle model predicted that while the 4 percent targeted improvement in freshwater productivity for steelhead would be met under current restoration plans, the 7 percent targeted improvement in freshwater productivity for spring/summer Chinook would not be achieved under current restoration scenarios. We have developed a number of potential restoration scenarios and identified actions that would meet or exceed survival targets for spring/summer Chinook salmon.



Model predictions indicated that Chinook targets will not be met under current restoration scenarios so we plugged different restoration scenarios into the life cycle model and estimated the response in terms of smolt abundance, freshwater productivity and adult escapement. We were able to find combinations of actions that would result in the desired target for Chinook.

Timelines

Bridge Creek IMW

Stage 1 was implemented in 2009 and Stage II restoration actions will install a series of treatments throughout 4 additional reaches on the lower 30 km of Bridge Creek in 2015. Pre-project data have been collected since 2006, and post-project monitoring should continue through at least 2018.

Entiat IMW

Restoration actions have been implemented in 2012 and 2014, with the next round scheduled for treatment in 2016 and 2017. The final round of actions will be implemented in 2020. ISEMP began monitoring in 2010 and monitoring should continue through 2023 at least.

Lemhi IMW

The first re-connection occurred in 2005 to restore connectivity at least seasonally to Bohannon Creek and the next reconnection is in the planning stage. ISEMP began intensive monitoring in 2009 which should continue through at least 2018.

Habitat Status and Trend

After 4 years of data collection under CHaMP we are able to confidently produce robust estimates of habitat status. Data and complete results are available at https://isemp.egnyte.com/dl/qKVQ8KYvbo and by request. Although the status estimates are robust, with only 4 years of data we cannot distinguish short-term year-year aberrations from long-term linear trends. At this point, any statistically significant metric change should be interpreted as a significant difference across the 4 years sampled to date, and should not be interpreted as a likely indication of future trends or be used to predict future status.

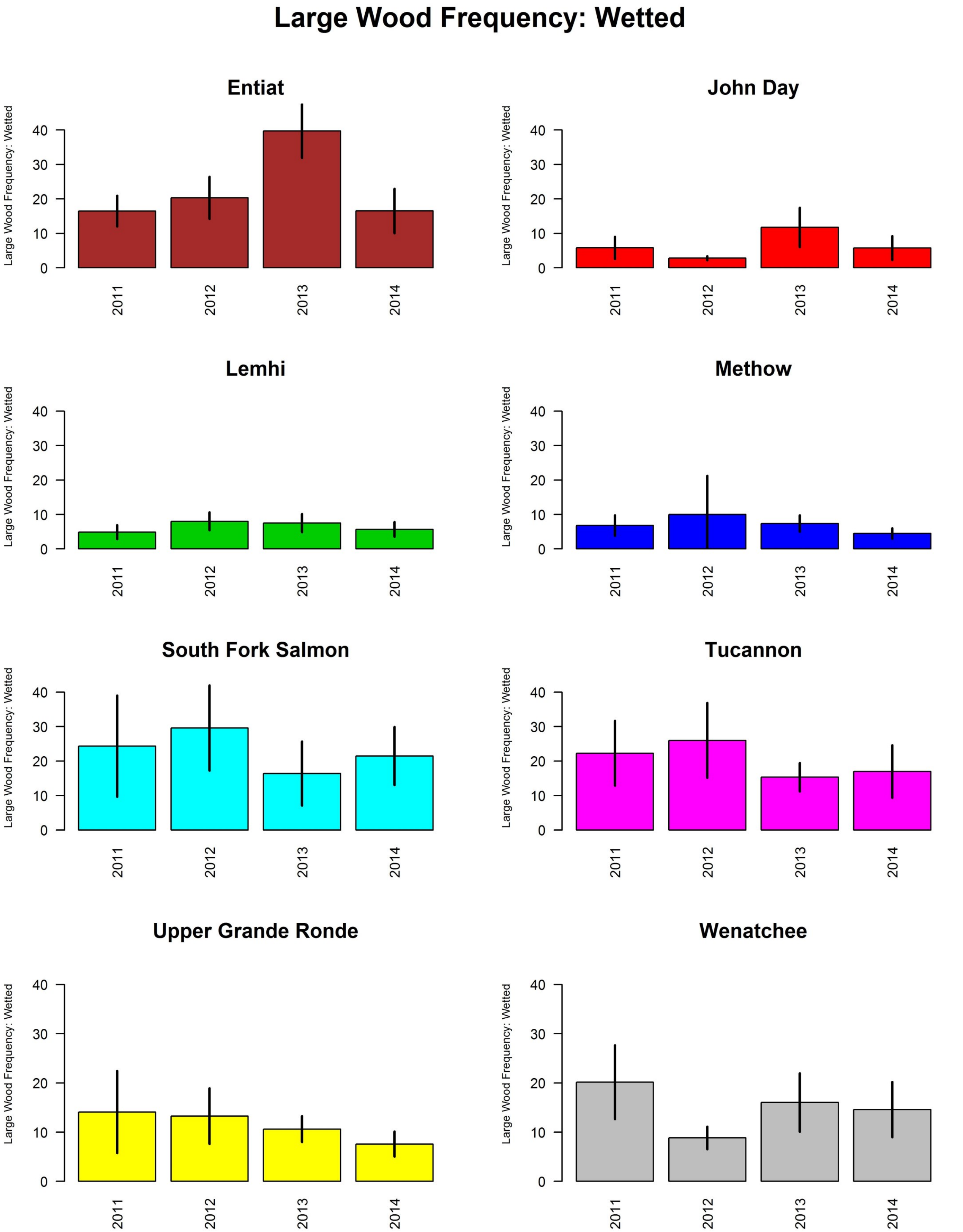
An updated a variance decomposition analysis (estimates the relative magnitude of the various variance components that sum to the total amount of variance observed in each CHaMP metric) showed that in 2014 crews implementing the CHaMP protocol continued to excel at collecting repeatable, standardized data and, in general, the amount of measurement noise, relative to other sources of variation, remained low and consistent with that observed in prior years.

Advancements also continued in the "lab", for example, we continued to refine and improve the Geomorphic Change Detection software to quantify changes in habitat status over time and test restoration design hypotheses. This software now does a better job of distinguishing real changes from noise, a significant improvement.

(Right) CHaMP successfully collected data from across 12 watersheds and at 34 AEM sites in 2014 and these data can be displayed and analyzed a number of ways. For example, the figure to the right shows the status by year (2011—2014) for eight watersheds (Entiat, John Day, Lemhi, Methow, South Fork Salmon, Tucannon, Upper Grande Ronde, and Wenatchee) for the estimated mean Large Wood Frequency: Wetted (1/m). This type of output is available for all the metrics generated under CHaMP.

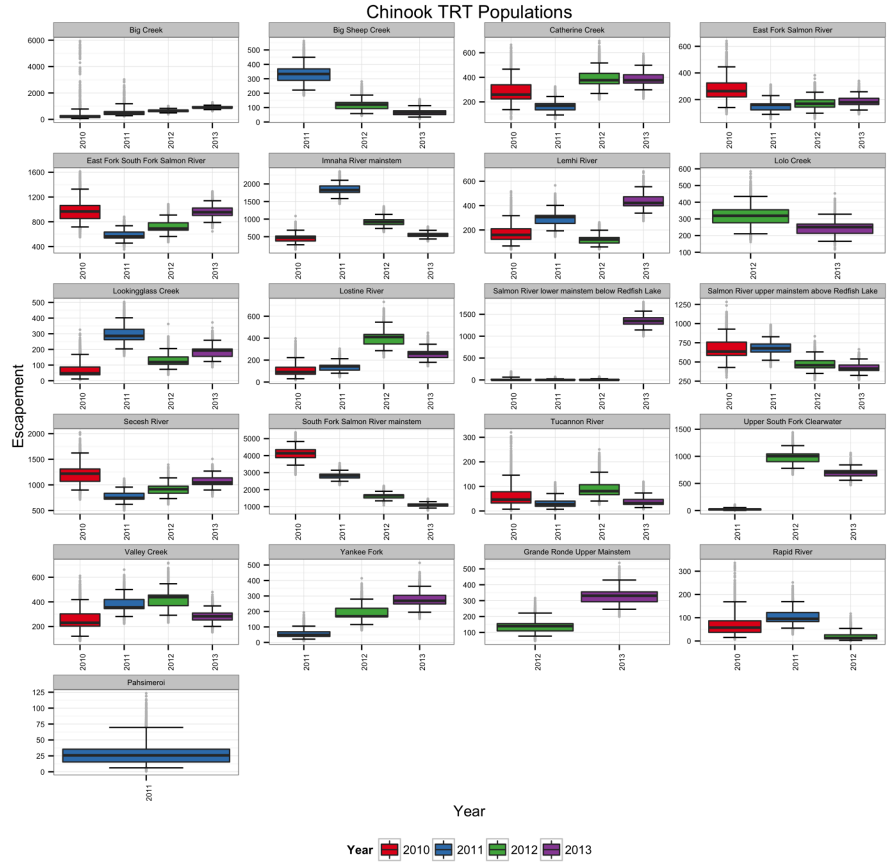
(Below) CHaMP personnel collect data on the amount of pool tail fines at a site as part of training at the annual CHaMP camp.

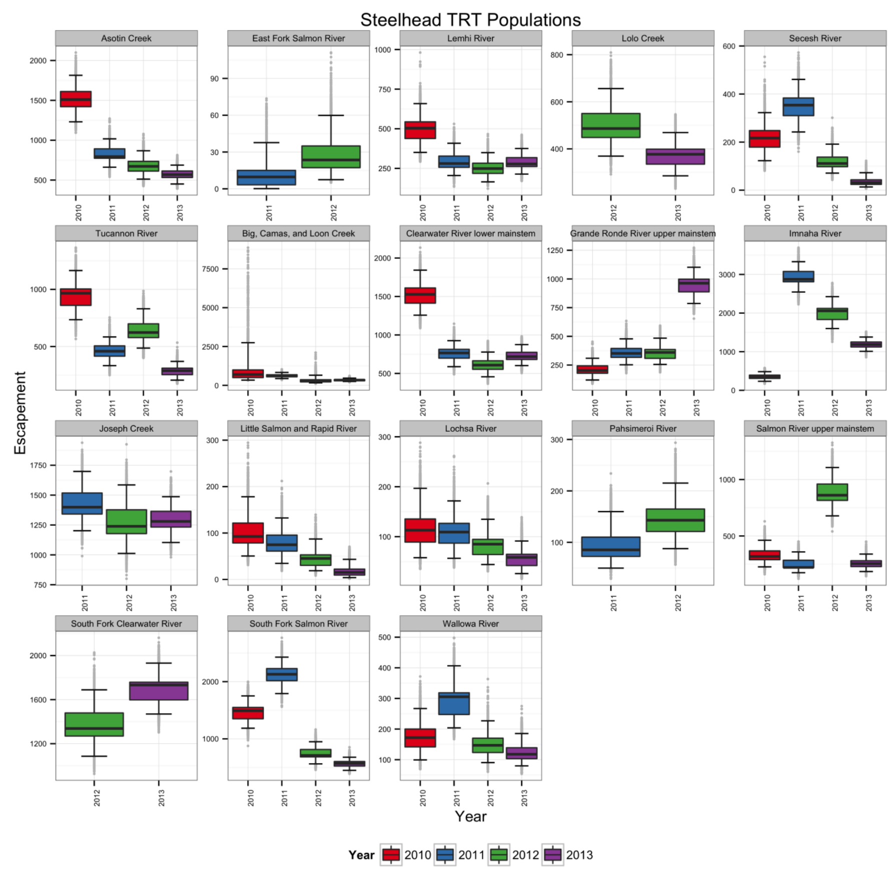

Annual CHaMP Field Training, Cove, OR. June 2015. Photo courtesy of Shelley K Photography


Large Wood Frequency: Wetted
Entiat
John Day
Lemhi
Methow
South Fork Salmon
Tucannon
Upper Grande Ronde
Wenatchee
Large Wood Frequency: Wetted
0
10
20
30
40
2011
2012
2013
2014

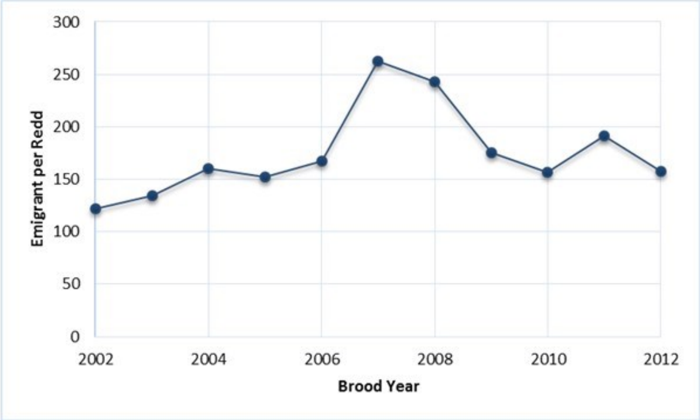
Fish Status and Trends

ISEMP personnel continued to collect and analyze data to estimate the status of parr in the Salmon and Entiat River subbasins in 2014 and to improve methodologies for collecting this data. This includes a spatially continuous approach developed in the Lemhi IMW that has improved estimates significantly by decreasing measures of uncertainty. Adult escapement estimates come from PIT-tag based methodology and spawning ground surveys. Estimates of parr abundance and adult escapement allow us to estimate productivity, for example, for spring/summer Chinook in the Entiat and Secesh. Neither population show much evidence for or against density dependence; however, these are small datasets and revisiting this analysis with more years of data should provide more insight into the populations' dynamics.

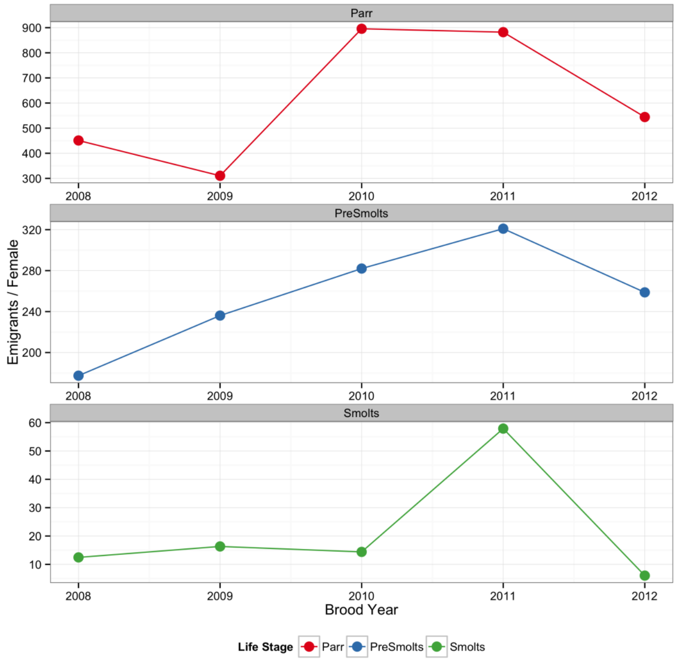




Spring/summer Chinook salmon (top) and steelhead (bottom) escapement estimates for TRT populations, generated using an approach developed by ISEMP personnel based on PIT tags and which has known statistical properties.



Time-series of productivity for spring Chinook salmon in the Entiat River subbasin, defined by emigrants per redd, 2002—2012



Time series of productivity for spring Chinook salmon in the Secesh River subbasin, defined by emigrants per adult female, 2008—2012.

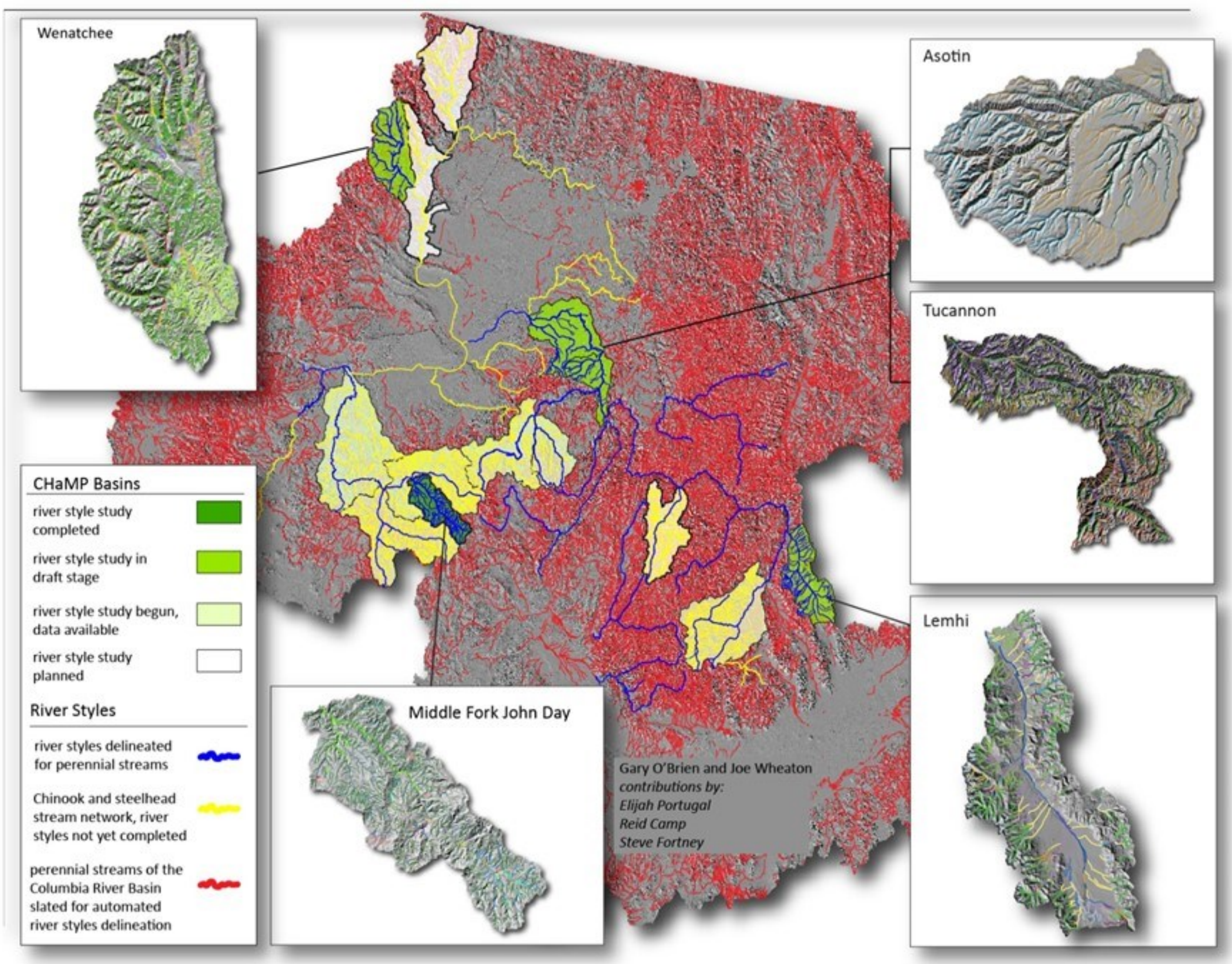
Sampling for juvenile Chinook and steelhead in the upper Lemhi River, 2014.

Analytical Tools to Expand beyond Sampled Watersheds

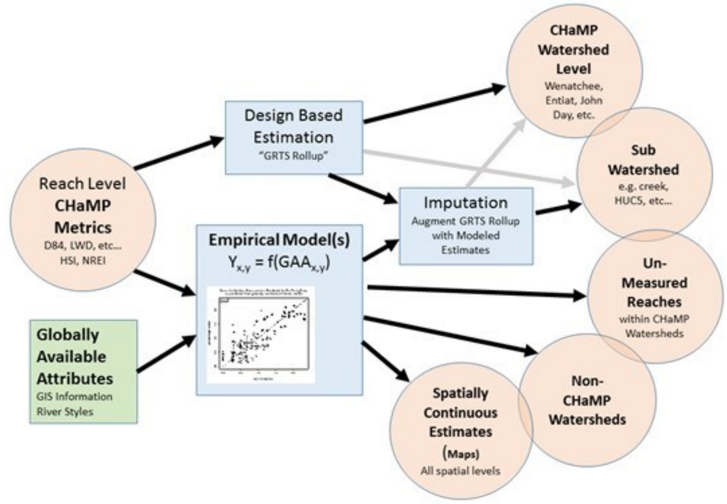
We made good progress in 2014 on advancing a variety of statistical techniques to extend point-level measurements to a variety of spatial scales, including watershed, subbasin, or ESU. We have developed and standardized empirical models that can be used to make direct estimates of CHaMP metrics at unmeasured reaches within watersheds, or in watersheds for which no CHaMP data exist. Of course, we urge caution in extrapolating models into unsampled watersheds, but cross-validation and residual analysis suggest that many of our empirical models do an excellent job of describing populations at the watershed level and extrapolation will be useful and appropriate.

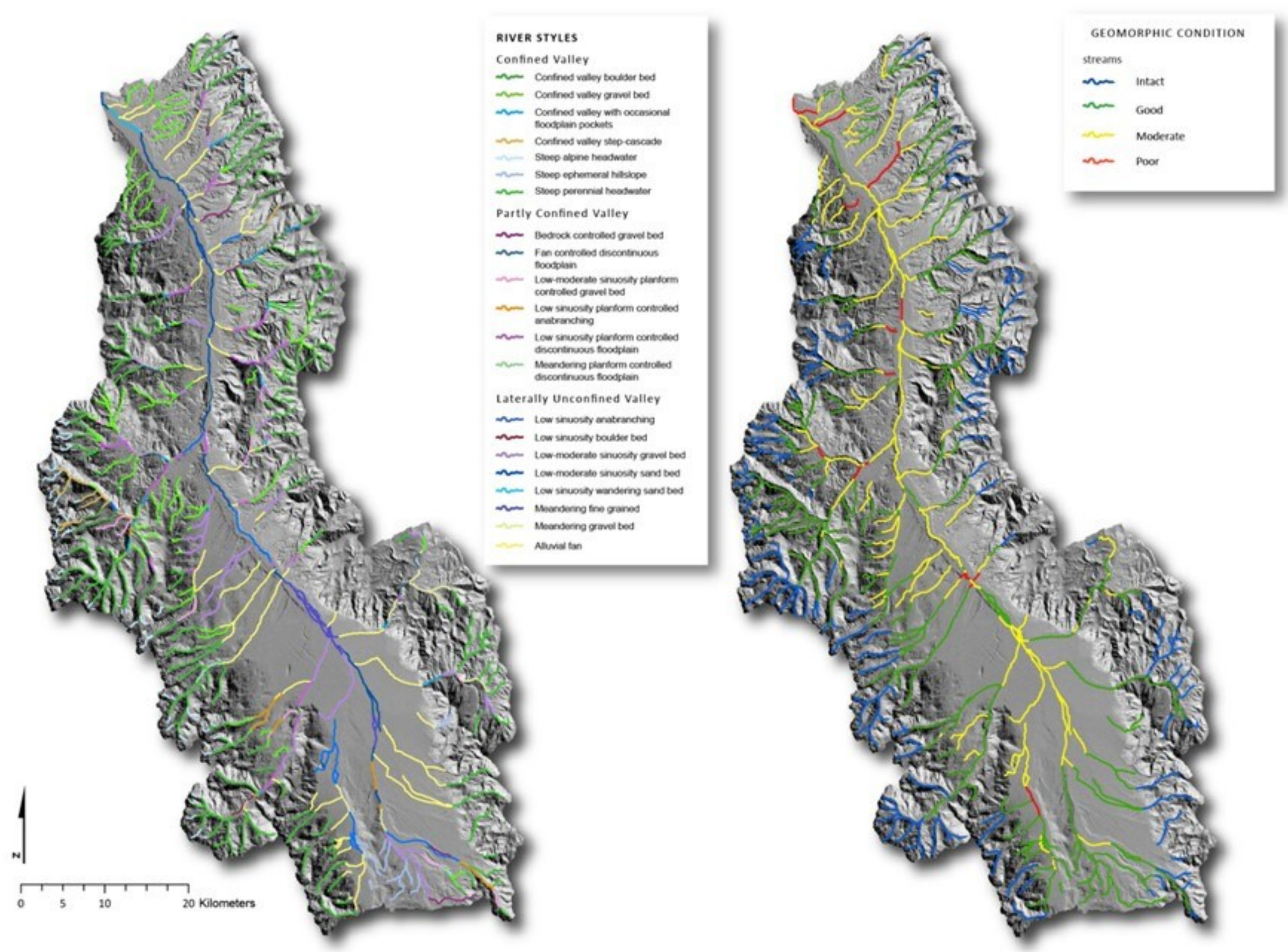
The outputs from automated tools used to delineate valley bottom and channel sinuosity show incredible promise. The River Styles framework, for instance, allows us to describe stream character and behavior and determine geomorphic condition, and feeds into our analysis of the recovery potential of streams in CHaMP watersheds. In 2015, we are focusing on using River Styles to produce condition maps to support the 2016 Expert Panel process and the 2018 AMIP process. We believe that our application of River Styles combined with new automated tools to characterize watersheds across the Columbia River Basin in terms of River Styles, will pave the way for determining river character and behavior, geomorphic condition, and river recovery potential across priority basins of the entire Columbia River Basin region.

Substantial progress was made in 2014 by CHaMP personnel applying River Styles in CHaMP subbasins throughout the Columbia River Basin, as shown in the map below. We are also making good progress identifying River Styles at the watershed scale, for example, the map below depicts the River Styles Stage 1 (character and behavior). and the map at right shows Stage 2 (geomorphic condition, right panel) defined for the Lemhi Watershed (HUC 8) in the southeast Idaho Batholith physiographic region.



Statistical tools for scaling habitat data from local to population scales. Circles show various spatial scales at which inference may be made. Blue boxes represent statistical tools used to translate from reach level CHaMP data to various spatial scales. The green box indicates globally available attributes - attributes available at all locations along the stream network, not just CHaMP sites.





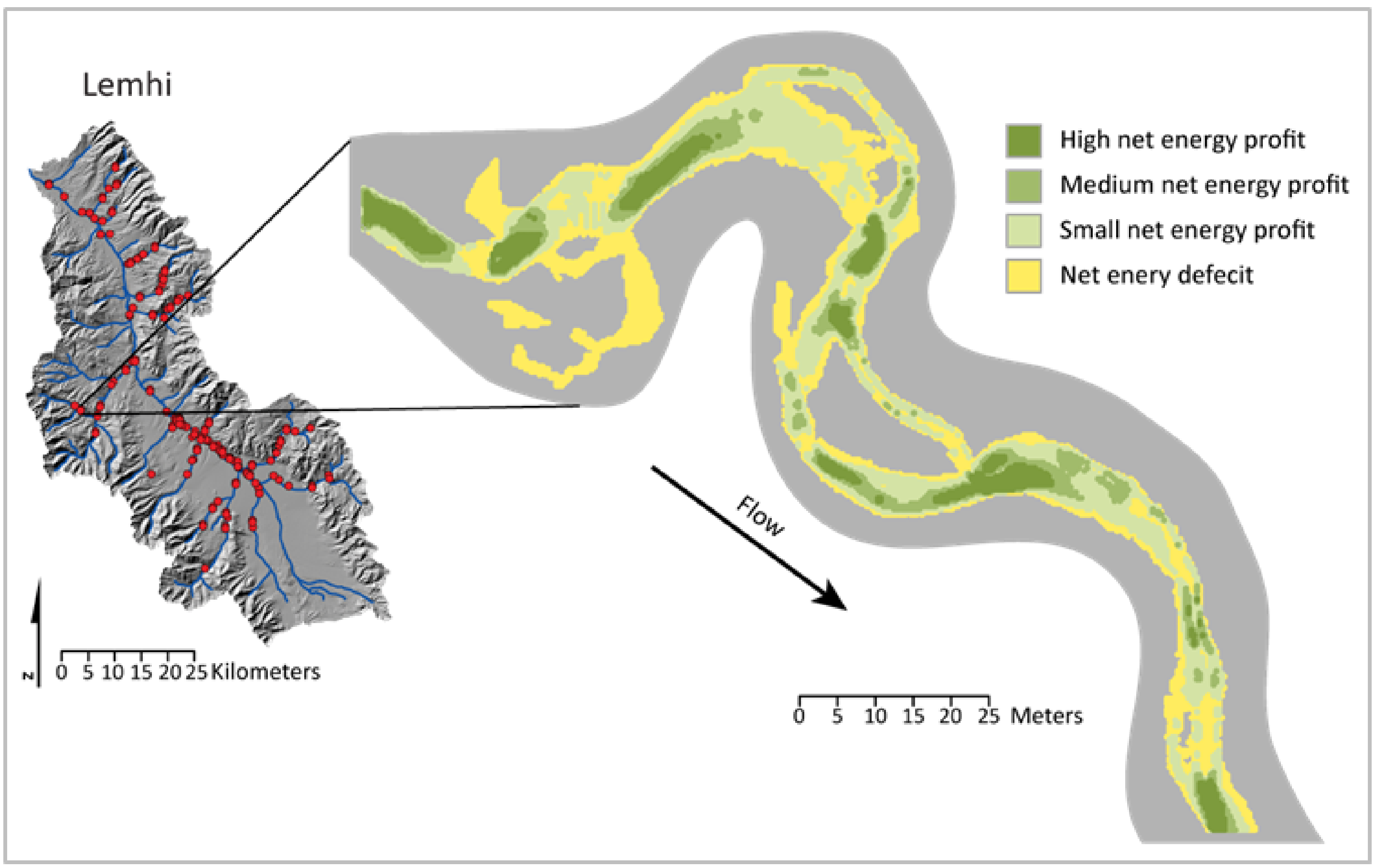
Fish-Habitat Relationships

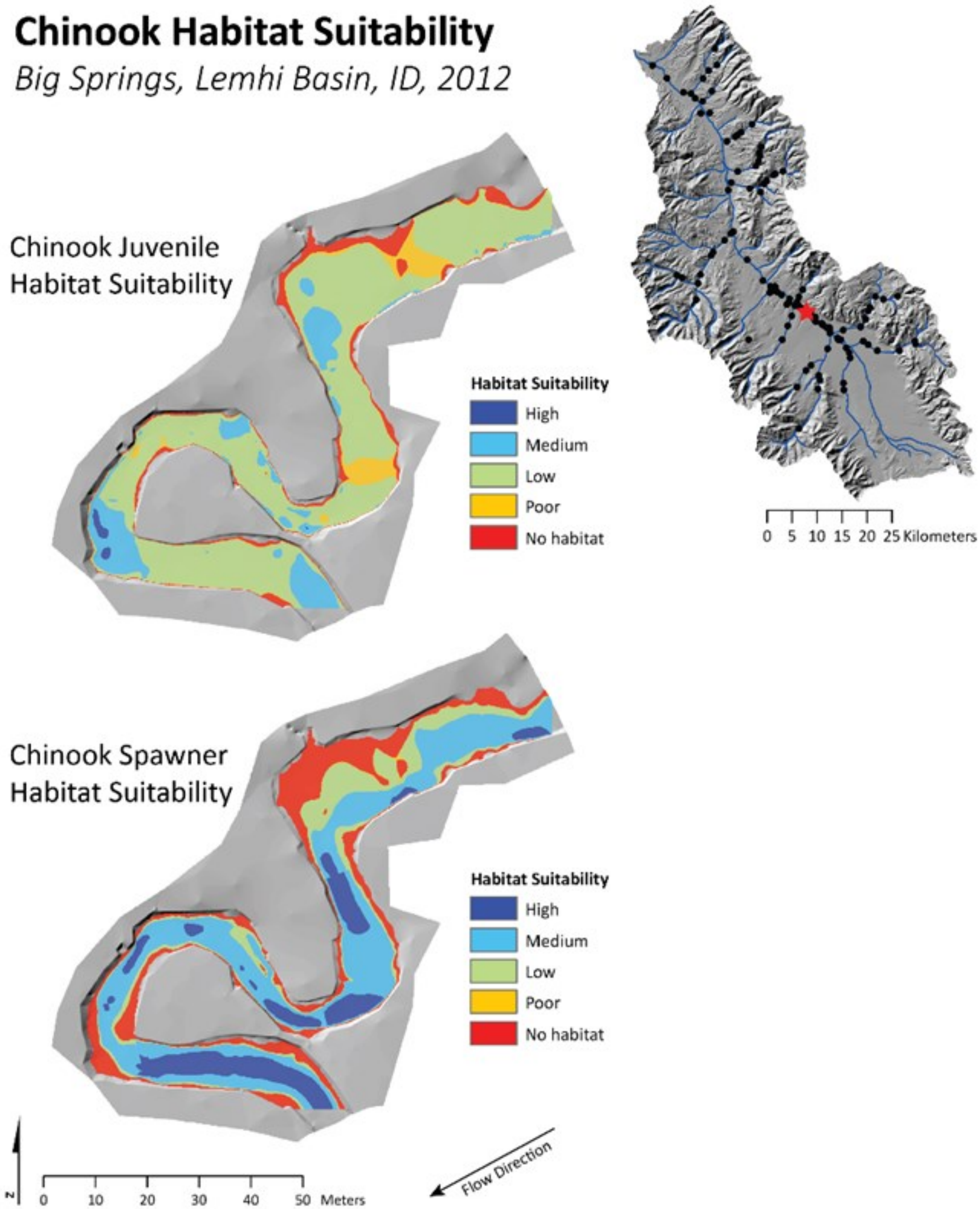
Estimating current carrying capacity for rearing parr and identifying the important habitat components that influence that capacity is necessary to effectively direct restoration actions as well as provide inputs for a variety of life cycle models. We made good progress in 2014 identifying fish-habitat relationships that can quantitatively predict capacity and survival and are robust across the Columbia River Basin landscape.

The NREI model continues to be central to advancing our understanding of fish-habitat relationships and providing parameters for life cycle modeling that allow us to link habitat changes to fish response. The model is now operational and has been used to simulate NREI and carrying capacity in the Asotin, Entiat, John Day, and Lemhi for 2011-2013. The NREI model currently produces a collection of outputs including raw NREI estimates, predicted fish locations, look-up tables of temperature- and drift-dependent capacity estimates, and plots displaying the spatial distribution of NREI estimates at CHaMP sites. These outputs can be used to compare habitat quality, fish capacity, and alternative habitat scenarios. We have been able to respond to watershed managers and provide reach/site-level carrying capacities for spring Chinook and/or steelhead where requested.

We made an incredible amount of progress in 2014 estimating habitat suitability and carrying capacity for spring Chinook and steelhead juveniles and adults through development of the Habitat Model. The Habitat Model is operational, and has been run using data from the Asotin, Entiat, John Day, Lemhi, Tucannon, Upper Grande Ronde, and Wenatchee for 2011-2013 for both spring Chinook and steelhead juveniles in the summer period and spawning adults. We are now working on modeling winter juvenile habitat, an over-looked limiting factor in many places, and building a new set of fuzzy inference-based criteria. As noted earlier in the use by managers section, the Habitat Model is being used by BPA engineers and habitat practitioners alike to guide restoration planning.

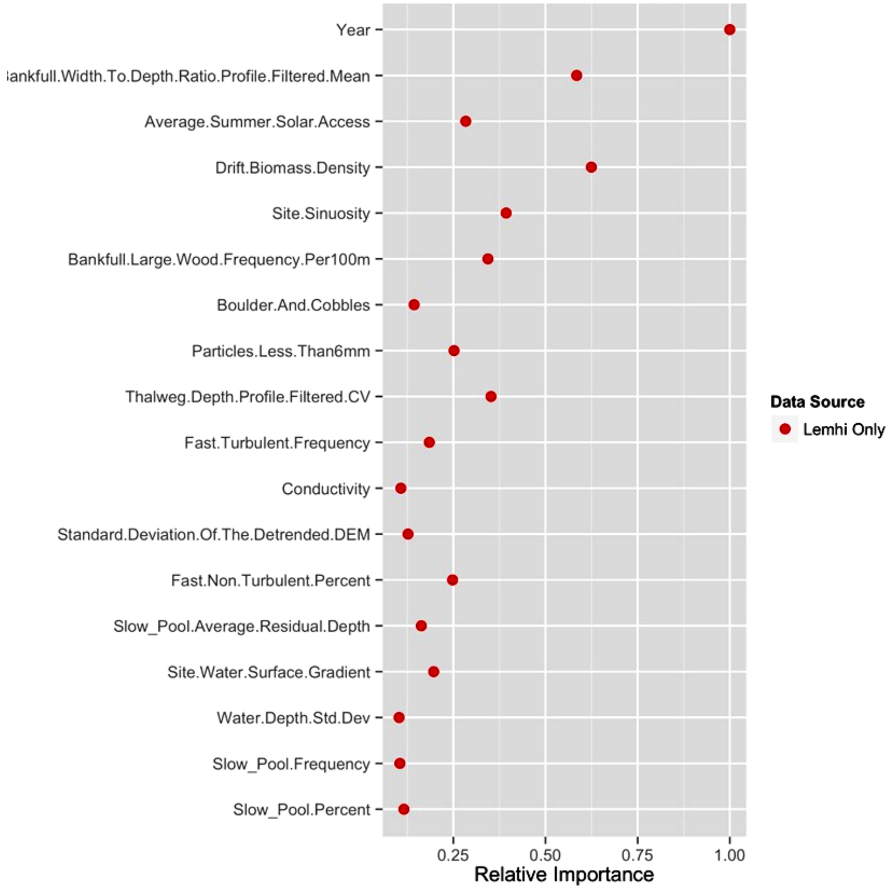
Using data collected at a CHaMP site, in this case from Big Springs Creek in the Lemhi basin, as input into the NREI model, we can predict and map the spatial distribution of good (capable of supporting fish growth, dark green areas) to poor (fish will lose weight, yellow areas) habitat for juvenile Chinook. This gives us an estimate of the carrying capacity of the reach which is used in as an input to various life cycle models. The small map shows the location (red dots) of CHaMP sites within the Lemhi and the location of Big Springs Creek.

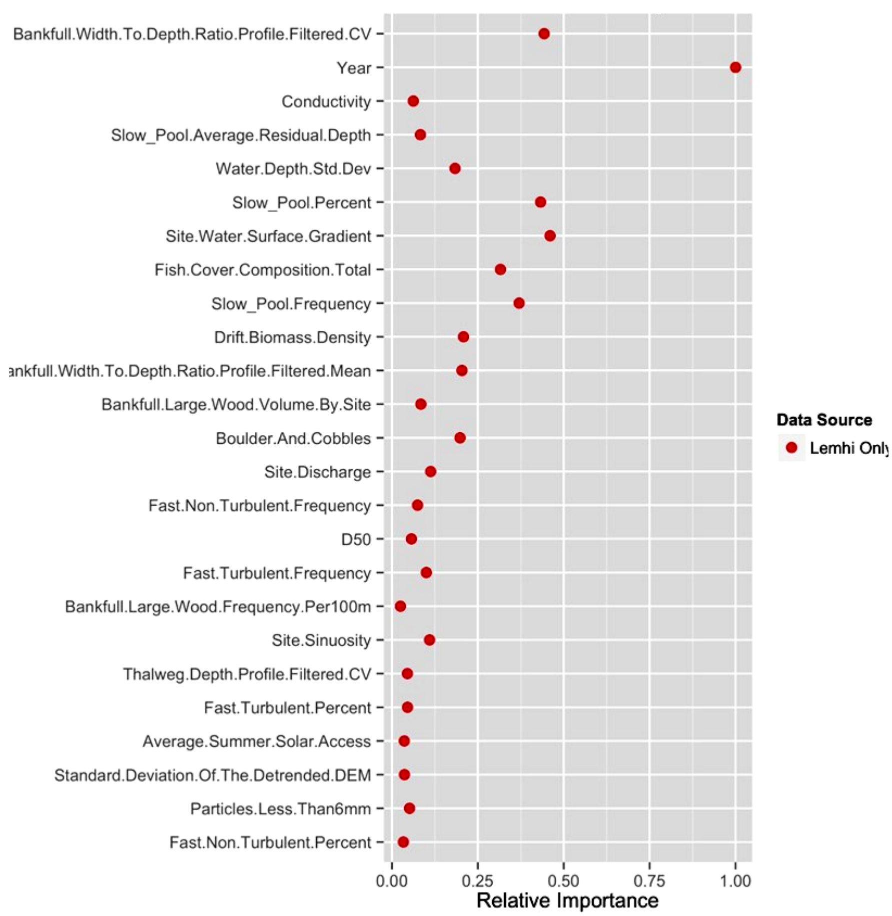




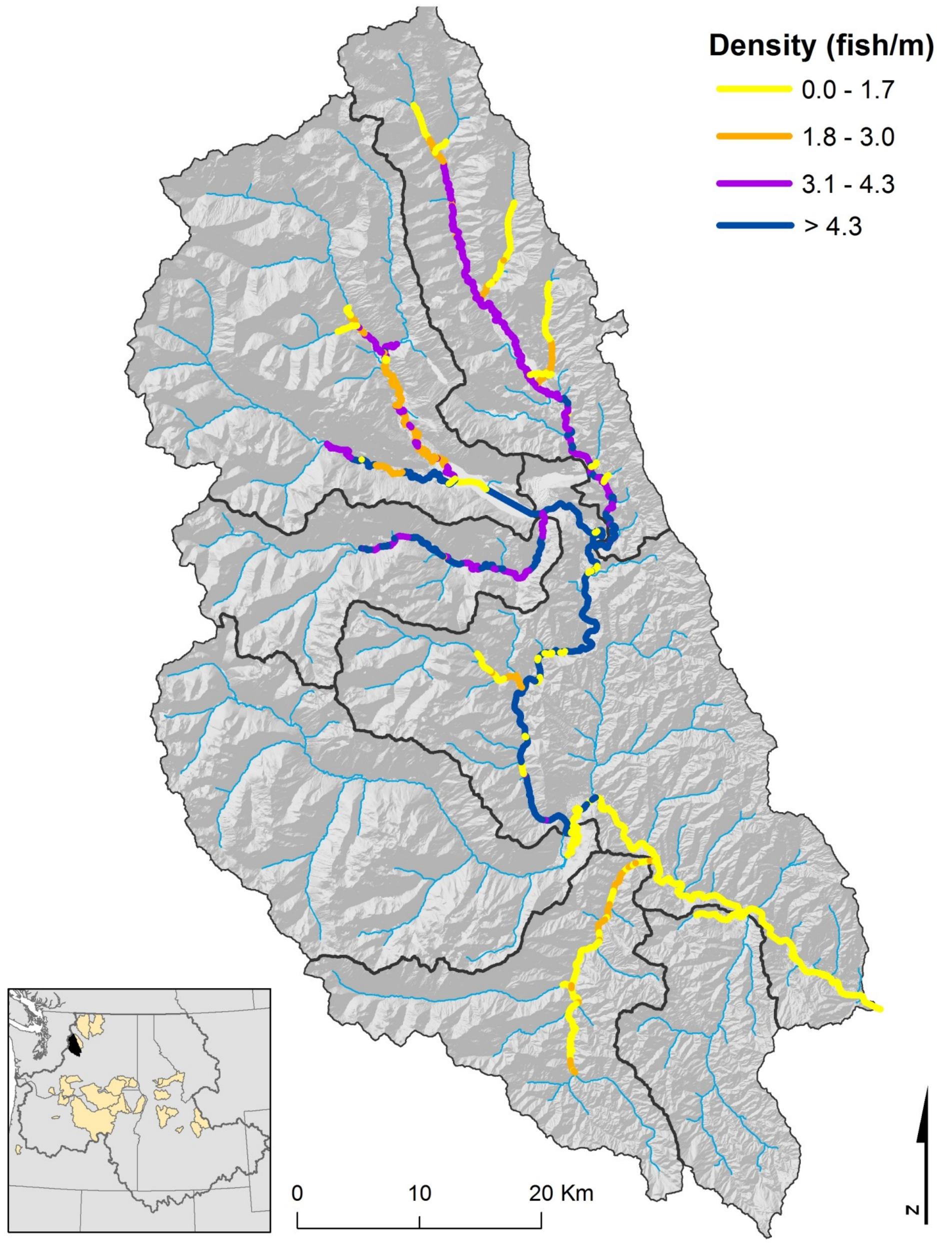
Output from the Habitat Model can be displayed as a map, making the data easily accessible. Here, spawner and juvenile Chinook habitat suitability rankings are generated using the Habitat Model for a Big Springs CHaMP site on the Lemhi. Selected CHaMP metrics combined with a hydraulic model are used to generate habitat suitability indices. These indices are ranked from high quality habitat (purple areas within the stream reach) to no suitable habitat available (red areas within the stream reach). This information can be used to guide where restoration actions should be targeted and at what life stage they should be designed to address, and assess the effectiveness of restoration actions using pre– and post-restoration CHaMP data.

ISEMP personnel also continued to refine work with quantile regression forests (QRF) over the course of 2014. QRF is a powerful statistical tool that allows us to visually examine the effect of a habitat metric on a fish response while assuming all the other habitat metrics remain at their mean values. We can also generate estimates of carrying capacity based on describing the entire distribution of predicted fish densities for a given set of habitat conditions, not just the mean expected density. This is a more realistic way of estimating carrying capacity since observed densities at the site scale are rarely equal to a site's carrying capacity due to unmeasured or unaccounted for variables. Presentations of this approach have been well received (e.g., to fisheries managers at the Upper Columbia Regional Technical Team), and comparisons of QRF output with NREI, the Habitat Model, and data collected by collaborators in the Upper Columbia has shown good correlation among the different methods. This suggests they are all driving toward the same truth about carrying capacity. We have also developed tools to extrapolate these estimates continuously across larger scales, for example, watersheds or subbasins.





These figures show the relative ranking of habitat metrics associated with quantile regression forest (QRF) carrying capacity estimates for steelhead (top) and Chinook (bottom) juveniles in the Lemhi River. This approach helps us to identify which metrics are most important for predicting carrying capacity. For steelhead the top four metrics that best predict carrying capacity are year, instream complexity, how sinuous the stream is, and the density of drift biomass. For Chinook the top four metrics are year, instream complexity, stream gradient, and the amount of pool habitat available. The fact that year is the most important predictor of carrying capacity underlines the necessity of long-term monitoring programs to capture the range of interannual variation.



This map illustrates the extrapolation of estimated carrying capacity using a quantile random forest approach for every reach in the Wenatchee within the Chinook domain. Capacity classes were chosen based on natural breaks in the distribution of capacity estimates. High density sites represent high carrying capacity and are shown in blue on the stream network, low density sites represent an estimate of poor carrying capacity and are shown in yellow on the stream network. This approach provides project sponsors with a powerful tool to identify priority reaches for restoration.

Tools to Leverage Fish Data

Over the course of 2014 we continued parameterizing and refining the ISEMP life cycle model, an approach to combining fish and habitat data to evaluate the influence of habitat on the freshwater life stages of Chinook and steelhead within the context of their whole life cycle. The model has been parameterized for both Chinook and steelhead in the Lemhi (the Idaho Office of Species Conservation has begun to use the model for work in the Lemhi River), for steelhead in the John Day, and parameterization is underway in the Entiat and Wenatchee. Coordination with other aspects of ISEMP's tool development (e.g., identifying fish-habitat relationships) has bolstered advancements in the life cycle model to allow us to tie a fish population response to changes in habitat. We anticipate output being available to project sponsors and collaborators for use in 2015.

In 2014 we also made good progress on tools that leverage different types of fish data collected by ISEMP and collaborators in the Columbia River Basin. We significantly advanced development of tools that allow fish managers to better estimate total adult escapement over a dam and partition the escapement into tributaries, while analysis on data from steelhead scale samples in the Lemhi has produced a tool to age emigrating steelhead so that they can be assigned to a brood year. This is potentially an invaluable tool for fisheries managers tracking the status and trend of steelhead populations, particularly for productivity. In fact, in 2014 ISEMP personnel were able to assign a brood year to every juvenile steelhead emigrating out of the Lemhi, and the tool has now been adopted in the Entiat IMW.

Collaboration With Other Habitat Monitoring Programs

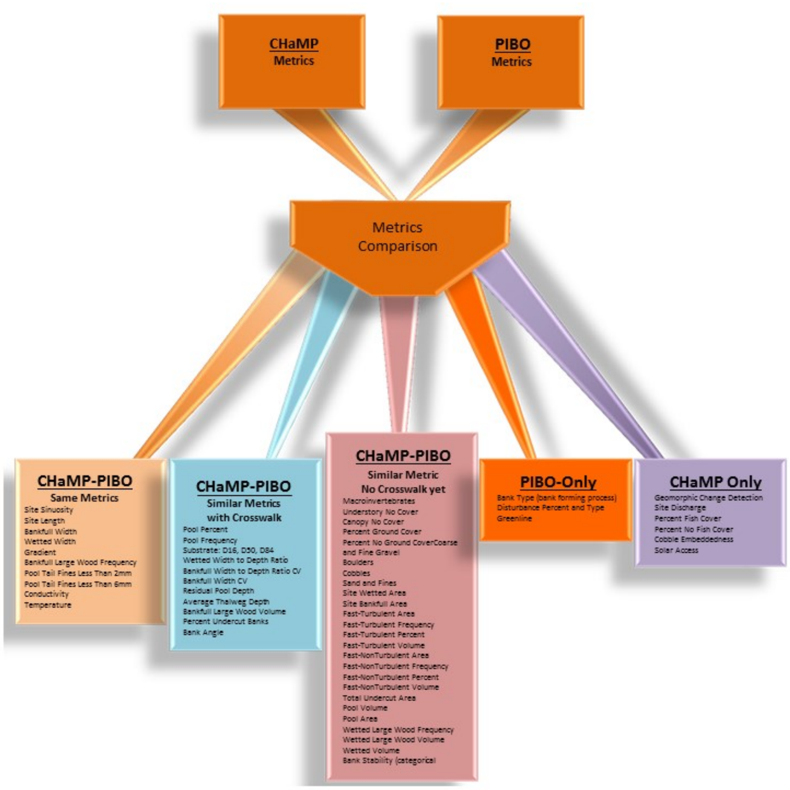
PIBO

In 2014 CHaMP and PIBO continued collaboration to develop an interoperable dataset using a set of metric data that are monitoring program independent. Initially, we had identified that it would be possible to crosswalk 24 metrics, with an additional 26 identified that could be transformed with a little more effort. To generate this dataset, some metrics needed to be transformed (12, linear transforms only), some needed to be constructed from measurements (2), while others (10) mapped directly from one program to the other (see CHaMP 2015). In the fall of 2014, CHaMP developed a demonstration project to show the ability to adjust or transform three univariate metrics (temperature, pool frequency, and large wood frequency) that have known mathematical relationships (crosswalks) between the two programs. Geographically, the scope of this effort was limited to three species and 5 ESUs: Snake River spring-summer Chinook, Upper Columbia spring-summer Chinook, Mid-Columbia steelhead, Snake River steelhead, and Upper Columbia steelhead. BioAnalysts, Inc. provided metric threshold determinations that Sitka Technology Group used with the shared CHaMP-PIBO metrics to create an interactive map application and color-coded displays. These displays were based on user-defined categorizations of quality; "rollup" areas were color coded based on simple characterizations of site-level surveys to estimate condition at successively larger scales, all the way up to the ESU and basin scale.

The CHaMP-PIBO data integration effort was an important first step in generating a regional approach to the management, distribution and reduction of stream habitat monitoring data. There is no reason that the CHaMP-PIBO experience should be unique; crosswalks between other metric sets could be developed and housed in the integrated data management system. This does not go all the way to the development of a data exchange template (MMX) for regional stream habitat data, but the crosswalk algorithms are a necessary component of an exchange format for relevant metrics and necessary for determining the extent to which the integration is possible. PIBO and CHaMP are moving beyond the MMX template idea to try cross-program analyses where each program's data are incorporated by the other program to increase coverage and sample size. To date, these analyses are not mature enough to report on, but the ability to support regional decision making with data from multiple regional monitoring programs is being developed.

AEM

CHaMP successfully intensified its coordination efforts with the regional AEM program in 2014 to ensure standardization between shared sites, metrics, and protocol elements, and to maintain the integrity of the CHaMP survey design while accommodating the addition of new AEM sites if requested. CHaMP training in 2014 was set up to accommodate an AEM-specific module and discussion, and crews from both programs benefitted from a combined CHaMP-AEM data collection application and new tablet platform as a result of collaboration between these two programs. Efficiencies were also realized through use of a common data management and QA/QC environment and tools.



Working with PIBO staff, CHaMP personnel identified metrics collected by both programs that did not need a crosswalk (peach box), metrics that required crosswalking (blue box), metrics that are only collected under PIBO (orange box) and only under CHaMP (purple box), and a list of metrics for which no crosswalk has yet been developed (pink box).

Recommendations

As the end of the current BiOp approaches in 2018 and ISEMP and CHaMP personnel continue to work to develop guidance for BPA on the best practices to monitor and restore Chinook and steelhead populations in tributary habitat across the Columbia River Basin to meet requirements called for in the BiOp, we recommend that both programs continue forward through 2018 without any major changes. With 4 years of high quality habitat data now in hand from CHaMP and the IMWs producing results on the effectiveness of restoration actions, we believe we are at a point in our journey where we can generate useful and usable tools for the management and policy communities. Our recommendations include:

1) Complete the IMW study designs as originally proposed (i.e., through 2018-20). We have yet to show increases in freshwater at the scale of a TRT population, although we have evidence from Bridge Creek and Little Springs Creek that habitat restoration can increase survival, productivity, and abundance. We are optimistic that habitat restoration can change freshwater productivity at larger spatial scales but we need to stay the course in order to document this.

2) Continue development of ISEMP and CHaMP tools that link habitat and fish and allow extrapolation to less densely sampled areas. This initiative will ultimately improve our ability to define limiting factors, quantitatively evaluate restoration potential, and simulate suites of restoration actions to support the identification and implementation of restoration actions with the highest probability of success at the lowest cost. Specific areas of focus should include:

- further validating model predictions using empirical data, and
- continuing to develop and test application of watershed-level context for all watersheds in the Columbia River Basin through extrapolation using empirical models, and the application of network models, such as River Styles.

3) Advance the development of data management and dissemination systems to render synthesis tools “usable” by non-project personnel, and facilitate quick and efficient distribution of program data and products. This effort would involve continuing work to improve user-interfaces and possibly training workshops.

4) ISEMP and CHaMP data and tools should be leveraged to identify locations where information limits the effective imputation of program results – where are the black holes, and what is the type or level of effort necessary to better support imputation?

5) Continue large scale monitoring and evaluation initiatives such as CHaMP and the LGR run decomposition, as these initiatives provide standardized metrics across a large spatial domain with known statistical qualities.

6) BPA should consider whether/how the tools developed by ISEMP and CHaMP could cost-effectively and reliably replace existing data streams, for example, replacing redd surveys with the PIT-tag based run decompositions.

ISEMP and CHaMP staff will also develop plans within the coming months to address the following comments and requests from the ISAB:

Publications

1) Develop manuscripts for publication that describe the lessons learned about habitat monitoring, including, for example, guidance on efficient methodology and fish habitat relations.

2) Address the cost-effectiveness of CHaMP surveys, which is of great interest inside and outside the Columbia River Basin.

3) Publish the novel statistical approaches to analyzing CHaMP data.

Management Questions

1) The Program assumes a lot can be done with habitat restoration to rebuild fish populations in the Basin. Are we getting closer to some answers, and realistically how long will it take before we can know whether habitat restoration alone can restore fish populations?

2) A lot of good work has gone into comparison, crosswalk, coordination and collaboration and they are really happening at a new level. Are we closing in on a consistent, efficient, repeatable methodology that can be shared and widely compared in time and space?

INTRODUCTION

This annual technical report to Bonneville Power Administration (BPA) covers Calendar Year (CY) 2014 of the Integrated Status and Effectiveness Monitoring Program (ISEMP; BPA Project 2003-017) and the Columbia Habitat Monitoring Program (CHaMP; BPA Project 2011-006). This is the first combined annual report from ISEMP and CHaMP; in previous years each project has produced its own annual report, but beginning with the 2014 annual report we will publish a combined report to reflect the intertwined nature of the work of ISEMP and CHaMP. While ISEMP and CHaMP are two distinct programs under BPA's Fish and Wildlife Program, key personnel, goals, objectives, and products are shared between them. In general, the same staff is responsible for the development and implementation of both programs, and data from both programs are leveraged to develop products to support decision makers. For example, CHaMP collects habitat data that ISEMP uses in the development of fish-habitat relationships and other products needed by the policy and management community.

A shared goal of ISEMP and CHaMP is to develop and export standardized monitoring and analytical approaches to help answer many of the RPAs posed in the 2008 BiOp related to the recovery of spring Chinook (*Oncorhynchus tshawytscha*) and steelhead (*O. mykiss*), including three management questions identified by BPA (see box). In this report we provide updates from CY2014 field work and analysis for both programs, layout a work plan for 2015 and beyond, and collate responses from past ISEMP and CHaMP reports to respond to ISAB, ISRP and Council reviews and questions. The report is structured around chapters focused on the main topics of analysis and development that ISEMP and CHaMP are currently working on and appendices.

CHaMP Camp, Cove, OR. 2015. Photo courtesy of Shelley K Photography

RPAs

RPA 50.1 Implement and maintain CRB PIT-Tag Information System

RPA 50.5 Provide additional status monitoring of SR B-Run Steelhead populations

RPA 50.4 Fund pilot studies in Wenatchee/Methow/Entiat

RPA 50.6 Review/modify existing fish pop status monitoring projects

RPA 51.1 Synthesize fish pop metrics thru Regional Data Repositories

RPA 52.4 Provide additional PIT-tag marking of UCR populations

RPA 56.1 Implement research in select areas of the pilot study basins

RPA 56.2 Implement habitat status/trend monitoring as component of pilot studies

RPA 56.3 Develop strategy for habitat status/trend monitoring for ESA fish

RPA 57.1 Entiat-Study ways to improve channel complexity and fish productivity

RPA 57.2 Lemhi-Study reduce entrainment and provide better fish passage

RPA 57.3 Bridge Creek-Study treatments of channel incision

RPA 57.4 Wenatchee/Methow/John Day-Habitat/fish productivity assessment

RPA 71.4 Implement standard metrics, business practices, and information collection

RPA 71.5 Coordinate further development and implementation of the other Hs

RPA 72.1 Participate and jointly fund support in regional coordination forums

RPA 72.2 Fund data system components to support inform management needs of Hs

Key Management Questions

KMQ1: What are the tributary habitat limiting factors preventing the achievement of desired objectives?

KMQ2: What are the relationships between tributary habitat actions and fish survival or productivity improvements, and what actions are potentially most cost effective?

KMQ3: Are tributary actions achieving the expected level of habitat improvement, and associated biological response?

CHAPTER 1: STREAM HABITAT IMPROVEMENT AND FISH POPULATION RESPONSE AT THE WATERSHED SCALE: RESULTS FROM THE INTENSIVELY MONITORED WATERSHEDS

Introduction

Achieving the statistical power to detect changes of potentially limited magnitude in freshwater productivity as a response to habitat restoration actions within the timeframe of the BiOp requires intensive sampling, but monitoring at such a level is neither desirable nor affordable everywhere. Intensively Monitored Watersheds (IMWs) are an effective and efficient approach to determine the nature and magnitude of any fish population response to restoration actions. ISEMP personnel are implementing three IMWs: Bridge Creek IMW in the John Day subbasin in Oregon, the Entiat River IMW in the Upper Columbia subbasin in Washington, and the Lemhi IMW in the Salmon River subbasin in Idaho (Figure 1). In this CY2014 annual technical report we present an update of analyses from each IMW.

Adaptive Management of Restoration Actions

River restoration often requires the development of new approaches and designs that are best tested and achieved through adaptive management (Downs and Kondolf 2002). Although adaptive management is frequently touted as an important part of the restoration process, it is very rarely integrated into restoration plans. ISEMP personnel have employed an active adaptive or "experimental" management approach that is meant to ensure a rapid progression toward restoration goals, while also maximizing information gain and learning through restoration implementation. For example, ISEMP personnel are applying an adaptive management loop to the restoration treatments in Bridge Creek that follows a 4 stage process: i) *planning*, ii) *doing*, iii) *evaluation and learning* and iv) *adjusting* monitoring and restoration treatment actions (Figure 2). The *evaluation and learning* stage includes annual evaluations of IMW monitoring information such as presented in this report, explicit evaluations of restoration structures and structure complexes, and periodic system-wide reviews of the IMW restoration experiments in which findings are disseminated in a variety of reports, peer-reviewed publications, (e.g., Pollock *et al.* 2007 and Pollock *et al.* 2014), and presentations at professional conferences (e.g., national AFS conference in Portland, 2015).

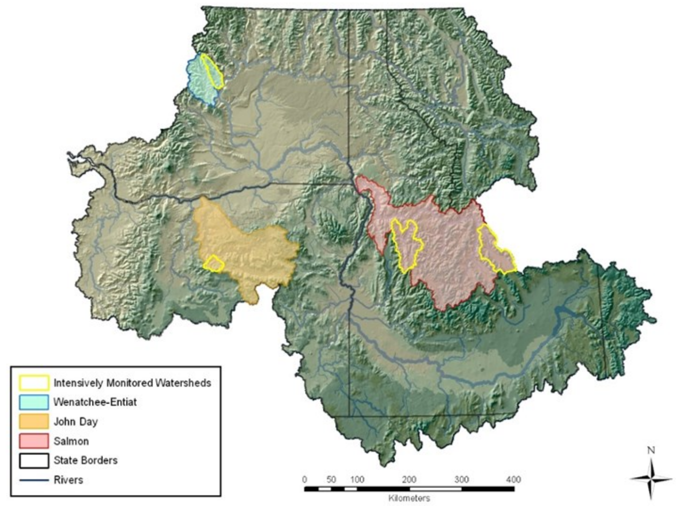


Figure 1. Location of the three Intensively Monitored Watersheds, Entiat River, John Day, and Lemhi River in the Columbia River Basin being implemented under Bonneville Power Administration's Integrated Status and Effectiveness Monitoring Program.

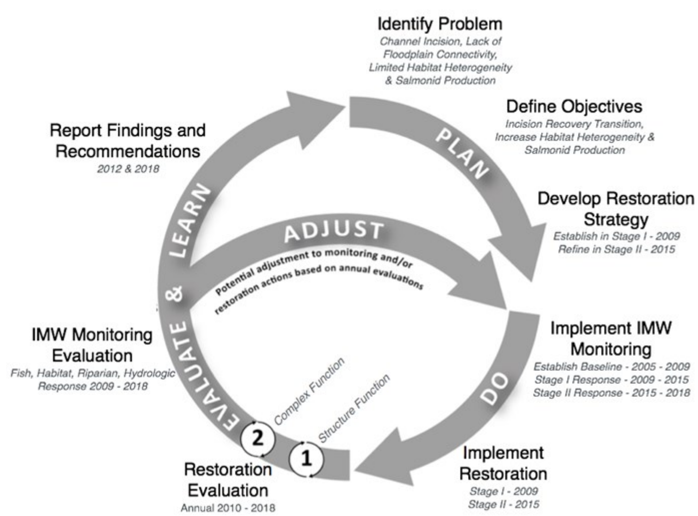


Figure 2. Conceptual diagram of the adaptive management process designed for the Bridge Creek IMW. Detailed sub-loops are entered for explicit evaluation of (1) individual BDA structures as well as (2) how structures work together in concert at the complex level.

Bridge Creek Intensively Monitored Watershed

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 3). 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. Despite its degraded state, 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.

Figure 3. Location of the Bridge Creek Intensively Monitored Watershed in the John Day subbasin, Oregon.

In 2009 the Oregon Department of Fish and Wildlife (ODFW) and NOAA-Fisheries drafted the Mid-Columbia Recovery Plan (NMFS 2009), where Bridge Creek was listed as one of 11 tributary Major Spawning Areas critical to the recovery of the Lower John Day steelhead Major Population Group, and was identified as having high restoration potential. The plan specifically identified potential factors limiting steelhead productivity on Bridge Creek, which include an overall lack of habitat diversity, quality, quantity, abundant channelization and streambank hardening, high substrate embeddedness, low summer flows, and high summer stream temperatures. Floodplain connectivity and riparian structure were also listed as having high restoration priority for the lower valley on Bridge Creek. Many of the limiting factors listed for Bridge Creek within the Mid-C Recovery Plan are characteristics associated with incised stream channels.

The Bridge Creek IMW is a long-term watershed-scale experiment monitoring stream and riparian habitat restoration and the response of a threatened population of steelhead. The Bridge Creek IMW represents a novel approach to restoration in which treatments, beaver dam analogs (BDAs), have been designed to mimic and work with the dam building activities of beavers in order to promote the recovery of incised stream channels (Figure 4 and Figure 5; for more detail see ISEMP 2011, 2012, 2013).

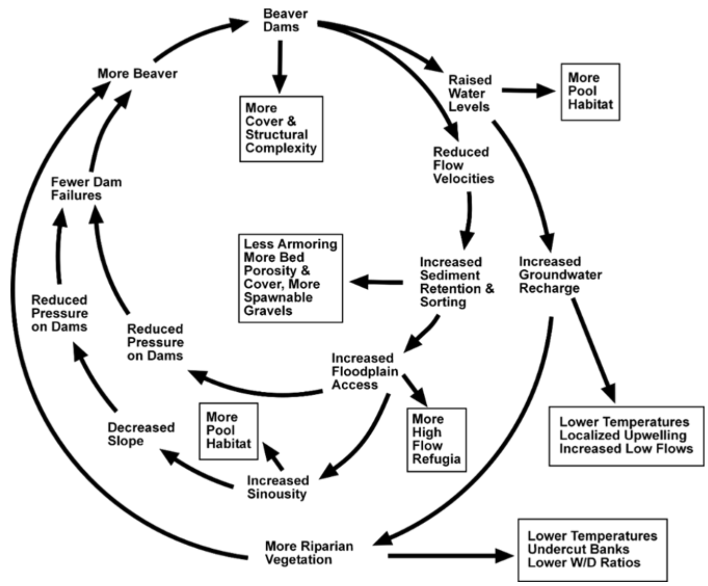


Figure 4. Beaver build dams in incised stream trenches that create positive feedback loops in terms of biological and physical processes that ultimately improve habitat for beaver, making it easier for them to sustain colonies and expand their population. These dam building cycles also improve salmonid habitat, highlighted in the boxes.

Figure 5. A beaver dam analog (BDA) used in the Bridge Creek IMW to encourage beaver to build dams on stable structures. Dams are expected to entrain substrate, aggrade the bottom, and reconnect the stream to the floodplain.

The Bridge Creek IMW project began with pre-project monitoring in 2006, and the first round of BDAs were installed in 2009 (Figure 6). Post-restoration monitoring as part of the IMW project is ongoing, and to date positive responses have been documented with regard to the condition of in-channel and riparian habitat, and the productivity of steelhead populations (Pollock *et al.* 2012). With a second stage of restoration scheduled for implementation in 2015, the Bridge Creek IMW project continues to provide valuable information that demonstrates the potential for stream restoration to improve salmonid population viability, the role of beavers in shaping salmonid habitat, and serves as a model for application of beaver-assisted restoration approaches within other impaired watersheds.

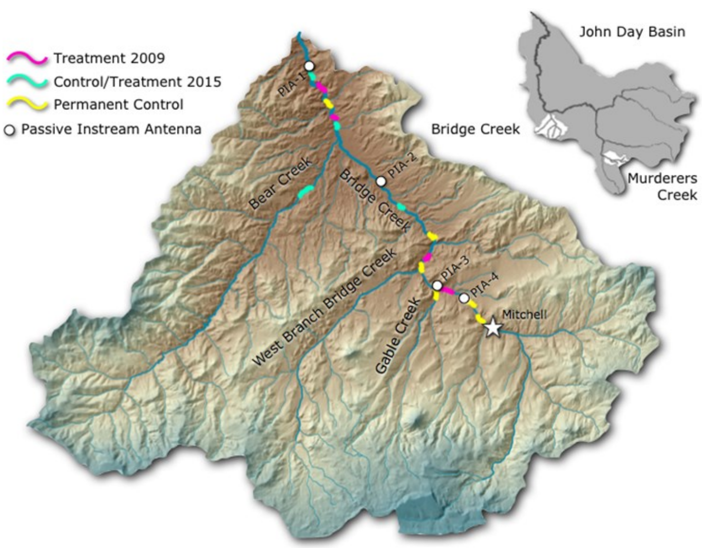


Figure 6. Map of the Bridge Creek IMW study area showing locations of Stage I treatment, proposed Stage II treatment, and permanent control monitoring reaches. The location of the external control watershed in Murderers Creek is also shown.

Hypotheses Tested

- Installation of BDAs will increase sediment retention, aggrade the stream bottom and cause a net aggradation effect in Bridge Creek.
- Water table elevation will increase as a result of the sedimentation and bed aggradation generated by BDAs.
- BDA installation will result in changes in vegetation composition, stem density, and beaver browse with distance from water
- Vegetation community structure and composition differs between sites and transects influenced by beaver activity and those not influenced by beaver activity.
- BDA installation will lower summer water temperatures.
- BDA installation will results in increases in juvenile steelhead abundance, growth, survival and productivity.

Restoration Implementation

More than 4 km of channel were treated with BDAs in the fall of 2009 throughout 4 incised treatment reaches each roughly 1 km in length (Figure 6). At the reach scale, structures were placed at a frequency to capitalize on all opportunities to promote aggradation and floodplain reconnection throughout the treatment area. Secondary structures were often placed a short distance downstream from a primary structure to avoid steep gradient drops within the treatment area that could potentially result in excessive scour, and limit the likelihood of head-cutting and undermining of structures upstream. Additionally, the presence of multiple structures in series provides capacity for beaver colonies to build multiple dams and establish a dam complex, which seems to provide additional resiliency in that the significance of any single dam failure is less important when an intact dam is in close proximity. This is important because beaver need a stable colony to consistently produce offspring. However, the dynamics of individual dam failure and evolution should not be confused as necessarily promoting an 'unstable colony' or 'unstable dam complex'. It takes 2 years to produce offspring, and if colonies fail in less than 2 years it limits the likelihood of colony persistence and of population expansion. In Bridge Creek, individual dam failure is so common (Demmer and Beschta 2008) that establishment of larger dam complexes and stable colonies have historically been rare.

Experimental Monitoring Design

The Bridge Creek IMW employs a Before-After-Control-Impact (BACI) experimental design for comparisons of physical habitat and steelhead responses between treatment and control reaches before and after the implementation of the restoration actions. The BACI design is also deployed in a nested hierarchy to compare restored and unrestored areas at the watershed, subwatershed, and reach scales. At the watershed scale, Bridge Creek is being compared to nearby Murderer's Creek, where intensive monitoring of steelhead populations and physical habitat conditions is ongoing. Within the mainstem of Bridge Creek comparisons are being made between control and treatment reaches. Pre-project data have been collected in Bridge Creek since 2006, and post-project monitoring will continue through at least 2018 (Table 1).

Table 1. Major monitoring components of the Bridge Creek Intensively Monitored Watershed study.

Type	Monitoring Component	Spatial Design	Temporal Design	Metric Description
Fish	Juvenile salmonid capture-recapture PIT-tag surveys	Roughly 1 km site within each control, treatment, and proposed treatment reach	Each site fished 3 times annually in summer, fall, and winter	Juvenile salmonid population estimates, survival, and growth rates
	Operation PIAs	4 PIA arrays spanning the mainstem Bridge Creek IMW study area	Continuous	Juvenile salmonid movement, smolt timing, smolt abundance. Adult salmonid spawning distribution
	Adult steelhead trap	Operated at the mouth of Bridge Creek	Peak of adult steelhead spawning from Feb, - May	Spawner estimate, smolts to adults, spawning distribution, hatchery - wild ratio
Habitat	CHaMP fish habitat and topographic surveys	Multiple sites within each control, treatment, and proposed treatment reach	Sites are sampled annually according to a rotating panel design	Fish habitat quality and complexity, topographic channel change. See https://www.champmonitoring.org for a complete list of metrics and descriptions
	Stream temperature	22 at the up and downstream end of each monitoring reach	Continuous	Pre-post treatment - control comparisons of temperature change, longitudinal change in stream temperature
	Groundwater elevation monitoring	Groundwater monitoring wells at 1 treatment and 1 control reach	Continuous	Pre-post treatment - control comparisons of groundwater elevation
	Riparian vegetation	Continuous throughout study area and high resolution transect-level monitoring within areas of interest (specifically floodplains and benches).	Continuous monitoring every ~5 years using remote imagery. Transect-level monitoring annually pre-restoration and at 1, 2, 3, and 5 years post restoration.	Classification of remote aerial imagery to quantify and locate areas experiencing changes in vegetation. Transect-level monitoring quantifies stem density, species composition (focusing in riparian indicator species), and beaver browse.
	Remotely sensed LiDar	Continuous throughout study area	Contracted flights in 2005 and 2013. Additional flights following Stage II restoration	Metrics expressing changes to channel and floodplain topography
	Remotely sensed aerial imagery	Continuous throughout study area	Contracted flights in 2005 and 2013. Additional flights following Stage II restoration	Metrics of riparian vegetation extent and density, changes to channel planform
	Photo points	3 to 5 fixed photo points in each control, treatment, and proposed treatment reaches	Repeat photos 2 times per year	Documentation of restoration effects to channel morphology and riparian vegetation
	Beaver dam distribution	Continuous throughout study area	Once per year during fall	Natural dam density, abundance, distribution, and persistence
	Rapid assessments of BDA structures and structure complexes	All existing BDA structures and complexes	Once per year during mid-summer	Variety of metrics describing BDA and complex function and potential to cause harm

Restoration Monitoring: Beaver Colony Establishment in Treatment Reaches

ISEMP personnel have conducted annual surveys since 2007 to document the distribution and abundance of natural beaver dams throughout the Bridge Creek IMW study area. These data build on an additional 20 years of beaver dam distribution data collected throughout the study area by the Bureau of Land Management that dates back to 1988 (Demmer and Beschta 2008). This time series of beaver dam distributions allows a unique opportunity to test whether the restoration approach developed for Bridge Creek is capable of altering channel and riparian characteristics in a manner that encourages establishment of stable beaver colonies.

We conduct beaver dam distribution surveys during late fall when beavers have finished building and repairing dams that may have breached during spring high flows. During the surveys the location of all actively maintained natural beaver dams and any BDAs that are actively being maintained and modified by beaver are recorded (Figure 7). Annual counts of beaver dam abundance are divided by monitoring reach length to calculate beaver dam density (dams/km) as a proximate measure of beaver abundance and colony establishment and persistence. Evaluation of the time series of beaver dam density throughout the treatment reaches demonstrates the efficacy of the BDA restoration approach in improving conditions and encouraging beaver colony establishment (Figure 8).

Figure 7. Example of beaver dam distribution and BDA structure utilization surveys showing BDA structures that are and are not being maintained by beaver as well as intact and failed natural beaver dams.

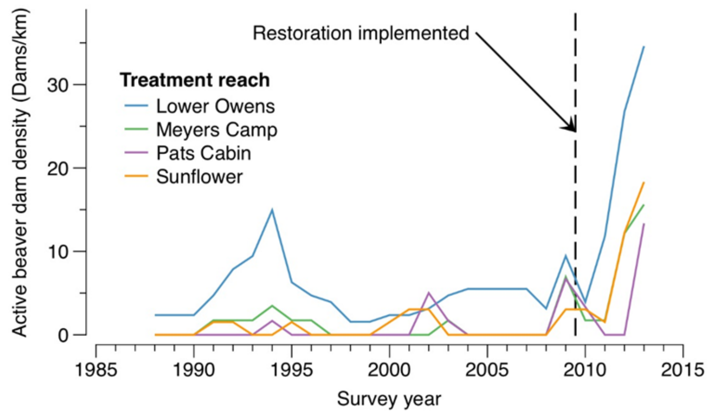


Figure 8. *Density (dams/km) of active beaver dams and/or BDA structures maintained by beaver within treatment reaches in the Bridge Creek IMW from 1988 – 2013.*

As of fall 2013 the density of actively maintained beaver dams and BDA structures are at the highest level observed in the 30-year time series of beaver dam distribution surveys for Bridge Creek. Although the treatment reaches have previously exhibited low densities of beaver dam building activity during distinct time periods, following installation of BDA structures dam densities have increased by several fold (Figure 8). For example, in the Lower Owens treatment reach beaver dam densities as high as 15 dams/km had previously been observed in the mid-nineties. However, following restoration dam densities within the reach have reached nearly 40 dams/km, a density of natural beaver dams that exceeds those observed within natural (untreated) dam complexes on Bridge Creek. Among all 4 of the treatment reaches dam densities did not immediately increase following restoration implementation. This delay in beaver utilization suggests that an initial year or two of channel aggradation may have needed to take place allowing high winter and spring flows to access adjacent terraces, thereby reducing the stream power and lowering the failure rate of natural beaver dams. Further, this pattern may also suggest that the treatment reaches now have established beaver colonies and the drastic increase in beaver dam abundance occurring in 2011, 2012, and 2013 may be due to reproduction by a stable colony.

Channel Aggradation and Geomorphic Change Detection

Topographic surveys are conducted annually in late fall using an rtkGPS (CHaMP 2015), beginning in 2009 shortly after the initial installation of BDA structures. Approximately 30 sites (~200 m of channel) are surveyed throughout treatment and control reaches according to a 2 year rotating panel design. Each survey is post-processed, subjected to a rigorous QA/QC procedure, and an orthogonal 10 cm resolution digital elevation models (DEM, Figure 9) is derived (CHaMP 2015). Geomorphic Change Detection 5.0 (GCD) software (http://gcd.joewheaton.org) is used to difference sequential DEMs and conduct a spatially-variable uncertainty analysis to robustly distinguish real changes from noise (Wheaton *et al.* 2010) and creating a DEM of Difference (DoD).

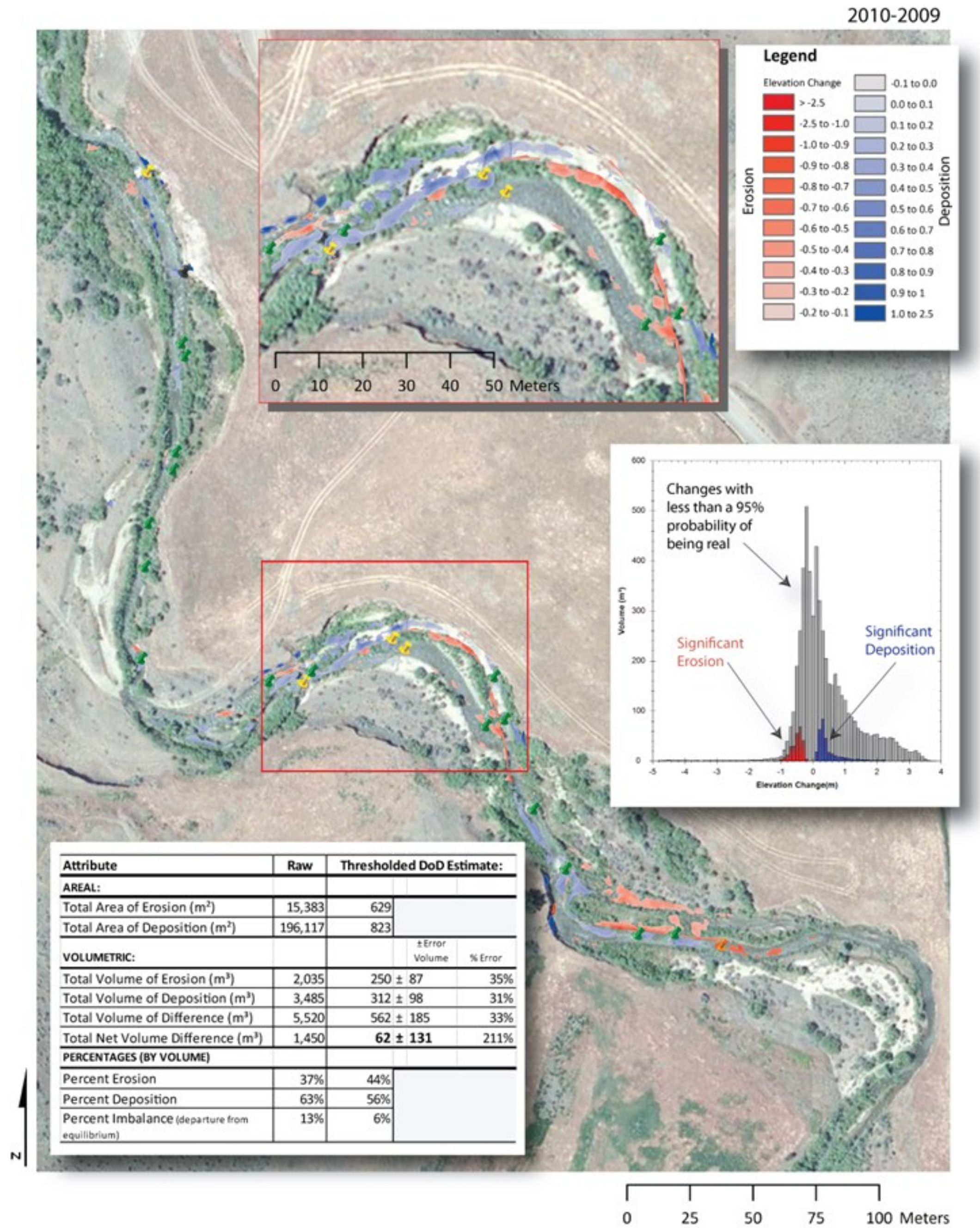


Figure 9. Using a digital elevation model (DEM) and geomorphic change detection (GCD) to construct a DEM of difference (DoD) at the Pats Cabin treatment reach. Map shows only those changes with a 95% or greater probability of being real, whereas the elevation change distribution and table show results of both statistically significant changes and those that cannot be distinguished from noise.

As hypothesized, results of GCD analysis from treatment and control reaches show evidence of strong net aggradation following treatment installation (Figure 10). In the initial years following restoration, both the average (among sites) and total change in sediment deposition is greater than the change in sediment erosion observed in the GCD analysis. In sharp contrast, control reaches on Bridge Creek continue to show greater sediment outputs than inputs, suggesting the BDA restoration structures are functioning to meet a primary restoration objective of increasing sediment storage and promoting channel aggradation.

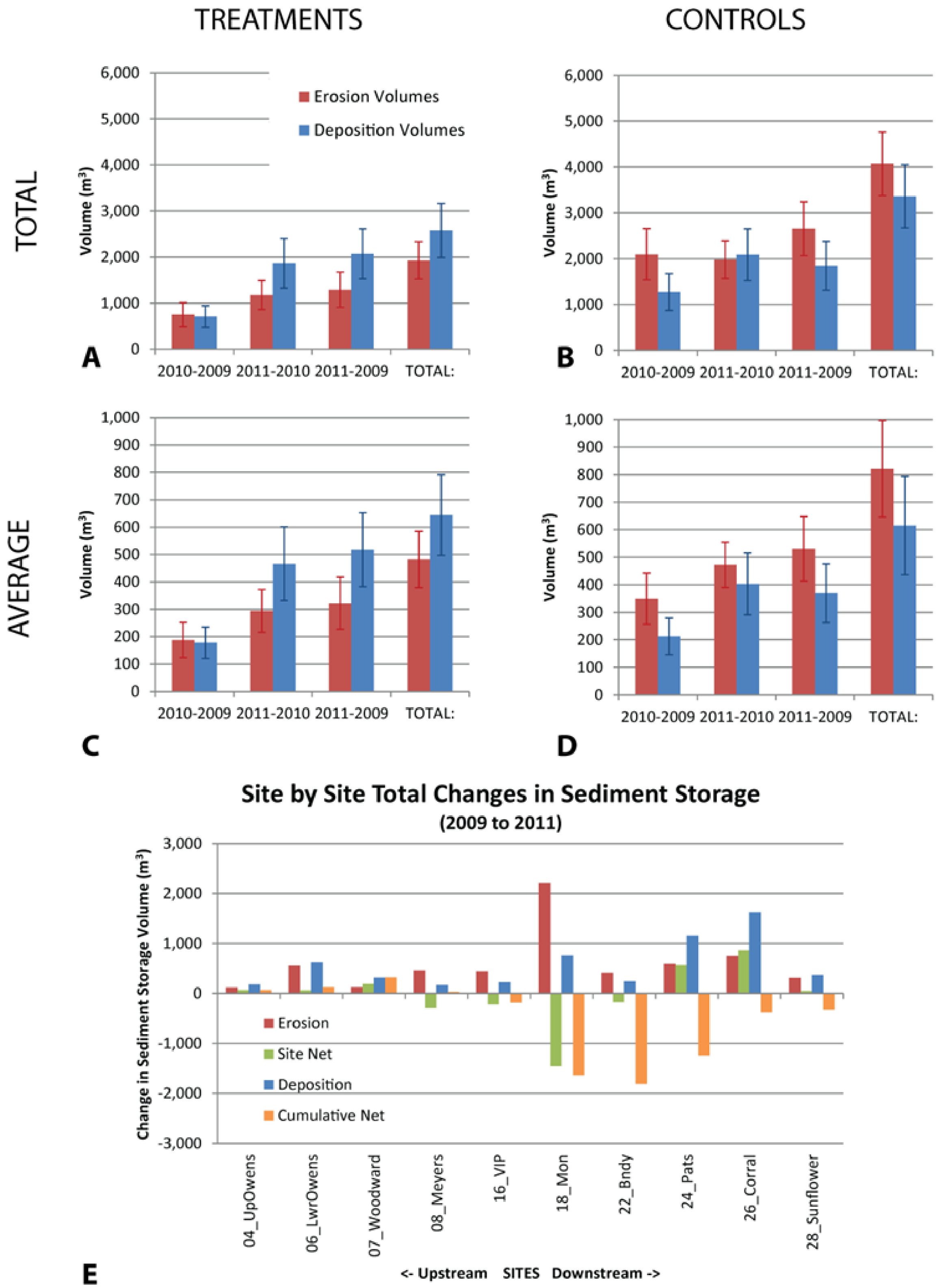


Figure 10. Geomorphic change at treatment and control sites from 2009 to 2011 in the Bridge Creek Intensively Monitored Watershed. All volumes reported are those with a 95% or greater probability of being real (error bars represent +/- one standard deviation of propagated error volumes).

In some treatment reaches, minor breaches and even major blowouts of individual beaver dams and/or BDA structures has occurred; however, there is little evidence of systematic breaching or blowouts of these dams leading to rapid channel degradation. Since BDA structures are installed in a series as a complex, minor dam failures generally result in sediment being transported and stored in downstream structures. Increased deposition observed in the treatment reaches has led to greater floodplain connectivity, habitat complexity, and improvements in overall system function associated with channel aggradation and incision recovery.

Hydrologic Connectivity and Groundwater Elevation

We have also hypothesized that water table elevation will increase as a result of the sedimentation and bed aggradation generated by BDAs (Pollock *et al.* 2012). Beaver dams may directly influence water table elevations by increasing inflow into groundwater aquifers from the stream channel via bank seepage or overbank inundation. Water impounded behind beaver dams expands the wetted area of the stream and the area of flooded soils; by decreasing current velocities dams increase sediment and organic matter retention and promote aggradation (Naiman *et al.* 1988). As sedimentation and aggradation occur, the spatial and temporal extent of overbank inundation should increase and bank storage should also increase as long as aquifers are available to accept recharge.

In 2006 we installed two groundwater monitoring well fields at a treatment and control site to monitor changes in groundwater elevation before and after restoration. Water depth and temperature data are collected in each well using HOBO Water Level Loggers (Onset Computer Corp., model U20-001-01) set to record data in one or two hour intervals. Each well field also incorporated one logger to record atmospheric temperature and pressure. Data from each logger are downloaded once or twice a year (typically in the spring and fall), and adjusted for barometric pressure using HOBOware software and data collected by the barometric logger.

Overall, water table elevations increased at both the treatment and control sites over time (Figure 11) but there is strong evidence of an additional upward trend in water table elevations at treated sites, even after accounting for the overall increase observed at both sites. Consistent with the increase in beaver dam abundance in the treatment reach in 2011 – 2014, an average annual increase of 0.17 m/year was observed in water table elevation in the treatment reach in post-restoration years. In addition, two wells that were dry during pre-treatment years contained water post-treatment. These results provide evidence

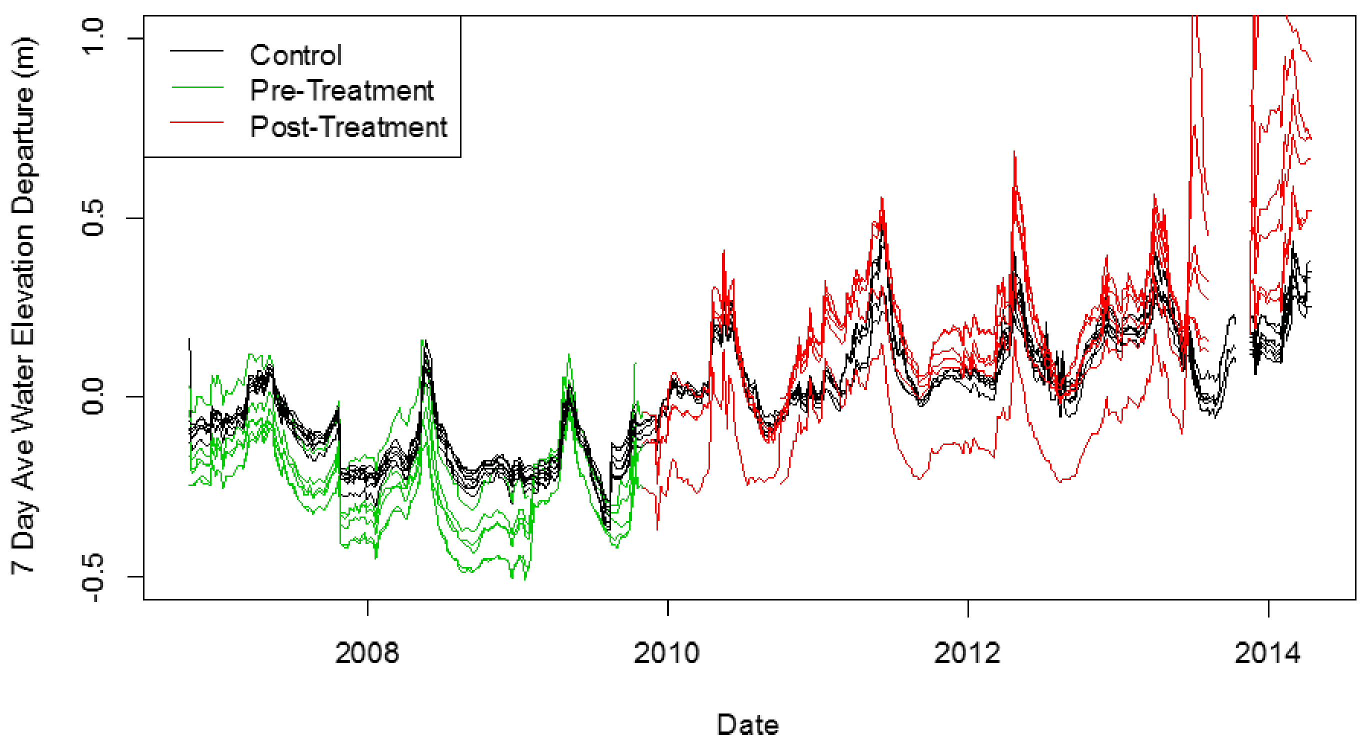


Figure 11. Water table elevation departure from average at the control wellfield (Upper Owens, black lines) and at the treatment wellfield (Lower Owens, colored lines). The green portion of the Lower Owens lines indicates water table elevations pre-treatment, while red indicates post-treatment. Each line represents data from one groundwater well. Seven-day mean water table elevations were calculated for each well, for each week in the time series. Water table elevation change was normalized by subtracting the average water table elevation value for each well over the entire time period analyzed from each weekly average. Thus, lines represent the departure from this overall average value.

that support the original hypothesis that sedimentation and aggradation generated by BDAs alters surface water patterns in such a way as to increase water table elevations in restored areas. An increase in water table elevation is expected to promote expansion of riparian vegetation and floodplain resources that provide important functions to salmonids, such as shading and surface water temperature regulation.

Riparian Vegetation: Pilot Study

ISEMP personnel completed a pilot study in 2013 to explore relationships between beaver activity and the composition and structure of riparian vegetation surrounding Bridge Creek. The study was designed to (1) examine changes in vegetation composition, stem density, and beaver browse with distance from water, and (2) to determine the extent to which vegetation community structure and composition differs between sites and transects influenced by beaver activity and those not influenced by beaver activity. A third goal of the study was to evaluate the field methods used and their effectiveness in collecting relevant data to inform future studies.

Surveys were conducted at four sites along Bridge Creek, including two control sites and two treatment sites. Treatments were installed in 2009, and natural beaver activity and beaver dams were present in all four sites in 2013. Vegetation at each site was assessed along belt transects that were 2 m wide and 10 m long, and which began near the edge of the stream. Additional data were also collected within four, 1 m^2 quadrats placed at 0, 3, 6, and 9 m along each belt transect. Data collected within each transect included both physical (e.g., distance from floodplain, presence of sediment deposition) and biological information (e.g., percent cover, number of stems, stem diameter).

Preliminary analyses of the 2013 pilot data are ongoing, but initial findings suggest that treatment sites with a high proportion of low floodplain host a vegetative community with a large proportion of willow and high stem densities closer to the stream (Figure 12). There also appears to be a strong relationship between the number of stems browsed by beaver and distance from the stream (Figure 13). Although this research is preliminary the information will be used to guide future restoration projects that hope to utilize beavers for channel and riparian rehabilitation, and will be used to answer questions related to the foraging requirements needed to sustain stable colonies.

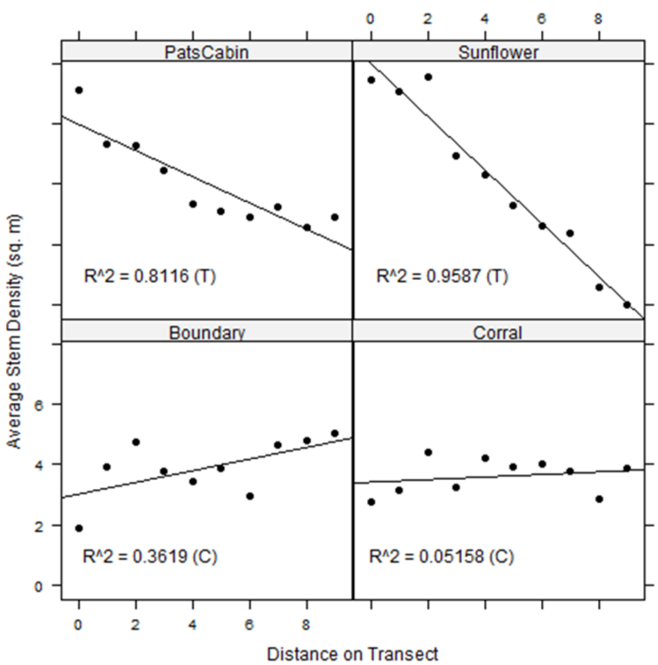


Figure 12. Average stem density of willow per square meter at a given transect distance. Distances are binned into one-meter sections, and values closer to zero are nearer to the edge of the stream. The number of stems in each distance bin in each transect at a site were summed, then divided by 2 to get the overall stem density at each distance (each distance bin was 2 m^2). Stem densities in each bin category were then summed across all transects at a site. Finally, the summed density in each distance bin was divided by the total number of distance bins at each site. Simple linear regression lines and R^2 values are shown for each site. T = treatment, C = control.

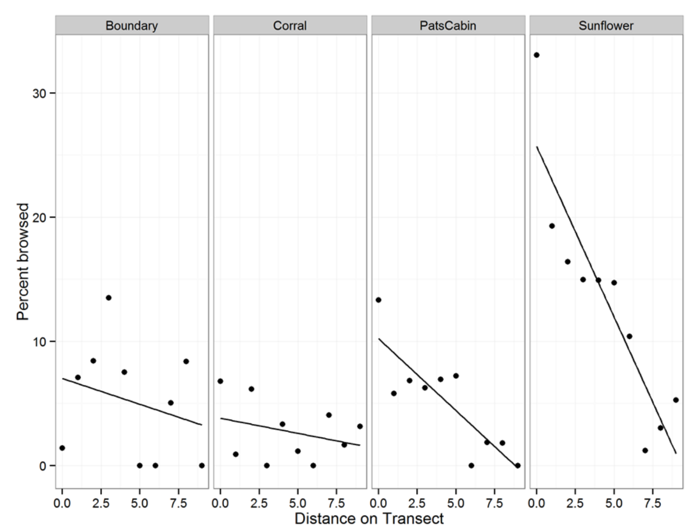


Figure 13. Percentage of all stems browsed by beaver at a given transect distance. Distances closer to zero are nearer to the edge of the stream. Percentages were calculated as the total number of browsed stems in a given distance bin across all transects, divided by the total number of all stems in that distance bin, multiplied by 100. Simple linear regression lines are shown for each site. R^2 values were 0.07 at Boundary, 0.09 at Corral, 0.72 at Pat's Cabin, and 0.81 at Sunflower.

Stream Temperature

We began temperature monitoring in 2006 and currently maintain 24 temperature loggers on approximately 27 km of the mainstem of Bridge Creek (Figure 14). The loggers are dispersed so as to ensure continuous monitoring at the upstream and downstream ends of all treatment and permanent control monitoring reaches. Loggers are also deployed on Gable Creek, Bear Creek, and on Murderers Creek which acts as a control watershed for the Bridge Creek IMW.

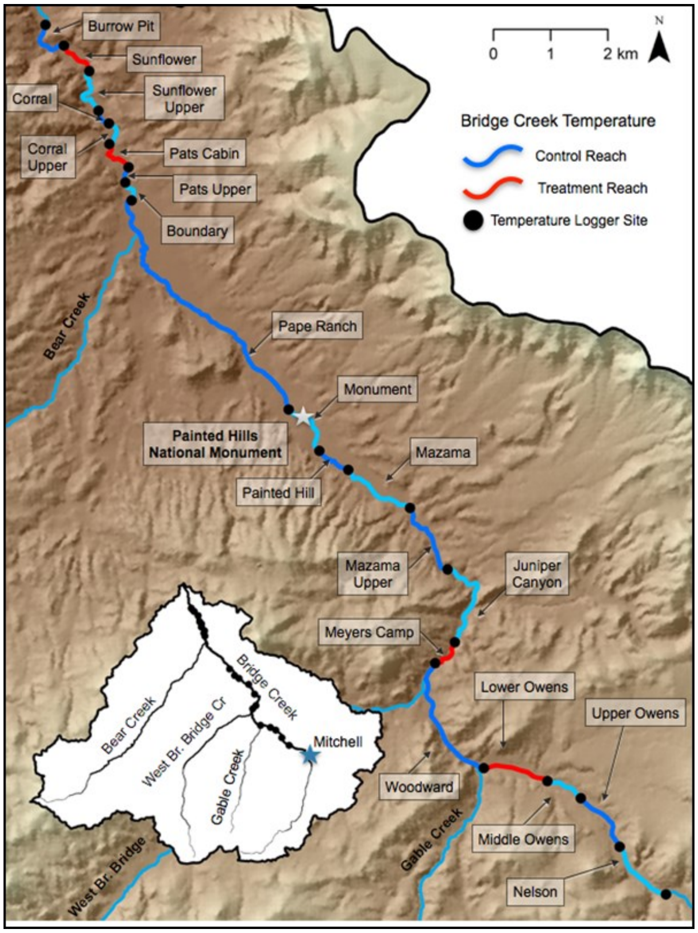


Figure 14. Distribution of temperature monitoring sites and monitoring reaches on the mainstem of Bridge Creek.

We performed a BACI comparison of the data by differencing the daily mean and daily maximum temperature measured at the downstream end of the two treatment reaches (Lower Owens and Corral) with that of the upstream control reach (Nelson Creek). The distribution of daily difference was then summarized for each month of the summer (Figure 15 and Figure 16). We found a reduction of maximum daily stream temperatures in heavily ponded sites relative to the control site with no beaver activity upstream of the temperature monitoring location.

Prior to 2009, and shortly after restoration implementation in 2010, Lower Owens exhibited maximum daily temperatures roughly 1 or 2 °C greater than Nelson Creek (Figure 15); however, as the abundance and extent of beaver ponds increased within the reach in 2012 - 2014, maximum daily temperatures shifted and on average became 1 to 2 °C cooler than Nelson Creek. A similar pattern is exhibited in the comparison that included the Corral reach. Average of maximum daily temperatures were reduced from roughly 4 to 6 °C greater than Nelson Creek to between roughly 1 to 3 °C greater in 2012, 2013, and 2014 (Figure 17).

It is noteworthy that beaver complexes seem to have little effect on stream temperature during June. This is likely due to higher surface discharge from runoff, so that the effects of groundwater exchange are diluted. In both comparisons reduction in maximum stream temperature were most apparent for July, August, and September when Bridge Creek is flowing at a low base flow discharge.

Another interesting pattern we observed through evaluation of average daily temperature values is that the response to increased abundance and extent of beaver dams was reduced or lacking. Closer evaluation using hourly temperature patterns both pre- and post-restoration reveals that the effect that beaver dams have on temperature manifests as a reduced range of temperature values for each day (Figure 17). Therefore, although the average daily temperature may not indicate a response to beaver ponds, maximum daily temperatures may be significantly reduced. In the case of Bridge Creek this has dramatic implications for juvenile steelhead habitat quality, as a reduction in maximum daily temperature by only a few degrees may prevent significant stress to fish populations during summer. High summer temperatures appear to play a major role in affecting the distribution of juvenile *O. mykiss* using Bridge Creek. Throughout all seasons juvenile *O. mykiss* abundances are much greater above the Pape Ranch where summer temperatures rarely exceed 24°C, or do so only for a short period each day. Further, during summer PIT-tagged juvenile steelhead are detected moving from the lower end of Bridge Creek into cooler reaches upstream of the Pape Ranch. This migration occurs in early to mid-July immediately following maximum daily stream temperatures exceeding 24°C.

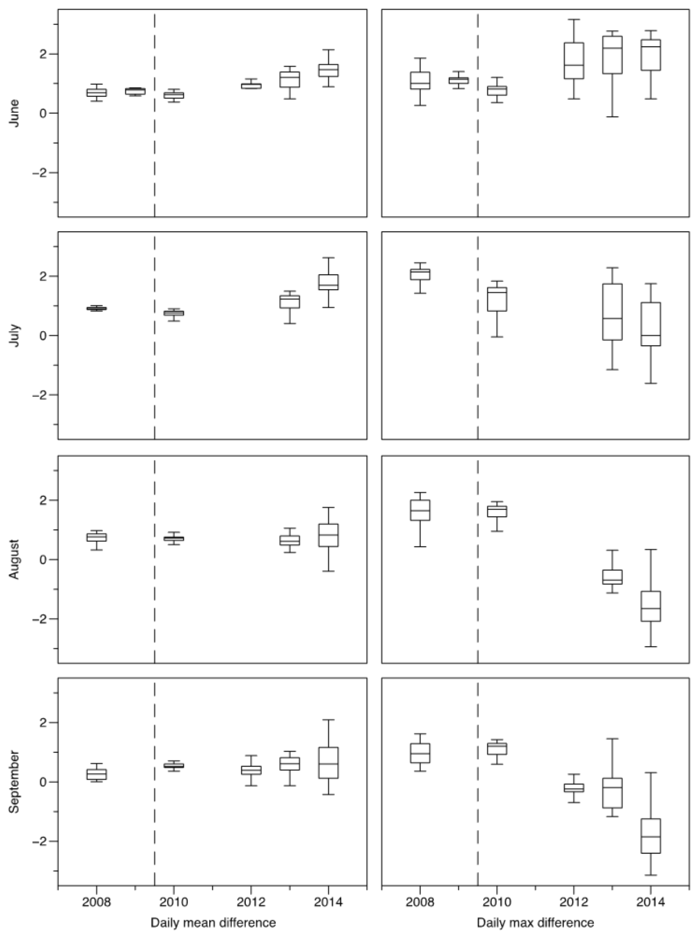


Figure 15. Distribution of pre- and post-treatment and control differences in daily mean and maximum temperatures for the Nelson Creek and Lower Owens Reaches.

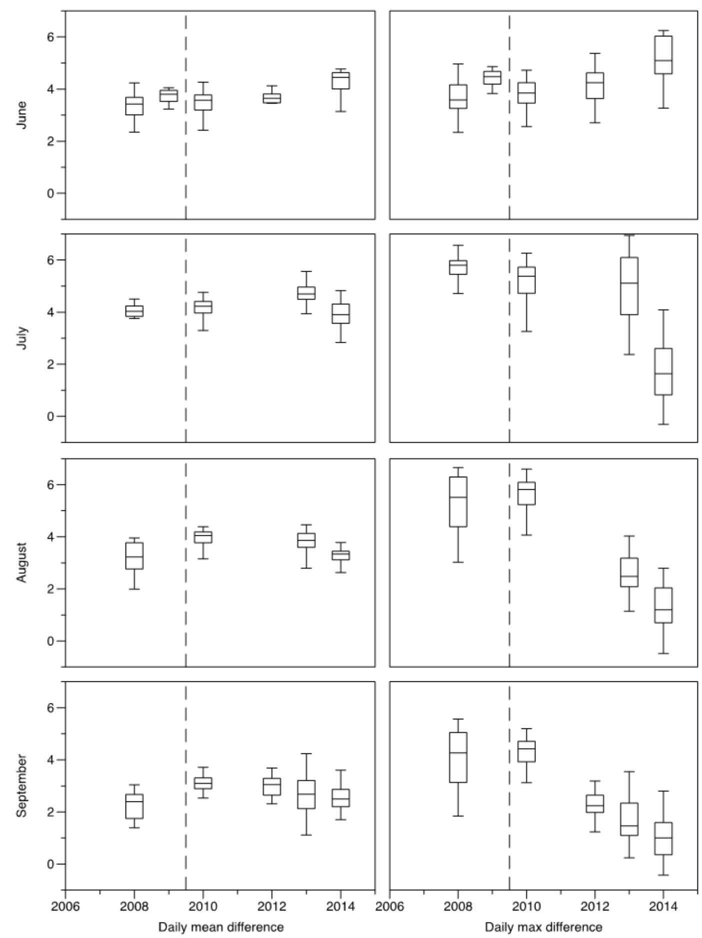


Figure 16. Distribution of pre- and post-treatment and control differences in daily mean and maximum temperatures for the Nelson Creek and Corral Reaches.

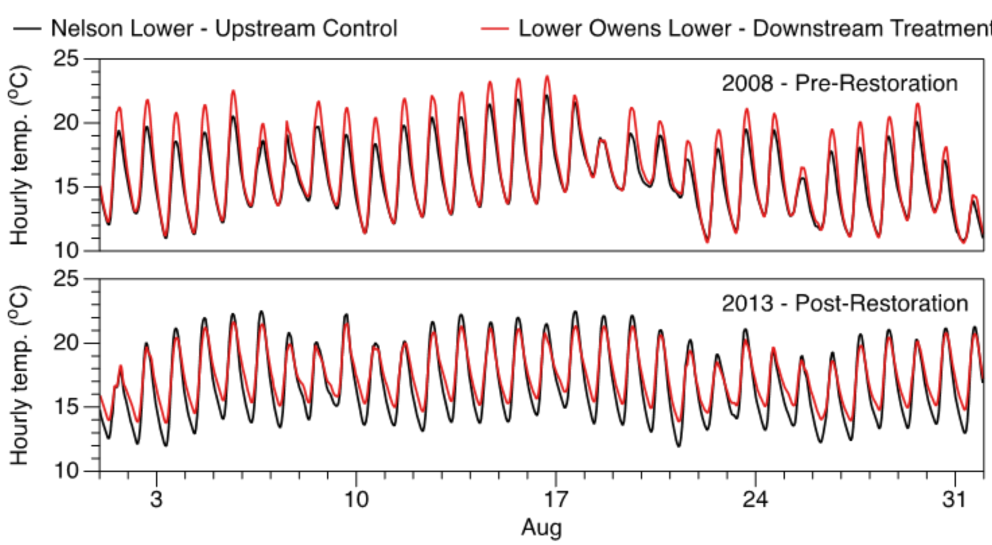


Figure 17. Comparison of hourly stream temperatures at the Nelson Creek control reach and downstream of the Lower Owens treatment reach before (2008) and after (2013) restoration implementation demonstrating a reduction in the range of daily temperature in reaches below beaver complexes.

Longitudinal Patterns of Temperature on Bridge Creek

Longitudinal patterns in stream temperature were summarized by differencing the daily mean and daily maximum temperature recorded at the downstream end of each monitoring reach from temperature recorded at the upstream end (see Figure 14 for monitoring reach names and locations). These daily differences measure the degree that each reach on Bridge Creek serves to increase, maintain, or decrease stream temperatures from upstream to downstream. Daily differences in mean and maximum temperature were then averaged for the summer months of July, August, and September for 2013 (Figure 18).

Similar to the BACI analyses presented above, longitudinal deviations are most significant for maximum daily temperature, and less so for mean daily temperature (Figure 18). Also, a higher percentage of stream reaches showed a decrease in maximum temperature during August and September. For example, in July of 2013 only 3 out of the 15 reaches exhibit decreasing maximum daily temperature and in August this number increased to roughly half of the monitoring reaches. This again highlights that moderation of stream temperature by hydrologic connectivity with the water table may be most significant at low flow conditions during late summer, precisely when temperatures are causing stress for juvenile salmonids.

Perhaps the most significant aspect of this summary of longitudinal temperature dynamics for the Bridge Creek IMW is that, despite extensive increases in wetted channel area, the reaches treated with BDA structures do not significantly increase summer temperatures relative to untreated sections of stream. Further, in many cases the reaches treated with BDA structures in 2009 are effectively moderating high summer stream temperatures that may be limiting the rearing potential of juvenile *O. mykiss*.

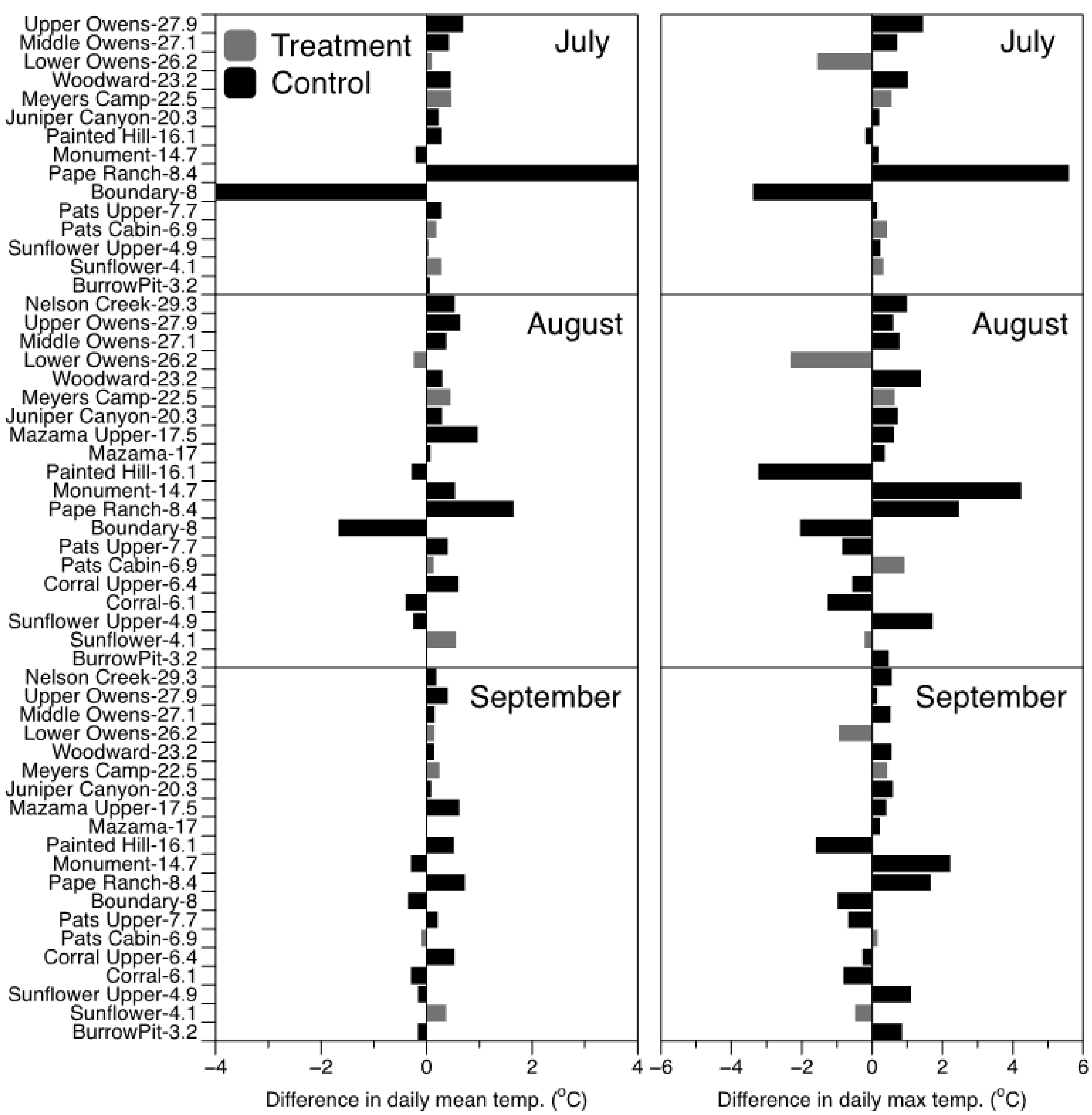


Figure 18. Mean longitudinal difference in mean and maximum daily maximum temperature for summer 2013.

Juvenile Steelhead Monitoring

ISEMP personnel began monitoring juvenile steelhead in 2007, spanning both pre- and post-restoration. Sites were established to represent treatment and control reaches at all scales within the overall experimental design, and are visited during spring (June), fall (September-October), and winter (December-February). This sampling schedule was established to capture seasonal variation, to avoid sampling during the high flow period of March-May, which is when steelhead spawning activity occurs, and to sample before juvenile steelhead out-migrate. Each site is sampled on two consecutive days in a sampling event. Juvenile salmonids are captured using an electrofisher, and untagged fish are implanted with a Passive Integrated Transponder tag (PIT tag). Mark and recapture of PIT-tagged steelhead is used to generate estimates of juvenile steelhead abundance, measure seasonal growth rates, and estimate steelhead survival and productivity.

While the multi-scaled experimental design utilized by the Bridge Creek IMW allows treatment and control reaches to be compared to one another, movement information indicates that juvenile steelhead utilize multiple sites on Bridge Creek during their rearing period. This movement suggests sites on Bridge Creek lack independence, and likely reduces the observed response between treatment and control reaches. An effective way to deal with this lack of independence among sites on Bridge Creek is to compare the average response of juvenile steelhead among the monitoring reaches on Bridge Creek to the external control reaches on Murderers Creek. Therefore the results presented here focus on comparing the difference in juvenile steelhead abundance, growth, survival, and productivity between treatment (Bridge Creek) and control (Murderers Creek) watersheds. Evaluating the difference between treatment and control watersheds also accounts for temporal variability in response metrics due to conditions not accounted for in the experimental design.

Juvenile Steelhead Abundance

To standardize abundance estimates among sites that differ in length abundance estimates are divided by site length in order to convert to juvenile steelhead density (no./100 m). With some exceptions, seasonal estimates of juvenile steelhead density on both Bridge Creek and Murderers have followed a similar pattern both before and after restoration implementation. However, following restoration the average difference in density between the two watersheds suggests that the restoration is having a positive effect on juvenile steelhead densities in Bridge Creek. Before restoration Bridge Creek exhibited juvenile steelhead density estimates that were lower than Murderers Creek by around 100 juveniles/100 m. Following restoration seasonal densities of steelhead on Bridge Creek have on average been greater than those observed on Murderers Creek (Figure 19). A particularly compelling aspect of this comparison is the low densities of steelhead observed in both Murderers and Bridge Creek during the summers of 2014, and the rapid increase observed in juvenile steelhead densities the following fall and winter in Bridge Creek. This pattern suggests that restoration actions in Bridge Creek have improved habitat conditions for juvenile steelhead and support a more resilient population.

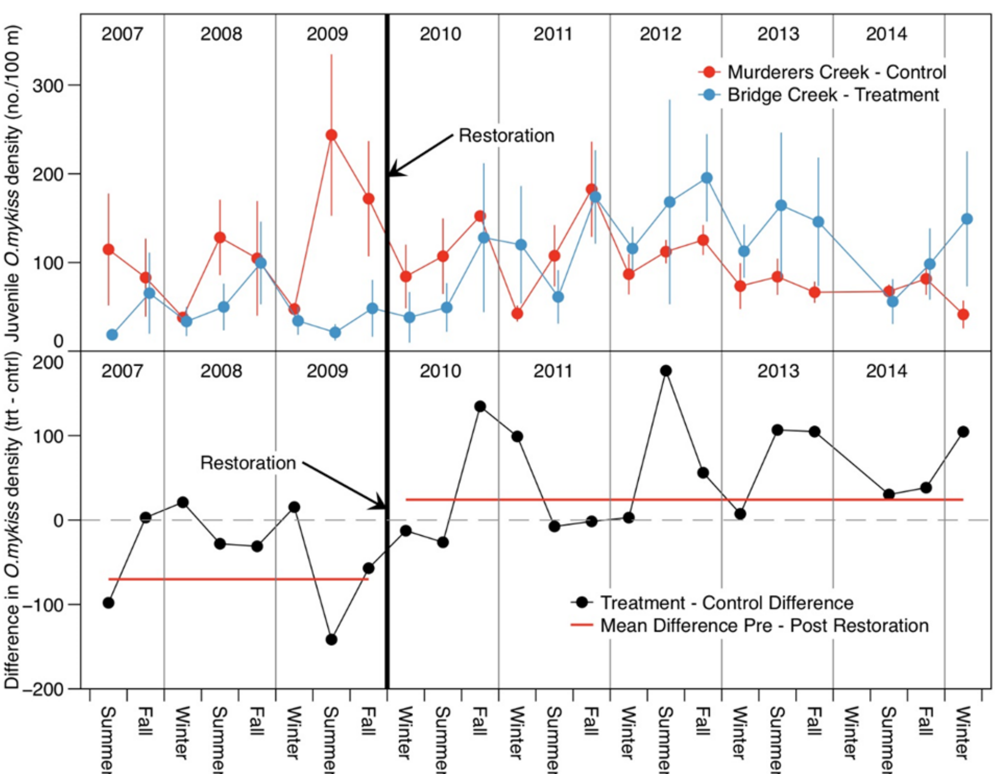


Figure 19. Seasonal density of juvenile O. mykiss pre- and post-restoration (top panel) on Bridge Creek (treatment) and Murderers Creek (control). Bottom panel depicts the seasonal difference in density (black line) and average difference pre- and post-restoration (red line) between treatment and control watersheds. Errors bars are one standard error.

Juvenile Steelhead Growth

Individual growth rates are standardized for fish size and the duration of the growth period (relative growth rate) by dividing the seasonal change in juvenile steelhead weight by fish weight and by the number of days in the growth period (i.e., grams of growth/gram of fish/day). Several patterns are apparent when comparing the growth rates between Murderers and Bridge Creek. First, growth rates fluctuate drastically by season, with highest growth occurring in each watershed during the winter growth period, which extends from February through June (Figure 20). Following restoration average juvenile growth rates on Bridge Creek show a slight decrease relative to Murderers Creek, possibly due to the higher densities of juvenile *O. mykiss* now present within Bridge Creek resulting in density dependent competition for food or other limited resources. Further, foraging in beaver ponds may be slightly less energy efficient for juvenile steelhead which would also explain the slight decrease in growth rates. This aspect of the response of juvenile steelhead highlights the need to monitor multiple response metrics such as abundance, survival, and productivity in order to fully understand the effects that restoration may have on fish populations.

Juvenile Survival

Juvenile steelhead survival is analyzed by developing encounter histories for each individually PIT tagged fish from active tagging and passive detections at PIAs which are modeled using the Barker model (Barker 1997, Barker *et al.* 2004) to produce estimates of survival in Program MARK (Cooch and White 2010). A comparison of seasonal survival pre- and post-restoration between Murderers and Bridge Creek lends further support that the restoration actions improve habitat conditions for *O. mykiss* populations (Figure 21). Prior to restoration, Murderers Creek exhibited a considerably higher probability of survival for juvenile *O. mykiss* than Bridge Creek; however, following restoration survival on Bridge Creek increased relative to Murderers Creek, and on average has almost an equal survival probability for juvenile steelhead. These finding are consistent with, and may explain, the increase in juvenile steelhead abundance that has been observed on Bridge Creek following restoration.

Juvenile Production

Production is perhaps the most integrated response of organisms to their environment and is an important metric for evaluating the effects of a restoration project. Here production is estimated as the product of the number of individuals in an area (density) times the growth of those individuals over time (growth) times the survival rate of the group over the time period.

Post-restoration monitoring has shown juvenile steelhead abundance and survival increased and growth slightly decreased relative to Murderers Creek; an evaluation of production puts these responses into context and reveals that while individual growth rates have decreased, increases in survival and abundance have resulted in an overall increase in the production of juvenile steelhead on Bridge Creek (Figure 22).

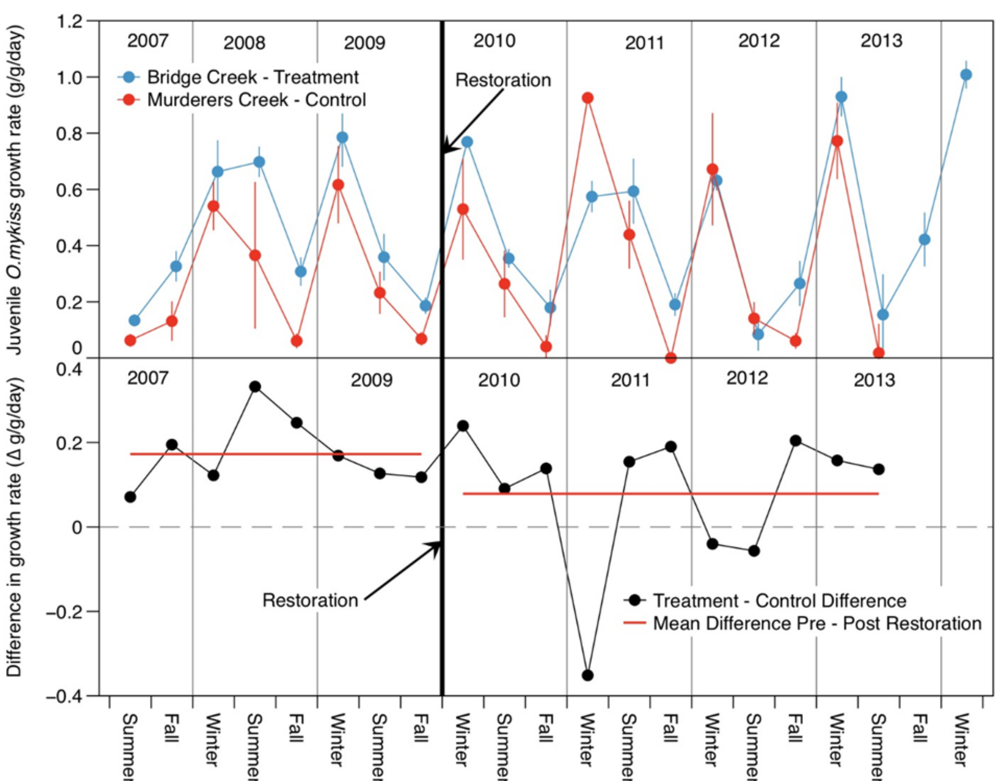


Figure 20. Seasonal relative growth rate (g/g/day) of juvenile O. mykiss pre- and post-restoration (top panel) on Bridge Creek (treatment) and Murderers Creek (control). Bottom panel depicts the seasonal difference in growth rate (black line) and average difference pre- and post-restoration (red line) between treatment and control watersheds. Errors bars are one standard error.

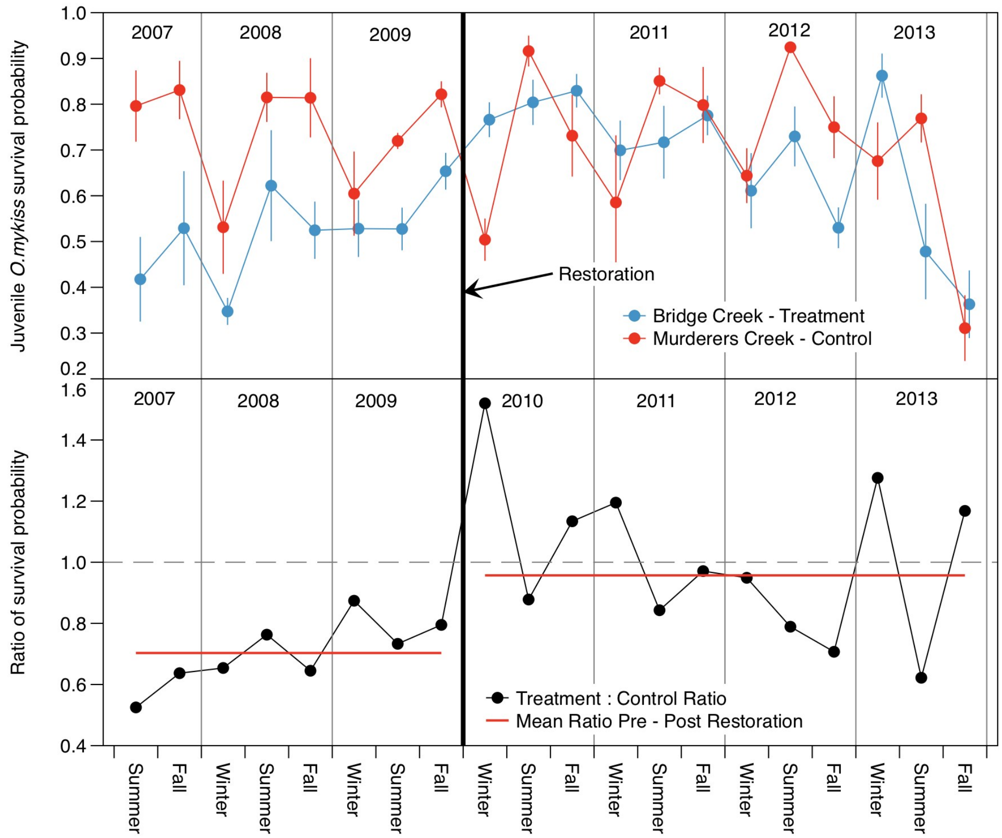


Figure 21. Seasonal probability of survival for juvenile O. mykiss pre- and post-restoration (top panel) on Bridge Creek (treatment) and Murderers Creek (control). Bottom panel depicts the seasonal difference in survival probability (black line) and average difference pre- and post-restoration (red line) between treatment and control watersheds. Errors bars are one standard error.

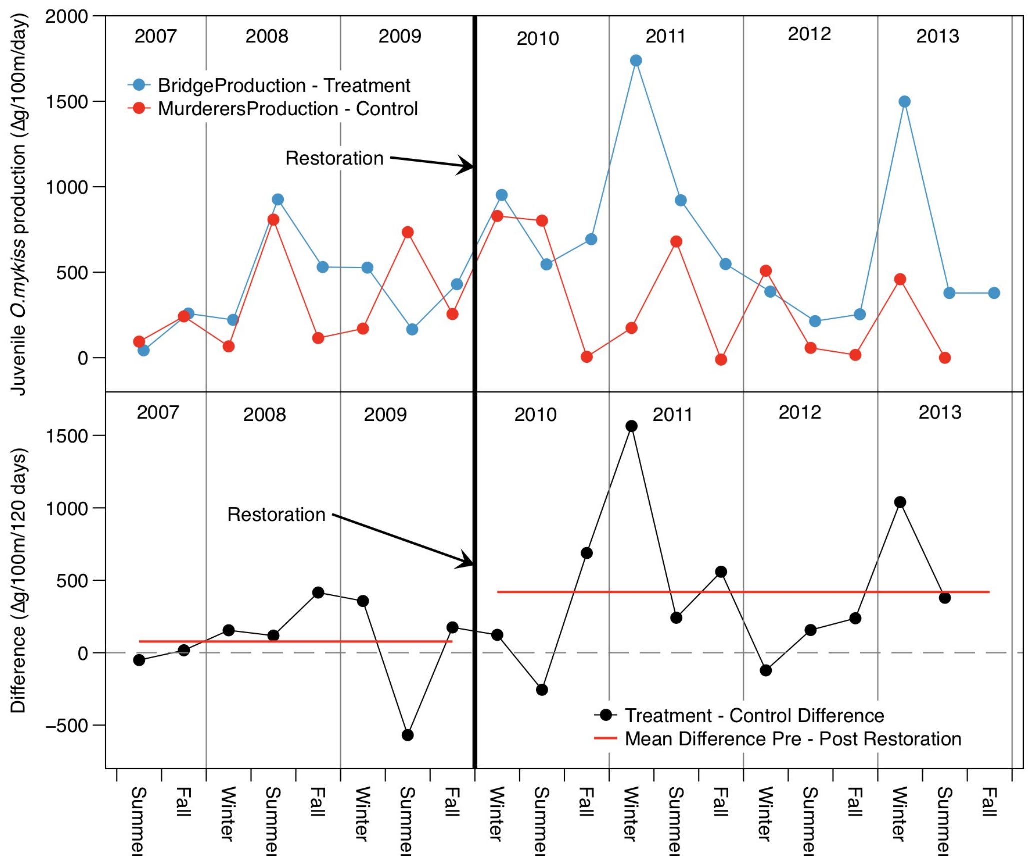


Figure 22. Seasonal production of juvenile O. mykiss pre- and post-restoration (top panel) on Bridge Creek (treatment) and Murderers Creek (control). Bottom panel depicts the seasonal difference in production (black line) and average difference pre- and post-restoration (red line) between treatment and control watersheds. Errors bars are one standard error.

Adult Steelhead Monitoring

Due to the local geology, the Bridge Creek IMW study area consistently flows with a high suspended sediment load which decreases water clarity and renders spawning ground surveys ineffective. Monitoring information related to adult steelhead using Bridge Creek is almost entirely gained through operation of an adult steelhead weir at the mouth of Bridge Creek, and operation of PIAs throughout the study area (Figure 23).

We began operating the adult steelhead weir on Bridge Creek during the spring of 2009. The trap is located approximately 1 km upstream of the mouth of Bridge Creek, and is generally installed at the beginning of March and removed by mid-May each year.

Figure 23. Adult steelhead trap near the mouth of Bridge Creek (left) and a tagging station (right).

During operation we check each trap box morning and evening. We move captured adult steelhead to a streamside anesthesia bath (MS-222), and once sedated record their length, origin, and sex and scan for a PIT-tag. We PIT tag untagged steelhead in their dorsal sinus. Following data collection and tagging fish recover from anesthesia in a freshwater bath prior to being released back to the stream in their direction of travel.

Adult Abundance

The weirs on Bridge Creek are not 100% efficient so we use a mark-recapture model to produce an annual estimate of adult steelhead abundance on Bridge Creek. We treat adult steelhead captured and tagged in the upstream trap as the mark (M) event of the mark-recapture estimate, while captures (C) and recaptures (R) are accounted for as steelhead are captured in the downstream trap. Adult steelhead population estimates produced using mark-recapture at the trap have ranged from as low as 455 in 2014 to as high as 833 in 2010 (Table 2).

Downstream migrating steelhead remain extremely difficult to capture in the downstream trap, which greatly affects the precision of adult abundance estimates. ISEMP personnel will be constructing an improved downstream trap box in 2015 which should increase the number of downstream captures and recaptures and help to improve the precision of adult steelhead abundance estimates.

Table 2. Annual mark-recapture population estimate (N and standard error) for wild origin adult steelhead estimated as fish captured in the upstream trap (Marks), and captured (Captures) and recaptured (Recaptures) in the downstream trap. No adult steelhead were recaptured during 2012 prohibiting a population estimate for that year.

Year	Marks	Captures	Recaptures	N (SE)
2009	126	84	19	540 (102)
2010	118	20	2	833 (385)
2011	39	101	6	582 (198)
2012	41	4	-	-
2013	106	9	1	535 (276)
2014	64	41	3	455 (159)

Evaluating if Beaver Dams Effect Adult Spawning Distribution

Through the combination of PIT-tagging adult steelhead as they enter Bridge Creek and the placement of PIAs, we are able to monitor the distribution of adult spawning steelhead throughout the study area to see if BDAs or natural beaver dams are acting as barriers to upstream migration. The distribution of adult spawning steelhead is estimated for each run year as the percent of the total steelhead detected above each PIA (see Figure 6 for PIA locations). A significant deviation in the distribution of spawning adults would indicate that BDA structures or natural beaver dams are creating barriers to upstream migration. Restoration structures were first installed on Bridge Creek in the fall of 2009, so the spring of 2009 is the only pre-treatment year of spawning distribution information. With the exception of 2013, the upstream spawning distribution of adult steelhead has remained relatively similar to those observed in the pre-treatment year of 2009 (Table 3, Figure 24). These observations suggest that the restoration treatments consisting of the installation of channel spanning BDA structures on Bridge Creek have not significantly reduced the upstream distribution of adult steelhead by acting as barriers to migration.

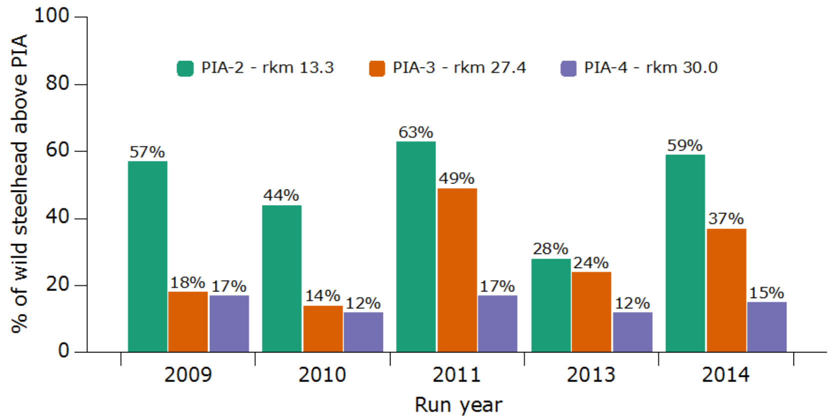


Figure 24. Annual spawning distribution of wild origin adult steelhead within the mainstem of Bridge Creek as percent of total tagged adults detected above each passive instream antenna (PIA).

On average 50% of the wild adult steelhead spawning in Bridge Creek will do so in the lower 13 km of Bridge Creek. This pattern highlights the need for continued restoration in the downstream reaches of Bridge Creek where two of the 2015 proposed treatment reaches are located. The restoration treatments should serve to improve the abundance and distribution of essential spawning gravels, as well as provide flow and temperature refugia for rearing juvenile steelhead. Also worth considering is the contrast between hatchery and wild origin steelhead spawning distributions: during most years the majority of hatchery steelhead (~ 80%) appear to spawn in the first 13 km of Bridge Creek below PIA-2. Wild origin steelhead appear to have a greater distribution throughout the watershed with greater than 50% detected at or above PIA-2 in most years.

Table 3. Number and percent of the total PIT-tagged adult wild and hatchery steelhead detected above each successive Passive Instream Array on Bridge Creek.

Year	Total adults above trap		Adults above PIAs				
	Hatchery	Wild	PIA	Hatchery	Wild	% Hatchery	% Wild
2009	16	110	2	3	63	19%	57%
			3	3	44	0%	18%
			4	0	19	0%	17%
2010	52	66	2	16	29	31%	44%
			3	14	21	4%	14%
			4	2	8	4%	12%
2011	4	35	2	0	22	0%	63%
			3	0	16	0%	49%
			4	0	6	0%	17%
2013	16	90	2	4	25	25%	28%
			3	4	14	13%	24%
			4	0	11	0%	12%
2014	4	59	2	0	35	0%	59%
			3	0	26	0%	37%
			4	0	9	0%	15%

Stage II Restoration Plan

The Stage II restoration actions propose installation of a series of BDAs throughout 4 additional reaches on the lower 30 km of Bridge Creek (Figure 25). Theses reaches were prioritized through review and analysis of extensive field surveys of fish and fish habitat and remotely sensed information. Additionally, in May 2014 ISEMP personnel and collaborators visited each proposed reach to specify the design and expected outcomes of each structure and structure complex. Although not included in this document, detailed specification information and videos for each structure and structure complex are available upon request from Bridge IMW personnel.

The Phase 2 reaches are well suited to the installation of constriction dams and complexes designed to accelerate incision trench widening (Table 4), and although the use of these designs is somewhat novel within Bridge Creek, Bridge Creek IMW personnel have applied these restoration techniques at similar projects as part of the Asotin IMW restoration project. The expected channel responses have been well documented and highly successful in achieving specific restoration objectives (see Wheaton *et al.* 2012 for a review).

A fourth treatment will be applied to Bear Creek, the largest tributary and an important steelhead spawning and rearing stream entering the lower valley on Bridge Creek. Bear Creek rarely maintains flows throughout the summer over a substantial proportion of its length, and the restoration treatments have been designed to encourage hydrologic reconnection of surface water with subsurface flows (Table 4).

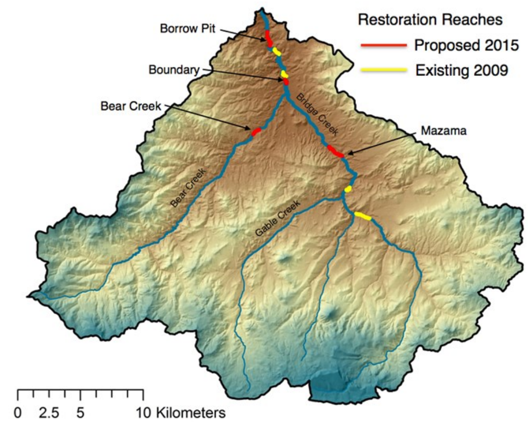


Figure 25. The location of the four existing and three proposed BDA restoration reaches in the Bridge Creek Watershed.

Table 4. Description of current conditions and restoration design and objectives for the proposed 2015 restoration reaches on Bridge Creek.

Proposed Reach	Current Conditions	Design and Objectives
Borrow Pit Reach Length: 1.3 km Structure Complexes: 11 Primary Dams: 10 Secondary Dams: 7 Constriction Dams: 27 Reinforced Existing Dams: 0	Incised phase 2 channel lacking inset floodplain development. Low habitat complexity, steep gradient, and low sinuosity. Little to no beaver activity	Complexes of constriction dams to widen incision trench, induce meanders, increase habitat complexity, and mobilize material for aggradation. Secondary dams to begin bed aggradation and improve conditions for beaver complex establishment
Mazama Reach Length: 1.6 km Structure Complexes: 19 Primary Dams: 20 Secondary Dams: 27 Constriction Dams: 21 Reinforced Existing Dams: 0	Mix of incised phase 2 and phase 3 channel with limited inset floodplain development. Moderately sinuous channel with low habitat complexity. Little to no beaver dam building activity	Complexes of constriction dams to widen incision trench, induce meanders, increase habitat complexity, and mobilize material for aggradation. Primary and secondary dams to begin bed aggradation and improve conditions for beaver complex establishment
Boundary Reach Length: 0.5 km Structure Complexes: 7 Primary Dams: 7 Constriction Dams: 0 Secondary Dams: 8 Reinforced Existing Dams: 2	Phase 3 channel with low floodplain connectivity and moderate habitat complexity. Moderate beaver dam building activity with high rate of dam failure.	Reinforced existing, primary, and secondary dams to disperse high flows reducing dam failure. Increase pond extent to encourage establishment of stable beaver complex
Bear Creek Reach Length: 0.7 km Structure Complexes: 3 Primary Dams: 7 Constriction Dams: 0 Secondary Dams: 2 Reinforced Existing Dams: 0	Intermittent and/or subsurface flow during low flow periods. Low habitat complexity and reduced riparian canopy	Primary and secondary dams designed to increase water storage, create pond habitat, and disperse flow to encourage groundwater exchange.

Entiat River Intensively Monitored Watershed

ISEMP's Entiat River IMW project is a long-term watershed-scale experiment to monitor restoration of instream fish habitat on the lower 26 miles of the mainstem Entiat River in the Upper Columbia subbasin in Washington (Figure 26), and the response of spring Chinook and steelhead (Nelle 2011). This IMW represents an engineered approach to restoration in which treatments are designed to increase instream complexity through the addition of wood or rock and the removal or breaching of levees to activate the floodplain. The Entiat River IMW began with pre-project monitoring in 2010, and the first of four rounds of habitat restoration actions were implemented in 2012. Effectiveness monitoring is ongoing, and to date some positive responses in instream habitat have been documented, and there are indications that spring Chinook and steelhead populations may be responding to restoration actions. A third round of restoration actions are scheduled for implementation in 2016 and 2017, and the final round of actions is targeted for 2020. At this point in the project, the Entiat River IMW is beginning to provide valuable information that demonstrates the potential for an engineered approach to stream restoration to improve salmonid population viability, and serves as a model for application of engineered restoration approaches within other impaired watersheds.

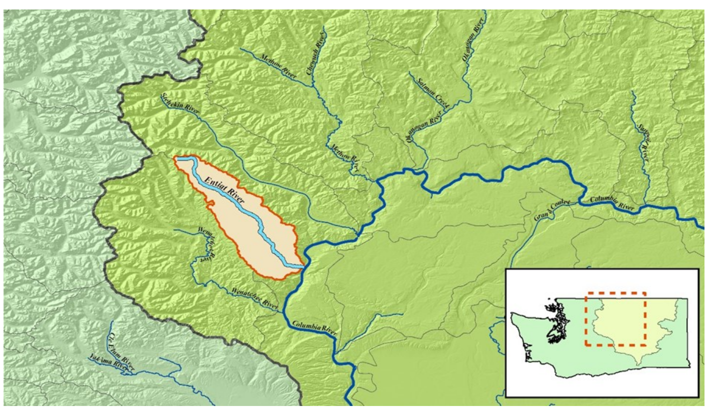


Figure 26. Location of the Entiat River Intensively Monitored Watershed in the Upper Columbia subbasin, Washington.

The Entiat River drains approximately 1,100 km^2 (425 mi^2) of the eastern slope of the central Cascade Mountains in Washington State, and is a tributary to the Columbia River (Figure 26). Wildfire, flooding, mass soil and debris movement, and land use have been the primary historic disturbance processes in the Entiat watershed. Land use has included floodplain and river channel modification projects and structures such as channel straightening/widening and diking, streamside vegetation disturbance, grazing, roading, agriculture, timber harvesting, transport of logs within the river channel, dams for log storage ponds and hydropower generation, residential development, and recreation (CCCD 2004). These land use actions have resulted in simplified channel conditions and created limiting factors for Upper Columbia spring Chinook, *O. mykiss*, and bull trout (*Salvelinus confluentus*). The Upper Columbia Spring Chinook and Steelhead Recovery Plan (UCSRB 2007) has determined that these populations have a high risk of extinction (more than 25% in 100 years), low abundance and productivity, and are at risk for diversity and spatial structure. Efforts to restore salmon and steelhead habitat in the Entiat River are guided by the Recovery Plan (UCSRB 2007), which recommends the use of instream structures (such as boulders and large woody debris) as immediate, short-term actions to increase habitat diversity in the Entiat River subbasin along with associated effectiveness monitoring which has been largely occurring under ISEMP[1].

The Entiat IMW is designed to answer five hypotheses in a timely and cost-efficient manner. The Entiat River subbasin is a relatively homogenous physical and biological environment and populations are likely suppressed by the lack of variability in habitat features as much as they are by the magnitude of any particular limiting factor. Therefore, our hypotheses test both the magnitude of the treatment effect as well as the change in variability in certain indicators resulting from the treatment effect.

H1(a): The implementation of instream channel modifications will, at the reach, valley segment and watershed level a) significantly improve the magnitude of physical habitat and macroinvertebrate indicators favorable to spring Chinook salmon and steelhead production and b) will increase the variability of these indicators.

H1(b): The implementation of side channel restoration projects will, at the reach, valley segment and watershed level a) improve the magnitude of physical habitat and macroinvertebrate indicators favorable to spring Chinook salmon and steelhead production and b) will increase the variability of these indicators.

H2: The combined effect of improvements in physical habitat and macroinvertebrate indicators will, at the reach, valley segment and watershed scale, a) increase the density, growth and survival of juvenile spring Chinook salmon and steelhead within the Entiat River subbasin, b) increase the number of emigrants of spring Chinook salmon and steelhead leaving the Entiat subbasin, and c) will increase the within-basin productivity of spring Chinook salmon and steelhead from the Entiat subbasin as measured by the number of emigrants-per-redd.

H3: Improvements in physical habitat and macroinvertebrate indicators will lead to life-stage specific changes in survival and growth rates in spring Chinook and steelhead during rearing, over-wintering, and emigration periods.

[1]U.S. Forest Service Pacific Northwest Station is conducting a project-level effectiveness monitoring program in the Entiat River, the Salmon Recovery Funding Board has rotating sites in the Entiat as part of its statewide monitoring program, and BPA's Action Effectiveness Monitoring program also has sites on the mainstem.

H4: The physical and biological effect of restoration actions varies with the type of restoration action.

H5: Restoring freshwater habitat in the Entiat River sub-basin will make a significant contribution toward the recovery of Entiat River spring Chinook and steelhead.

Restoration Implementation

The focus of the IMW is the lower 26 miles of the mainstem Entiat River, starting at the boundary with the U.S. Forest Service and going to the mouth of the Entiat at the confluence with the Columbia River. We are using a hierarchical-staircase design to guide where and when restoration actions are implemented to support comparisons between treatments and controls within the Entiat River, similar to that employed in Bridge Creek. The Entiat design uses the Bureau of Reclamation's tributary assessment (BOR 2009) that identified three distinct valley segments encompassing 17 geomorphic reaches along the lower 26 miles of the mainstem Entiat River (based on gradient and geomorphology) to delineate the hierarchical framework (valley segments) and controls and treatments (geomorphic reaches) (Figure 27). Each geomorphic reach has been identified as a permanent control (based on the Bureau's assessment that there were limited restoration opportunities) or as a temporary control that will eventually be treated. Restoration actions are staggered through time: targeted reaches in valley segment 3 were treated in 2012, targeted reaches in valley segment 1 were treated in 2014, and targeted reaches in valley segment 2 are scheduled for treatment in 2016 and 2017. The final round of actions will be implemented in valley segment 1 in 2020, by which time all feasible instream restoration actions will have been implemented.

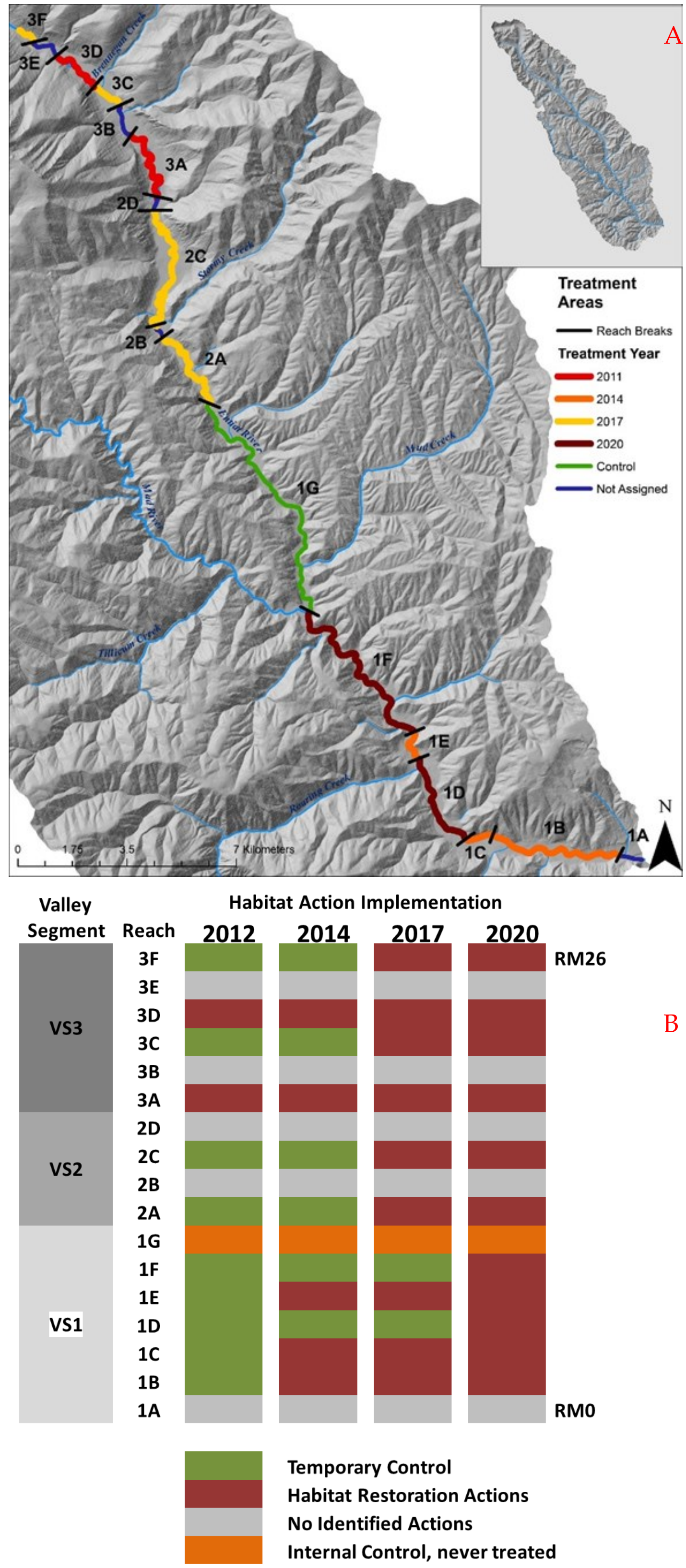


Figure 27. Location and timing of restoration actions in the Entiat River IMW. Treatments are stratified by valley types. Geomorphic reaches colored red were treated in 2012, orange geomorphic reaches were treated in 2014, yellow reaches will be targeted for treatment in 2016 and 2017, and burgundy reaches in 2020. The Mad River is being used as the internal control and is not targeted for habitat restoration (Panel A). Panel B illustrates how all targeted reaches will be treated by 2020, converting temporary controls (green) into treatments (red) and maintaining internal control sites (orange) that are never treated.

Statistical Monitoring Design

The Entiat River IMW statistical design also employs a BACI design in a nested hierarchy to compare restored and unrestored areas at the watershed, sub-watershed, and reach scales that was described for the Bridge Creek IMW. We will compare the Entiat River to the Chiwawa River[2] in the Wenatchee River subbasin at the watershed-scale and we are using the main tributary to the Entiat, the Mad River, as an internal control. We have been collecting pre-project data since 2010 (Table 5), and monitoring ideally would continue through 2023.

Table 5. Major monitoring components conducted as part of the Entiat River IMW study

Type	Monitoring Component	Spatial Design	Temporal Design	Metric Description
Fish	Juvenile salmonid capture-recapture PIT-tag surveys	300 m site within each control, treatment, and proposed treatment reach	Each site fished 2 times annually in summer and winter	Juvenile salmonid population estimates, survival, and growth rates
	Operation of Passive Instream Antennas (PIAs)	6 PIA arrays bracketing the 3 valley segments on the mainstem Entiat River and one at the mouth of the Mad River	Continuous	Juvenile salmonid movement, smolt timing. Adult salmonid spawning distribution
	Rotary screw trap	Operated at the mouth of the Entiat River	March - November	Production and productivity estimates
	Steelhead and Spring Chinook spawning ground surveys	Census counts	Steelhead February-June; spring Chinook August - September	Number of spawning steelhead and spring Chinook, spawning distribution
Habitat	CHaMP fish habitat and topographic surveys	Multiple sites within each control, treatment, and proposed treatment reaches	Sites are sampled annually according to a rotating panel design	Metrics of fish habitat quality and complexity, as well as metrics of topographic channel change. See https://www.champmonitoring.org for a complete list of metrics and descriptions
	Stream temperature	Collected at CHaMP sites	Continuous	Pre-post treatment - control comparisons of temperature change, longitudinal change in stream temperature

[2]The Chiwawa is the location of long-term monitoring of fish populations and physical habitat conditions and has been used as a comparison with the Entiat by local geologists and biologist in the past.

Habitat Restoration Monitoring

As part of the Entiat IMW we have been measuring the characteristics of fish habitat in the mainstem Entiat and Mad Rivers in order to quantify the responses to fish habitat improvement projects. Since 2011, we have conducted fish habitat surveys at 67 unique sites (300 m in length) during low flow conditions (July through September) using the CHaMP protocol. The first of four pulses of habitat improvement actions were implemented in the summer of 2012. Three projects - Dillwater, Tyee, and 3D - were constructed in two of the six geomorphic reaches (3A and 3D) of valley segment three, spanning approximately 21% of the total length of the valley segment. Actions included side channel reconnections, addition of boulders and large woody material, and riparian planting.

We performed two types of analyses to determine if the habitat improvement projects were effective. We analyzed topographic data using the GCD software in order to quantify the effects of treatments on changes to channel morphology (the physical template of fish habitat) measured as changes in erosion and deposition. We normalized change detection metrics by survey area to be able to compare different numbers of treatment and control sites that had different survey extents. We analyzed two control-treatment pairs: (1) sites in valley segment 3 that overlapped treatments were compared to sites upstream, and (2) control sites in valley segment 2 were compared to all sites in valley segment 3. We also analyzed metrics generated from the measurement of sediment, wood, channel complexity, and geomorphic units using a multivariate analysis of variance (MANOVA).

From the GCD analysis we found little change in sediment between treatment and control sites within valley segment 3 between 2012 and 2013 (Figures 28 and 29), nor was there a change when we compared valley segments 2 and 3 (Figures 30 and 31). The lack of geomorphic response may be due to several factors: projects were not of sufficient magnitude to cause geomorphic change, or the snowmelt flood in 2013 was too small to cause changes in channel morphology, or that when we compared the two types of control-treatment groups analyzed using GCD, both the control and the treatments groups of sites in the across -valley segment comparison had greater amounts of change than the within-valley segment control and treatment groups of sites. This difference may be due to the fact that the across-valley segment comparison contains more sites.

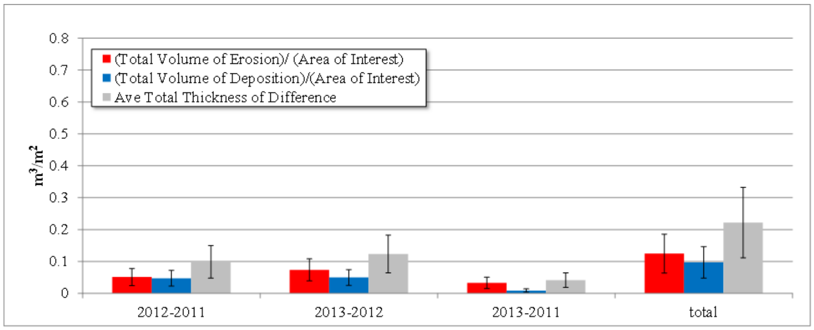


Figure 28. Geomorphic change measured as erosion and deposition at habitat monitoring control sites in valley segment 3 upstream of reaches treated in 2012 as part of the Entiat IMW.

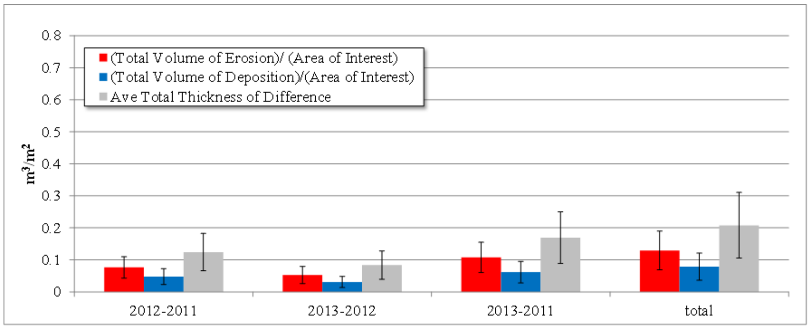


Figure 29. Geomorphic change measured as erosion and deposition at habitat monitoring sites in valley segment 3 within reaches that were treated in 2012 under the Entiat IMW.

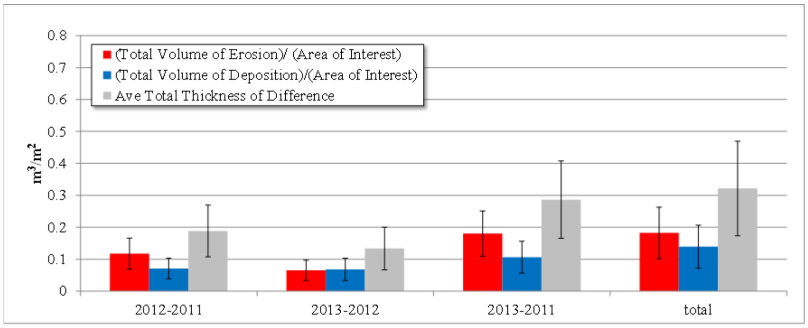


Figure 30. Geomorphic change measured as erosion and deposition at monitoring sites in valley segment 2, a control for treatments implemented in valley segment 3 in 2012 under the Entiat IMW.

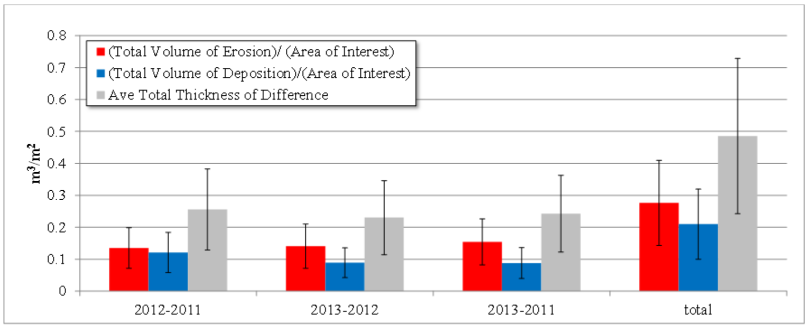


Figure 31. Geomorphic change measured as erosion and deposition at monitoring sites in valley segment 3 targeted for habitat restoration in 2012 under the Entiat IMW.

To determine if there were responses in other physical habitat characteristics, we also conducted a multivariate analysis of variance (MANOVA) on selected metrics related to channel complexity, substrate, pools, and large wood. We tested for a difference among sites in treated reaches (3A and 3D) and sites in control reaches (2A, 2C and 3C) for each year (Figure 32, Table 6). No significant difference was found among the treatment and control sites for habitat complexity, pool frequency or depth or the amount of large wood instream (Table 6). However, there was a significant difference among treatment and control sites for sediment in 2013 (Table 6, Pillai's Trace = 0.52, F = 3.329, p = 0.032).

We also tested for a difference in habitat metrics before and after treatment. There was a significant change in two of the four categories of habitat metrics (Table 8). Pool frequency and average pool residual depth were significantly greater in the post-treatment group of sites (Table 8, Pillai's Trace = 0.32, F = 7.51, p<0.01), and the frequency of large wood measured within the bankfull channel was significantly greater (Table 8, Pillai's Trace = 0.17, F = 3.20, p = 0.05). No significant difference was found in the amount of sediment or habitat complexity at sites before and after treatment.

In summary, as a result of the treatments that occurred in 2012, our reach-scale monitoring efforts detected an increase in

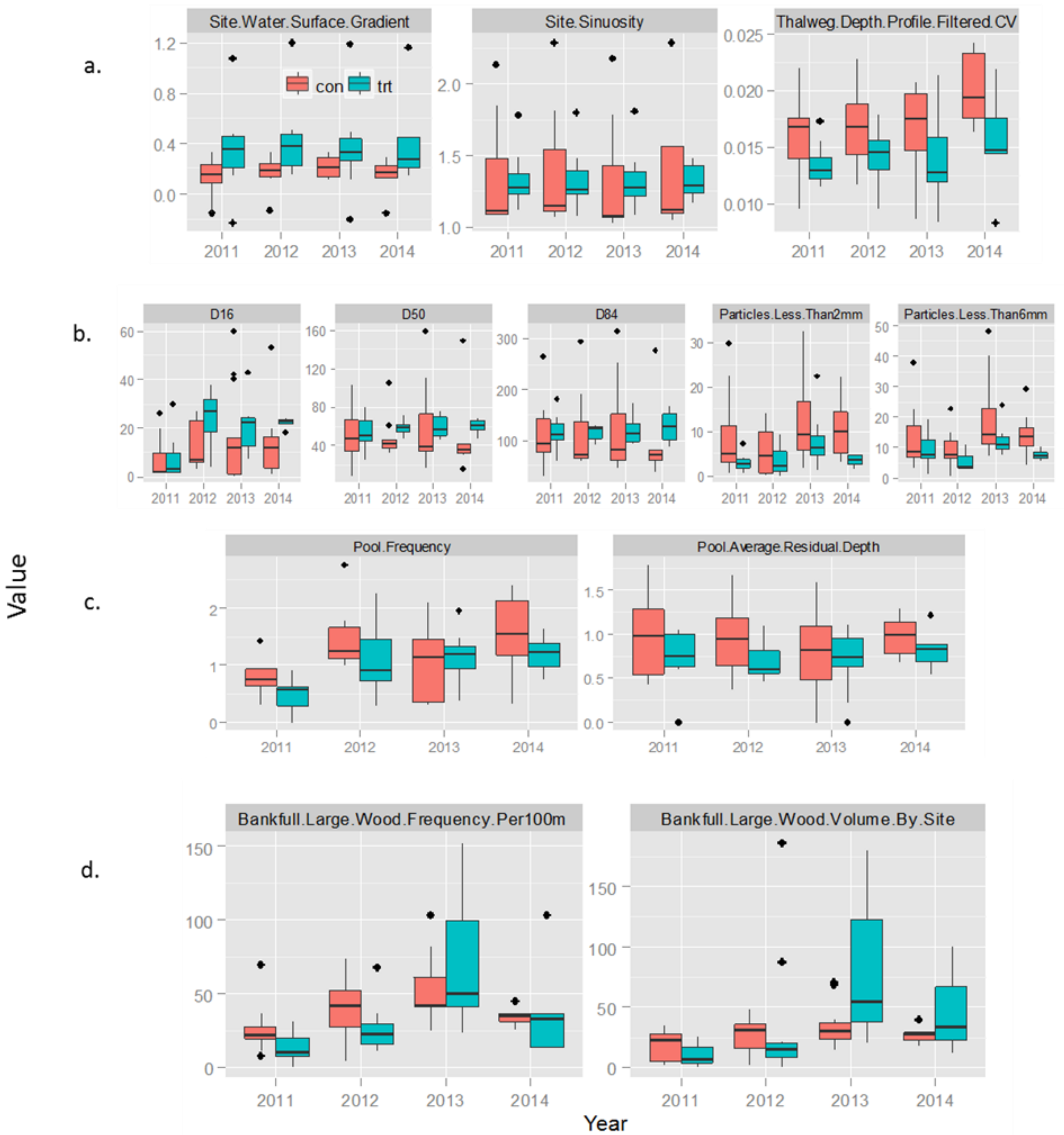


Figure 32. Comparison of selected habitat metrics representing (a) habitat complexity, (b) sediment, (c) pools, and (d) large wood loading between control and treatments sites for each year from 2011 to 2014. Boxes represent the inter quartile range, the central bar represents the median, the whiskers represent the standard error, and the points are outliers outside 1.5 times the distance of the inter quartile range. [Note: red bar diagrams are the control sites (2A, 2C and 3C) and blue are treatment sites (3A and 3D)].

Table 6. Analysis of selected habitat metrics at control and pre-treatment sites in the Entiat IMW using a multivariate analysis of variance (MANOVA) for each year 2011 – 2014.

Year	Habitat Categories	Habitat Metrics	Df	Pillai	Approx F	Num Df	Den Df	Pr(>F)
2011	Complexity	Site Water Surface Gradient, Site Sinuosity, Thalweg Depth Profile Filtered CV	1	0.279	2.196	3	17	0.126
2012			1	0.251	1.229	3	11	0.345
2013			1	0.173	1.113	3	16	0.373
2014			1	0.311	0.901	3	6	0.494
2011	Sediment	D16, D50, D84, Particles Less Than2mm Particles Less Than6mm	1	0.487	1.900	5	10	0.181
2012			1	0.421	1.311	5	9	0.341
2013			1	0.526	3.329	5	15	**0.032**
2014			1	0.735	2.78	5	5	0.143
2011	Wood	Bankfull Large Wood Frequency per100m, Bankfull Large Wood Volume By Site	1	0.203	2.293	2	18	0.130
2012			1	0.184	1.352	2	12	0.295
2013			1	0.033	0.291	2	17	0.751
2014			1	0.222	1.00	2	7	0.415
2011	Pools	Pool Frequency, Pool Average Residual Depth	1	0.221	2.553	2	18	0.106
2012			1	0.361	3.384	2	12	0.068
2013			1	0.221	2.405	2	17	0.120
2014			1	0.162	0.676	2	7	0.539

the frequency of pools, residual pool depth, and the amount of wood, but we did not see any changes to channel morphology in the year following treatment. Future topographic monitoring may yield insight into the effect of these types of treatment on channel morphology.

Table 7. Comparison of the statistical parameters of the multivariate analysis between pre and post restoration for the treatment sites (3A and 3D) and control sites (2A, 2C and 3C).

Habitat Category	Habitat Metrics	Treatment (T) or (C)Control	Comparison	Multivariate Analysis Results (MANOVA)					
				Df	Pillai	Approx.F	num.Df	den.Df	P
Complexity	Site Water Surface Gradient, Site Sinuosity, Thalweg Depth Profile Filtered CV	T	Pre and post	1.00	0.01	0.15	3.00	31.00	0.93
		C	Pre and post	1.00	0.06	0.61	3.00	27.00	0.61
Sediment	D16, D50, D84, Particles Less Than2mm, Particles Less Than6mm	T	Pre and post	1.00	0.21	1.29	5.00	24.00	0.30
		C	Pre and post	1.00	0.22	1.52	5.00	27.00	0.22
Wood	Bankfull Large Wood Frequency per100m, Bankfull Large Wood Volume By Site	T	Pre and post	1.00	0.32	7.51	2.00	32.00	**0.00**
		C	Pre and post	1.00	0.15	2.48	2.00	28.00	0.10
Pools	Pool Frequency, Pool Average Residual Depth	T	Pre and post	1.00	0.17	3.20	2.00	32.00	**0.05**
		C	Pre and post	1.00	0.03	0.42	2.00	28.00	0.66

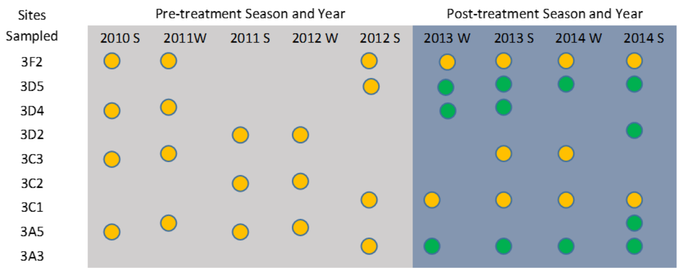
Juvenile Spring Chinook and Steelhead Population Monitoring

We began monitoring juvenile spring Chinook and steelhead in 2010, establishing sites to represent treatment and control reaches at all scales within the overall experimental design (Figure 27). Eleven annual sites represent each of the geomorphic reaches of the mainstem Entiat River, and three annual sites represent the Mad River outside control reach. An additional 18 rotating sites (six annually; three year rotation) are distributed between the three valley segments of the mainstem Entiat. Juvenile monitoring sites are visited 2 times annually during summer (August), and winter (February/March). This sampling schedule was established to capture the seasonal variation in fish behavior between summer and winter and also to avoid sampling during steelhead spawning (March-June), avoid the high flow period of June-July when flows are too high to allow safe in-river sampling, and to be sampling when juvenile spring Chinook are large enough to tag and before outmigration begins. Within each sampling event a 300 m site is either sampled on two consecutive days as part of a mark-recapture effort, or sampled once with a single pass electrofishing effort, and a ratio estimator is then used to estimate abundance at single pass sites. Juvenile salmonids are captured using backpack electrofishing, seine netting (beach or snerd) or angling in the summer period, or by nocturnal stalk netting in the winter. Each captured Chinook and steelhead is scanned for a PIT tag, and if it is not tagged it is implanted with a new PIT tag in the body cavity.

Only one season of post-implementation fish monitoring has occurred since the summer 2014 actions were implemented (winter 2015), so current analysis is limited to pre- and post-treatment for actions implemented in valley segment 3 in 2012 (5 years of pre-treatment data and 2 years of post-treatment data). A total of 40 fish sample events occurred between 2010 and 2014 in valley segment 3, of which 12 sites were in reaches that received restoration actions in 2012 (Table 8). We estimated abundance from sites in valley segment 3 using the Chapman modification to the Petersen estimate as presented in Van Den Avyle and Hayward (1999), and tested the validity of the estimates using the Robson and Regier (1964) method.

We performed a factorial ANOVA analysis to determine if (a) restoration actions resulted in an increase in abundance of juvenile spring Chinook and steelhead at the reach or watershed scale after habitat actions were implemented in reaches in valley segment 3 in 2012, and (b) whether there was a difference in fish response by season. Due to the short time series of data post-treatment and the limited number of reaches treated this analysis should be considered preliminary.

Table 8. The number of sites sampled for juvenile Chinook and steelhead abundance in valley segment 3 of the Entiat River IMW before and after habitat restoration actions were implemented in the summer of 2012. Yellow circles are control sites, green circles are sites that were in reaches receiving restoration actions. S = Summer, W = Winter season of sampling.



We found no significant difference in estimated Chinook abundance between control and pre-treatment sites (ANOVA $F_{2,39} = 0.01$, $P = 0.92$), nor in abundance at sites across years (ANOVA $F_{2,29} = 0.13$, $P = 0.88$) nor a significant in interaction between site × year (ANOVA $F_{2,39} = 0.36$, $P = 0.70$) (Figure 33).

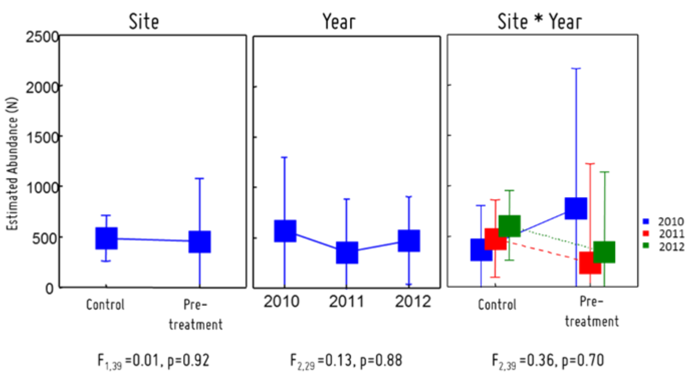


Figure 33. Abundance of juvenile Chinook at control and pre-treatment sites in valley segment 3 in the Entiat River IMW 2010-2012.

Similarly, we did not find a significant difference in spring Chinook abundance estimates at sites within reaches 3A and 3D as a result of habitat restoration actions (ANOVA, $F_{1,9} = 0.89$, $P = 0.37$). There was no significant effect of season on Chinook abundance (ANOVA, $F_{1,9} = 0.75$, $P = 0.41$), nor a significant in interaction between treatment and season (ANOVA $F_{1,9} = 3.65$, $P = 0.09$) (Figure 34).

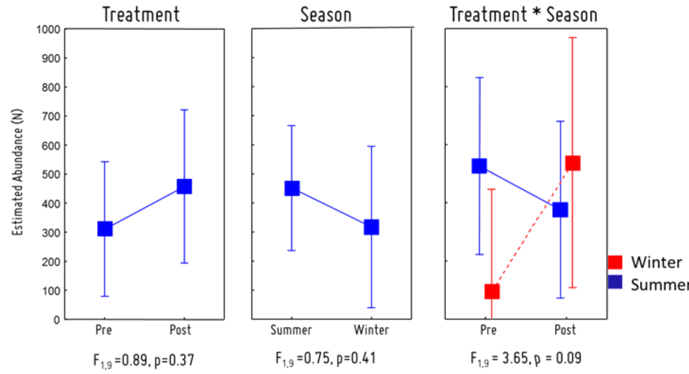


Figure 34. Abundance of juvenile Chinook at sites pre- and post-treatment in valley segment 3 in the Entiat River IMW 2010-2012.

We did not find a significant change in Chinook abundance at control sites in valley segment 3 in the pre- and post-treatment years (Figure 35; ANOVA $F_{1,10} = 0.07$, $P = 0.79$), nor a significant difference in Chinook abundance at control sites across summer and winter ($F_{1,10} = 3.86$, $P = 0.08$), or a significant in the interaction of treatment and year interaction ($F_{1,10} = 0.02$, $P = 0.89$) (Figure 35).

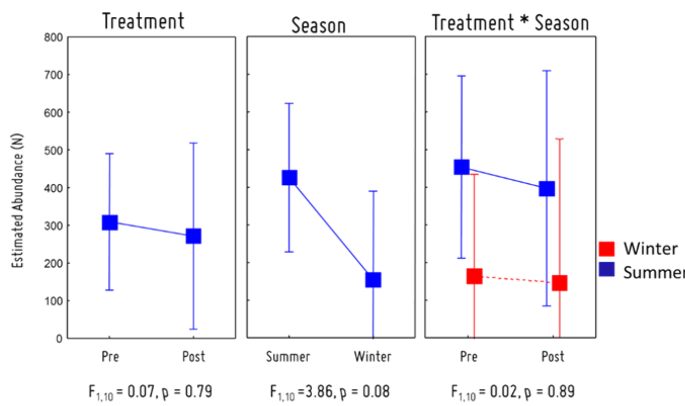


Figure 35. Abundance of juvenile Chinook at control versus treatment sites in valley segment 3 in the Entiat River IMW after habitat restoration actions in 2012.

We were unable to conduct a preliminary comparison of pre- and post-treatment abundance for steelhead because reliable site level abundance estimates could not be derived for several of the pre-treatment sites; however, a comparison of post-treatment steelhead abundance across seasons showed no significant difference (Figure 36; ANOVA $F_{1,10} = 3.86$, $P = 0.48$). Future analyses will be possible for steelhead as the abundance data time series is increased, and as additional treatment reaches are incorporated into the analysis. Post-hoc analyses may also provide abundance estimates for some pre-treatment sites that are currently lacking data.

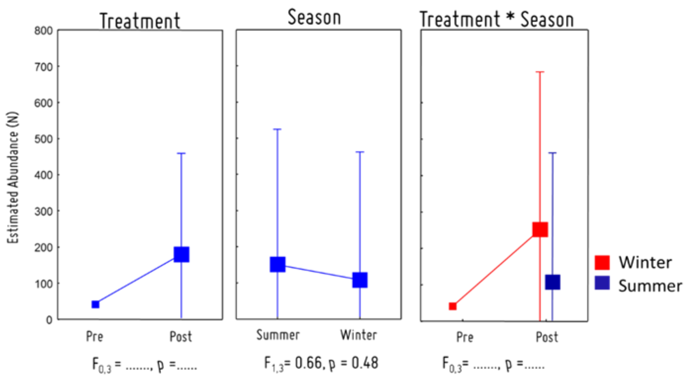


Figure 36. Abundance of juvenile steelhead at sites pre- and post-treatment in valley segment 3 in the Entiat River IMW 2010-2012.

In conclusion, despite such a limited time series of data to analyze and a lack of significance in the results, patterns in the data suggest that restoration actions have improved Chinook abundance at sites within restored reaches, especially in winter. More years of post-treatment monitoring will improve the dataset, as will more treatments, for example, in the near-term being able to include monitoring data collected in the 2015 field season from sites in reaches treated in 2014.

Juvenile Spring Chinook and Steelhead Survival Results

Improvements in rearing and winter habitat as a result of habitat restoration actions in the Entiat IMW may translate into increased juvenile survival. We are estimating survival probabilities using PIT tag technology where parr are tagged during the summer and winter fish sampling events and are redetected through a variety of means: either as a recapture at a subsequent sampling event, as a resight at the PIAs, or as a capture at the rotary screw trap at the mouth of the Entiat River during emigration. Re-sighting by the PIAs is the primary method for re-encountering PIT tagged fish. The six arrays on the mainstem bound the three valley segments, potentially allowing us to estimate survival probabilities at the valley segment level given enough data (Figure 37).

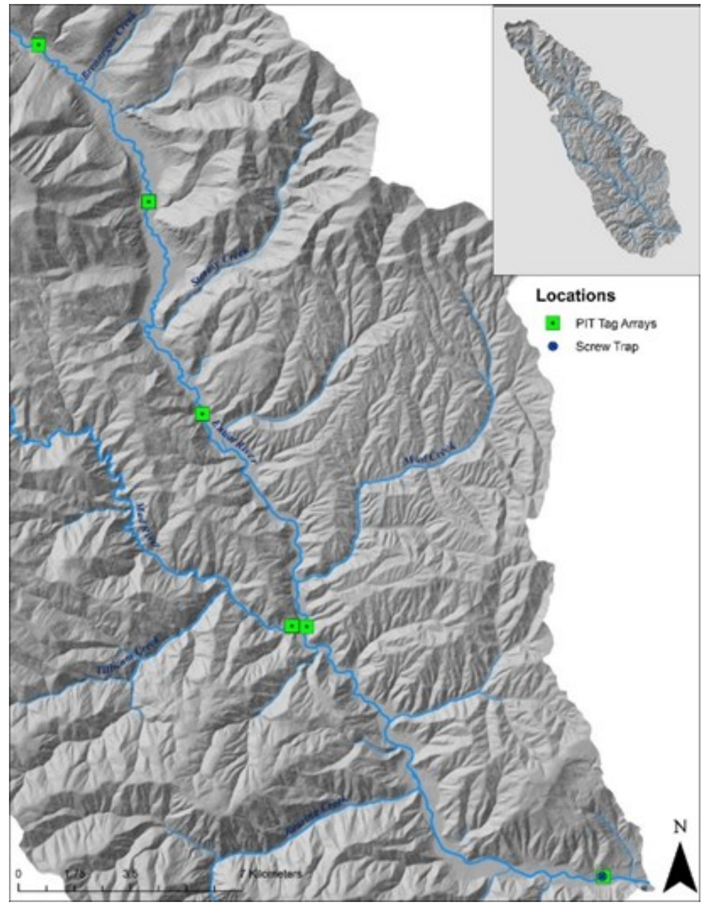


Figure 37. Location of fixed Passive Instream Arrays on the Entiat and Mad Rivers, Washington.

We used the Barker model (Barker 1997, Barker *et al.* 2004) within Program MARK (Cooch and White 2010) to estimate juvenile survival probability. Estimates for Chinook salmon and steelhead are apparent survival rather than true survival probability, in that they include known emigrants from the study area whose contribution to the likelihood function is removed after their final encounter, as well as unknown emigrants that could cause a negative bias in survival estimates.

To date, consistent model convergence has not been possible for summer survival estimates for Chinook due to their life history (Chinook leave the Entiat River quickly as yearlings or sub-yearlings resulting in relatively low numbers of marked fish and re-sights) but winter survival has been estimated for both species at the subbasin (Figure 38) and valley segment level (Figure 39 and Figure 40) and we have tested for differences in survival probabilities by valley segment and by year (Figure 41).

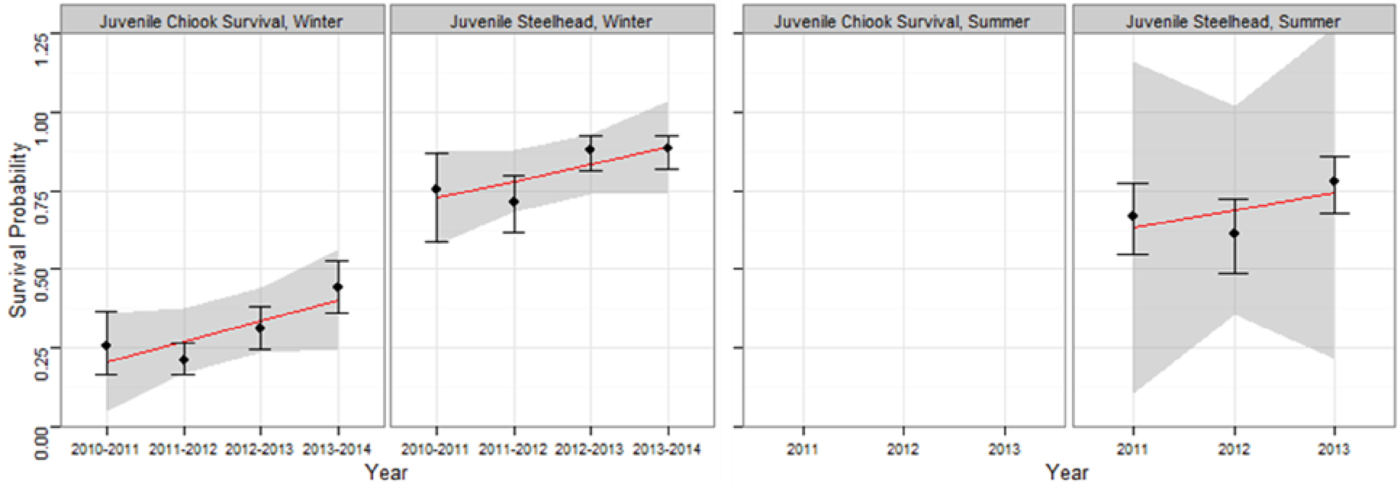


Figure 38. Winter and summer survival probability of juvenile Chinook salmon and steelhead 2010- 2014 at the subbasin scale. The bar plot represents mean survival probability and the 95% confidence interval of the mean for each year while the gray area and the red line represent the 95% confidence interval of the mean of the survival probability over time.

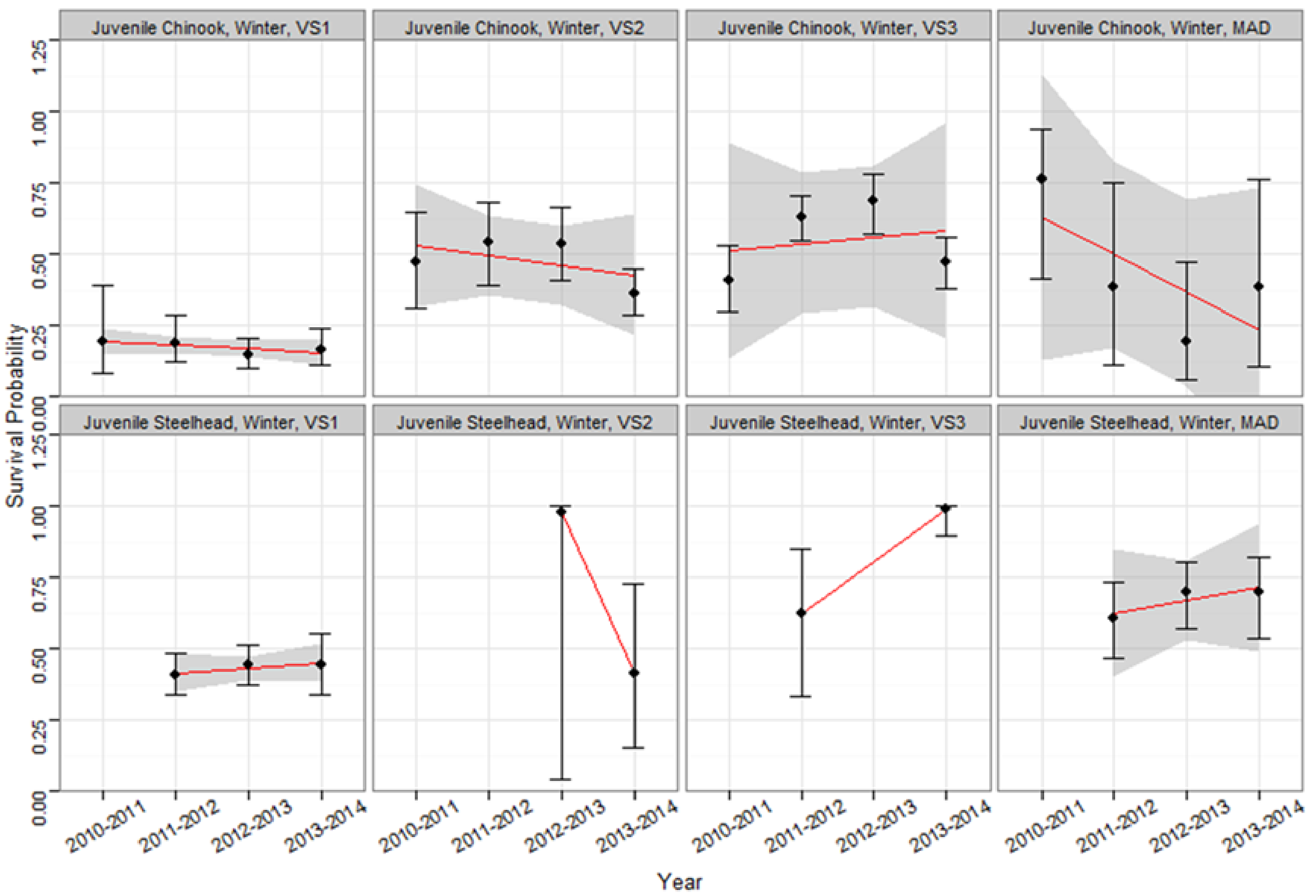


Figure 39. Winter survival probability of juvenile Chinook salmon and steelhead for the three valley segments (VS1, VS2, and VS3) and Mad River 2010- 2014. The bar plot represents mean survival probability and 95% confidence interval. The gray area and the red line represent the 95% confidence interval of the mean of the survival probability over time. Note: linear trend over time could not be computed at a two valley segments (VS2 and VS3) for the winter period due to insufficient data points.

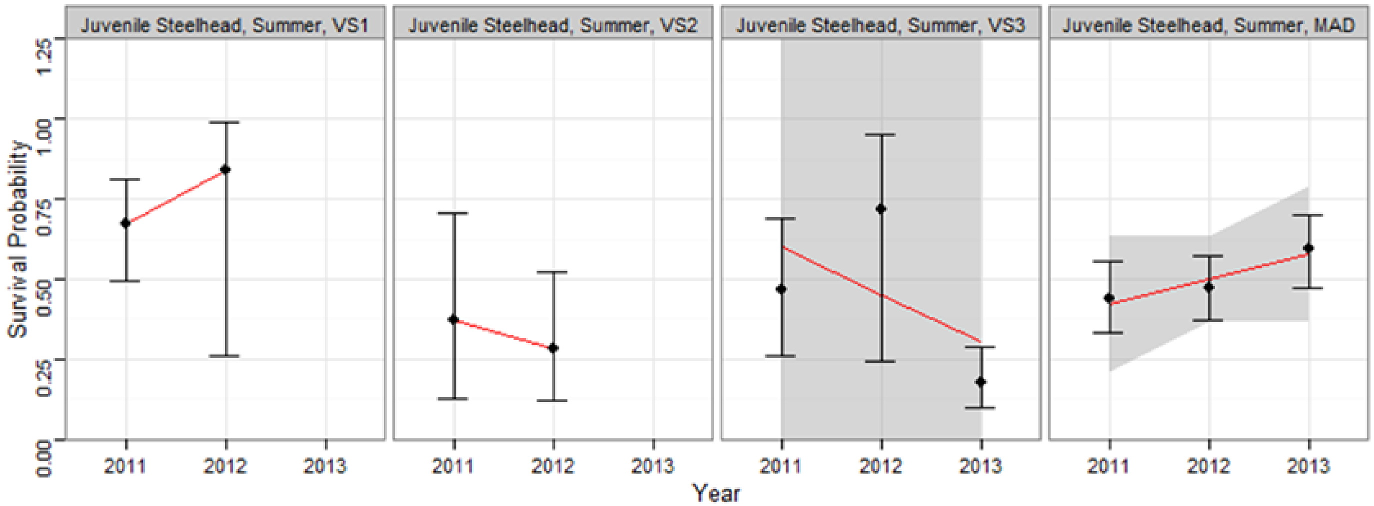


Figure 40. Summer survival probability of juvenile steelhead at three valley segments (VS1, VS2, and VS3) and Mad River for 2011- 2013. The bar plot represents mean survival probability and its 95% confidence interval. The gray area and the red line represent the 95% confidence interval of the mean of the survival probability over time. Note: linear trend of the survival over time could not be estimated for valley segments 1 and 2 due to insufficient data points.

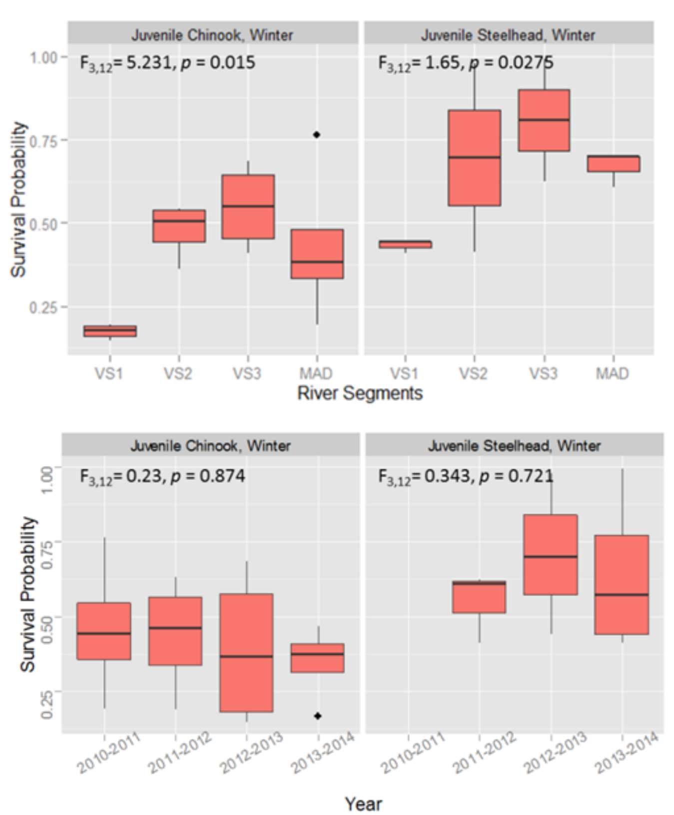


Figure 41. Survival probability (winter) with site (upper panel) and year (lower panel) for Chinook salmon and steelhead. Boxes represent the interquartile range, the central bar represents the median, the whiskers represent the standard error, and the points are outliers outside 1.5 times the distance of the interquartile range.

Side Channel Habitat

Side channel habitat has been identified as important for rearing and flow refugia of juvenile salmon and steelhead in the Entiat River. ISEMP collaborators U.S. Fish and Wildlife Service Mid-Columbia Fishery Resource Office (MCFRO) have been conducting juvenile Chinook and steelhead monitoring for abundance, survival and growth at 5 side channels along the mainstem Entiat under ISEMP (Figure 42). Sites considered for the off-channel habitat study were limited to habitats distinctly separate from the main river channel where 1) flow was perennial, 2) the site was accessible year round, and 3) physical site conditions supported the PIT tag antenna monitoring requirements of the study. The duration of monitoring varies by side channel (Table 9) partly as a result of when restoration actions where implemented: two of the side channels were either created (Tyee) or enhanced (3D) as part of the restoration work in 2012 in valley segment 3; the remaining 3 side channels (Wilson's, SanRay, and Harrison's) are existing habitat features of varying complexity in valley segment 1. MCFRO personnel conducted mark-recapture fish sampling in summer, fall and winter using backpack electrofishing, seining, and hand-netting over two consecutive days, and used block nets at the top and bottom of each site for the duration of the mark-recapture period and temporary PIT tag detection arrays at the inlet and outlet of each side-channel site for the duration of the study. ISEMP personnel conducted habitat surveys using CHaMP protocols in the summer. Preliminary analysis results are presented for Chinook in side channels; work on steelhead, which is more complicated given their life history, is underway and will be presented in a subsequent report.

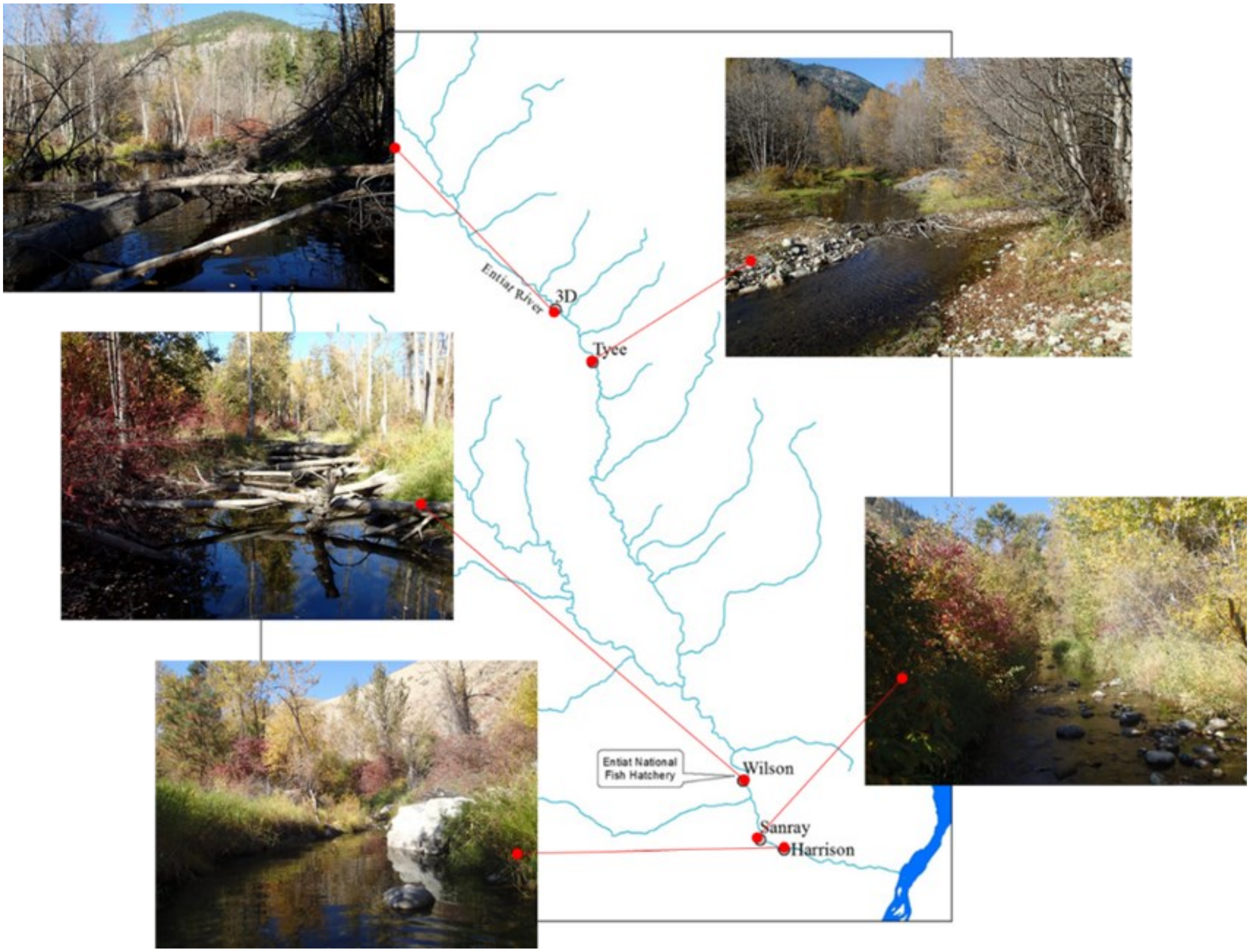


Figure 42. Location of side channels monitored in the Entiat River Intensively Monitored Watershed, Washington.

Table 9. Side channels monitored under the Entiat River IMW, location within valley segments, description, and start date of monitoring.

Side Channel	Valley Segment	rkm	Date Completed	Length (meters)	Date Monitoring Started
3D	3	42.7	2012	402	March 2013
Tyee	3	38.0	2012	304	August 2013
Wilson	1	11.0	2006	285	Fall 2011
SanRay	1	7.0	Naturally occurring	117	Fall 2011
Harrison	1	6.0	2008, 2012	515	March 2013

Juvenile Chinook Density in Side Channel Habitat

We found an apparent seasonality of use of side channels by juvenile Chinook, where density appears to be greater in the summer (average 0.50 ± 0.53 parr/m^2 (mean ± 95% CI)) than in the winter (0.02 ± 0.02 parr/ m^2)(Figure 43). Average parr density was higher in valley segment 1 side channels than in valley segment 3 side channels in the summer, a difference that was less pronounced by fall. However, by the winter, parr density was higher in valley segment 3 side channels than those in valley segment 1. This use pattern could be a reflection of the life history of Chinook in the Entiat, where we see a large scale movement downstream to valley segment 1 after emergence in valley segment 3. Many of the Chinook that move downstream emigrate as sub-yearlings and are not available to use the valley segment 1 side channels; Chinook that elect to stay in valley segment 3 and winter in the Entiat (emigrating the following spring as yearlings) are available to utilize the side channel habitat.

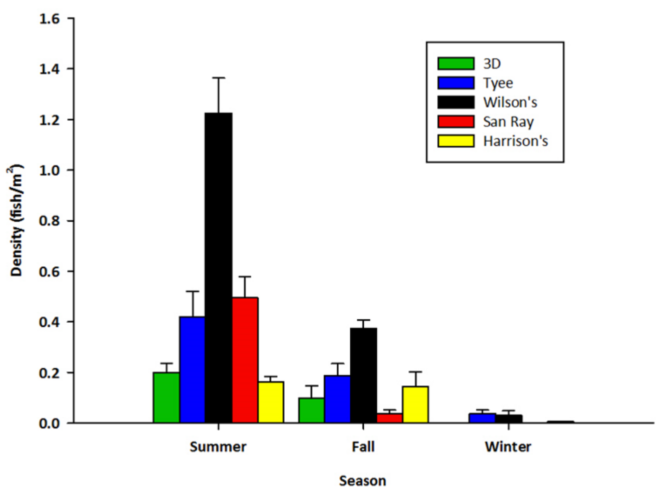


Figure 43. Chinook parr densities (±95% CI) in side channel habitat on the Entiat River, Washington from August 2013 - March 2014.

Juvenile Chinook Survival in Side Channel Habitat

We estimated apparent survival probability for juvenile Chinook for each side channel by season using the Barker model (Barker 1997, Barker *et al.* 2004) but could not estimate survival for steelhead due to insufficient steelhead numbers. Overall, survival of juvenile Chinook in side channels was higher during the summer period (August-October 2013) than the winter period (October 2013-March 2014) (Figure 44). Apparent survival was also higher in valley segment 3 side channels (3D and Tyee) compared with valley segment 1 side channels (Wilson's, Harrison's, and SanRay) in both summer and winter.

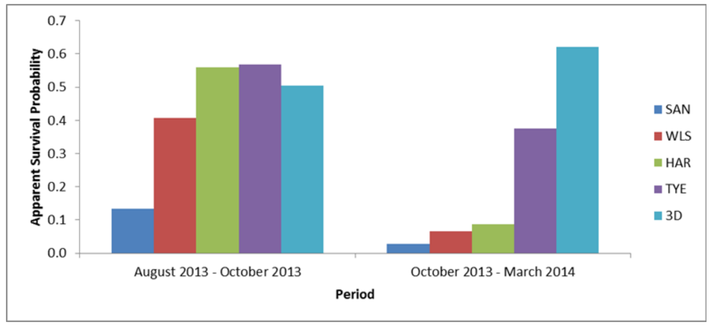


Figure 44. Summer (August – October) and winter (October – March) apparent survival probabilities of juvenile Chinook salmon in side channel habitat in the Entiat River, Washington, 2013-2014.

Size and Condition of Juvenile Chinook in Side Channel Habitat

Juvenile Chinook have the greatest growth rate over the summer period and as survival is linked to size (larger fish survive better) there are advantages to being able to grow quickly before the winter or emigration. We found significant differences in summer fish length and weight between some side channels and the adjacent mainstem sampling sites, but not others. Juvenile Chinook in valley segment 3 side channel 3D were significantly larger than those in the adjacent mainstem; however, juvenile Chinook sampled in the Tyee side channel were smaller than those in the adjacent mainstem (length ANOVA: $F_{2,1711} = 375.70$, $P < 0.001$, weight ANOVA: $F_{2,1711} = 412.46$, $P < 0.001$; post hoc comparisons t-tests all $P < 0.001$) (Table 10). A clear pattern was not obvious among side channels and the mainstem in valley segment 1 although significant differences do exist in length and weight (length ANOVA: $F_{3,2998} = 45.09$, $P < 0.001$, weight ANOVA: $F_{3,2998} = 35.09$, $P < 0.001$): the largest fish by fork length were sampled in the SanRay side channel (based on pairwise comparisons, all $P < 0.05$; Table 10); however, Chinook sampled in the Wilson and Harrison side channels were significantly lighter than those in the mainstem (Table 10). Chinook sampled in the winter in the mainstem in both valley segment 1 and 3 were longer and heavier compared with Chinook sampled in the side channels (Table 10), which may suggest that side channel habitat is not preferred in the winter and the larger fish are selecting for the mainstem.

Table 10. Mean (+95% CI) fork length (mm) and weight (g) of Chinook parr from side channels and their corresponding mainstem valley segment sites (valley segment 1 and valley segment 3) in the Entiat River, Washington. Mainstem sites were not sampled in the fall. There are no data for SanRay, Harrison's, and 3D in the winter due to an insufficient number of fish sampled.

		Summer		Fall		Winter	
Valley Segment	Site	Fork Length (mm)	Weight (g)	Fork Length (mm)	Weight (g)	Fork Length (mm)	Weight (g)
VS1	Mainstem	67.0 ± 0.5	3.77 ± 0.11	-	-	90.0 ± 0.9	7.27 ± 0.24
	Wilson's	65.7 ± 0.6	3.26 ± 0.10	73.7 ± 0.9	4.37 ± 0.17	77.0 ± 3.9	4.85 ± 0.82
	SanRay	69.6 ± 0.9	3.82 ± 0.17	76.3 ± 3.3	4.85 ± 0.75	-	-
	Harrison's	54.1 ± 1.3	1.67 ±0.16	72.9 ± 4.4	4.65 ± 0.81	-	-
VS3	Mainstem	63.1 ± 0.7	3.17 ± 0.11	-	-	89.7 ± 0.8	7.78 ± 0.22
	Tyee	54.3 ± 0.7	1.70 ± 0.08	63.6 ± 1.4	2.93 ± 0.22	76.1 ± 2.9	4.69 ± 0.62
	3D	76.2 ± 0.8	5.30 ± 0.18	81.5 ± 1.1	6.33 ± 0.24	-	-

Occupancy Time for Juvenile Chinook in Side Channel Habitat

Movement rates out of the side channels can give an indication of the longevity of use, and we found that an average of 39% in the summer (Figure 45) and 38% in the fall (Figure 46) of tagged juvenile Chinook moved out of the side channel they were sampled in within a day or were not detected again in that side channel. Another 29% moved out in the summer and 34% in the fall between 1 and 10 days after marking. Juvenile Chinook salmon sampled in valley segment 3 side channels appear to reside in the side channel for a longer period of time than those sampled in valley segment 1. A rapid movement out of the side channels could be due to a tagging/handling effect, or it could reflect use patterns based on side channel habitat characteristics or resource availability.

In summary, the effectiveness of side channels in restoring salmonid populations is likely driven by several factors such as the side channel's habitat characteristics and location in the watershed, as well as the life history of the juveniles using them. Our next steps with respect to determining the effectiveness of side channel restoration include:

- Analyzing steelhead data for juvenile abundance, density, survival, occupancy of side channels
- Characterizing side channel habitat from the CHaMP data and looking for fish-habitat relationships to explain the various responses observed so far in the Chinook metrics.
- Including the side channel fish data in the overall analysis of treatment effect at the valley segment and watershed scale.

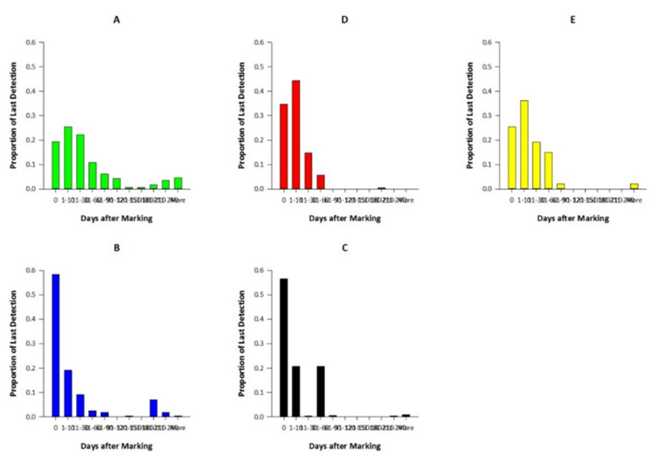


Figure 45. Days after summer sampling that a proportion of PIT tagged juvenile Chinook salmon were last detected in their respective side-channels. (A) 3D (B) Tyee (C) Wilson's (D) SanRay (E) Harrison's.

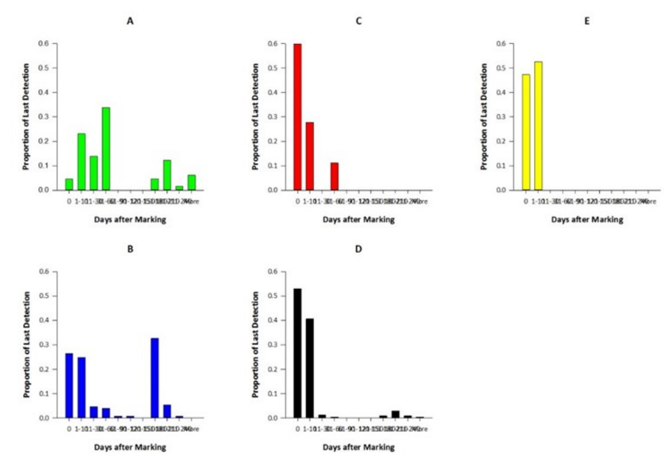


Figure 46. Days after fall sampling that a proportion of PIT tagged juvenile Chinook salmon were last detected in their respective side-channels. (A) 3D (B) Tyee (C) SanRay (D) Wilson's (E) Harrison's.

Lemhi River IMW

The Lemhi River IMW is located in the Salmon River Basin, Idaho, and is designed to assess the effectiveness of ongoing collaborative habitat restoration actions that aid in the recovery of the Lemhi River population of spring/summer Chinook salmon and steelhead. The goals of this project have two components: 1) evaluation of habitat actions at the scale of the Lemhi River and 2) development of tools that can be applied throughout the Columbia River Basin. At the scale of the Lemhi River, the primary goal of the program is a quantitative evaluation of whether habitat restoration actions achieved the 4% for steelhead and 7% for spring/summer Chinook salmon improvements in freshwater productivity (i.e., smolts per female spawner) as identified in the 2008 BiOp (NOAA 2008). We have hypothesized that improvements in the quality and/or quantity of existing habitat will increase the productivity and spatial connectivity of anadromous and resident (e.g., rainbow trout) salmonid populations. Thus, if out-of-subbasin survival does not exhibit density dependence, more smolts will translate into more adults.

Chinook salmon and steelhead in the Lemhi River have been declining for decades and the 10-year (2000-2009) geometric mean of Chinook salmon spawners is 96 (38-582), and 0.94 (0.59-1.52) recruits/spawner (Ford et al. 2010). The mean for spawners during 2009-2013 was slightly higher at 220, ranging from 120 in 2012 to 337 in 2011(ISEMP 2013). For steelhead, the average spawners from 2009-2012 is 493 (421-630 range; ISEMP 2013; insufficient data meant escapement estimates for previous years was not available). The ICC-TRT target for Chinook salmon is a minimum threshold of 2,000 spawners and 1.34 recruits/spawner, whereas the target for steelhead is 1,000 spawners and 1.14 recruits/spawner (ICTRT 2007).

Three primary factors are believed to impose significant constraints on the viability of steelhead and Chinook salmon in the Lemhi River: (1) out-of-subbasin mortality, primarily resulting from passage mortality associated with the FCRPS; (2) loss of access to tributary habitat resulting from channel dewatering; and (3) decreased mainstem habitat quality resulting from decreased flow. Similarly, the productivity and survival of resident salmonids are believed to be negatively impacted by the isolation of tributary habitats, which results in: (1) loss of population connectivity; (2) decreased access to cool water refugia, and (3) impedance of spawning migrations due to the untimely dewatering of tributary habitats.

The objectives of the Lemhi IMW and associated Restoration Plan fall into four broad categories (QCI 2005):

1. Remove or reduce upstream and downstream migration barriers to fish and provide access to available spawning and rearing habitat by:

- providing flow to maintain hydraulic and ecological connectivity between the mainstem and tributaries so that fish have access to historically productive habitat;
- providing flow in the lower reach of the Lemhi River so that adults and juveniles can freely migrate in and out of the Lemhi subbasin;
- removing physical obstructions (e.g., irrigation berms and push-up dams) that limit localized movements of upstream and downstream migrating fish; and
- minimizing entrainment into irrigation ditches that do not provide adequate rearing habitat and do not functionally reconnect with the Lemhi River or tributaries.

2. Maintain or enhance riparian conditions characteristic of good habitat to ensure that adequate vegetation persists to provide shade, increase bank stability and protection, decrease sediment input, and promote the recruitment of large woody debris.

3. Decrease sediment and temperature, provide quality substrate, and increase the abundance and quality of off-channel habitat, and increase pool frequency and quality to improve productivity and survival.

4. Tributary reconnections: there are a total of 31 tributaries to the Lemhi River, and with the exception of Hayden Creek and Big Springs Creek, in most years all are dewatered in their lower reaches during the irrigation season, and are therefore isolated from the Lemhi River. Tributaries contain habitat that is believed to be important for the persistence of fish in the Lemhi basin. As such, a primary focus of the IMW is to re-establish tributary connectivity so fish may access habitat in these watersheds.

The habitat improvements in the Lemhi River are aggressive and occur at multiple spatial scales so that the effect size of the actions is anticipated to be sufficient for resolution at reach, subpopulation and aggregate population scales, and are thus ideally suited for effectiveness monitoring (Table 11). In addition, the diversity of habitat actions enables a well-designed study to assess the effects of multiple classes of habitat actions (e.g., flow enhancement, tributary reconnection etc.) using the same infrastructure and effort.

A total of 22 types of habitat restoration actions are being actively implemented in the Lemhi River (Table 11). The majority of these actions are intended to enable anadromous salmonids to access and utilize tributary habitat that was disconnected from the mainstem Lemhi as a result of irrigation withdrawals. Lemhi River managers identified “high priority” watersheds as primary targets for reconnection/restoration efforts based on existing information describing habitat conditions in concert with the logistical feasibility of obtaining successful tributary reconnections; for example, the number of flow enhancement or alternative water diversion projects necessary to maintain instream flow, and taking into account their costs. ISEMP personnel, in collaboration with the co-managers and federal agencies, are tasked with evaluating the effectiveness of high priority watershed reconnections, and identifying whether additional tributary reconnections (“moderate priority” watersheds) will be necessary to achieve the freshwater productivity improvements identified in the BiOp.

The core of the Lemhi IMW monitoring strategy is a life cycle model (QCI 2005) that views freshwater productivity as a function of habitat quantity and quality (for details see Chapter 6). Implementation of ISEMP in the Salmon River Subbasin is structured to estimate fish and habitat metrics to directly populate the model. The life cycle model requires fish population data for life-stage specific juvenile abundance, productivity/survival, growth/condition, spatial distribution, and adult escapement across habitat classes and within treated and untreated stream reaches.

Sampling Design and Methodology

In order to effectively monitor the diversity of habitat restoration projects and quantify multi-scale fish-habitat relationships an extensive sampling design both in sampling and infrastructure is required (Figure 47). The Lemhi is broken into 16 strata, associated with specific restoration actions and different ecological conditions. The mainstem Lemhi is divided into upper and lower mainstem reaches divided at the confluence of Hayden Creek. The lower mainstem reach is a highly channelized and does not have any Chinook spawning and minimal steelhead production, whereas the upper mainstem has a majority of the Chinook spawning for the watershed and significant resident rainbow spawning. Hayden Creek is considered the reference reach due to the lack of restoration planned there. Each tributary is designated as an individual stratum in order to isolate the effects of restoration actions, most importantly hydraulic reconnections.

The sampling frame is designed to incorporate all areas where steelhead and Chinook are likely to be observed. We completed extensive surveys throughout the watershed to define the upper extent of steelhead (or resident rainbow's in areas without hydraulic connection to mainstem) and Chinook salmon in collaboration with IDFG in 2009 and 2010. As reconnections are completed and sub-populations have time to redistribute throughout the watershed, we will periodically repeat the extensive surveys to ensure the sampling frame is correct. It is important to note that Chinook salmon only spawn in Hayden Creek and the upper mainstem Lemhi River.

Three RSTs are deployed in the Lemhi: at the mouth of Hayden creek, in the upper Lemhi River just upstream of the confluence with Hayden Creek, and in the lower mainstem River near 5 km from the confluence with the Salmon River. These locations existed historically and support sampling of two Chinook sub-populations. Seventeen PIAs are located throughout the watershed at locations important to pre/post tributary connections. Arrays are located near each RST and near the confluence of each tributary that has been or will be connected in the next 10 years. Additionally, we have installed several arrays to determine the efficacy of passage restoration projects.

In order to estimate life-stage specific survival and abundance, we have implemented two novel designs in order to

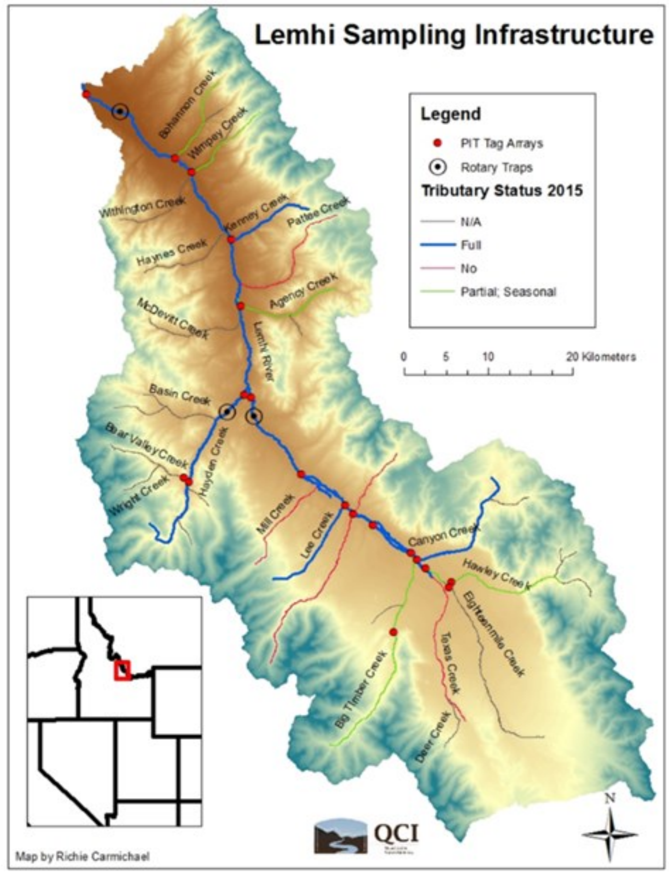


Figure 47. Distribution of juvenile capture and fixed fish sampling infrastructure in the Lemhi River IMW. Tributary status is associated with hydraulic connectivity to the mainstem Lemhi River: N/A - not in sampling frame; Full - complete connectivity throughout the year; Partial - connectivity during some months, usually high flow periods.

better understand fish life-stage specific abundance and survival (see QCI 2013 for greater detail): 1) Decomposition of aggregate Lower Granite Dam (LGR) adult spring/summer Chinook salmon and steelhead escapement into sex and age-structured population/tributary specific escapement estimates (see Chapter 6), and 2) Remote-site juvenile enumeration and tagging surveys to estimate the standing crop of juvenile salmonids prior to emigration. Remote-site juvenile surveys are used to representatively capture, enumerate, and PIT tag juvenile spring/summer Chinook salmon and *O. mykiss*. Surveys are distributed throughout the geographic range occupied by anadromous salmonids using a GRTS design (Stevens and Olsen 2004). Distribution of this effort using GRTS enables the estimation of life-stage specific juvenile abundance for individuals greater than 50 mm at the population scale (i.e., parr and subsequent life-stages) and in individual treatment and reference locations. Additionally, a continuous sampling design was initiated in 2013 to help with logistical issues associated with reach-based sampling. Using spatially explicit locations for every tagged, recaptured or resighted fish using mobile PIT tag equipment as well as continuously sampling fish allows the estimation of micro-habitat use as well as seasonal stream use (Figure 48).

Table 11. Lemhi River Intensively Monitored Watershed planned restoration actions.

Conservation Measure	Title	Geographic Area	Objective	Description
CM – 01	Lemhi River Tributary Reconnects	Lemhi River Tributaries	Fish Passage	Provide hydraulic and ecological connectivity between the Lemhi River and 10 tributaries
CM – 02	Removal of Irrigation Structures and Road Culverts that Inhibit Fish Passage	Basinwide	Fish Passage	Identify fish passage problems and improve fish passage throughout the Lemhi River basin
CM – 03	Fish Screening to Reduce Entrainment in Irrigation Canals	Basinwide	Fish Passage	Screen irrigation ditches to reduce entrainment and associated mortality of fish in tributaries
CM – 04	Eliminate Ditch Return Threats	Basinwide	Fish Passage	Prevent fish from entering irrigation ditches from the downstream end
CM – 05	Riparian Grazing Management	Basinwide	Riparian Habitat Protection	Improve riparian zones along the Lemhi River and tributaries to rehabilitate fish habitat
CM – 06	Enhance Side Channels and Secondary Rearing Channels	Middle Reach Upper Reach	Stream Habitat Improvement	Provide fish access to side channels to enhance spawning habitats and juvenile rearing capacity
CM – 07	Lemhi River Stream Channel Rehabilitation	Lower Reach Middle Reach Upper Reach	Stream Habitat Improvement	Restore large segments of the Lemhi River to improve habitat condition for spawning and rearing
CM – 08	Pool Development	Lower Reach Middle Reach Upper Reach	Fish Passage and Stream Habitat Improvement	Improve fish passage and rearing in the Lemhi River by increasing the number of pools
CM – 09	Maintain Biologically Sufficient Conditions for Fish Passage in the Lower Lemhi River	Lower Reach	Fish Passage	Minimum continuous stream flows below the L6 diversion and modifications to the river channel would be used to maintain biologically adequate fish passage for access to the middle and upper river reaches and tributaries
CM – 10	Upper Lemhi River Chinook Salmon Assessment	Upper Reach	Stream Habitat Improvement	McFarland stream flow and fish performance study
CM – 11	High Volume Flow to Improve Instream Habitat Conditions	Upper Reach	Stream Habitat Improvement	Provide high volume stream flows to maintain stream channel complexity and rehabilitate fish habitat
CM – 12	Maintain fish Passage in the Lower Reaches of Hayden Creek	Hayden Creek	Fish Passage	Preserve continuous flows that historically have been available in lower Hayden Creek for migrating fish

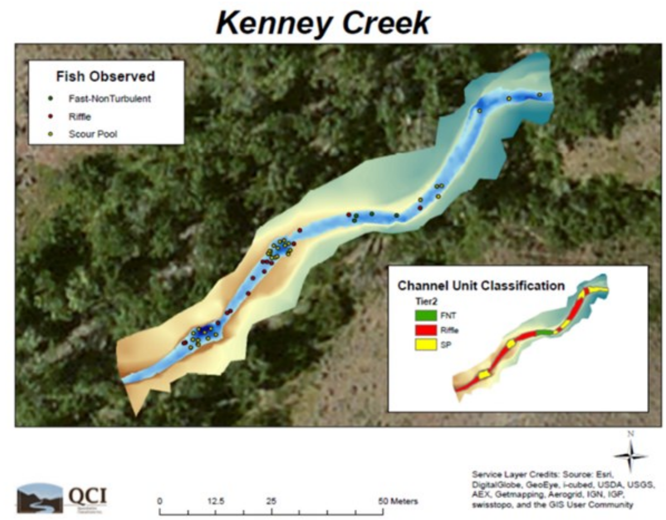


Figure 48. An example of how spatially continuous PIT tag detection survey data can be joined to habitat attribute data collected during spatially continuous fish capture effort. This example utilizes habitat collected at a CHaMP site, overlaid with the channel units identified for individual fish detections.

Comparison against GRTS-based abundance estimates from prior years demonstrates that the spatially continuous sampling approach generally produces more precise estimates. Additionally, the mobile PIT tag detection efforts in coordination with habitat data collected simultaneously with the mark survey allow reach and tributary scale associations between fish residency and habitat features (Figure 48). Subsequent recapture of tagged juveniles in remote-site surveys and RSTs accompanied by "re-sight" information from strategically located PIAs enables life-stage specific survival estimates to be generated for the population and in treatment and reference locations. We also collect tissue and scale samples from PIT-tagged individuals enabling an estimate of the sex and age structure of the standing juvenile population, which can be contrasted with the age and sex structure of emigrating juveniles. Over time, this contrast in age structure will allow us to partition resident versus anadromous *O. mykiss* production.

ISEMP personnel began fish monitoring in 2009 with collection of fry, parr, and presmolts from brood year 2008. In 2010, PIA installations provided data for estimation of adult escapement, starting with the 2009-2010 brood year for steelhead and 2010 brood year for spring/summer Chinook salmon.

Habitat Monitoring

Habitat monitoring, using the CHaMP protocol beginning in 2011, is distributed within the same GRTS sample frame as the fish monitoring effort. This overlap allows fish and habitat relationships to be developed at the site, tributary, and population scales. These efforts allow estimation of habitat attributes in treatment and reference areas that can be used to directly populate the watershed model and develop fish and habitat relationships. Habitat surveys are conducted from June – October annually.

The combination of remote site and population-level sampling (continuous, site-based, LGR, RSTs, PIAs) allows us to examine fish and habitat relationships at a range of spatial and temporal scales. This provides the opportunity to describe life-stage specific responses to individual habitat actions and to accumulated effects of multiple habitat actions at the population level, while controlling for environmental variation that might otherwise obscure those relationships or reduce our ability to detect change.

Seven life-history stages are modeled in the life cycle model: egg, fry, parr, pre-smolt, smolt, adult, and spawner (see Chapter 6). The "pre-smolt" stage is used to define fish that are actively migrating in the fall. In the Lemhi River, Chinook salmon migrate by age-1, whereas *O. mykiss* may spend up to 5 years in freshwater before migrating to the ocean. We classify fish that become sexually mature in freshwater as resident spawners, which may occur at any pre-smolt age. We assumed 40% of *O. mykiss* remain in the Lemhi watershed as rainbow trout, and 5% of male Chinook salmon become resident jacks. Anadromous adults of both species can remain at sea for up to 3 years, based on ages determined from scales collected from fish at LGR. The number of adults returning to spawn is corrected by the harvest rate (0.07 for both Chinook salmon and *O. mykiss*) and the survival for adults passing through the hydrosystem in the Columbia and Snake rivers (Figure 49 and Table 12). All survival and carrying capacity inputs to the model are found in Table 13. Please note that some estimates cannot be directly estimated and literature values are used.

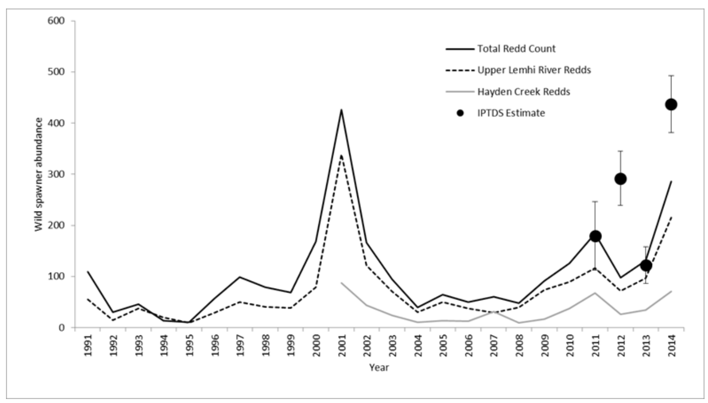


Figure 49. Chinook salmon adult abundance trends in the Lemhi River from redd counts and escapement estimates using PIAs for run years 2011-2014.

Table 12. Lemhi River steelhead escapement estimates using PIT tags and the Lower Lemhi River PIA site.

Species	Population	Spawn Year	Est	SE	Lower 95% CI	Upper 95% CI	CV
Steelhead	Lemhi River	2010	501	78	370	667	0.154
Steelhead	Lemhi River	2011	300	53	198	396	0.174
Steelhead	Lemhi River	2012	251	47	166	355	0.185
Steelhead	Lemhi River	2013	287	41	212	369	0

O. mykiss in the Lemhi River have three distinct life stages: anadromous, fluvial, and resident (tributary). For this report we will not try to elucidate the relationships between them and restoration projects; however, most tributaries with large numbers of resident rainbow do provide smolts into the FCRPS.

Estimating Carrying Capacity

Under baseline conditions, we assumed spawner capacity in the lower Lemhi River section was zero for both Chinook salmon and *O. mykiss* because redds have not been observed for either species in this section. Similarly, we set capacity for spawners and eggs in the Salmon River to zero because we were not interested in how potential fish from this area are contributing to the Lemhi River population, but the consequences of fish wintering and rearing in this section for the population could be significant.

To estimate parr and presmolt capacity, we used quantile regression forests (Meinshausen 2006) to determine how fish densities changed with various habitat metrics (for methodology see Chapter 6) using fish and habitat data collected by ISEMP and CHaMP from 379 site-visits in the Lemhi between 2009 and 2014. Landscape and environmental factors associated with different spatial and temporal scales can influence the distribution, abundance, and survival of each life stage so we used the predicted 90th quantile from the fitted quantile regression forest model to estimate capacity, and applied those estimates accordingly to strata and channel type continuously through the watershed using the ground-truthed Beechie channel type GIS layer and CHaMP habitat estimates (Figure 50 and Figure 51). Total capacity by species per stream was calculated by multiplying the stratum's land use/channel type length (Table 14) by the carrying capacity estimate (Table 15 and Table 16).

Table 13. Lemhi life cycle model input variables and the associated data sources.

Sub-basin	Life stage	Chinook		Steelhead		Reference
		Survival	Capacity	Survival	Capacity	
Hayden	Egg	0.423	272.23	0.68	272.33	Gebhards 1961; Bjornn 1978
Hayden	Fry	0.49	3.04	0.29	2.46	Bjornn 1978
Hayden	Parr	0.5	0.174	0.54	0.1	
Hayden	Age-0 presmolt	0.7	0.174	0.64	0.051	
Hayden	Age-1+ presmolt	0.7	0.174	0.87	0.051	
Up Lemhi	Egg	0.423	660.18	0.68	660.18	Gebhards 1961; Bjornn 1978
Up Lemhi	Fry	0.49	3.04	0.29	2.46	Bjornn 1978
Up Lemhi	Parr	0.53	0.365	0.51	0.2	
Up Lemhi	Age-0 presmolt	0.51	0.365	0.61	0.1	
Up Lemhi	Age-1+ presmolt	0.7	0.365	0.81	0.1	
Low Lemhi	Egg	0	0	0	0	
Low Lemhi	Fry	0.49	1.05	0.29	2.46	Bjornn 1978
Low Lemhi	Parr	0.69	0.87	0.33	0.035	
Low Lemhi	Age-0 presmolt	0.76	0.187	0.66	0.035	
Low Lemhi	Age-1+ presmolt	0.7	0.187	0.91	0.035	
Salmon	Parr	0	1.08	0.23	1.08	
Salmon	Age-0 presmolt	0.384	0.4	0.66	0.4	
Salmon	Age-1+ presmolt	0.735	0.4	0.79	0.4	
Snake/Columbia	Juvenile dam passage	0.53	infinity	0.39	infinity	Haeseker et al. 2012
Estuary/Ocean	Early adult	0.061	infinity	0.12	infinity	McClure et al. 2008 (cited in Honea et al 2009)
Ocean	Adult	0.8	infinity	0.8	infinity	McClure et al. 2008 (cited in Honea et al 2009)
Snake/Columbia	Adult dam passage	0.806	infinity	0.77	infinity	McClure et al. 2008 (cited in Honea et al 2009)

Figure 50. The relative ranking of habitat metrics associated with quantile regression forest (QRF) carrying capacity estimates for steelhead/rainbow juveniles in the Lemhi River.

Figure 51. The relative ranking of habitat metrics associated with quantile regression forest (QRF) carrying capacity estimates for Chinook salmon juveniles in the Lemhi River.

Table 14. Lemhi River sampling frame by stream (stratum), land-use type, and channel type (meters).

Stream	LandUse	Cas-cade	Con-fined	Island Braid-ed	Meander-ing	Plane-bed	Pool-riffle	Step-pool	Straig ht	Total
Agency	SubTotal			112		5,489	11,341	1,426		18,368
	Private			112		2,192	7,746	130		10,180
	Public					3,296	3,596	1,296		8,188
Big Eightmile	SubTotal				64	8,364	1,653			10,081
	Private				64	7,413	1,238			8,715
	Public					951	415			1,366
Big Springs	SubTotal						7,796			7,796
	Private						7,796			7,796
Big Timber	SubTotal		457	8,000	1,462	9,117	8,971	255	2,243	30,505
	Private			6,493	1,462	2,072	1,654		1,143	12,825
	Public		457	1,507		7,044	7,317	255	1,100	17,680
Bohannon	SubTotal	1,237		73		7,294	1,093	5,591		15,288
	Private			73		6,754	964	3,850		11,641
	Public	1,237				540	129	1,741		3,647
Canyon	SubTotal				198	5,344	15,494			21,035
	Private				198	3,335	3,807			7,341
	Public					2,008	11,686			13,695
Hawley	SubTotal				3,740	6,918	12,365	293		23,316
	Private				3,740	198	2,472			6,410
	Public					6,720	9,893	293		16,906
Hayden	SubTotal		2,001	6,853		6,577	2,089	1,747	5,148	24,414
	Private		159	6,317		431			3,679	10,586
	Public		1,842	536		6,146	2,089	1,747	1,468	13,828

Stream	LandUse	Cas-cade	Con-fined	Island Braid-ed	Meander-ing	Plane-bed	Pool-riffle	Step-pool	Straig ht	Total
Kenney	SubTotal			238		4,899	1,334	1,961		8,433
	Private			238		1,853	815			2,905
	Public					3,047	520	1,961		5,527
Lee	SubTotal						1,103			1,103
	Private						1,103			1,103
Lemhi Mainstem	SubTotal		288	63,587	19,877		6,033		1,753	91,538
	Private		288	63,258	19,877		6,033		1,728	91,183
	Public			329					25	354
Little Springs	SubTotal				6,962					6,962
	Private				6,962					6,962
Mill	SubTotal						1,111			1,111
	Private						1,111			1,111
Pattee	SubTotal	124		71		6,003	4,541	1,995		12,734
	Private			71		243	1,014			1,328
	Public	124				5,760	3,527	1,995		11,406
Texas	SubTotal				14,679					14,679
	Private				14,679					14,679
Wimpey	SubTotal	4,431		62		4,413	990	3,444		13,340
	Private			62		3,895	990	646		5,593
	Public	4,431				518		2,798		7,746

Table 15. Chinook salmon stream carrying capacity (fish/meter) by channel type.

Stream	Cascade	Confined	Island Braided	Meandering	Plane-bed	Pool-riffle	Step-pool	Straight
Agency			4.85		1.71	2.12	0.89	
Big Eightmile				4.45	3.11	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	

Table 16. Steelhead/rainbow stream carrying capacity (fish/meter) by channel type.

Stream	Cascade	Confined	Island braided	Meandering	Plane-bed	Pool-riffle	Step-pool	Straight
Agency			3.085408		1.897663	1.966667	1.923932	
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 Eight-mile	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	

Survival

We used a combination of literature derived values and empirical data to estimate juvenile survival, whereas adult survival was taken strictly from literature (Table 17). Chinook salmon and steelhead egg and fry survival were set as constants across all subpopulations (see below). Chinook egg survival was taken from Gebhard (1961)[3]. For survival from the parr, age-0 and -1+ presmolts to the next life stage, we used passive detections from fish tagged from 2009-2013 (n=52,664 for Chinook; n=41,008 for steelhead). Fish in these stages were placed into brood year. For Chinook salmon, brood year was based on size and timing of initial tag. This was justified because >99% of juveniles leave the Lemhi River at age-1. For *O. mykiss*, we used age data determined by scale analyses combined with size and timing of the tagging event. Survival at the smolt stage, identified as downstream migration through Snake and Columbia River dams, was taken from literature values. We considered survival for adults by ocean age.

Juvenile migrant survival was estimated using TribPit (Lady et al. 2014), which accounts for the probability that a fish may reside in a watershed for multiple years before migrating downstream. We estimated the probability of a Chinook juvenile migrating and surviving to four locations: location of release (Hayden Creek, upper Lemhi, lower Lemhi) to the next detection location. This equates to using either the RST or PIAs at mouth of Hayden creek or bottom of upper mainstem Lemhi, depending on the sub-population, or the lower Lemhi RST or PIAs. Lower Granite Dam was the last location used for estimation and all fish were defined as smolts at that location regardless of migration timing. The upper mainstem Lemhi and Hayden Creek had similar survival estimates for area and life-stage except for overwinter survival (Table 18). Upper Lemhi fish overwinter survival was estimated at 5.7% (average over three years) versus Hayden Creek Fish at 18.1%. This difference could be significant for planning future restoration efforts.

[3]This estimate was taken from a single redd in the Lemhi River

Table 17. Lemhi Watershed Model input variables and the associated data sources.

Subbasin	Life Stage	Survival	Capacity	Survival	Capacity	Reference
Hayden	Egg	0.423	272.23	0.68	272.33	Gebhards 1961; Bjornn 1978
Hayden	Fry	0.49	3.04	0.29	2.46	Bjornn 1978
Hayden	Parr	0.5	0.174	0.54	0.1	
Hayden	Age-0 presmolt	0.7	0.174	0.64	0.051	
Hayden	Age-1+ presmolt	0.7	0.174	0.87	0.051	
Up Lemhi	Egg	0.423	660.18	0.68	660.18	Gebhards 1961; Bjornn 1978
Up Lemhi	Fry	0.49	3.04	0.29	2.46	Bjornn 1978
Up Lemhi	Parr	0.53	0.365	0.51	0.2	
Up Lemhi	Age-0 presmolt	0.51	0.365	0.61	0.1	
Up Lemhi	Age-1+ presmolt	0.7	0.365	0.81	0.1	
Low Lemhi	Egg	0	0	0	0	
Low Lemhi	Fry	0.49	1.05	0.29	2.46	Bjornn 1978
Low Lemhi	Parr	0.69	0.87	0.33	0.035	
Low Lemhi	Age-0 presmolt	0.76	0.187	0.66	0.035	
Low Lemhi	Age-1+ presmolt	0.7	0.187	0.91	0.035	
Salmon	Parr	0	1.08	0.23	1.08	
Salmon	Age-0 presmolt	0.384	0.4	0.66	0.4	
Salmon	Age-1+ presmolt	0.735	0.4	0.79	0.4	
Snake/Columbia	Juvenile dam passage	0.53	infinity	0.39	infinity	Haeseker et al. 2012
Estuary/Ocean	Early adult	0.061	infinity	0.12	infinity	McClure et al. 2008 (cited in Honea et al 2009)
Ocean	Adult	0.8	infinity	0.8	infinity	McClure et al. 2008 (cited in Honea et al 2009)
Snake/Columbia	Adult dam passage	0.806	infinity	0.77	infinity	McClure et al. 2008 (cited in Honea et al 2009)

Table 18. Chinook salmon juvenile survival estimates by Lemhi River sub-population; GRA - Lower Granite Dam.

Upper Lemhi Mainstem Sub-Population	2012	2009	2008	Average
Parr	58.0%			
Parr/Presmolt to Mouth of Lemhi	84.5%	96.0%	100.0%	93.5%
Over - Winter Lower Lemhi Reach	7.5%	7.4%	2.2%	5.7%
Spring to Mouth of Lemhi	82.2%	91.9%	76.8%	83.7%
Fall Migrant to GRA	38.0%	34.3%	28.6%	33.6%
Spring Migrant to Granite (both rearing areas)	70.7%	75.4%	48.6%	64.9%
Spring to GRA, Upper Lemhi Over-Winter	73.7%	75.6%	52.7%	67.3%
Spring to GRA, Lower Lemhi Over-Winter	31.4%	78.7%	75.1%	61.7%
Hayden Creek Sub-Population	2012	2009	2008	Average
Parr	60.1%			
Parr/Presmolt to Mouth of Lemhi	47.3%	62.9%	60.1%	56.8%
Over - Winter Lower Lemhi Reach	18.7%	10.9%	24.8%	18.1%
Spring to Mouth of Lemhi	78.0%	68.2%	100.0%	82.1%
Fall Migrant to GRA	31.5%	28.5%	38.1%	32.7%
Spring to GRA, Upper Lemhi Over-Winter	74.1%	82.2%	94.8%	83.7%
Spring to GRA, Lower Lemhi Over-Winter	79.2%	66.5%	63.7%	69.8%

Fish movement/migration

We can leverage the behavior of migratory fish and allow for movement between strata as juveniles or adults because we are using 16 strata. After iterating the Beverton-Holt equations across all life stages, a fraction of the population at the fry, parr, pre-smolt, and spawner life stages can be re-assigned to another site using matrices of migration probabilities. Adult fish are directed to return to their natal sites to spawn. This functionality allows for colonization of new habitat, changes in spatial distribution that might accompany habitat restoration, and/or general movement observed at different life stages.

We observed three distinct movement patterns for Chinook salmon, which we account for in the model from estimates of migrants at screw traps and PIAs. Chinook migrant estimates by life-stage are shown in Table 19. Table 20 shows the temporal distribution of juvenile migrates by percent of migrants estimates; however, to understand the percent of total migrant population before mortality we adjusted the calculations for survival (Table 21). These data are consistent with what Bjornn (1978) observed. First, a group of parr would leave the Lemhi River during July and August and begin migrating to the ocean, overwintering in the mainstem Salmon River downstream of the Lemhi. Thus far, none of the tagged fish that migrate during this period have returned as adults. Second, "presmolts" would migrate downstream to overwinter, and then continue their migration to the ocean the following spring. This ranged from fish leaving tributaries to the mainstem Lemhi River or continuing to the Salmon River, and presmolts from the mainstem Lemhi River migrating to the Salmon River. Third, age-1 smolts would leave the Lemhi River in March and April, and travel quickly through the downstream dams. These three patterns are consistent with those observed in the Pahsimeroi River, Idaho, (Copeland and Venditti 2009). Observations of fry and parr at the downstream trap are usually <0.01% of the population, so we assumed fry or parr did not migrate into the Salmon River.

Table 19. Rotary screw trap Chinook salmon juvenile migrant estimates for three Lemhi River locations. NR = data has not been reported, NP=life-stage not present, Lower Lemhi Trap did not provide estimates for 2009-2011.

Location	Life Stage	Brood Year					
		2013	2012	2011	2010	2009	2008
Upper Lemhi	Adult	97	75	116	89	74	39
	Fry	NR	445	1372	799	3796	0
	Parr	NR	461	354	783	444	15
	Presmolt	NR	6128	26858	18818	39634	5905
	Smolt	NR	5167	5387	2654	3710	1143
	Total Migrants		12201	33971	23054	47584	7063
Hayden Creek	Adult	34	26	68	37	17	9
	Fry	NR	16447	15507	13763	7468	0
	Parr	NR	665	1571	1657	1953	22
	Presmolt	NR	9476	17501	16739	8053	10590
	Smolt	NR	1468	947	826	983	1172
	Total Migrants		28056	35527	32984	18457	11784
Lower Lemhi	Adult	131	101	184	126	91	48
	Fry	NP	NP				NP
	Parr	NP	NP				NP
	Presmolt	17,056	14,583				11,623
	Smolt	7,334	6,519				4,999
	Migrants/Female	372	418				693

Table 20. Juvenile Chinook salmon migrants as a percent of total life-history population upstream of rotary screw trap in the Lemhi IMW, 2008 – 2012.

Location	Life Stage	Year					
		2012	2011	2010	2009	2008	Average
Upper Lemhi	Fry	1%	1%	1%	2%		1%
	Parr	1%	0%	2%	0%		1%
	Presmolt	38%	72%	78%	84%	72%	69%
Hayden Creek	Fry	25%	17%	16%	15%		18%
	Parr	3%	4%	5%	9%		5%
	Presmolt	77%	90%	91%	81%	82%	84%

Table 21. Percent of sub-population life history estimated to migrate past rotary screw trap in the Lemhi IMW.

Location	Life Stage	Percent of Sub-population					
Upper Lemhi	Fry		3.6%	4.0%	3.5%	8.0%	0.0%
	Parr		3.8%	1.0%	3.4%	0.9%	0.2%
	Presmolt		50.2%	79.1%	81.6%	83.3%	83.6%
	Smolt		42.3%	15.9%	11.5%	7.8%	16.2%
Hayden Creek	Fry		58.6%	43.6%	41.7%	40.5%	0.0%
	Parr		2.4%	4.4%	5.0%	10.6%	0.2%
	Presmolt		33.8%	49.3%	50.7%	43.6%	89.9%
	Smolt		5.2%	2.7%	2.5%	5.3%	9.9%
Lower Lemhi River	Presmolt	69.9%	69.1%				69.9%
	Smolt	30.1%	30.9%				30.1%

An important difference between Hayden Creek and the upper Lemhi River migrants is the dominate life-stage at migration. The majority of migrants are fry from Hayden Creek and the majority of migrants for the upper Lemhi River are pre-smolts. Important for restoration planning, Hayden Creek fry spend more time than other migrants in the lower mainstem Lemhi River, which has lower survival rates than rearing in Hayden Creek or upper Lemhi.

Movement within and between strata in the Lemhi is very common and we illustrate the percent of tagged steelhead and Chinook, adjusted for observation detectability, in Table 22 and Table 23, respectively. Rearing habitat for Chinook is extremely important in the restoration actions in the Lemhi, and the existing distribution was updated to the tributary upstream extents. These data were utilized in the migration matrix of the life cycle model. Chinook abundance estimates (when available) and observed numbers are shown in Table 24. This information is extremely important when validating juvenile rearing and migration into recently reconnected tributaries.

Table 22. Percent of steelhead/rainbow trout that immigrate or emigrate from Lemhi tributary streams and mainstem. Total tags are all tags placed in fish during that calendar year. E = Emigration, I = Immigration.

Stream	Calendar Year											
	2009		2010		2011		2012		2013		2014	
	E	I	E	I	E	I	E	I	E	I	E	I
Agency	0.0%	0.0%	0.0%	0.0%	1.1%	0.0%	0.0%	0.0%	0.4%	0.0%	1.4%	0.0%
Big Eight-mile	0.0%	0.0%			0.0%	0.0%			0.0%	0.0%	0.0%	0.0%
Big Springs					3.6%	0.2%	2.4%	1.0%	1.4%	0.0%	1.3%	0.0%
Big Timber	0.0%	0.0%	23.2%	0.0%	18.2%	0.6%	26.5%	0.0%	2.4%	0.1%	0.0%	0.0%
Bohannon			0.0%	0.0%	21.0%	8.4%	19.0%	4.8%	6.8%	1.8%	6.8%	0.0%
Canyon	0.9%	0.0%	6.4%	0.0%	4.7%	0.0%	23.8%	0.6%	6.2%	0.0%	6.3%	0.0%
Hawley	0.0%	0.0%	2.9%	2.9%	0.0%	0.0%	0.0%	0.0%	1.6%	0.0%	0.6%	0.0%
Hayden	27.0%	4.4%	50.6%	9.2%	51.8%	6.4%	64.4%	7.3%	45.1%	8.8%	32.9%	6.6%
Kenney	3.6%	3.6%	2.3%	0.9%	5.0%	5.0%	5.5%	5.1%	2.4%	1.7%	0.0%	0.0%
Lee											7.0%	0.0%
Lemhi	0.0%	7.6%	0.0%	5.6%	0.0%	5.2%	0.0%	5.9%	0.0%	20.5%	0.0%	8.8%
Little Springs	0.0%	0.0%			52.0%	0.0%	41.0%	2.3%	26.2%	1.0%	28.7%	0.0%
Texas					0.0%	0.0%	0.0%	0.0%				
Wimpey	4.2%	4.2%	3.0%	3.0%	1.8%	1.8%	0.0%	0.0%	16.0%	3.3%	22.3%	0.0%

Table 23. Percent of parr tagged in the Upper Lemhi and Hayden Creek that were observed/detected in subsequent tributaries. Total tags fish captured include all methods. Roving Tags are just those tags placed by electrofishing or seining. Juvenile migrations are from rotary screw traps on the Upper Mainstem Lemhi or Hayden Creek. NR = not yet reported.

Stream	Calendar Year					
	2009	2010	2011	2012	2013	2014
Big Eightmile						0.14%
Big Springs				0.39%	0.15%	0.66%
Big Timber				0.67%	0.03%	0.09%
Bohannon						0.02%
Canyon				0.60%		0.04%
Kenney				0.03%		
Lee						0.05%
Little Springs				0.10%	0.16%	0.02%
Wimpey				0.01%	0.05%	
Total Tags available	4,016	10,429	8,401	7,198	8,764	8,020
Roving Tags available	247	1,786	972	1,195	622	458
Juvenile Migration Estimates	18,847	66,041	56,039	69,498	40,257	NR

Table 24. Observed and estimated numbers of Chinook salmon juveniles that migrate into reconnected tributaries. Shaded areas denote tributary reconnection status: dark - connected; light- seasonally connected, none - not connected.

Stream	Season	Calendar Year					
		2009	2010	2011	2012	2013	2014
Big Eightmile	SubTotal						11
	Summer						11
Big Springs	SubTotal				28	13	53
	Fall			842*		3	336*
	Spring						1
	Summer				224*	10	
Big Timber	SubTotal				48	3	7
	Fall				47		
	Summer				1	3	7
Bohannon	SubTotal						2
	Summer						2
Canyon	SubTotal				43		3
	Fall				39		1
	Summer	141*		58*	71*		2
Kenney	SubTotal				2		
	Fall				2		
Lee	SubTotal						4
	Summer						4
Little Springs	SubTotal				7	14	2
	Fall				6	10	
	Summer				1	4	2
Wimpey	SubTotal				1	4	
	Summer				1	4	

Results from Modeling Different Restoration Scenarios

Results from previous model runs used very similar input scenarios and data as described here and illustrated the immediate success of individual projects (Little Springs Creek Restoration Project, and a large upper Lemhi mainstem channel reconstruction and side channel project). The Little Springs Restoration Project estimated that Chinook survival increased from 29% to 80% for migrants to the mouth and estimated that steelhead/rainbow populations increased from 110 to 1,297 over 3 years (QCI 2014). This result is very encouraging and is evident from base stream flow where the stream outflow stays hydraulically connected throughout the year.

We have also shown that at the population scale, to date, the ISEMP life cycle model predicts that habitat restoration actions completed in the Lemhi River from 2009-2013 will be adequate to exceed the 4% increase in freshwater productivity for steelhead identified in the supplemental BiOp. However, the model predicts that the 7% freshwater productivity increase target for spring/summer Chinook salmon will not be achieved (Figure 52). To address this shortfall, we developed a number of potential restoration scenarios to identify actions that would meet or exceed survival targets for spring/summer Chinook salmon. Modelled actions included a suite of site-specific actions such as channel reconstruction and tributary reconnections in Texas Creek. Together, these actions resulted in model estimates that would meet the survival target for spring/summer Chinook salmon (Figure 53).

We modeled several new scenarios using the updated carrying capacity estimates and migration estimates to understand their effect on mainstem entrainment from diversions, increases in productivity in the lower mainstem Lemhi to improve overwinter and migration survivals, and the addition of new spawning and rearing areas (either through restoration actions decreasing water temperatures or tributary reconnection). The new scenarios are focused on Chinook salmon (Table 25). Figure 54 shows the sample frame and extent of tributaries modeled.

Two scenarios focused only on the effect of increasing juvenile rearing area in the Lemhi, assuming 1) a change from base conditions to current conditions, and 2) all potential streams in sampling frame in historical condition. For scenario 1, escapement increased by 7.8% and productivity increased by 4%. Scenario 2 showed an increase in escapement by 12.2% and 6% in-

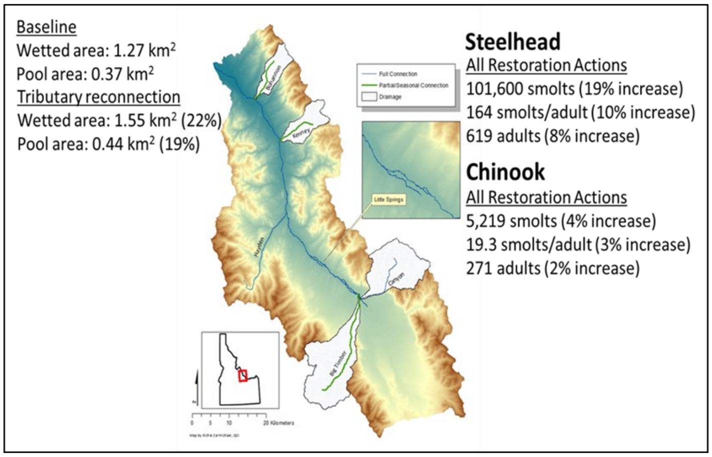


Figure 52. Estimated change in habitat available to anadromous salmonids as a result of tributary reconnections in the Lemhi, and estimated change in smolt abundance, freshwater productivity (smolts/female spawner), and adult escapement accompanying ch1anges in habitat quantity and quality from Lemhi River habitat restoration for actions from 2009 to 2012.

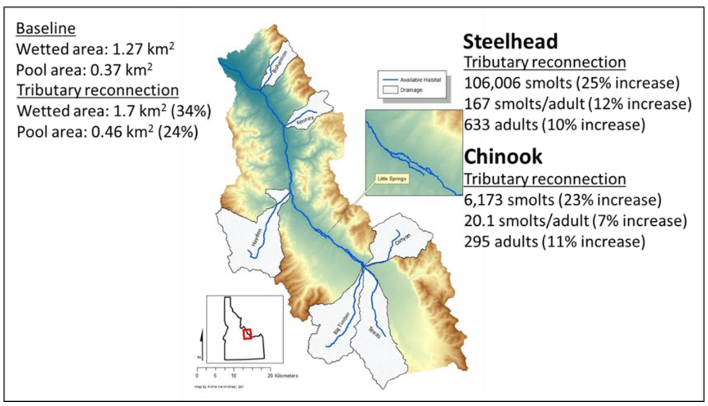


Figure 53. Estimated change in available habitat and subsequent estimates of smolt abundance, freshwater productivity (smolts/ female) and adult escapement from in-stream, habitat restoration and the reconnection of Texas Creek in addition to currently reconnected Lemhi River tributaries.

crease in productivity. Three scenarios were focused solely on improving spawning conditions including 1) improving in-stream conditions (decreased water temperature) in Big Springs Creek to allow Chinook spawning, 2) improving lower river channel conditions to half of the spawning potential of the upper river, and 3) a combination of both 1 and 2. All three scenarios improved escapement numbers, but productivity only increased by the addition of spawning in Big Springs.

By changing both spawning and rearing conditions, all three scenarios resulted in significant escapement and productivity increases and all met the population restoration goals. The biggest difference was attributed to assuming all areas in the Lemhi watershed were accessible; however, since it is unlikely that Chinook salmon juveniles would use all ecological conditions, nor would adults spawn in the small tributaries, this scenario was modeled for comparison only.

In order to understand the influence of mainstem productivity, we modeled two scenarios: 1) decrease in Chinook juvenile migrant survival by 25%, and 2) increase in lower mainstem Lemhi survival by 10%. This can be interpreted as a change in ecological conditions due to temperature and/or instream restoration to improve habitat complexity, or the direct influence of screening diversions on improving survival of migrants by decreasing entrainment. Both scenarios had significant influence on escapement and productivity, suggesting that mainstem conditions are the most important drivers to Chinook population stability. Most Chinook spend a majority of their freshwater life in the mainstem Lemhi, making conditions in the mainstem integral for any population improvements. Figure 55 and Figure 56 illustrate the current Chinook salmon distribution determined by roving and/or mobile surveys. Most notable are the increases in rearing areas in Little Springs, Big Eightmile, Lee, Big Timber and Canyon Creeks after streams were at least partially hydraulically reconnected to the mainstem Lemhi River. We utilized this information to determine the extent of existing use of Chinook salmon rearing in the Lemhi.

Conclusions

Our results from the last 6 years of monitoring have shown important changes in the Lemhi Chinook population due to restoration actions. Most notable is the increase in juvenile rearing habitat by opening up additional areas. The potential impacts of future restoration should focus on lower mainstem Lemhi restoration efforts to improve conditions for rearing and overwinter survival. Predictions showed the greatest population changes are influenced by a combination of improved conditions for rearing and spawning. Currently there are only a few potential areas that Chinook adults can access; however, water quality (Big Springs, lower mainstem Lemhi), channelization (lower mainstem Lemhi) and the lack of hydraulic connection (Texas Creek) preclude additional production from spawners.

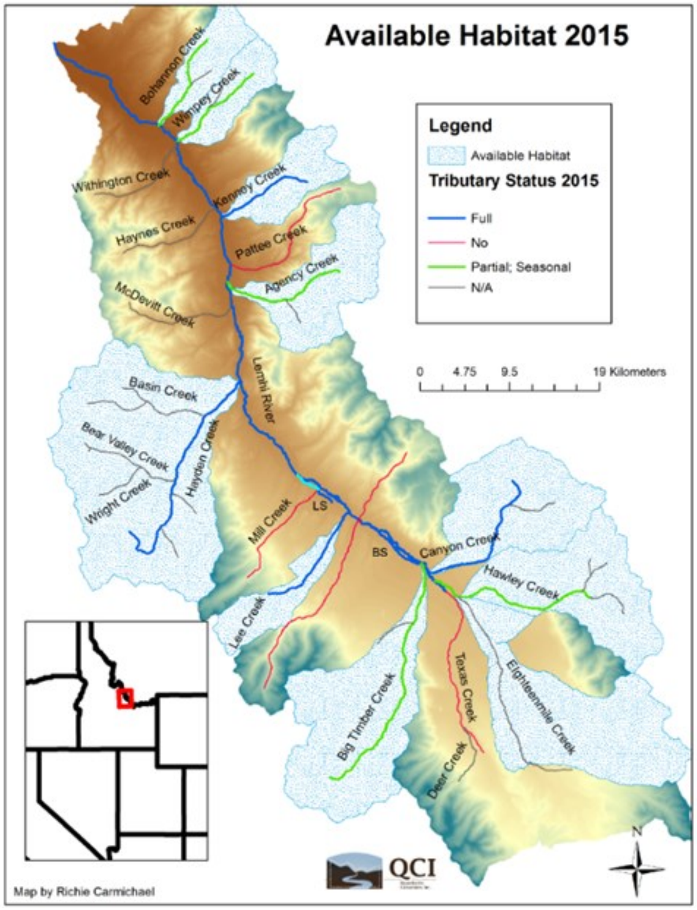


Figure 54. Lemhi River tributaries used for modeling restoration changes to the Chinook population. Tributary status: Full=hydraulic connection year-around to mainstem Lemhi; No= no hydraulic connection, Partial or Seasonal - on downstream area of tributary or during specific flow regimes the stream hydraulically connected; N/A = Not in monitoring or restoration design.

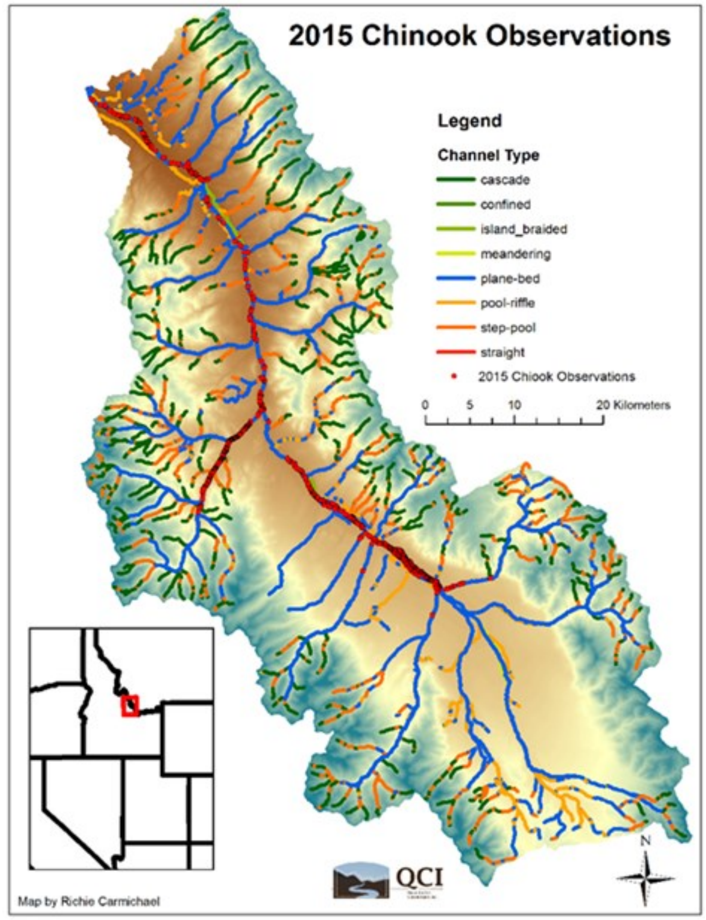


Figure 55. Chinook juvenile distributions in the Lemhi River and tributaries from roving surveys and mobile detections from 2009-2015.

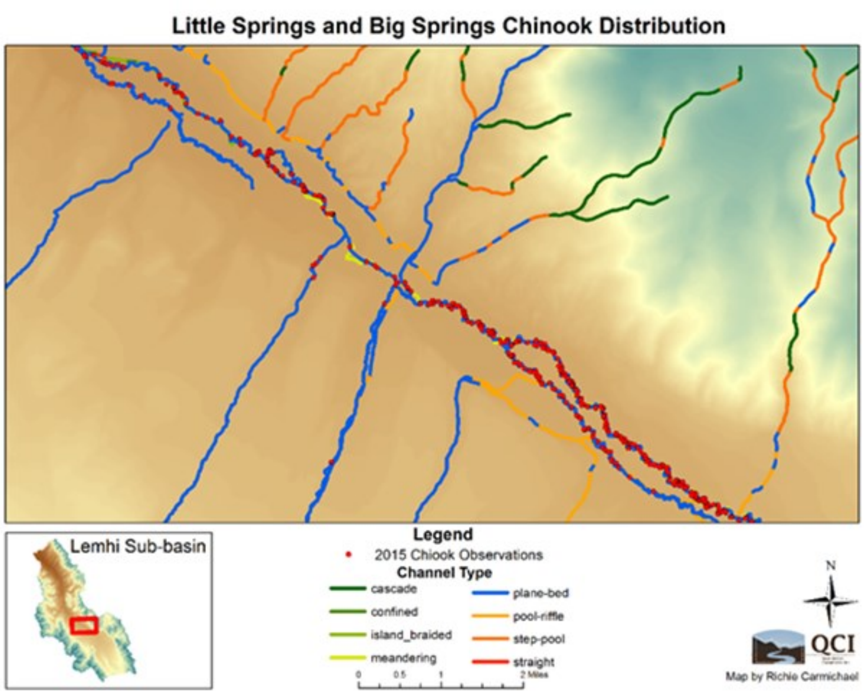


Figure 56. Chinook juvenile distributions in Little Springs Cr and Big Springs Creek and surrounding tributaries from roving surveys and mobile detections from 2009-2015.

Table 25. Habitat restoration scenarios for Lemhi watershed model runs. Stream types are Base= connected before restoration started; Accessible = where Chinook salmon juveniles can currently use; Potential = all areas with habitat suitable for Chinook salmon, if hydraulically connected an individual fish could access.

Stream Type	Target Streams	Direction	Change from Base	Life Stage	Spawners	Escapement	Smolts/ Adult
Historically Connected (Base)	Mainstem Lemhi, Hayden Creek, Big Springs Creek (no spawning)	None			706		
Accessible	Current Chinook Juvenile Distribution	Increase	21,962 m	Rearing	761	7.8%	4%
All potential streams	Full Sample Frame; all streams	Increase	163,110 m	Rearing	792	12.2%	6%
Accessible	Big Springs	Increase	7,796 m	Spawning	780	10.5%	6%
Accessible	Lower Mainstem	Increase	47,119 m	Spawning	795	12.6%	4%
Accessible	Lower Mainstem, Big Springs	Increase	54,915 m	Spawning	802	13.6%	7%
Accessible	Texas	Increase	14,679 m	Spawning, Rearing	869	23.1%	7%
Accessible	Lower Mainstem, Big Springs, Texas	Increase	65,594 m	Spawning, Rearing	905	28.2%	8%
All potential streams	Full Sample Frame; all steams*	Increase	163,110 m	Spawning, Rearing	1,598	126.3%	12%
	*Did not model spawning in Lower Mainstem						
Accessible	Upper and Lower Mainstem Productivity	Decrease	-25%	Survival	86	-87.8%	3%
Accessible	Lower Mainstem Productivity	Increase	10%	Survival	802	13.6%	6%

Recommendations

We recommend that based on the promising results that are coming out of the ISEMP IMWs that we stay the course and continue to implement and intensively monitor in the John Day, Entiat and Lemhi IMWs. Given the difficulty of effecting significant habitat change that results in responses from the fish population, we believe that the Bridge Creek IMW and Little Springs Creek in the Lemhi IMW have shown remarkable results that managers and policy makers could use to guide implementation in other watersheds where incision or lack of tributary connection are issues. The Entiat IMW is half way through the planned implementation schedule and results to data suggest that actions need to aim for creating immediate and large changes in the instream habitat to effect a significant response. The Lemhi IMW has also demonstrated the power of the ISEMP life cycle model to predict if proposed actions will produce the desired results, or to suggest alternative actions that would reach fish population targets. Based on comments from reviewers a comparison of the different approaches used and identifying the level of effort that is appropriate for monitoring responses at different spatial scales would be useful.

CHAPTER 2: HABITAT STATUS AND TRENDS

Introduction

CHaMP is expressly designed to monitor the status and trends of freshwater tributary habitat for salmonids. In 2014, we completed our fourth year of monitoring (Figure 57). Crews applying the protocol visited over 400 sites in 2014 (Table 26). Since 2011, other programs and collaborators have also been applying the protocol (Figure 58, Table 27). For example, it is being used by ISEMP and BPA's Action Effectiveness Monitoring (AEM) program under different study designs, and to support other watersheds' sampling objectives.

In this chapter we present a summary of our analysis methodology and select results based on 2011-2014 data, at the watershed spatial scale. With only four years of data it is not yet possible to distinguish long-term linear trends from short term year-year aberrations: any statistically significant metric change should merely be interpreted as a significant difference across the four years sampled thus far, and should not be interpreted as a likely indication of future trends or used to predict future status. Given the amount of data CHaMP generates annually, we cannot present status and trends results for all metrics in all CHaMP watersheds at all spatial scales herein. Complete results are available at https://isemp.egnyte.com/dl/qKVQ8KYvbo. Requests for status and trends for additional metrics and/or spatial domains may be made to South Fork Research (Matt Nahorniak; matt@southforkresearch.org).

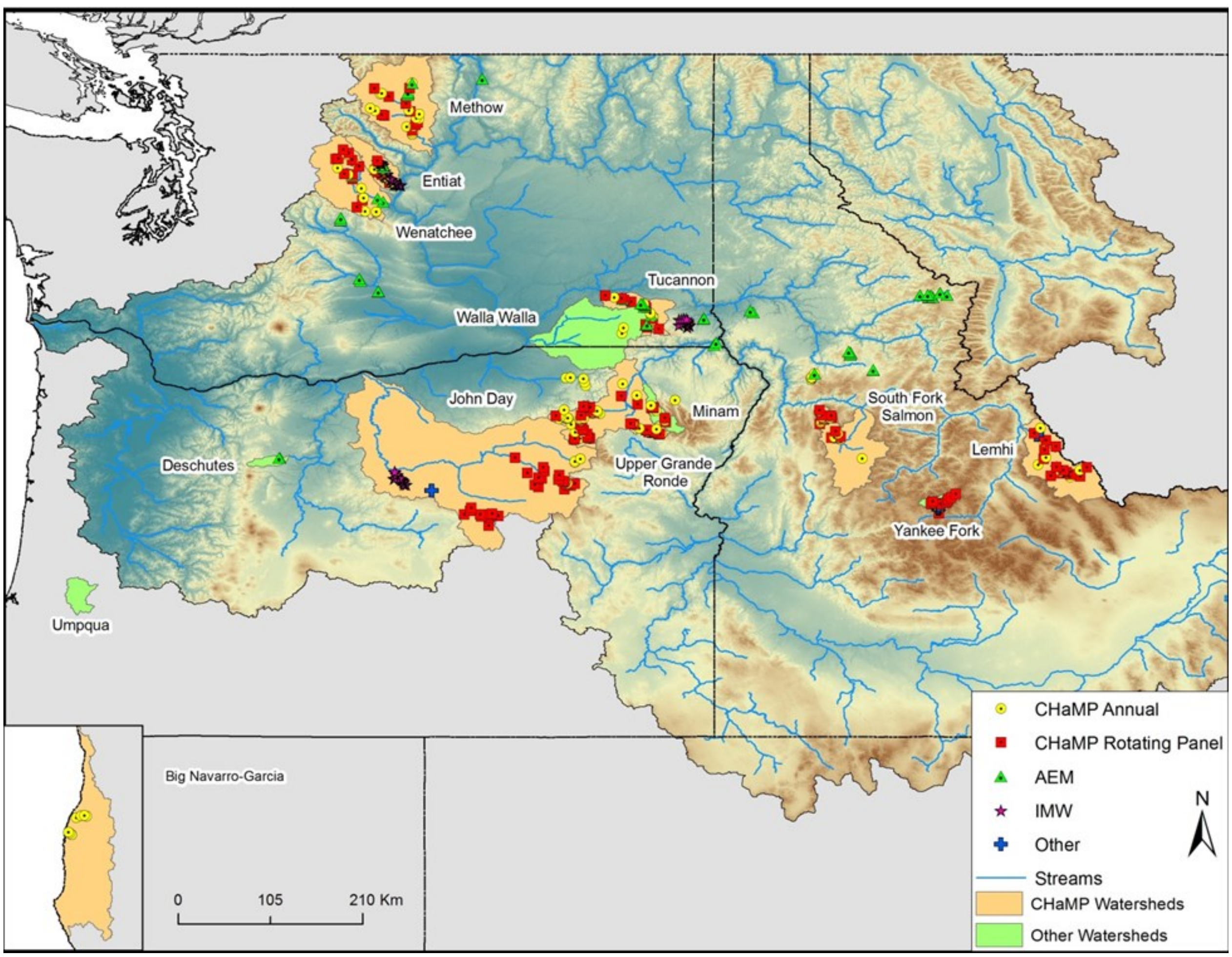


Figure 57. Locations and types of sites visited by crews implementing the CHaMP protocol in 2014. ISEMP and BPA's Action Effectiveness Monitoring (AEM) program are using the protocol under different study designs; the CHaMP protocol also supports other watersheds' sampling objectives. For example, in 2014 the California Department of Fish and Wildlife (CDFW) and the Pacific States Marine Fisheries Commission (PSMFC) continued their application of CHaMP methods in the coastal Big Navarro-Garcia watershed.

Table 26. Number of sites visited by crews implementing the CHaMP protocol in 2014.

Watershed	Status and Trend	IMW	AEM	Other	2014 Visit Total
Asotin				18	18
Big Creek, CA				9	9
Entiat	16	33		5	54
John Day	29	25	2	3	59
Lemhi	23	3			26
Methow	25				25
Minam	10				10
South Fork Salmon	25				25
Tucannon	25			3	28
Upper Grande Ronde	57			10	67
Wenatchee	25				25
Yankee Fork	10			15	25
Other			34		34
2014 Visit Total	245	61	36	63	405

Table 27. Summary of unique site visits made by crews implementing the CHaMP protocol 2011 – 2014.

Watershed	Status and Trend	IMW	AEM	Other	Four-Year Total
Asotin				22	22
Big Creek CA				39	39
Entiat	37	62		5	104
John Day	165	51	2	3	221
Lemhi	105	11			116
Methow	45			4	49
Minam	15				15
South Fork Salmon	55				55
Tucannon	49				49
Upper Grande Ronde	121			10	131
Wenatchee	54				54
Yankee Fork	20			15	35
Other			58	6	64
Four-year Total	666	124	60	104	954

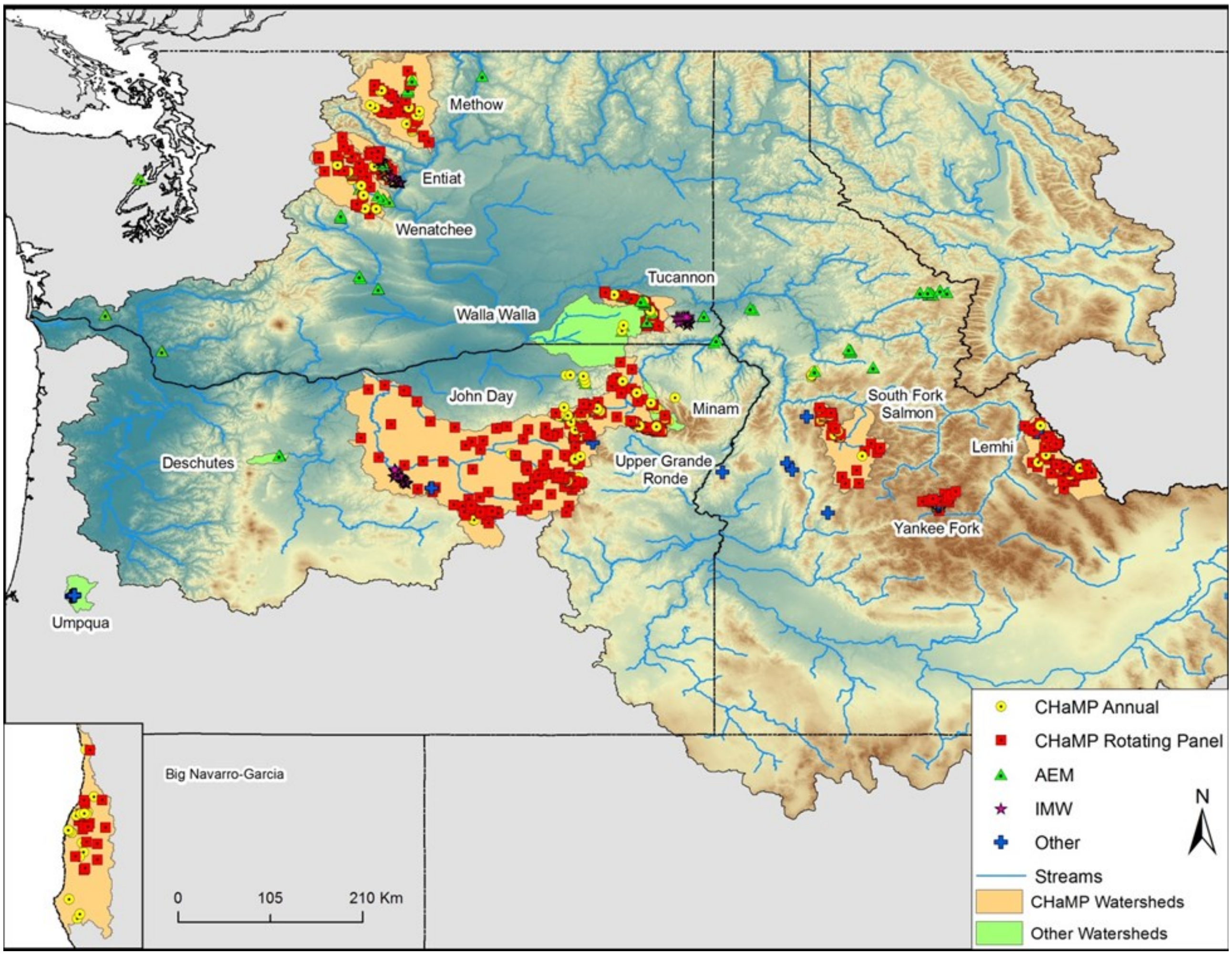


Figure 58. Locations of unique sites visited by crews implementing the CHaMP protocol from 2011-2014.

Metric generation methods

CHaMP sampling designs incorporate spatially balanced, stratified random sampling, where (in most CHaMP watersheds) strata are defined as combinations of valley class (source, transport, or depositional) and ownership type (public or private). Within each stratum, equally probable, spatially balanced sampling is done. Sample inclusion probability may vary across the different strata.

Spatial balance in sample design is achieved via use of a GRTS sample selection algorithm (Stevens and Olsen 2004). In sampling of a spatial resource, sample points very close to each other tend to be more alike (spatially correlated), thus there is limited additional information content added to a sample when, for example, a second sample point occurs very close to an existing sample point. A spatially balanced sample tends to spread out the sample points more uniformly across space, increasing the amount of independent information present in each individual sample point. GRTS sampling, specifically, provides a spatially balanced sample while also maintaining the robust sample properties of simple random sampling. The spsurvey package (Kincaid and Olsen 2013) for the R statistical programming language is used to analyze status and trends for GRTS sampling performed by CHaMP. Spsurvey elegantly incorporates sample design into the analysis, and properly accounts for the GRTS sampling design and spatial autocorrelation estimates in the variance estimates.

For the purposes of CHaMP data analysis, we define ***status*** as the distribution of a CHaMP metric over a specified spatial domain and time range. For spatial domain, we here present selected results at the watershed level. However, status can also be defined at sub watershed levels (HUC5 within a watershed, individual tributary creeks, etc.) or across multiple watersheds (i.e. the entire interior Columbia basin covered by CHaMP sampling). Time ranges considered may include individual years, as well as time averaged status over each year completed thus far in CHaMP sampling (2011-2014). When estimating status over multiple years, we first average the metric of interest at the site level to obtain a single average response at each site over the time period of interest, then analyze the data using spsurvey using the single average response for each site.. ***Trend*** is defined as the average of site level linear trend over time, for a given metric, over a specified spatial domain. Currently we have four years of CHaMP data. For sites sampled annually, we typically have four visit year measurements – one for each year. For three year rotating panel sites, we have either one or two visit year measurements per sites. For all sites that contain more than one visit year, we can estimate a linear trend by regressing the metric as measured at each site against time (in years). Note that, at the site level, there is high uncertainty in a trend estimate made from a regression of either 2 or 4 data points. These individual site level trend estimates are then analyzed using spsurvey, just as is done for status, as described above, to estimate a distribution of trends across the spatial domain of interest.

Extreme caution should be applied when interpreting estimates of trend, given that only four years of data are available. Small year to year differences may show up as trends, but in reality these "trends" may only reflect short term aberrations year to year, rather than long term linear changes. With only

Table 28. Status and trend estimates for selected CHaMP metrics in the Wenatchee watershed. Full results are available at: https://isemp.egnyte.com/dl/qKVQ8KYvbo.

Metric	N	Status			Trend		
		Mean	95% LCB	95% UCB	Trend	95% LCB	95% UCB
Alkalinity	44	52.25	44.67	59.84	-4.00	-14.60	6.60
Bankfull Depth Avg	44	0.46	0.39	0.54	-0.01	-0.02	0.01
Bankfull Width Avg	44	11.80	9.01	14.59	-0.22	-0.40	-0.04
Discharge	44	1.79	1.36	2.22	-0.21	-0.68	0.26
Fast NonTurbulent Percent	44	14.33	9.66	18.99	-2.27	-5.51	0.98
Fast Turbulent Percent	44	47.90	41.91	53.88	0.56	-3.16	4.29
Fish Cover: Artificial	44	0.12	0.01	0.23	0.03	-0.02	0.07
Gradient	44	2.61	1.91	3.32	0.02	-0.01	0.05
Large Wood Frequency: Bankfull	44	35.93	27.83	44.03	5.49	1.59	9.39
Large Wood Frequency: Wetted	44	21.31	17.06	25.56	1.05	-1.08	3.18
Residual Pool Depth	44	0.50	0.37	0.63	0.03	-0.001	0.06
Sinuosity	44	1.23	1.18	1.28	0.00	-0.01	0.01
Slow Water Frequency	44	3.8	2.7	5.0	0.42	0.29	0.54
Slow Water Percent	44	30.02	24.60	35.43	3.12	1.19	5.04
Substrate < 2mm	40	17.96	12.46	23.45	1.78	0.14	3.42
Substrate < 6mm	40	31.72	19.96	43.49	2.21	-0.64	5.06
Substrate Est: Boulders	44	8.84	6.29	11.39	-0.58	-1.81	0.66
Substrate Est: Coarse and Fine Gravel	44	42.07	36.97	47.17	0.27	-0.85	1.39
Substrate Est: Cobbles	44	23.54	19.11	27.96	1.41	-0.25	3.06
Substrate Est: Sand and Fines	44	24.0	20.2	27.7	-1.18	-3.48	1.12
Substrate: D50	45	46.11	35.19	57.02	2.96	-2.66	8.58
Substrate: D84	45	121.28	96.32	146.25	5.77	-4.82	16.35
Thalweg Depth Avg	44	0.41	0.33	0.48	-0.02	-0.02	-0.01
Thalweg Site Length	44	258.2	212.1	304.3	1.3	-0.5	3.1
Wetted Depth SD	44	0.17	0.14	0.20	-0.01	-0.01	-0.01
Wetted Width To Depth Ratio Avg	44	28.57	24.78	32.35	0.29	-0.40	0.98
Fish Cover: Aquatic Vegetation	39	0.84	0.48	1.20	0.06	-0.68	0.80
Percent Undercut by Area	39	1.6	0.5	2.8	-0.08	-0.76	0.61
Percent Undercut by Length	33	3.36	2.05	4.68	0.55	-1.52	2.61
Substrate: Embeddedness Avg	35	9	5	12	2.8	0.4	5.2
Substrate: Embeddedness SD	35	10	7	13	1.8	-0.7	4.3

four years' of data, it is not possible to distinguish short term year-year aberrations from long term trends. Thus, any statistically significant trend observed to date should merely be interpreted as a significant difference across the four years sampled thus far, and should not be interpreted as a likely indication of future trends or used to predict future status. After a full nine years' worth of CHaMP sampling have been completed, we will have a significantly better ability to distinguish long term trends from year-year aberrations.

Status and Trends Results

The example status results in Table 28 represent the four year average status for each CHaMP metric at within the Wenatchee watershed. Non-zero trends (likely to be year-year aberrations rather than long term linear trends) are highlighted in blue. Results are also summarized for individual years, and within individual watersheds. Detailed results for Large Wood Frequency, Wetted, by year and by watershed (for select watersheds) are displayed by Table 29. Complete results for key CHaMP metrics, covering individual years, the average status and trend from 2011-2014, in all CHaMP watersheds, are available at https://isemp.egnyte.com/dl/qKVQ8KYvbo.

In addition to results in tabular form, we are able to generate plots showing status by year, by watershed, for each of the selected key CHaMP metrics. These plots include estimates of the mean by year as well as 95% confidence intervals for the mean for each metric. Complete status and trend results are not presented here; instead, example results are provided, as well as explanations of status and trend calculations that are being performed. Complete results in graphical form are available at https://isemp.egnyte.com/dl/qKVQ8KYvbo.

Examples included on the next page are for Large Wood Frequency: Wetted (Figure 59) and Substrate: D50 (Figure 60).

Table 29. Detailed Summary for Large Wood Frequency: Wetted, in selected watersheds. Full results are available at https://isemp.egnyte.com/dl/qKVQ8KYvbo.

			Status					Trend		
Population	**Year**	**N**	Mean	Std Error	CV	95% LCB	95% UCB	Trend	95% LCB	95% UCB
	2011	73	40.8	6.8	0.8	27.5	54.2			
	2012	52	32.5	3.9	0.8	24.8	40.2			
Entiat	2013	72	53.6	4.3	0.8	45.2	62.1			
	2014	46	30.2	3.7	0.9	22.9	37.5			
	Avg. of All Years	100	40.8	2.9	0.8	35.2	46.5	3.962	0.508	7.416
	2011	55	15.8	2.6	1.4	10.7	20.8			
	2012	72	20.4	5.2	1.6	10.2	30.7			
John Day	2013	55	19.1	3.8	1.1	11.6	26.5			
	2014	26	12.9	2.9	0.9	7.3	18.6			
	Avg. of All Years	116	18.1	2.3	1.0	13.6	22.5	-2.825	-5.388	-0.262
	2011	25	15.1	2.4	1.2	10.3	19.8			
	2012	31	15.0	2.8	1.1	9.6	20.4			
Lemhi	2013	29	12.1	1.7	0.9	8.8	15.5			
	2014	22	10.6	1.9	1.0	6.8	14.4			
	Avg. of All Years	68	12.9	1.5	1.2	9.9	15.9	-0.288	-1.716	1.14
	2011	22	29.9	5.1	0.7	19.8	39.9			
	2012	19	22.5	4.0	0.7	14.6	30.3			
Wenatchee	2013	22	40.0	5.3	0.7	29.6	50.4			
	2014	20	28.4	7.6	1.1	13.6	43.2			
	Avg. of All Years	44	35.9	4.1	0.7	27.8	44.0	5.487	1.587	9.388

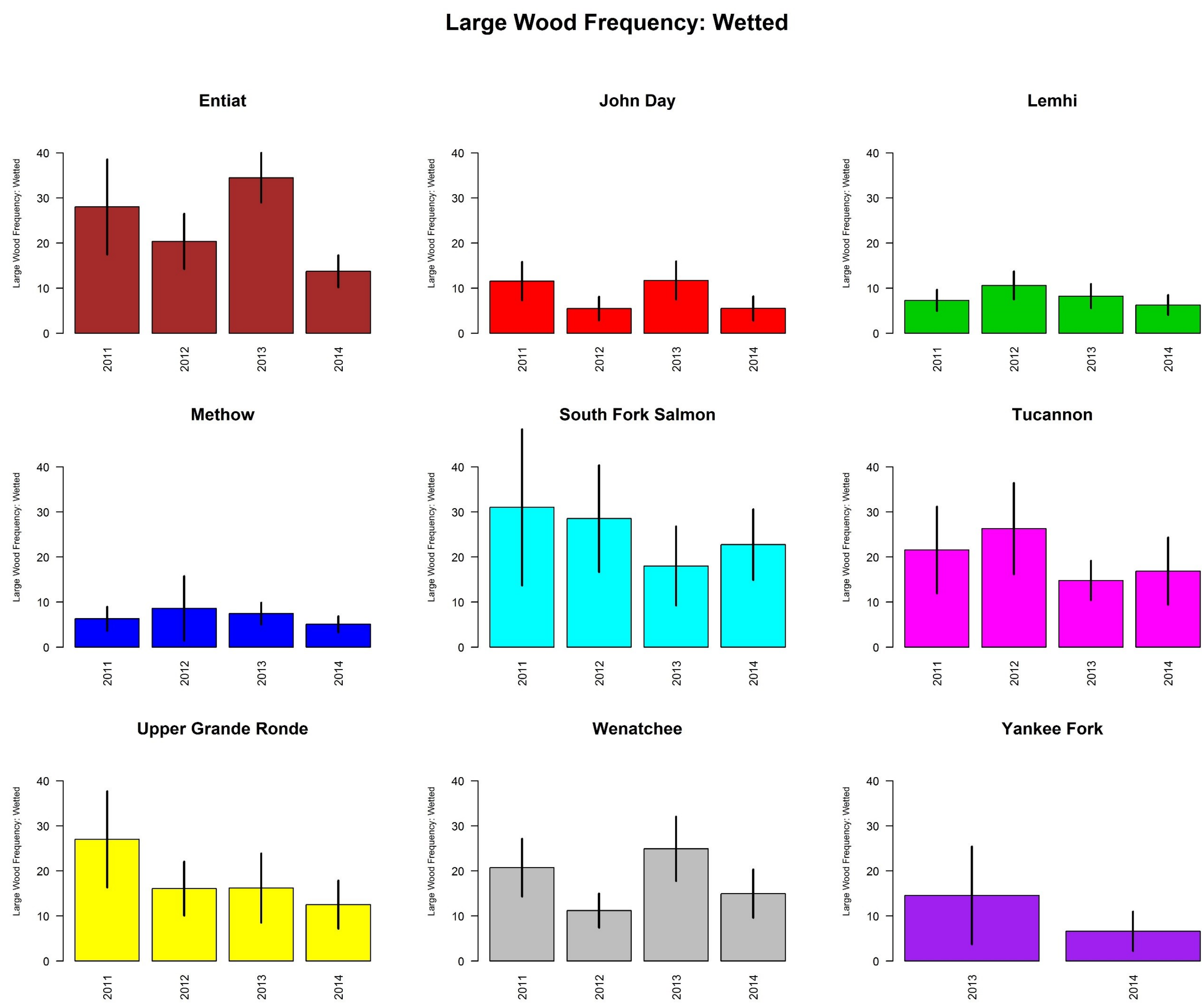


Figure 59. Estimated mean Large Wood Frequency: Wetted (1/m), by watershed x year. Black lines indicates 95% confidence intervals for the mean.

Variance Decomposition

In addition to status and trend estimation, a variance decomposition analysis has been updated to estimate the relative magnitude of the various variance components that sum up to the total amount of variance observed in each CHaMP metric. Variance components assumed for the model are as follows:

σ^2_y: Year-Year (common across all sites in all valley classes in all watersheds)

σ^2_w: Watershed-Watershed variance

σ^2_{vc}: Valley-Class to Valley Class variance

σ^2_s: Site-Site variance

σ^2_e: Measurement Error (Independent for all sites, all measurements, all years)

The lmer function in R is used to estimate components of variance. Inverse probability bootstrapping (IPB), a technique developed within CHaMP is used to account for design weights in the original sampling plan. IPB sampling is a methodology developed specifically to support CHaMP data analysis (manuscript is currently in review).

Measurement noise is assessed via a subsample of CHaMP sites that are re-visited more than once in a given season. Differences in within year, within site responses are taken as measurement noise, and may be due to crew-crew variability or error, but may also reflect real variability, reflective of changes that occur within a sampling season. However, given that year-year variance components tend to be small, it is reasonable to assume that within year temporal components of variability are also small, and thus regarding within year variation as measurement noise is likely a reasonable assumption.

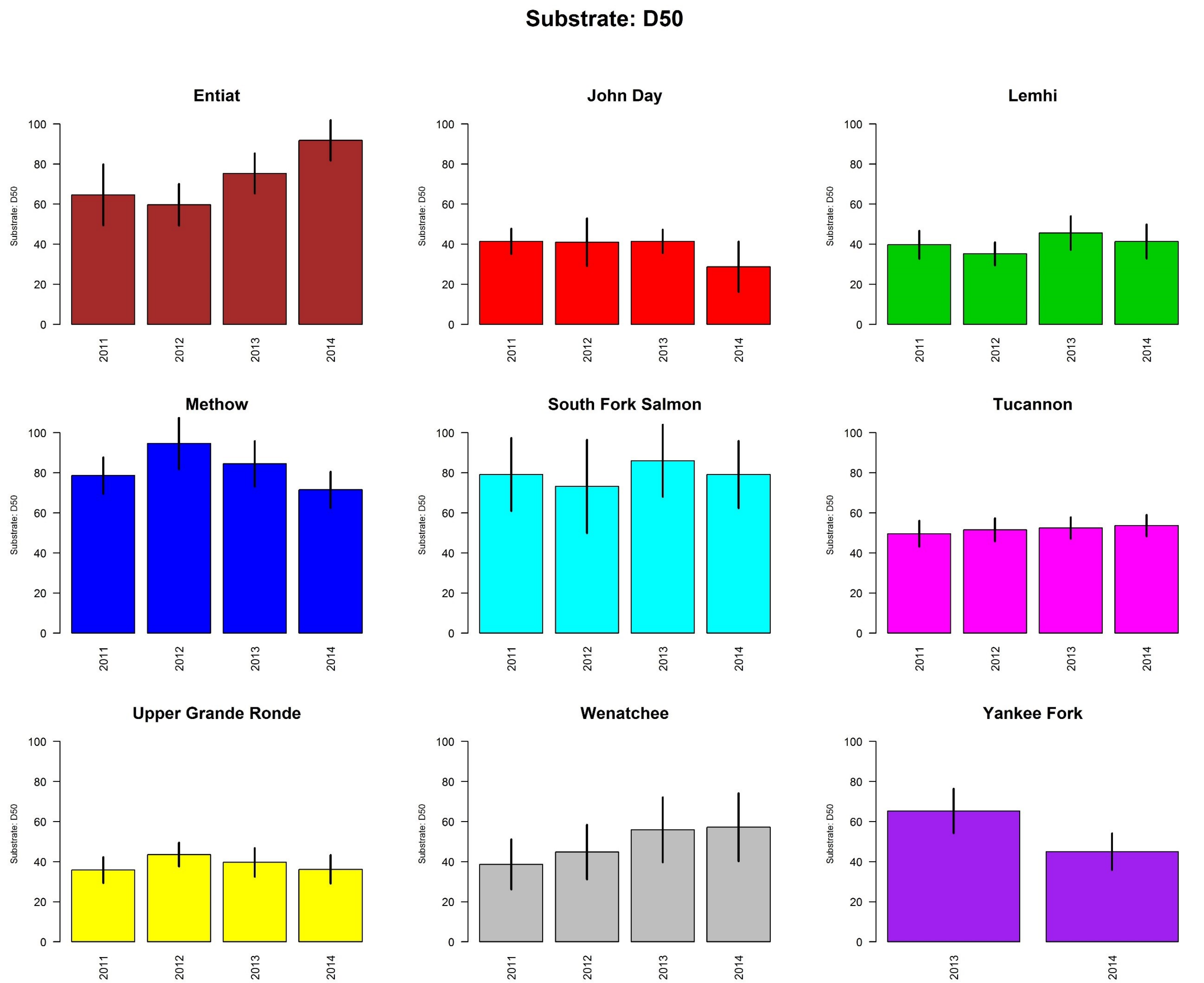


Figure 60. Estimated Substrate D50: Median Pebble Size (mm) by watershed x year. Black lines indicates 95% confidence intervals for the mean.

The variance decomposition is serves several purposes. First, it highlights which metrics, if any, have problematically high measurement noise relative to overall variance. Such metrics are typically addressed via improved sampling procedures or metric modification. In addition, the variance decomposition provides insight into how to any necessary modifications to the sampling design are to be done. For example, we observe that typical year-year variation is a small component of the overall variance. This suggests that less additional information is gained by sampling sites annually, as might be gained by sampling more total sights, but sampling then less often, given a consistent total sampling effort.

Results from 2014 for the updated variance decomposition are provided in Figure 61 on the next page. In general, the amount of measurement noise, relative to other sources of variation, is low, and consistent with what we observed in this analysis in prior years (2011-2013; see Ward et al. 2012, CHaMP 2013, CHaMP 2015). Large wood volume in wetted fast turbulent water is where the highest amount of noise is observed. This may be due to compounding errors in not only the volume of wood measured, but in the visit-visit variability in the assessment of reach type (which may vary with discharge, and not therefore be purely measurement error be real variability). Even at this highest level of measurement noise, the impact on precision of estimates at watershed, or other multi-site spatial scale ups, is minimal because measurement noise tends to be minimized at larger samples sizes, while signal strengths are increased. Therefore, we remain confident that additional years of sampling will improve our ability to assess status and eventually detect long term linear trends, if they exist.

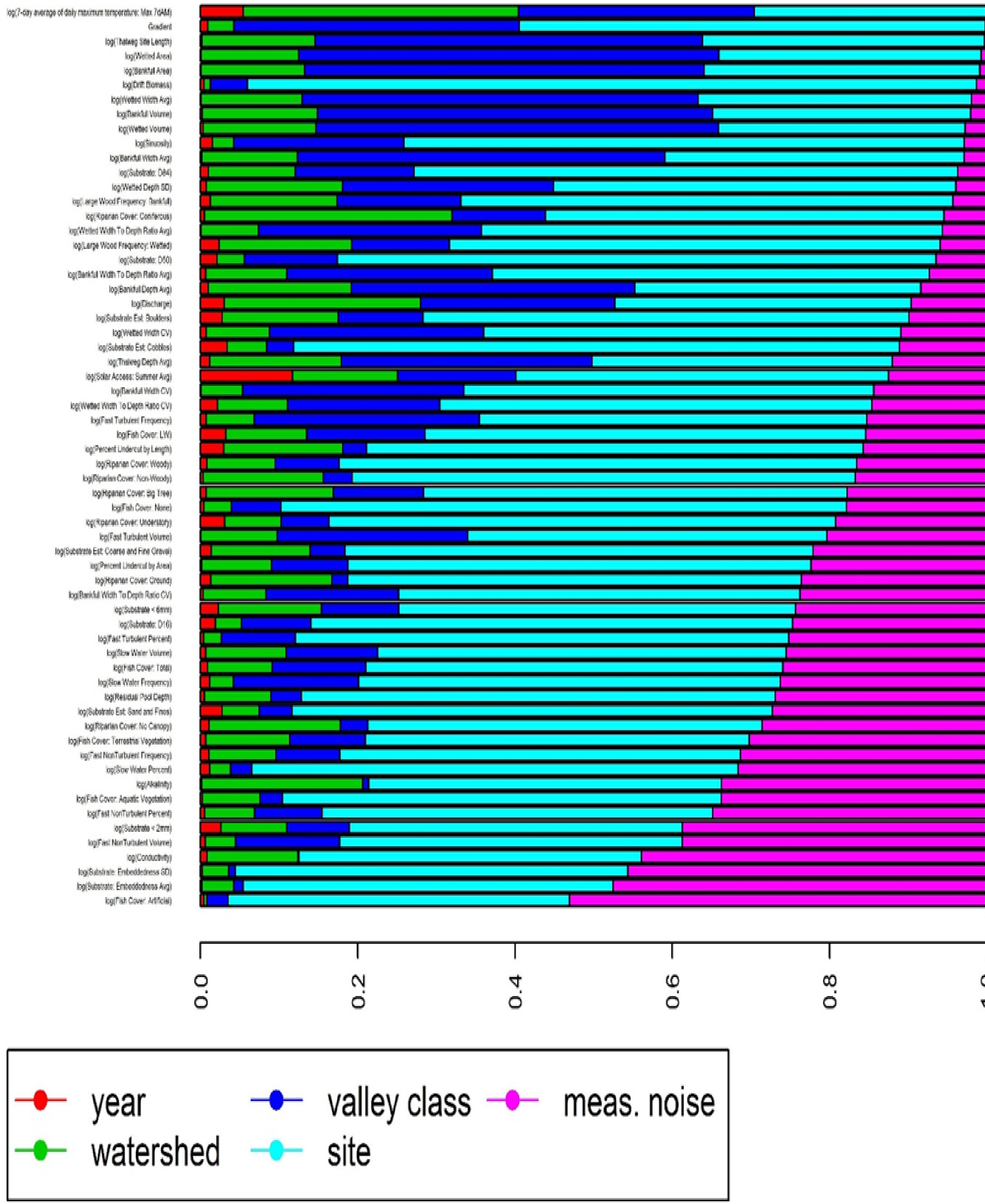


Figure 61. CHaMP Metrics: Estimated components of variance as of 2014.

Geomorphic Change Detection

In 2014 we continued to employ Geomorphic Change Detection (GCD) 5.0 software (http://gcd.joewheaton.org); see CHaMP 2012, 2013, 2015) to analyze CHaMP DEMs from repeat topographic surveys at a site to quantify changes in habitat status over time, and test restoration design hypotheses (see Chapter 1). We refined the GCD software to difference sequential DEMs using spatially-variable uncertainty analysis, in order to robustly distinguish real changes from noise (Wheaton et al. 2010) in the DEM of Difference (DoD, Figure 62) output products that the software produces.

Briefly, our approach uses a spatially-variable, probabilistic, minimum Level of Detection ($_{min}$LoD) to account for errors propagated from the individual DEMs into a DoD. We developed a fuzzy inference system (FIS) to estimate errors in each DEM on a cell-by-cell basis. The FIS accounts for the tradeoff between the completeness of sampling coverage (point density used as proxy) and topographic complexity (slope used as proxy) while keeping track of instrument-reported 3D GPS point quality. The FIS estimates a spatially-distributed metric of surface reliability expressed as a vertical elevation error for each DEM. Basic error propagation is then used to incorporate errors from each concurrent 1 m resolution DEM into the DoD calculations. This propagated error term is used to define the probability that elevation changes measured between two successive DEMs are real by calculating a *t* score to compare the DoD differences against the minLoD defined by the propagated error. The DoDs are then thresholded at a 95% confidence interval, so that only changes estimated as having 95% or higher probability of being real are included in the in the change detection results. The Bridge Creek IMW example presented in Chapter 1 provides a point of reference for the detailed FIS discussion that follows.

The FIS calculates reach-scale volumetric change in storage by multiplying all elevation changes in the DoD by the cell area and accounting separately for erosion and deposition areas. The quantities described below can all be calculated for any control volume of interest; for example, the entire reach or just a defined sub-area of the reach. The net change in sediment storage (ΔV_{DoD}) is defined as the sum of all the deposition volumes minus the sum of all the erosion volumes:

$$\Delta V_{DoD} = \sum V_{Deposition} - \sum V_{Erosion}$$

ΔV_{DoD} over some time period Δt (epoch) represents the sediment budget (or expression of conservation of mass):

$$\frac{Q_{b_{in}} - Q_{b_{in}}}{\Delta t} = \frac{\Delta V_{DoD}}{\Delta t}$$

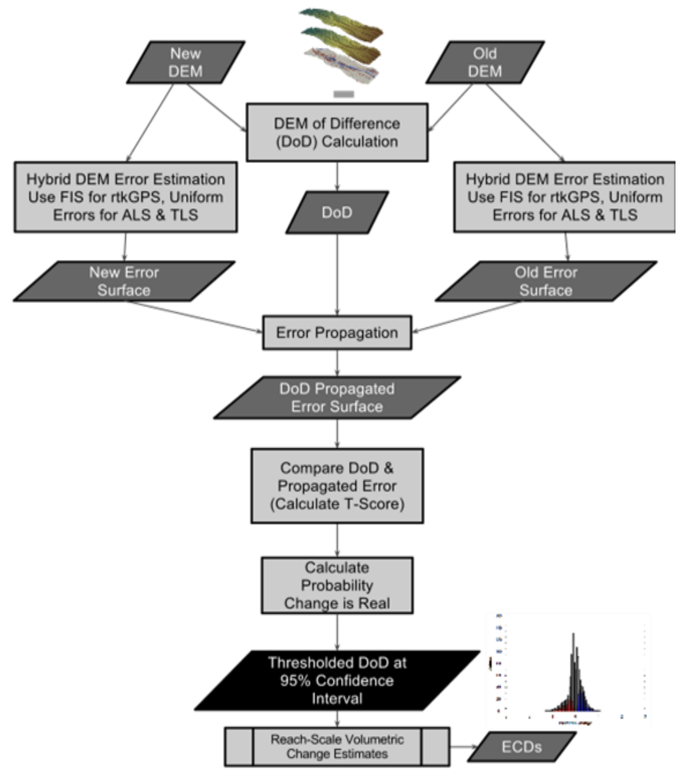


Figure 62. *Geomorphic Change Detection (GCD) workflow.*

where $Q_{b_{in}}$ and $Q_{b_{out}}$ are the bedload flux into and out of the control volume (typically a study reach). By contrast to the net change in sediment storage (ΔV_{DoD}), we also have the total bulk change in sediment storage ($\sum V_{DoD}$), which is simply the sum of the erosion and deposition change in storage volumes as opposed to the difference. In the Bridge Creek IMW case study presented in Chapter 1, bedload flux data were not available; therefore only net and total changes in sediment storage is reported. However, the summed volumes of erosion ($\sum V_{Erosion}$) and deposition ($\sum V_{Deposition}$) both spatially integrate the net changes in sediment storage over the course of the epoch. In terms of the alluvial sediment store that makes up the valley fill, $\sum V_{Erosion}$ represents withdrawals from storage whereas $\sum V_{Deposition}$ represents deposits to that storage.

The methods described above are useful for establishing confidence that 'real changes' are reliably being distinguished from noise.

As of 2014, we have constructed a GCD model engine that is built into the CHaMPMonitoring.org data management system, which automatically runs GCD analysis for every repeat topographic survey at a site. Therefore, we have been able to generate GCD results in a standardized, automated manner for all CHaMP watersheds with repeat visits. We have analyzed GCD data in the Bridge Creek IMW for the early years following restoration from 2009 to 2010 and 2010 to 2011. Change detection analyses for more recent years are currently being processed and will be disseminated in future reports and publications. We have also performed analyses in other CHaMP watersheds and ISEMP IMWs (e.g., Tucannon, Asotin, Entiat, Methow) and plan to continue analysis work in other watersheds. The BPA's regional AEM program has adopted the CHaMP topographic survey protocol so that the AEM effort can generate DoD products to evaluate the reach-scale effectiveness of different suites of habitat restoration actions.

Recommendations

We believe, based on four years of annual CHaMP metric variance decomposition analysis and protocol refinements to improve metric capability, that we are able to produce standardized, repeatable, and precise habitat measurements and metrics. Accordingly, we recommend that sampling and variance decomposition continue under existing CHaMP frameworks for the remainder of the 9-year study design to improve our ability to describe status, detect long-term linear trends, and continue refinement of fish-habitat synthesis products that rely on robust CHaMP datasets.

CHAPTER 3: FISH STATUS AND TRENDS

In this chapter we summarize the status and trend data ISEMP personnel have collected from the Salmon and Entiat River subbasins related to parr, smolt, and adult abundance, and productivity through time. Status and trends monitoring in the John Day River subbasin under ISEMP has been discontinued since 2014 due to budget constraints, but ODFW, with whom ISEMP partnered on this monitoring effort, continues to conduct fish and habitat monitoring in the subbasin. However, ISEMP personnel are implementing an extensive juvenile steelhead monitoring program within Bridge Creek as part of the IMW project. Juvenile steelhead are monitored within treatment and control reaches using a series of capture-recapture surveys at which time captured juvenile steelhead are implanted with a PIT tag. While it is impractical to operate a smolt trap at the mouth of Bridge Creek, we have been developing analytical approaches that will be used to expand the counts of PIT-tagged steelhead smolt as they pass over PIAs at the mouths of tributary watersheds such as Bridge Creek. Smolt estimates will be combined with estimates of adult abundance from the trap to produce metrics of steelhead productivity such as smolt to adult ratios and smolts per female. These estimates will be available for all years in which the PIAs and adult trap have been in operation (2009 – present), and will extend into future years.

Estimation of Parr Abundance at the Subbasin Scale

We have sampled for parr in the Secesh and Lemhi rivers since 2009 and in the Entiat River subbasin since 2005. In the Entiat, sampling is conducted at the site scale using a GRTS sampling design (Stevens and Olsen 2004) following the plan laid out in the Upper Columbia Monitoring Strategy (Hillman 2004, 2006). Sites have been stratified using two schemes over this time period: from 2006 – 2010 strata were delineated by anadromous/resident zones and 5 gradient classes; from 2011 – present strata have been delineated by geomorphic valley class and ownership. Additionally, within the Entiat IMW (2010 - present), strata are defined by geomorphic reach based on the Bureau of Reclamation tributary assessment (BOR 2009). The target frame has shifted several times over the period of 2004-2014, but in general the frame has shrunk over time as we have better determined the extent of anadromy. Also the number of sites sampled each year has varied due to budget constraints or reallocating effort into focused studies to answer a particular question, such as quantifying crew variability for example. For this analysis the most parsimonious frame (2013) was adopted to support comparisons across years, leading to some data not being included. Lastly, sites were sampled using snorkel counts from 2004 – 2009, and depletion surveys and mark-recapture surveys using electrofishing from 2010 - 2013.

In the Lemhi and Secesh the GRTS approach was also employed from 2009 – 2012; however, from 2013 on ISEMP ceased sampling for parr in the Secesh, and switched to a spatially continuous sampling methodology in the Lemhi (see Chapter 6 for more details). For each site survey, an estimate of abundance for each species is made using different methods depending on the sampling methods used.

- Depletion surveys - the abundance estimators described in Seber (1982) are used for two and three pass depletions.
- Mark-recapture surveys - the modified Chapman estimator is used.
- Single pass surveys, or surveys with low numbers of recaptures that provided unreliable abundance estimates - a ratio estimator has been developed from the other mark-recapture or depletion surveys, which was then applied to the number of fish caught in the initial first pass sample.
- We delete any repeat visits to a site within each year and use data from one visit/site/year.

We calculated an estimate of parr abundance using the statistical software R, including the spsurvey package for the GRTS-based survey data, whereas we estimated total parr abundance from spatially continuous surveys in the Lemhi (years 2013-2014) using SPAZ software (Arnason and Station 1996). In both the Entiat and Lemhi the steelhead frame was assumed to be identical to the 2013 CHaMP frame, while the spring/summer Chinook frame was restricted to those areas of the 2013 CHaMP frame identified as accessible to anadromous fish.

For the Entiat data, we estimated abundance for each species using different methods depending on the sampling methods used in each survey.

- Snorkel surveys - an inflation factor based on one developed by Tracy Hillman (pers. comm.) but modified by ISEMP (based on data from the Wenatchee) was applied.
- Depletion surveys - the abundance estimators described in Seber (1982) were used for two and three pass depletions.
- Mark-recapture surveys - the modified Chapman estimator was used.
- For surveys with low numbers of recaptures that provided unreliable abundance estimates with the modified Chapman estimator a ratio estimator was developed from the other mark-recapture surveys, which was then applied to the number of fish caught in the initial marking sample.

Status of Parr Abundance

Figure 63 shows the status of parr abundance in the Lemhi and Secesh from 2009 – 2014. Due to the inability to perform a recapture pass in the lower mainstem of the Lemhi in 2013, where a majority of spring/summer Chinook parr are suspected of rearing, estimates of spring/summer Chinook parr in 2013 are not available.

For Entiat River Chinook and steelhead parr estimates we see an increase in Chinook abundance estimates from 2009 to 2010 and drop in steelhead abundance estimates from 2010 to 2011 (Figure 64); however, these overlap with the implementation of CHaMP in 2011 which changed the stratification scheme of how sites were selected, plus there was a change in sampling methodology in 2010 from snorkeling to electrofishing. Although we have tried to account for these differences by providing a best estimate of abundance at each site, regardless of sampling method, the large shifts in abundance for both Chinook and steelhead, albeit in different directions, during the same period that stratification, sampling methods and sample size all changed seems unlikely to be a coincidence. At the moment, it is unclear how to test this hypothesis, or how to correct for any unintentional introduction of bias. Previous work in the Wenatchee which compared parr estimates to redd counts and smolt estimates suggested that years where the fish surveys were conducted with electrofishing led to parr estimates that were more consistent with smolt estimates and redd counts. This suggests the last 4-5 years may be a more unbiased estimate of the total number of Chinook and steelhead parr in the Entiat.

Estimating Trends in Parr Abundance

Estimating salmonid population trends is a difficult business since it requires a long-term dataset that encompasses multiple salmonid generations, preferably collected with a stable protocol. The most difficult part may be determining exactly what question a trend analysis is meant to answer, as different approaches should be taken to address different questions. Several studies have suggested the need for 15-30 years of data from 30-60 different sites in order to detect a 2% decline at a statistical level of alpha = 0.8 to estimate long-term trends (Wagner *et al.* 2013). Very different approaches should be employed to detect effects of restoration actions, and ISEMP will continue to investigate best practices in these realms.

An additional consideration is the many methods available to estimate trend. We investigated three methods for estimating trend on the Entiat parr abundance data: (1) the GRTS method, which assumes a linear trend at each site and rolls up those

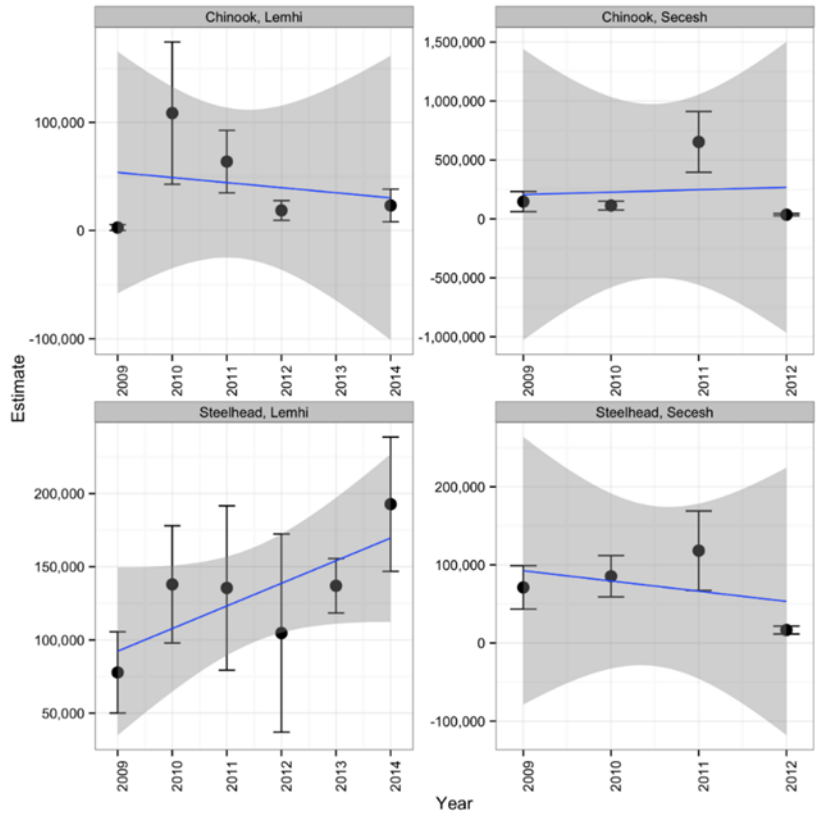


Figure 63. Estimates of the status of parr in the Lemhi and Secesh River subbasins, with 95% confidence intervals, by species and river 2009 - 2014.

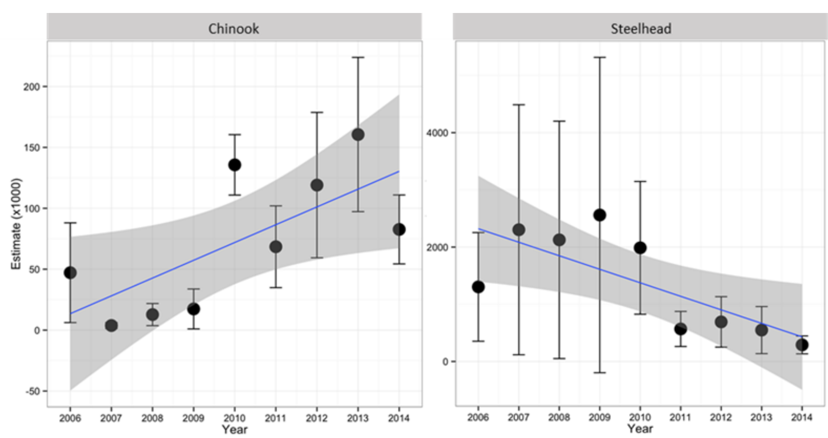


Figure 64. Estimates of the status of Chinook parr (left panel) and steelhead parr (right panel) in the Entiat River subbasin, with 95% confidence intervals 2006 – 2014. The blue line and gray area depict the best fit linear trend, with 95% confidence intervals.

trend estimates in the same manner that we roll up abundance estimates from sites to a larger scale; (2) the multivariate auto-regressive state-space method (MARSS) which estimates a population growth rate from multiple locations assuming they all have a single underlying trend, and (3) the regression method which takes the annual status estimates and fits a regression line to them. The regression method does not account for the uncertainty in the annual estimates and out of the three methods investigated is probably the least rigorous, but it is also the simplest and is often applied. Each method of estimating a trend in the salmonid population in the Entiat River subbasin resulted in a different answer (Table 30).

The fact that the three different methods produces three different results suggest there is not enough data to estimate a true population growth rate, or that the current data is too noisy to do so without a longer time series. While we have been collecting fish population status and trend data in the Entiat River subbasin for 8 years, that equates to only about two generations in a salmonid life cycle. A longer time series that adds more salmonid generational data will likely better support a trend analysis. ISEMP personnel are continuing to develop recommendations for which approach is the most rigorous to use when estimating trends in salmonid populations.

Table 30. Estimates of Chinook and steelhead population growth rate in the Entiat River subbasin from data collected 2006 – 2014.

Method	Growth Rate	
	Chinook	Steelhead
GRTS	-0.12	-0.28
MARSS	0.06	0.06
Regression	0.34	-0.23

Estimating the Number of Smolts Emigrating from the Lemhi, Secesh and Entiat River Watersheds

Lemhi and Secesh

Data about smolts comes from three RSTs in the Lemhi, and one in the Secesh River (Figure 65 and Figure 66). The upper Lemhi RST (LRW) and Hayden Creek RST (Hayden) were operated by QCI from 2008 - present. The lower Lemhi RST (L3A) was operated by IDFG through 2012, although the data from 2012 was not deemed reliable. QCI has operated a trap in the lower Lemhi (L3AO) since 2013. The trap in the Secesh has been operated by the Nez Perce tribe from 2008 to the present, although no data was available from the 2011-2012 migration year. The earlier years of data from the lower Lemhi RST (trap L3A in years 2008-2010) are considered less than ideal for a variety of reasons. In the future, methods will be developed to predict the data from the lower trap based on the traps in the upper Lemhi and Hayden creek; however, with only 2 years of reliable data to date to base this on, such an analysis will require more years of data.

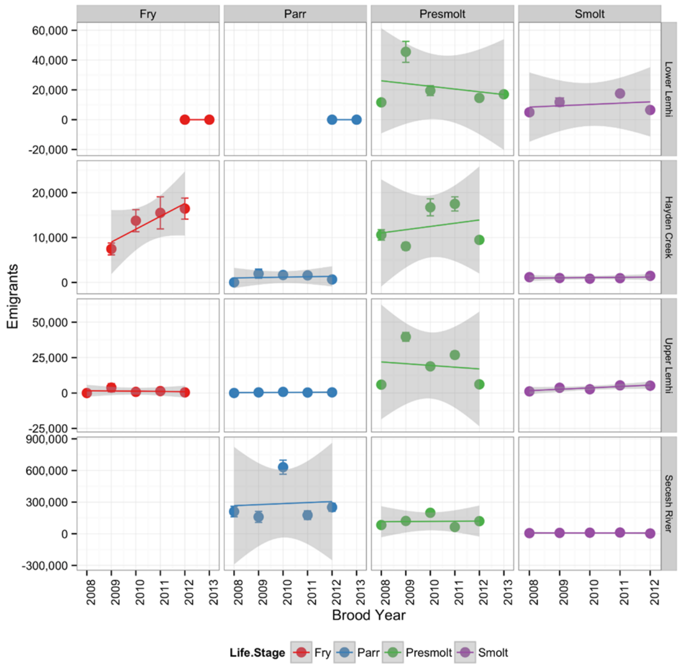


Figure 65. Estimates of the status of spring/summer Chinook smolts in the Lower Lemhi, Hayden Creek, Upper Lemhi and Secesh River, with 95% confidence intervals, by life stage and screw trap, 2008 - 2013.

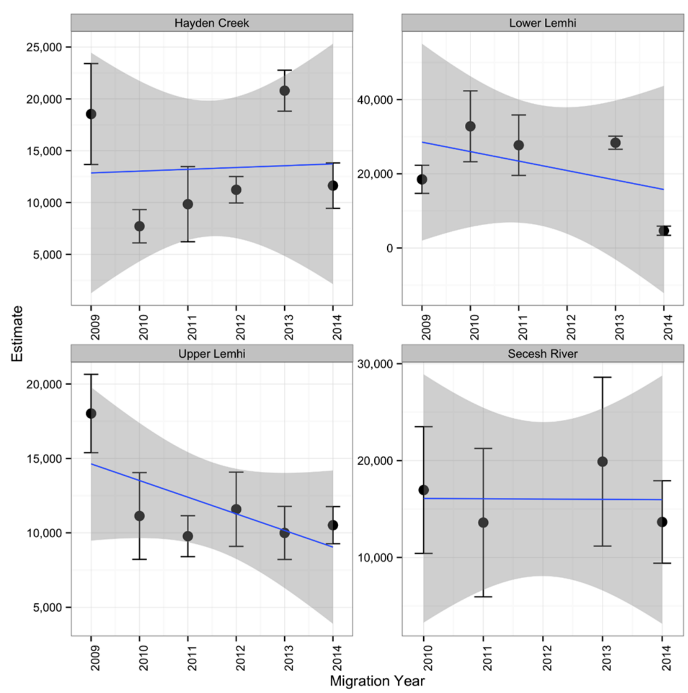


Figure 66. Estimates of the status of steelhead emigrants from the Lower Lemhi, Hayden Creek, Upper Lemhi and Secesh River, with 95% confidence intervals, for each screw trap, 2009 – 2014.

Entiat River Subbasin

Smolt data in the Entiat comes from traps operated at river km 6 from 2003 – 2009 (upper trap, run by MCFRO for hatchery evaluation purposes) and at the mouth of the mainstem (lower trap, ISEMP-funded) run by the MCFRO from 2007 – present (Figure 67).

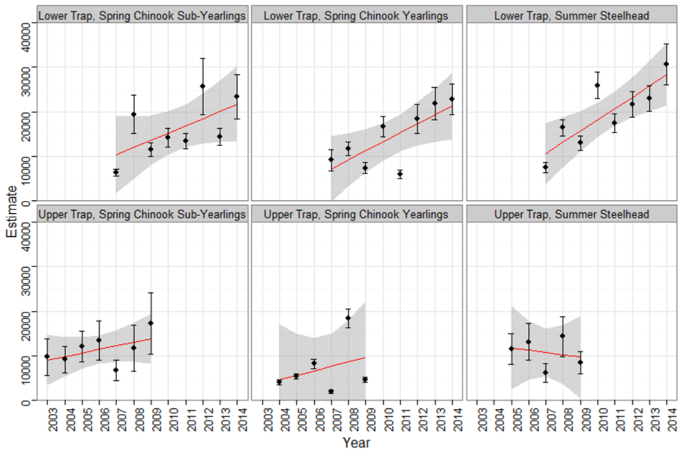


Figure 67. Estimates of the status of spring Chinook sub-yearlings, yearlings and summer steelhead emigrants from the Entiat River upper rotary screw trap 2003 - 2009, and lower screw trap 2007 – 2014, with 95% confidence intervals.

Estimating Adult Escapement

Adult escapement estimates for the Lemhi and Secesh River subbasins (Tables 31—34) come from the PIT-tag based methodology described in detail in Chapter 6. For the Entiat, adult escapement estimates have historically been derived from spawning ground surveys: steelhead surveys are funded by ISEMP and conducted by the MCFRO (Figure 68), while spring Chinook surveys are conducted by the MCFRO as part of their hatchery program (Figure 69). In recent years Washington Department of Fish and Wildlife has implemented estimating adult escapement for the Upper Columbia, including the Entiat River, for steelhead using same PIT-tag based methodology as employed in the Lemhi (2014).

Table 31. Estimates of adult spring/summer Chinook salmon escapement to the Lemhi River subbasin 2010 – 2013.

Spawn Year	Estimate	Std. Error	Lower 95% CI	Upper 95% CI	CV
2010	181	68	65	320	0.357
2011	290	51	199	392	0.175
2012	122	36	58	194	0.282
2013	438	55	348	562	0.124

Table 32. Estimates of adult spring/summer Chinook salmon escapement to the Secesh River subbasin 2010 - 2013.

Spawn Year	Estimate	Std. Error	Lower 95% CI	Upper 95% CI	CV
2010	1201	182	886	1584	0.150
2011	780	87	635	965	0.111
2012	923	100	736	1115	0.107
2013	1076	90	914	1264	0.084

Table 33. Estimates of adult steelhead escapement to the Lemhi River subbasin 2010 – 2013.

Spawn Year	Estimate	Std. Error	Lower 95% CI	Upper 95% CI	CV
2010	501	78	370	667	0.154
2011	300	53	198	396	0.174
2012	251	47	166	355	0.185
2013	287	41	212	369	0.142

Table 34. Estimates of adult steelhead escapement to the Secesh River subbasin 2010 – 2013.

Spawn Year	Estimate	Std. Error	Lower 95% CI	Upper 95% CI	CV
2010	213	54	118	313	0.247
2011	345	56	243	461	0.162
2012	120	32	64	185	0.256
2013	37	16	12	72	0.408

Entiat River Subbasin

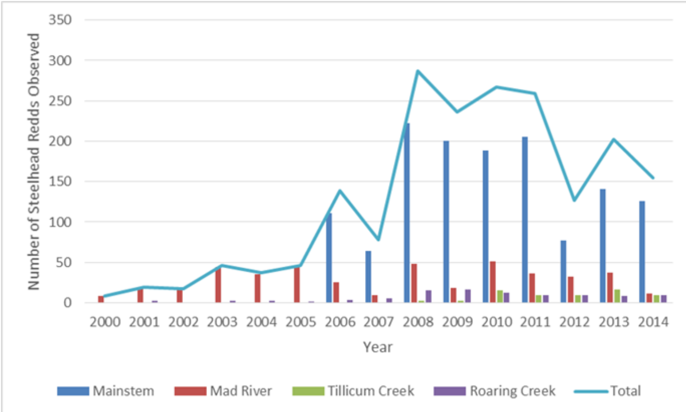


Figure 68. Steelhead redd counts from the Entiat River subbasin 2000 – 2014.

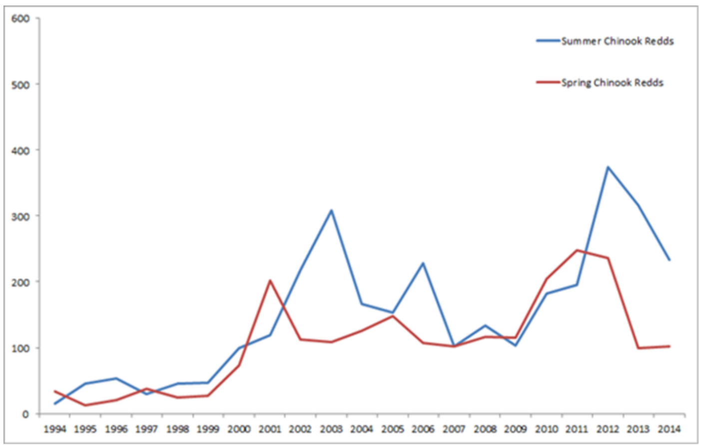


Figure 69. Summer and spring Chinook redd counts from the Entiat River subbasin 1994 – 2014. Data courtesy of the U.S. Fish and Wildlife Service Mid-Columbia Fishery Resource Office.

Productivity

ISEMP personnel have calculated productivity for spring/summer Chinook salmon in the Secesh for 5 brood years, 2008-2012, by calculating the number of smolts, by life-stage, per estimated females who escaped to spawn in the Secesh (Figures 70—72). For years with PIT-tag based estimates of adult escapement, we estimated the number of females by applying the sex ratio of PIT-tagged adults (with known sex) thought to have escaped to either the Lemhi or the Secesh to the escapement estimate of those areas. PIT-tag based estimates of adult spawners only go back to 2010, so we used estimates from the DIDSON surveys (Kucera and Tribe 2008, Kucera and Tribe 2009) and used sex ratios from carcass surveys for brood years 2008-2009 (Venditti *et al.* 2011, Venditti *et al.* 2012) to estimate productivity. Productivity estimates for spring Chinook in the Entiat have been calculated based on emigrants per redd (Figure 73); estimates for steelhead based on brood year identified by scale analysis are being developed and will be reported in the future.

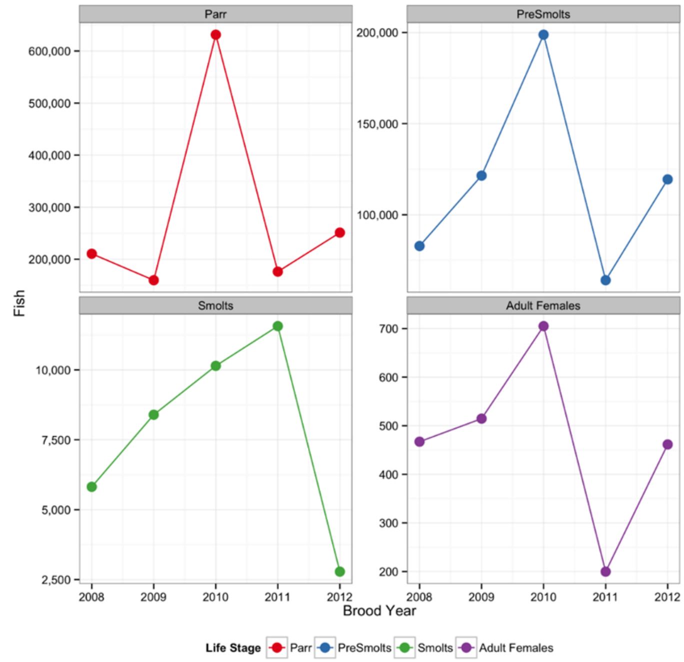


Figure 70. Estimates of spring/summer Chinook salmon abundance, by life stage and brood year in the Secesh River, 2008 - 2012.

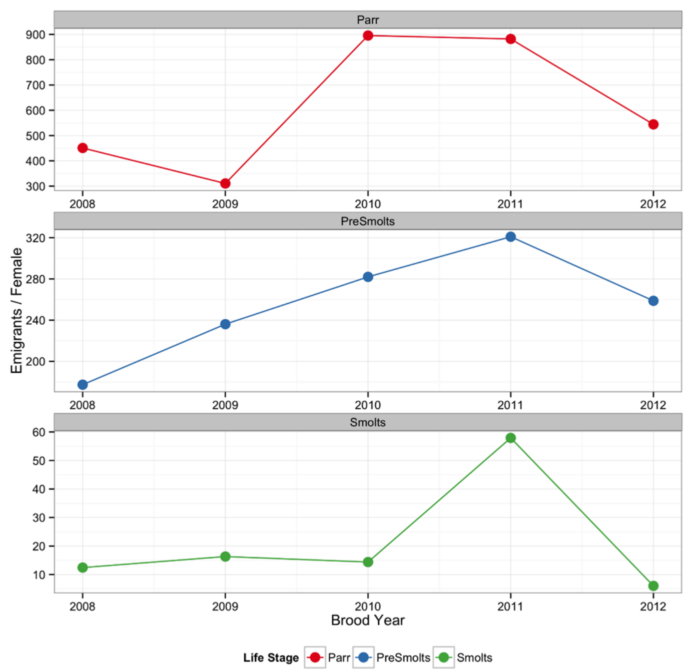


Figure 71. Time-series of productivity of spring/summer Chinook salmon in the Secesh River, defined by emigrants per adult female, 2008 - 2012.

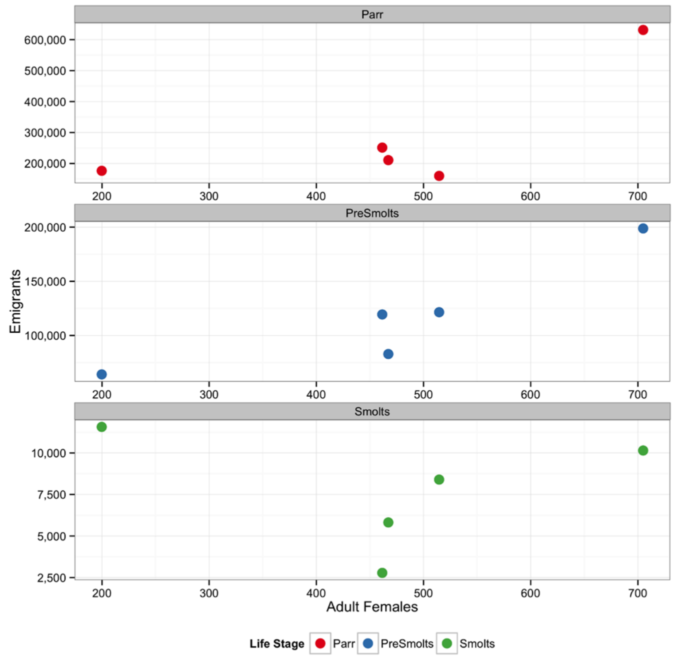


Figure 72. Emigrants (by life stage) of spring/summer Chinook salmon in the Secesh River, plotted against estimated adult females.

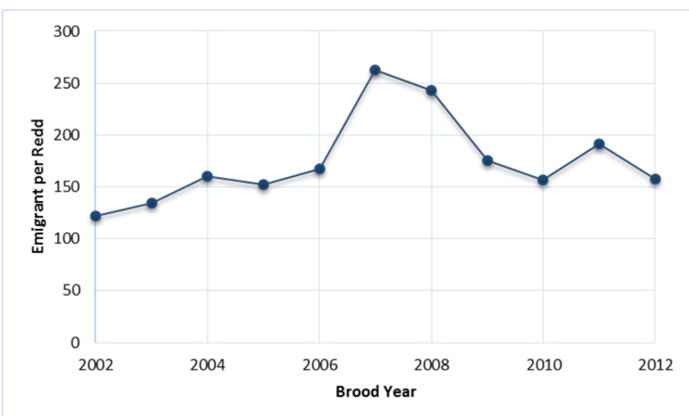


Figure 73. Time-series of productivity of spring Chinook salmon in the Entiat River, defined by emigrants per redd 2002 - 2012.

There is little evidence for or against density dependence in the productivity by life-stage for spring/summer Chinook salmon in the Secesh or the Entiat. However, this is a small dataset, only five brood years for the Secesh for example, and revisiting this analysis with several more years of data may provide more insight into the population dynamics.

Total Escapement for the Salmon River Subbasin

Estimates of total wild escapement over LGR are presented in Table 35 and Figure 74 and for spring/summer Chinook salmon escapement estimates for Technical Recovery Team populations (Figure 75) and steelhead escapement estimates for Technical Recovery Team populations (Figure 76). The methodology used to generate these estimates is described in detail in Chapter 6.

Table 35. Estimates of escapement of adult spring/summer Chinook and steelhead over Lower Granite Dam 2010 – 2013 using a PIT tag based methodology.

Species	Year	Escapement Estimate	CV
Chinook	2010	27927	0.050
	2011	24761	0.025
	2012	21918	0.049
	2013	19610	0.049
Steelhead	2010	43539	0.051
	2011	40111	0.047
	2012	30818	0.091
	2013	20833	0.045

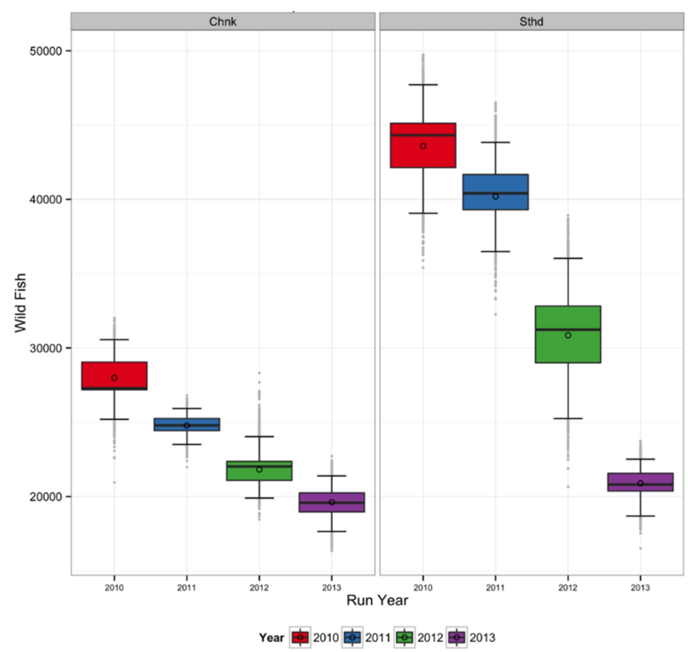


Figure 74. Boxplots showing the posterior distributions of total escapement for each combination of species, Chinook and steelhead, and year. Middle lines and boxes depict the mode and 50% highest posterior density intervals, while circles mark the median. Whiskers represent the 95% highest posterior density intervals, and points are outliers beyond that interval.

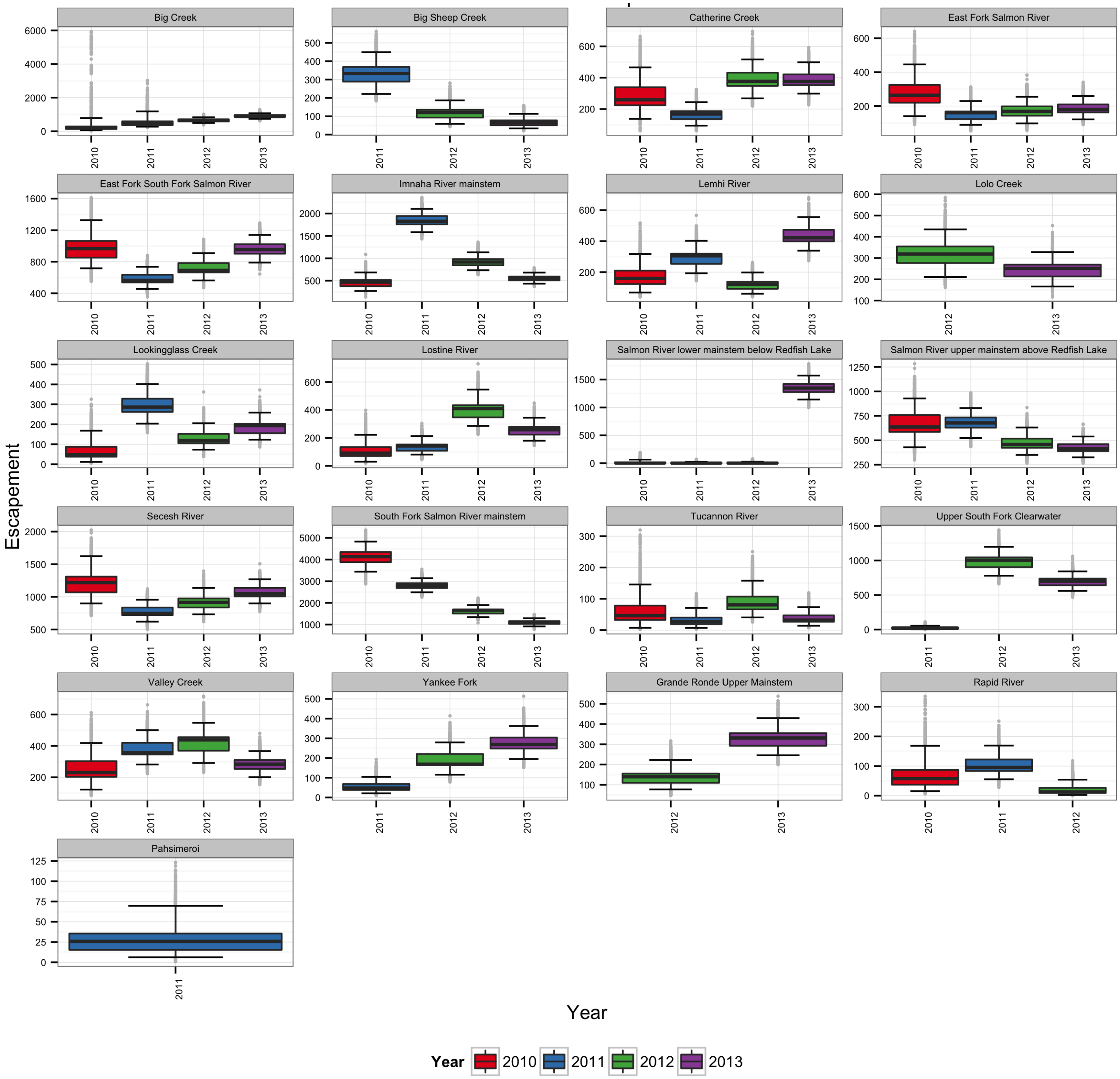


Figure 75. Boxplots showing the posterior distributions of the spring/summer Chinook salmon escapement estimates for Technical Recovery Team populations. Middle lines and boxes depict the mode and 50% highest posterior density intervals. Whiskers represent the 95% highest posterior density intervals, and points are outliers beyond that interval. Colors correspond to different years.

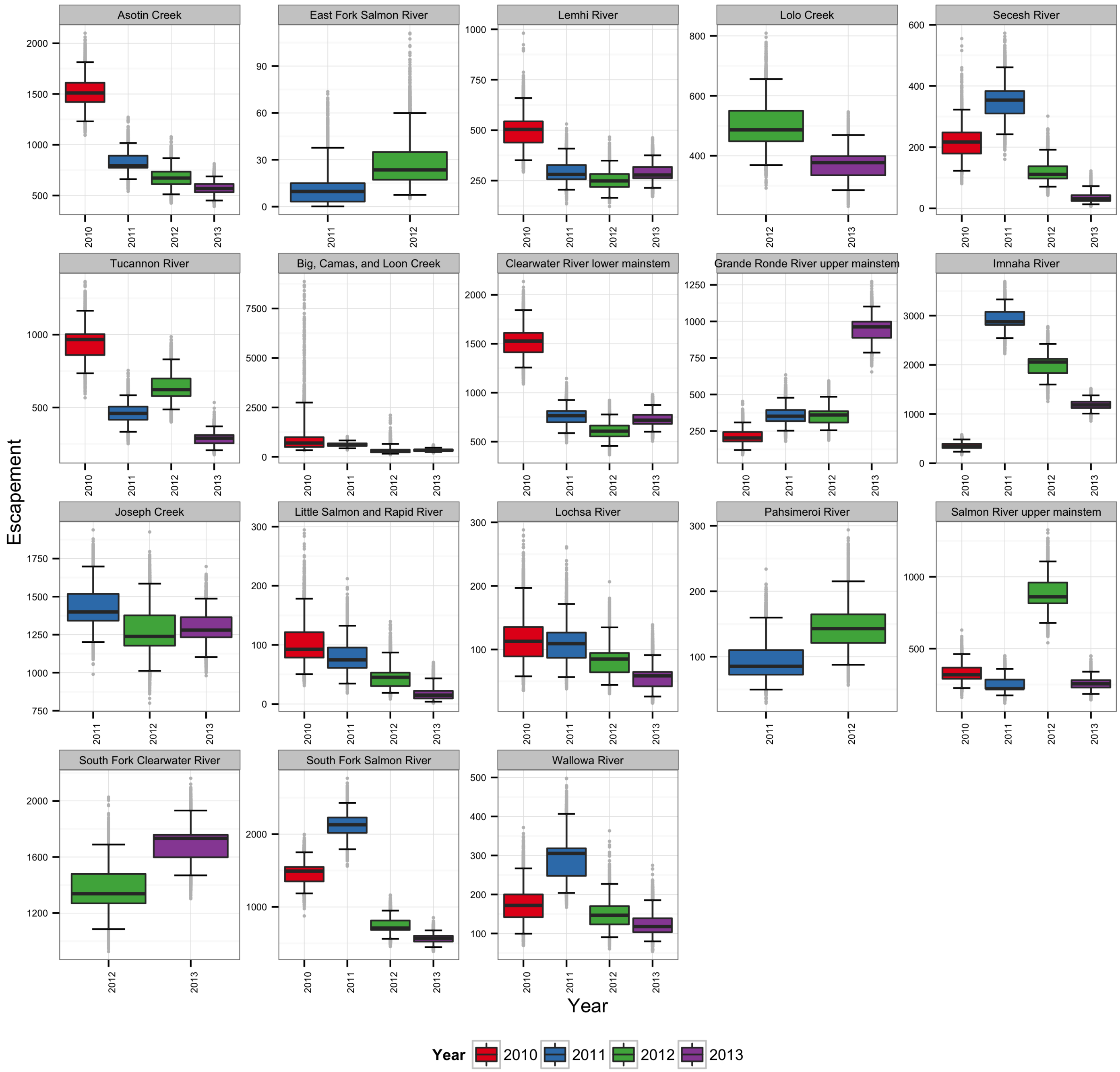


Figure 76. Boxplots showing the posterior distributions of the steelhead escapement estimates for Technical Recovery Team populations. Middle lines and boxes depict the mode and 50% highest posterior density intervals. Whiskers represent the 95% highest posterior density intervals, and points are outliers beyond that interval. Colors correspond to different years.

Recommendations

Based on the developments in protocol and study design that have led to more accurate estimations of the status of parr and adult escapement we recommend that ISEMP continues to collect these data series to build long-term datasets that will support the robust estimation of trends. As we discussed, estimating trend is a difficult business, and ISEMP is committed to developing analysis tools and guidance for the RME program on this matter. In response to reviewers' comments, we also recommend that we compile a review of sampling techniques, including describing the biases associated with each method (e.g., snorkeling, depletion, mark-recapture and single pass surveys), and show how life cycle models developed in the IMWs can be used to generate fish population statistics in unsampled basins.

CHAPTER 4: EXTRAPOLATING SITE-LEVEL DATA

Introduction

An important part of ISEMP and CHaMP's work is the development of statistical methods and analytical tools for estimating habitat condition in watersheds and domains of interest where sampling is not currently occurring. In this chapter we describe how we are applying the River Styles framework (Brierley and Fryirs 2005; see CHaMP 2013, 2015) in CHaMP watersheds in the Columbia River Basin, in order to develop a mechanism by which we can further the validation of our continuous network models of stream character, behavior, and geomorphic condition for all watersheds. The sections that follow present ISEMP-CHaMP River Styles advancements from 2014 and discuss the utility of this approach for model validation in sampled areas, and data extrapolation to unsampled areas.

River Styles

In 2014 we furthered development and testing of a River Styles Procedural Tree to standardize our application of the River Styles framework in CHaMP watersheds (Kasprak and Wheaton 2012; O'Brien and Wheaton 2015). We developed a common procedural tree to allow us to effectively cross-walk information between watersheds in a standardized manner, and to improve consistency across the CHaMP-ISEMP network. As of 2014, we have applied and tested our River Styles Tree across a number of CHaMP watersheds (e.g., Tucannon, Asotin, Grande Ronde, Wenatchee, Yankee Fork). In 2015 we will continue to advance application of the River Styles framework and procedures in additional CHaMP watersheds. Data from this effort will further our ability to validate the automated tools and continuous network model products that we are developing to support management and policy decision-making.

Methods

Our methods for determining a "first cut" for River Styles and reach breaks are explained in detail in O'Brien and Wheaton (2015). As described by Brierley and Fryirs (2005), any substantial change in the valley setting, channel planform, floodplain extent, or geomorphic unit configurations define a new River Style. Major changes in valley confinement and planform characteristics are also used to define the reach breaks. Initial reach breaks are determined using Google Earth imagery and other higher-resolution remotely sensed datasets, such as DEMs. The breaks are then mapped in ArcMap using a 24k NHD streamline layer as a base and validated in the field to create an initial River Styles reach break map. Our common procedural tree is applied to create River Styles trees. The trees contain all of the river types that we identified in laterally confined, partly confined, and laterally unconfined valley settings (Figure 77). Each attribute that is present in these settings is added lower in the tree

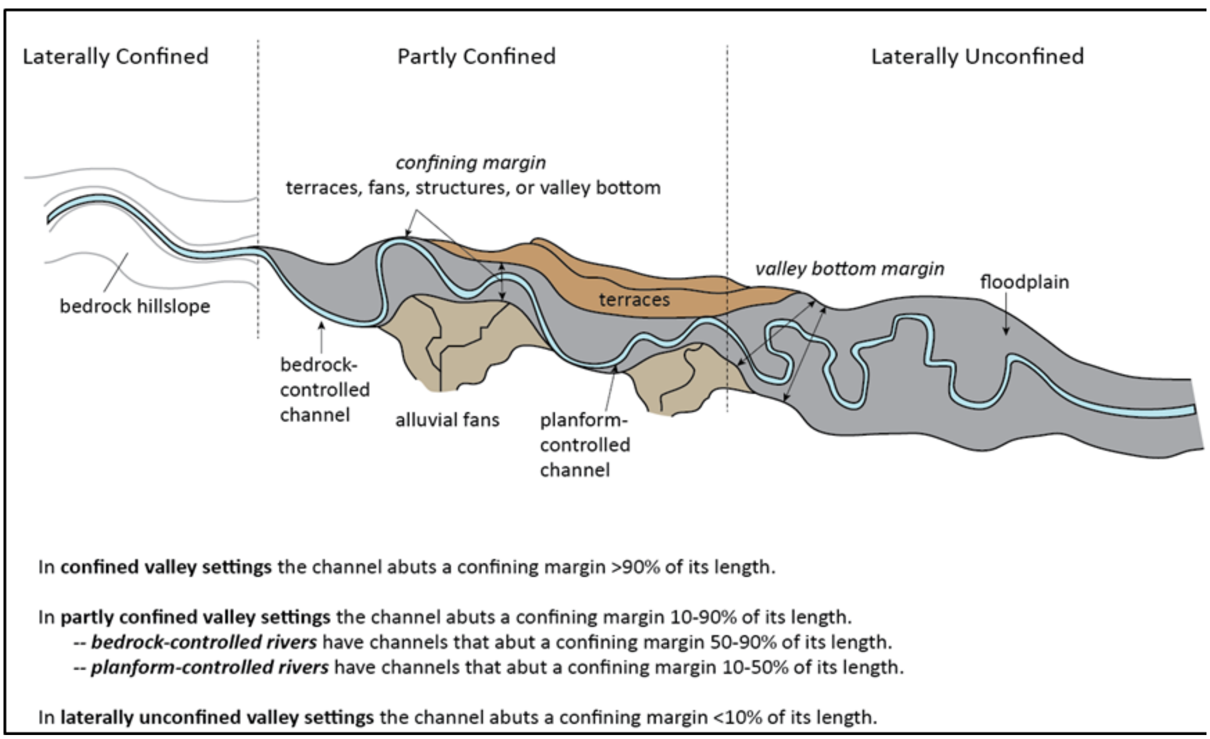


Figure 77. Schematic showing levels of channel and valley confinement used to determine valley setting and define River Styles (from O'Brien and Wheaton 2014; modified from Brierley and Fryirs 2005).

based on finer scale of analysis and size of nested geomorphic attributes. As an example, we applied our procedural framework to plot a tree and designate River Styles in laterally unconfined valleys in the Asotin Watershed (HUC 8) (Figure 78).

We are pursuing River Style characterizations at multiple scales and have completed River Styles Stage 1 (delineation of river character and behavior) for the large physiographic regions and subbasins of the interior and upper CRB for major perennial trunk streams that span important Chinook and steelhead domains (Figure 79). In the case of individual CHaMP watersheds we use a higher stream density in our analyses, which are primarily focused on fish-bearing perennial streams and are extended past Stage 1 to Stage 2 (geomorphic condition; Figure 80), and Stage 3 (river recovery potential).

Results

We have completed Stage 2 and 3 River Styles analyses in the Middle Fork John Day, Pine Creek, Asotin and Lemhi watersheds, and are pursuing completion of the remainder of priority watersheds in the CRB. The procedural tree used in the John Day watershed has been adopted as the working version for CRB subbasins. River Styles trees for the types identified throughout the Blue Mountain, Idaho Batholith, and Upper Columbia regions of the CRB have been completed and are available upon request.

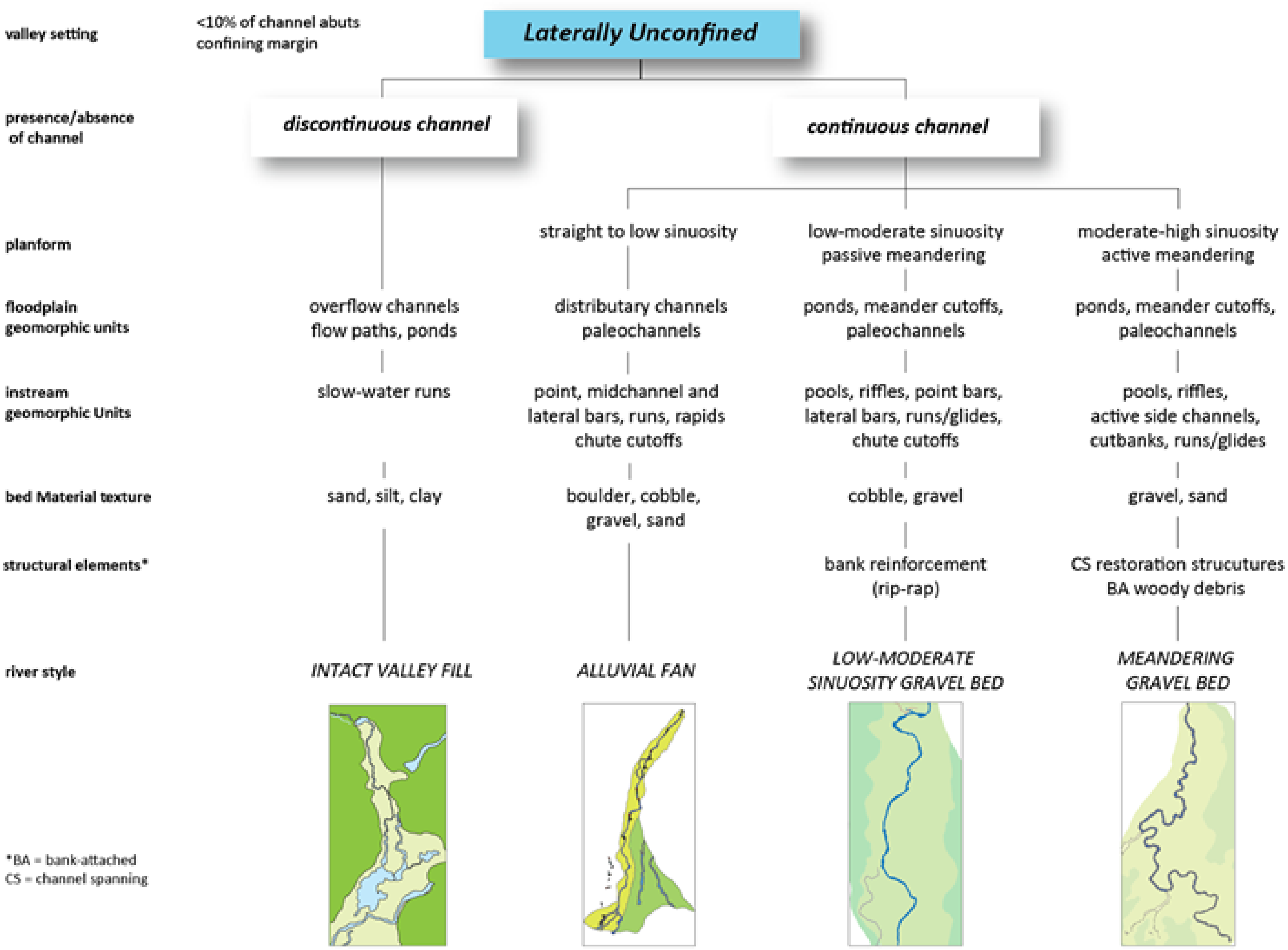


Figure 78. River Styles tree developed for laterally unconfined valleys in the Asotin Watershed, southeast WA. Reproduced from Camp (2015).

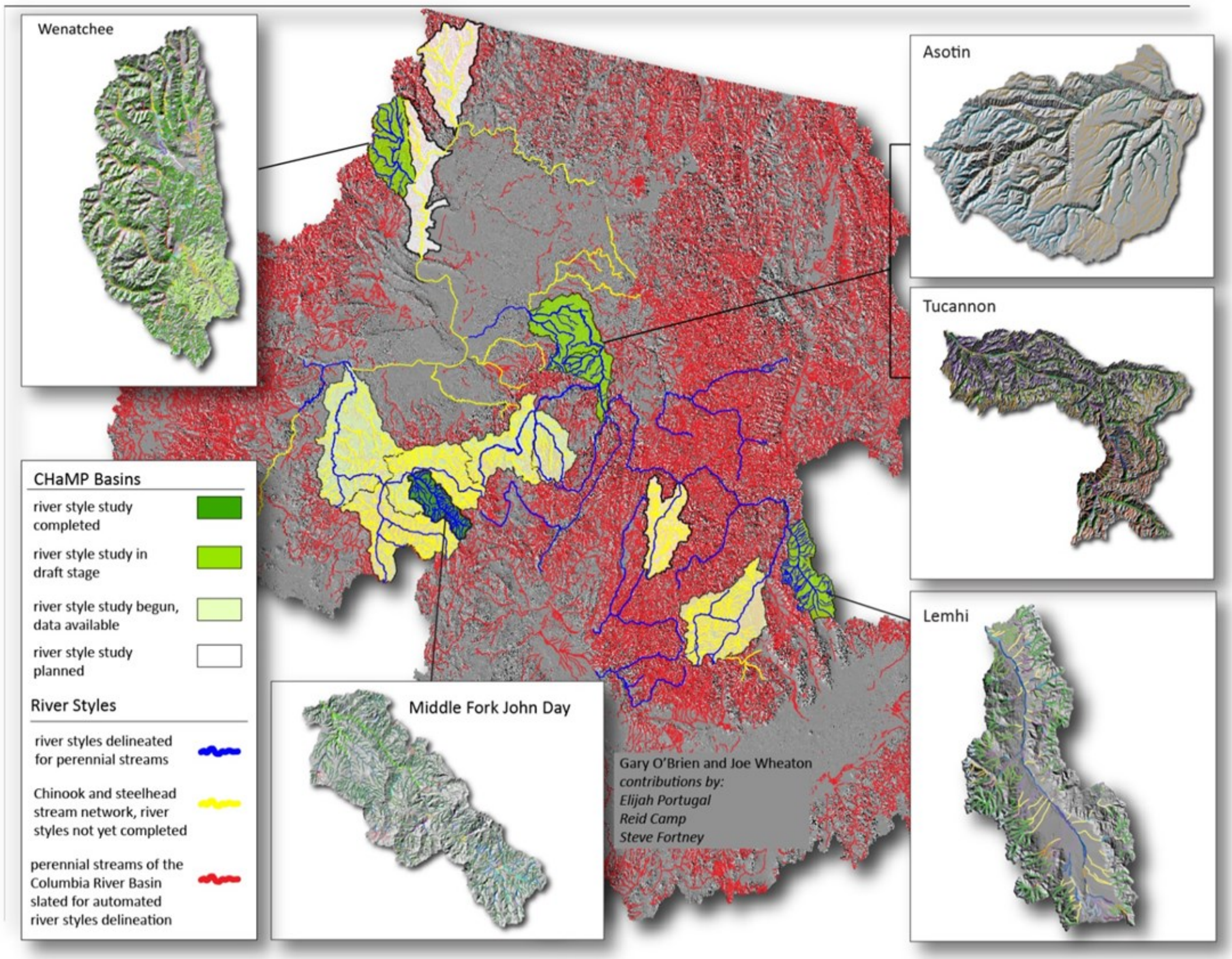


Figure 79. Progress of River Style analyses in CHaMP subbasins throughout the Columbia River Basin.

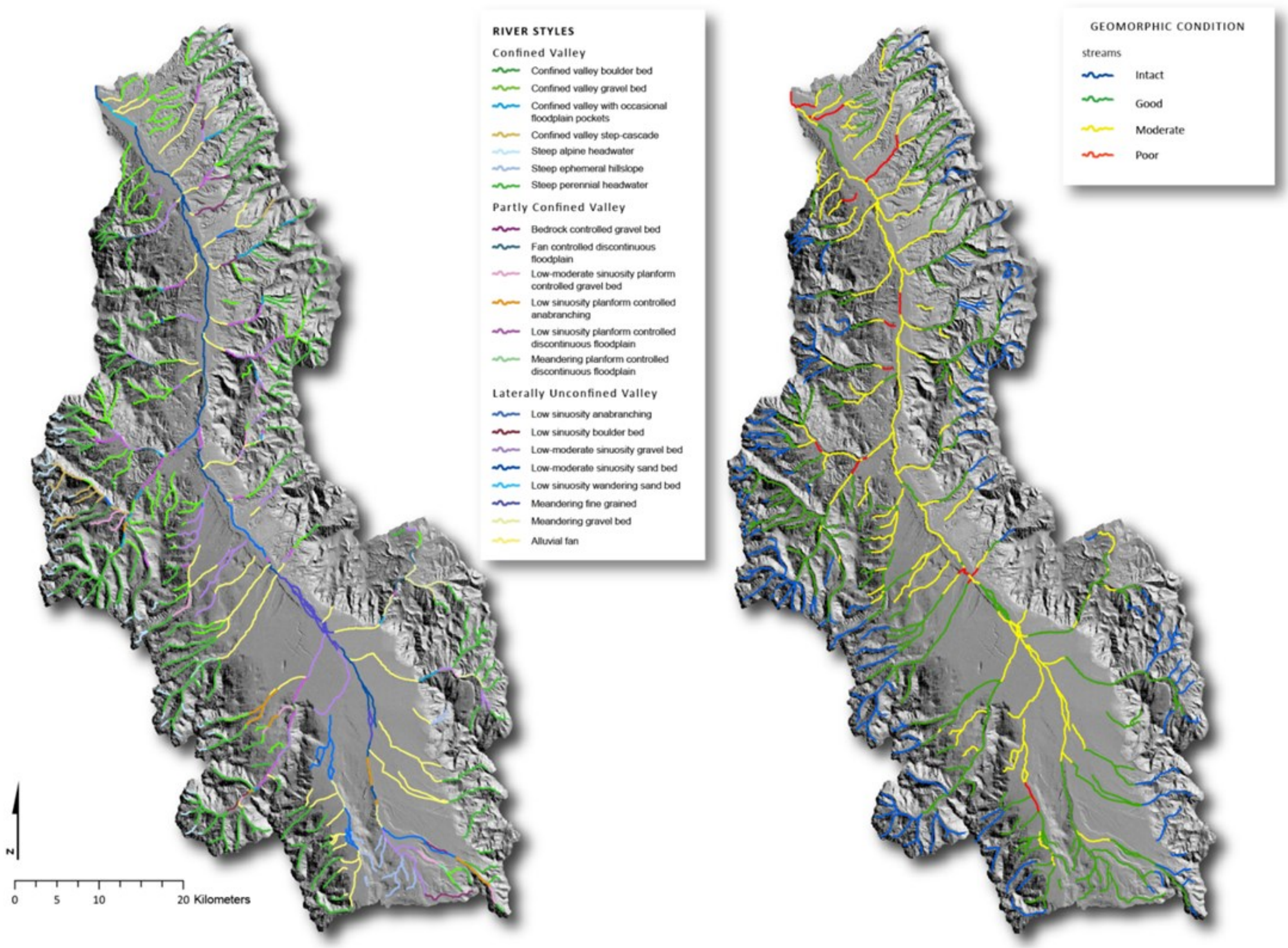


Figure 80. River Styles Stage 1 (character and behavior) and Stage 2 (geomorphic condition) defined for the Lemhi Watershed (HUC 8) in the southeast Idaho Batholith physiographic region.

Using Manual River Styles Designations for Automated Tool Testing

We are using valley setting information that we have developed manually to test a Valley Bottom Extraction Tool (V-BET), following the method of Gilbert et al. (2015). Initial V-BET validation and comparison of valley bottom outputs show excellent agreement between valley settings determined manually, the proximity of reach breaks, and their corresponding river styles. Testing is underway for completed products in the Middle Fork John Day and the Lemhi (Figure 81).

Summary

Landscape units provide geomorphic, geologic and biophysical control contexts for river character and behavior. We are using River Styles to define that context and validate continuous network maps of condition and recovery potential. The River Styles Procedural Tree we developed to represent the physical structure and organization of River Styles across CHaMP basins, and create standardization in the way trees and River Styles are developed and determined, is operational and being applied in CHaMP watersheds.

To date, the first three large regions (Idaho Batholith, Blue Mountains, and Upper Columbia) have been surveyed manually and River Styles trees produced. Ongoing efforts in the Asotin, Tucannon, Middle Fork John Day, Wenatchee and Lemhi Watersheds have used this procedure, and Kirsty Fryirs reviewed and approved it in late 2014.

The outputs from automated tools that we have developed to delineate valley bottom and channel sinuosity (Gilbert et al. 2015) show incredible promise (e.g., O'Brien et al. 2015). We believe that our application of River Styles combined with our application of new automated tools to characterize watersheds across the CRB in terms of River Styles, will pave the way for determining river character and behavior, geomorphic condi-

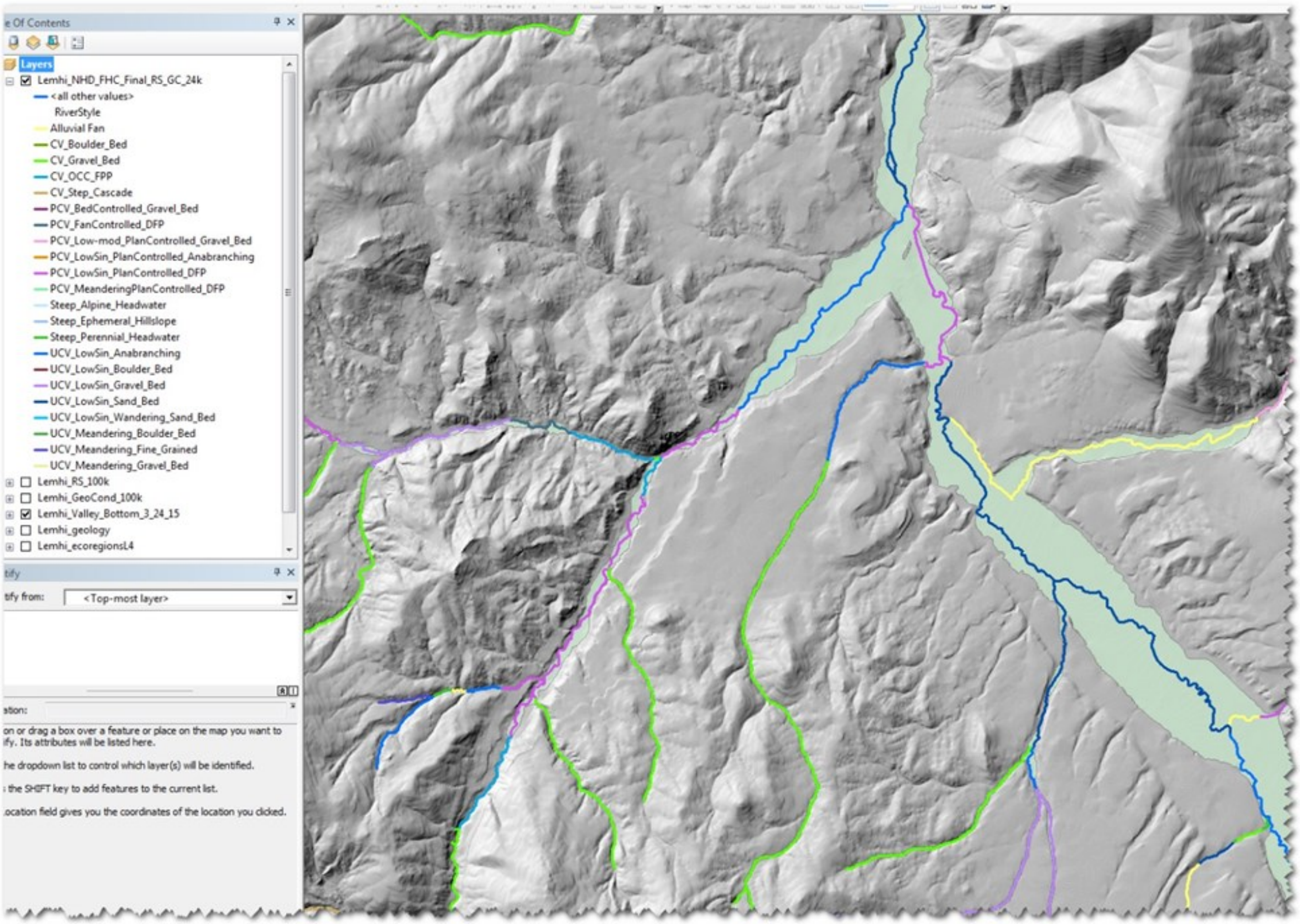


Figure 81. Screen shot of Lemhi Watershed river styles coupled with valley bottom delineation (Gilbert 2015) using the Valley Bottom Extraction Tool (V-BET).

tion, and river recovery potential across priority basins of the entire CRB region.

Effort in 2015 will be focused on continuing River Styles work and validation to enable the production of condition maps to support the 2016 Expert Panel process and 2018 AMIP process.

Recommendations

We recommend completing application of the River Styles framework and our procedural tree in all CHaMP watersheds, with the priority being additional work in the Columbia Plateau and North Cascades level 4 ecoregions, as these regions comprise the "heart" of the CRB and contain the majority of CHaMP and ISEMP study basins. These important level 4 ecoregions also encompass all of the River Styles projects currently in progress. We also recommend that River Styles work continue in other CHaMP watersheds (e.g., the Yankee Fork). Continued collaboration by geomorphologists and automated tool development will remain essential for cross-validation, and the production of reliable synthesis products to support the extrapolation of site-level CHaMP habitat data to larger scales and unsampled watersheds.

Geomorphic Unit Tool (GUT)

We have developed and are in the process of testing a Geomorphic Unit Tool (GUT) that will allow us to automate a portion of our River Styles implementation effort crews to support implementation of the process-based hierarchical geomorphic unit classification system (Figure 82) that we have developed with the authors of River Styles (Gary Briereley and Kirstie Fryirs). In short, we are working to generate geomorphic unit datasets directly from the CHaMP topographic data, rather than subjectively in the field. Our approach is rule-based and, as of

2014, largely automated (Figure 82).

Geomorphic unit modeling requires topographic data (e.g., LiDaR DEM or total station DEM) as the initial input. The final output geomorphic unit map can feed into other CHaMP products and analyses, such as interpreting habitat suitability maps (Figure 83). Each workflow step is detailed in the following subsections.

Evidence Rasters

Evidence rasters are the lines of evidence that a given cell is a particular geomorphic unit. Evidence rasters are derived

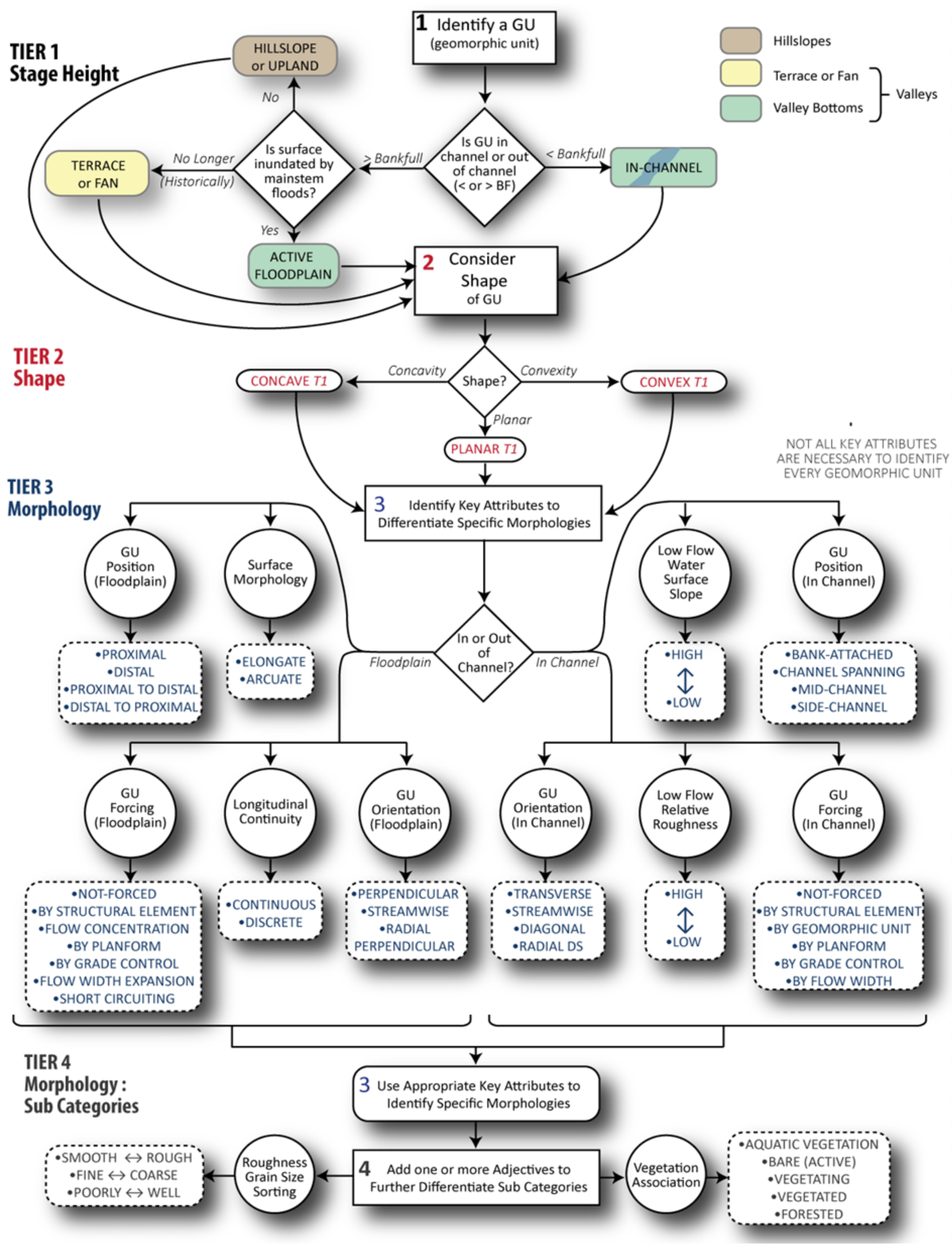


Figure 82. Three tiered geomorphic unit classification system.

from the input topographic data using various raster analyses (e.g., DEM surface slope). In general we use the same lines of evidence that an individual would use to delineate units in the field. For example, the evidence rasters used to delineate the active floodplain are slope, relief, and height above bankfull (all which should be relatively low). Many of the evidence rasters are normalized in order to use rules that do not have to be scaled across sites. In the context of concavities, the evidence raster are bankfull channel, normalized bankfull depth, and normalized concavity (Figure 84).

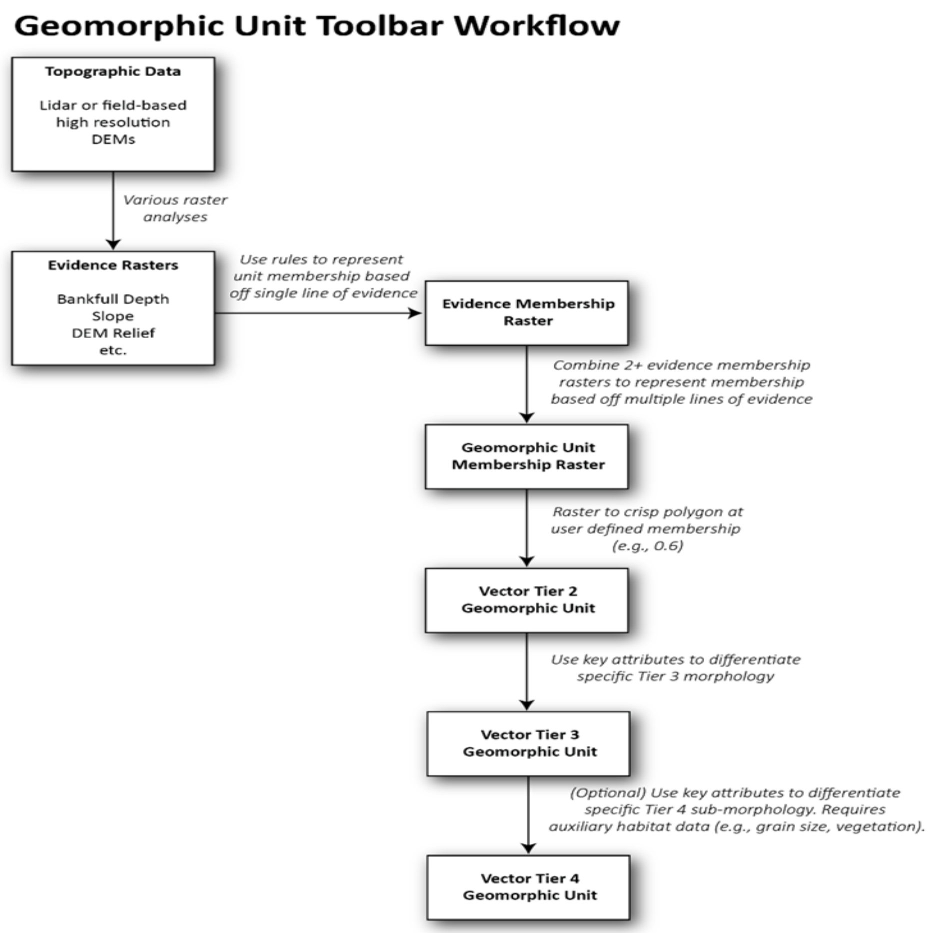


Figure 83. Tiered geomorphic unit delineation workflow diagram. CHaMP will delineate to tier 3 using a semi-automated workflow. Tier 4 delineation is optional and requires integration of auxiliary habitat data, such as grain size and vegetation associations.

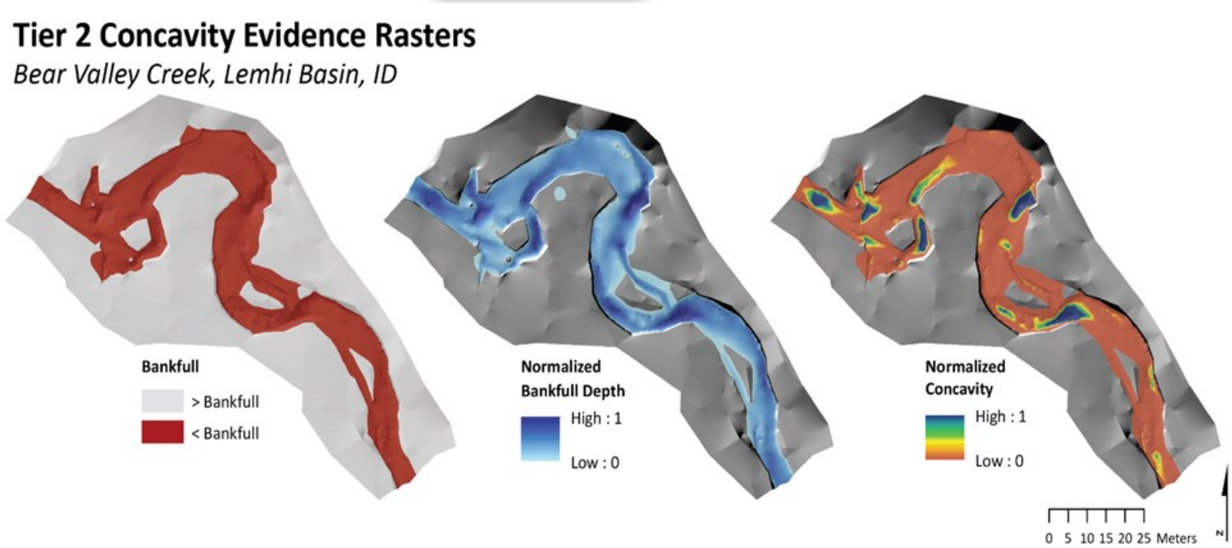


Figure 84. Concavity evidence rasters: bankfull, normalized bankfull depth, and normalized concavity. Data are for Bear Valley Creek, Lemhi River basin, Idaho (CBW05583-028079, Visit 1029) which is a 160 m long CHaMP site. The survey data are from the 2012 field season.

Evidence Membership Rasters

Evidence rasters are converted to evidence membership rasters using thresholds and transform functions (Figure 85). Transform functions are a set of conditional statements (or rules) that are applied to the evidence raster to output an evidence membership raster. Thresholds are derived either heuristically or empirically from raster value distributions across sites. We currently use linear transform functions due to their ease of interpretation, but alternative shapes, such as sigmoidal curves, could be used in the future. The output evidence membership raster represents the degree to which a given cell is a particular geomorphic unit (e.g., a concavity) based off of a single line of evidence (Figure 86).

Geomorphic Unit Membership Rasters

Rather than rely on a single line of evidence to model geomorphic unit membership we combine several (2+) individual evidence membership rasters to create a single geomorphic unit membership raster (e.g., concavity membership raster). The individual evidence probability raster are combined multiplicatively.

Vector Geomorphic Units

The resulting unit probability rasters are converted into a crisp tier-2 geomorphic unit polygon at any user defined value. Subsequently the area of each polygon is calculated. The area threshold varies based on geomorphic unit type (e.g., < 0.5 m2 for banks; < 1 * [bankfull width] m2 for convexities). Polygons with an area below the threshold are discarded and may then be classified as a different unit type.

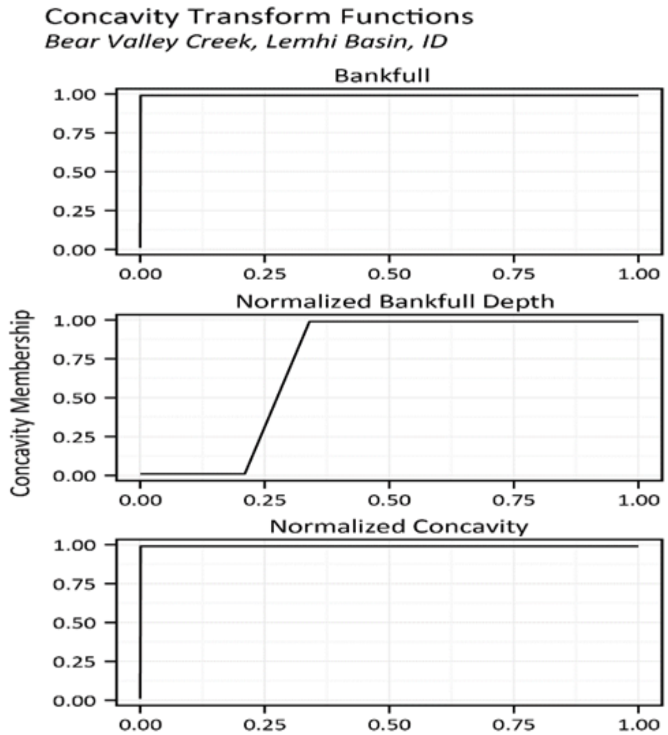


Figure 85. Concavity transform functions: bankfull, normalized bankfull depth, and normalized concavity. Transform functions are used to calculate geomorphic unit membership for each input evidence raster. The majority of the transform functions are 'hard set', however some (e.g., normalized bankfull depth) are based on the raster summary statistics.

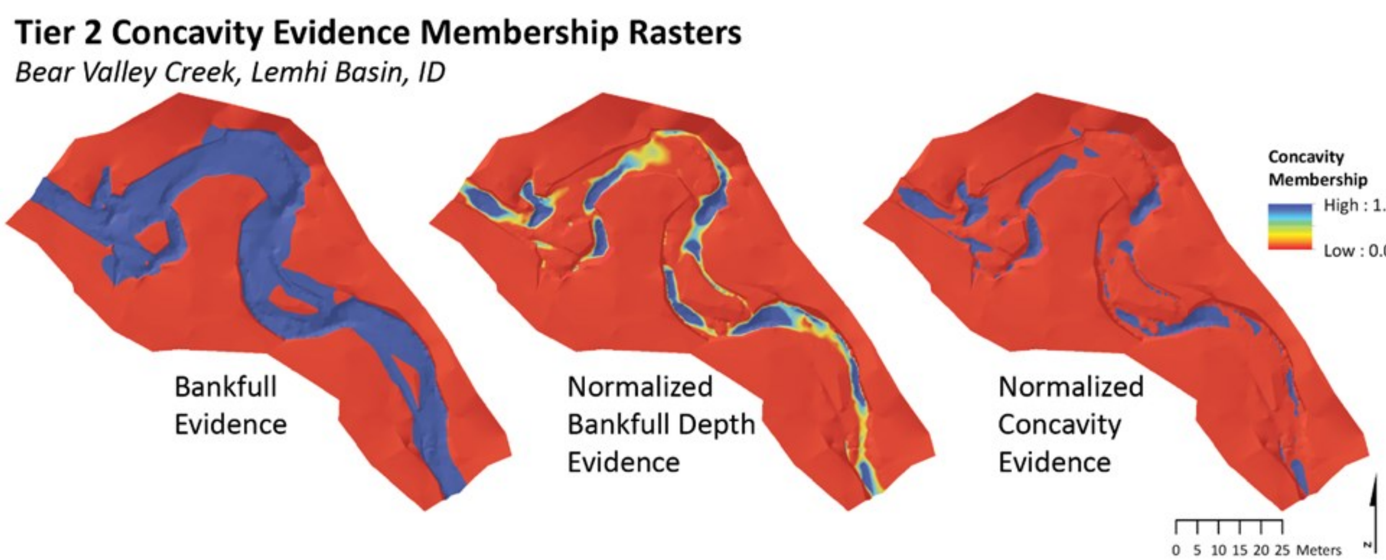


Figure 86. Evidence membership raster for bankfull, normalized bankfull depth, and normalized concavity. These rasters represent concavity membership based on a single line of evidence. Data are for Bear Valley Creek, Lemhi River basin, Idaho (CBW05583-028079, Visit 1029) which is a 160 m long CHaMP site. The survey data are from the 2012 field season.

Tier 3 describes the specific morphology of the geomorphic unit based on key attributes. These key attributes include: low water surface slope, unit position, unit orientation, low flow relative roughness, and unit forcing. The latter two attributes require auxiliary habitat information (e.g., grain size, location of forcing elements). Therefore, we are currently only using the first three attributes to determine specific morphology. Orientation is simply determined by whether a unit is longer than wide (streamwise) or wider than long (transverse). Radial and diagonal are two rare cases that we are currently not modeling for. Position is based off of water's edge, instead of bankfull, so that position can be dynamic with stage. However, each CHaMP survey will Criteria for determining positions is:

- Channel spanning: within 10% average wetted width of 2 water edges
- Bank-attached: within 10% average wetted width of only 1 water edge
- Mid channel: not within 10% average wetted width to any water edge

Example Results - Lemhi

Modeled geomorphic unit results are showing for CHaMP site CBW05583-028079 which is located on Bear Valley Creek in the Lemhi River basin, Idaho. The tier 2 geomorphic unit membership rasters, and crisp unit polygons at 50% membership, are depicted in Figure 87 and Figure 88, respectively.

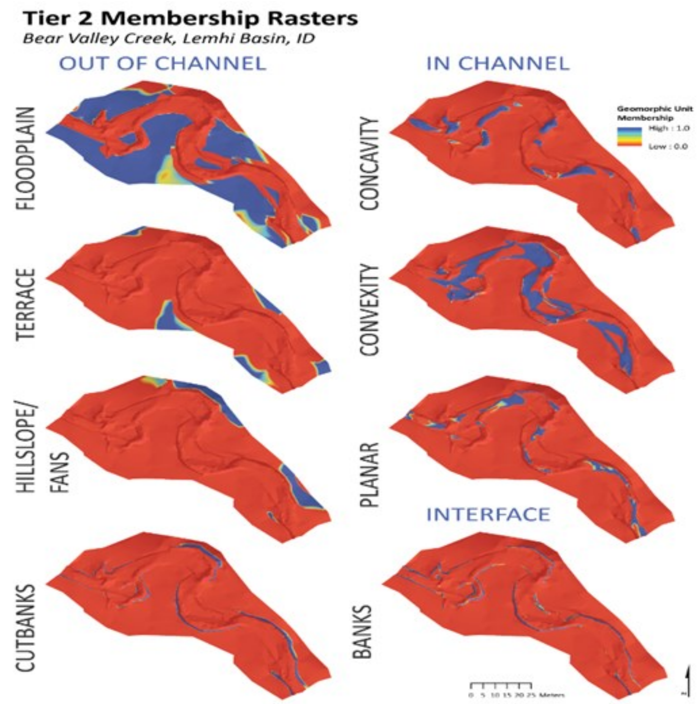


Figure 87. Tier 2 geomorphic unit membership rasters. Unit membership ranges from 0 (no membership, red) to 1 (full membership, blue). Data are for Bear Valley Creek, Lemhi River basin, Idaho (CBW05583-028079, Visit 1029) which is a 160 m long CHaMP site. The survey data are from the 2012 field season. See Figure 7 for watershed context of this site.

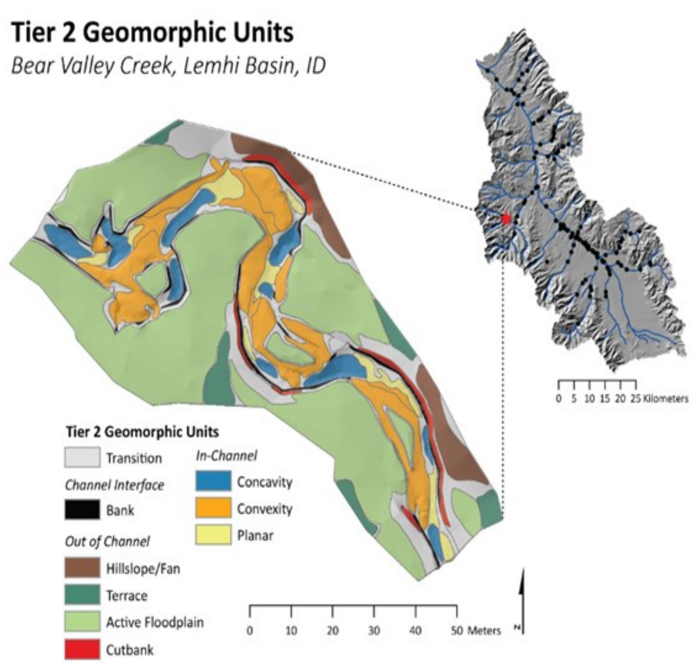


Figure 88. Tier 2 'crisp' geomorphic units at > 0.5 membership (left). Data are for Bear Valley Creek, Lemhi River basin, Idaho (CBW05583-028079, Visit 1029) which is a 160 m long CHaMP site. The survey data are from the 2012 field season. Lemhi watershed map (right) is provided for context. Black circles show location of all CHaMP sites in the basin. Red star shows location of Bear Valley Creek site CBW05583-028079.

Validating modeled geomorphic units

We conducted a validation test comparing modeled geomorphic units (thresholded at a 50% membership) with a desktop manual classification using 2012 CHaMP data for a Bear Valley Creek site (site CBW05583-028079). We completed the desktop manual in ArcGIS following the classification diagram shown in Figure 82 and using several map layers (e.g., 10 cm contours, hillshade, slope raster, aerial imagery). The manual classification was treated as the more accurate geomorphic unit representation than either the field or automated classification. Transition units were removed prior to the comparison since these units are highly dependent on the chosen membership value threshold. Modeled cutbanks were re-classified as banks because cutbanks were not included as a unit in the original manual classification. A cell-by-cell agreement matrix was calculated, showing the percentage of correctly classified and misclassified cells (Table 36). We found high agreement among several units, including: concavities (80%), convexities (88%), banks (88%), and floodplain (92%). Hillslope/fan agreement was 64% with modeled cells also classified as floodplain (22%) and bank (10%). The latter two classifications were due to, respectively, cells being low sloping and high sloping but close the channel. We infer the manual classification may have overly generalized hillslopes and that, in this instance, the model performed better. Planar agreement was 50% with modeled cells classified as convexity (46%), concavity (3%), and bank (1%). There can be subtle differences between slight convexities and planar units. The results here indicate we need to work on discriminating these features.

Recommendations

Tier 2 delineation in GUT is at an 'Operational' stage meaning the tool has a streamlined code capable of being run by experienced analysts, and generally has a semi-automated workflow, with key stages of user input and QA/QC. Tier 3 delineation in GUT is at a 'Prototype' meaning the tool has some code that the analyst who developed it can run the analysis, but it is generally a very manual workflow and has not been automated. Recommendations moving forward into 2015 include:

- Complete Tier 3 classification workflow. Our current goal is to classify as many different Tier 3 units as possible. However, we may find that we can only define units based on unit position and orientation, which would limit the overall number of units we can classify.
- Test GUT using a larger sample of CHaMP sites (i.e., a diverse sample of sites, e.g., bankfull width category, slope, channel type)
- Develop and test tool automation
- Continue to improve/trouble shoot the automated GUT for use at CHaMP sites.
- Automate for all previous visits. In the long-term, replace channel units with geomorphic units, but always be able to crosswalk these geomorphic units back to channel units.

Table 36. Tier 2 manually classified vs modeled geomorphic unit cell-by-cell agreement matrix. High percentages between identical categories indicates high agreement. Transition units were omitted from the validation since, in the model, these units are highly dependent on the threshold membership value. Data are for Bear Valley Creek, Lemhi River basin, Idaho (CBW05583-028079, Visit 1029) which is a 160 m long CHaMP site. The survey data are from the 2012 field season. See figure 7 for watershed context of this site.

		Manually Mapped						
		In Channel			Interface	Out of Channel		
		Concavity	Convexity	Planar	Bank	Floodplain	Hillslope/Fan	Terrace
Modeled	Concavity	80%	0%	3%	1%	0%	0%	0%
	Convexity	6%	88%	46%	9%	0%	0%	0%
	Planar	13%	6%	50%	0%	0%	0%	0%
	Bank	2%	1%	1%	88%	1%	10%	0%
	Floodplain	0%	5%	0%	2%	92%	22%	0%
	Hillslope/Fan	0%	0%	0%	0%	0%	64%	0%
	Terrace	0%	0%	0%	0%	7%	4%	0%

Hydraulic Modeling: Building the Link Between Site-level CHaMP Measurements and Mechanistic Habitat -Capacity Models

Hydraulic models are a key linkage being used in CHaMP to relate stream hydraulics to juvenile salmonid population dynamics. We have used CHaMP data to develop hydraulic models for the majority of more than 600 reaches at which the CHaMP program collects habitat data for use in ISEMP-CHaMP products such as NREI and habitat suitability index (HSI) models. The hydraulic modeling approach we have developed aims to provide hydraulic models capable of supporting our research, in terms of precision accuracy, as well as in the ability to generate unique hydraulic models at multiple flow conditions for every site.

Multiple CHaMP data sources are used as inputs to the hydraulic models (see CHaMP 2015). All information used to generate model inputs are generated as part of default CHaMP data collection procedures. To date we have successfully completed more than 1000 hydraulic models, distributed across each sample year and primary CHaMP watershed (Table 37). Model results are publically available via champmonitoring.org.

Modeling Strategy

Our primary objective is to generate field estimates for depth and velocity, defined over the maximum practical spatial extent of each of our CHaMP surveys. Other hydraulic model outputs, such as vorticity and bed shear stress, are simply functions of the velocity and depth fields. Our additional challenge was to devise a modeling strategy that, in addition to providing accurate and spatially fine results, enables automation of the modeling process in order to produce thousands of hydraulic models. There are more than 600 unique CHaMP sites, and each CHaMP site will be surveyed from between three and nine times throughout the life of the CHaMP study design. Thus, at minimum, we will have several thousand models to run. In addition to generating each hydraulic model, we also seek to generate easy to interpret quality control feedback, informative of the success and accuracy of each model.

We use Delft-3D Flow (http://oss.deltares.nl/web/delft3dto) to model fluid flow at our CHaMP reaches. Delft-3D flow is an open source, freely available software with modeling capabilities for free surface flows across a wide range of spatial scales (Deltares 2013a). We chose Delft 3D not only because it is open source and freely available, but also because it is highly flexible and capable of modeling fluid flows that meet our current needs and potentially a broader suite of needs well beyond our current objectives. In addition to supporting the capabilities required for CHaMP hydraulic modeling, Delft 3D was selected as it is capable of being run in batch mode, suitable for our need to model large numbers of CHaMP reaches.

Inputs required for the hydraulic modeling are derived from CHaMP data, including DEMs for both reach level bathymetry and water surface level elevation, an estimate of surface roughness, and water discharge rate. Required inputs describing modeled geometry, boundary conditions, initial conditions, fluidic properties, and numerical parameters are input to Delft 3D Flow as a series of input text files (Deltares 2013a).

Table 37. Number of successfully completed hydraulic models by watershed, year combination.

CHaMP Watershed	Visit Year			
	2011	2012	2013	2014
Methow	19	13	22	19
Entiat	24	33	48	32
Wenatchee	19	17	20	19
Tucannon	20	24	24	24
John Day	52	77	51	39
Upper Grande Ronde	70	46	52	54
Lemhi	39	46	35	21
South Fork Salmon	32	26	21	20
Totals	275	282	273	228
	1,058.00			

Our modeling strategy reflects our objective to model high numbers of sites across a range of conditions, rather than intensively study a small number of sites. Where practical, we opted for simplicity, generally at the expense of computational efficiency. For example, we use simple rectilinear computational grids, rather than curvilinear or adaptive mesh grids. Our grid spacing is often finer than needed for much of the modeled flow, thus the computational intensity is perhaps greater than would be required if using a curvilinear grid. However, given the abundance of computational power available, in automating the process we found it vastly more effective to use simple rectilinear grids at the expense of computational efficiency, rather than add the complexity of attempting to automate and validate curvilinear grids for every site modeled.

For pre- and post-processing, we created scripts written in the R programming language (R Core Team, 2014). The preprocessing script reads a comma separated value (csv) files containing: the DEM, which is generated from surveyed bathymetry; the water surface elevation digital elevation model (WSEDEM), also generated from the CHaMP survey; a csv file describing the surveyed thalweg location; and a file containing discharge, the 84th percentile value for site pebble size distribution (D84), and additional meta-data used for book-keeping and process tracking. The R script converts the input data into a series of input files formatted for the Delft-3D Flow software package. The R script also generates a file of meta-data to be passed through the process and augmented during post-processing, and a suite of quality assurance plots form which the user can quickly confirm that boundary conditions and input files have been generated correctly.

In addition to the standard set of Delft 3D input files, our pre-processing script generates a batch file script and accompanying xml file that enable running of the Delft 3D software in batch mode, bypassing any need for manual operation within the Delft 3D graphical user interface. This process is necessary to meet our high volume automation objective. Our pre-processing script also generates a macro that, after the Delft 3D Flow simulation is complete, runs the Delft 3D supplied Quickplot tool (Deltares 2013b) to convert Flow 3D results into a text format that can be read by a post-processing R script.

Quality Assurance

A quality assurance plot showing the grid extents, wetted area, thalweg, upstream boundary condition extent and downstream boundary extent is also generated by the pre-processing script (Figure 89). This plot provides a quick visual confirmation that the input files have been read successfully and that boundary conditions are appropriate. An additional R script conducts post-processing of the Delft 3D Flow results. Delft 3D Flow output, converted from Delft 3D Flow into a set of text files using the Quickplot tool, is read back into R, and results are interpolated back onto the spatial points defined on the original DEM grid. A csv output file of results is generated, as are a series of contour plots used for quality assurance. These QA plots visually display field results showing velocity, depth, water surface elevation, and bed shear stress. Plots of spatially explicit estimates of error in modeled depth are also calculated as the difference between surveyed depth and modeled depth. Plots of this error provide a quick visual assessment of model accuracy.

Grid Spacing

During development, we found we could consistently run models on computational grids containing approximately 500,000 grid points. Beyond that, memory requirements and computational requirements limited our ability to run successful models. Therefore, grid spacing used for the computational grid varies by the size of stream reach being modeled, such that we attempt to use as fine of a grid as computational practical without exceeding the 500,000 grid point limit. Additionally, we limit grid spacing options such that grid spacing is either a multiple of the 0.1 m DEM grid spacing, or an integer fraction of the DEM spacing, such that allowable computational grid spacing includes values such as .4, .2, .1, .05, or 0.025 meters, etc. (Figure 90). We found this simplified that process of translating and/or interpolating data to and from the DEM grid to the computational grid.

To ensure our resulting grid spacing is sufficiently fine, we compared results of simulations run at grid spacing as described above, to those run at coarser grid spacing, across a variety of CHaMP reaches. Varying grid spacing demonstrated that the grid spacing, as determined from our algorithm, appears to be sufficiently fine. Doubling the grid spacing resulted in only minor deviations in velocity fields and corresponding depth

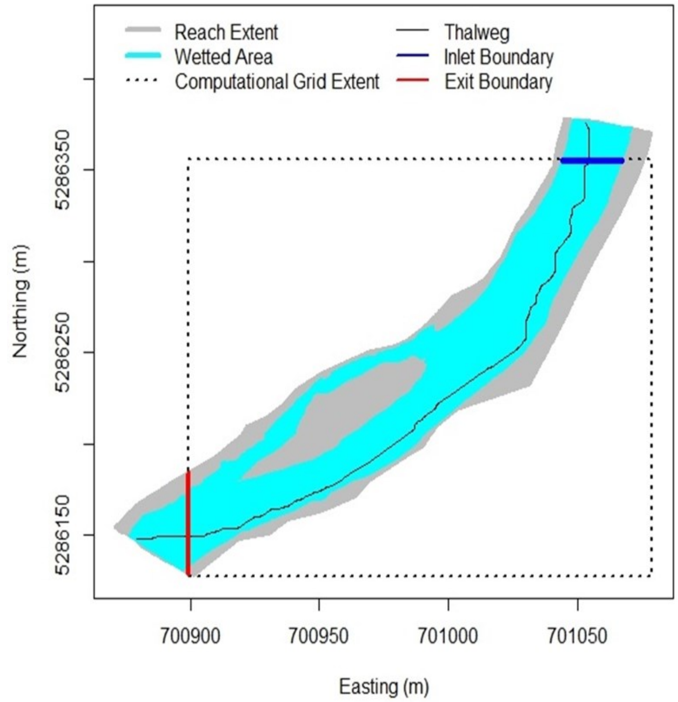


Figure 89. Extent of reach surveyed, computational grid extents, and inlet/exit boundary locations.

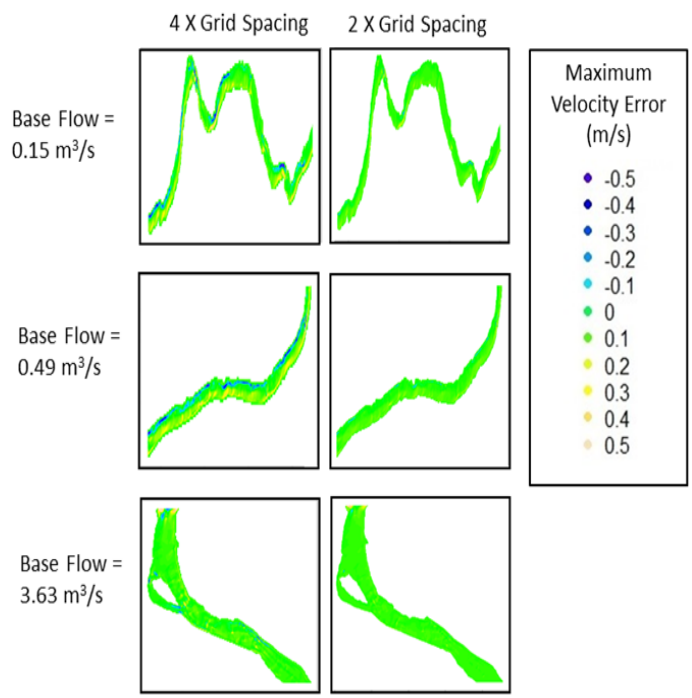


Figure 90. Velocity magnitude differences, relative to simulations at default grid spacing, for simulations performed at 4X and 2X the default grid spacing, for low, medium, and high flow CHaMP reaches.

fields. As grid spacing is further increased to 4X the default grid spacing, we observe significant differences in velocity and depth fields, indicating that grid spacing should not be coarsened to this level.

Boundary Conditions

Boundary conditions are specified at the upstream and downstream computational boundaries (Figure 89). The upstream and downstream boundaries are determined by determining which boundary (North, South, East, or West) the thalweg crosses at the upstream and downstream ends, respectively. Note that the inlet and outlet can occur on the same edge of the computational grid.

Hydraulic discharge in m^3/s is specified at the upstream computational boundary. This boundary condition is applied over the wetted length of the computational boundary crossing the upstream portion of the stream reach. Wetted length is defined as any portion of the inlet boundary with positive water depth, as determined by the difference in water surface elevation DEM and the DEM of bathymetry. The total discharge is distributed along each cell of the inlet boundary such that the volume flow rate at each cell is proportional to the measured water depth.

Water surface elevation is specified as the downstream boundary condition. Water surface elevation at the downstream boundary is estimated as the average water surface elevation at all wetted (according to the WSEDEM) points along the exit boundary. We specify the downstream boundary at all points along the exit face where the elevation is equal to or lower than the downstream water surface elevation, unless such points were already defined as inlet boundary locations. Because the inlet and outlet boundary conditions are specified at edges along cardinal directions, there is, in most cases, some boundary condition specification error due to the fact that the flow direction of the boundaries is rarely orthogonal to the boundary. In some cases, the wetted boundary edge, specified as a boundary condition along one edge only, may actually extend around a corner to an adjacent edge. Thus, as in most fluidic modeling, caution should be used in use and interpretation of modeled results near the computational grid boundary. However, experimentation with boundary conditions suggests that boundary condition errors typically propagate no more than 2-3 wetted widths upstream from the exit boundary or downstream from the inlet boundary.

Initial Conditions

Initial conditions for the model are set such that the water level at all points is set to the water level at the downstream boundary condition. Where the bed elevation is greater than the downstream boundary condition, no water is present at t=0. Initial velocity is zero for all wetted areas at t=0. While we recognize that computational time required to reach a steady state solution could potentially be improved by setting initial water levels closer to those as surveyed, we found that the steady state solution was not dependent on initial conditions.

Surface Roughness and Model Calibration

It is impractical, both in terms of computational power and our ability to create DEMs at high enough precision, to include features in a DEM that can be described as “surface roughness”: pebbles, small rocks, etc. In CHaMP streams, surface roughness is primarily driven by the distribution of pebble sizes in the substrate, especially in the shallower, higher velocity channel units. Because features at this spatial level cannot be modeled directly, a model correcting for surface roughness is necessary. From the options available in the Delft 3D Flow software, we chose the White-Colebrook (Colebrook and White 1937; Colebrook 1939) model. Surface roughness in the X and Y directions are inputs to this model. We assume equal roughness in the X and Y directions, and that we can use information about pebble size distribution CHaMP as a proxy for surface roughness.

Pebble size distribution is measured at all CHaMP sites by randomly selecting pebbles at a series of transects across the wetted width of the stream. The 16th, 50th, and 84th percentile pebble sizes are reported. We expect that hydraulic fields on the scale of interest are more affected by larger pebbles than smaller pebbles. Therefore, the CHaMP metric for the 84th percentile pebble size (D84) was considered as a potential indicator of surface roughness to use as inputs to the Delft 3D Flow model. We assumed some scalar value of D84 would provide a reasonable proxy for surface roughness, and thus used a scalar multiple of D84 as a means of model calibration. The scaling factor was varied over a range of values from 1 to 8, and resulting velocity and depth fields modeled at each scalar value were compared to a series of validation points, where velocity and depth were measured directly at a series of points along a series of transects, at a subset of CHaMP sites. We used 31 sites where validation data were collected, with from three to six transects collected per site, and up to 21 points per transect directly measured. The selection of our scalar on D84 to be used to input surface roughness was selected as the value that minimized the overall error when comparing modeled results to validation results.

Using a scalar multiplier on the CHaMP metric D84 proved effective at calibrating the model. As the multiplier was increased, modeled depth tended to decrease, while modeled velocity tended to increase (Figure 91). At a D84 multiplier of approximately 3.0, velocity and depth errors are minimized. We therefore use this multiplier for all CHaMP sites. Because our intention is to model thousands of CHaMP site / visit combinations, we use a single value for all sites, rather than attempt to optimize on a site by site basis.

Simulation Time

Typically in computational fluidic modeling, simulations would be run through simulated time until the user was satis-

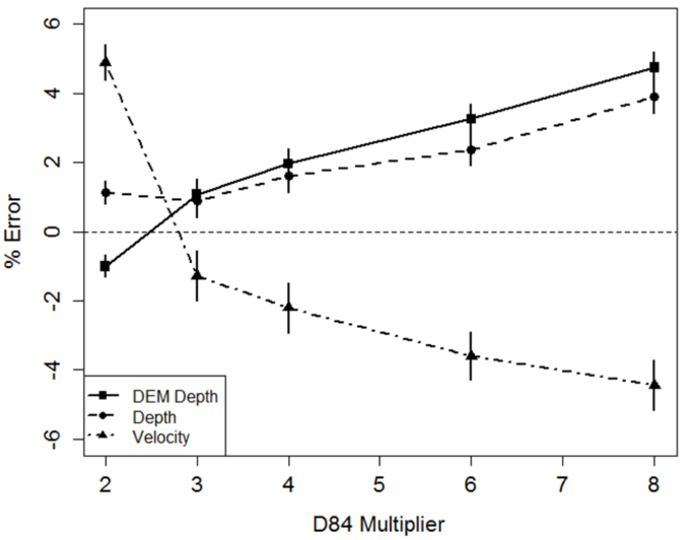


Figure 91. Estimated mean error at validation locations vs multiplier applied to scale D84 as surface roughness input to model. Error is defined as the percent difference between modeled values for a) depth as measured in the DEM survey, b) direct depth measurements at validation points, and c) direct velocity measurements at validation points. Vertical bars indicate 95% confidence bounds.

fied, via some quantitative feedback, that the solution has reach a steady or quasi-steady state. Because our objective is to automate high numbers of simulations, we developed a conservative rule for total simulation time over which to run the simulations. From the DEM and water surface elevation DEM information, we estimate the volume of water present in the reach to be modeled. We run the simulation such that the rate of discharge multiplied by the simulation time is equal to twice the total water volume of the reach. Typically the simulation reaches a steady state when the total discharged volume is roughly equal to the total site volume, and we've found that doubling this simulation time seems to provide ample margin to ensure all simulations reach a steady state solution.

Computation Time

For CHaMP reaches modeled thus far, the clock time required to run each simulation can vary from a few minutes to as long as 24 hours, depending on the size of the reach and the discharge rates. Typically large reaches take longer to model, and sites with low discharge rates require longer times to model. Modeling all CHaMP sites in this manner is not practical at a single computer. Therefore, we take advantage of distributed cloud computing, running a single site on a single instance of the solution code set on each of multiple copies, accessed via the cloud.

Outputs

We have selected a list of outputs as deliverables from our model (Table 38). Outputs are recorded on a regularly spaced 10 cm grid, at all points that are either a) wetted, according to the hydraulic model solution, or b) wetted, according to the original crew survey. Outputs included for each grid point include: X and Y location (where X and Y are northing and easting, respectively), velocity vectors in X and Y directions, as well as resulting velocity magnitude, depth, and depth error (estimated as the difference between depth estimated in the survey, and depth estimated via the hydraulic model). In addition, a set of higher level attributes such as shear stress are output. These are calculated from the velocity and depth fields within the Delft 3D program.

Results

We have been successful thus far at creating more than 600 hydraulic models. For the CHaMP sites we have attempted to model to date, we have successfully generated hydraulic model results for more than 97% of sites. Model failures and other problem areas are discussed below. Publically available results for all sites modeled are available at *champmonitoring.org*.

Table 38. CHaMP site hydraulic modeling output written to each row of the .csv output file. The output file contains one row for each point on a uniform 0.1 m rectilinear grid overlaying the CHaMP site.

Output	Description	Units
X, Y	Geographic Cartesian coordinates for Northing and Easting, respectively, in meters	m
X Velocity, Y Velocity	X and Y vector components of velocity	m/s
Velocity Magnitude	Magnitude of resultant velocity vector	m/s
Depth	Water depth	m
WSE	Elevation of water surface, above sea level	m
Bed Level	Elevation of bed, above sea level	M
Bed Shear X, Bed Shear Y	X and Y vector components of bed shear stress	N/m^2
Depth Error	Difference between surveyed depth and modeled depth	m

Plots of velocity and depth, as measured at selected transect location for selected sites, compared to plots of modeled velocity and depth, as well as depth as surveyed and reflected in the DEM, show generally good agreement between modeled and measured values (Figure 92 and Figure 93). In general, modeled velocity and depth profiles are much smoother than direct measurements. Depth profiles as measured in the survey process, as reflected in the DEM, are also much smoother for modeled depth than directly measured depth. This is as expected, since the survey process operates at spatial scales larger than small localized features such as rocks, cracks, woody debris, etc. The precision of the modeled values are reflective of the DEM. Nevertheless, we find that, while the modeled values

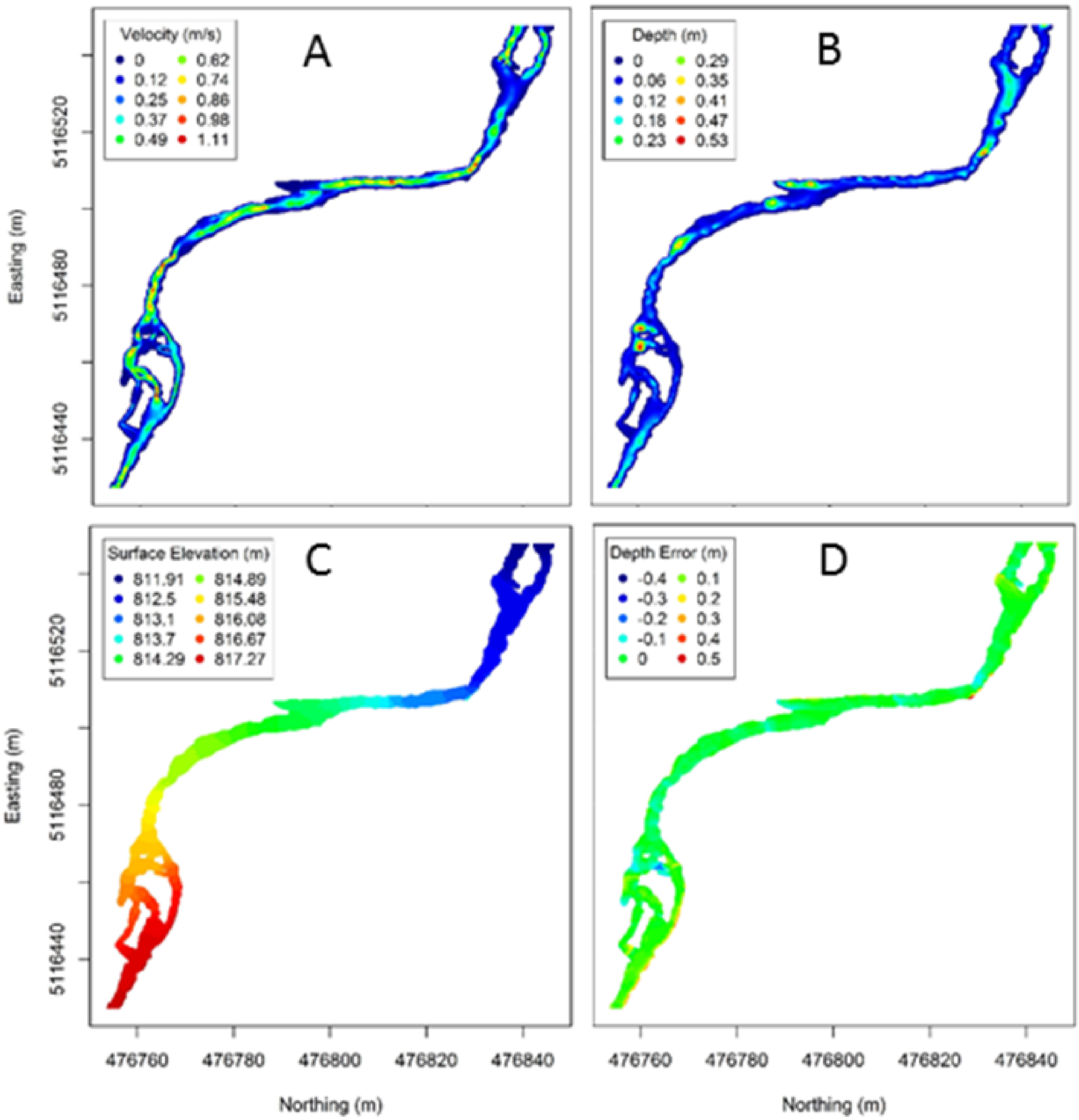


Figure 92. Velocity (A), depth (B), surface elevation (C), and the depth error estimated as the difference between surveyed depth and modeled depth (D), for CHaMP site ASW00001-SF-F5_P3BR.

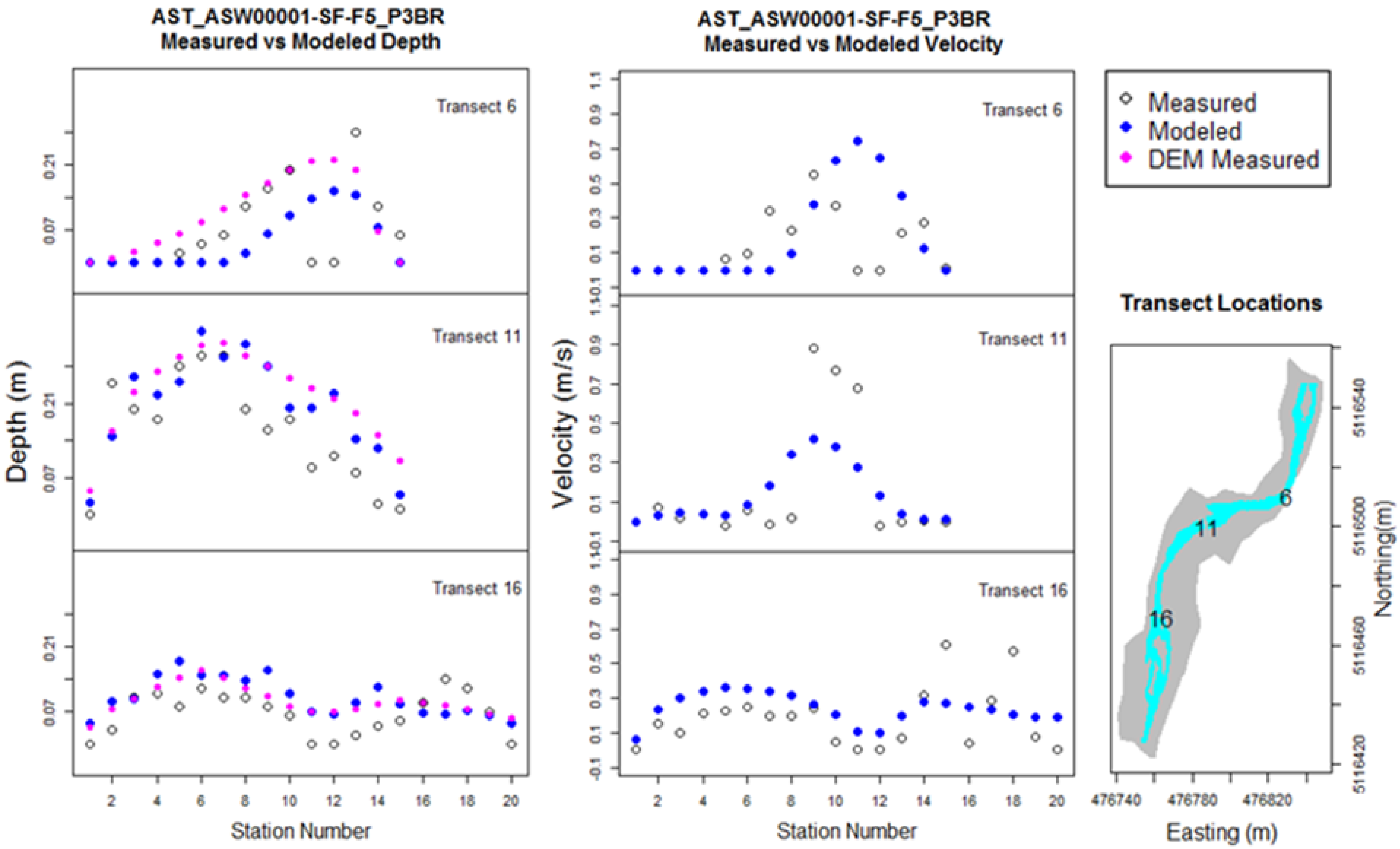


Figure 93. Example modeled depth and velocity compared to measured depth and velocity. DEM measured depth is depth derived from the DEM survey. Measured depth and velocity are direct measurements at transect locations. Transect locations are shown in inset map.

are smoothed out, they tend to match the overall depth and velocity profiles measured directly.

Problem Areas

Examination of spatial plots of the difference between modeled depth surveyed depth, as reflected in the DEM and water surface elevation DoDs, suggest that, for most sites, modeled results accurately reflect measured values; however, there are some riparian features have been problematic thus far: undercut banks, and large, sometimes porous woody structures. While undercut banks and porous woody structures are assessed as part of the CHaMP protocol, sufficient spatial detail is not recorded from which to include dimensions of such features in the DEMs; thus, impacts to the flow and velocity fields directly resulting from such features are not reflected in the hydraulic models.

For example, at CHaMP site ENT0001-1E3, the survey crew noted and photographed a large tree that had fallen across most of the width of the channel near the downstream extent of the reach. CHaMP bathymetry measurements do not include fallen trees or other woody debris (although spatially non-explicit measures of woody debris are obtained by survey crews); however, the impact of such woody debris is reflected in the surveyed water surface elevation. As the hydraulic model is driven by bathymetry, discharge rates, and boundary conditions, it cannot account for the effect of the fallen log. Therefore, we see large, localized depth field errors, and presumably analogous errors in the velocity field, immediately upstream of the location of the fallen log (Figure 94). In this case, the anomaly occurs near the downstream boundary condition; thus, the error is propagated as an underestimate of depth upstream of the anomaly, rather than an overestimate of downstream depth below the anomaly, or a combination of the two.

Undercuts present another problem for CHaMP hydraulic models. DEMs from CHaMP bathymetry surveys do not capture undercut area or depth; the DEM reflects a stream bank that runs vertically down from the edge of the overhanging bank, rather than an undercut bank. At CHaMP site ASW00001-NF-F4_P1BR, the crew observed and photographed a considerable undercut bank, and this undercut is not represented in the DEM. The modeled reach has, at undercut cross sections, a smaller than actual cross section width at wetted depths. The modeled flow is therefore more constrained than the actual flow, and the resulting modeled depth is greater the actual depth near the undercut locations (Figure 95). It should be ex-

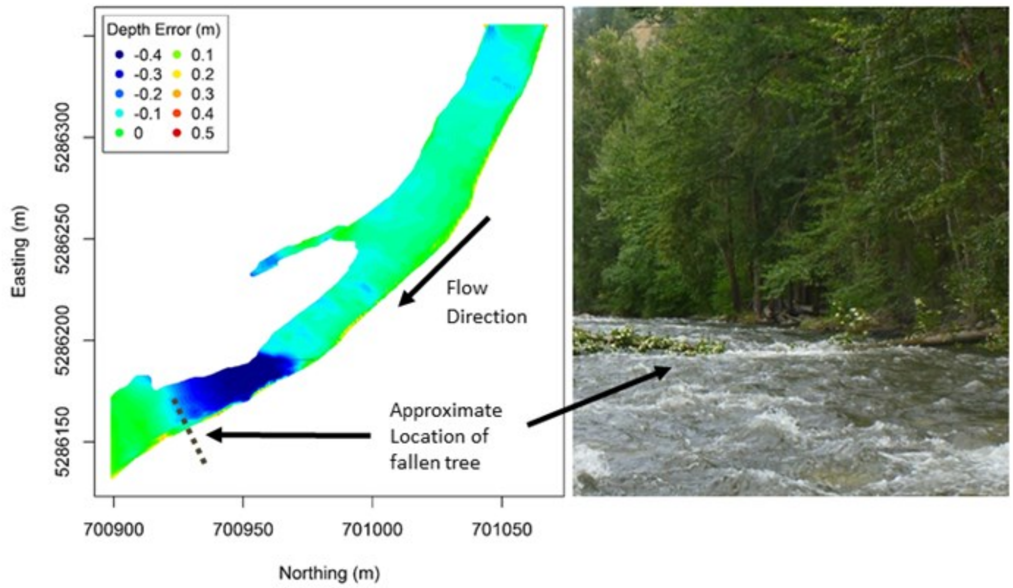


Figure 94. Depth error, with respect to surveyed depth, for CHaMP site ENT0001 -1E3. Localized area where modeled depth is underestimated, likely due to a fallen log in river. Logs, shrubs, and other woody debris is not reflected in the DEM, thus the increase in water surface elevation upstream from the log is not reflected in the hydraulic model.

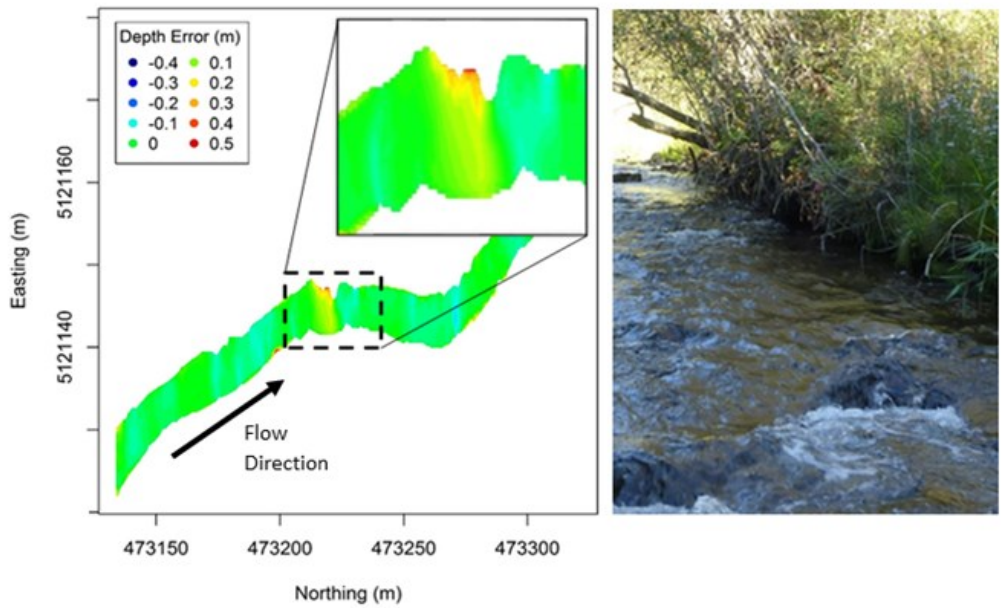


Figure 95. Depth error, with respect to surveyed depth, for CHaMP site ASW00001-NF-F4_P1BR. The undercut bank, on the river left, is not reflected in the DEM. The hydraulic model therefore over-predicts depth because the modeled cross sectional area is less than the actual cross sectional area, by the amount of area in the undercut.

pected that modeled velocity at these locations is greater than actual velocity as well.

Modeling Across Ranges of Discharge Rates

Future plans for CHaMP hydraulic modeling include estimating flow and velocity fields over a range of discharge rates, to model flow conditions different from those present when the DEM data were obtained. Problematically, the downstream water surface elevation is unknown, thus the downstream boundary condition cannot be specified accurately. While CHaMP is considering other data sources from which to estimates the downstream boundary condition, we can presently provide insight into the maximum range and extent of error propagation resulting from assumed, rather than measured, downstream boundary conditions. To examine this, we modeled three CHaMP reaches – one each for relatively high flow, medium flow, and low flow rate streams, at both twice and half their measured discharge, using two downstream boundary conditions for each: One boundary condition specified no change in downstream water surface elevation, while the other specified a change in downstream water surface elevation such that the wetted cross sectional area is scaled in proportion to the change in discharge. In reality, it is reasonable to expect the correct downstream water surface elevation to be somewhere in between these two extremes, as the flow can be expected to get somewhat deeper and somewhat faster at increased discharge rates. The difference in velocity and/or depth fields between these two extremes provides a worst-case bound of the extent of error induced by an unknown, assumed downstream boundary condition, as well as a worst case limit for the extent of such error propagation. We found that, at worst case, the extent of errors introduced by varying discharge at an unknown downstream water surface elevation (Figure 96) were limited to a few wetted widths upstream, suggesting that, with caution and awareness, reaches can be modeled across a range of discharges.

Summary

We have to-date generated hydraulic models for more than 600 CHaMP reaches, estimating depth and velocity fields for the discharge rates at the time of measurement. We believe these results are accurate and precise enough to be utilized in the development of salmonid habitat models such as NREI and HSI, and this work is currently in process. Hydraulic model results are publically available via *champmonitoring.org*, and we encourage their use in additional applications as researchers see fit.

Scaling and Expanding CHaMP Habitat Data across Spatial Scales

Interest in habitat characteristics throughout the interior Columbia basin may exist at a range of spatial scales, from the scale of localized site level restoration actions, to entire tributary watersheds, to the interior Columbia basin as a whole. Spatial scales of interest may be driven by management requirements such as assessment of local watersheds, or by biological drivers, such as the spatial domain utilized by a given salmonid species

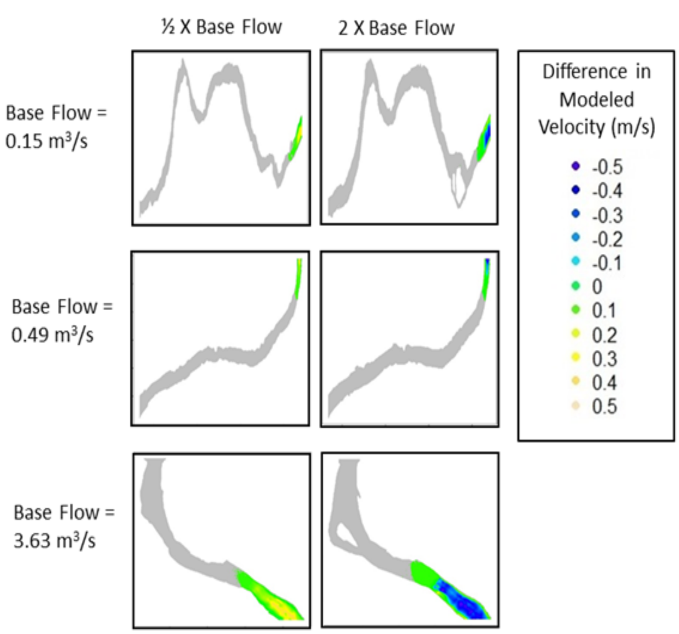


Figure 96. Maximum velocity error and extent of error propagation at low, medium, and high flow rate CHaMP reaches, resulting from assumed exit boundary condition when modeling at discharge rates with unknown downstream water surface elevation. Maximum velocity errors are estimated as the velocity field differences (a-b) between modeled velocities at where downstream water surface elevations are assumed: a) unchanged from base flow, and b) downstream water surface elevation is adjusted such that exit boundary wetted areas are scaled proportional to discharge. Gray indicates no change in modeled velocities.

at given life stage. Users may even wish to use CHaMP data to, where possible, make predictive estimates into interior Columbia watersheds sparsely sampled, or even not un-sampled. The CHaMP program exploits a variety of statistical tools to make accurate inference possible across this broad range of spatial scales (see CHaMP 2015).

CHaMP data are collected at discreet, unconnected stream segments, less than 1 km in length, randomly distributed in a spatially balanced fashion throughout each CHaMP watershed. Stratified sampling is used, with strata generally defined as combinations of valley class (source, transport, or depositional) and ownership type (public or private). All statistical analysis of CHaMP data must take into account sampling design to give proper weight to each data point. Though measurements are taken at points along a network, these point estimates are only one spatial scale of interest. In addition to point estimates at CHaMP sites, we wish to make estimates of the distribution of CHaMP metrics at spatial scales including: entire CHaMP watersheds, sub-watershed scales composed of multiple reaches, unmeasured reaches within CHaMP watersheds, and at any of these spatial scales within non-CHaMP watersheds within the upper Columbia basin. In addition, we wish to make spatially continuous estimates of CHaMP metrics and products, in the form of spatially explicit maps.

We employ a variety of statistical techniques to take these point level measurements and extend them to estimates at a variety of spatial scales (Figure 97). The spatial scale of interest drives the choice of most appropriate statistical tool. For estimation of status and trend across broad spatial scales, design based estimation is generally most appropriate. For continuous estimates, or estimates at unmeasured locations, we build empirical models that relate globally available attributes to CHaMP metrics or products, and use those models to predict CHaMP metrics or products beyond directly measured locations. At locations where we have limited direct measurement of CHaMP metrics, we can utilize imputation methods that augment estimates from design based estimation with empirically modeled data.

Design Based Estimates ("GRTS Rollups")

For estimation of status and trend across fairly broad spatial scales, over which there are about 15 or more sample sites distributed broadly over the spatial scale of interest, design based analysis (commonly referred to as "GRTS Rollups" within CHaMP; see Ward et al. 2012, CHaMP 2013, CHaMP 2015) is generally the best statistical tool. Such tools seamlessly integrate sampling design into the analysis to provide unbiased status and trend estimates. The R packages spsurvey (Stevens and Olsen 2004) is widely used within CHaMP, as well as for a wide range of ecological monitoring applications, optimal precision in status and trend estimates. More detailed summaries of status and trends for key CHaMP metrics is found in Chapter 2.

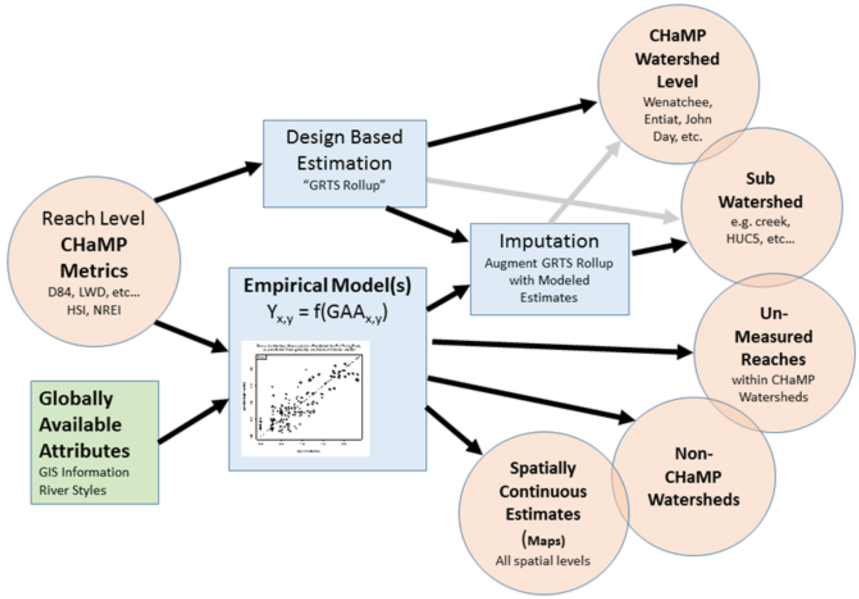


Figure 97. Statistical tools for scaling habitat data from local to population scales. Circles show various spatial scales at which inference may be made. Blue boxes represent statistical tools used to translate from reach level CHaMP data to various spatial scales. The green box indicates globally available attributes - attributes available at all locations along the stream network, not just CHaMP sites.

Empirical Model-Based Estimates

Often spatial scales of interest considerably more fine than the watershed level are of interest for research or management. When these spatial scales consist of less than 15 or so directly measured CHaMP sites, standard design based estimators may not provide sufficient precision. In this case, we make use of empirically derived relationships that relate globally available attributes to measured CHaMP metrics. For example, throughout the upper Columbia basin, we have continuously available information on valley class, human disturbance, natural landform classifications, elevation, drainage area, slope, etc. For many CHaMP metrics, we have built empirical models that relate measured CHaMP metrics (e.g., Fast Turbulent Frequency) to these globally available attributes (see CHaMP 2015), and then can use these empirical models to estimate CHaMP metrics at unmeasured reaches.

For simple linear models, we use model assisted regression to properly account for sampling design in the construction of empirical models. For more complex modeling techniques, we have developed a methodology called inverse probability bootstrapping (IPB) to properly account for sampling design while using model based statistical techniques. An ideal empirical model is unbiased across spatial scales of interest, and relationships observed within the data set over which the model is fit must be consistent at any spatial scale where the model is to be applied. Careful analysis of residuals must be performed, and in some cases models must be optimized to the spatial scale over which they're to be used for prediction.

Imputation: Subwatershed-scale Estimates

At spatial scales where some direct CHaMP level measurements do exist, we utilize an imputation methodology in order to augment the limited information from direct CHaMP measurements with predicted CHaMP metrics based on empirical models. The imputation process recognizes that the directly measured data is generally better data than the modeled data, in that the modeled data has an additional element of uncertainty, quantifiable as the standard error in modeled prediction (assessed via cross validation).

In our imputation process, we use spsurvey to generate a GRTS based estimate (Kincaid and Olsen 2013) of the distribution (mean and standard deviation) for the metric of interest at the spatial scale of interest. This estimate is then used as an informed prior in a Bayesian model, for which the modeled data are taken as additional data, and the model includes a random term for measurement noise added to the modeled data. Where an empirical model provides reasonable information content on the unmeasured sites, we can considerably increase precision in imputed estimates versus GRTS based estimates alone (CHaMP 2015).

Extrapolation: Extending Estimates of CHaMP Metrics to Unmeasured Reaches or Watersheds

The empirical models described above can also be used to make direct estimates of CHaMP metrics at unmeasured reaches within watersheds, or into watersheds for which no CHaMP data exists. Caution must of course be exercised, especially when extrapolating models into un-sampled watersheds, as we must assume that the empirical relationships observed are constant within and external to our CHaMP watersheds. In general this assumption may not be true. The more our empirical relationships describe spatially constant underlying physical laws, the less risk there is in this assumption. However, cross validation and residual analysis (see CHaMP 2015) has suggested many of our empirical models do an excellent job of describing populations at the watershed spatial level; thus extrapolating watershed level distribution estimates into un-sampled watersheds may indeed be useful and appropriate.

Continuous Estimates (Maps)

We are also able to generate spatially continuous estimates from the empirical models. In this case, the modeled estimates are used to fill in gaps between measured sites. From these spatially continuous estimates, we are able to create maps explicitly showing the estimated spatial distribution of CHaMP metrics (Figure 98). 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.

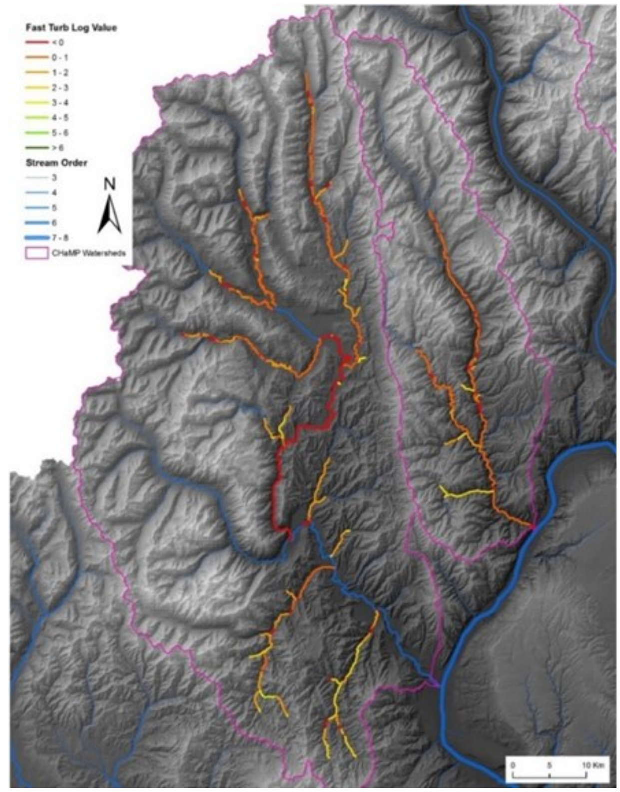


Figure 98. Spatially continuous estimate of log(Fast Turbulent Spacing) in the Entiat and Wenatchee watersheds.

Objectives for 2015

In 2015 we will improve our empirical models relating CHaMP metrics, primarily by including additional globally available covariates, including River Styles, as explanatory variables. In addition, we will consider more sophisticated modeling techniques, including kriging techniques that take advantage of spatial autocorrelation, where it exists. Higher level, multivariate CHaMP products, such as HSI and NREI capacity models, will be included in these analyses, and we will produce spatially continuous maps showing estimated capacity across portions of the interior Columbia River basin. Preliminary results for capacity, based on HSI, are currently available for select CHaMP watersheds (Figure 99).

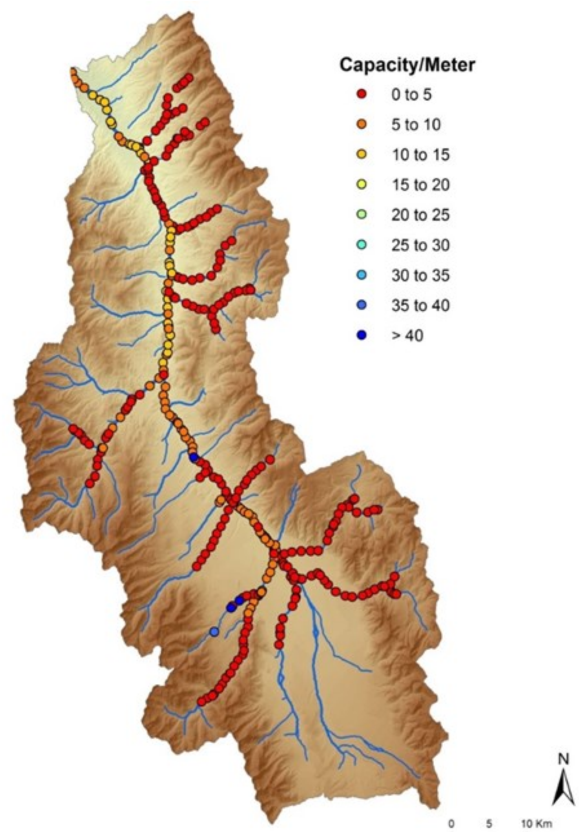


Figure 99. Spatially continuous estimate of juvenile steelhead carrying capacity in the Lemhi River subbasin, as estimated by HSI modeling.

Bringing Standardization to Habitat Sampling Methods, Metrics and Tools across the CRB

We have supported the standardization of regional and project-specific habitat monitoring programs by developing a standardized habitat monitoring protocol, study design, and workflow that are trained, reviewed and updated annually by the CHaMP development team, which includes data analysts, collectors, program managers, and collaborator staff. We leverage the development team to manage and review existing CHaMP metrics and updates, and new metrics that were identified during the pilot implementation period. Our team developed and continues to update program metadata, including standardized data dictionaries used to inform the champmonitoring.org data repository, data collection applications (e.g., iPad applications) and the River Bathymetry Toolkit (RBT; see Ward et al. 2012, CHaMP 2013, CHaMP 2015).

Protocol Standardization, Improvements and Refinements

We reinforce existing and new methods in the CHaMP protocol annually through an intensive 10-day field training. All veteran and new participants are requested to attend for the full training period to reinforce protocol methods and tool applications, to prevent protocol drift, and to ensure all crews conducting CHaMP surveys in that year are well versed in any protocol or tool changes from the previous year. In 2014 we introduced three notable changes to the CHaMP protocol related to Large Woody Debris (LWD), macroinvertebrate drift sampling, and side channels.

We replaced our method of placing LWD pieces in size classes with measuring and estimating discrete lengths and diameters for each individual piece. Prior to 2014, we enumerated the count of LWD pieces and placed them into size classes based on length and diameter. We made the decision to measure a subset of LWD pieces in order to provide more precise measurements of LWD, and to assist field technicians with calibrating their eyes for subsequent LWD length and diameter estimates. Discrete measurements and estimates also provide more precise calculations for metrics relating to LWD volumes.

We reintroduced macroinvertebrate drift sampling at all CHaMP sites, after a 2013 ISEMP-CHaMP study reaffirmed the benefit of, and need for, drift sampling at sites to support fish-habitat relationship modeling efforts (Weber et al. 2014; CHaMP 2015). The changes that we made to the drift protocol added emphasis on checking nets regularly for clogging, and clarified how to select proper net placement locations and correctly install nets.

Over the last 4 years, CHaMP has made strides in characterizing and quantifying side channels, particularly small side channels that contain <16% of flow at a site. In 2014 we added a Small Side Channel unit to our channel unit hierarchical classification system to provide a distinct, spatially accurate way to designate small side channels. This in turn allows relevant metrics, such as percent side channel area and small side channel length, width, area, and volume, to be automatically calculated through the CHaMP River Bathymetry Toolkit (RBT).

Metadata Documentation

In 2014, we made notable progress in metadata documentation of CHaMP metric definitions. By tracking critical crosswalks among the RBT and CHaMPmonitoring.org metric calculations, we greatly improved the consistency of tools that were developed concurrently by different ISEMP-CHaMP team groups. Our collaborative effort with PNAMP and the BPA AEM program to maintain consistent terminology and data dictionaries is a reflection of the overall collaborative nature of the ISEMP and CHaMP programs, which in turn provide a reliable resource for tool development platforms. CHaMP metadata, such as metric definitions, quality assured metric lists, and calculations are also available and utilized by CHaMP collaborators and data consumers, thereby making it easier for regional entities and analysts to understand CHaMP data.

In 2015, we plan to continue and improve the documentation of our study designs, metrics, analytical tools, and summary products so they are consumable by policy makers, managers, data analysts, and data collectors. This means providing the appropriate level of metadata in the appropriate form for each type of user.

Sampling Design

Since 2012, we have made minimal changes to CHaMP sampling designs. Our ability to make design changes has been enhanced via the Sample Designer tool. Each year we review the individual watershed designs of collaborators to determine if changes are warranted to meet watershed-specific objectives or logistical constraints. As one of our overarching objectives is trend detection over a 9 year monitoring period, any sampling design changes are carefully considered such that long-term CHaMP objectives are not compromised.

In 2014, CHaMP sampling design changes were limited to slight updates to the Wenatchee and Tucannon designs to meet local project objectives. For 2015, we recommend continuing centralized management of habitat sampling designs through CHaMP to balance individual watershed designs with our long-term habitat monitoring objectives and design, and to promote collaboration with other monitoring programs (e.g., AEM) to evaluate potential ramifications of sampling design changes.

Topographic Toolbar

In 2014, we made updates to the CHaMP Topographic Toolbar to enable the generation of islands, side channel centerlines, and side channel cross sections for both wetted and bankfull extents. Since 2011, the CHaMP protocol has called for sampling large and small side channels, but until 2014, we did not explicitly account for length and width measurements of these areas in the topographic survey features and metrics. In 2014 we also improved crew workflow by enabling them to process total station files within the Toolbar, which allowed more detailed tracking of topographic survey quality and errors, and removed the need for crews to process surveys in a second piece of software (ForeSight).

To provide a smoother user experience for crews processing their data, we wrapped the CHaMP Transformation Tool into the Toolbar; this tool was previously in a secondary toolbox in GIS, separate from the Toolbar. We also made additional metadata and code improvements. For example, we added the ability to track processing steps and errors in log tables to provide important information about crew processing workflow, survey errors, and software bugs to the CHaMP quality assurance team and programmers. Lastly, our programming development team improved communication by utilizing a standard code repository to efficiently track versioning and facilitate organized, collaborative development.

River Bathymetry Toolkit (RBT)

We leveraged many of our Topographic Toolbar updates in our 2014 updates to CHaMP RBT processing and metric generation procedures. Over the CHaMP pilot period, we developed new side channel metrics, including side channel width, side channel width to depth ratio, braidedness, and side channel length. In 2014 we tested our ability to calculate and automate production of these metrics, which will likely be released to the public in 2015. These important metrics will be a standard part of the 2015 CHaMP effort.

CHaMPMonitoring.org (CM.org)

In 2014, we made updates to various aspects of CM.org and its dataflow management tools. These included updating the data broker, and enhancing tablet data logger applications in order to standardize protocol management between the CHaMP and AEM programs, improving crew user interfaces, and increasing the rate and ease with which crews were able to conduct post-processing, data QA, and uploading data for storage and higher-level review.

Metric Integration between Programs

At the request of BPA, CHaMP personnel devoted resources in fall 2014 to develop and populated a mock database to house data from three metrics shared by the CHaMP and PIBO programs and translate data collected from one program to data from the other, in order to explore data sharing and broaden analytical product options to support management decisions.

While conceptually, data sharing sounds like a reasonable objective to achieve, several reasons exists that would prevent the ability of each program to consume data from the other program. First, because each program was designed to address their management questions, the survey designs were necessarily and substantially different. This can prevent combining data in a meaningful manner given the complexity and arrangement of fish habitat across stream networks. Second, while several metrics appear to be compatible across the two programs, several metric are not. These metrics that cannot be cross-walked may be key to address programmatic management questions. Therefore, metrics alone do not answer these questions, rather the synthesis of multiple forms of data is necessary. Consequently, both CHaMP and PIBO practitioners believe that a limited initial demonstration of the utility of data sharing should be conducted before committing additional limited resources to a project that is conceptual.

BioAnalysts staff developed threshold values for species and life stages of interest for three common program metrics. This effort was limited to three univariate metrics that have known mathematical relationships between CHaMP and PIBO: stream temperature, pool frequency, and large wood frequency. In addition, we geographically limited the effort to three species and 5 ESUs; Snake River Spring-Summer Chinook, Upper Columbia Spring-Summer Chinook, Mid-Columbia Steelhead, Snake River Steelhead, and Upper Columbia Steelhead. Sitka Technology Group developed a broadsheet for BPA as a deliverable from this mock exercise, in order to identify limitation, assumptions, and potential next steps if BPA determines that additional investment a common database framework and metric integration is a priority.

CHAPTER 5: FISH—HABITAT RELATIONSHIPS

One of our primary efforts in ISEMP is to utilize various sampling and modeling techniques to relate habitat condition to fish survival and habitat capacity. These relationships are developed using data from the Lemhi, Secesh, John Day, and Entiat Rivers. Finding fish-habitat relationships that can quantitatively predict capacity and survival and are robust across the Columbia River Basin landscape will enable improved identification of limiting factors, support restoration planning by targeting specific habitat improvements, and allow estimates of anticipated changes in freshwater productivity accompanying the implementation of habitat restoration plans. Additionally, reliable fish-habitat relationships are central to our ability to predict fish responses to habitat restoration actions in watersheds that do not have intensive fish monitoring programs, and are a fundamental input into salmonid life cycle models.

Juvenile survival and productivity are a function of habitat carrying capacity and improving juvenile salmonid carrying capacity through tributary restoration actions is a key component of the 2008 BiOp for endangered Chinook and steelhead populations in the Pacific Northwest (NWFSC 2008). Estimating the current carrying capacity for rearing parr and identifying the important habitat components that influence that capacity is a necessary challenge to effectively direct restoration actions as well as provide inputs for a variety of life cycle models.

ISEMP and CHaMP personnel are continuing to develop and implement several approaches for estimating juvenile carrying capacity, including methods based on a subset of the habitat data collected by CHaMP that develop capacity predictions from first principles, and a statistical approach that uses selected habitat metrics with fish metrics.

Net Rate of Energy Intake

Linking the capacity for rivers to support fish (i.e., carrying capacity or growth potential) to habitat attributes within river segments is an integral component of understanding the current status of a watershed to support fish, and evaluating potential options to improve habitat to increase fish populations. The Net Rate of Energy Intake (NREI) model uses CHaMP data (drift biomass, drift sample size class information, temperature, channel unit D_{50}), ISEMP data (fish population information), and Delft3D hydraulic model output (built from CHaMP discharge measurements and topographic surveys) to estimate energy intake rates and carrying capacities for spring Chinook and steelhead juveniles at CHaMP sites. The model estimates NREI as the balance of potential energy a fish could consume in a specific stream location and the metabolic costs of maintaining that position (i.e., swimming). We use a foraging model to predict potential food energy intake (Hughes and Dill 1990, Hughes *et al.* 2003, Hayes *et al.* 2007), which is limited by a temperature-dependent estimate of maximal consumption (Hanson *et al.* 1997), and a swim costs model to estimate the energetic costs of swimming (Hanson *et al.* 1997, Hayes *et al.* 2007). The NREI is the difference of energy intake and swim costs. More details on NREI development are available in previous ISEMP reports (ISEMP 2011, 2012, 2013).

ISEMP personnel are using the NREI model to inform habitat condition maps, estimate carrying capacities, and evaluate alternative habitat scenarios (e.g., changes in temperature or topography). The NREI model contributes to habitat condition maps by estimating the potential energetic profitability, an important measure of habitat quality, at many possible foraging locations at individual CHaMP sites. The model estimates carrying capacity by counting the number of foraging locations predicted to support fish (i.e., locations where NREI > 0) and systematically controlling for the effects of territorial behavior (e.g., Imre *et al.* 2004). Currently, NREI estimates of carrying capacity are used to parameterize salmonid life cycle models being developed for key ISEMP basins (see Chapter 6). To evaluate alternative habitat scenarios we compare NREI-generated habitat condition maps and predicted capacities to help assess the benefits or detriments of changes to habitat. For example, we could generate habitat condition maps and capacity predictions under both current and reduced temperature regimes to help understand the potential benefits of restoration that is aimed at reducing water temperatures.

We recently completed a series of improvements to the NREI model to increase its efficiency and utility. They are:

- *Rectilinear hydraulic inputs*: We eliminated a cumbersome step in the Delft3D-NREI workflow by adapting the model to accept rectilinear hydraulic inputs rather than curvilinear hydraulic inputs.
- *More efficient foraging model*: We redesigned the foraging model to work with the new hydraulic input format and significantly decreased foraging model processing time.
- *Chinook model added*: Previous NREI model simulations were only for steelhead, and we have now used bioenergetics parameters for Chinook to create a Chinook NREI model.
- *Multiple drift-temperature scenarios from a single simulation*: Previous versions of the model required individual simulations for each unique drift-temperature combination of interest, and we have now modified the model to estimate NREI and carrying capacity for many combinations of drift and temperature values simultaneously. In addition to cutting down on the number of simulations required for each CHaMP site, this also provides a more complete picture of

fish energetics in less time and provides end users with the flexibility to evaluate different scenarios without a full NREI model re-run.

- *Output visualization*: We created improved visualizations to display NREI and carrying capacity estimates across a range of drift-temperature combinations.

The NREI model is now operational and we have used the model to simulate NREI and carrying capacity for 244 visits in four CHaMP watersheds (Asotin, Entiat, John Day, and Lemhi) and 3 sample periods (2011-2013). The NREI model currently produces a collection of outputs including raw NREI estimates, predicted fish locations, look-up tables of temperature- and drift-dependent capacity estimates, and plots displaying the spatial distribution of NREI estimates at CHaMP sites (Figure 100). These outputs can be used to compare habitat quality, fish capacity, and alternative habitat scenarios for any of these visits. We have already responded to requests from watershed managers for reach/site-level carrying capacities for spring Chinook and/or steelhead.

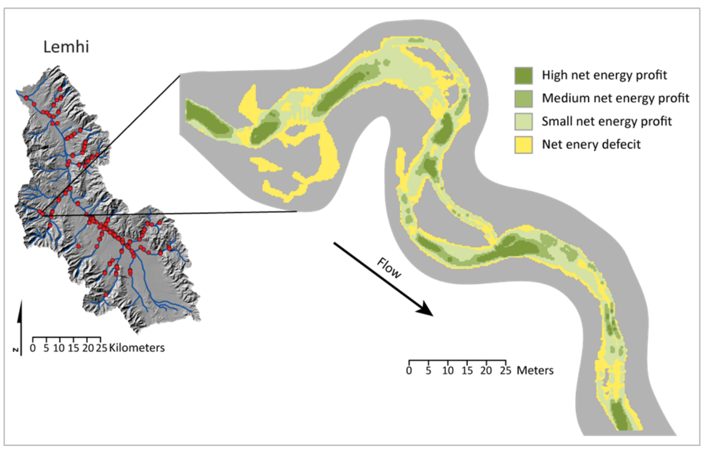


Figure 100. Example output map showing the spatial distribution of NREI estimates at a CHaMP site (CBW05583-028079) in the Lemhi basin.

The NREI model typically estimates NREI at the 20-cm or 30-cm cell scale depending on site size, although 10-cm scale modeling can be conducted at the expense of increased processing time. Current NREI outputs are generated for each whole CHaMP site that is modeled, but channel unit scale output will be possible in future versions of the model. Future planned analyses include validating the model, for example, comparing observed and predicted fish densities using quantile regression, and taking advantage of newly simulated visits. We will continue to refine and improve the model, including accounting for drift depletion by foraging fish, interspecific competition, and calibrating the model using observed growth data.

Habitat Model

The Habitat Model uses CHaMP data (channel unit D_{50}, topographic surveys), Delft 3D hydraulic model output (water depth and velocity), and suitability curves (e.g., Maret *et al.* 2006) to estimate habitat suitability and carrying capacity for juvenile spring Chinook and steelhead and adults. Suitability curves are used to predict habitat suitability for a specific species and life stage based on abiotic factors, such as depth, velocity, and/or substrate types (Figure 101). Scores for each are combined into a global HSI score that is calculated on a cell-by-cell basis and can be translated into reach-scale estimates of weighted usable area and carrying capacity.

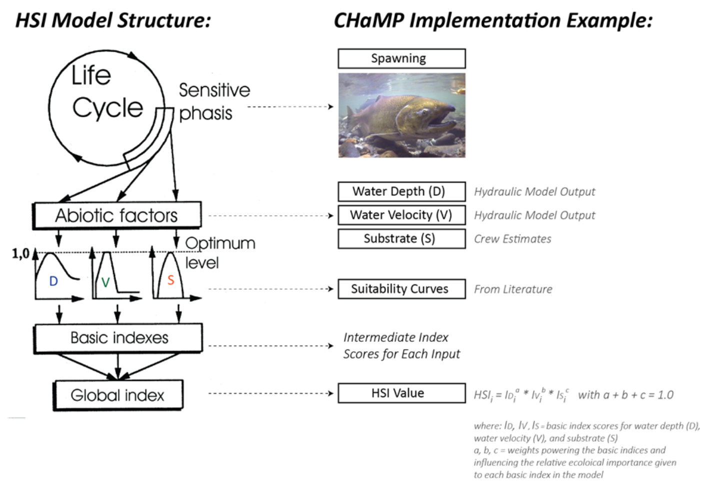


Figure 101. HSI model structure and an example of how CHaMP is implementing spawning HSI models. Figure adapted from Leclerc 2005.

The Habitat Model can be used to: (1) generate habitat condition maps, (2) simulate before/after habitat restoration and alternative climate/flow scenarios, (3) estimate reach-scale carrying capacity and (4) identify limiting factors for restoration planning. It can also aid in evaluating other spatial relationships, such as determining the proximity of rearing to spawning habitat.

The Habitat Model is operational, and has been run using data from 385 site visits in seven different CHaMP watersheds (Asotin, Entiat, John Day, Lemhi, Tucannon, Upper Grande Ronde, and Wenatchee) for 2011-2013. The model has been run for juvenile spring Chinook and steelhead in the summer period and spawning adults. We are currently using preference curves developed from previous studies by the U.S. Forest Service and Bureau of Indian Affairs in the Salmon River Basin (EA Engineering, Science and Technology Inc. 1991a, 1991b; Rubin *et al.*, 1991; R2 Resource Consultants 2004) that were used in instream flow studies (e.g., Maret et al 2006; Morris and Sutton 2007). An example of current output from the Habitat Model is shown in Figure 102. We are in the process of compiling curves to model winter juvenile habitat, considered an over-looked limiting factor by many regional biologists, and building a new set of fuzzy inference-based criteria.

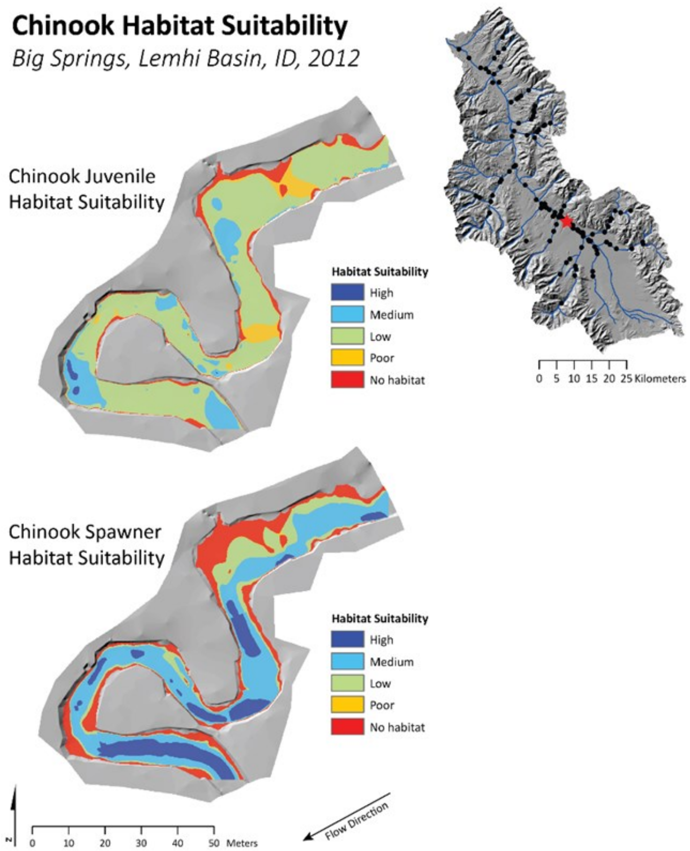


Figure 102. Example of Chinook Habitat Model output. Data are from the 2012 field season from Big Springs Creek, Lemhi River basin, Idaho (LEM00001-Big0Springs-6, Visit 551), a 200 m long CHaMP site. The Lemhi watershed map (top right corner) is provided for context. Black circles show location of all CHaMP sites in the basin. Red star shows the location of Big Springs site LEM00001-Big0Springs-6.

Habitat Model Products

Products from the Habitat Model include cell-by-cell habitat suitability scores and weighted usable area (WUA), where weighted usable area is calculated as:

$$WUA = \sum HSI_i * Area_i$$

and i represents the i^{th} cell in the HSI raster. Since WUA is a site-specific value, we calculate the normalized WUA (nWUA) as $WUA/total\ wetted\ value$. This allows us to compare reach-level suitability scores across different sites and basins. We are deriving estimates of redd and juvenile carrying capacity from CHaMP/ISEMP data, where juvenile carrying capacity is calculated as WUA/juvenile territory size and juvenile territory size is derived using fish length and a species-specific equation adopted from the literature. Spawning females are also known to exhibit territoriality, and although there is limited information on the scope of spawning territories in the primary literature, published grey literature suggests that territory size can be reasonably approximated using 4*redd area (Cramer and Ceder 2013; Keeley and Slaney 1996). As such, we calculated redd capacity as WUA/(4*redd area). Fish habitat suitability is modeled at the 10 cm cell scale. Weighted usable area, nWUA, and juvenile/redd capacity are calculated for each species for each study reach (Figure 103). We are not calculating capacity at the channel unit scale currently, but this may be implemented in future versions of the model.

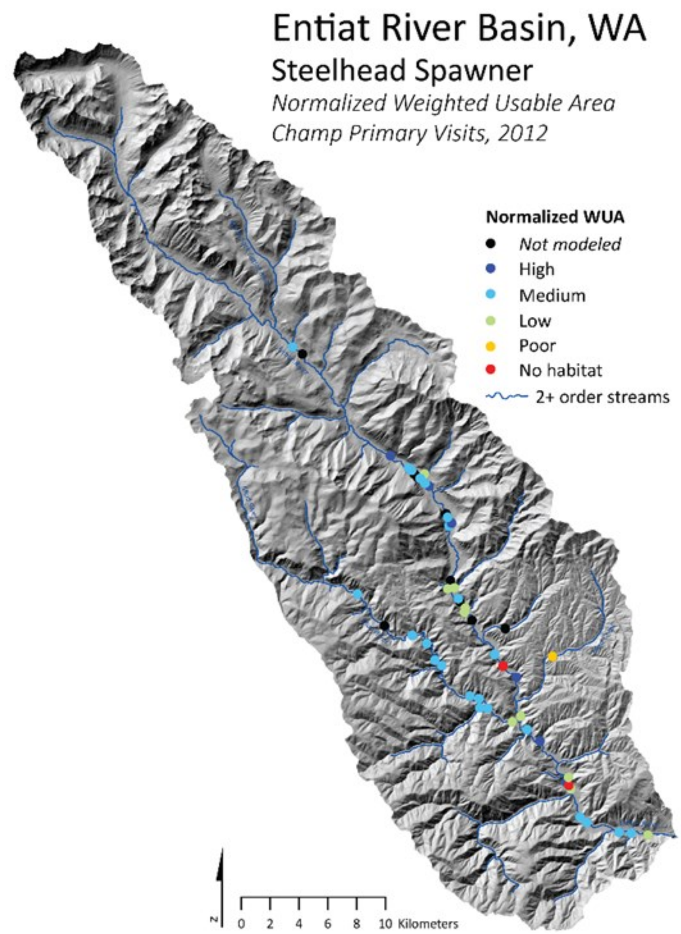


Figure 103. Chinook spawner normalized weighted usable area in the Entiat River subbasin, Washington, based on survey data from the 2012 field season. Sites with higher normalized weighted usable area have more habitat available for Chinook spawners.

Reach-level summary metrics (i.e., WUA, nWUA) are available for 385 CHaMP sites, allowing us to compare habitat quality for each species and life stage across seven CHaMP basins. Juvenile and redd capacity estimates are available for the Asotin, Entiat, John Day, and Lemhi basins, and redd capacity estimates are available for the Entiat, John Day, and Lemhi River subbasins.

Future work includes development and validation of a fuzzy habitat model, continuing to improve the automated Habitat Model, and continuing to update the HSI and fuzzy habitat model curve library, including integrating winter juvenile HSI models into our current HSI library. We will be modeling winter juvenile habitat at sites where hydraulic model output is available in the coming year.

Quantile Random Forest Approach

As well as the Habitat Model and NREI modeling approaches ISEMP personnel are investigating statistical tools to generate habitat condition maps via site-level capacity estimates, based on the assumption that higher parr densities correspond to better habitat. Observed densities at the site scale are rarely equal to a site's carrying capacity due to unmeasured or unaccounted for variables and to address this issue we are investigating a quantile regression forest (QRF) analysis. Random forest models have been shown to outperform more standard parametric models in predicting fish-habitat relationships in other contexts (Knudby *et al.* 2010). Quantile regression forests share many of the benefits of random forest models such as the ability to capture non-linear relationships between the independent and dependent variables, naturally incorporate interactions between covariates, and work with untransformed data while being robust to outliers (Breiman 2001; Prasad *et al.* 2006). They also describe the entire distribution of predicted fish densities for a given set of habitat conditions, not just the mean expected density. QRF has been used in a variety of ecological systems to estimate the effect of limiting factors (Terrell *et al.* 1996).

We used a QRF approach to empirically derive estimates of parr carrying capacity as a function of several potential limiting factors for a range of subbasins within the interior Columbia basin. We used ISEMP fish data from 2011 – 2013 (data collection and density estimation described in Chapter 3) as well as several collaborating agencies, including USFWS, ODFW, and Columbia River Inter-Tribal Fisheries Commission. Habitat data was collected under CHaMP and matched with fish abundances from the same year at the same site. We removed any duplicated site visits within each year giving a total of 712 sites available for analysis. We also demonstrate here how to link reach-level estimates of capacity to larger spatial scales.

For both spring/summer Chinook salmon and steelhead, we fit a QRF model to the site-scale fish density and 28 habitat metrics, using the *quantregForest* package in R (Meinshausen 2012, R Core Team 2014). Habitat metrics were chosen to encompass a variety of fish habitat measurements (e.g., large woody debris, pool frequency, depth, substrate, flow, etc.) while minimizing correlation between metrics. After fitting this model, we then predicted the 95% quantile of fish density at every CHaMP site, using habitat metrics that had been averaged across years for both species. We used the 95th quantile as a proxy for carrying capacity.

From a quantile regression forest, we can visually examine the marginal effect of each habitat metric on the quantile of interest through a partial dependence plot. These plots show the predicted effect on the response variable (e.g., 50th or 95th quantile) as one habitat metric changes, assuming all the other habitat metrics remain at their mean values. In Figure 104, predicted carrying capacity for spring/summer Chinook parr increases as stream complexity increases, and also as the density of large woody debris increases. For both habitat metrics, there is a maximum estimate capacity.

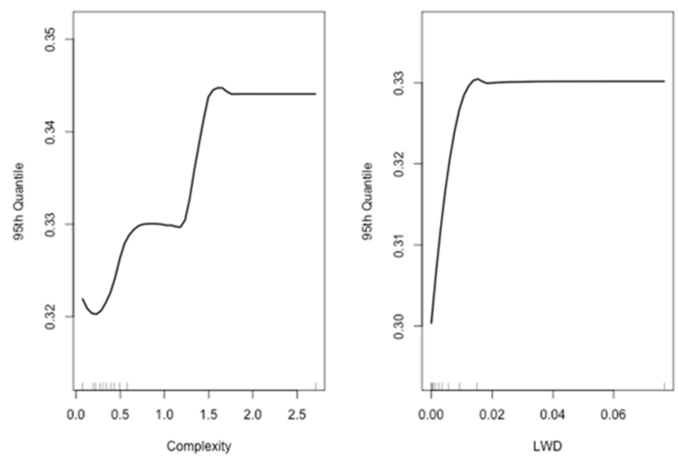


Figure 104. Partial dependence plots showing the marginal effect of complexity (defined as the CV of bankfull width:depth) and large wood (defined as the density of large wood volume within bankfull) on predicted carrying capacity of juvenile Chinook. Tick marks along the bottom indicate the deciles of the observed data.

Site-scale Carrying Capacity

Predicted capacities ranged from 0.83 - 29.33 fish/m for Chinook and 0.82 - 5.56 fish/mfor steelhead (Table 39).

Table 39. Estimated mean carrying capacity (fish per meter) values using a quantile random forest approach in each of the major CHaMP subbasins.

Subbasin	Mean Predicted Carrying Capacity (fish/m)	
	Chinook	Steelhead
Entiat	5.71	2.29
John Day	2.42	2.27
Lemhi	3.81	2.16
Methow	5.56	2.85
South Fork Salmon	7.88	1.90
Tucannon	5.07	2.81
Upper Grande Ronde	3.19	1.82
Wenatchee	4.55	1.89

Subbasin-scale Carrying Capacity

We have extrapolated capacity across the entire CHaMP domain for both steelhead and Chinook but here we present results for Chinook carrying capacity in the Wenatchee River subbasin as an example (Figure 105). Annual capacity estimates were averaged across time for any site visited for more than one year to predict an average capacity.

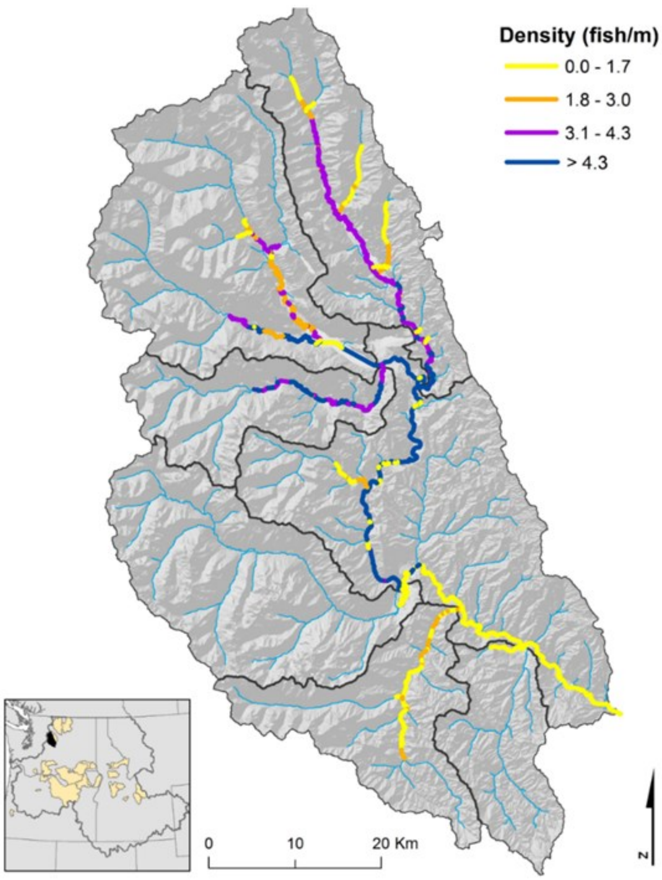


Figure 105. Estimated carrying capacity using a quantile random forest approach for every reach in the Wenatchee within the Chinook domain. Capacity classes were chosen based on natural breaks in the distribution of capacity estimates.

Comparisons with Other Methods: Chiwawa River

We compared these QRF-based estimates of spring Chinook salmon carrying capacity to spawners, parr, and smolt data from the Chiwawa River, Wenatchee River subbasin from 1991 – present brood year (BioAnalysts pers. comm.). The QRF estimates of spring Chinook parr capacity are similar to those estimated from fitting a Beverton-Holt curve to parr and spawner data to existing Chiwawa data (dark red line in Figure 106).

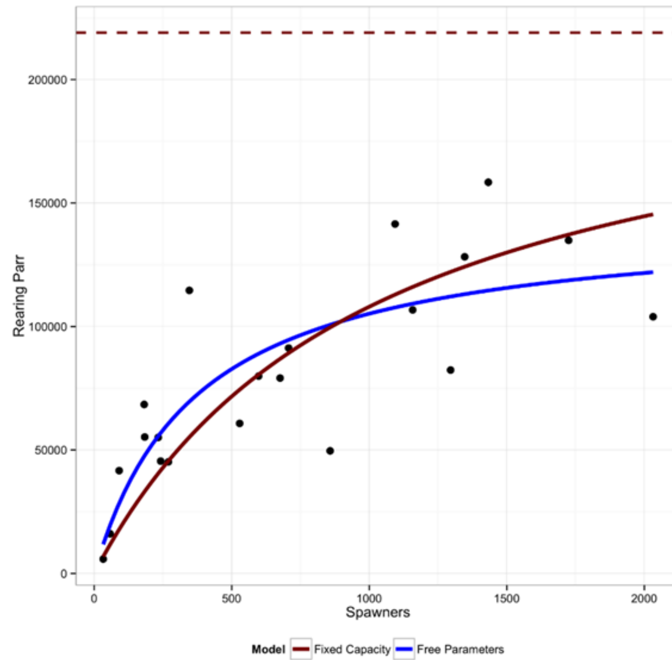


Figure 106. Estimated numbers of spring Chinook salmon spawners and rearing parr in the Chiwawa River (data provided by BioAnalysts Inc.). Blue line is the Beverton-Holt fit to those data. Dark red line is the fit when the capacity parameter is fixed at the QRF estimated value (dashed line).

Comparing QRF with NREI and the Habitat Model

We also compared QRF estimates of juvenile steelhead carrying capacity for 44 site visits in the John Day and Asotin subbasins with estimates from the Habitat Model and NREI. These empirical and mechanistic approaches were strongly correlated, although as expected the Habitat Model and NREI were generally higher than QRF estimates (Figure 107). Although the inputs are very different for these methods, the correlation between them suggests they are all converging at the same truth about carrying capacity.

We believe that quantile regression provides a robust methodology to estimate the upper threshold of possible fish density for a given set of habitat characteristics using observed fish and habitat data. The breadth of ISEMP and CHaMP data make these results and predictions robust across a wide range of habitats found within the interior Columbia basin, and the fact that QRF estimates of capacity are so strongly correlated with other methods is very encouraging. The combination of all these methods indicates a strong weight of evidence for the results. In the next year we will be working to improve the analyses by incorporating data from 2014, narrowing the list of habitat metrics used in the QRF model, improving and expanding the list of possible globally available attributes used in the extrapolation model, and testing capacity predictions against other independent datasets.

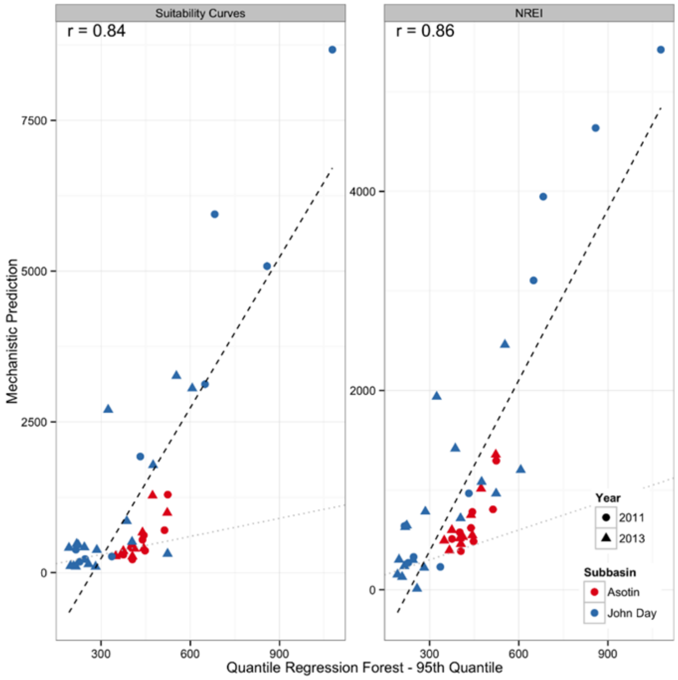


Figure 107. Estimates of total capacity from a quantile regression forest model, plotted against estimates of total capacity from habitat suitability curves and NREI. The dashed line shows the best-fit regression between the two estimates. The dotted line shows the one-to-one line.

Recommendations

Based on the encouraging results from the work completed so far with NREI, the Habitat Model and Quantile Regression Forests, we recommend that ISEMP continues to advance these tools, using more years of data from CHaMP and ISEMP and other datasets from watersheds outside of those sampled by ISEMP and CHaMP. Validating these models is of outmost importance so that we can expand their use outside of the tributary or watershed that they were developed in. We also recommend that we continue to develop the extrapolation tools that we have described here so that site-level data can be used to tell managers and policy makers about the fish populations at the scales at which they are managed.

CHAPTER 6: TOOLS TO LEVERAGE FISH DATA

Life Cycle Modeling

Life cycle models (LCMs) provide a useful framework for leveraging the fish population and habitat monitoring data collected by CHaMP and ISEMP, as well as other BPA-funded RM&E projects, to help identify an optimal portfolio of targeted restoration actions aimed at recovering listed salmon and steelhead populations. They are also a powerful tool to help answer key management questions, and are particularly well suited for basins in which restoration opportunities are numerous and resources are limited. Accordingly, LCMs are being developed for spring Chinook and steelhead in the Entiat River subbasin, spring/summer Chinook and steelhead in the Lemhi River subbasin, and for the major population group (MPG) of steelhead occupying the John Day Basin.

The ISEMP life cycle model is an extension of the life stage specific Beverton-Holt model (Moussalli and Hilborn 1986) modified to explicitly link survival to habitat attributes (Sharma et al. 2005). Development of this modeling approach largely relied on decomposition of readily obtainable remote sensing data (e.g., land-use classifications) into instream habitat features, for which capacity and quality were "scaled" relative to a reference condition. Similarly, prior model implementation relied on indirect measures of adult abundance (e.g., redd counts) and indices of juvenile abundance (e.g., snorkel surveys). This approach sufficiently demonstrated the applicability of the model as a tool to assess the effectiveness of habitat actions; however, it was clear that more detailed information would be necessary to detect the relatively small changes in freshwater survival resulting from habitat restoration within the timeframe of the BiOp evaluation. Therefore ISEMP personnel have developed a LCM that has a flexible population model framework capable of incorporating life-stage specific demographics, movement dynamics, fish-habitat relationships, and various restoration scenarios (Figure 108, QCI 2005 and Nahorniak 2013). The framework of a multistage Beverton-Holt stock recruitment model (Moussalli and Hilborn 1986) was applied to estimate the number of individuals that transition to a new life stage ($N^{i+1,t+1}$) based on the number of individuals in the previous life stage ($N^{i,t}$), the survival rate or productivity ($p^{i,t}$) to the next stage, and the capacity ($c^{i,t}$) of habitat to support individuals at a specific stage:

$$N_{k,i+1,t+1} = \frac{N_{k,i,t}}{\frac{1}{p_{k,i,t}} + \frac{1}{c_{k,i,t}} N_{k,i,t}} \quad (1)$$

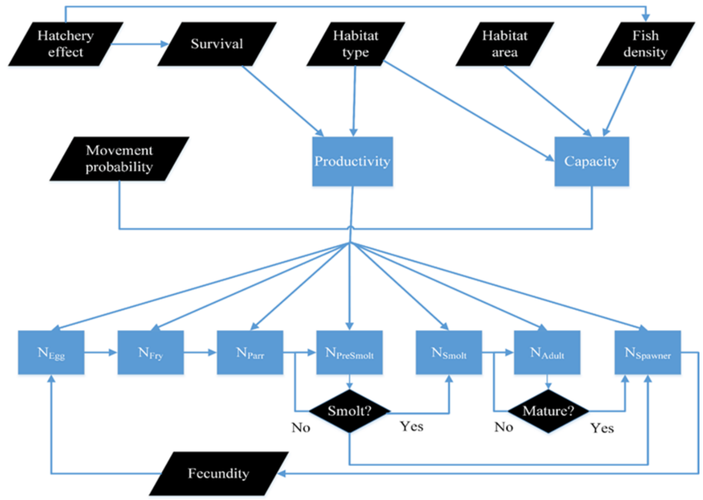


Figure 108. Conceptual diagram of the ISEMP life cycle model illustrating how habitat and hatchery effects influence life-stage specific biological responses of salmonids. Parallelograms = data, rectangles = processes derived from equations, and diamonds = probabilities that fish at a given age will smolt during the presmolt life stage or mature during the adult life stage. Flexibility to model the complexities of the steelhead life history for the life stages in the gray box are shown in the bottom box.

Given a set of candidate restoration actions, three tasks must be completed before key management questions can be informed using an LCM approach:

(1) Develop and parameterize a population model accurately capturing the dynamics and life history attributes of the target population under current baseline conditions.

(2) Identify quantitative links between habitat conditions and population parameters and build appropriately into the model structure.

(3) Accurately model the response of habitat to proposed restoration actions, both in terms of the expected magnitude of change and its temporal domain (i.e., how long will it take for benefits to be realized).

Here, we report on progress made toward these tasks in 2014-15 in the John Day. Model output from the Salmon River subbasin is reported in Chapter 1; Entiat and Wenatchee models are under way and will be reported on in future publications.

John Day Steelhead LCM

Our goal is to develop an analytical framework that integrates freshwater habitat measurements with realized stage-to-stage survival probabilities to help shape decisions pertaining to key management questions. We designed the John Day steelhead LCM to provide a quantitative means of assessing specific restoration alternatives focused on improving habitat in freshwater spawning and rearing areas. We translate habitat measurements collected at the reach scale (CHaMP 2015) into estimates of carrying capacity and population productivity using mechanistic fish-habitat models (e.g., NREI, Habitat Model, Chapter 5), to parameterize the LCM (Figure 109). The final step is the extrapolation of reach-scale estimates to the river network by population.

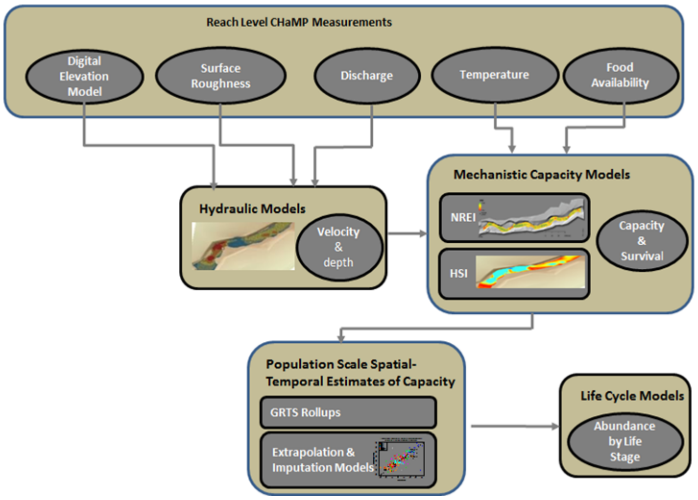


Figure 109. Relationships and workflow between CHaMP habitat data collection, capacity modeling, within-basin extrapolation, and life cycle modeling.

The John Day steelhead LCM is structured hierarchically to account for population dynamics and fish-habitat relationships at multiple spatial scales to allow for the concurrent modeling of multiple, interacting population segments (e.g., independent populations within an MPG) that are linked via dispersal[5]. By linking model parameters to summaries of reach-scale population dynamics (e.g., capacity and survival) we can evaluate the impact of addressing limiting factors, specifically for spawning and rearing habitat, for salmonid populations that vary in importance throughout a given watershed. The John Day is uniquely situated in that nearly all model inputs can be estimated empirically from data collected for ongoing projects and/or long-term monitoring efforts.

The ISEMP model is able to accurately capture variation in *O. mykiss* life history by (1) enabling pre-smolts to 'opt out' of anadromy and instead remain in natal tributaries to become resident rainbow trout, whilst continuing to contribute reproductively to the mixed anadromous/resident population, (2) allowing for the anadromy/residency 'decision' to be made on a gender-specific basis, consistent with available empirical evidence (e.g., Ohms *et al.* 2014), and (3) allowing for mature *O. mykiss* to survive after spawning and make multiple reproductive contributions over a lifetime.

[5]The current version of the John Day steelhead LCM includes only one segment of the John Day MPG, the Middle Fork.

Given that fish populations and habitats typically show spatial structuring within river networks, additional factors must be considered when reach-level estimates are used to represent a population that operates at coarser spatial scales. For the John Day steelhead LCM the parameter values used are obtained by first extrapolating reach-scale estimates of life-stage specific parameters to similar river segments within the network, where similarity is determined using continuously available river and watershed attributes (e.g., physical habitat structure, primary production, water temperature). Subsequently, predicted parameter values are integrated across the entire watershed (e.g., capacity estimates are summed) to calculate basin-scale estimates (Figure 109). This network extrapolation allows us to generate population-level estimates of parameters that are consistent across LCMs implemented throughout the Columbia River Basin. The network extrapolation is built from spatially explicit predictions of habitat factors driving fish populations, thereby providing the link between habitat conditions and populations that is necessary for evaluating management alternatives. For example, using parameter values from network extrapolations, managers can evaluate the population response expected to result from specific habitat modification alternatives (e.g., length of stream improved, location of restoration, type of restoration, etc.) within the context of current population dynamics.

In 2014 completing the John Day steelhead LCM centered primarily on two tasks, 1) parameterizing for baseline conditions, and 2) building habitat–population parameter linkages. A secondary task was preliminary modeling of a riparian restoration scenario. We chose the Middle Fork John Day population for this phase of work as the basin has several years of fish in/fish out-type data (e.g., Banks *et al.* 2014), CHaMP habitat and ISEMP fish surveys, and a complete catchment-wide River Styles assessment (O'Brien and Wheaton 2014).

Baseline Condition Parameter Estimation

To estimate carrying capacity parameters for the John Day steelhead LCM, which models population dynamics at the watershed scale, we first estimated the capacity for each CHaMP reach to support spawning adults (Habitat Model) and rearing juveniles (NREI; see Chapter 5 for details). These reach-scale estimates were then extrapolated to the network scale and summarized for input into the model. The estimate of total juvenile capacity was decomposed into parr and pre-smolt fractions for initial model runs based on the average parr/pre-smolt proportions from annual fish sampling activities (i.e., ISEMP, ODFW).

The productivity parameters are based on life stage-specific survival estimates made predominantly from PIT-tag mark-recapture observations for juvenile and adult steelhead, both in the John Day River Basin and at mainstem Columbia River sites. A combination of fish and habitat monitoring datasets have been compiled and used to estimate baseline condition model parameters (Table 40), including (1) parr/pre-smolt survival, 2007-2013 calendar years (ISEMP); (2) John Day Dam-to-

Bonneville Dam smolt-to-adult return rates, 2006-2011 out-migration years (McCann *et al.* 2014), (3) Middle Fork trap-to-John Day Dam out-migrant survival estimates, 2004-2014 (Banks *et al.* 2014; ODFW, unpublished data), (4) CHaMP habitat survey data (i.e., Habitat Model and NREI capacity estimates and egg-to-fry survival), calendar years 2011-2014, and (5) basin-level estimates of smolts per spawner (and smolt age composition), brood years 2008-2011 (Banks *et al.* 2014; ODFW, unpublished data), with the variable calendar/brood year ranges reflecting the onset of different monitoring projects (or study design changes).

Two modifications are needed to the estimates of the productivity parameters required to run the model before they can be adopted as the final inputs for use in scenario evaluation:

(1) Mark-recapture survivals (for some stages) may be modified to better reflect *maximum* productivity given that these empirical estimates implicitly include density dependence whereas the model treats them otherwise, and

(2) We still need to embed the link between survival and habitat conditions. Given that survival is strongly dependent on the size, or growth, of an individual (Coleman and Fausch 2007, Letcher *et al.* 2015), we will likely exploit a habitat condition -> growth -> survival pathway in order to establish an overall habitat–survival connection. Indeed, preliminary analyses of John Day steelhead mark-recapture data indicate that juvenile *O. mykiss* of larger size survive at a higher rate than their smaller counterparts. This, combined with an emerging method to predict basal food web production continuously across the Middle Fork of the John Day network, shows considerable promise as a platform for testing the effects of specific management actions on both the capacity and productivity, at least for the parr/pre-smolt stages.

Network Extrapolation

To be useful in LCMs, parameter values must correspond to a spatial scale that is consistent with the one at which population dynamics and management scenarios play out. For the John Day steelhead LCM we are currently modeling the steelhead population occupying the Middle Fork John Day River watershed. As a result, a key process in the parameterization of the model is the extrapolation of reach-scale estimates of capacity and productivity to the river network occupied by *O. mykiss*. The process of extrapolating reach-scale parameter values is a work in progress and a primary component needed to finalize the model. However, to demonstrate the usefulness of extrapolating reach-scale parameter values to a continuous river network occupied by a population of management concern, initial extrapolation was conducted by determining the average carrying capacity for each of the 15 identified River Styles channel

Table 40. Model parameters and origin for the John Day steelhead life cycle model. 'N/A' corresponds to stages for which capacity will be set to a large, non-limiting value (i.e., due to an assumption of no density dependence or a lack of information to specify otherwise); 'BY' = brood year; 'CY' = calendar year; 'HSI' = habitat suitability index; 'NREI' = net rate of energy intake.

	Source of Parameter Estimates		
Life Stage(s)	Capacity	Productivity (Survival)	Others
Spawners	N/A	From ongoing PIT-tag studies of Bonneville-to-tributary survival	Fecundity from published length-fecundity relationship
Egg	HSI-based estimate of spawner capacity	Computed from published % fine sediment vs. egg-to-fry survival relationship and CHaMP sediment data	
Fry	N/A	Back-calculated from smolts/spawner estimates and survivals for #2 and #4	
Parr/Pre-smolt	Juvenile Habitat Model and NREI models	From ISEMP/CHaMP PIT-tag data	Age-specific emigration probability derived from age- & BY-specific out-migrant estimates, BY egg escapement, CY survivals
Smolt	N/A	Cormack-Jolly-Seber trap-to-John Day Dam survival estimates	
Ocean Rearing Juvenile/Adult	N/A	Published John Day Dam-to-Bonneville Dam SARs	Age-specific maturation probability from return-at-age data

classifications (see Chapter 4). Subsequently, the capacity for the Middle Fork John Day basin to support steelhead redds and juvenile steelhead was calculated as the sum of the total number of individuals for a specific life stage supported by each river segment (calculated as segment length x River Styles specific density). For example, we estimate that river segments in the Middle Fork John Day River can support age-1+ juvenile steelhead (≥ 100 mm) at an average density of 2.7 fish per meter. This average density at carrying capacity represents a weighted average of fish densities for each River Style accounting for the total proportion of the river network classified as a given River Style. Figure 110 depicts the spatially explicit carrying capacity predictions from the NREI model extrapolated to the river network based on the reach River Style classification.

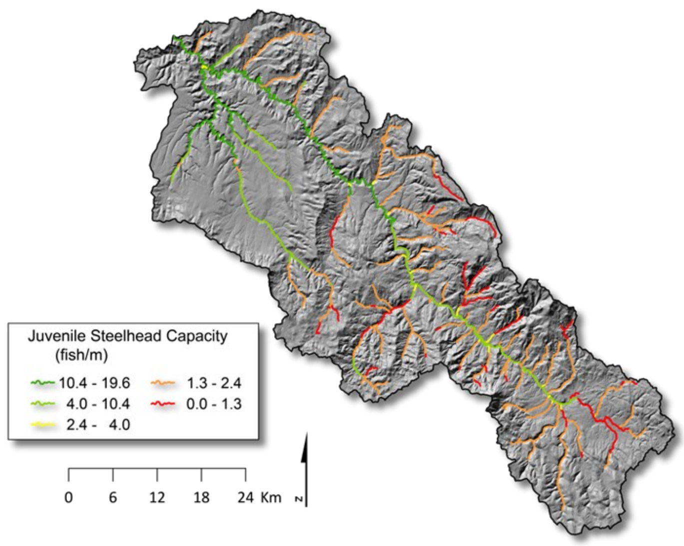


Figure 110. Results from a River Styles-based extrapolation of NREI-based estimates of reach-scale juvenile capacity to estimate the total juvenile carrying capacity for the middle Fork John Day River, OR.

Ongoing work on extrapolating reach-scale estimates of parameter values to the continuous river network is focused on using structural equation models (SEMs). Structural equation models provide a powerful tool to quantitatively describe relationships between biotic and abiotic factors affecting fish populations and capacity and productivity that serve as inputs for the John Day steelhead model, while providing a tool to account for the spatial covariance of SEM model inputs. Using SEMs, we can develop an *a priori* list of models to extrapolate reach-scale parameter values to the river network and select the extrapolation model that uses the most informative predictor variables (i.e., biotic and abiotic factors) coupled with the appropriate covariate structure for these variables by standard model selection procedures. Structural equation models are being used to predict fish capacity and productivity for segments throughout the river network using model inputs that account for the assemblage of different habitat units occurring in distinct River Styles, as well as variation in water temperature and prey availability across the river network (Figure 111).

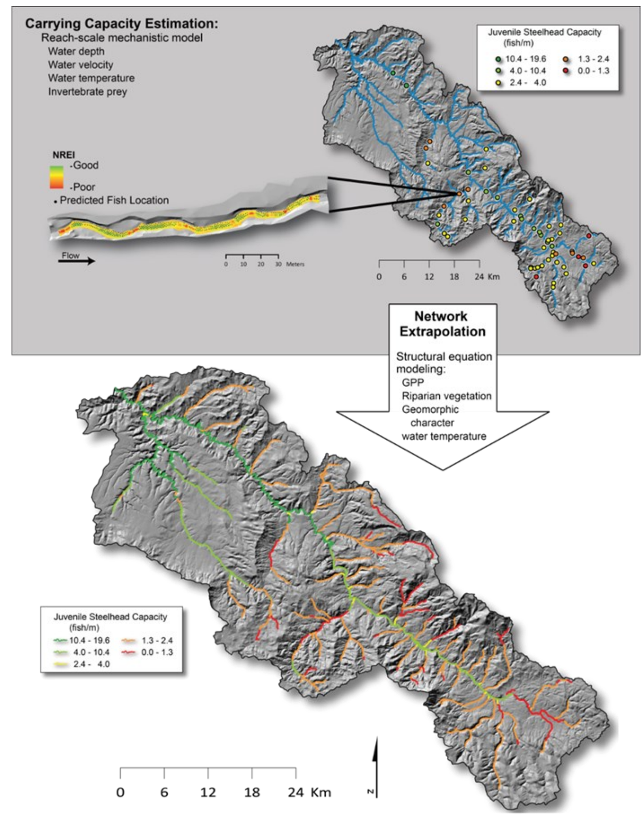


Figure 111. Conceptual illustration of the process by which spatially explicit predictions of population parameters are derived continuously for the river network. Initially, habitat conditions at all monitored river reaches are translated into fish metrics by a mechanistic fish habitat model (net rate of energy intake model, top panel). Then structural equation modeling is used to extrapolate reach-scale fish metrics using input variables available continuously for the river network, to create continuous maps of predicted fish metrics (e.g., carrying capacity, lower panel) that can be used to identify limiting factors throughout the river network.

Initial Model Testing and Preliminary Deterministic Runs

We ran the model using current condition inputs and compared key outputs to high-level population statistics to verify that the model inputs were correctly specified and behaving as expected. We compared the smolts per spawner, smolt age composition, smolt-to-adult return rate (SARs), and adult return age composition values computed from model output to those associated with the datasets/years used in model parameterization. We observed strong correspondence between model output and empirical data in all model runs.

As a second test of our LCM framework, we used the deterministic version of the model to simulate the response of the Middle Fork John Day steelhead population to the temperature improvements expected to result from complete riparian restoration and instream flow recovery. We created two model runs to approximate this scenario: (1) 'Capacity Only' where we changed the parr/pre-smolt capacity estimate used to reflect

NREI outputs with a 4 °C reduction in temperature, and (2) 'Capacity + Productivity' where we used the capacity change from (1) plus an assumed 5% (absolute) improvement in juvenile (parr/pre-smolt) productivity given that NREI output indicated that a 40% increase in growth potential may occur under restored temperature conditions.

Model runs predicted that improvements in juvenile rearing capacity alone would do little to reverse the downward trend for the simulated population, whereas combined juvenile capacity and productivity improvements may translate into a 5-fold higher abundance of spawners at the end of a 20-year time horizon (Figure 112). It is important to note, however, that these scenarios were developed for initial testing purposes only and do not account for any of the other habitat improvements that may also occur when riparian restoration is achieved (e.g., improved instream sediment conditions, wood loading, etc.).

Future Work

Considerable progress has been made towards the development of the John Day steelhead LCM. Next steps include reviewing, updating and finalizing the suite of model parameters, including point estimates and their associated uncertainty (i.e., for stochastic simulations). Concurrently, we will estimate and incorporate the life history parameters needed to properly model a resident *O. mykiss* population segment, including repeat spawning by adult steelhead and gender-specific probabilities of assuming anadromous or resident life histories. Once these steps are completed, we will conduct a full-scale analysis of model sensitivity within the plausible range of parameter values. Finally, we will expand the model's coverage to the entire John Day Basin steelhead MPG, with spatially explicit modules for each of the basin's five independent populations (i.e., Upper Mainstem, Lower Mainstem, South Fork, North Fork, and Middle Fork John Day subbasins).

Work on the John Day model is informing model development for steelhead in the Entiat and Wenatchee River subbasins through collaboration between ISEMP personnel, and ISEMP-wide the next steps for development of the LCM are to incorporate fish-habitat relationships beyond the simple extension of empirical observations, developed through the Habitat Model, NREI, and QRF, and to incorporate better estimates of capacity rather than historical values and empirical data, neither of which may be good estimates of contemporary capacity.

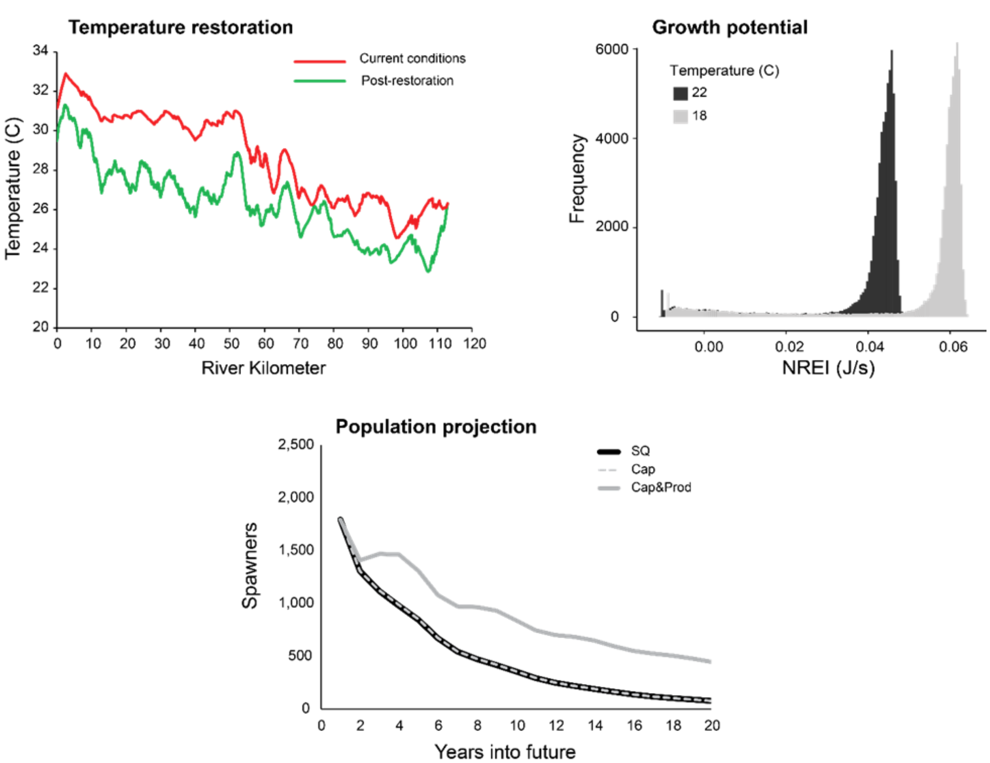


Figure 112. Results from a test restoration scenario evaluation completed using the draft Middle Fork John Day parameterization of the John Day steelhead life cycle model. We approximated the thermal benefits expected from full implementation of the Middle Fork John Day riparian re-vegetation/restoration plan (upper left panel; from Oregon DEQ's Heat Source model) by re-computing NREI-based juvenile capacity estimates (upper right panel) under an assumed 4°C reduction in average temperature. We then simulated the Middle Fork John Day population for 20 years under current (SQ = Status Quo) and improved temperature conditions (bottom panel), where 'improvement' was modeled based on (i) the new NREI capacity prediction alone (Cap) and (ii) assuming a simultaneous improvement in capacity and productivity (Cap+Prod).

Estimating Total Adult Steelhead and Chinook Escapement

Estimating the escapement for adult steelhead across their range is extremely difficult because of variable environmental conditions during the upstream spawning migration period. We are also limited in our ability to install weirs or reliably observe redds due to the high gradient streams and larger rivers in which steelhead typically spawn. A proliferation of PIAs and improved technologies over the past decade has allowed researchers to tag adult migrating salmonids and detect them in strategic locations within a river network. ISEMP personnel have developed a method using this technology that provides estimates with appropriate uncertainty of total natural origin steelhead and spring/summer Chinook escapement to various tributaries using detections of PIT tagged fish marked at a dam.

The number of mainstem dams in the Columbia River Basin provides relatively easy access to migrating adult salmonids at traps located in fish ladders. Tagging in strategic locations means a large number of populations can be estimated at one time, for example, Priest Rapids Dam is used to intercept adult salmon and steelhead during their migration up the Upper Columbia River (See 2014), and adults are tagged at Lower Granite Dam (LGR) on the Snake River. By combining the estimated tagging rate from mainstem adult PIT tagging locations with subsequent upstream tributaries detections at PIAs we can estimate the total spawning population of salmon and/or steelhead at each detection location using two independent models: the first estimates total escapement over a dam with uncertainty, while the second estimates the probabilities that a fish moves along certain paths of the stream network above the dam. By combining the two, we obtain escapement estimates that incorporate the uncertainty of the total escapement as well as the uncertainty of the movement probabilities. Here we focus on the first model estimating total escapement of natural origin fish over LGR.

Our goal is to estimate the number of natural origin steelhead and spring/summer Chinook that cross LGR each week. We use data from window counts on the fish ladder and from a trap within the fish ladder (Schrader *et al.* 2013, data from Columbia Basin Research Data Access in Real Time (DART). The window usually operates 16 hours a day (4am - 8pm), occasionally dropping to 10 hours (6am - 4pm). Observers at the window record every fish they see crossing the window during those hours, and differentiate between clipped and unclipped steelhead (but not Chinook). Counts are made every day the fish ladder is open to fish passage.

The trap samples the run by opening 4 times an hour for a length of time that is set by the daily trap rate. This rate is set with a goal of capturing a certain number of wild fish, but may change throughout the season due to water temperature and flows. Fish caught in the trap are PIT tagged and genetic information is taken to determine wild from hatchery fish. The number of fish caught in the trap each week divided by the known trap rate (seconds the trap is open/total seconds in a week) provides another estimate of the total number of fish that crossed the dam that week.

All data is summed to a weekly time-step. The nighttime passage rate is estimated based on PIT tags known to be crossing the dam each week. The window counts are then adjusted by this rate to provide one estimate of the total number of fish crossing the dam that week. The number of fish in the trap is divided by the weekly trap rate to provide another estimate of the total number of fish. These two estimates are combined in a state-space model to provide a final estimate of the number of fish crossing the dam each week. That estimate is then multiplied by the proportion of those fish that are wild, based either on morphology or genetics, to calculate the number of wild fish crossing the dam. This proportion is derived from the fish caught in the trap. Finally, the number of wild fish is reduced by the proportion of fish that have re-ascended the dam that week, based on PIT-tag data. This results in an estimate of the number of unique wild fish crossing LGR on a weekly time scale (Figure 113).

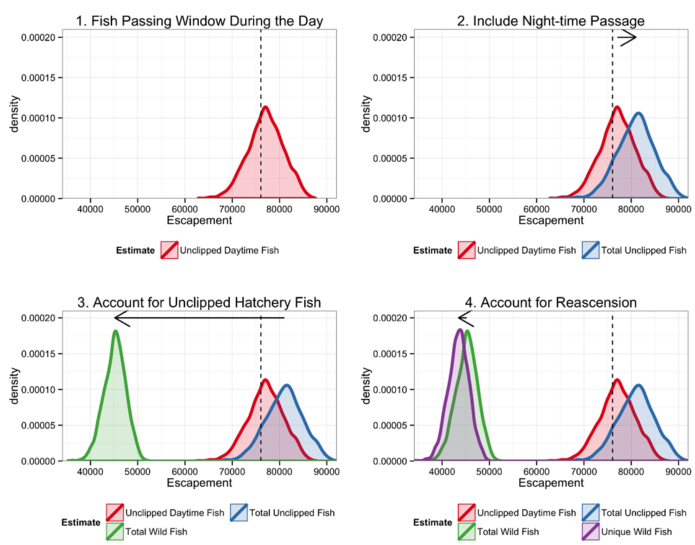


Figure 113. Example of how different components of the model are incorporated and lead to a final estimate of the escapement of unique wild fish.

Both estimates of weekly passage are assumed to be corrupted observations of the true number of fish crossing the dam that week. The log of the true number of fish crossing on week t, X_t, is modeled as a random walk (Shumway and Stoffer 2010).

$$\begin{aligned} \log(X_t) &= \log(X_{t-1}) + e_t \\ e_t &\sim \mathcal{N}(0, \sigma_X^p) \end{aligned}$$

The number of fish caught in the trap, Y_t^T, is modeled as a binomial process based on the known trap rate that week, ν_t, and the estimated number of fish crossing the dam that week,

X_t, multiplied by a correction factor, $1/\gamma$. This correction factor was included to explain any consistent bias between window counts and trap counts, which initial analyses showed was necessary. The number of fish counted at the window, Y_t^W, is modeled as a negative binomial, to account for the possibility of additional error when the counts are very large. θ_t is the proportion of fish crossing the dam during the hours when the window is open for counting.

$$\begin{aligned} Y_t^T &\sim \text{Bin}\left(v_t, X_t * \frac{1}{\gamma}\right) \\ p_t &= \frac{r}{r + (X_t * \theta_t)} \\ Y_t^W &\sim \text{NegBin}(p_t, r) \end{aligned}$$

In addition, there are two other processes that must be accounted for. The first is the proportion of fish that cross the dam while the window is closed (night passage rate), and the second is the proportion of fish that are crossing the dam multiple times (re-ascension rate). Both rates can be estimated from the PIT tagged fish of wild origin that are crossing the dam each week. In addition, we have some information about average monthly night passage rates from a previous study (1997 - 2003, 2007, Fish Passage Center). The logit of the re-ascension rate, η_t, is modeled as an AR(1) process.

$$\begin{aligned} \text{logit}(\eta_t) &= \alpha_\eta + \phi_\eta * \text{logit}(\eta_{t-1}) + f_t \\ \alpha_\eta &= \mu_\eta(1 - \phi_\eta) \\ f_t &\sim \mathcal{N}(0, \sigma_\eta) \end{aligned}$$

For the model, we are interested in the daytime passage rate, the complement of the nighttime passage rate. The daytime passage rate on week t, θ_t, is a combination of the rate estimated from monthly historical data collected from 1997 - 2003 (only 2002-2003 for spring/summer Chinook) and 2007 and the weekly observed rate based on PIT tagged fish. The historical monthly rate, θ_t^{hist}, is estimated from a binomial model, based on the number of fish that passed the dam while the window was open, z_t^{day} and the total number of fish that passed that month M_t. The monthly daytime passage rate is assumed to be consistent across those historic years.

$$z_t^{day} \sim \text{Bin}(\theta_t^{hist}, M_t)$$

The weekly observed daytime passage rate, θ_t^{PIT}, is also estimated from a binomial process based on the number of PIT tags observed to cross the dam during the daytime hours while the window is open, y_t^{day}, and the total number of PIT tags observed to cross the dam at any point that week, N_t. The logit of the observed daytime passage rate, θ_t^{PIT}, is modeled as an AR(1) process to help smooth the estimated values.

$$\begin{aligned} y_t^{day} &\sim \text{Bin}(\theta_t^{PIT}, N_t) \\ \text{logit}(\theta_t^{PIT}) &= \alpha_\theta + \phi_\theta * \text{logit}(\theta_{t-1}^{PIT}) + g_t \\ \alpha_\theta &= \mu_\theta(1 - \phi_\theta) \\ g_t &\sim \mathcal{N}(0, \sigma_\theta) \end{aligned}$$

These two estimates of daytime passage rates, θ_t^{hist} and θ_t^{PIT}, are combined through a weighted average to determine the daytime passage rate utilized in the state-space model, θ_t. The weighting each day, ρ_t, is predetermined and depends on the number of fish that crossed the dam that week compared to the average weekly number of fish observed to cross during that month from the historical data so that if a large number of PIT tags cross the dam on a particular week, ρ_t will favor the information coming from θ_t^{PIT} but if there are few PIT tags observed on a given week, ρ_t will give more weight to θ_t^{hist}.

$$\theta_t = \rho_t * \theta_t^{hist} + (1 - \rho_t) * \theta_t^{PIT}$$

The model was fit using the JAGS program (Plummer 2009), run with R software (R Development Core Team 2009). Uninformative priors were used for $X_1, \sigma^P, \sigma_\eta, \sigma_\theta$ (Uniform(0,10)), $\phi_X, \phi_\eta, \phi_\theta$ (Uniform(-2,2)), as well as $\mu_\eta, \mu_\theta, \theta_t^{hist}, \gamma$ (Beta(1,1)), and r (Gamma(shape=0.01, scale=0.01)).

Window Versus Trap Estimates

To justify the use of both window and trap-based estimates, a comparison of the weekly estimates of the total number of wild fish using window counts or trap counts divided by the trap rate showed the two align quite well (Figure 114). Trap estimates are generally slightly higher, which is to be expected as window counts do not account for fish crossing the dam at night. However, there is a consistent bias for spring/summer Chinook in 2012, with the trap generally estimating a higher weekly escapement than the window. Therefore, we incorporated a bias parameter, γ, into the model in such a way that we assume the window counts are more reflective of the true escapement.

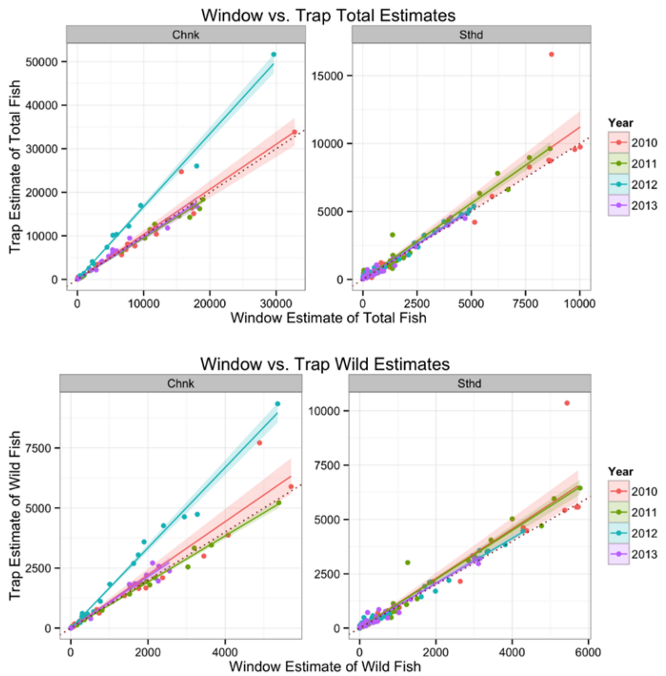
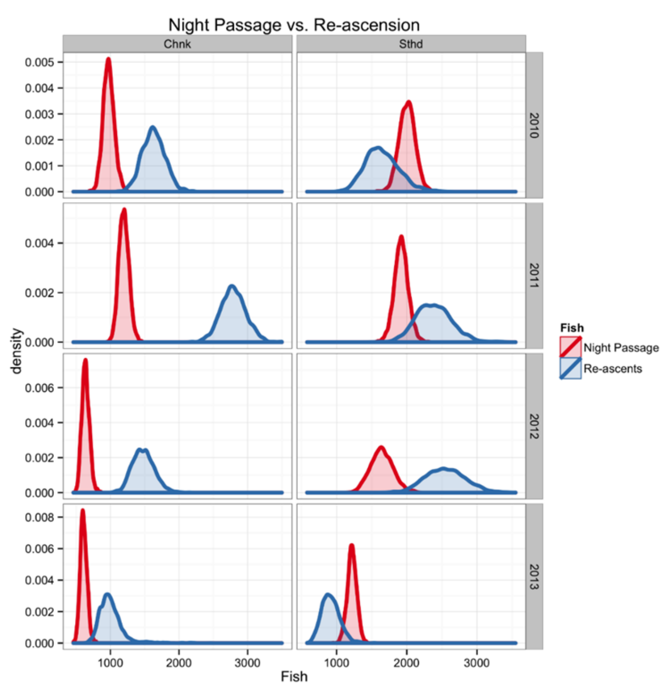


Figure 114. Window estimates (x-axis) vs trap estimates (y-axis) for total fish (top row) and wild fish (bottom row). Best linear fit with standard errors are shown for each year.

Night Passage versus Re-ascension

We examined the number of fish estimated to cross the dam at night, while the window is closed, and the number of fish estimated to have re-ascended the dam each week. Summing this over the course of the entire season, we evaluated whether the two cancel each other out. Some years they are close, but in most years they are not identical (Figure 115).



We then estimated the total wild escapement over LGR (see Chapter 3 for results). The estimate of natural origin fish depends upon genetic data, and not just morphological indicators. This means that when combined with the second stage of modeling, movement probabilities, it is possible that our estimates of escapement will be lower than where biologists are differentiating between wild and hatchery fish using morphology (e.g., weirs). The parental-based tagging program only started several years ago so the first few years in this dataset (2010 and 2011) have incomplete genetic tags, leaving many fish to be classified by morphology alone. By 2012 the vast majority of returning hatchery fish had a genetic marker in the database, and by 2013 all fish did (per comm., Michael Ackerman, IDFG). ISEMP personnel have also developed estimates of sex and age structure based on PIT tags for these TRT populations (not presented here) which can be used to assess freshwater productivity metrics such as smolts per female.

Estimating Adult Escapement into Tributaries

The goal of this project is to estimate adult escapement of natural origin spring/summer Chinook salmon and natural origin steelhead to various tributaries above a dam, for example, LGR, using detections of fish PIT tagged at LGR. To do so, we have developed two independent models: the first estimates total escapement over LGR with uncertainty (described above), while the second estimates the probabilities that a fish moves along certain paths of the stream network above LGR. By combining the two, we obtain escapement estimates that incorporate the uncertainty of the total escapement as well as the uncertainty of the movement probabilities. Here we focus on estimating movement probabilities and combining those with total escapement estimates to calculate escapement to various tributaries within the Snake River basin. Escapement estimates can be further partitioned by sex and age using tissue and scale samples collected from adults as they are PIT tagged at LGR.

Hierarchical Patch-Occupancy Model

We uploaded adult spring/summer Chinook salmon and steelhead PIT tag data by spawning year from DART to PTAGIS as a registered tag list, and then queried PTAGIS for the complete tag history (Interrogations, Recaptures, and Mortalities) for all tags within each list. We then constructed PIT tag detection histories for the minimum and maximum observation date by observation site for each PIT tag beginning at LGR and moving in the upstream direction, organized by major river basin then by tributary (Figure 116).

As a rule, each PIT tag can only follow a single branch. The minimum observation date was the primary variable used to define observations; however, if a tag was observed in multiple

Figure 115. Posterior estimates of the number of fish who crossed the dam at night and the number of ascending fish who were re-ascending the dam.

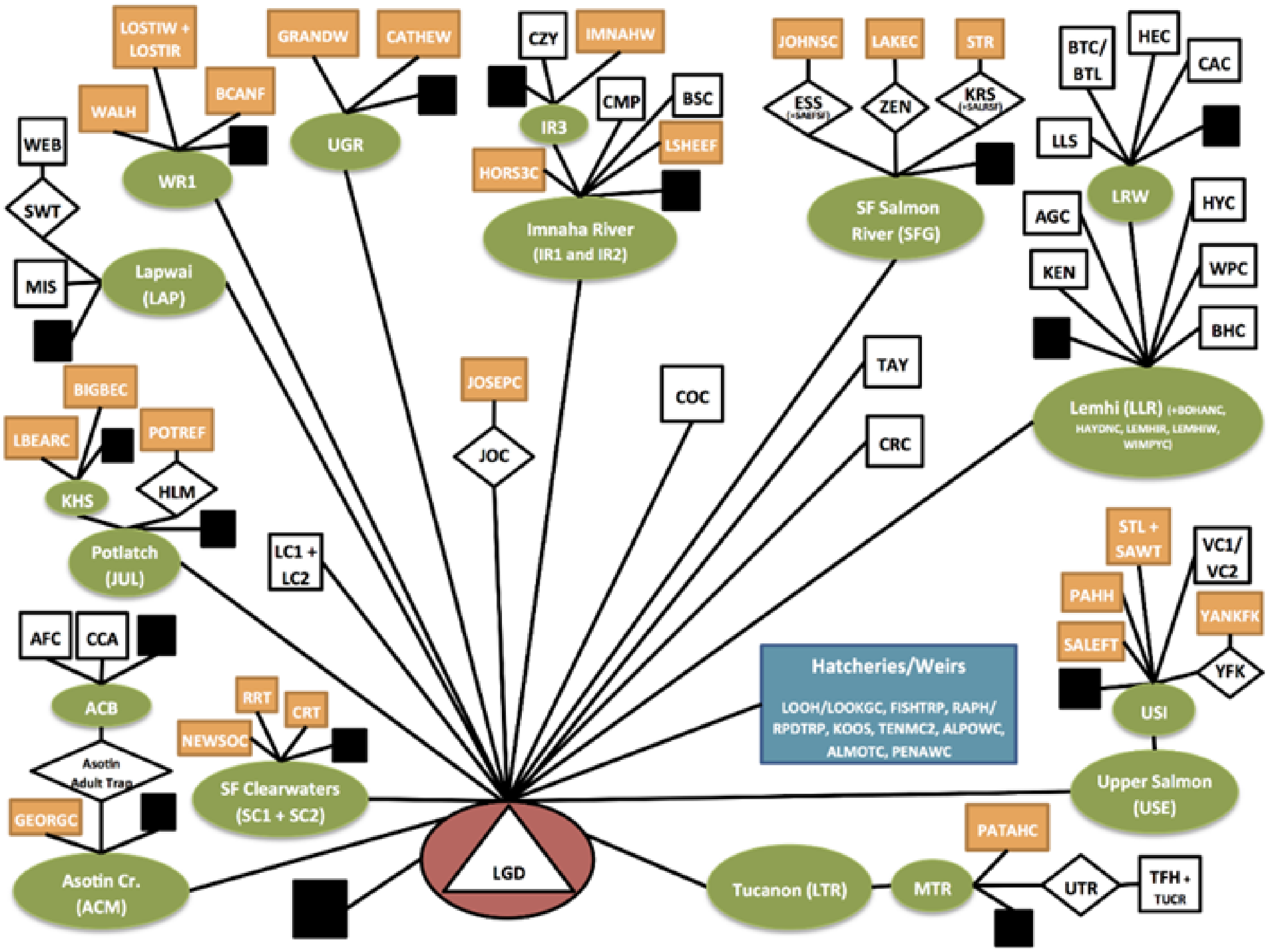


Figure 116. The hierarchical branching model diagram for steelhead crossing Lower Granite Dam. Ovals are branching points whose detection is informed by their own array and upstream detections. Rectangles are terminal bins. Diamonds are locations where fish may be detected as they move upstream. Orange rectangles have a fixed detection probability of 100%.

branches (dip-in or post-spawn behavior), the tag was assigned a branch based on the complete detection history or was assigned a branch or final (spawning) location based on a comparison of the minimum and maximum observation dates between sites. For example, if a tag was observed at the Joseph Creek array (JOC) and then later observed at the upper Grande Ronde array (UGR), the tag would be assigned to UGR and the detection at JOC would be deleted. Additionally, an individual tag may be observed within multiple tributaries within a basin such as Hayden Creek (HYC) and the upper Lemhi River (LRW). In such cases, the minimum and maximum observation dates were used to assign the tag to a spawning tributary and to delete the observations within the other tributary.

After the detection histories are validated, the detection dates by site for each tag are converted into zeros (not detected at the site) and ones (detected at the site). The data is then filtered based on the results of parental-based tagging to remove any unmarked hatchery origin fish inadvertently tagged at LGR.

Because the stream network can be observed as a hierarchy of rivers and tributaries (e.g., branched spatial arrangement) the spatial distribution of steelhead populations can be modeled using nested patch occupancy models (Royle and Dorazio 2008). These models are ideal for estimating hierarchical transition probabilities that are used to represent movement of individuals through river networks. Application of these models to PIA detections is complicated because detection efficiencies typically are less than 100%. We resolve the issue of imperfect detections by modeling the location of tagged individuals with an underlying state variable. Specifically, we distinguish between and estimate both the detection probability at a PIA and the occurrence probability of fish. The detection probability at each PIA (the probability of observing a tagged fish, given that it is present (i.e., the PIA efficiency), is:

$$p \quad = Pr(y_i = 1 \mid z_i = 1),$$

and the occurrence probability is:

$$\psi \quad = Pr(z_i = 1).$$

This separation of detection and occurrence probabilities leads to a state-space model that models the state of occurrence, z_i, as a function of ψ and the observation, y_i, as a function of both z_i and p.

$$z_i \quad \sim \text{Bern}(\psi)$$
$$y_i \quad \sim \text{Bern}(z_i p)$$

For a single confluence, with J upstream branches, the occurrence probabilities of a fish in each branch are constrained by a Dirchilet distribution so that the probabilities sum to one above that confluence. Upstream of each confluence contains a "not seen" patch ($J+1$) that represents fish that may have spawned in the mainstem between the confluence and the detection sites in any of the upstream branches, fallen back below that confluence and gone undetected, experienced pre-spawning mortality, or been harvested.

$$\psi_{j,\ldots,J,J+1} \quad \sim \text{Dir}(1,1,\ldots,1)$$

A Dirichlet specified with 1 for each tributary represents an uninformative prior on transition probabilities. This model is a series of nested patch-occupancy models where the nested structure mimics the branching nature of the stream network and the locations of the detection infrastructure. Detection locations not located at the confluence of several branches are modeled with a single ψ, with an uninformative prior of $\text{Beta}(1,1)$, representing the probability that a fish migrates past that particular point on the stream.

To simultaneously estimate occurrence and detection, this model requires either two antennas at any detection site, or upstream detections. If neither is available, the probability of detection cannot be estimated and must be fixed at 100%, which provides a conservative estimate of the number of fish passing that detection site. Several sites have more than two antennas. For sites with three antennas, detections from the middle antenna were combined with detections from the upstream antenna, and for sites with four antennas, the two downstream antennas were combined, as were the two upstream antennas. This minimizes the number of detection probabilities that the model must estimate, while still using all detections.

The model was fit using the JAGS program (Plummer 2009), run with R software (R Development Core Team 2009). Uninformative priors were used for all ψ's (Dirchlet(1,1...,1) or Beta (1,1)) and p's (Beta(1,1)).

This model makes the following assumptions:

- •Tagged and untagged steelhead have similar behavior patterns
- •Mortality is the same between tagged and untagged steelhead (no tagging mortality of any type)
- •The last place fish is detected is the location of spawning (e.g., they didn't fall back and spawn somewhere else)
- •Detection arrays were functioning during spawning times at each location
- •We can determine operation times during migration
- •Tags functioned properly until spawning, there was no chronic efficiency decreasing over time (once tagged, tag will always be seen)
- •No tag loss or decreasing tag efficiency
- •All fish returning to different populations have similar run timing, or once tagged, all marked fish mix randomly

To estimate escapement to various tributaries, samples of the posterior of total escapement past LGR are multiplied by appropriate combinations of occurrence probabilities. For example, the probability of a fish moving to Webb Creek (past the WEB array), is the product of the probability of moving along the Lapwai branch (past LAP), along the Sweetwater branch (past SWT) and into Webb Creek (past WEB) $(\phi_{LAP} * \phi_{SWT} * \phi_{WEB})$. This product of probabilities is then multiplied by the estimate of total escapement past LGR to estimate escapement to Webb Creek.

Estimates of escapement to any terminal location with a single array (orange boxes in Figure 116) will be biased low (and be overly precise) if the detection probability of that final detection site is less than 100%. Without a double array or upstream detections, there is no information in the data to estimate detection probabilities for these sites. Fixing the detection probability to 100% is equivalent to assuming that all marked fish swimming past that site are detected, therefore precluding any possibility of more marked fish slipping past the site undetected. If independent estimates of a site's detection probability can be made with appropriate uncertainty these estimates can be brought into this model as priors (which will not be updated by the data in this model). To date, no such independent estimates have been included.

Technical Recovery Team (TRT) Population Estimates

We presented results at the TRT population scale for spring/summer Chinook and steelhead in Chapter 3 (Figure 75 and Figure 76). It should be noted that as the detection infrastructure has changed since the beginning of this project, the interpretation of the results needs to be examined carefully. For example, the estimates for the Grande Ronde River upper mainstem only include Catherine Creek in 2010-2012, because that was the only detection site within that population. In 2013, the instream array UGR became operational, and allowed for an estimate of escapement covering a larger spatial area. There are several other examples like this for both spring/summer Chinook salmon and steelhead.

Comparisons with Independent Estimates

We gathered as many independent estimates of escapement from our collaborators as possible. These included estimates at weirs, estimates based on redd counts, DIDSON surveys, and possibly other methods as well. For some weirs, we found counts at the weir as well as estimates of total escapement above the weir. Some of these estimates were provided with uncertainty, some were not. We endeavored to match up PIT-tag based estimates from the same spatial scale to make comparisons (Figure 117 and Figure 118).

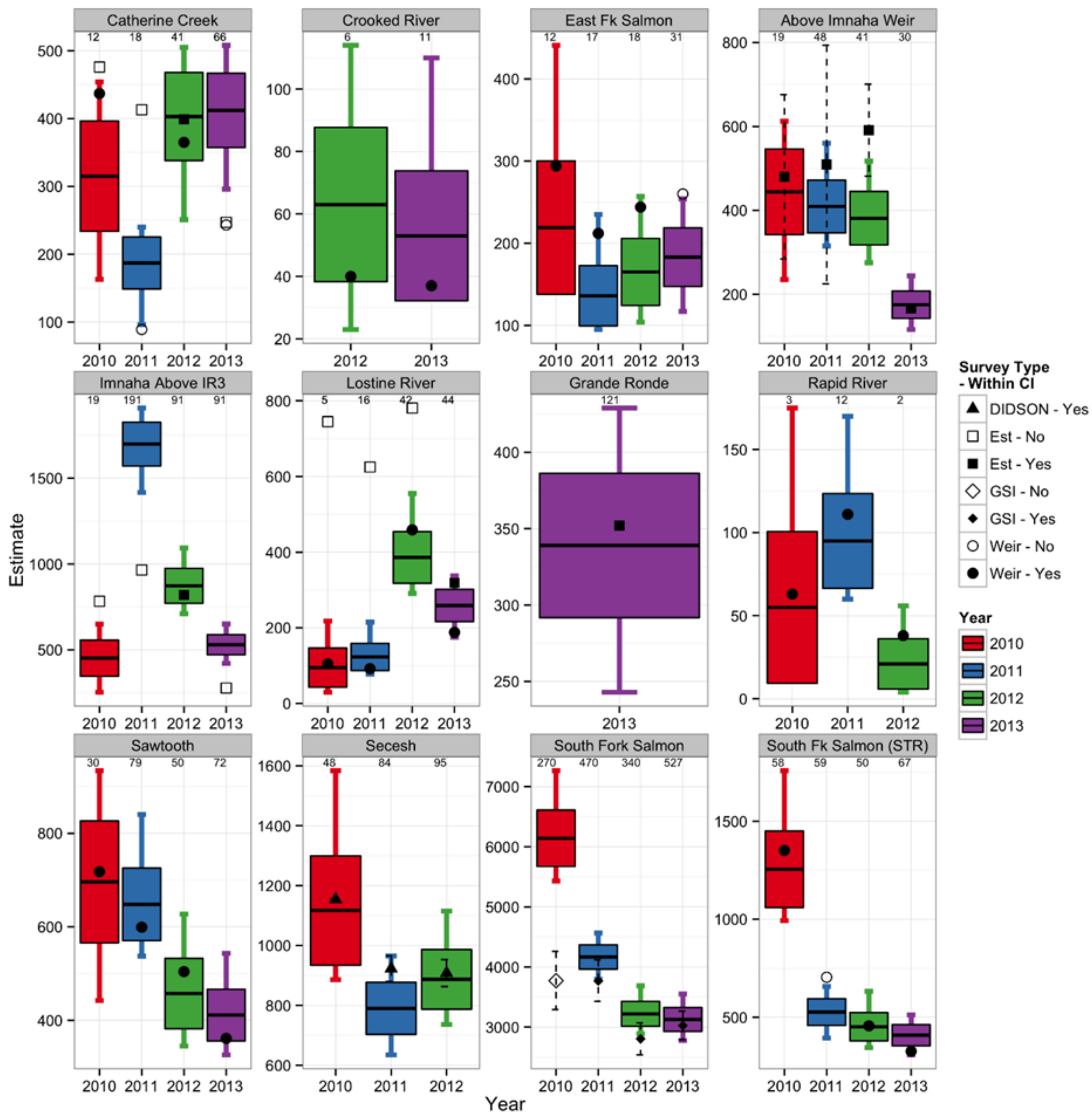


Figure 117. Comparisons between PIT-tag based estimates (boxplots) and independent estimates (black points) for natural origin spring/summer Chinook salmon. Middle lines and boxes depict the mode and 50% highest posterior density intervals. Whiskers represent the 95% highest posterior density intervals, and points are outliers beyond that interval. Colors correspond to different years. Different shapes correspond to different methods used in the independent estimates, and dashed black lines show uncertainty when it was provided. Filled shapes indicate overlapping confidence intervals between PIT-tag based and independent estimates, while open shapes indicate non-overlapping confidence intervals. The small numbers above each boxplot show how many PIT tags were detected within that spatial domain and were used in the PIT-tag based estimates.

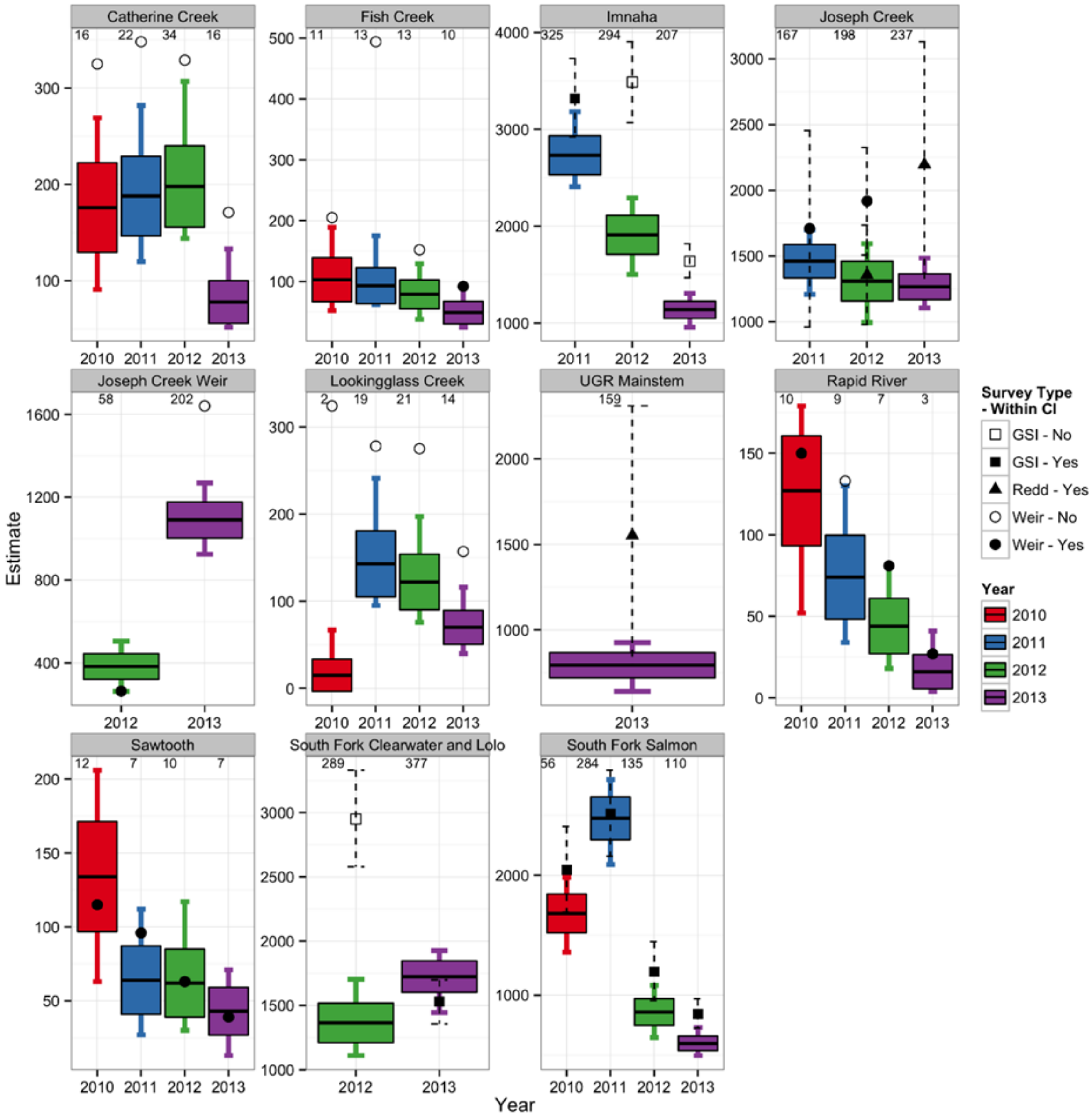


Figure 118. Comparisons between PIT-tag based estimates (boxplots) and independent estimates (black points) for natural origin steelhead. Middle lines and boxes depict the mode and 50% highest posterior density intervals. Whiskers represent the 95% highest posterior density intervals, and points are outliers beyond that interval. Colors correspond to different years. Different shapes correspond to different methods used in the independent estimates, and dashed black lines show uncertainty when it was provided. Filled shapes indicate overlapping confidence intervals between PIT-tag based and independent estimates, while open shapes indicate non-overlapping confidence intervals. The small numbers above each boxplot show how many PIT tags were detected within that spatial domain and were used in the PIT-tag based estimates.

This PIT-tag-based escapement estimate provides sex and age-structured escapement estimates across the entire Snake basin, wherever PIAs have been installed. Individual TRT populations can be parsed into finer spatial detail, depending on the density of detection infrastructure. In general, these estimates match up very well with independent estimates of escapement. We anticipate making several improvements to this model, including:

- Incorporating independent estimates of weir efficiencies. Current model runs have assumed 100% probability of detection at a weir, which may be inaccurate in some places and some years. This potentially leads to underestimates of escapement past those weirs; however, to make this improvement we require the cooperation and timely reporting of weir efficiencies by many regional partners.
- Testing the feasibility of time-varying movement probabilities. One of the assumptions currently made is that a fish that passes LGR early in the season has the same probability of moving along each river branch as a fish that passes LGR late in the season. Building time-varying movement proba-

bilities into the model would allow us to test for this possibility, as well as safeguard against non-representative sampling at the LGR trap due to extended trap closures at certain times of the season.

- Improving the estimates to the Tucannon. Currently we report estimates of escapement to the Tucannon that reflect the portion of the Tucannon population that reach LGR before swimming back downstream to the mouth of the Tucannon. We do not account for fish that enter the Tucannon without reaching LGR.
- Continuing to work with regional partners to determine the cause of any mismatches between these estimates and independent estimates. We have begun this process, including helping to provide estimates of uncertainty to certain independent estimates.

Determining the Age of Emigrating Steelhead

ISEMP personnel are developing approaches to assigning age to emigrating steelhead so that they can be assigned to a brood year. In the Lemhi we have made advances in assigning a brood year to every juvenile steelhead emigrating out of the Lemhi past the lower Lemhi RST. Based on age data from a sample of scales from a larger sample of juvenile fish, we fit a two-stage model involving a mixture model that estimates the distribution of age classes, based on length, as well as a linear model describing how length is predicted by known age class. We then applied this model to the larger sample of juvenile fish to estimate the age composition of the population. We have length data from fish sampled in the Lemhi since 2009, but we only have age data for a subset of those years, and not at all screw traps. We restricted this analysis to data from the lower screw trap because we were focused on known emigrants; fish from the lower trap were only sampled for age in 2010 (Table 41). Since only one age-4 steelhead was caught it was grouped with the age 3 steelhead.

Table 41. The number of steelhead sampled in each age group caught at the lower Lemhi rotary screw trap in 2010.

Age of Steelhead	Number of Steelhead
0	5
1	261
2	164
3	33
4	1

Fish that were scaled for age analysis were not necessarily a random sample of the population so using a straightforward multinomial model to estimate age based on length would lead to biased results (analysis by K.E. See). To address this issue we took a two stage approach:

(1) We fit a mixture model to all of the length data for each year. We assumed the length data came from a random sample of the population, and was therefore representative of the population for that year. The distribution of observed lengths is a combination of distinct distributions, one from each age class. Using the mix function from the library $mixdist$ in R (Macdonald and Juan Du 2012, R Core Team 2014), we estimated the parameters of this mixed distribution of lengths. We fit models that assumed a mixture of normal, lognormal, or gamma distributions, and chose the most likely based on a chi-squared test. This model also estimates the proportion of fish that make up each age class (π_a).

We fit a linear model that predicts the fork length based on the known age class, using data from all of the aged fish. This provided an estimate of the mean, μ_a , and standard deviation, σ_a , of fork lengths within each age class (a). To date, we have not incorporated any covariates (e.g., month) into this model.

$$\begin{array}{rcl} y_t \mid (X = a) & \sim & N(\mu_a, \sigma_a^2) \\ \mu_a & = & \alpha_a + B_a + e_a \\ B_a & \sim & N(0, \sigma_a^2) \\ e_i & \sim & N(0, \sigma_{err}^2) \end{array}$$

We can then estimate the probability a fish of a given length is in each age class using Baye's theorem:

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

where $f(y \mid X = a)$ is a normal pdf with mean μ_a and standard deviation σ_a , $P(X = a)$ is equal to π_a which comes from the mixture model, and $f(y)$ is the pdf of that mixture model as well. Once we have estimated probabilities of being in each age class for each fish, we assign an age class by drawing from a multinomial distribution with those probabilities.

Area under the curve (AUC) metrics were derived to assess predicted ages. An AUC of close to 100% suggests very few predictive errors, while an AUC of 50% suggests the predictions are no better than a random guess.

We used a parametric bootstrap on the total smolt estimates by year, and the ages assigned to each fish caught in the RST (to calculate bootstrapped proportions of age class by year). Multiplying those together provides an estimate with standard error of the total steelhead emigrants by age class each year that incorporates the uncertainty in the total emigrant estimate made at the RST and the uncertainty in assigning ages to each fish caught in that screw trap.

For 2010, a mixture of normal distributions was determined to be the best fit to the data (Figure 119). This mixture model also produced probabilities of being in each age class conditional on fork length as depicted in Figure 120.

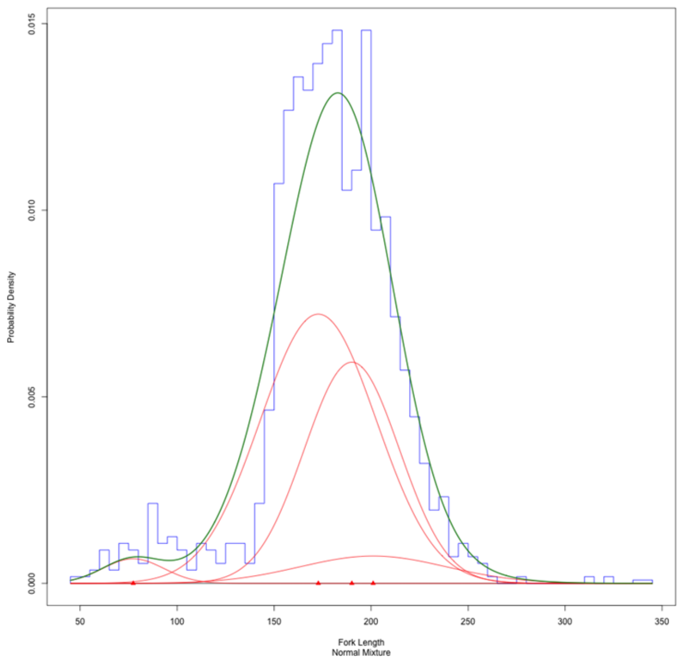


Figure 119. Histogram of fork lengths from the lower Lemhi RST in 2010, with best fitting mixture model. Red lines correspond to each age class, and the green line is the sum of those distributions. Triangles mark the mean of each age group's distribution.

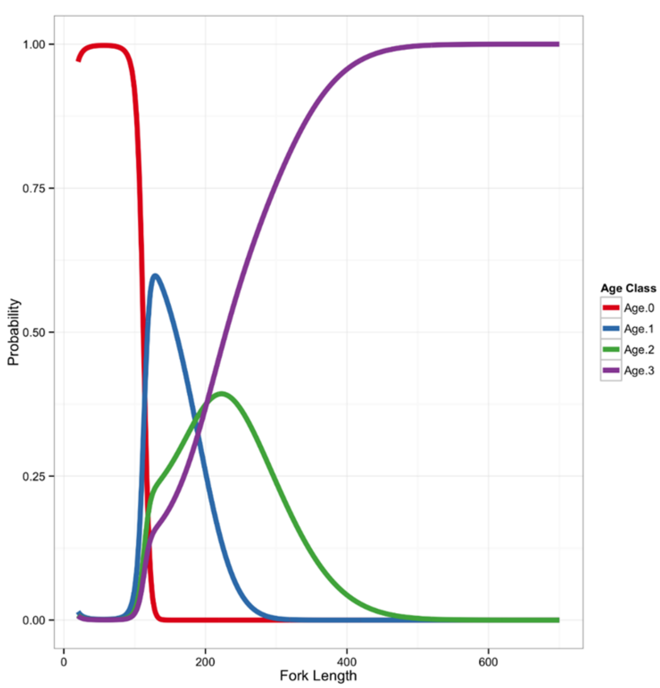


Figure 120. Probabilities of a steelhead smolt being in each age class given its fork length based on data collected at the lower Lemhi rotary screw trap in 2010.

As seen in Table 42, this model was able to differentiate between age classes fairly well. The worst results were differentiating between ages 2 and 3, which is not surprising.

Table 42. Multiclass area under the curve estimates for differentiating between each pair of age classes for steelhead smolts collected at the lower Lemhi rotary screw trap in 2010.

Age.1	Age.2	Area under the Curve Estimate
0	1	77.97
0	2	80.85
0	3	81.18
1	2	58.68
1	3	60.53
2	3	51.87

Predicting the age class of all fish without a scale sample was done by applying Baye's theorem, which allows us to examine the estimated age composition of steelhead smolts for each year (Table 43). Applying that to the total emigrant estimate from mark-recapture work at the rotary screw trap, we estimate total emigrants by age class (Table 44, Figure 121).

Table 43. Age composition (percent) of Lemhi steelhead in each year after predicting ages of all fish caught in the lower Lemhi screw trap.

Year	Percent of Steelhead in Each Age Class			
	Age.0	Age.1	Age.2	Age.3
2009	8.4	31.3	31.8	28.4
2010	2.5	54.1	36.4	7.0
2011	6.6	33.0	60.3	0.1
2014	24.8	29.4	44.5	1.2

Table 44. Estimates (standard error) of steelhead emigrants by age class and migration year from the Lemhi River 2009 – 2014.

Year	Age of Steelhead Emigrants			
	Age.0	Age.1	Age.2	Age.3
2009	1,546 (203)	5,761 (647)	5,851 (578)	5,226 (523)
2010	815 (166)	17,337 (2,726)	11,626 (1,615)	2,227 (230)
2011	1,780 (314)	8,847 (1,152)	16,275 (2,761)	31 (22)
2014	1,168 (158)	1,385 (198)	2,094 (301)	56 (19)

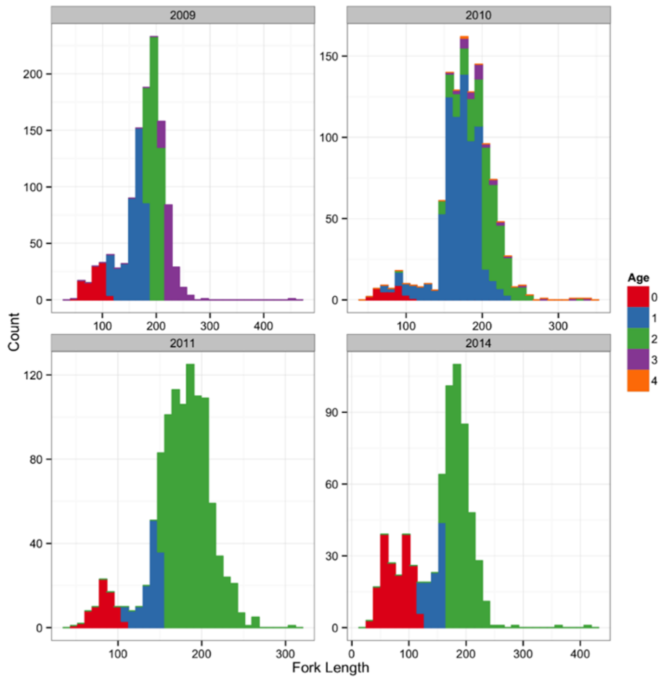


Figure 121. Observed fork lengths for steelhead smolts emigrating from the Lemhi River, with colors indicating the predicted age classes, by migration year.

We were able to describe the distribution of age classes for the one year when we had a sample of fish aged at the lower trap (2010). For other years that do not have aging data available we can apply the same length-at-age relationship to length data, although this assumes the same length-at-age relationship across years. An area for further study is whether to apply the same proportion of age classes from 2010 to other years, or to use unconditional mixture model fits to estimate those proportions. Preliminary analyses indicate that each approach produces different results, and we are pursuing this as the next step for this analysis.

It should also be noted that differentiating fish of lengths greater than 180 mm is difficult since the probabilities of such fish being age 1, 2, or 3 are very similar (Figure 120). This results in overlapping densities of predicted age classes (Figure 121).

Predictions for other years rely on mixture model fits to length data, unconditioned on any age data (since it was not available). If we fit such an unconditioned mixture model to 2011, the estimated proportions of age classes would be very different. Note that in Table 43, age 1 steelhead are estimated to account for 54% of the fish coming out of the lower Lemhi in 2010, while in other years that age group is estimated to be closer to 30%. Either 2010 was truly an anomaly, or mixture models are not fitting well for the other years, in that they are not describing the relative proportions of age classes well.

These results should not be considered final due to the uncertainty around whether to use the conditional mixture model fit from 2010 and apply those age proportions to other years, or to use the unconditional mixture model fits from those other years. Further investigation into which choice is more appropriate, or whether incorporating growth into the mixture model will resolve this issue, is needed.

Recommendations

Based on the promising results presented here on the development of the life cycle model, estimating adult escapement over a dam and into tributaries, plus tools to help fisheries managers manage their steelhead populations better by being able to assign an age class to all emigrating smolts, we recommend that this work continues. For the John Day steelhead life cycle model we recommend that we continue to refine the model so that it can be used rigorously at larger scales and address transferability to other populations such as the Umatilla and the Yakima. We recommend that work continues to refine the PIT-tag based escapement estimates, including working with collaborators on data sharing to facilitate production of estimates using the model, and that BPA consider promoting this approach in a pilot manner with managers where appropriate. Finally, we also recommend pursing refinement of the method to determine the age of steelhead form length data, including working with ISEMP partners in the Entiat and John Day to validate the approach.

CHAPTER 7: ISEMP/CHAMP 2015 WORK PLAN

Here we present an outline of the work plan for ISEMP and CHaMP in 2015. A key deadline of September 30, 2015 applies to all of the tasks outlined so as to be available to the Expert Panel process scheduled for 2016. Further work plan development will occur, based on feedback from the management community on these work products during 2015, and with an eye to future key management decision making timelines. The 2015 Work Plan has four major components:

- Survival - PIT tag-based juvenile survival estimation. **Goal:** Provide guidance on how to best estimate survival, including how fish behavior, data needs, etc. may affect alternatives. Table 45 outlines the timelines by subbasin for survival estimation.
- Fish-habitat modeling. **Goal**: To develop a flexible fish habitat modeling (FHM) environment that supports fish abundance and habitat capacity estimates that can be used in life cycle modeling. Table 46 outlines the steps in 2015 to reach this goal.
- Life cycle models. **Goal**: To develop a life cycle model (LCM) for use as a habitat action effectiveness evaluation and planning tool. Table 47 outlines the steps in 2015 to reach this goal.

Table 45. Timeline of completion for survival estimates by subbasin.

Subbasin	Goal	Steps	Comments	Deadline
Upper Columbia	Estimate survival of Chinook parr in the Entiat.	Estimate seasonal survival using Barker model of juvenile Chinook at multiple spatial scales.	Survival estimates can be used in life-cycle model.	4/1/2015
Upper Columbia	Estimate survival of juvenile steelhead in the Entiat.	Parameterize a Barker model to estimate age-structured seasonal survival of steelhead.	Survival estimates can be used in life-cycle model.	5/1/2015
Upper Columbia	Multi-state mark-recapture-resight survival and movement model for Entiat steelhead.	Estimate seasonal outmigration probabilities for each valley segment population of summer-tagged steelhead. Estimate seasonal survival probabilities for steelhead rearing in the Entiat for 1, 2, 3 years before outmigration	Estimates can be used in life-cycle model.	7/30/2015
Upper Columbia	Draft life cycle model parameterized for Entiat spring Chinook	Incorporate survival and movement estimates and NREI and HSI output into life cycle model	Needs review by local collaborators and iterative runs to refine	6/30/2015
Upper Columbia	Draft life cycle model parameterized for Entiat steelhead	Incorporate survival and movement estimates and NREI and HSI output into life cycle model	Needs review by local collaborators and iterative runs to refine	9/30/2015
Upper Columbia	Estimate survival of Wenatchee juvenile steelhead 2011-2013, possibly with age structure.	Parameterize a Barker model to estimate seasonal survival Determine whether age structuring is feasible in absence of scale samples (use Entiat data or lit values?) or just use FL as a proxy.	Survival estimates can be used in life-cycle model.	6/30/2015
Upper Columbia	Estimate seasonal survival and fine scale migration patterns of Chinook parr in the Little Wenatchee.	Estimate survival juvenile Chinook from July through March Parameterize NREI and define fish habitat relationships for extrapolation to subbasin	Survival estimates can be used in life-cycle model. Data useful to both ISEMP and WDFW/NWFSC modeling efforts	9/30/2015
John Day	Estimate survival of two age classes of juvenile steelhead in South Fork and Middle Fork of John Day	Estimate seasonal survival for two size/age classes and modify Barker model to incorporate length-at-tagging as covariate in smaller size class survival		4/1/2015
Salmon Basin	Determine if survival of Chinook parr can be estimated in the Lemhi using a Barker model	Compile data and fit Barker model.		1/1/2015
Salmon Basin	Determine if survival of Chinook parr can be estimated in the Lemhi using a Bayesian state-space model	Develop a Bayesian state-space model that accounts for parr movement between several spatial areas, imperfect detectability and probability of smolting as sub-yearlings Compare results to those from Barker model in Release 1	Potentially the biggest improvement over the Barker model is the ability to explicitly model movement between spatial areas	3/1/2015
Salmon Basin	Estimate survival of Chinook parr in Secesh subbasin.	Based on results from above work, determine best approach (Barker or Bayesian state-space) to estimate survival	Survival estimates in this reference condition habitat could provide inputs to the life-cycle model as density-independent productivity	5/1/2015
Salmon Basin	Estimate survival of steelhead in Lemhi	Estimate seasonal and age-specific survival rates of steelhead in the Lemhi, potentially using 3 seasons and 2 or more age/size classes. Choose best approach based on results	The steelhead model will most likely incorporate more spatial areas than the Chinook model	7/1/2015

Table 46.Timeline of development for a fish-habitat modeling environment.

Model	Goal	Steps	Comments	Deadline
Habitat Suitability Curves	Validation of HSC-based fish-habitat model using available fish data	Compare redd capacity estimates with GPS redd data Spawner electivity analysis Model sensitivity analysis Automate HSC	Validation results will determine which HSC curve types are most appropriate for which basins.	Complete
Fuzzy Inference System	Development and validation of FIS approach	FIS input membership functions will be based off of literature and expert opinion Determine inputs to include in the FIS model	This process should include input from CHaMP and ISEMP collaborators	4/1/2015
Fish Habitat Modeling	Compare HSI, FIS and NREI			9/30/2015
	Develop displays of FHM outputs to target audiences		e.g., Expert panels, graphic output	
Net Rate of Energy Intake	Continued validation of NREI capacity estimates and calibration of NREI model			4/1/2015
	Demonstration of potential to extrapolate NREI capacity estimates in the Middle Fork of the John Day			
	Operational to production model		Summary statistics (carrying capacity and NREI distributions) for all CHaMP sites with necessary input data.	
	Develop an automated version of the NREI model capable of being run by CHaMP analysts			
Quantile Regression Forests	Estimate summer parr rearing capacity for Chinook and steelhead for all CHaMP sites in Columbia River Basin	Compare results with those from more standard linear quantile regression	Use available landscape-level data as covariates in a linear model, then use that linear model to predict capacity everywhere those landscape-level data are available	1/30/2015
		Explore extrapolation options	To particular subbasin using GRTS-based design estimators; or use available landscape-level data as covariates in a linear model, then use that linear model to predict capacity everywhere those landscape-level data are available; or test spatial stream network models to account for spatial autocorrelation, potentially using some landscape-level covariates as well (e.g., River Styles, gradient, etc.)	6/30/2015
		Iterative improvements	Determine best fish metric to use as response (e.g. fish/m, fish/m^2, fish/sqrt (wetted area), biomass) Incorporate temperature data/predictions, and/or primary productivity predictions Run separately for different size or age structure (year 1 steelhead vs year 2) Incorporate other species as covariates	9/30/2015

Continuous Gross Primary Production Model	Move to operational status	Predictions of GPP at the network segment scale	John Day plus other CHaMP watersheds	4/1/2015
		Use network primary production model, in conjunction with water temp model and River Styles, to extrapolate NREI and survival model output to parameterize watershed specific life cycle model for the South Fork and Middle Fork of the John Day		
		Develop relationship between water temperature, GPP, and River Styles and NREI estimates of carrying capacity to predict capacity for segments lacking topographic data		
Extrapolating Metrics to the Stream Network	Provide spatially explicit site-level estimates of selected CHaMP metrics and products within CHaMP watersheds			3/1/2015
	Provide spatially explicit site-level estimates of selected CHaMP metrics and products for qualifying locations within interior Columbia basin watersheds not covered by CHaMP			3/1/2015
	Develop models linking NREI and HSI to globally available attributes	Initial extrapolation of HSI-based capacity estimates	Using currently available attributes and NREI and HSI-based capacity estimates at all/sufficient number of representative CHaMP sites	5/30/2015
		Extrapolation of HSI-based capacity estimates using River Styles	Assess error and bias, determine "best" model, repeat extrapolation	6/30/2015
		Initial extrapolation of NREI-based capacity estimates		5/30/2015
		Initial extrapolation of NREI-based [density independent] survival	Using currently available attributes and NREI-based survival estimates, estimate density independent survival at population specific parr-smolt spatial scales	6/30/2015
		River Styles-enhanced extrapolation of NREI-based capacity estimates		7/30/2015
		River Styles-enhanced extrapolation of NREI-based survival		7/30/2015
		Estimate NREI- and HSI-based capacity at LCM spatial scales		6/1/2015
	Use predicted responses to augment GRTS-based estimates at LCM spatial scales.	Use measured data combined with modeled data to produce best possible mean and spatial variance estimates for a given spatial region, with estimates of standard errors	Key to developing fish-habitat relationships and to parameterizing life cycle models.	8/30/2015
Temperature Modeling	Update 2013 to full year spatially continuous estimates of stream temperature for all CHaMP watersheds			Complete
	Generate spatially continuous stream estimates for the first half of 2014 for all CHaMP watershed			2/15/2015
	Update documentation	Streamline, document, and consolidate R code for complete modeling cycle		2/28/2015
	Model sensitivity analysis	Identify where and when sites added/dropped from basin data collection for maximally robust temperature estimation in order to give specific recommendations for field season 2015 (logger density, placement, update frequency, supplementation with non-CHaMP data)		3/1/2015
		Can models be borrowed across watersheds?		4/1/2015
		Compare and contrast temperature model outputs: MODIS, Issak models, Heat Source		4/1/2015

Table 47.Timelines and objectives for the development of the ISEMP life cycle model.

Subbasin	Goal	Steps	Comments	Deadline
Programmatic	Iterative improvements to the LCM	Identify how to include HSI, NREI, and QRF into the model Troubleshoot updated model. Update model with advancements in fish/habitat relationships from parallel work on HSI, NREI, and QRF. Test predictions against empirical observations annually. Evaluate which metric (biomass, density, age-structured abundance, etc.) is most informative and widely applicable Model/effectiveness monitoring testing/validation across ISEMP and other watersheds		9/30/2015
Salmon River- Lemhi IMW	Code fish/habitat relationships into LCM, generate restoration scenarios	Code LCM to accept HSI to estimate spawning capacity and identify suitable juvenile rearing habitat (areal extent) Code the model to accept capacity estimates derived from NREI and QRF Code LCM to allow changes to habitat capacity as a function of altering one or more metrics included in QRF (incorporating correlation across other metrics)		2/30/2015
	Test fish/habitat relationships by predicting fish biomass as a function of habitat characteristics, compare to empirical estimates	Use re-coded LCM to simulate alternative restoration strategies for the Lemhi		4/30/2015
	Complete LCM publication	Add fish/habitat relationships and simulations to current draft, submit to journal for review. Alternatively, publish existing manuscript and seek a second publication that incorporates fish/habitat relationships		5/30/2015
Upper Columbia	Develop draft watershed production model for Entiat spring Chinook	Utilize results from multi-state mark-recapture-resight model		4/28/2014
	Working version of the LCM for Entiat Spring Chinook	Present working model to RTT and AMIP groups Prepare manuscript for publication Have draft manuscript reviewed internally (ISEMP) and externally (RTT, USFWS) Submit for publication		6/30/2015
	Draft LCM for Entiat River steelhead	Work with ISEMP partners to develop approaches to tie steelhead fish -habitat relationships into the LCM. Review status of various model outputs needed for Wenatchee steelhead, e.g., hydraulic model, NREI, HSI, QRT. Review various approaches on the table now (NREI, QRT) and choose one to pursue for steelhead in the Wenatchee Work with ISEMP partners on running HSI model for Wenatchee steelhead. Write documentation for all new and updated features.		9/30/2015
	Draft LCM for Wenatchee steelhead	Compile encounter history for survival analysis Work with ISEMP partners to develop approaches to tie steelhead fish -habitat relationships into the LCM. Review status of various model outputs needed for Entiat steelhead, e.g., hydraulic model, NREI, HSI, QRT. Review various approaches on the table now (NREI, QRT) and choose one to pursue for steelhead in the Entiat Work with ISEMP partners on running HSI model for Entiat steelhead. Write documentation for all new and updated features.		9/30/2015

John Day	Complete initial parameterization of steelhead LCM for the Middle Fork John Day River	Initial survival estimates will reflect variation in survival in survival owing to fish size for Age 1 individuals and static survival for Age 2+ steelhead. Initial parameterization for production parameters will not reflect spatial variation throughout the watershed, but be held constant. Initial estimates of juvenile capacity will be based on NREI modeling efforts in the SF and MF John Day. Use River Styles classification of the Middle Fork John Day as first pass estimates of watershed capacity. Initially the average capacity estimate for each River Style will be used as a means to extrapolate site-scale NREI capacity estimates to the river network. Initial estimates of egg capacity will be based on the hurdle model developed by Falke et al. (2013)		Complete
	Update parameterization of juvenile capacity and survival	Use network primary production model, in conjunction with water temp model and River Styles, to extrapolate NREI and survival model output to parameterize watershed specific LCM for the Middle Fork of the John Day. Develop relationship between water temperature, GPP, River Styles and NREI estimates of carrying capacity to predict capacity for segments lacking topographic data. Link size-specific survival estimates to factors influencing growth across the watershed Evaluate other drivers of survival for age 2 + fish (site-level standing crop, habitat attributes [CHaMP metrics], temperature)	These potential drivers of survival will be evaluated using an Information Theoretic Approach under the Barker model framework.	4/30/2015
	Re-parameterize JD LCM to use HSI estimates of redd capacity	Test model predations against observed steelhead data for the Middle Fork John Day River Parameterize initial model for the South Fork John Day River	Provide a direct, mechanistic link to fish habitat and allow users to evaluate the effects of potential habitat manipulation on fish populations	
Lower Granite Dam Run Reconstruction	Produce peer-reviewed manuscript	Estimate number of wild adult steelhead and Chinook crossing Lower Granite Dam Estimate number of adult steelhead and Chinook escaping to various tributaries in the Snake basin Combine top 2 bullets to estimate wild escapement	Some portions already, and completed some simulation testing. Will need to be updated with latest version of model, and probably require additional simulation testing	2/1/2015
	Iterative improvements	Model must be updated each year to account for changes in PIT tag sighting infrastructure. Funding and oversight to be handed off to a steering committee composed of representatives of the various collaborating tribes and agencies.	Similar model was built for Upper Columbia steelhead	9/30/2015

CHAPTER 8: RESPONSE TO ISAB/ISRP QUESTIONS

Here we present answers to a compilation of ISAB/ISRP questions from over the years on a range of topics from coordination to study design to analytical methods for ISEMP and CHaMP.

Coordination

How has CHaMP supported the coordination and standardization of regional and project-specific federal, state, tribal and/or non-government monitoring programs and elements, including: metrics, sample designs, data collection protocols, data dictionary, metadata, and data access, i.e., web-based data and information sharing and management strategy for water, fish, and habitat data?

CHaMP has established a regional status/trend/project habitat-monitoring program in the Columbia River Basin. The CHaMP design includes stream habitat metrics, a watershed-scale status and trends sample design that can be adapted for effectiveness monitoring (e.g., AEM), data collection protocols, a data dictionary, metadata and a data management system, and a data distribution process. At the heart of the program is a web-based information system, www.champmonitoring.org. The site is used to manage the survey design, work flow of the data collection season, the QA process, metric generation and data dissemination. CHaMP is coordinated with other regional monitoring programs through data sharing and the documentation of methods on the PNAMP site, monitoringmethods.org. The CHaMP response design was built with the capacity to generate other regional programs' metrics, either through the incorporation of identical metrics (e.g., PIBO wood metrics, ODFW habitat units), or the development of metric generation scripts that use CHaMP measurements to construct these metrics (e.g., EMAP thalweg metrics). The basis for many CHaMP metrics is a detailed topographic survey of each monitoring reach so data from CHaMP can be integrated into regional topographic data sets, such as those generated with LiDAR. The topographic survey also has the potential of producing yet to be developed stream habitat metrics since the resulting Digital Elevation Model (DEM) can be resampled according to new metric protocols, for example, as we learn how to better describe "stream complexity".

Protocol and Metric Evaluations

What has CHaMP learned about the appropriateness and value of its protocols and habitat parameters through testing in select basins with fish and habitat monitoring, including those that are undergoing active restoration?

Based on a number of focal studies, CHaMP habitat metrics have been shown to be good predictors of habitat quality/quantity in terms of predicting rearing and spawning capacity (long term average maximum capacity based on a particular resource limitation, such as space or food) and juvenile growth potential. In approaching this question, we believe it is useful to distinguish primary CHaMP metrics—those that are measured/quantified directly in the field—from 'advanced' or model-derived CHaMP metrics.

Utility of primary CHaMP metrics

Numerous empirical evaluations have used CHaMP habitat metrics derived from topographical surveys (i.e., from the digital elevation models constructed from survey data) and auxiliary data collected at CHaMP sites. This effort has demonstrated that across a wide variety of modeling approaches (linear regression, Boosted Regression Trees, Regression Forest, bioenergetic and mechanistic modeling), CHaMP metrics have proven to be good predictors of fish density, survival or growth.

Where both fish and habitat (using CHaMP surveys) have been sampled in conjunction (695 sites for steelhead and 294 for Chinook), different machine learning regression approaches have been used to identify fish habitat relationships, with very encouraging results. Quantile regressions using these datasets are being used to estimate carrying capacity that will be used as inputs into the ISEMP life cycle model. Temperature, solar radiation, and conductivity collected during CHaMP survey are being used to predict gross primary production, (GPP) which in turn is a strong predictor of fish growth and abundance. CHaMP metrics used in a rapid assessment to describe the physical properties of stream channels, temperature, and GPP (see above) explained 65% of the variation in fish density across 35 km of intermittently sampled river reaches in the Middle Fork John Day. Also, Gallagher et al. (2015) links variables measured using CHaMP protocols to the abundance of juvenile steelhead and coho salmon in northern California.

Utility of model derived CHaMP metrics

In addition to studies that link primary CHaMP metrics to fish abundance/distribution, efforts to quantify relationships between fish distribution, abundance, survival, and 'advanced' CHaMP metrics are continuing to be pursued. Model derived metrics are not measured directly in the field but rather are derived from analytical or mechanistic simulation models. Of note, high resolution DEMs derived from CHaMP surveys allow development, testing, and application of powerful analyses ranging from microhabitat to multiple reach comparisons to describe fish. For example, differencing of DEMs derived between successive CHaMP surveys are a powerful means for geomorphic change detection (GCD) (Wheaton et al. 2010). Hydraulic models, parameterized from field topography, substrate, and stream-flow measurements, and validated at multiple sites, are now automated for all CHaMP site visits. Hydraulic model output and substrate are used to predict habitat suitability and carrying capacity for juvenile and adult salmon and steelhead. The U.S. Geological Survey uses CHaMP metrics to populate a primary productivity model for the Methow River subbasin in Washington. Using bioenergetics models, Weber et al. (2015) demonstrated that temperature and drift, as collected by CHaMP, were a strong predictor of juvenile steelhead consumption and growth (Weber 2009). The NREI model incorporates hydraulic models, drift, temperature, bioenergetics and foraging models to describe habitat quality and estimates of carrying capacity for juvenile steelhead and Chinook (Wall 2014, Rosenfeld et al. 2014). One way to demonstrate the utility of integrating the physical and bioenergetics models is that we can compare differences in current and manipulated (artificial changes to site topography due to potential restoration actions) DEMs to reflect stream restoration efforts. DEM pairs (current and manipulated) were used in NREI simulations to demonstrate the predicted benefit of the potential restoration actions to habitat quality and carrying capacity of juvenile steelhead (Wall 2014). Tools have been developed to efficiently integrate CHaMP data into these models, allowing for model predictions at several hundred sites. A current, high-priority task of the CHaMP development team is to further validate NREI and the Habitat Model predictions of juvenile and adult (spawning) capacity. In locations where survival estimates from mark-recapture information using the Barker model (Conner et al. 2015) were feasible, length was shown to be a strong covariate, allowing growth models such as Weber et al. (2015) or NREI to be used to also predict survival. Capacity and survival estimates using these modelling approaches are currently incorporated into a life cycle model created by ISEMP. This life cycle model provides a way to relate current, potential, expected, and restored fish habitat, as described by CHaMP, to population dynamics of salmon and steelhead.

How has CHaMP compared its protocols with those of other groups to resolve differences in habitat monitoring approaches and elements?

Through the course of the CHaMP pilot study many efforts were made by CHaMP to compare its protocol to that of other groups. For example, in 2012 we conducted a protocol and metric comparison with USFS's PIBO monitoring program. The goal of that study was, based on results, to recommend changes and actions that balanced project specific logistics with progress towards regional approaches to collecting, analyzing, reporting and sharing aquatic habitat monitor data. To date, CHaMP has made several adjustments to specific field collection methods to align data more completely with the USFS's PIBO monitoring program. Also, we have piloted a joint data management system between these two programs and are engaging with PNAMP on the task of developing regional habitat metric data exchange templates - several steps forward toward the goal of a region-wide stream habitat monitoring effort. The process of aligning metrics and indicators is an ongoing effort underway within regional programs.

How can different approaches to design, data collection, data storage, and data analysis provide a test of the efficacy of scaling up from past efforts while still allowing and encouraging other promising, or well proven, efforts to continue?

The use of a common survey design (GRTS) facilitates, in a scale-independent manner, the merging or integration of monitoring measurements and metrics. CHaMP in no way pioneered the use of a common survey design but certainly has taken advantage of the extensive history of environmental monitoring programs being based on GRTS. CHaMP has participated in the development of GRTS application tools that are now available online (monitoringresources.org) and has worked with the GRTS development group to extend the current toolset to facilitate the inclusion of legacy monitoring locations and integrate multiple statistical survey designs. CHaMP's extensive use of protocol-based metadata allows the efficient sharing of similar measurements and metrics. For example, CHaMP is collaborating with USFS's PIBO monitoring program to develop network predictive models of CHaMP metrics that make use of PIBO metrics for calibration and validation. A similar analysis was undertaken by ODFW and CRITFC to allow the use of CHaMP data in the ODFW HabRate watershed evaluation process.

Describe how CHaMP has considered the value of "non-standard" metrics and methods (2011 ISRP)

Metrics and products from CHaMP are evaluated based on performance and utility. A variance-decomposition is completed annually on all metrics to evaluate performance criteria, including repeatability, signal to noise ratio, and spatial variation. If metrics do not meet performance criteria then the collection methods and/or calculations are re-evaluated to determine whether improvements are feasible and worthwhile. A much broader set of tools and criteria are used to determine metric utility, but CHaMP has generally defined a metric as useful if it informs fish-habitat models and relationships, effectiveness monitoring parameters, or enables relating CHaMP monitoring to other habitat monitoring programs.

The spatially explicit topographic data produced by CHaMP is sometimes considered 'non-standard' in the aquatic habitat monitoring community. However, both CHaMP and ISEMP have developed software and novel metrics to maximize the utility of these spatially explicit and continuous datasets. For example, IMWs have used changes in topography over time captured by GCD metrics and topographic-based products to document changes in topography coinciding with restoration actions (effectiveness monitoring). This has been done in the Entiat, Asotin, and Bridge Creek IMWs. Additionally, the Asotin IMW monitoring group has used the topographic products from CHaMP surveys for project planning. Manual manipulations of the CHaMP topography were done to mimic restoration actions in several locations and then a hydraulic model was run to predict outcomes of actions. Restoration locations that maximized desired outcomes were then selected for implementation. The 'non-standard' topographic products have also been used as source data for the Habitat Model, NREI, and bioenergetics modeling. The CHaMP topography and substrate metrics are used as inputs to a near-3d hydraulic model that produces spatially explicit estimates of velocities and water depth at a site. Although the hydraulic model outputs are valuable stand-alone products to hydrologists and restoration planners, the outputs, along with additional substrate metrics, can also be used as inputs to a Habitat Model as well as an NREI model to produce additional monitoring metrics based on fish-habitat relationships. The CHaMP metrics and products feeding these models are therefore considered 'valuable' by the program. CHaMP has also used suites of metrics in Quantile Regression Models to determine which metrics correlate to patterns in fish density both within and across CHaMP watersheds. Lastly, CHaMP generates metrics that are commonly produced by other large scale monitoring programs, including PIBO and AREMP, providing opportunities to integrate metrics across programs. This integration allows programs to extend the spatial or temporal extents of their management models to areas or time frames that were not directly measured through their program.

Field Implementation

Has it been possible for a 3-person CHaMP crew to sample an average of a site per day, particularly in remote areas and/or at sites with complex channel morphology?

On average 80% of CHaMP sites can be sampled in a day by a three person crew. This is variable depending on site accessibility and site complexity. As with any habitat program, remote sites add a degree of logistical complexity that makes it difficult to achieve a one-day sampling objective, when, for example, crews spend 3-4 hours hiking to a site. At complex sites, we encourage crews to take a higher density of points in the topographic survey when they have encountered a complex section of stream, which will add time to the job of measuring a site's topography. The goal is to best capture the channel topography in the most efficient fashion possible.

Describe whether CHaMP has been able to transfer data collection and tool expertise to CHaMP collaborators, including whether they will eventually have the staff expertise not only to collect the data using CHaMP protocols, but to effectively understand and use the modeling programs and other analytical tools to support and document the benefits of their habitat restoration programs.

To date, several steps have been taken to transfer knowledge of CHaMP data collection protocols and analysis tools to relevant parties. Firstly, collaborators participate in an annual preseason training each year, during which intensive training on field surveying, data processing, and data management is provided. Training has been attended by numerous collaborators beyond the original scope of the Interior Columbia River Basin, including northern California (CDFW), the Umpqua, and extension to the Aquatic Effectiveness Monitoring Program (AEM) monitoring project-based restoration throughout the Pacific Northwest. Secondly, CHaMP software developers (North Arrow Research and USU) have provided in-person training sessions and published online modules for CHaMP-affiliated and non-affiliated parties, on GCD, which is one of CHaMP's more developed and freely available analysis tools. This model of knowledge transfer is envisioned for other CHaMP analysis tools, such as the CHaMP Habitat Model toolkit. CHaMP-affiliated staff have also presented to, and coordinated analysis products with, technical staff from collaborating organizations such as the affiliates of the Biological Opinion's Adaptive Management Implementation Plan (specifically the AMIP Life Cycle Modeling group), the Expert Panel groups that assess tributary habitat improvement projects (Bureau of Reclamation, BPA and NOAA led process), and regional stakeholder groups (e.g., Middle Fork John Day Intensively Monitored Watershed group and Regional Technical Team of the Upper Columbia). Lastly, during this year's training workshop (1-10 June, 2015), CHaMP will hold a first-ever 'advanced user' session designed to introduce collaborators and interested parties to the latest in tool development.

Study Design

How will the results obtained from monitoring individual sites within a watershed be "rolled up" … to advance generalizations about status and trends in habitat condition for the watershed as a whole?

There are two methodologies CHaMP uses to summarize monitoring results, from site-level measurements to inference at larger spatial scales, including the watershed scale. At the watershed scale, sample sizes are generally large enough ($n > 20$) that design-based inference is the most appropriate statistical technique. Design-based inference provides robust and accurate, unbiased estimates for the distribution of CHaMP response metrics, and requires no distributional assumptions. We use the spsurvey package in R, which is specifically tailored to the analysis of GRTS-based sampling designs such as those used in CHaMP.

In watersheds where sample sizes are small, or in areas where we want to make estimates at sub-watershed spatial scales and are limited to small sample sizes, we either replace or augment design-based estimates with model-based estimates. To do this, we relate CHaMP metrics to globally available attributes (GAAs) – attributes describing geomorphology, temperature, biology, River Style, etc., that have been quantified continuously across the entire spatial domain of interest. Where GAAs are correlated with CHaMP metrics, we can generate statistical models that can be used to predict CHaMP metrics at unmeasured CHaMP sites. These models may be purely empirical, but are most effective when the GAA and the form of the derived model reflect underlying processes governing CHaMP metrics. While there are benefits to model-based approaches, caution must be exercised as these methods may be less statistically robust. Distributional assumptions, presence or absence of spatial autocorrelation, etc., must be considered in the construction of models to avoid introduction of bias into parameter estimates. In addition, sampling design considerations must be incorporated into a model based analysis just as in a design –based analysis.

Given an empirical model relating CHaMP metrics to GAAs, estimates at any spatial scale can be made by using the models to predict CHaMP metrics at all sites within the spatial domain of interest and then infer population-based statistics (mean and distribution) for the domain of interest. In some cases, we choose to augment limited field measurements by imputing with modeled data. The imputation process takes into account the higher precision of the field data and the additional error present in the modeled data, and attempts to generate the best estimate possible for the spatial domain of interest. CHaMP has used Bayesian hierarchical modeling to perform the imputation process.

How can the results from CHaMP watersheds be extrapolated to unmonitored watersheds within the interior Columbia River Basin?

Each CHaMP watershed is unique, due to a combination of geomorphic, biologic, and other attributes, but the distributions of such attributes are largely overlapping from one watershed to the next. Furthermore, spatial differences in Interior Columbia River Basin watersheds are driven, in large part, by different relative proportions of attributes that are common across the basin as much or more than by a different fundamental set of underlying attributes unique to each watershed. Because of this, we contend that by building high quality models relating these underlying attributes to CHaMP results, we can not only describe unsampled areas within intensively measured CHaMP watersheds, but also extend these models to unmeasured regions within the basin.

We have compiled, and continue to add to, a repository of GAAs for network segments throughout the basin. Many of these GAAs were selected as they have biological meaning to fish and habitat-fish relationships (e.g., primary production, temperature, geomorphic character, beaver assessments). Where we empirically relate CHaMP results to GAAs, we can use models to estimate CHaMP results (base-level CHaMP metrics as well as higher level CHaMP products such as NREI estimates) at locations where direct CHaMP results have not been measured. To the extent that our empirical models reflect the underlying physical processes influencing fish populations (e.g., bottom-up regulation, physiological tolerances, foraging dynamics), we expect limited bias in the application of models built from data-rich watersheds into data-poor or no data watersheds or other regions.

The extent to which we will be successful in accurately extrapolating to unmonitored watersheds is dependent on the level of correlation between GAAs and CHaMP results, and the consistency of such correlations across measured and unmeasured watersheds. While we expect the use of informative and biologically important GAAs to at least partially reflect underlying processes, we nevertheless recognize that empirical models are necessarily imperfect, and we therefore rely on robust statistical modeling practices to quantify the potential for error and bias in empirically modeled estimates. Models constructed to estimate responses at unmeasured sites within a sampled watershed are assessed using validation techniques such Jackknife cross-calibration. Models used to extrapolate into un-measured watersheds are built from data from all available measured watersheds to ensure watershed-to-watershed differences can be quantified, and cross-validation is assessed at the watershed level (e.g., leave one out cross-validation where watershed, rather than site, is the spatial level of cross-validation) to quantify the potential for bias due to watershed specific variations. In addition to cross-validation practices, opportunistic validation data sources (such as USFS's PIBO monitoring data) will be utilized to quantify differences between extrapolated estimates and directly measured results

Describe what CHaMP has learned about being able to make watershed status and trends estimates of habitat quantity and quality through evaluations of sampling intensity and the number of sites (more sites/less intensity vs. few sites of high intensity), and how site selection is influenced, if at all, by proximity to ongoing instream or riparian restoration actions.

First with respect to the status estimates of watershed or coarser level roll-up – a status estimate characterizes the distribution of attribute 'scores' across the spatial scale of interest; as such, it captures the variation among sites. Variation within sites (either temporal or as a result of the measurement process) during the index window can confound the estimate of among site variation. Consequently, CHaMP's design includes revisits to sites during the index window to estimate the relationship between site to site variation and noise. A basic statistical premise is to put sampling effort where variation is greatest. CHaMP's analyses covering the first 4 years of data indicate that the relative balance between revisit intensity (approximately 10 % of sites) provides a good balance between revisit intensity vs. site intensity.

Second, with respect to trend estimation, CHaMP's goal is to estimate the magnitude of trend among a variety of attributes after 9 years. CHaMP's basic design calls for a sample size of 45 sites at which a watershed roll-up 'status' estimate of trend could be determined. CHaMP balances the allocation of sampling effort to sites visited annually and sites visited on a 3-year cycle (some sites visited 9 times, and others visited 3 times in 9 years). One alternate could be to allocate higher intensity to all sites (i.e., visit all sites annually at the expense of reduction of sample size at which trend could be estimated). More annual visits to a site provides a better site trend estimate, at the expense of a greater number of sites at which trend would be estimated. A second alternate could be to eliminate all annual sites, and only sample sites on a 3 year cycle. One of the purposes of including annual sites is to measure short-term yearly variation that would be missed if sites were only visited on a 3-year cycle. The current design allows us to begin to look at yearly variation and the effects, for example, of reducing the number of annual sites. So far, indications are that power to detect the 9-year trend estimate will be relatively unaffected by shifting some annual sites to 3-year panels; however, retention of some annual sites gives us the option of picking up any short-term trend or rapid changes and assists us in evaluating temporal variation among sites.

With respect to the question about whether site selection is affected by proximity to restoration actions – we can think of events that affect sites, whether intentional (as in restoration) or unintentional (accidents; natural events). We are interested in the aggregate effects of all these types of events (large and small) on the watershed level roll-up. If we were to alter the design to exclude/include specific events, we could bias the roll-up estimates. However, with respect to planned restoration actions, it is feasible to take them into account during the design process as we have done for the Tucannon in which sections of the mainstem were scheduled for treatment. We could identify these as treatment/control strata that could be embedded within the broader scale watershed design, and therefore taken into account in the watershed level roll-ups.

Examine/discuss the difficulties and potential benefits in incorporating ad hoc data when trying to extrapolate to other areas.

Careful construction of sample design and documentation of sampling histories is necessary to calculate site-level sample inclusion probabilities for all sites within CHaMP watersheds. In order to calculate unbiased estimates of CHaMP parameters (or higher-level analysis products derived from site-level metrics), or conduct model-based analysis that, for example, relate GAAs to CHaMP metrics or relate CHaMP metrics to fish population dynamics, these sample inclusion probabilities must be calculable. Failing to account or properly calculate sample inclusion probabilities can, and generally does, result in biased parameter estimates.

If, however, other data sources come from designed studies, and given that the metrics are reasonably similar, are repeatable across sampling programs, and that sample units are equivalent, one can combine data from two or more datasets and accurately calculate sample inclusion probabilities for the combined dataset.

In cases where ad-hoc data are, at the site level, obtained with equivalent monitoring, but the site or sites are not selected from a probabilistic design, but rather are selected on other criteria, the data may be used, but we must assume the sample inclusion probability is 100%. The weight given any site in an analysis is proportional to the inverse of the sample inclusion probability, thus a selected site, while included in the analysis, will have an extremely low weight in the analysis. Very little increase in information content is obtained from adding non-probabilistic sites to the analysis.

Informally, there may be opportunities to use ad hoc or other non-CHaMP data as validation points to test empirical models fit to CHaMP data. Ideally, validation data would have the same probabilistic nature as the data used to fit the model, but this is not generally the case for ad hoc data. Thus, only limited credence should be given to such an analysis. However, a very strong model may be valid over a large range of spatial or temporal space, and as such this should be qualitatively demonstrable using ad hoc data.

Fish-Habitat Relationships

Explain what value CHaMP can provide to verify assumptions about relationships between habitat conditions and fish populations.

Relationships between habitat conditions and fish populations have proven difficult to describe or are inconsistent across large spatial scales. CHaMP/ISEMP, by implementing novel approaches to measuring fish habitats across the multiple spatial scales used by fish populations, has been able to develop new approaches to describing the relationship between fish and the physical environment they occupy. In addition, CHaMP/ISEMP has also been applying previously developed empirical and mechanistic approaches to predicting fish abundances to new, larger spatial scales to test whether these models are suitable at spatial scales more closely aligned with fish populations.

CHaMP measures and describes lotic habitat using novel approaches that generate detailed fine-scale metrics based on the topography of river channels. Topographic habitat surveys are coupled with reach-scale surveys of structural elements that force aquatic habitat (e.g., riparian vegetation, in-channel coarse wood) and factors that drive prey resource availability for juvenile salmonids (e.g., drifting invertebrates, primary production, solar inputs), allowing CHaMP/ISEMP to simultaneously evaluate the importance of physical and biological factors as they interact to influence fish populations. The description of channel topography using total stations has allowed CHaMP/ISEMP to extend the spatial scale at which mechanistic models describing fish-habitat relationships (e.g., NREI, Habitat Model) can be applied.

CHaMP measures aquatic habitat at many spatial scales using a vetted survey design to provide sound statistical inferences that represent the extent of variation present in the survey extent (Columbia River Basin as well as individual tributary watersheds). This allows CHaMP/ISEMP to test hypotheses about how aquatic habitat structures fish populations across much of the Columbia River Basin, incorporating a variety of ecoregions, land uses, land ownerships, and geomorphic settings. As a result, CHaMP/ISEMP is in a position to evaluate the influence of large-scale drivers of fish-habitat relationships, which have previously made broad generalizations difficult.

An important contribution of CHaMP and ISEMP has been to develop approaches to empirically and mechanistically predict fish abundance and survival as a function of habitat quantity and quality at the scale of river reaches. Furthermore, CHaMP/ISEMP has pioneered new approaches to make continuous predictions of fish populations across river networks that can be integrated in life cycle models to evaluate fish-habitat relationships across many spatial scales (from habitat/geomorphic units to river networks). Comparisons of these multiple approaches will test how robust the currently accepted assumptions are about fish-habitat relationships.

What kinds of analytical methods are being used to relate habitat status and trends to fish status and trends, and why? How are CHaMP habitat parameters being used to determine whether restoration actions are influencing improvement in habitat characteristics and survival of specific fish life-stages, VSP parameters?

The analysis of fish and habitat data is being pursued using a complement of approaches, including:

(A) Statistical models that relate sampling estimates of fish abundance to variables measured within CHaMP reaches; this includes purely empirical approaches, like quantile regression forests, as well as quasi-mechanistic approaches such as structural equation models;

(B) Mechanistic models that make spatially explicit predictions about the suitability of different locations within CHaMP reaches using field measurements (drift, temperature, substrate, discharge, topography) and the results from hydraulic model simulations;

(C) Network-scale statistical models that predict monitoring metrics [(A) and (B)] at unsampled locations so that fish-population-scale inferences can be made;

D) Network-scale process-based models (geomorphic assessments, temperature, and stream productivity models) that predict monitoring metrics [(A} and (B)] at unsampled locations so that population-scale inferences can be made, and

(E) Population-simulation models which use the results from analyses (A) – (D), as well as other fish monitoring datasets, to make predictions of population performance under current and anticipated future habitat conditions; such models will include relationships between habitat conditions and population productivity/capacity.

Together, these approaches provide a basis for relating fish population status to habitat status at three temporal domains. First, by exploring fish–habitat relationships across CHaMP sites/subbasins at a snapshot in time (e.g., (A) – (D) above), a space-for-time view of fish and habitat status is gained. From this, sites/subbasins characterized by poor habitat conditions and low abundance/productivity can be readily identified. Secondly, once a long-term record of CHaMP-based fish and habitat metrics ((A) - (D) above) is constructed for a particular subbasin, temporal trends in both fish population status and habitat condition can be quantified and compared. Further, depending on the restoration design (i.e., timing of actions, presence of controls, etc.) for a basin, causal relationships may be examined using existing (BACI) and/or novel (e.g., Bayesian hierarchical intervention analysis) intervention analysis approaches. Finally, given an expectation about how habitat conditions will trend in the future (e.g., due to planned restoration), the life cycle modeling framework can provide insight on the related population response. Importantly, this last step provides a basis for comparing observed outcomes to expectations, a necessary component of adaptive management.

Project Effectiveness

How effective [considering scope, scale, duration and cost of implementation] are various treatment types and BMPs for addressing identified habitat impairments?

CHaMP and ISEMP are involved in the evaluation of a number of stream restoration project types, but CHaMP and ISEMP are in no way systematically evaluating stream habitat treatment types and BMPs, so it isn't clear that these projects would be in a position to address this question. This type of question may not even be appropriate for the project to tackle since issues of "cost of implementation" seem best handled at the level of the Environment, Fish and Wildlife Program as a whole. However, CHaMP and ISEMP will be able to address the efficacy of particular stream habitat rehabilitation approaches with respect to changes in physical and biological stream components and the resultant fish population response.

How can project planners and implementers use habitat-fish synthesis products to evaluate whether specific or general restoration strategies are effective in a geographic area?

Designers and implementers of stream habitat restoration projects, as well as those developing watershed-level restoration strategies can make use of many CHaMP/ISEMP data and analysis products. Network and watershed estimates of stream habitat condition, either single monitoring metrics such as <D50> or <Residual_Pool_Depth>, or the synthesis metrics such as <summer_rearing_capacity> are directly consumed by groups assessing watersheds for status and trends at reach or subwatershed scales. In the Asotin, the mechanistic benefit to fish of specific restoration actions designs have been demonstrated through the application of the NREI model predictions of stream habitat quality. NREI predictions based on pre- and post -restoration DEMs illustrate mechanistically how changes in stream physical and biological components such as bed form, water temperature, and drift biomass result in altered rearing or spawning capacity, or juvenile survival. In the Middle Fork John Day and the Lemhi Rivers, restoration scenarios were used to parameterize life cycle models to understand the extent and timing of the population response to action implementation as well as to test a suite of restoration actions. he Habitat Model Workbench is being used in a manner similar to that of the NREI modeling described above to explore the potential fish benefit of specific restoration action geometries. Habitat Model predictions of stream habitat quality based on DEMs produce information-rich estimates of habitat quality based on individual restoration action features, and as such, can be used as a design tool to tune site specific restoration concepts. At the watershed scale, River Style assessments can be used to develop overall restoration action plans. The basic components of an River Styles assessment contain the current geomorphic condition (RS -1), the degree of impairment or condition variant (RS-2) and the restoration potential (RS-3). The restoration potential can be refined into a restoration plan (RS-4) by applying local priorities and constraints (Brierley & Fryirs, 2005).

In evaluating restoration effectiveness, how do CHaMP and ISEMP propose to accommodate factors affecting fish populations in non-wadeable areas downstream of CHaMP sampling sites, including the mainstem, estuary and ocean?

Tools and techniques developed in ISEMP/CHaMP provide the basis for the evaluation of DEM-based information collected from other platforms. For example, red and green LiDAR, photogrammetry, structure for motion, and sonar can collect the necessary information to build DEMs. Utah State University has used these tools to develop bathymetry from sonar, for example, and is testing this in large rivers such as the Colorado. The University and Grand Canyon Monitoring and Research Center are developing approaches to sample substrate with the sonar information as well. The tool ISEMP/CHaMP developed can seamlessly use DEMs collected from these other approaches. In fact, the River Bathymetry Toolkit (RBT) was originally initiated on stream channel topography gathered from green LiDAR.

CHaMP/ISEMP are part of the AMIP life cycle modeling program, and as such, are working to develop of tributary life cycle module that can integrated with regional work on standardizing extra-tributary impacts. This collaboration will allow CHaMP to simultaneously evaluate the relative impacts of restoration actions conducted both within tributaries and in downstream habitats. Simultaneously, CHaMP/ISEMP are able to provide AMIP collaborators with life-stage specific capacity and productivity parameter estimates that are explicitly linked to tributary habitats where salmon and steelhead spawn and juvenile rearing occurs. While integration of tributary and extra-tributary life cycle model structures are being developed and integrated, CHaMP/ISEMP is poised to evaluate restoration effectiveness by using population responses that are independent of downstream impacts (i.e., smolts/ female) or to parameterize life cycle models incorporating tributary specific spawning and rearing conditions with empirically derived SARs that integrate downstream impacts.

How will CHaMP facilitate an evaluation of project effectiveness in watersheds where treatments are not experimentally controlled?

In carefully controlled experiments, causality can be inferred for the various treatments as the driver of differences in response between control and treatment groups. Where treatments are not experimentally controlled and purely observational data are analyzed, as in the IMWs within the CHaMP sample frames, causation cannot be directly inferred. Nevertheless, careful analysis of the data in conjunction with mechanistic biological modeling can provide strong evidence from which to evaluate restoration project effectiveness.

Typically, causation can be supported and defended from observational data when responses are observed, sound scientific judgement is used to support the notion that a specific effect caused the observed response, and all reasonable alternate causes are eliminated through careful analysis. In CHaMP, changes in CHaMP metric responses before and after treatment can be observed whether or not the treatment is from a controlled experiment. We can pose the questions: might there be other factors, besides treatment, driving the observed response? Are these measured? From the data can we eliminate these other potential factors as driving the response? Are these other factors present in untreated sites, and do we see the same response at those sites? Following this line of questioning, we can hope to eliminate most non-treatment causes for the observed response, and may fairly argue that the treatment is the likely driver. While not as robust as a controlled experiment in attributing cause, this is certainly a common practice across all sciences.

Additionally, for some potential responses, CHaMP has developed mechanistic models, such as NREI, Habitat Model, etc. These mechanistic models, in some cases used as inputs for the life cycle models, can be used to make testable predictions. If observed responses to restoration treatments are consistent with those predicted a-priori from mechanistic models, this lends further credence to the argument that the treatment is indeed the driver of the response rather than merely a spurious coincidence.

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Wheaton, J. M., J. Brasington, S. E. Darby, J. Merz, G. B. Pasternack, D. Sear, and D. Vericat. 2010. Linking geomorphic changes to salmonid habitat at a scale relevant to fish. River Research and Applications 26(4):469–486.

Wheaton, J. M., J. Brasington, S. E. Darby, and D. A. Sear. 2010. Accounting for uncertainty in DEMs from repeat topographic surveys: improved sediment budgets. Earth Surface Processes and Landforms **35**:136-156.

APPENDIX A1: DATA LINKS AND RESOURCES

Data and tools produced by CHaMP are available through multiple websites and are subject to change as updates are made to the program. Although these products are currently affiliated with CHaMP, much of the developmental work behind these tools has been facilitated by ISEMP. Contact Carol Volk (carol@southforkresearch.org) for additional information on data accessibility and availability.

Monitoring Data and Metadata

CHaMP Monitoring Website

The primary function of this website is to support CHaMP crew data collection, quality assurance, and data review. It also serves as a central portal for protocol, reports, presentations, and training material. Information is best accessed by logging in but user permissions may restrict availability of some information. New users may request a login via the website. The following information is available:

Documents: The Documents page provides downloadable resources such as **presentations**, **reports**, **training material**, and **GIS data,** including geodatabases of GIS data within CHaMP watersheds (e.g., DEMs, hydrography, Land Use-Land Cover, Ownership, CHaMP sampling domains, ESA fish populations, etc.).

Site evaluations: The Site Evaluations page provides downloadable information on site accessibility for crew data collection. This information is also used for the GRTS estimation process.

Data Exports: The Data Export page is the fastest location to download **field measurement data, visit metrics, visit information**, and **temperature logger data** collected by crews. MS Access databases of field-collected measurement data are available for standard or custom user download.

Spatial Data-FTP site: CHaMP collaborators can access spatial data, such as **topographic survey data**, **DEMs, RBT outputs**, and **hydraulic model outputs** from an FTP site. The FTP site is a convenient method for downloading large batches of visit data. Contact us for additional access information.

CHaMP Survey Designs: The Monitoring Sample Designer is an online tool that is currently used to store **survey design details**, including **year-by-year planned sampling** lists, downloadable **GIS files of sites**, and **metadata on design decisions**.

Survey designs: CHaMP survey designs are listed in the Monitoring Sample Designer under the CHaMP Monitoring Program.

Sampling domains (GIS frames): The target sampling domain used for survey design development is available as part of the background information of the Survey Design. Review the survey design to access specific metadata on the GIS frames.

Models

Hydraulic model

The Delft-3D hydraulic model utilized to produce velocity and depth estimates for CHaMP sites is available by contacting Matt Nahorniak (matt@southforkresearch.org). Hydraulic model outputs are available via the FTP site. We are currently processing all quality assured visits with topographic data through the hydraulic model and data availability subject to change.

Network temperature models

The MODIS network temperature model is used to generate spatially and temporally continuous estimates of stream temperature within CHaMP watersheds. This stream network-based temperature models utilizes MODIS Land Surface Temperature calibrated using CHaMP site temperature data and are available for 2011, 2012, and 2013. 2014 will be available in Fall 2015 as the 2015 field season will include temperature downloads for August-December 2014. These models are currently available upon request and include temperature estimates along streams within CHaMP watersheds every 8 days. Contact Kristina.mcnyset@noaa.gov for additional information and data access.

Habitat Model

A library of habitat suitability curves, including those utilized by CHaMP, and the functionality to run different curves against CHaMP visit data have been packaged into the Habitat Model software. The software is available for use but still undergoing developmental improvements. CHaMP is currently running a set of standard habitat suitability curves for each visit with hydraulic model outputs for 2011-2014. These data will likely be available on champmonitoring.org by the fall of 2015, but are currently available upon request. Contact Sara Bangen (Sara.bangen@gmail.com) for additional information.

NREI

Information about the NREI model currently under development for use with CHaMP surveys and preliminary model outputs are available upon request from Eric Wall (c.eric.wall@gmail.com); however, there are many assumptions and watershed-specific decisions on fish size and temperature ranges that should be reviewed prior to data use and therefore available data should be considered preliminary outputs.

Tools

CHaMP Toolbar

The CHaMP Toolbar is utilized by CHaMP crews to process topographic data collected using the CHaMP protocol in a standardized and streamlined workflow. This toolbar packages a variety of tools, including the Transformation Tool, various River Bathymetry Toolkit functions (e.g., wetted extent generation and cross sections), and standard GIS processing tools (e.g., clipping functions, TIN generation, and DEM generation).

River Bathymetry Toolkit (RBT)

The core calculation of many CHaMP metrics is based on functionality of the River Bathymetry Toolkit, which is a collaborative development effort serving multiple interested parties. RBT exists as both a desktop-level toolbar and command-line executable. Contact Philip Bailey (philip@northarrowresearch.com) for additional information.

RBT Workbench

The RBT workbench is a tool developed to manage analyst batch processing of surveys through analytical tools. This is one of the newest tools utilized by CHaMP and is available for use but is still undergoing developmental improvements.

GCD Toolbar

The core functionality of the Geomorphic Change Detection Toolbar is utilized to produce several CHaMP metrics. CHaMP topographic data that has been processed by RBT can be opened via the GCD toolbar for result review.

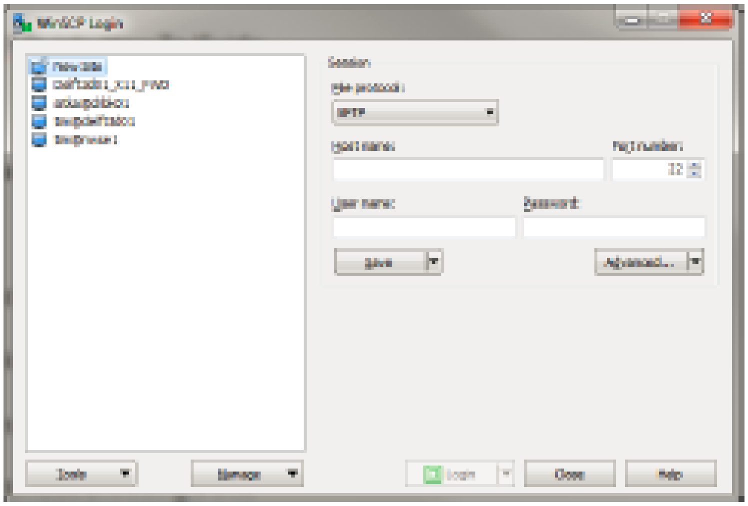
Retrieving CHaMP data using the FTP site:

WinSCP: A Quick HowTo Guide

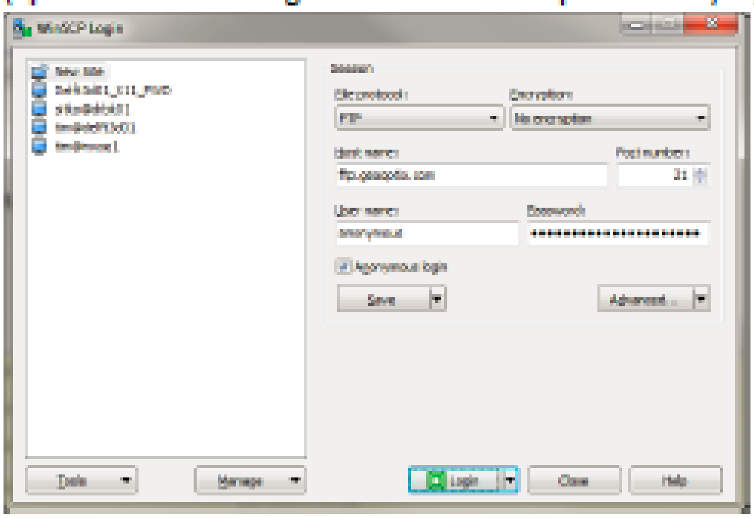
Last update by tim@sitkatech.com 2015.01.19

Site Management

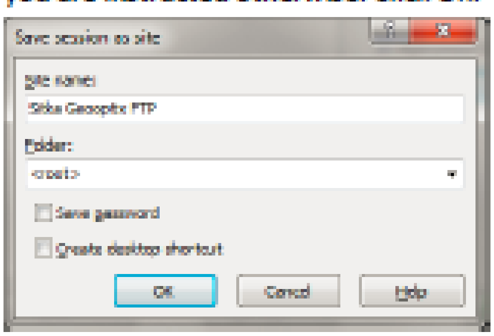
1. Launch WinSCP. This will display the Site Management dialog



2. Create a new FTP site by filling in the Session options shown (update fields according to Sitka information provided to you)

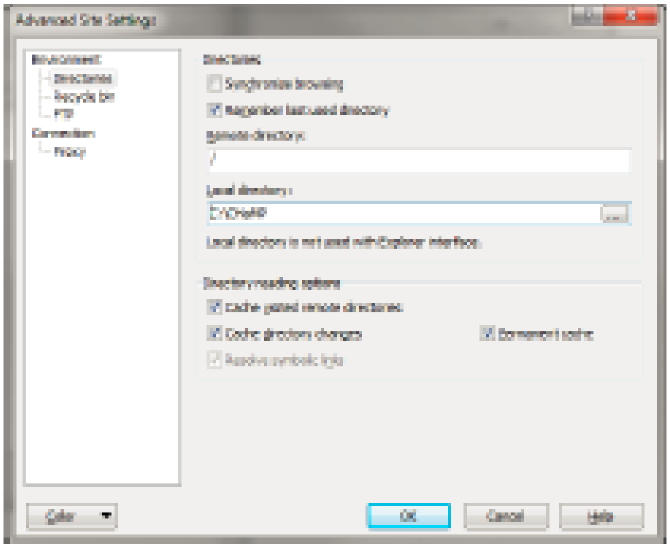


3. Click the Save button and enter a meaningful Site Name. Leave the Folder set as <root> unless you are instructed otherwise. Click OK.



Optional: Update settings for your local folder (where you want the files to be downloaded)

1. Right click the site you just made and select Edit.
2. Click the Advanced button.
3. Click the Directories sub-element on the right and change the Local Directory setting to a directory of your choice.



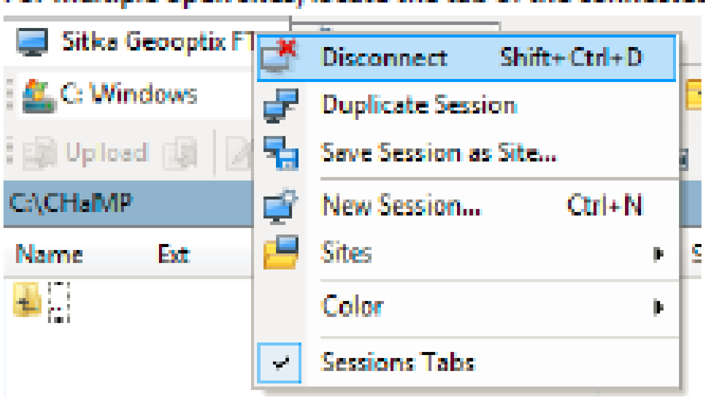
4. Click OK. Click Save

Connecting to a Site

1. Launch WinSCP (as necessary)
2. Select the Site by clicking it once.
3. Click the Login button

Disconnecting from a Site

1. For multiple open sites, locate the tab of the connected site, right click and select "Disconnect"

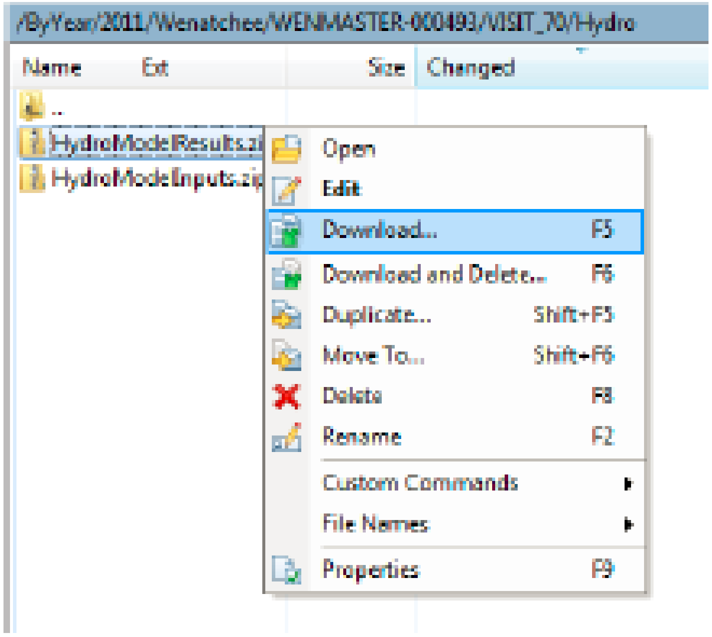


2. If you have no other sites open, you will be shown the Site Manager. Click close to exit the application.

Alternately, you may exit the app by simply closing the main window.

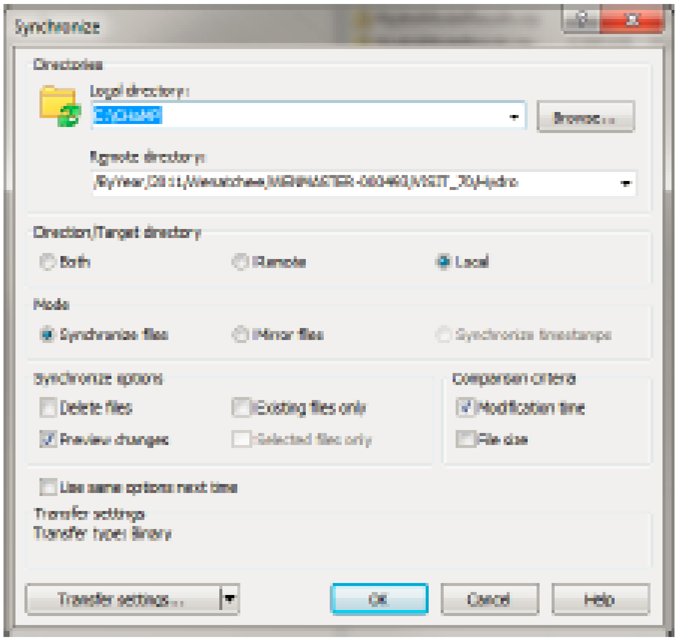
Downloading from a Site

1. Launch WinSCP, connect to a site. Navigate to the file/folder you wish to download using the right content pane.
2. (as needed) Navigate in the left content pane to the destination for your download.
3. Select the contents to be downloaded in the right pane. Drag and drop to the left pane. (alternately, right click on any selected item and choose Download)

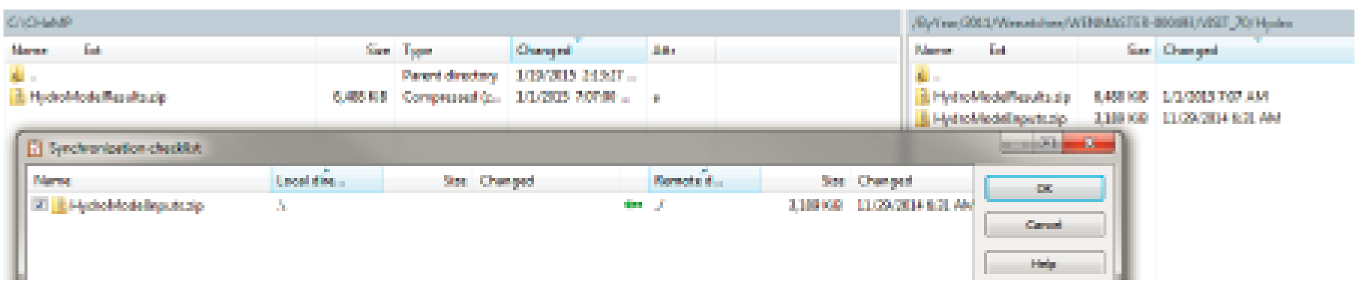


Synchronized Downloading from a Site

1. Launch WinSCP, connect to a site, use the right content pane to locate the files to download.
2. Use the left content pane to locate the files that have been downloaded, but require updating/refreshing.
3. Click the Commands menu and select Synchronize.
4. Look over the settings displayed, in general, no changes should be required. Click OK to see a list of files to be updated/transferred.
   Note: Always be sure that the Direction/Target Path is set to "Local" or data may be deleted from your system.

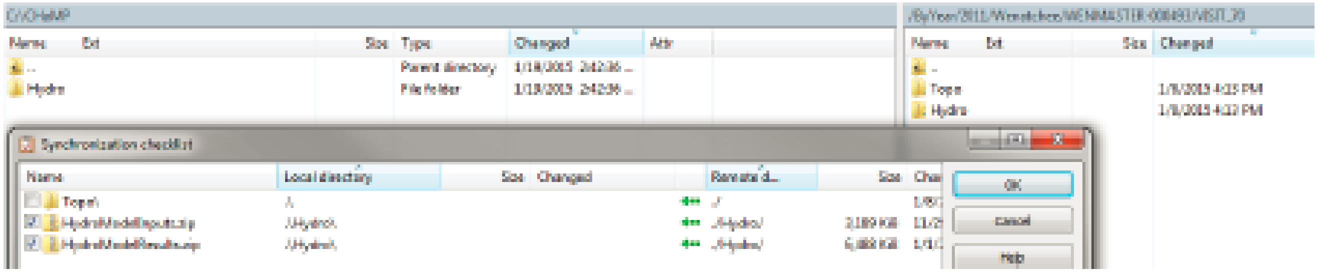


5. Example list of files to synchronize from the site to workstation.
   Notice that the file found on the right (remote), but not existing on the left (local) has been automatically selected for download.
   Click OK to initiate the download of all files listed.



6. This operation applies recursively to directories as well. Uncheck any directories/files that are not of interest.
   In the example below we do not want to download 'Topo', so it is unchecked.
   Notice also that the 'Hydro' directory exists in both the left and the right panes. Therefore the contents of the local Hydro directory will be updated with the contents of the remote Hydro directory.
   If there Directories that do not exist on the left, will be created as needed. If the Topo directory

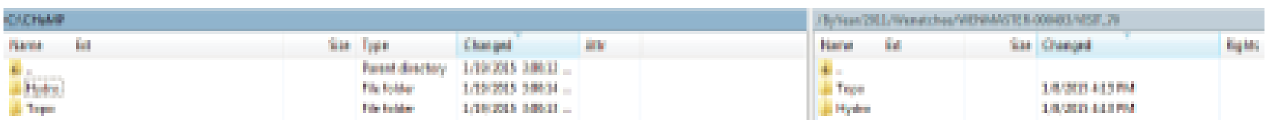
was checked, because it does not exist locally, it will be created and all contents downloaded.



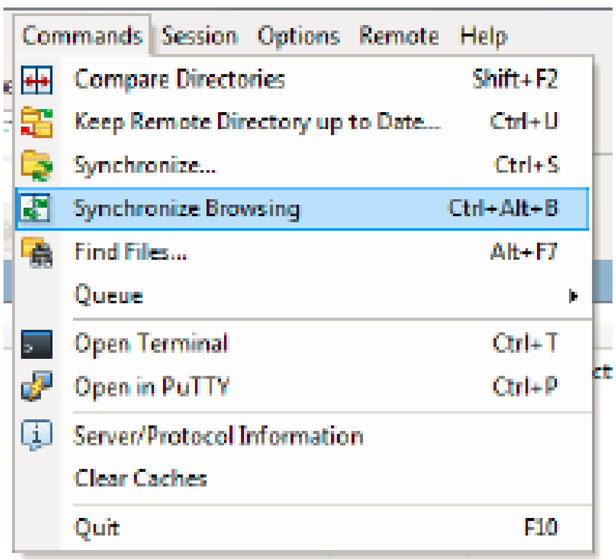
Synchronized Browsing a Site

About: If you have downloaded an entire directory tree from an FTP host, synchronized browsing is a convenient way to save time navigating to a file or folder of interest.

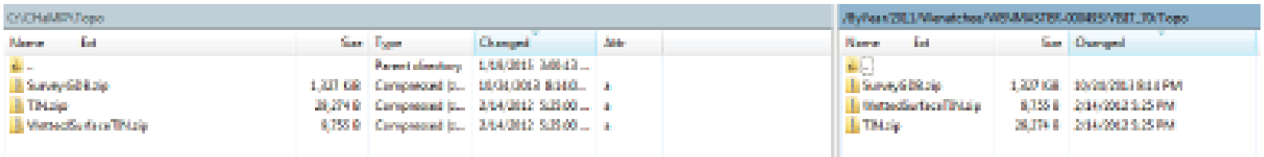
1. As usual, launch WinSCP, connect to a site.
2. Start each pane at the top level of the directory tree you have previously downloaded such that each side has the same starting point for browsing.



3. Click the Commands menu and select "Synchronized Browsing"



4. Now navigate in either the left or right pane and you will notice that the opposite pane will be automatically updated to match.
   In the example below, we have navigated into the Topo folder.
   In the file listing header which shows the path, we can see that both the left and right panes are in the Topo folder.



Notice that that the folders above the Topo folder do not match. That is not necessary. So a key point with this feature is that only the folders you navigate need to match.
(Also, the files of the folders do not need to match either even though they do in this example).

Final Notes

WinSCP is a feature rich free tool that can be used for FTP, SFTP, and SCP (Linux) copying, updating, and synchronizing of files. The help pages are very user friendly as well. If you have questions this guide does not answer, the first place to look is the online help. Just click the Help button in whatever task you are doing or select "Contents" from the Help menu.

Enjoy!

Sitka Technology Group - Operations Dept.

APPENDIX A2: PUBLICATIONS

Bangen, S., J. Wheaton, N. Bouwes, B. Bouwes, C. Jordan. 2014. A methodological intercomparison of topographic survey techniques for characterizing in-stream habitat. Geomorpholgy 206:343-361.

Bangen, S., J. Wheaton, N. Bouwes, C. Jordan, C. Volk, M.B. Ward. 2014. Crew variability in topographic surveys for monitoring wadeable streams: a case study from the Columbia River Basin. Earth Surface Processes and Landforms DOI: 10.1002/esp.3600.

Bouwes, N., J. Moberg, N. Weber, B. Bouwes, C. Beasley, S. Bennett, A.C. Hill, C.E. Jordan, R. Miller, P. Nelle, M. Polino, S. Rentmeester, B. Semmens, C. Volk, M.B. Ward, G. Wathen, and J. White. 2011. Scientific protocol for salmonid habitat surveys within the Columbia Habitat Monitoring Program. Prepared by the Integrated Status and Effectiveness Monitoring Program and published by Terraqua, Inc., Wauconda, WA.

Camp, R. and Wheaton, J.M. 2014. Streamlining Field Data Collection With Mobile Apps. EOS. 95 (49): 453-454. DOI: 10.1002/2014EO490001

CHaMP. 2013. The Columbia Habitat Monitoring Program: 2012 Second Year Lessons Learned Project Synthesis Report (2011-006). Prepared for the Bonneville Power Administration by CHaMP. Published by Bonneville Power Administration, Portland, OR. 63 pages.

ISEMP. 2014. The Integrated Status and Effectiveness Monitoring Program: Annual Report for Fiscal Year 2013. Prepared by ISEMP for the Bonneville Power Administration. Published by Bonneville Power Administration. 46 pages.

Hall, J., M. Pollock, S. Hoh, C. Volk, J. Goldsmith, C. Jordan. Evaluation of deep-planting and herbivore protection methods to restore riparian vegetation in a semiarid watershed without irrigation. Restoration Ecology. In Press.

Hall, J., M. Pollock, S. Hoh. 2011. Methods for Successful Establishment of Cottonwood and Willow Along an Incised Stream in Semiarid Eastern Oregon, USA. Ecological Restoration 29(3): 261-269.

Ohms, H., M. Sloat, G. Reeves, C. Jordan, and J. Dunham. 2013. Influence of sex, migration distance, and latitude on life history expression in steelhead and rainbow trout (Oncorhynchus mykiss). Canadian Journal of Fisheries and Aquatic Sciences. 70: 1–11 dx.doi.org/10.1139/cjfas-2013-0274

Pollock, M., J. Wheaton, C. Jordan, N. Bouwes, C. Volk. 2014. Channel incision evolution models that incorporate beaver and restoration activities. BioScience 64:279-290

Pollock, M., Beechie T , Wheaton JM, Jordan C, Bouwes N, Weber N, and Volk C. 2014. Using Beaver Dams to Restore Incised Stream Ecosystems. Bioscience. DOI: 10.1093/biosci/biu036.

Pollock M, Wheaton JM, Bouwes N and Jordan CE. 2012. Working with Beaver to Restore Salmon Habitat in the Bridge Creek Intensively Monitored Watershed: Design Rationale and Hypotheses. NOAA Technical Memorandum, NOAA Northwest Fisheries Science Center, Seattle, WA, 108 pp.

Rosenfeld, J., N. Bouwes, C. Wall. S. Naiman. 2013. Successes, failures, and opportunities in the practical application of drift-foraging models. Environmental Biology of Fishes. DOI 10.1007/s10641-013-0195-6

Ruesch A., C. Torgersen, J. Lawler, J. Olden, E. Peterson, C. Volk, D. Lawrence. 2012. Projected Climate-Induced Habitat Loss for Salmonids in the John Day River Network, Oregon, U.S.A. Conservation Biology. DOI: 10.1111/j.1523-1739.2012.01897.x.

Volk, C., K. Barnas, Y. Lucero. 2014. Why is data sharing in collaborative natural resource efforts so hard and what can we do to improve it? Environmental Management. 53:883-893, DOI 10.1007/s00267-014-0258-2

Ward, M.B., P. Nelle and S.M. Walker. (editors). 2012. CHaMP: 2011 Pilot Year Lessons Learned Project Synthesis Report. Prepared for the Bonneville Power Administration by CHaMP. Published by Bonneville Power Administration, Portland, OR. 95 pages.

Ward, E., B. Semmens, D. Phillips, J. Moore, and N. Bouwes. 2011. A quantitative approach to combine sources in stable isotope mixing models. Ecosphere 2(2):art19. doi:10.1890/ES10-00190.1

Weber, Nicholas P. 2009. Evaluation of Macroinvertebrates as a Food Resource in the Assessment of Lotic Salmonid Habitat. All Graduate Theses and Dissertations. Paper 286. http://digitalcommons.usu.edu/etd/286

Weber, N. P., N. Bouwes, and C. E. Jordan. 2014. Estimation of salmonid habitat growth potential through measurements of invertebrate food abundance and temperature. Canadian Journal of Fisheries and Aquatic Sciences 71(8):1158–1170.

Wheaton J., C. Garrard, C. Volk, K. Whitehead. 2012. A Simple, Interactive GIS Tool for Transforming Assumed Total Station Surveys to Real World Coordinates - The CHaMP Transformation Tool. Computers & Geosciences. 42: 28-36. DOI: 10.1016/j.cageo.2012.02.003.

Wheaton JM, Brasington J, Darby SE and Sear D. 2010. Accounting for Uncertainty in DEMs from Repeat Topographic Surveys: Improved Sediment Budgets. Earth Surface Processes and Landforms. 35 (2): 136-156. DOI: 10.1002/esp.1886

In Review◎. Bennett S, Bouwes N, Wheaton JM, and Camp R*. Adapting Adaptive Management for Testing the Effectiveness of Stream Restoration: An Intensively Monitored Watershed Example. Submitted to Fisheries.

In Review◎. Hough-Snee, N., Kasprak, A., Rossi, R., Bouwes, N. and Wheaton, J., Hydrogeomorphic and biotic drivers of instream wood differ across sub-basins of the Columbia River Basin, USA. Submitted to River Research and Applications.

In Review◎. Larsen, D.P. et al. The Columbia Habitat Monitoring Program (CHaMP) survey design in the Columbia River Basin: development and lessons learned on how best to spread your monitoring dollars across the landscape.

In Review◎. Wheaton, J.M. et al. Organizing stream network data: a comparison of the River Styles approach to other commonly used methods.

In Review◎. Wheaton JM, Fryirs K, Brierley G, Bangen S, Bouwes N, and O'Brien G. Geomorphic Mapping and Taxonomy of Fluvial Landforms. Submitted to Geomorphology.

In Revision◎. Schaffrath K, Belmont P and Wheaton JM. Landscape-scale geomorphic change detection: Quantifying spatially-variable uncertainty and circumventing legacy data issues. For submission to Geomorphology.

In Press. Nahorniak, M. Using inverse probability bootstrap sampling to eliminate sample induced bias in model based analysis of unequal probability samples.

ISEMP-CHaMP TECHNICAL PAPERS 2015

As well as product development for use by managers and decision-makers, ISEMP and CHaMP are also focusing on submitting manuscripts to relevant peer-reviewed journals. Manuscripts we anticipate being ready for submission by September 2015 include:

Measurable fish responses to habitat restoration actions in Bridge Creek, John Day subbasin, OR. Topic: Juvenile salmonid abundance, growth and survival are all affected by the restoration treatments implemented in the Bridge Creek IMW. Lead: Nick Bouwes. Status: Draft

Linking habitat management actions to fish response through reach-scale net rate of energy intake (NREI). Topic: Demonstration that stream habitat change affects the bioenergetics of rearing salmonids in an understandable and predictable manner. Lead: Carl Saunders. Status: Draft

Lemhi Intensively Monitored Watershed Life Cycle Model: an approach to integrate fish biology with population process dependent on habitat condition to support the development and testing of management action scenarios. Topic: Large-scale stream restoration actions can take years to impact populations; however, using life cycle models, estimates of the restoration benefit can be made based on our current knowledge of the system. Lead: Chris Beasley. Status: Draft

A modelling approach to allocate steelhead adult escapement over the Lower Granite Dam to upstream tributary populations: improving upon the art of steelhead redd surveys. Topic: Estimating population-level abundance for steelhead in the Snake River ESU has not been possible before based on traditional spawning ground based methods; however, using PIT tags it is now possible to generate estimates for most of these populations. Lead: Chris Beasley. Status: Draft.

Entiat Intensively Monitored Watershed Life Cycle Model for spring Chinook and steelhead: an approach to integrate fish biology with population process dependent on habitat condition to support the development and testing of management action scenarios. Topic: Large-scale stream restoration actions can take years to impact populations; however, using life cycle models, estimates of the restoration benefit can be made based on our current knowledge of the system. Lead: Shubha Pandit. Status: Draft

An overview of the history, development, implementation and utility of the Columbia Habitat Monitoring Program (CHaMP). Topic: CHaMP was developed to meet a general need to describe stream habitat quality and quantity for salmonids based on a number of existing methods. Lead: Mike Ward. Status: Draft

Estimating spring Chinook survival and movement using a multi-state modeling approach in the Entiat Intensively Monitored Watershed, WA. Topic: Method to estimate survival and movement of juvenile salmonids in the context of a watershed restoration project. Lead: Shubha Pandit. Status: Draft

Developing an effective model for predicting spatially continuous, daily stream temperatures from remotely sensed land surface temperatures within the John Day Basin, Oregon. Topic: Stream temperature varies across the network and through time, but is only monitored at points – can stream temperature be estimated in a spatially-temporally continuous manner? Lead: Kris McNyset. Status: Draft

Extrapolating site-level metrics collected under the Columbia Habitat Monitoring Program (CHaMP) across the stream network. Topic: Habitat monitoring occurs at points across the stream network, but management decision need to be made across the entire watershed – can stream habitat metrics be estimated at stream reaches that aren't monitored? Lead: Matt Nahorniak. Status: Draft.

Hydraulic modeling of river and stream reaches sampled in the Columbia Habitat Monitoring Program. Topic: An automated method to generate hydraulic models at habitat monitoring sites from generic topographic data. Lead: Matt Nahorniak. Status: Draft

Geomorphic change detection at the reach scale using tools developed by the Columbia Habitat Monitoring Program (CHaMP) in the Columbia River Basin. Topic: Repeated topographic surveys can be used to capture spatially explicit geomorphic change, which is an important descriptor of stream habitat. Lead: Joe Wheaton. Status: Draft

Automating Habitat Suitability Indices and their application to life cycle models. Topic: Stream habitat metrics have been evaluated against fish density by a number of methods, resulting in a wide range of potential habitat suitability relationships that can now be generated automatically from CHaMP survey data. Lead: Sara Bangen. Status: Draft

Evidence of floodplain reconnection following installation of beaver dam analogues in Bridge Creek, John Day Basin (OR), a small and incised stream. Topic: Restoration of floodplain processes in the Bridge Creek IMW has resulted in dramatic changes in riparian vegetation and ground water levels and temperature. Lead: Carol Volk. Status: Draft

Classifying watersheds in the Columbia River Basin. Topic: Watersheds in an area as large as the Columbia River basin vary naturally in their geomorphic and climatological condition as well as the degree of human disturbance; organizing this diversity into "classes" helps to explain variation in monitoring data. Lead: Thom Whittier. Status: Draft

Development of a network primary production model using data collected under the Columbia Habitat Monitoring Program (CHaMP) in the John Day subbasin. Topic: A driver of variation in stream habitat quality is the degree to which primary production varies across the network; modeling primary production will support modeling fish use of stream habitat. Lead: Carl Saunders. Status: Draft

Comparing the Relative Performance of Downstream Migrant Abundance Estimation Methods Through Simulation. Topic: comparative examples of the relative performance of commonly used abundance estimation methods for emigrating juveniles, their bias, and whether the confidence intervals generated consistently capture true abundance. Lead: Steve Tussing. Status: Accepted for publication in the North American Journal of Fisheries Management pending responding to reviewers' comments.

Effect of fish length at tagging on estimating survival probabilities for juvenile anadromous salmon in the Columbia River basin. Topic: the effect of size-at-tagging on survival estimation leveraging data from multiple mark-recapture PIT tagging efforts in the across the ISEMP subbasins from 2006-2014. Lead: Shubha Pandit. Status: Draft.

Intensive monitoring of a rearing spring Chinook cohort in the Little Wenatchee River, Columbia Basin, Washington. Topic: In 2014, ISEMP initiated an intensive monitoring study in the Little Wenatchee River in order to contribute towards parameterizing a life cycle model, and inform future study designs. Lead: Keith van den Broek. Status: Draft.

APPENDIX A3: LIST OF MODEL INPUTS, OUTPUTS AND ECOLOGICAL CONCERNS ADDRESSED

Model	Model Status	Model Inputs	Model Outputs	Ecological Concern
River Bathymetry Toolkit (RBT)	Production	CHaMP Topographic Survey	Metrics of channel topography and dimensions, such as sinuosity, gradient, areas, and volumes	Channel Structure and Form Peripheral and Transitional Habitats
Geomorphic Change Detection	Production	Topography	Metrics and locations of areas where topography has changed, such as areas and volumes of erosion and deposition, within a site.	Channel Structure and Form Sediment Conditions
Hydraulic Model	Production	Topography Discharge Substrate Size	Spatially explicit models describing velocities and depths at a site.	Channel Structure and Form
Habitat Suitability Index	Operational	Topography Substrate Size Hydraulic Model: Velocity Hydraulic Model: Depth	Percent Weighted Usable Area for Juveniles and Spawners Capacity: WUA/Juvenile territory size Capacity: WUA/Redd Area	Habitat Quantity
Fuzzy Inference System (FIS)	Prototype	Topography Substrate Fish Cover Undercuts Hydraulic Model: Velocity Hydraulic Model: Depth	Percent Weighted Usable Area for Juveniles and Spawners Capacity: WUA/Juvenile territory size Capacity: WUA/Redd Area	Habitat Quantity
NREI	Operational	Drift Biomass Topography Temperature Hydraulic Model/Velocity	NREI	Food Channel Structure and Form
Growth Potential	Conceptual	Drift Biomass Temperature Metric	Growth Potential (g/day)	Food
Geomorphic Unit Tool (GUT)	Prototype	Topography	Locations and type of channel units within a site based on geomorphic definitions	Habitat Quantity Channel Structure and Form

Model	Model Status	Model Inputs	Model Outputs	Ecological Concern
River Styles	Prototype	Confinement Sinuosity Grain Size Riparian Condition	Continuous classification of stream network into River Styles classes, which are based on geomorphic characteristics, landscape context, and condition assessments	Channel Structure and Form
MODIS Temperature Model	Production	MODIS Land Surface Temperature Validation: 8 day average stream temperature	Continuous 8 day average temperatures along a stream network for each 8 day period throughout the year.	Water Quality
Gross Primary Production (GPP)	Conceptual	Temperature Conductivity Solar Input	Estimate of gross primary production along a stream network.	Food
Structural Equation Models (SEM)	Prototype	River Styles NREI Discharge LWD Conductivity Gross Primary Production	Habitat Capacity	Habitat Quantity Riparian Condition Channel Structure and Form Sediment Condition Water Quality and Quantity
Quantile Regression Functions (QRF)	Production	Site-scale CHaMP metrics, including NREI and HSI modeled metrics, and fish densities	Distributions of fish densities based on habitat metrics that can be used to infer habitat capacity.	Habitat Quantity Riparian Condition Channel Structure and Form Sediment Condition Water Quality and Quantity
GRTS	Production	Site-scale CHaMP metrics, including NREI and HSI modeled metrics	Weighted minimum, mean and maximum of input estimates for the spatial scale of interest (e.g. watershed or population)	Habitat Quantity Riparian Condition Channel Structure and Form Sediment Condition Water Quality and Quantity
Life Cycle Model	Operational	Habitat capacity and/or individual CHaMP metrics that are locally important for fish based on limiting factors or life stages	Life stage and species specific estimates of fish production (based on growth and survival metrics)	

APPENDIX A4: REGRESSIONS FOR CHAMP-PIBO METRIC CROSS-WALK

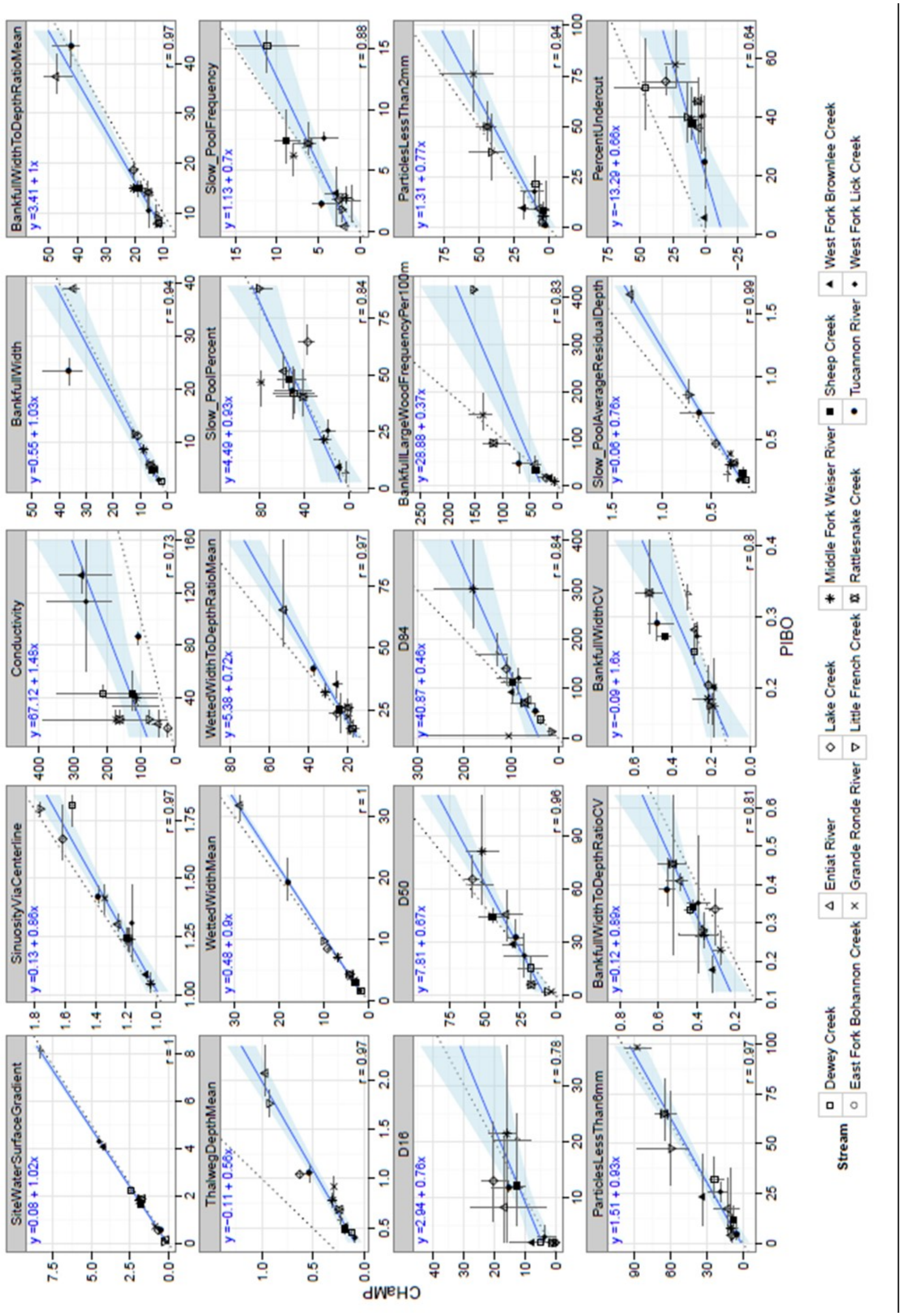


Figure A4.1. Regressions performed with a set of metrics collected by CHaMP and PIBO to determine the best procedure for creating crosswalks between metrics.

APPENDIX B1: RESPONSE TO ISRP AND NORTHWEST POWER AND CONSERVATION COUNCIL DOCUMENTS AND LETTERS

As well as the questions from the ISRP/ISAB that we address in detail in Chapter 8, here we address questions that have been posed by the ISRP as part of a programmatic review of ISEMP and CHaMP (ISRP document 2013-02) and from the Council's June 17, 2013 letter to BPA. Their comments indicate that the current effort by both programs is scientifically sound and a much needed part of the overall monitoring and evaluation needs of the Council's Fish and Wildlife Program. Additionally, the ISRP had the following recognitions and recommendations for ISEMP and CHaMP (in blue), to which we have responded below:

"ISEMP has become one of the most important monitoring programs in the Columbia River Basin. Because it employs a variety of novel techniques, it is essential that ISEMP collaborate with other large-scale monitoring efforts to maximize data sharing and opportunities for learning."

ISEMP continues to work with collaborators in the Columbia River Basin to share data and enhance learning. For example, our development of LGR escapement estimates requires interagency collaboration and cooperation in both the Snake River and Upper Columbia basins. In the Snake River, we rely on Idaho Department of Fish and Game (IDFG) to provide genetic analysis for all fish caught in the LGR trap to determine which PIT tags should be considered valid natural origin fish. We also coordinate with collaborators to ensure reporting on the same schedule of PIT tag observations to PTAGIS to improve the consistency with which we can provide these estimates. In the Upper Columbia, the Washington Department of Fish and Wildlife (WDFW) has implemented an ISEMP product to support annual escapement estimates of steelhead into the Wenatchee, Entiat and Methow Rivers using the LGR approach. We also collaborate with WDFW to collect spring Chinook data in the Little Wenatchee River that was identified as useful to both ISEMP and WDFW's development of life cycle models. In the John Day the Oregon Department of Fish and Wildlife's (ODFW) Northeast-Central Oregon Research and Monitoring Program relies upon adult steelhead capture data from ISEMP's Bridge Creek adult weir to assess the status of ESA-Listed Middle Columbia River summer steelhead. The Bridge Creek weir provides the only point of capture for adult steelhead in the Lower Mainstem John Day River population. Data from this weir are used to estimate the proportion of hatchery origin adult steelhead (PHoS) spawning naturally in the Lower Mainstem John Day River population. Estimating PHoS is a key component of monitoring the recovery of Middle-Columbia steelhead, and ODFW would not be able to successfully accomplish this in the Lower Mainstem John Day without the Bridge Creek weir. In addition to monitoring PHoS, the weir also provides annual escapement estimates which can be used to evaluate inter-annual trends observed in spawning ground surveys of summer steelhead throughout the John Day River basin. We are also continuing to collaborate with regional partners (e.g., USFS' PIBO and BPA's AEM) to identify potential data collection and application efficiencies, which is described in detail in the Executive Summary.

"To facilitate coordination and collaboration ISEMP, along with other major monitoring organizations, should promote annual meetings to exchange results and lessons learned".

We have hosted annual meetings to share results and lessons learned from ISEMP's monitoring efforts on an almost annual basis in various locations to make them available to as many collaborators as possible. For example:

- A data analysis meeting with collaborators in Wenatchee WA, November 2008
- An ISEMP annual meeting in Boise, ID, 2013
- A fish monitoring lessons learned meeting in Portland OR, March 2012
- Most recently at the ISEMP and CHaMP Analysis and Synthesis Workshop February 2013 in Portland, OR. During this two day meeting ISEMP/CHaMP staff gathered feedback from managers and policy decision-makers, monitoring practitioners and collaborators on products being developed for managers and habitat restoration practitioners.

Additionally, both ISEMP and CHaMP have maintained a strong presence in regional and national professional meetings, such as presentations at the Upper Columbia Science Conference in 2013, presentations at the PNAMP IMW conference in March 2013 in Portland, OR, and presentations at chapter, regional and national American Fisheries Society (AFS) conferences annually. In fact, ISEMP is hosting a full-day symposium at the 2015 national AFS meeting in Portland, August 2015, titled "Recent Advances in Establishing Fish-Habitat Relationships in Lotic Systems", and chaired by ISEMP's Nick Bouwes and Keith van den Broek. In addition to presentations on the work ISEMP is doing, contributed papers from other researchers have been accepted into the symposium so that a broad range of work will be presented to a wide audience.

"The ISRP should continue to review ISEMP progress reports as they become available".

We are committed to making ISEMP's annual progress reports available to as wide an audience as possible, including the ISRP. Progress reports are available on BPA's website http://www.cbfish.org/, as well as on ISEMP's website http://isemp.org/.

"The ISRP continues to support Intensively Monitored Watersheds as venues for establishing relationships between habitat restora-

tion and fish populations. New watersheds to be designated as IMWs should meet strict criteria for experimental design, including well-situated treatment and control sites, statistically sound sampling regimes, careful selection of response metrics, and commitment to long-term evaluation".

We are pleased that the ISRP continues to support IMWs as we are excited about the results that emerging from the ISEMP IMWs and have presented updated results from all three IMWs in Chapter 1.

"CHaMP should continue its efforts to consolidate and streamline habitat measurements, as well as eliminate metrics that do not provide useful information. Excellent progress has been made, and additional work will result in a set of protocols that can be employed in a wide variety of locations".

Over the course of 2014 CHaMP continued to put considerable effort into consolidating and streamlining habitat measurements and workflow, and is now at a point where the CHaMP protocol could be implemented in a wide variety of locations and produce a standardized, repeatable and information-rich dataset. Details on advancements in this area are available in Chapter 2.

"We recommend that CHaMP be open to inclusion of metrics that go beyond the characterization of physical habitat, such as additional measures of food webs and the condition of watersheds outside the boundaries of streams and their immediate riparian areas".

CHaMP continues to apply its three inclusion rules for metrics and indicators:

1) Information Content: Habitat metrics and indicators must provide information directly related to salmonid productivity, including survival and growth, as documented by peer reviewed literature, modeling, or existing data analysis.

2) Data Form: Habitat metrics and indicators must provide statistical information with robust data quality. The data generated for a prospective metric must be repeatable, detect heterogeneity, and have adequate properties for modeling/statistics (e.g., variance distributions must meet statistical assumptions for modeling or testing).

3) Feasibility: Habitat metrics and indicators need to be generated by field tools or software that are readily implementable as of the time field testing in fall 2010 (i.e., does not rely on future technological advances). Feasibility is also bounded by the need to fit all survey work within a three-person-day field survey at 80-90 percent of all sites likely to be encountered.

While we do not argue that other stream habitat metrics and indicators such as chemical contaminants can affect fish population response, we do not believe that water quality metrics beyond the water chemistry parameters included in the CHaMP protocol play as significant a role in the mechanistic fish-habitat relationship models and predictions that we are developing. For example, CHaMP resumed drift macroinvertebrate sampling in 2014 because we found this metric to be more capable for supporting ISEMP models than benthic macroinvertebrate data, which are often used as an indicator of water quality (CHaMP 2015), and that the value of the drift metric increased when it was used in multivariate products.

"The ISRP suggests that CHaMP look for opportunities to improve collaboration with other habitat monitoring efforts to improve sampling efficiencies and promote coordination with organizations having similar interests (e.g., PACFISH/INFISH Biological Opinion Effectiveness Monitoring Program [PIBO] and the Aquatic and Riparian Effectiveness Monitoring Plan [AREMP]; water quality monitoring programs)".

In 2014 CHaMP and PIBO continued collaboration to develop an interoperable dataset – that is, a set of metric data that are monitoring program independent. Initially, we had identified that it would be possible to crosswalk 24 metrics, with an additional 26 identified that could be transformed with a little more effort. To generate this dataset, some metrics needed to be transformed (12, linear transforms only), some needed to be constructed from measurements (2), while others (10) mapped directly from one program to the other (see CHaMP 2015). In the fall of 2014, CHaMP developed a demonstration project to show ability to adjust/transform three univariate metrics (temperature, pool frequency, and large wood frequency) that have known mathematical relationships (cross-walks) between the two programs. Geographically, the scope of this effort was limited to three species and 5 ESUs; Snake River Spring-Summer Chinook, Upper Columbia Spring-Summer Chinook, Mid-Columbia Steelhead, Snake River Steelhead, and Upper Columbia Steelhead. BioAnalysts, Inc. provided metric threshold determinations that Sitka Technology Group used with the shared CHaMP-PIBO metrics to create an interactive map application and color-coded displays. These displays were based on user-defined categorizations of quality; "rollup" areas were color coded based on simple characterizations of site-level surveys to estimate condition at successively larger scales, all the way up to the ESU and basin scale.

The CHaMP-PIBO data integration effort was an important first step in generating a regional approach to the management, distribution and reduction of stream habitat monitoring data. There is no reason to expect that the CHaMP-PIBO experience is unique; crosswalks between other metric sets could be developed just as easily and also housed in the integrated data management system. This does not go all the way to the development of a data exchange template (MMX) for regional stream habitat data, but the crosswalk algorithms are a necessary component of an exchange format for relevant metrics and necessary for determining the extent to which the integration is possible. PIBO and CHaMP are moving beyond the MMX template idea to try cross-program analyses where each program's data is incorporated by the other program to increase coverage and sample size. To date, these analyses are not mature enough to report on, but the ability to support regional decision making with data from multiple regional monitoring programs is being developed.

CHaMP successfully intensified its coordination efforts with the regional AEM program in 2014 to ensure standardization between shared sites, metrics, and protocol elements, and to maintain the integrity of the CHaMP survey design while accommodating the addition of new AEM sites if requested. CHaMP training in 2014 was set up to accommodate an AEM-specific module and discussion, and crews from both programs benefitted from a combined CHaMP-AEM data collection application and new tablet platform as a result of collaboration between these two programs. Efficiencies were also realized through use of a common data management and QA/QC environment and tools.

"The ISRP finds that CHaMP's pilot phase has shown sufficient progress that potential expansions of the suite of sites visited is justified, but with caution as sampling protocols continue to be refined and funding for field crews grows".

The Council's recommendation for CHaMP continuation was conditional upon CHaMP remaining in a pilot phase until there is stability in the data collection protocols and the evaluation analysis has been developed, and has undergone further ISRP and Council review. In the 2013 CHaMP Lessons Learned report (CHaMP 2015) we showed how stability in CHaMP's data collection has been achieved and the advancements that have been made in evaluation analysis and concluded that CHaMP was ready to move beyond the pilot phase. In this report we detail further progress that has been made, especially in the area of evaluation analysis (Chapter 4). The Council also asked in its June 17, 2013 letter to BPA if CHaMP had been implemented through "*an incremental approach, consistent with the ISRP's review conclusions (i.e., pilot effort)*"; we believe that the information laid out in the CHaMP 2013 Lessons Learned report shows how CHaMP did follow the ISRP's recommendations and how it is ready to move beyond the pilot phase.

"As with ISEMP, the ISRP would like the opportunity to review CHaMP progress reports as they become available".

We are committed to making CHaMP's annual progress reports available to as wide an audience as possible, including the ISRP. Progress reports are available on BPA's website http://www.cbfish.org/, as well as on CHaMP's website https://www.champmonitoring.org/.

In its letter to BPA dated June 17, 2013, the Council supported the continued implementation of ISEMP and CHaMP and made several requests of both programs:

"…*explain how these tributary habitat monitoring and evaluation activities link to and integrate into the monitoring, evaluation, reporting and data management effort for the entire program, including for the tributaries (ISEMP, CHaMP and AEM), the estuary (CEERP), artificial production (such as the CHREET proposal); Bonneville's data management framework, the Coordinated Assessment (CA) data sharing effort, and other large scale aquatic monitoring programs occurring within the Basin that are funded by other agencies such as PIBO and AREMP.*"

The tributary based monitoring efforts underway in ISEMP and CHaMP are not integrated with the CEERP program in the estuary, or the artificial production assessment programs such as CHREET due to these programs having fundamentally different geographies and objectives. However, ISEMP and CHAMP are working with other large-scale aquatic monitoring programs within the tributary environment that are funded by other agencies, mostly AREMP and PIBO. ISEMP and CHaMP have run methodological comparisons of response designs between a number of stream monitoring programs (2008, 2011), have worked with PNAMP on identifying the potential for cross-walk/data sharing on a metric by metric basis, and most recently, are coordinating with PIBO to integrate CHaMP and PIBO metrics when generating watershed-scale estimates of habitat condition. In addition, CHaMP is collaborating with PNAMP to further the discussion of data exchange formats for habitat metrics, an effort analogous to the work by the Coordinated Assessment to facilitate sharing of fish population data.

"…*the submission and the review in 2015 should be used for a comprehensive consideration of whether and how to transition CHaMP out of the pilot phase..*"

2013 was an important milestone for CHaMP in that it marked the completion of the third year of CHaMP's initial 3-year sampling panel and, arguably, conclusion of the "pilot" project phase. The implementation of CHaMP was initiated under a "pilot" designation following discussions with the ISAB/ISRP and their concerns regarding the development of a new region-scale habitat monitoring program based on response and survey designs that were considered not fully established. Thus, the initial footprint (Wenatchee, Entiat, Methow, John Day, Tucannon, Lemhi, Upper Grande Ronde, South Fork Salmon) was to be considered a trial run for the program before any additional watersheds were brought into the sampling design which was designed to represent the tributary habitat data needs of the BiOp RPA and the AMIP life cycle modeling task as suggested by the BiOP RME Working Group. The "pilot" implementation of CHaMP was assumed to be less than the full set of sites necessary, so that after sufficient confidence in the methodology was generated, a more complete sampling of the interior Columbia River basin would be undertaken. While the cautious approach to the launch of the program certainly was warranted – an enormous investment was being considered based on the development work done within ISEMP – the terms of the "pilot" designation were not specified, nor were the evaluation criteria to indicate moving the projects designation to post-pilot, or production. In order to facilitate the review of CHaMP's status we developed a set of evaluation criteria in the form of a series of questions that we feel adequately demonstrate that the CHaMP pilot has met its objectives. These evaluation criteria were presented in detail in the 2013 CHaMP Lessons Learned report (CHaMP 2015).

Given three complete monitoring evaluation cycles and the extensive QA/QC processes implemented by the CHaMP team on all aspects of the project (protocol, training, field data collection gear, data capture, data cleaning, data stream, data management, analysis methods), we feel that the project has matured to a sufficient degree that it meets the technical expectations of a robust, dependable stream habitat monitoring method. Furthermore, we feel that CHaMP implementation groups, collaborators, and the ISEMP analysis efforts have demonstrated the utility of the CHaMP data to resolve critical uncertainties for both tributary habitat and salmon population management efforts. As such, we consider the pilot implementation phase of CHaMP to be complete, that the robustness and utility of the method has been adequately demonstrated, and that the Environment, Fish and Wildlife Program can confidently implement CHaMP to address key management question as called for in the FCRPS Biological Opinion and other programmatic directives.

"…the submission and the review in 2015 should be used …..to confirm or alter the timeline for completion and end of the Program funded IMW studies and the evolution of the rest of the ISEMP project."

The ISEMP IMW studies in Bridge Creek in the John Day, Entiat River in the Upper Columbia, and the Lemhi River in Idaho are each at different points in their implementation. In the Bridge Creek IMW, restoration actions (installing structures to encourage beavers to build dams) were first implemented in 2009, with the next round scheduled for 2015. Fish response monitoring is planned through 2018, although benefits would likely still be accruing for decades.

In the Entiat River IMW the first round of restoration actions (increasing instream complexity and floodplain connectivity through an engineered approach) were implemented in 2012, followed by another round in 2014. The next round of actions are scheduled for implementation in 2016 and 2017, and the final round of actions would be implemented in 2020, at which time all feasible instream habitat actions would have been completed and the lower 26 miles of the mainstem Entiat River (below the U.S. Forest Service boundary) could be considered restored to the extent possible given societal and budget constraints. Fish response monitoring should continue through 2023, although benefits would likely still be accruing for decades.

In the Lemhi IMW the first round of habitat restoration actions (reconnection of de-watered tributaries) occurred in 2005, and ISEMP began intensive effectiveness monitoring in 2009. The next round of reconnections are being planned now. Fish response monitoring should continue through 2020, although benefits would likely still be accruing for decades. For all three IMWs, the lessons learned about the long-term response of salmonid populations to habitat restoration actions would be invaluable.

We are eager to discuss with BPA, the Council and ISRP how ISEMP should evolve in the future, and to help with this discussion we have laid out a task timeline that identifies work that needs to be completed through 2018. From 2015 – 2018 we believe that the guiding principle should be to work with validated monitoring and analysis methods that are scientifically defensible to show their application to answering management questions in a meaningful way. Regional management and planning timelines and analytical product requirements are the drivers for ISEMP and CHaMP near- and long-term work planning. The primary objectives of the 2015 - 2018 work planning cycle are the 2016 Expert Panel process, the 2017/2018 FCRPS BA/BiOp, and the 2008 FCRPS BiOp AMIP process.

Through 2015 ISEMP and CHaMP have been focused on capacity building such that data streams are consumed by robust analyses. The next critical phase of the work is to completely assess summary and synthesis products such that we can have confidence in the decision support capacity of these tools. We need to properly vet all the tools used in decision support so that the management community is motivated to move from current practices. This step requires managing adoption inertia, as well as managing the risk of introducing many new tools.

While CHaMP and ISEMP are primarily focused on developing tools and analyses related to status and trends and effectiveness monitoring, a critical component of ISEMP's work is the development of summary products codifying the relationships between tributary habitat actions and fish survival or productivity, and identifying which actions are most cost-effective at addressing habitat impairments. It is these products – maps of habitat capacity, maps of stream restoration potential, relative rankings of restoration scenarios – that need to be validated with the management community and developed to a production level, thus closing the loop between RME and on-the-ground actions. The application of the ISEMP-CHaMP summary products to the tributary habitat work-flow is expected to increase the efficiency and efficacy of the stream habitat restoration program and build confidence in the RME program's value.

Key work areas that will be developed fully in annual work plans for 2016 – 2018 are:

- IMW completion timeline, with suggested target end dates
    - Lemhi River IMW 2018 - 2020
    - Entiat River IMW 2020 - 2023
    - Bridge Creek IMW 2019 - 2021
- Habitat monitoring work products
    - Columbia River basin-wide assessments of tributary habitat quality/quantity across populations/ESUs
    - Application of watershed-scale assessments to decision-making processes
    - Outreach/integration with local and regional groups
- Fish monitoring work products
    - Capacity estimates across populations/ESUs of the Columbia River basin
    - Freshwater survival estimates as a function of tributary habitat quality and quantity across populations/ESUs of the Columbia River basin
- Tributary habitat management work products

- Life cycle models based on tributary habitat quality/ quantity
- Restoration potential summary products across populations/ESUs of the Columbia River basin
- Application of watershed-scale decision-support tools to decision-making processes
- Outreach/integration with local/regional groups
- Geospatial data management structure
- Data management supporting watershed and network product development and display.

The Council's letter (June 17, 2013) also stated that CHaMP's overarching goal should be to develop and implement is "*a cost-effective, standardized, independent, statistically valid approach for evaluating habitat effectiveness*". The 2013 CHaMP Lessons Learned report (CHaMP 2015) laid out in detail how CHaMP has achieved this. In this 2014 report we provide updates to the work that was described in the 2013 report and further show how CHaMP has achieved the goal set for it by the Council.

The Council also requested a clear statement of "*how NOAA and Bonneville, working with other relevant participants, further developed the analytical, evaluation and reporting elements of the habitat effectiveness monitoring and evaluation effort to accompany the CHaMP monitoring, consistent with the ISRP's review conclusions*". The ongoing development of analytical, evaluation and reporting elements was described in detail in the 2012 ISEMP Annual Report (ISEMP 2013), which reported on tools and products shared with collaborators and other interested parties at the ISEMP and CHaMP workshop in Portland in February 2013. There we presented maps and graphics that display summarized metrics and data at various spatial scales to address different needs, and described the development of a translation of ISEMP and CHaMP data and metrics into a format that is directly useful to addressing management questions, is salmon-centric and bio-physically informed, and is built upon direct linkages, consistencies and efficiencies.

We are continuing to develop map products that can be used to track progress towards goals and support adaptive management over time. Using data collected under CHaMP we are creating geomorphic and habitat condition maps to aid in the assessment of habitat status and trend that can be synthesized into restoration priority maps for specific populations. We are also creating population condition maps based on the geomorphic and habitat condition information with fish data collected under ISEMP, and we continue to develop recovery potential maps of different areas in a watershed, population or ESU that use both ISEMP and CHaMP data.

APPENDIX B2: LETTERS OF SUPPORT

Subject: Letter of support for the Columbia Habitat Monitoring Program (CHaMP) regarding its usefulness to the Columbia River Inter-Tribal Fish Commission's habitat monitoring efforts

Date: June 11, 2015

From: Dale A. McCullough, Ph.D., Senior Fishery Scientist
Seth White, Ph.D., Fishery Scientist
Casey Justice, M.S., Fishery Scientist
Monica Blanchard, M.S., Fishery Biologist

CRITFC has been a participant and active collaborator in the development of CHaMP from the outset of the program (i.e., 2011), although we receive support to conduct CHaMP and related monitoring work through Columbia River Accords funding to the tribes. CRITFC sees significant value in monitoring habitat conditions at a basin scale (i.e., focusing on in-channel ecological characteristics, as well as riparian and watershed conditions) as a means to assess whether salmon habitat conditions and the overall environmental framework under which in-channel conditions respond are improving. In addition, we are assessing whether any net improvements that do occur would translate into increased spring Chinook survival and abundance at the scale of the listed populations. CRITFC has been developing, by use of CHaMP monitoring procedures, a database with quantitative annual and tri-annual (i.e., rotating panel set of monitoring sites measured every 3 years) habitat conditions that can be used to establish a statistically reliable baseline condition and trends that can be distinguished from natural background variation. CRITFC uses CHaMP techniques to perform both highly precise topographic mapping of stream channels in study sites that can be used to identify changes as well as refined estimates of key fish habitat characteristics at channel unit and reach levels. The habitat characteristics measured will be used to identify progress at a basin scale in improving spring Chinook habitat conditions. We believe that this approach is an essential one, especially in watersheds that have a complex spatial distribution of past restoration activities with varying times since action implementation, making it infeasible to find appropriate site-specific control sites for targeted restoration sites.

In the course of working with other CHaMP participants, CRITFC soon learned that there were numerous advantages to working with the other CHaMP program participants. Some of these advantages can be detailed below:

1. CHaMP monitoring, data generation, analysis, and application provide a regionally consistent habitat monitoring protocol that guides the collection of raw habitat data; convenes a community of scientists developing aspects of habitat monitoring methodologies; and provides a more quantitative, meaningful alternative to use of expert panels charged with assessing progress in improving Chinook and steelhead habitat.

2. CHaMP provides us the information that we can use to drive a life cycle model (LCM). This model would be the best alternative to an expert panel process to simply reach consensus on the quality of habitat and trends in this habitat quality.

3. CHaMP provides logistical support of habitat data and its collection via maintaining an organized database that can be accessed by all CHaMP participants, providing basic statistical summaries of habitat metrics that can be used in conducting other analyses, providing tools to assist in the

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QA/QC process, and maintaining performance of field equipment by recalling gear for recalibration and repair.

4. A unique component of CHaMP is the application of cutting edge tools for habitat monitoring based on topographic surveys and remotely-sensed information. These tools include GIS macros for processing topographic survey data, Total Station software for data collection, River Bathymetry Toolkit (RBT) analytical capabilities so we can use our own field surveys to model water surface area at different flows, Net Rate of Energy Intake (NREI) models that can be used to estimate carrying capacity linked to potential food availability and hydraulic conditions of depth and velocity, a hydraulic model for calculating habitat suitability indices and calculating weighted usable area along with other outputs, and use of MODIS satellite data to predict distributed temperature metrics on a stream network level. These represent high-level products that our organization could not have produced without significant additional levels of resources.

In summary, we have found our collaboration with CHaMP to be highly beneficial to our habitat monitoring program in the upper Grande Ronde basin—benefits which would likely extend across the region to other entities choosing to adopt CHaMP methodologies. This fruitful collaboration has freed up time and other resources and allowed us to expand our research program beyond the scope of evaluating habitat status and trends to include a more refined understanding of land management effects on fish habitat conditions and populations. We hope CHaMP is also supported by regional leadership based on the value-added products and services provided by the program.

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Gene Shippentower
46411 Ti'Mine Way
Pendleton, OR 97801
541.429.7287

June 12, 2015

RE: CTUIR support of CHaMP monitoring protocols

Dear ISAB/ISRP Review Committee,

CTUIR has implemented fish habitat restoration in five sub-basins; the Grande Ronde, John Day, Tucannon, Umatilla, and Walla Walla Rivers. As a result, we developed and implemented a plan to assess the effectiveness of CTUIR habitat improvements restoration with respect (*Oncorhynchus tshawytscha*), summer steelhead (*Oncorhynchus mykiss)* and bull trout (*Salvelinus confluentus*) populations in each of the five sub-basins. These three species were chosen as barometers of ecosystem function and health for a multitude of other species. The detection of measurable changes in biotic conditions, specifically changes to growth, survival and abundance of various salmon life stages, is inherently linked to habitat. Such changes in physical habitat have been tracked by CTUIR staff using CHaMP protocols for our Biomonitoring project (#2009-014-00) in sites located in five sub-basins. The benefits of collecting CHaMP habitat data in conjunction with the biological data generated in this study are needed in order to gain the greatest understanding of mechanistic relationships of restoration actions. For example, CTUIR is planning to use applications such as the habitat suitability index model, geomorphic change detection model, hydraulic model, bio-energetics model, and temperature models can make available inference on how well CTUIR restoration actions are relative to; reducing the effects of limiting factors, determining fish/habitat relationships, and what habitat actions are increasing fish survival and productivity.

CHaMP monitoring protocols are preferred because the data are consistent within and among our biomonitoring sites and consistent with our co-managers in the basins we work. Other benefits of implementing CHaMP for CTUIR are:

- Monitoring covering a large part of the ceded territory with a standardized format.
- Provides a baseline of detailed information on current status and trend of key first foods species.
- Knowledge of habitat conditions is based on field data and does not rely on expert opinion.
- Data collected is an intensive effort at a scale not previously undertaken.
- Data provides a level of information on first food species habitat that has not been available to the Tribe since the start of European settlement.
- CHaMP is a collaborative effort that builds on partnerships between Tribal, Federal, State, and private landowners. The scope of the data collected could not have been achieved by any one agency/department alone in the same time as this partnership.
- These partnerships are developing to increase data sharing and the scale of restoration efforts undertaken in the ceded territory.

Treaty June 9, 1855 ~ Cayuse, Umatilla and Walla Walla Tribes

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- GIS tools being developed by the CHaMP team using the data collected will provide information to habitat restoration managers on:
  - Geomorphic change detection.
  - Integration of LIDAR data with stream survey data.
  - Hydraulic modelling.
  - Fish Habitat modelling.
  - Habitat suitability indexes.
  - Temperature modelling.
  - Statistical analysis.
  - Mapping river bathymetry.
  - Net Rate of Energy Intake.
- The information collected by CHaMP can be compared to other biological monitoring efforts to evaluate the effectiveness of habitat restoration efforts.
  - This information can then be used to guide restoration managers on future geographic locations (focus areas), and on what techniques are successful in achieving the goals of the River Vision.

In addition to the list of benefits from the CHaMP program, CHaMP is unique and stands apart from many of the programs we have worked through over the years. The CHaMP group is a great example of researchers cooperating effectively with tribes, state and federal agencies, and private companies to develop and test cost effective methods to answer specific questions associated with the general uncertainties of fish and habitat relationships. The need to address these uncertainties has been longstanding and was clearly recognized in the early 1980s (and before). The need for answers will not diminish if the questions are not addressed. Without CHaMP, progress had been slow. Quantifiable regional integration of research findings was impossible because methods were not consistent. Without CHaMP, individual research groups were inefficient because they independently evaluated the same questions with small sample sizes and with inconsistent methods and approaches. Without CHaMP, knowledge gained by individual groups from both success and failures were rarely shared so the processes were unnecessarily repetitive.

The culture of the CHaMP group is remarkably positive and open. The group actively promotes a unified approach from a wide diversity of scientists from many disciplines. It is rare to observe so many people working as effectively together as the CHaMP collaborators have. Cooperation is high, innovation is encouraged and rewarded. The development, enhancement and testing of ideas and methods is progressive and accelerated. This type of group culture is effective and successful and has been associated with significant breakthroughs and progress in scientific history. CHaMP developers have openly collaborated to embrace, adapt and enhance past efforts by universities, agencies, consultants, and tribes working independently. The development of CHaMP has not replaced those ongoing efforts but has integrated and enhanced them.

Like a vineyard, the investment in developing CHaMP was substantial. Just like a vineyard, the full return of the investment in CHaMP will mature with time. There have been significant products and benefits realized from the investment already, but much more will come. CHaMP testing will likely show that some habitat actions are not ineffective. That should be viewed as a successful outcome that will help manager's direct resources to successful actions. CHaMP testing will also show that some monitoring methods are less effective than other monitoring methods. That would also be a successful result that will lead to the abandonment of

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poor methods and development of better tools. A consistent regional approach to developing effective monitoring methods and employing those methods to assess salmonid habitat enhancement projects has been a critical need for many years.

Thank you very much for your time and consideration of our continued support for CHaMP and its continued development, evaluation and enhancement.

Sincerely,

Gene Shippentower

Treaty June 9, 1855 ~ Cayuse, Umatilla and Walla Walla Tribes