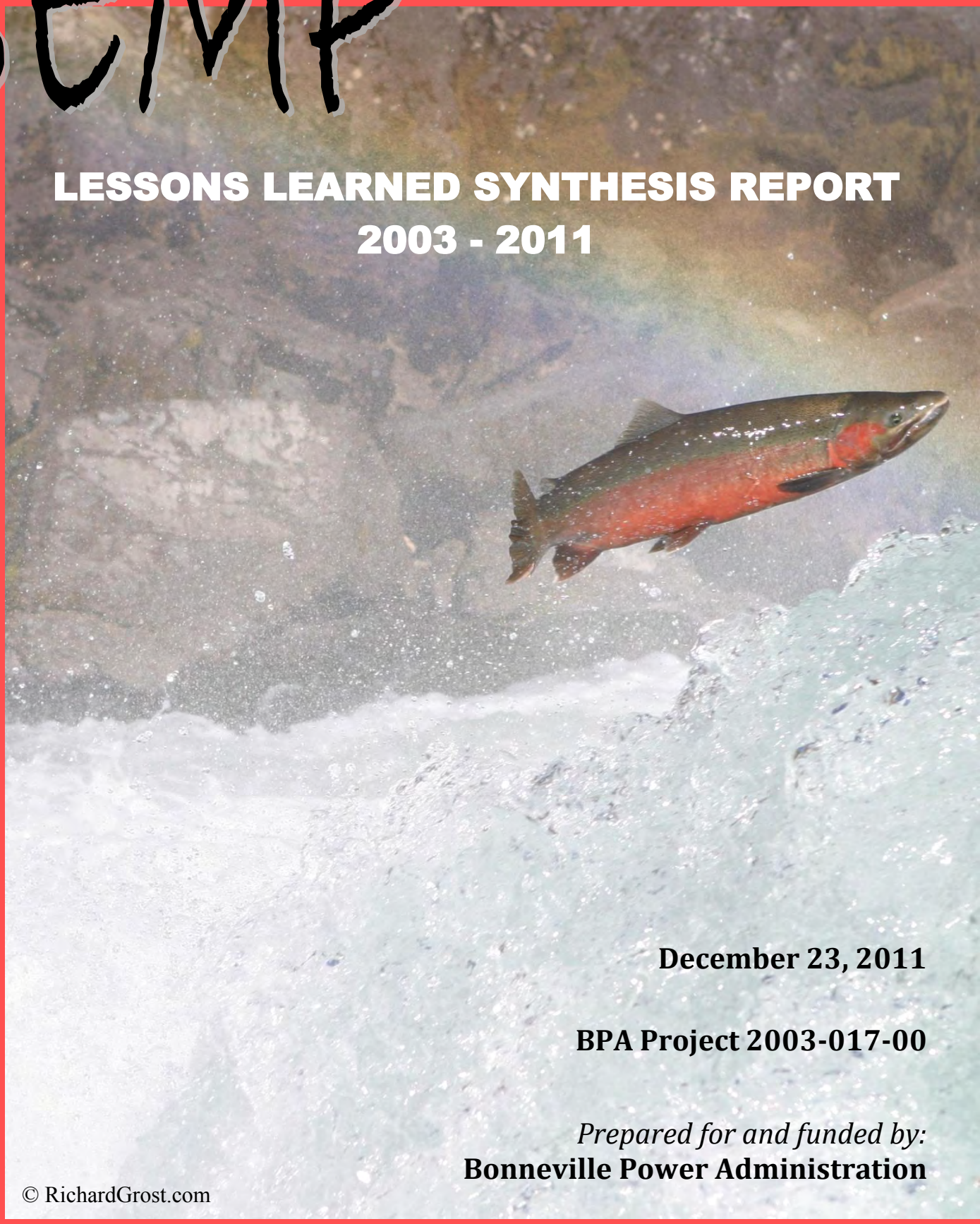
# ISEMP

INTEGRATED STATUS AND EFFECTIVENESS MONITORING PROGRAM



# LESSONS LEARNED SYNTHESIS REPORT
## 2003 - 2011

**December 23, 2011**

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# TABLE OF CONTENTS

# CHAPTER I: EXECUTIVE SUMMARY

> The ISEMP 2011 Synthesis Report tells a simple story:
>
> - A forum of policy managers and technical experts need to work together to develop an analytical framework to answer the fundamental management question. (Chapter 2)
> - ISEMP provides a suite of the technical tools that can be built into the analytical framework (Chapter 3)
> - Policy managers must assist the technical experts in building a decision making structure within which these tools can operate (Chapter 4)

Policy makers, resource managers, technical and science staff and the stakeholder community all recognize that the fundamental management question for ESA listed salmonids in the Columbia River basin is, "are habitat restoration actions effective in helping salmon populations recover?" The same group also recognizes that the community must engage collectively to answer the question. Engagement will take different forms, reflecting the differing roles of each participant, from policy to technical. To engage collectively requires a forum around which the technical, policy, implementation, and management objectives and roles can be reconciled; however, a forum alone is not sufficient to answer these questions -to bring the myriad roles and responsibilities of all elements of the fundamental management question together necessitates an analytical framework.

An analytical framework that underlies the fundamental management question of, "are habitat restoration actions effective," has both technical and policy elements. This framework must link stream habitat quality and quantity to beneficial changes in fish populations in a predictive manner and must be able to resolve the differences between suites of management actions and the resultant habitat change those same actions might affect in different watersheds. Most importantly, the framework must be founded on technical products that are informed by, and useful for, the management and policy community. Therefore, the framework itself must be a collective outcome of the collaboration envisioned by a technical-policy-management forum.

The Integrated Status and Effectiveness Monitoring Program (ISEMP) has been developing data collection, reduction, analysis, and interpretation products with the explicit goal of supporting the development of an analytical framework linking habitat quality and quantity with fish population processes. In this report we present a suite of the outputs, presented as Decision Support Products. These Decision Support Products range from simple descriptive methods for condensing and distributing volumes of field based measurements to predictive models that output the population capacity and productivity of a watershed based on habitat conditions. While each of these Decision Support Products is a technical and scientific advancement, they are not useful unless given a context in which they can be applied to the fundamental management question. The Decision Support Products form the basis of the analytical framework, yet require the collaboration of a technical-policy-management forum to be fully developed into a useful resource management tool.

Since the presentation of ISEMP's suite of Decision Support Products to a broad audience requires technical and non-technical descriptions of our work this report is divided into two major sections (main document and an appendix). The main part of the document introduces

the goal of ISEMP's work on the fundamental management question, the range of methods and technical approaches that are necessary to develop our products, and graphical presentation of the products and their outputs. The appendix is the complete technical development of the Decision Support Products, including details of data reduction, analysis and model development. In general, the policy audience will be most interested in the main document, while the technical consumer may benefit from the context provided in the main document, but may be most interested in the detailed background presented in the appendix.

# CHAPTER II: INTRODUCTION

**Fundamental Management Question:**

Are habitat restoration actions effectively helping salmonid populations recover, and, if so, which actions are the most cost-effective?

*Watch the series of boxes throughout this report. They point out the management implications of the work described on that page. When pieced together they outline the technical elements of our suggested approach to the Analytical Framework.*

The question at the heart of the contemporary debate surrounding salmonid management in the Columbia Basin is "Are habitat restoration actions effectively helping salmonid populations recover, and, if so, which actions are the most cost-effective?" All key management questions essentially boil down to refinements or nuances of this fundamental question.

Nearly 10 years ago, scientists at NOAA and elsewhere recognized the seeming intractability of the above question and proposed a program to systematically answer some of the scientific mysteries, starting with "what's the best way to measure habitat?" and "what's the best way to measure salmonid populations?" ISEMP was the result, and in this report we will focus on the achievements of ISEMP since its inception in 2003 and show how those achievements are relevant to answering this question in both the science and policy arenas, as both are inherent elements of the question.

The scientific elements of this question are about measuring habitat, fish populations, and the effects of action types, designing habitat restoration actions, and identifying and answering scientific uncertainties such as what aspects of fish habitat are important, and what about a fish population is important?

The policy element is about defining "recovery" and "effectiveness," providing a scale for "cost effective" and deciding what level of effort is enough. It also involves providing a forum for deciding, as a society, what the answers are when the technical or policy details are not clear.

Today, the fundamental management question on the role of tributary habitat management in salmonid population impact mitigation is decades old, and directed research projects have been underway for almost the same time period, but the question apparently remains intractable. As recently as December 2011 at the Columbia Habitat Monitoring Program Workshop in Portland, policy makers recognized the ongoing need for, and their responsibility to provide, a forum for deciding how to generate answers to this question and committed to moving the process forward.

In this report, we describe approaches to help this forum make progress using scientific raw materials to make decisions, frameworks for thinking about and interpreting the results, and how to funnel that information to the policy makers. We show how ISEMP has built the tools necessary to complete the analytical framework and that ISEMP is ready to work with policy managers to shape this framework to generate pragmatic answers to the key management questions.

## Developing the Technical Structure of the Analytical Framework: A Key ISEMP Task

The key technical task to address the fundamental management question and one of the major goals in ISEMP is to develop an analytical framework, based on measures of habitat quality and quantity, to predict the expected effects of various habitat restoration activities on fish population productivity and capacity.

Developing this analytical framework requires several intermediate steps. First, we need to identify what we can measure about the habitat that is correlated to fish abundance, growth and survival. Next, we have to determine what spatial and temporal scales are appropriate for linking habitat and fish populations using those habitat measures. For example, we can measure things like the density of large wood at one site, but fish move and may utilize habitat up- or downstream that has different densities of large wood. Therefore, the density of fish at a particular site should be linked to the density of large wood at a spatial scale larger than a site. Performing this linkage involves determining how to best measure fish and habitat characteristics that show their connections and to separate changes in fish populations due to restoration actions from those due to ongoing trends in their populations or their environment. With this understanding, we have built mechanistic models that describe those links between habitat and fish mathematically, and on the appropriate spatial and temporal scales. Experimental manipulations are then used to test the change in fish productivity caused by restoration actions that is predicted by these models.

The fish-habitat models can also be used as decision support tools to identify limiting factors for fish populations. This allow us to appropriately target restoration actions, to quantitatively compare the cost and expected biological benefit of various action plans, and to assess the actual fish productivity changes due to the habitat restoration actions that were taken. To build the analytical framework described above requires developing and testing variations of these decision support tools to determine which are most appropriate for the information that has been collected and to guide collection of critical, but missing information about fish and habitat conditions.

## Three Paths to Fish-Habitat Relationships

While the primary objective of ISEMP's work is to develop an analytical framework that relates habitat quality and quantity in a spatially explicit manner to fish population response in the tributary environment, there is no single best way to build this critically important decision support tool. In fact, ISEMP's work to connect stream habitat condition to fish population response takes three distinct paths, each with its own strengths and weaknesses, but each key to developing an overall framework.

The three basic methods ISEMP employs are descriptive, empirical methods, mechanistic models and experimental manipulations (Figure 1).

**Conceptual Model for the Analytical Framework**

- *Descriptive empirical methods*
- *Mechanistic models*
- *Experimental manipulations*

Figure 1 illustrates all of the technical elements that ISEMP recommends are built into the analytical framework.

Descriptive empirical methods are based on correlation and regression models and are the most common tool for data exploration. These methods are fundamentally descriptive and exploratory in that no preconceived relationship between predictor (habitat) and response (fish) metrics is required. The modeling process is used to reveal situations where habitat and fish metrics co-vary in a consistent fashion, can be used to generate hypotheses, but the methods alone can never "prove" that habitat conditions cause changes in fish population processes.

Mechanistic, predictive models are based on an assumption that everything is known about the connection between input (habitat conditions) and output (fish population) such that the input information is more like a scenario and the output is the predicted result of that scenario playing out in the real world. Mechanistic models are standard tools for generating and testing hypotheses and often underlie large-scale management actions as the basis for predictions in adaptive management schemes. While predictive models have incredible appeal – they can see the future – their limitations are easy to understand as their output can be no better than the sum of the knowledge incorporated in the rule-sets that relate input to output.

Formal, experimental manipulations are the classic tool of the scientific method for determining cause and effect. In this case, manipulations of habitat conditions are evaluated for fish population responses relative to un-manipulated areas; hence a strong, clear picture of the effect a particular habitat change has on fish populations is developed. With proper experimental design, fish-habitat relationships developed through experimental manipulations can be applied to domains not directly involved in the original experiment, within reason. We know that fish and stream habitat have regional patterns (coast vs. interior, mountain vs. plateau), so it is reasonable to expect that fish-habitat relationships also vary regionally. Thus, relationships developed through rigorous experimental manipulations can be applied regionally, but should be extended to other regions with caution.

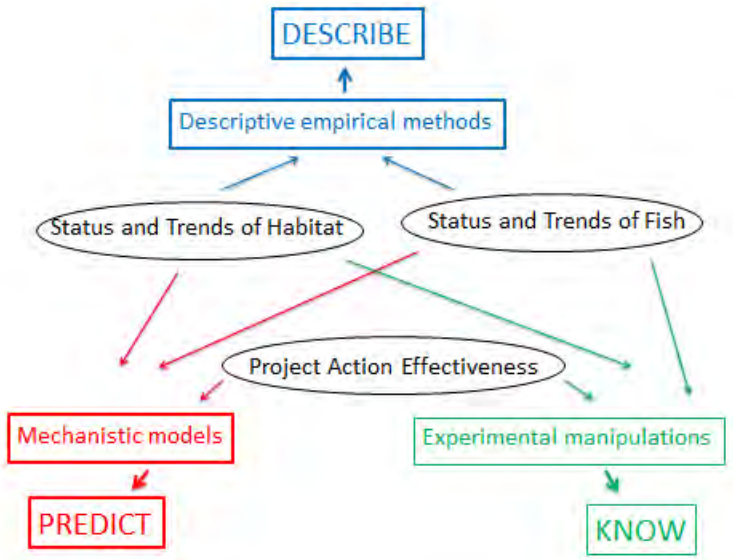



Figure 1. The three methods used by ISEMP to connect fish and habitat data that are the basis of the analytical framework proposed to managers for use as a decision-making tool to answer management questions.

Given the limitations of descriptive and predictive methods the ideal approach seems to be experimental manipulation, since the outcome is unambiguous knowledge that can be directly applied in support of management decision making. Unfortunately, there are practical limitations on experimental manipulations relevant to fish-habitat relationships – experiments can be expensive and can only be performed in rare cases where technical and social conditions in a watershed are suitable. In reality, all three methods are needed to get to an answer. Regression approaches give us the clearest insight, unfiltered by preconceptions, into what are the most meaningful metrics/indicators to measure and generate hypotheses. Mechanistic models tell us why those relationships exist, and tell us what we need to change/restore to achieve a desired result, and give us a framework for testing the hypotheses uncovered through empirical methods. Both descriptive and predictive approaches can be combined within a decision support tool, which if correctly applied should guide the implementation of experimental manipulations (identify limiting factors, assess alternative strategies for restoration, and apply those strategies within an analytical design that supports quantitative effectiveness statements). ISEMP's strategy is to hybridize the methods to most efficiently arrive at fish-habitat relationships that are the basis for a robust, predictive decision support tool to guide the implementation and evaluation of a tributary habitat management strategy for the FCRPS Biological Opinion (BiOp) and the EFW Program.

## Relevance, Extent, Timing, Contrast - The Cornerstones of Effective Habitat Monitoring

Four features – relevance, extent, timing, contrast – are the core of ISEMP's work on the BiOp Habitat Strategy. Considering these features addresses the question of how can the right actions be implemented in sufficient quantity and arrangement such that their impact can be detected in populations of listed salmonids in the Columbia River basin?

The BiOp tributary habitat strategy depends on the ability to identify stream habitat impairments and to restore/conserve these habitats in a manner that benefits fish populations. Habitat restoration programs can't demonstrate the benefit of their work because they are not doing the right thing and cannot detect the effect of doing the right thing. Habitat management actions that don't address an ecological limiting factor may benefit overall stream condition, but will not be reflected in fish abundance or productivity. However, even if habitat features that limit fish populations are identifiable and can be addressed with restoration or conservation actions (relevance), detecting the expected change in the fish can be extremely challenging, primarily due to issues of magnitude – the magnitude of the actions' footprint relative to that of the fish population (extent), the magnitude of the actions' realized benefit relative to when it is expected (timing), and the magnitude of the actions' impact that is separable from coincidental, confounding factors (contrast).

**Habitat Action Improvements**
The fundamental management question would be easier to answer if habitat actions were implemented with four features in mind. Actions need to:

- Address a limiting factor.
- Be large enough in magnitude to have an appreciable. affect.
- Happen in a small window of time.
- Be distinguishable from other confounding factors.

# CHAPTER III: ISEMP DECISION SUPPORT PRODUCTS

ISEMP monitoring directly supports identifying fish and habitat status; identifies, detects and tracks habitat impairments across the landscape; links habitat condition to fish populations; and is changing habitat conditions to change fish population status. All of these activities directly support the development of an analytic framework that managers can use to help make decisions. In this section we provide examples of how ISEMP's work can be used to support and build decision-making tools.

**Outputs for Managers**

- Data collection and reduction products
  - Run decomposition/escapement estimates
  - ISEMP Juvenile Monitoring Protocol
  - Habitat quality and quantity metrics
  - Variance decomposition
  - Habitat Impairment Predictions
- Synthetic products – synthesizing habitat to predict fish
  - Intrinsic potential maps
  - Habitat change detection
  - Watershed production models
  - Fish—habitat relationships
  - Fish—habitat relationships: abundance
  - Fish—habitat relationships: location
  - Fish—habitat relationships: growth
  - Growth potential model
  - Carrying capacity prediction model

## Run Decomposition: Estimating Escapement Using Instream PIT Tag Detection Arrays

Recent advances in instream PIT array construction and analytical techniques now allow scientists to place arrays in streams to estimate the number of upstream migrating fish, something that historically had been difficult to do. These advances meet BiOp requirements for "fish-in" numbers and for estimates that have "known statistical properties" (i.e., associated estimates of uncertainty).

**Run Decomposition**
Using PIT Tag detection systems provides cost-effective estimates of "fish-in" numbers at many tributaries, helping to meet multiple BiOp requirements. Originally developed by ISEMP in the Snake River, co-managers in Washington are adopting this approach in the Upper Columbia in 2012. Managers are finding run decomposition results useful in evaluating recovery planning assumptions.

Utilizing existing trapping facilities at Lower Granite Dam, ISEMP tags a known representative fraction of adult spring/summer Chinook salmon and steelhead and collects biological information (age, sex, genetics, etc.). Subsequent detection of these adults as they pass instream PIT tag arrays enables an estimate of total escapement above that point with accompanying estimates of uncertainty for the purpose of decomposing the run-at-large into population and/or tributary specific escapement estimates. Escapement estimates generated this way are reported in Tables 1 and 2.

Table 1. Escapement estimates and 95% confidence intervals for steelhead generated by ISEMP adult tagging and PIT tag array interrogation. Independent estimates are generated by weirs and provided to validate escapement estimates generated by PIT tag arrays.

| | 2009-2010 | | | 2010-2011 | | |
|---|---|---|---|---|---|---|
| **Tributary** | Estimate | 95% CI | Independent Estimate | Estimate | 95% CI | Independent Estimate |
| **Potlatch River** | 784 | 621-992 | | 739 | 443-1541 | |
| **Fish Creek (Lochsa River weir)** | 246 | 129-434 | 205 | | | |
| **Asotin Creek** | 1687 | 1407-1963 | 1,500 | 973 | 778-1224 | 1,128 |
| **Rapid River (weir)** | 136 | 72-235 | 150 | | | |
| **South Fork Salmon River** | 1795 | 1527-2081 | | 2,980 | 2,654-3361 | |
| **Secesh River** | 298 | 169-558 | | 433 | 250-738 | |
| **Big Creek** | 753 | 431-1914 | | 745 | 562-960 | |
| **Lemhi River** | 630 | 455-928 | | 503 | 346-736 | |
| **Valley Creek** | 237 | 155-411 | | 270 | 190-382 | |
| **Upper Salmon (Sawtooth weir)** | 138 | 76-226 | | 79 | 36-147 | 98 |
| **Lapwai Creek** | | | | 455 | 262-1340 | |
| **Joseph Creek (Grande Ronde River)** | | | | 1,663 | 1420-1921 | |
| **Cow Creek (Imnaha River)** | | | | 161 | 94-250 | |
| **Imnaha River** | | | | 3,516 | 3167-3897 | |

Table 2. Escapement estimates and 95% confidence intervals for spring/summer Chinook salmon generated by ISEMP adult tagging and PIT tag array interrogation. Independent estimates are generated by weirs and provided to validate escapement estimates generated by PIT tag arrays.

| | 2009-2010 | | | 2010-2011 | | |
|---|---|---|---|---|---|---|
| **Tributary** | Estimate | 95% CI | Independent Estimate | Estimate | 95% CI | Independent Estimate |
| **South Fork Salmon River** | 7,005 | 6,655-7,355 | | 4,749 | 4,326-5,201 | |
| **Secesh River** | 1,308 | 1,165-1,451 | | 779 | 745-791 | |
| **Big Creek** | 285 | 150-411 | | 449 | 290-689 | |
| **Lemhi River** | 262 | 243-281 | | 337 | 230-470 | |
| **Valley Creek** | 235 | 191-281 | | 460 | 380-560 | |
| **EF South Fork Salmon River** | 1,026 | 2,731-4,169 | 1,032 | | | |
| **Imnaha River** | | | | 2,421 | 2,124-2,716 | |

## Monitoring Juvenile Salmonid Populations

Salmon Subbasin ISEMP juvenile abundance estimates were initiated in 2009 and are confined to the South Fork Salmon River (SFSR) and Lemhi River subbasins which are targeted for effectiveness and/or status and trend evaluation. For the purposes of this report, we focused efforts on reporting juvenile

**ISEMP Juvenile Monitoring Protocol**
This protocol allows us to estimate juvenile abundance, growth and survival in the habitats most often targeted for restoration actions; namely, the freshwater habitat where fish reside prior to emigration in a standardized fashion among subbasins.

abundance estimates for the TRT identified population of spring/summer Chinook salmon and steelhead in the Secesh River (SFSR) and the subpopulations of spring/summer Chinook salmon and steelhead in the Lemhi River.

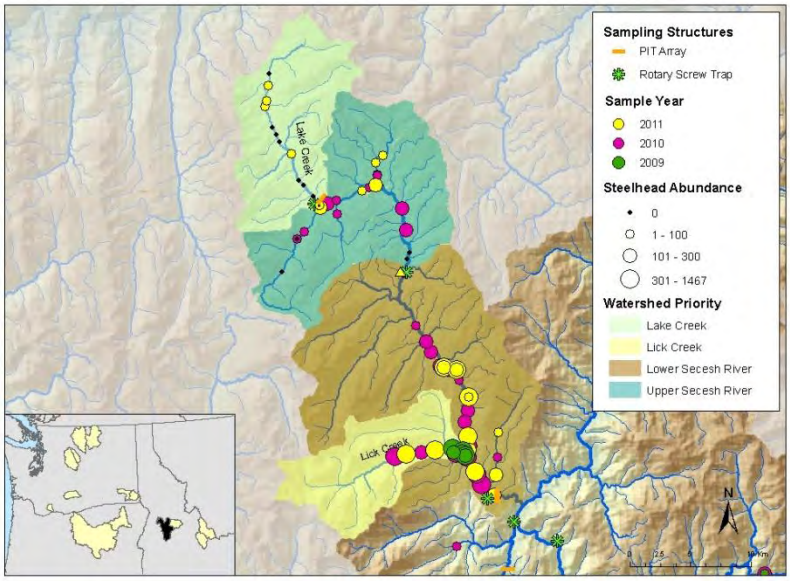



Figure 2. Location of juvenile sampling infrastructure and distribution and abundance of juvenile steelhead obtained via remote juvenile surveys in the Secesh River.

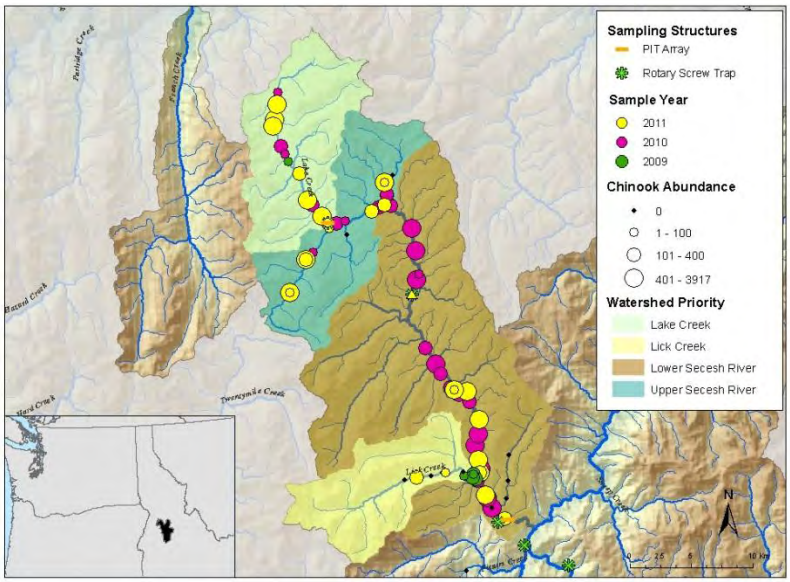



Figure 3. Location of juvenile sampling infrastructure and distribution and abundance of juvenile spring/summer Chinook salmon obtained via remote juvenile surveys in the Secesh River.

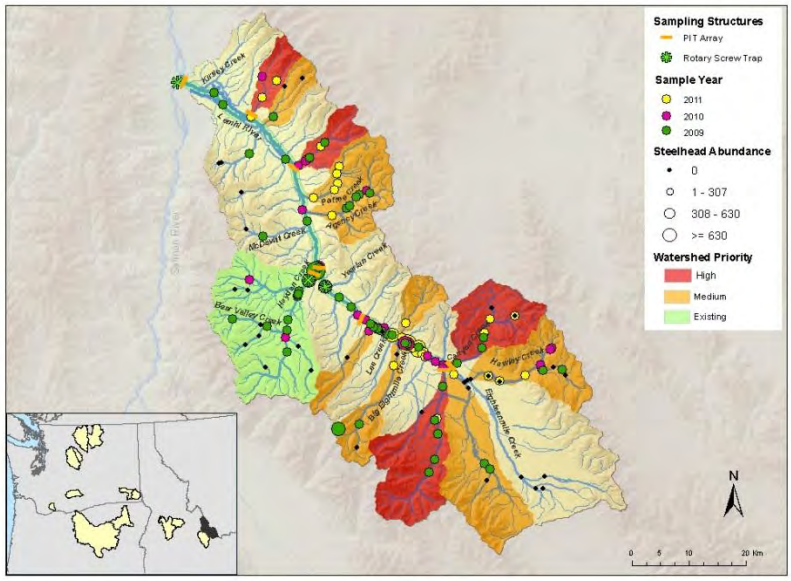



Figure 4. Location of juvenile sampling infrastructure and distribution and abundance of juvenile steelhead obtained via remote juvenile surveys in the Lemhi River.

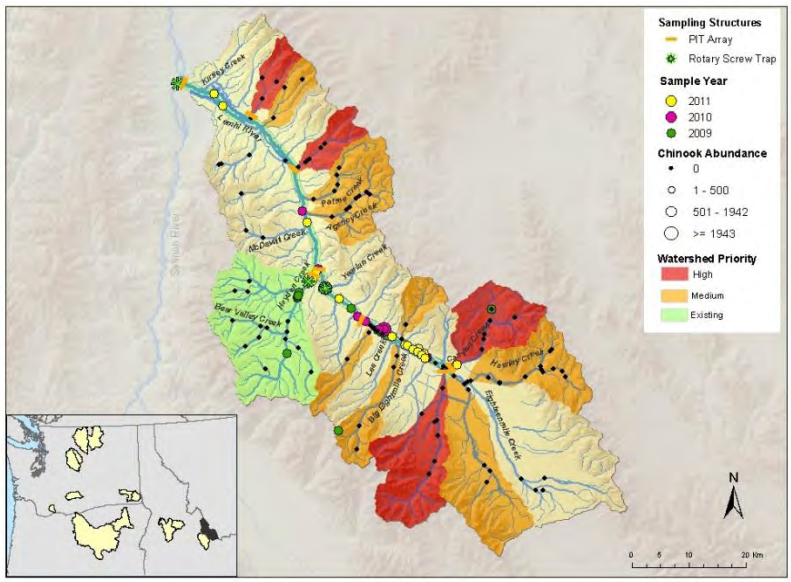



Figure 5. Location of juvenile sampling infrastructure and distribution and abundance of juvenile spring/summer Chinook salmon obtained via remote juvenile surveys in the Lemhi River.

## Habitat Status and Trend Data

**Habitat quality and quantity metrics**
Information rich data allows spatial and temporal pattern detection. Stream habitat measurements are only useful if the data can show differences between watersheds, types of streams, types of disturbance and are repeatable.

The quantity and quality of tributary stream habitat varies by ESA listed salmonid population across the Columbia River basin. The variability in habitat condition across the Columbia River basin occurs naturally by ecoregion, but is also strongly influenced by human activity. To be useful to managers, current monitoring activities must be able to see differences in key habitat metrics between population watersheds, between levels of human-caused disturbance, and through time as habitat management actions play out. Most importantly, monitoring programs must generate habitat condition metrics that can consistently quantify spatial and temporal patterns that arise from natural variation and human impacts – only then are these metrics useful for management purposes. Quantifying habitat condition allows managers to estimate the current and historic salmonid population capacity and productivity of these watersheds and to plan and track the implementation of mitigation strategies.

By stream or watershed habitat status, we generally mean a snapshot of the spatial variation in habitat conditions throughout the stream network derived from a survey of sites throughout the network. ISEMP and CHaMP habitat monitoring protocols have proven able to detect differences in the status of metrics among watersheds. For example, Figure 6 shows a comparison among watersheds for two metrics, fine sediment and pool area.

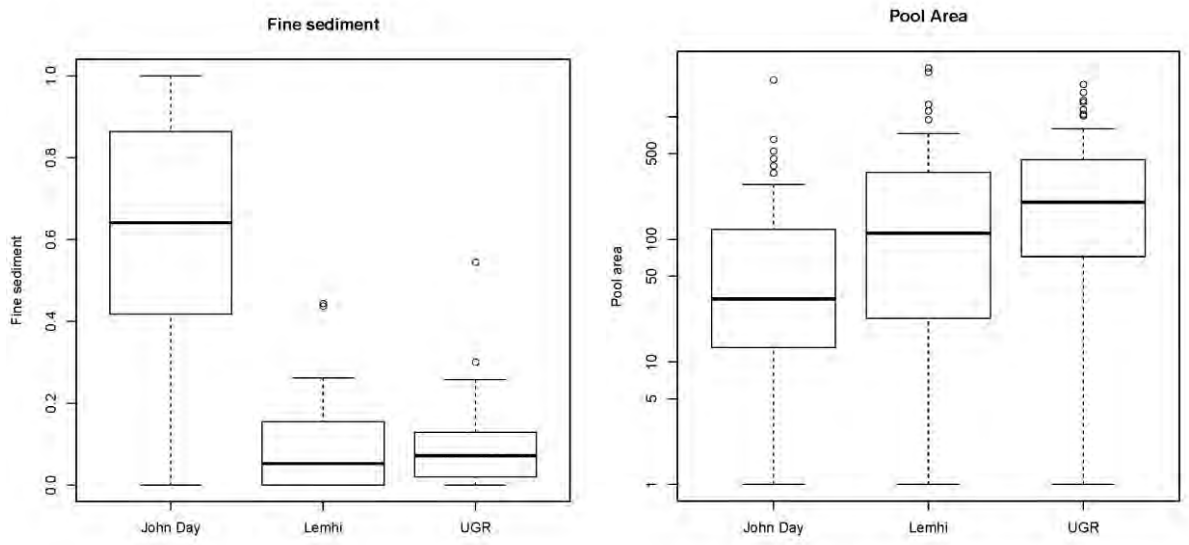



Figure 6: Boxplots of fine sediment (top) and pool area (bottom) of habitat metrics in the John Day, Lemhi and Upper Grande Ronde (UGR) watersheds.

There is a clear difference between fine sediment levels in the John Day network compared with that in the Lemhi and Upper Grand Ronde. On the other hand, for some attributes, spatial differences are small, as the plot of pool area illustrates.

ISEMP and CHaMP habitat monitoring protocols are also sensitive to detecting trends in habitat metrics. Trends are any patterns of change over time, usually with respect to change across years. We can measure and report trend as an underlying 'average' trend across all sites in a region – is habitat condition changing in my watershed – or, trend might be expressed as a series of site specific trends derived from revisiting the same set of sites over time. The distribution of site specific trends might have a mean of 0, in which case we would conclude that there is no regional trend, or the mean might be positive or negative, indicating a regional trend of improving or degrading habitat condition. Figure 7 shows the trend in average stream depth at ISMEP monitoring reaches in the Wenatchee River, at the subwatershed level and at the individual monitoring sites.

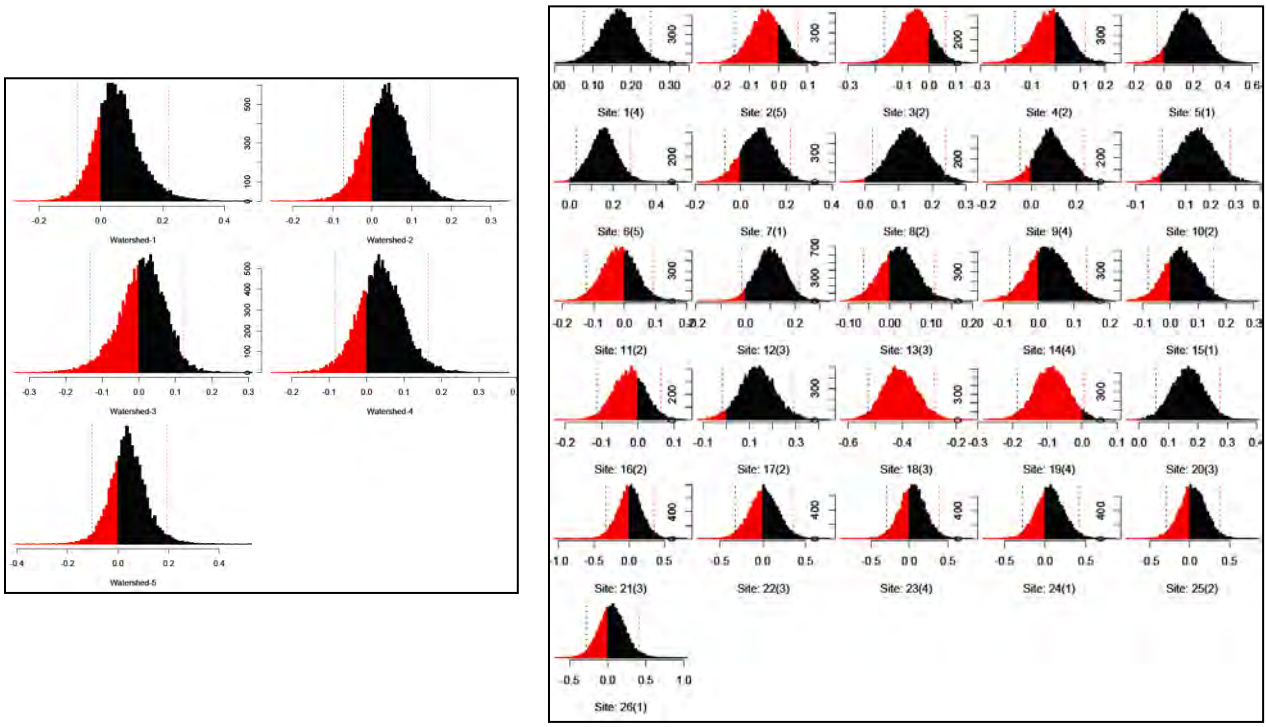



Figure 7. Trend in bankfull depth watersheds and monitoring reaches in the Wenatchee River subbasin over the period 2004 - 2009. Left-hand panel illustrates trend pattern in 5 subwatersheds of the Wenatchee Basin. Color coding reveals the probability that a negative (red) or positive (black) trend is detectable. There is evidence for a positive trend in bankfull depth in four of the watersheds, but not for one of them (watershed-3), based on a visual inspection. Right-hand panel illustrates trend at individual monitoring sites. Those sites with nearly all black or red indicate a high probability of either a positive or negative trend, respectively.

In order to evaluate how well we can determine status and trends, we need a framework that describes important components of variation and survey designs that allow us to determine those

**Variance Decomposition**
Breaking down variability within scientific measurements and metrics reduces uncertainty in subsequent management decisions.

components. The framework that ISEMP and CHaMP uses decomposes variability in a hierarchical fashion:

- **Spatial variation** describes the fundamental site to site differences
- **Yearly temporal variation** consists of two parts: common yearly variation across all sites in the domain as might be driven by external factors such as climate or ocean conditions (**coherent temporal variation**), and independent variation among sites: each site's yearly variation is independent of other sites yearly variation (**interaction variation**).
- **Residual (or extraneous) variation** introduced during the yearly sampling season comes from: a) temporal variation within the sampling season, b) an imprecise sampling or measurement protocol, or c) crew to crew differences in applying a standard protocol.

Properly designed surveys, like that used in ISEMP and CHaMP, allow us to estimate these important components of variation and to estimate their influence on estimates of status and trends. Example summaries illustrate the range in relative magnitude of these variance components for a few of the habitat attributes commonly measured in ISEMP and CHaMP (Figures 8 and 9).

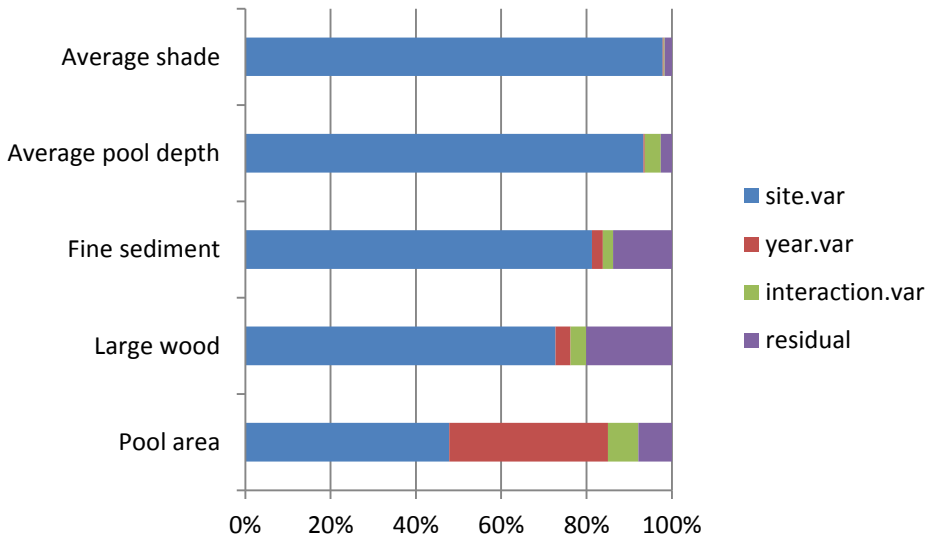



Figure 8. The relative proportion of total variation that is attributable to site, concordant, interaction, and residual variation, Wenatchee 2004 – 2010.

As can be seen in Figure 8, site variance comprises a large portion of total variation for average shade and average pool depth, indicative of a relatively clear "site" signal. For large wood and fine sediment, the residual component of variation is relatively large pointing to possible poor protocol performance. The pool area metric demonstrates a large concordant year component of variation implying that trend detection power for this metric will be low, and that there might be some external factors driving its magnitude.

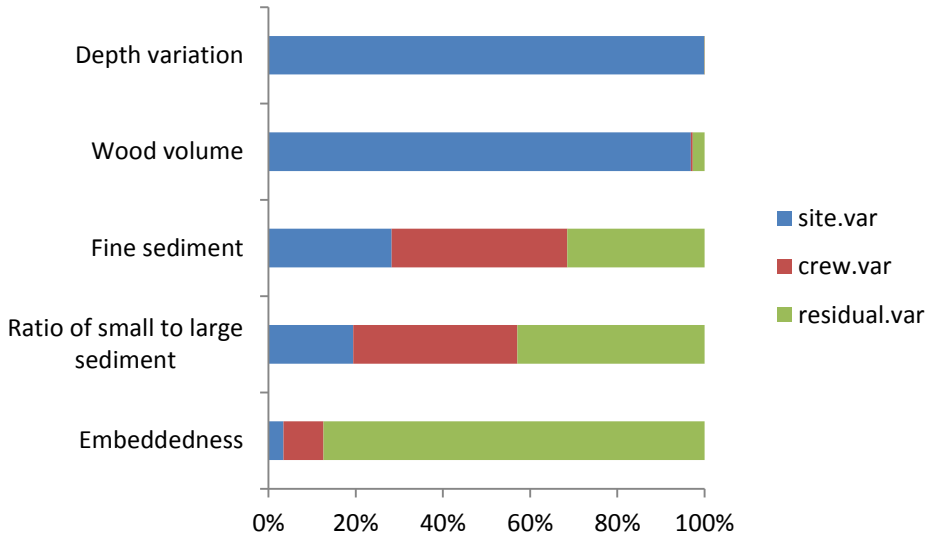



Figure 9. The relative proportion of total variation that is attributable to site, concordant, interaction, and residual variation from six sites in the Upper Grande Ronde during a short time interval with data collected using the CHaMP protocol.

Three important variance components are summarized here: site variance across the six sites surveyed for this study, crew variance (what fraction of the total variance could be attributed to different crews applying the same protocol), and residual variance (which most likely covers the variance associated with one crew applying the same protocol repeatedly during a short time interval). Large crew variance (e.g., for fine sediment and ratio of small to large sediment) implies that additional training might be useful. Large residual variance for the embeddedness metric implies that the sampling protocol is difficult to implement even by the same crew suggesting a problem with the repeatability of the measurement protocol itself. Depth variation and wood volume metrics perform quite well with low crew and residual variability.

## Habitat Impairments and Ecological Limiting Factors: Human Disturbance on the Landscape

Several modeling approaches have been presented to link attributes of fish populations (abundance, productivity, survival) to habitat conditions. The habitat attributes used in the models are derived from measurements made at sites. In order to "solve" the models, we need measurements of local habitat condition.

**Habitat Impairments Predictions**

Identifying habitat impairments systematically is critical for effective restoration planning. ISEMP's use of remote sensed data, landscape classification, and field data allows for identification, verification and progress tracking. Furthermore,

**Bang for the Buck**

Linking habitat impairment and intrinsic potential predictions leverages cost-efficiencies. Efficient restoration planning must consider both the degree of impairment and the potential value to fish for every action.

However, we would also like to predict where habitat conditions are expected to be good or poor to efficiently guide habitat restoration planning. The goal is to develop spatially explicit models of expected habitat condition so that we can create maps that show spatial patterns in expected good or poor habitat condition. These maps will allow us to target restoration actions in areas where habitat is expected to be in poorest condition and will allow us to track recovery toward an "acceptable" habitat condition.

USBOR and NOAA have developed a landscape classification that organizes watersheds (6th field HUCs) into classes with common natural features and classes with common "disturbance" features. This classification allows us to ask if there are relationships between habitat measurements and disturbance gradients and can these relationships provide insight into a framework for identifying spatial patterns in degraded networks? This disturbance gradient is based on four landscape attributes: proportion of 6th field HUC that is in urban land use, proportion in agricultural land use, proportion of impervious surface, and road density. Using monitoring data from CHaMP and assigning each sampling location a disturbance score (Best, Good, Moderate, and Poor) and a geomorphic valley type (Mountain and Floodplain/Constrained) illustrate the gradients between the observed habitat and expected habitat condition. There is a clear gradient in habitat condition as one progresses from sites classified as best toward those sites classified as poor.

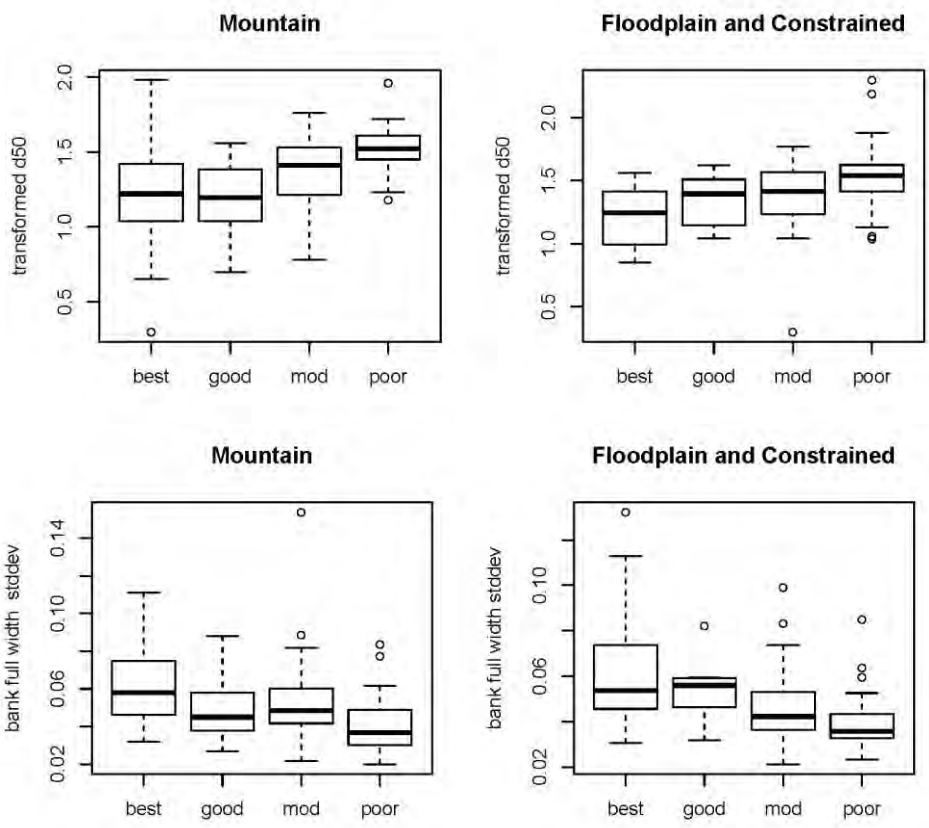


Figure 10. The relationship between two measures of habitat condition (d50, a measure of fine sediment, and standard deviation of bankfull width) and disturbance gradient for two classes of streams across four disturbance classes.

This kind of information can be used in two ways: as a tool for targeting restoration and for tracking recovery. Both the variables used to develop the disturbance gradient and the geomorphic classes are landscape features and can be mapped across the entire domain. These maps can display the spatial pattern in stream networks in the various condition classes, indicative of the locations where highest probability of poor habitat condition would be expected (Figure 11). These are areas where restoration could be targeted. The overall impact of restoration can then be tracked by the progression of the distribution of habitat metrics at restored locations toward those at the "best" sites.

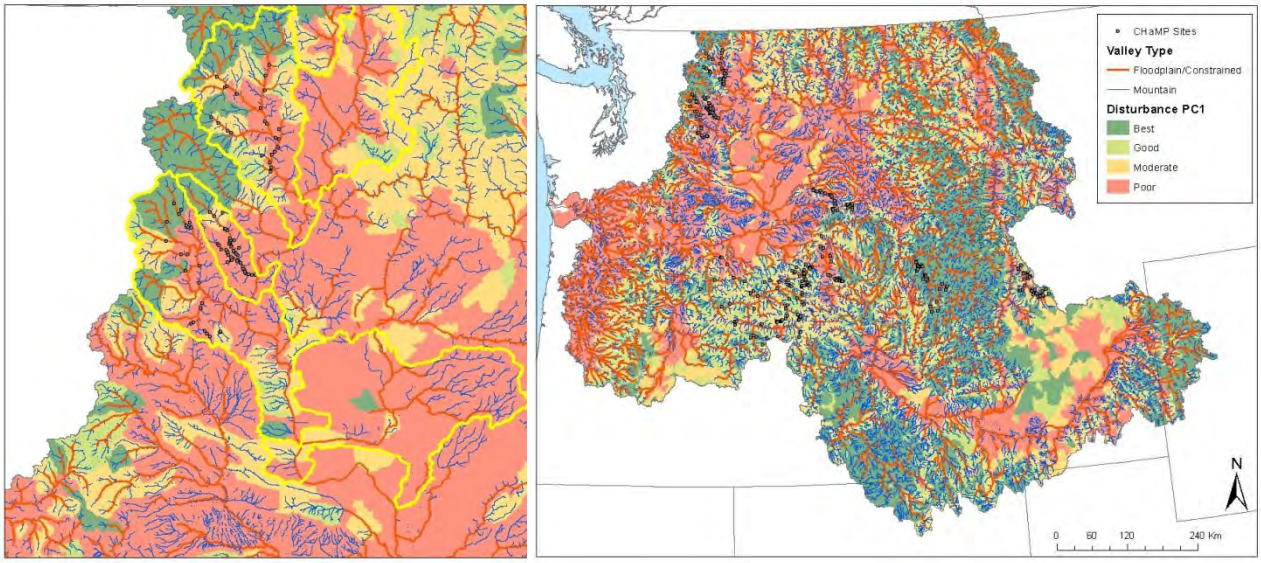



Figure 11. These maps illustrate where the probability of finding poor habitat condition is likely to be high and therefore where habitat restoration might be concentrated. The stream network classified into two geomorphic groups: Mountain and Floodplain/Constrained) because patterns of disturbance and recovery goals could differ. The lower panel is a closer look at the Upper Columbia portion of the upper panel.

## Temperature Impairment and Intrinsic Potential

**Intrinsic Potential Maps**
Intrinsic potential maps tease apart "natural conditions" from "human impairment."

Summer stream temperature is thought to limit salmonid productivity in many parts of the interior Columbia River basin. In parts of the basin, summer stream temperatures are naturally higher than those tolerated by cold water fishes, but in other parts of the basin, human activity such as water withdrawals, riparian corridor modification and stream channel simplification has resulted in elevated stream temperatures. Due to the interactions of naturally occurring warm summer streams and the landuse factors that unnaturally elevate stream temperatures, identifying stream

temperature impairments, and thus habitat mitigation opportunities, is not a simple case of measuring water temperature.

ISEMP has developed continuous stream temperature models based on remotely sensed data that predict daily minimum, maximum and mean stream temperature for all stream reaches over the past decade. By establishing risk criteria based on duration and magnitude of exposure to elevated summer stream temperatures, we can map the current occurrence of potential habitat impairment. Linking these maps with salmonid habitat intrinsic potential (IP) from the Interior Columbia Technical Recovery Team (ICTRT), we can predict the spatial locations (stream reach), degree of impairment (risk score), and relative priority for mitigation actions (risk score x IP score; Figure 12). An example from the John Day River basin shows that while roughly 50% of the steelhead domain in the basin is in high risk for summer thermal impacts stream reaches, only half of that extent has high intrinsic potential (Figure 13). Intersecting temperature risk modeling with IP extent allows managers to identify reaches and subwatersheds to target for mitigation actions and to prioritize suites of potential actions by expected benefit to salmonid populations.

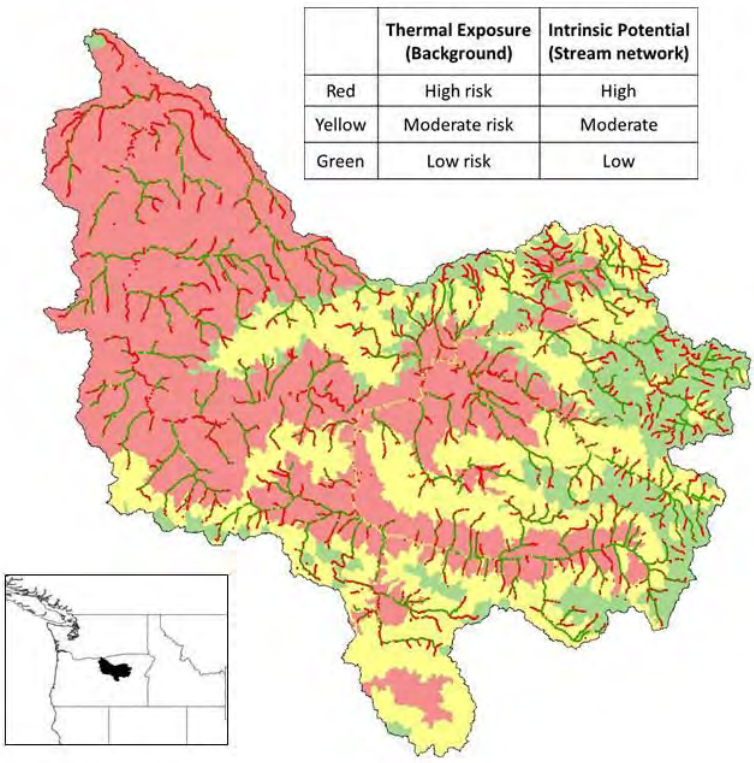



Figure 12. John Day River basin summer thermal impairment risk (background color) and Intrinsic Potential rating (stream color).

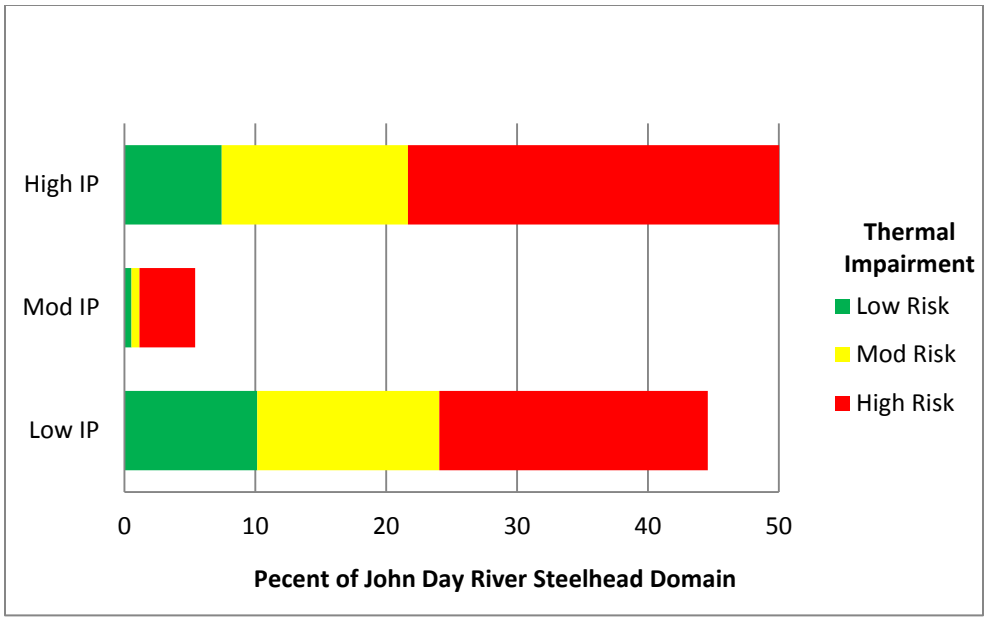



Figure 13. Relative proportion of the John Day River basin steelhead domain of Low, Moderate and High Intrinsic Potential (IP) falling in Low/Moderate/High summer thermal impairment conditions.

## Habitat Change Resulting from Restoration Actions: Bridge Creek Change Detection

Within the semi-arid interior Columbia River basin, channel incision is a widespread problem that degrades stream habitat by increasing channel gradient, reducing channel complexity, and disconnecting the floodplain resulting in a loss of groundwater storage capacity and riparian vegetation. This, in turn, leads to reduced based flows, increased summer stream temperatures, and a loss of spawning and rearing habitat. Instream and floodplain habitat within Bridge Creek has been degraded by channel incision. ISEMP is implementing the Bridge Creek IMW study to restore large sections of instream and riparian habitat along the lower 31 km of Bridge Creek sufficient enough to cause a detectable population-level benefit to steelhead.

**Habitat Change Detection**
Habitat change detection is a graphical and intuitive way to see changes in stream habitat caused by restoration actions. This product is literally a three-dimensional picture of the stream – that shows "changes." Successfully quantifying the change in habitat quality and quantity is necessary if we are to link restoration actions to benefits for fish populations.

Currently beaver occupy Bridge Creek where they build dams that aggrade the stream channel (deposition of sediments behind beaver dams that raise the stream bed). However, the current lack of large wood results in unstable dams that have a short life span. We are attempting to cause aggradation of the incised stream trench to restore floodplain connectivity by installing a series of instream beaver dams support structures (BDSS; vertical wood post driven into the stream bottom; Figure 14) designed to assist beaver in the construction of stable longer lasting dams.

Figure 14. An example of a beaver dam support structure (BDSS) 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 first step of the restoration design was implemented in 2009 where 84 structures were installed in four treatment reaches, leaving six reaches that will act as controls until they are treated in 2013. Because our goal is to aggrade the system, we must be able to describe this change.

The primary change detection metric in this project is the DEM of difference, or the difference of digital 3D maps of the channel constructed before the actions and one conducted after implementation. The DEM of difference is the change in stream bed elevation within the stream channel (Figure 15). Each point in the stream bed topography is evaluated before and after the treatment. A negative value (represented in red) indicates erosion, where a positive value (new elevation is higher than old; represented in blue) indicates deposition, and neutral change (represented as white). This is done for every point to create a surface, and a distribution of the actual elevational changes.

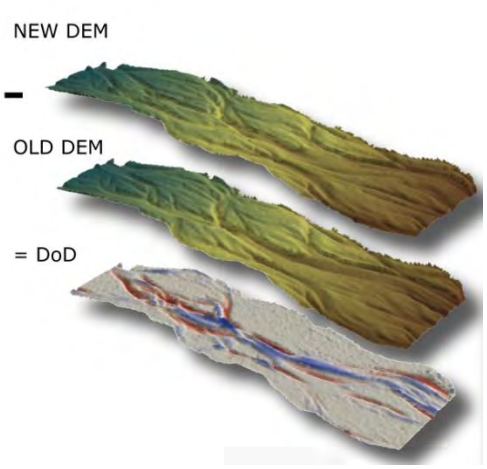



Figure 15. Concept of DEM differencing.

One year after installation of the BDSS, 30% were colonized by beaver, beaver activity was present in all treatment reaches, and beaver had expanded into a treatment reach previously unoccupied. In general, deposition occurred behind beaver dams and BDSSs, with scour pools forming downstream. DEMs of difference capture this general pattern clearly (Figure 15). We were also able to describe how the channel changed as a response to the actions, revealing the amount of deposition in treatment reaches was positive (the channel aggraded; Figure 16).

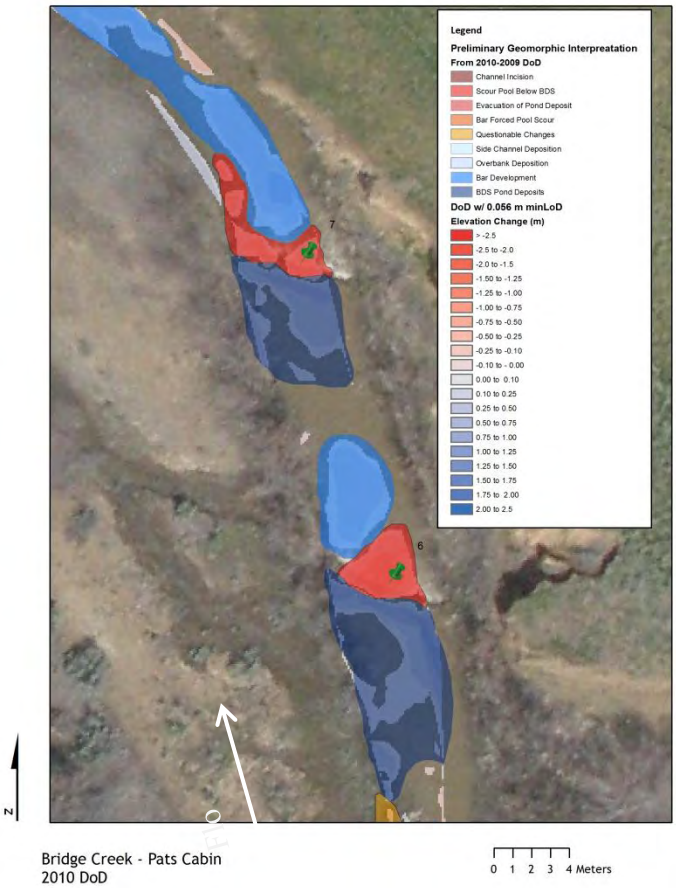



Figure 16. DEM of difference (post-restoration minus pre-restoration) from topographic surveys for a portion of treatment reach in Bridge Creek. Pushpins represent structure location. Blue color represents aggradation (deposition of sediments), and red represents erosion. General pattern was to have deposition behind structures, scour pool below structures, and deposition of the scour downstream from the pools.

After only two years we are not able to assess whether fish populations have responded, although we did observe Chinook salmon spawning in Bridge Creek for the first time one year post- restoration. We have been able to demonstrate that the stream channel has responded as expected (but at a much faster rate) to the restoration action, while at the same time display the utility and interpretation of this monitoring approach that has been adopted by ISEMP and CHaMP.

## Watershed Production Model: Lemhi

The watershed model evaluates the effectiveness of habitat restoration actions on salmonid salmon and is a useful platform to relate habitat restoration actions to the freshwater productivity of anadromous salmonids. As a decision support tool the modeling approach is easily generalized to locations/situations that are not "data-rich" when compared to the subbasins where ISEMP is implementing it. We are investigating how to most cost-effectively apply this modeling approach as a standard tool across the Columbia Basin, providing a quantitative approach for identification of limiting factors, restoration alternatives, and quantitative effectiveness evaluations necessary to service the information needs of the BiOp.

In terms of policy and management, the watershed model provides several useful products:

- It identifies factors that limit freshwater productivity at specific life-stages[1], enabling habitat restoration actions to better target problems and conversely to avoid habitat initiatives that are unlikely to address primary limiting factors;
- It identifies the types and magnitude of habitat alteration most likely to improve freshwater productivity;
- It provides a platform to evaluate alternative restoration actions to identify/prioritize actions most likely to cost-effectively improve freshwater productivity;
- It translates habitat quantity and quality to fish abundance, namely identifying reasonable expectations for total production;
- It identifies the types of monitoring most likely to detect changes in habitat conditions and freshwater productivity within a specified period of time;
- It provides an analytical tool to quantitatively evaluate change in habitat conditions and freshwater productivity; and
- It can be used to predict adult escapement taking into account ocean conditions, harvest, and hatchery impacts.

**Watershed Production Models**
A watershed production model is the analytical framework for prediction fish population response to changes in watershed-level habitat quality and quantity.

The utility of the watershed model can be demonstrated by its application in the Lemhi River (Salmon Subbasin, ID). The Lemhi River is substantially influenced by irrigation withdrawals. At the initiation of the Salmon Subbasin ISEMP project in 2009, only two of the 30 major tributaries of the Lemhi River were hydraulically connected to the mainstem Lemhi River year around (Figure X). Regional management agencies identified the loss of tributary habitat as a factor limiting the productivity of spring/summer Chinook salmon and steelhead. The 2008 BiOp suggests that

[1] Life-stages refer to changes in the ecological requirements of salmonids, defined by differences in the type of habitat required to meet growth requirements. For juvenile salmonids this is illustrated by a number of stages, including the transition from redds at emergence to active feeding stations as parr. At a coarser scale, this concept is also illustrated by the transition from headwater habitat to mainstem habitat during active emigration.

Phase One habitat restoration actions (including those restoration actions implemented between 2000 and 2006) are anticipated to achieve a 0.5% increase in Chinook salmon survival by 2017. The BiOp indicates that Phase Two habitat actions (to be implemented from 2007 through 2017) are anticipated to result in a 20% increase in Chinook salmon survival by 2017. Following this finding, significant BPA funding has been allocated towards projects aimed at "reconnecting" tributary habitat historically important for Chinook salmon and steelhead production. The large spatial scale and aggressiveness of these tributary reconnections makes the Lemhi River an ideal case-study for ISEMP. As it is unlikely that funding and logistics will enable the reconnection of all Lemhi River tributaries, how then do managers choose which tributaries should be the focus of restoration efforts? Are there alternative or additional habitat restoration actions that could prove effective at increasing freshwater productivity?

Initially, managers identified "high priority" watersheds as primary targets for restoration efforts. This prioritization was based on existing information describing habitat conditions modified by the logistical feasibility of obtaining successful tributary reconnections (e.g., number and cost of flow enhancement or alternative water diversion projects necessary to maintain instream flow). ISEMP, in collaboration with the co-managers and federal agencies is tasked with evaluating the effectiveness of Phase One, and identifying whether additional tributary reconnects will be necessary to achieve the freshwater productivity improvements necessary to achieve the goals identified in the BiOp.

ISEMP will provide policy and management a tool in the form of a watershed model that will enable a consistent methodology to identify limiting factors, identify the most cost-effective and logistically viable suite of restoration actions to address those limiting factors, and rigorously document the resulting change in freshwater productivity. As importantly, the application of this tool will enable managers to identify why habitat restoration investments to date have or have not delivered the anticipated benefits to fish survival.

In this section we illustrate how the model can be used to address BiOp related management questions. However, the results presented in this section must be prefaced by a caveat. The model is life-stage specific and brood-year based, meaning that it requires estimates of adult escapement and subsequent juvenile production attributable to those adults. Given that the ISEMP was initiated in 2009 in the Salmon subbasin, we currently have data for less than one complete brood year of spring/summer Chinook salmon and steelhead. The first complete brood year of production estimates will occur following juvenile emigration in 2012 for spring/summer Chinook salmon and 2014 for steelhead. While the results presented in this section utilize all data collected to date, data for incomplete brood years utilizes values from literature as necessary.

The model yields a number of estimates that are useful in a management context, for the purposes of this section we focused on changes in juvenile production (smolts per female; Table 10) predicted following the reconnection of all high priority tributaries and all high and moderate priority tributaries. Anticipated changes in juvenile and adult abundance accompanying restoration alternatives are illustrated in Figures 17 and 18.

Table 3. Percent change in spring/summer Chinook salmon productivity (smolts/female) estimated given the reconnection of high priority tributaries and high and moderate priority tributaries.

| Restoration Scenario | Percent Change |
|---|---|
| Existing Habitat | 0% |
| High Priority Reconnections | 11% |
| High and Moderate Priority Reconnections | 13% |

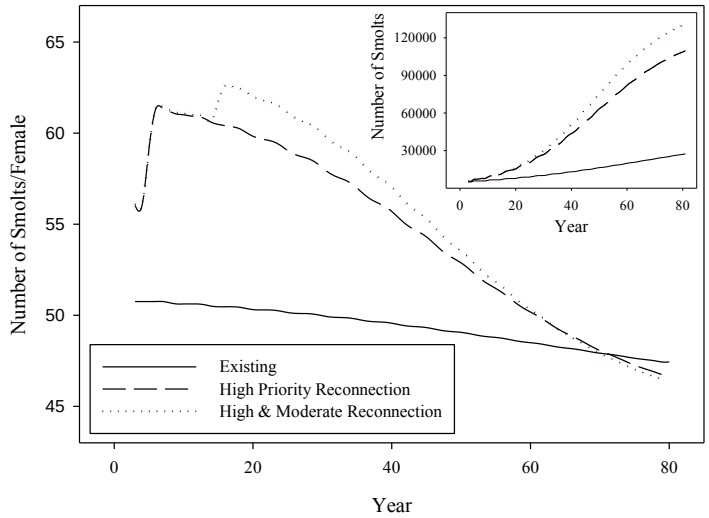



Figure 17. Number of smolts per female and total estimated smolt production (inset) given existing habitat, reconnection of high priority tributaries, and addition of high and moderate priority tributaries.

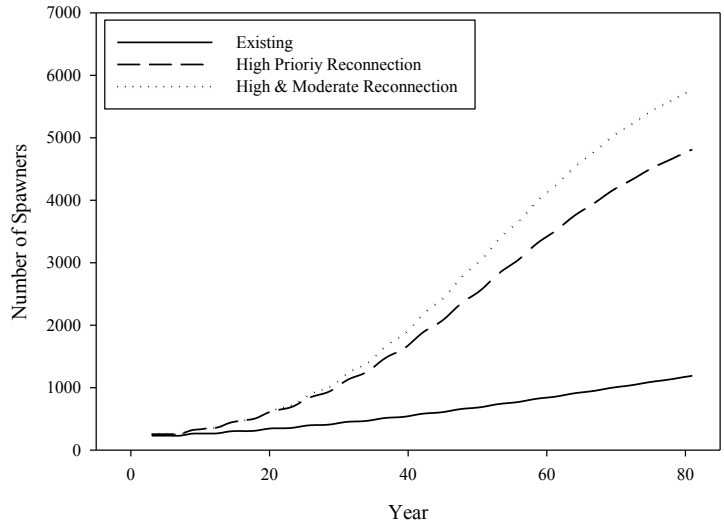



Figure 18. Number of spring/summer Chinook salmon adults returning to the Lemhi River given existing habitat, reconnection of high priority watersheds, and reconnection of high and moderate priority watersheds.

Data will be insufficient to fully populate the watershed model until 2013 for spring/summer Chinook salmon and at least 2014 for steelhead. Nonetheless, the provisional model results described above are useful for illustrating how the watershed model is a useful tool for evaluating the objectives of the BiOp as they relate to offsite mitigation in the form of habitat restoration. The model results support the conclusions of the Lemhi Conservation plan which identified the quantity and quality of juvenile rearing habitat as the primary factor within the Lemhi River that limits freshwater productivity. Although data are insufficient to evaluate the benefits of individual tributary reconnections, the classification of tributaries into high and moderate groups is largely supported by model results. Provisional model results also suggest that improvements in habitat quality in reconnected tributaries may be required to achieve the targeted 20% improvement in freshwater productivity described in the BiOp. By 2013 habitat and fish sampling in the Lemhi will be sufficient to support model evaluations aimed at identifying what habitat reconnection and/or improvement scenarios will most cost-effectively produce estimated improvements of at least 20% in freshwater productivity. These results will be available to support the 2013 BiOp comprehensive check-in and 2017 BiOp evaluation.

## Habitat-Juvenile Salmonid Abundance Relationships using Wenatchee ISEMP Data

Habitat monitoring programs need to measure those habitat characteristics which best predict fish population parameters such as abundance, growth and survival. Habitat monitoring should also inform the development of restoration actions so those actions fix the right aspects of habitat that produce more fish. To determine which habitat metrics are most important in predicting fish population parameters and therefore which should be included in a habitat monitoring protocol, ISEMP compared fish densities and a suite of habitat characteristics in the Wenatchee River subbasin from 2004 to 2010.

**Fish Habitat Relationships**
ISEMP has identified decision support products in four categories of fish—habitat relationships including:

- Abundance
- Growth
- Location

Figure 19 shows the relative importance of 15 habitat metrics identified from an original 23 metrics as most important for predicting the density of juvenile Chinook. They are listed from most to least important metrics, with most important at the top. The most important, the year effect (which accounts for differences in spawner abundances as well as environmental conditions not included among the predictor variables) is about twice as important for predicting juvenile Chinook density as gradient or a measure of temperature.

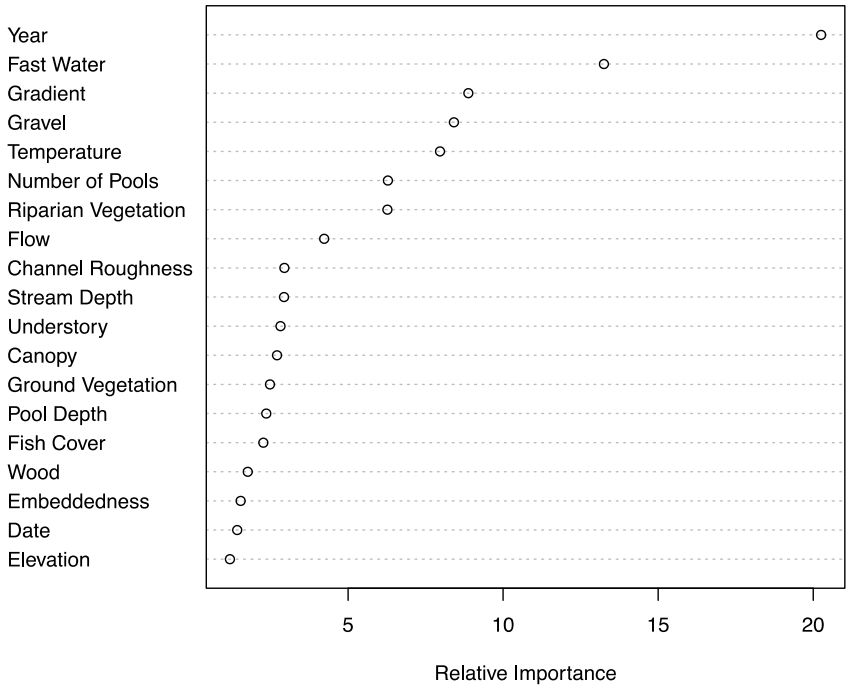



Figure 19. The relative importance of various habitat metrics in predicting the density of juvenile Chinook using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010 analyzed using a boosted regression tree approach.

**Fish Habitat Relationships – Abundance**
The abundance of rearing juvenile salmon can be predicted with suites of habitat measures allowing us to ask "Do habitat metrics predict fish density and do these fish-habitat relationships tell us what are "good" and "bad" habitat conditions?"

The fact that year is the most important variable predicting juvenile Chinook density underlines the necessity of monitoring habitat for more than one or two years in order to get a reliable picture of juvenile densities: densities in any one year could be very misleading.

Effective habitat restoration actions need to target limiting factors in any given tributary. The above ISEMP analysis was taken further to tease apart the "limiting factors" question. When we look at the relationships between the most relevant habitat metrics and Chinook density (Figure 20) several thresholds become apparent that can be used to identify limiting factors and provide quantifiable goals for habitat restoration work. For example, predicted values of the density of juvenile Chinook are high for low values of fast water, decline steadily for mid-range values and level off at higher values. This implies that sites with less than 5% fast water are important for juvenile Chinook and that restoration actions should target sites with too much fast water area, i.e., restoration actions should create slow water refugia.

The amount of gravel is another habitat characteristic that has a clear threshold relationship to Chinook density (Figure 20). Chinook density jumps from a low to a high value

once the percent coarse gravel crosses a threshold near 35%. This clearly shows how the amount of gravel at a site can be a factor limiting the density of juvenile Chinook.

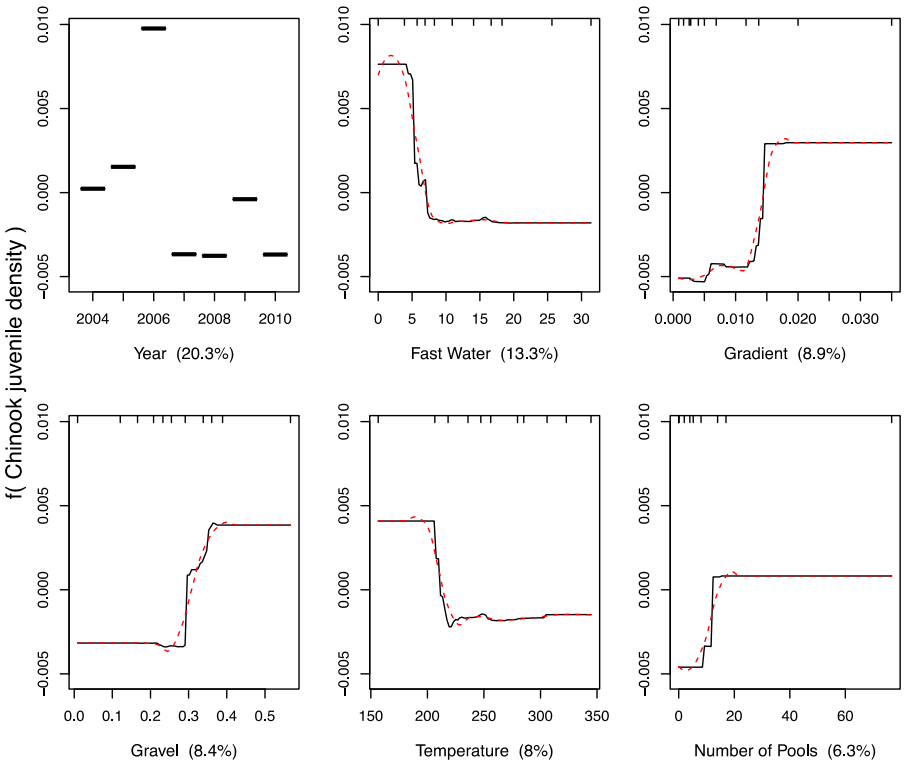



Figure 20. Partial dependence plots showing the effect of the six most important habitat metrics identified using a RBT on juvenile Chinook densities using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010.

Since different species have different habitat needs restoration actions need to account for the target species. ISEMP monitoring in the Wenatchee was able to detect these differences. Figures 21 and 22 show how steelhead respond to a different set of habitat metrics than Chinook and at different thresholds that are consistent with differences between the species.

Figures 21 and 22 also show how we can answer the question "How much restoration is enough?" If a restoration action limited the amount of fast water to less than 5% of the surface area, kept stream gradient to more than 0.015, reduced thermal input to less than 200, increased gravel to about 35%, and provided 15 or more pools per mile, then this work suggests that that action or suite of actions should maximize the density of Chinook at that site. However, additional work needs to be done to more specifically define threshold levels and to confirm consistency outside of the Wenatchee subbasin before these results should be used in management decision-making. Nonetheless, this represents an analytical framework for habitat and fish status and trend data that can be used to help answer the question "*Are habitat restoration actions effectively helping salmonid populations recover?*"

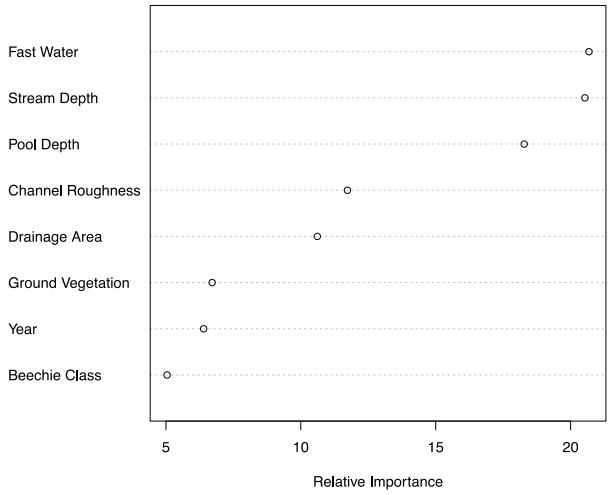



Figure 21. The relative importance of various habitat metrics in predicting the density of juvenile steelhead using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010.

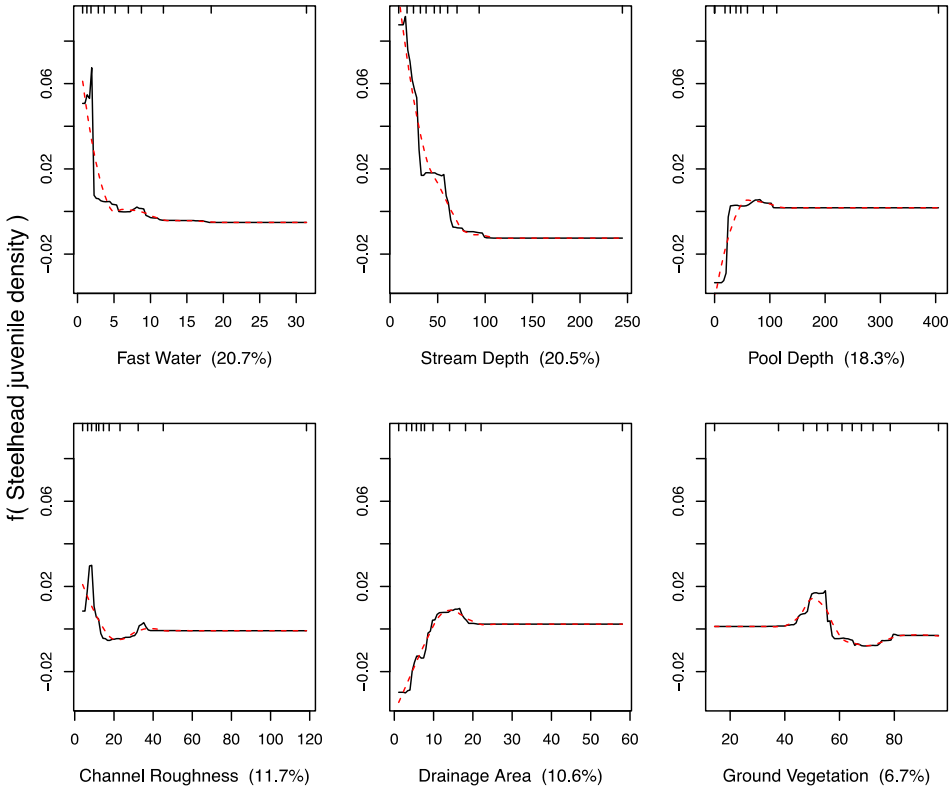



Figure 22. The effect of the four most important habitat metrics on juvenile steelhead densities using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010.

Figures 23 and 24 show the predicted densities of juvenile Chinook based on the amount of fast water and the percentage of coarse gravel respectively.

**Fish—habitat relationships: location**
Multi-dimensional maps of fish habitat can be used to represent status and trends, validate predicted values with actual observations, and can help identify or evaluate limiting factors.

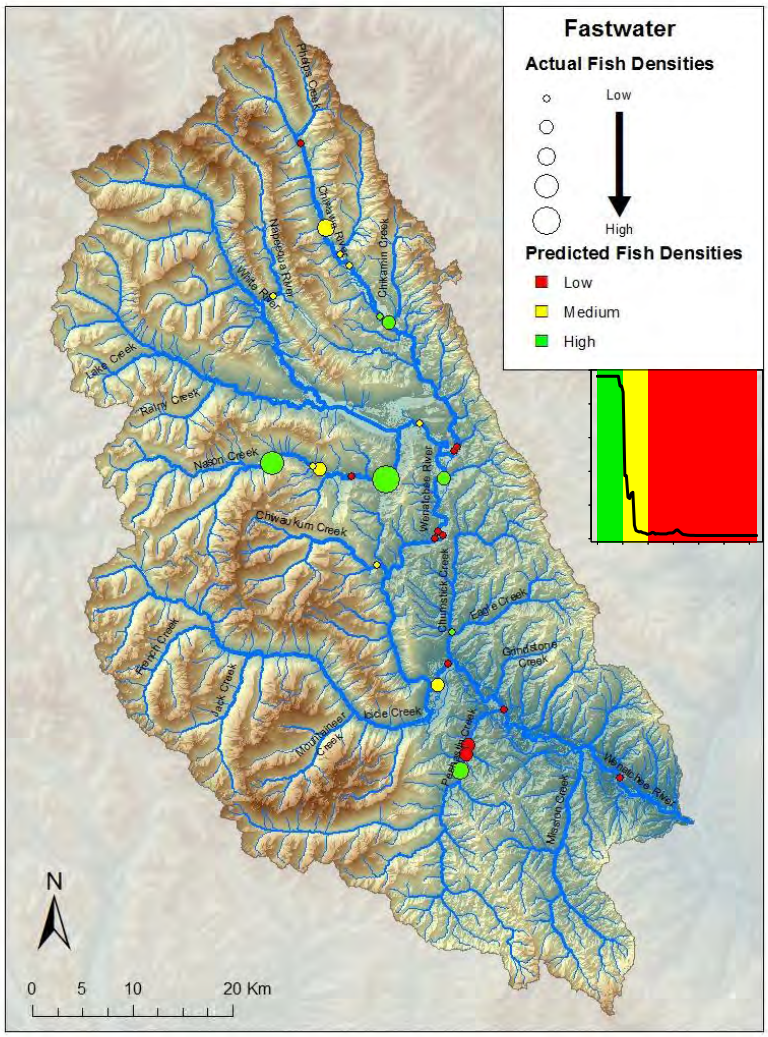



Figure 23. Observed juvenile Chinook densities, averaged across years, and the predicted densities, based on the amount of fast water at a site. The inset plot shows the relationship between fast water (x-axis) and predicted fish density (y-axis). Less fast water predicts high fish densities (green), more fast water predicts low fish densities (red).

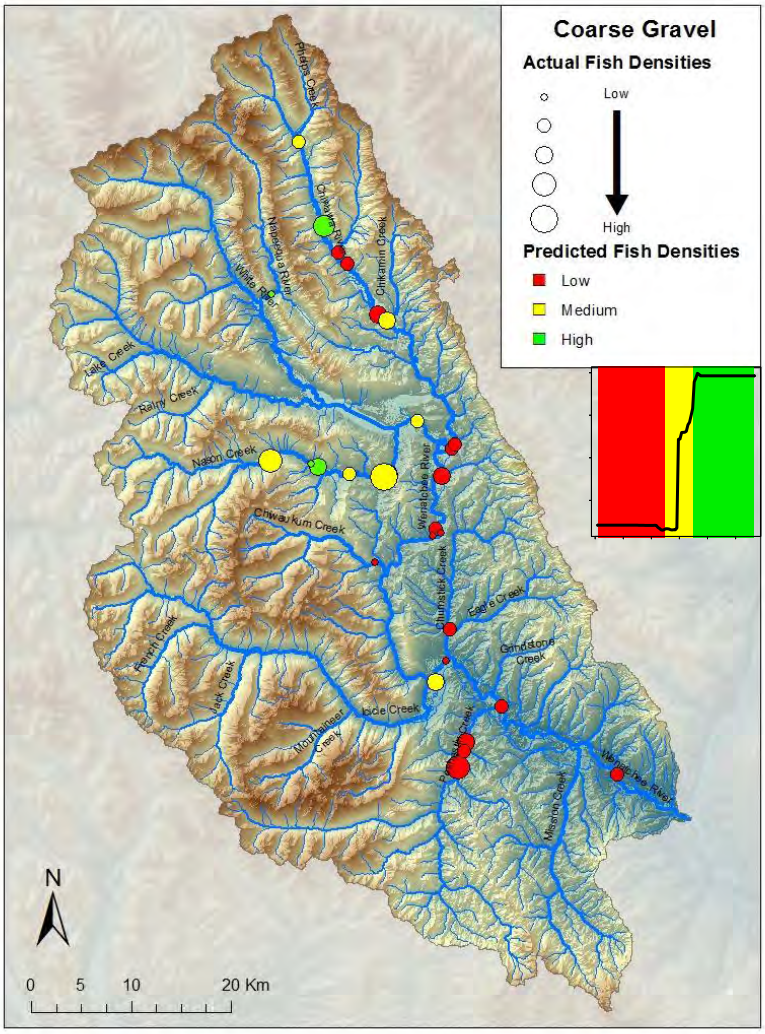



Figure 24. Observed juvenile Chinook densities, averaged across years, and the predicted densities, based on the percentage of coarse gravel at a site. Larger circles correspond to higher observed densities. The color corresponds to the predicted density. The small inset plot shows the relationship between coarse (x-axis) and predicted fish density (y-axis). Sites with low percentages of coarse gravel are predicted to have low fish densities (red area), and sites with high percentages of coarse gravel are predicted to have high fish densities (green area).

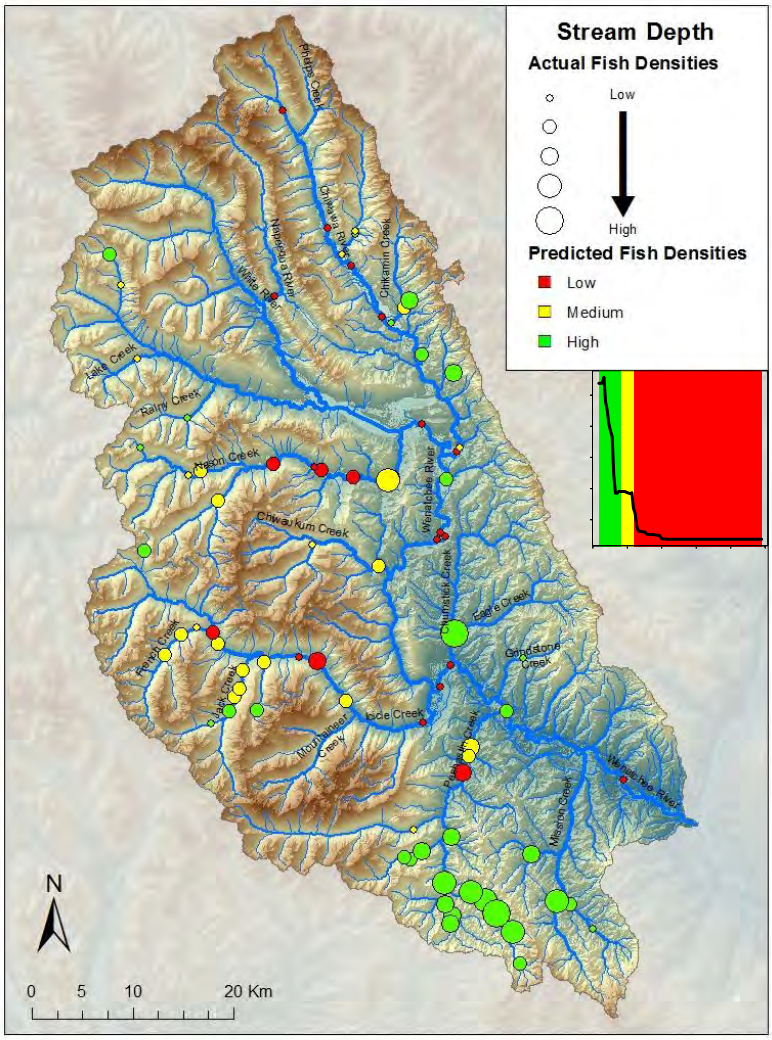



Figure 25. Observed juvenile steelhead densities, averaged across years, and the predicted densities, based on the average stream depth at a site. Larger circles correspond to higher observed densities. The color corresponds to the predicted density. The small inset plot shows the relationship between stream depth on the x-axis and predicted fish density on the y-axis. Sites with shallow stream depths are predicted to have high fish densities (green area), and sites with deeper stream depths are predicted to have low fish densities (red area).

When compared with the observed densities, the predictions based on a single habitat metric match the observed data fairly well. The predictions based on the entire suite of habitat metrics match even more closely. Steelhead may be more adaptable to local conditions, making it more difficult to predict their densities from a single habitat measure, as seen in Figure 25.

## Juvenile Salmonid Growth Lemhi ISEMP and CHaMP Data

If aspects of a restoration effort are expected to influence factors that control growth (e.g., temperature or primary productivity) then growth could be a metric used to measure habitat restoration effectiveness. ISEMP has collected growth data in the Secesh and Lemhi from 2009-2011 and the analysis below shows that it is possible to measure growth well enough to see differences among and within watersheds.

**Fish—Habitat Relationships: Growth**
Marking and recapturing rearing juvenile salmonids allows the estimation of growth rates. ISEMP results show that growth rates differ between watersheds. If growth rates can be changed with restoration, these tools will help us detect and demonstrate the change.

The Secesh River in Idaho is a largely pristine watershed relative to the Lemhi, with colder water, higher gradient, and a substantially better developed canopy than most of the Lemhi River. In contrast, the Lemhi, although heavily disturbed, is spring-fed, highly productive, and generally maintains a more moderate temperature regime. Consequently, we would expect growth rates to be substantially lower in the Secesh even under ideal conditions. As can be seen in Figures 26 and 27, monitoring was indeed able to detect a difference in growth rates between and within watersheds: Chinook grow slower in the Secesh mainstem and tributaries compared to the Lemhi and its tributaries including Hayden Creek (Figure 26).

Ongoing ISEMP research is working to link differences in growth to finer-scale resolution in habitats (e.g., within subbasins) to better focus where and how to implement habitat actions.

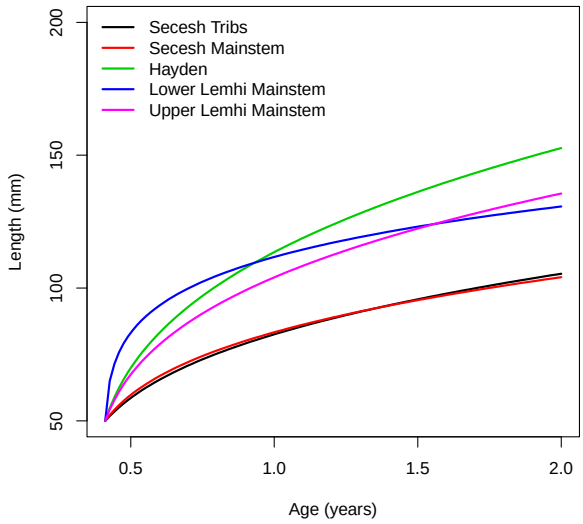



Figure 26. Fitted growth curves using mark-recapture data of Chinook from the Salmon River basin 2009-2011.

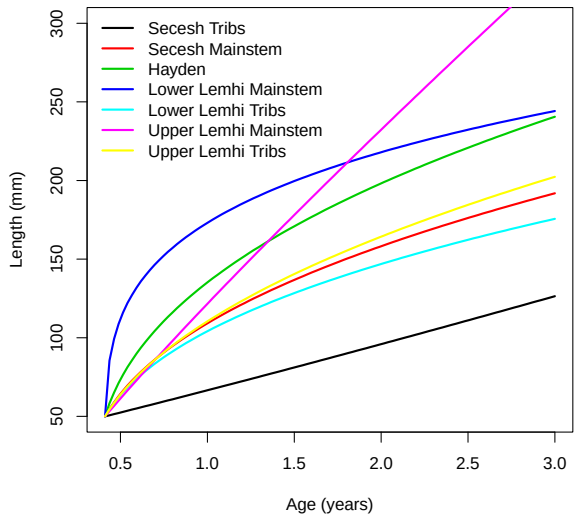



Figure 27. Fitted growth curves using mark-recapture data of steelhead from the Salmon River basin 2009-2011.

## Growth Potential Models: Synthesis of the Benefits of the Middle Fork IMW Study.

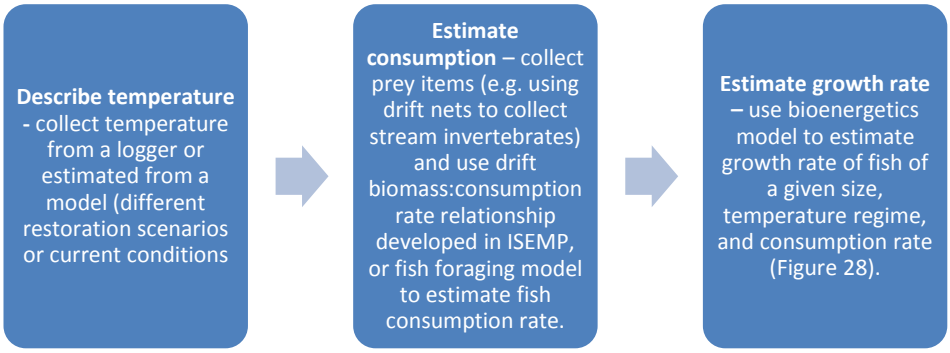
In ISEMP, we have developed a simple model to estimate juvenile steelhead growth potential of a stream or stream reach based on water temperature and biomass of drifting invertebrates.

**Growth Potential Models**
These ISEMP tools use stream temperature, measures of fish food and a bioenergetics framework to show where restoration actions may be able to relax limitations on fish growth and survival.

Temperature is a focus of the ISEMP John Day Pilot project because temperature is an integrative response across multiple external and internal stream factors, it is sensitive to multiple human disturbances, and is crucial in influencing salmonid production.

Several models estimate stream temperature, including a model developed by ISEMP to estimate temperature throughout the John Day and Wenatchee (see Habitat impairments temp model section). Another model, Heat Source, is used throughout the state of Oregon by the Department of Environmental Quality (ODEQ) to complete their TMDL process. This model estimates temperature over several scenarios, including current conditions, natural historic conditions, and the effects of different stream restoration alternatives such as actions implemented in the Middle Fork John Day IMW.

We have combined our growth potential model with these temperature models to assess limiting factors and expected benefits to stream restoration for the Middle Fork IMW. The basic work flow is shown below:




The models synthesize the expected benefits of restoration to the stream channel, riparian area, and water use that are translated to changes in the thermal environment (the restoration focus), and the resulting changes in the performance of the steelhead population. The growth potential model performed well in the watersheds where it was developed (as part of ISEMP's Bridge Creek IMW) (Figure 28). The growth potential model suggested that improvements due to the Middle Fork IMW would increase growth rates.

Because of the inability to replicate IMWs, a mechanistic understanding of the casual relationship between restoration and fish response, such as this approach provides, is required to extrapolate the lessons learned from a watershed restoration approach to other systems.

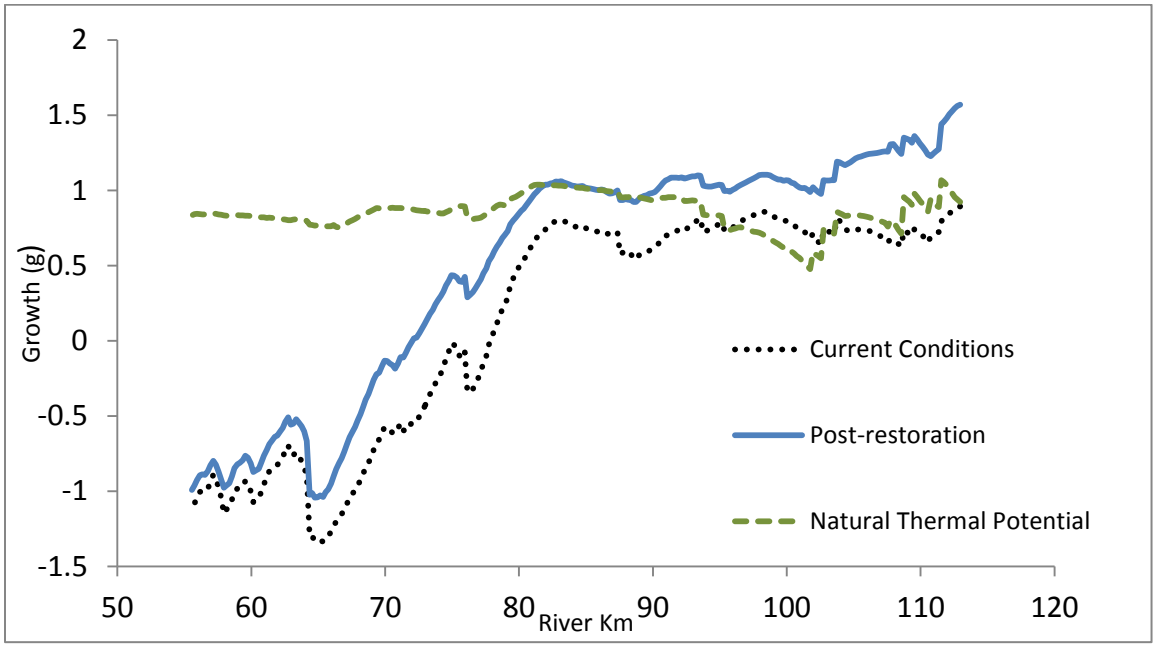



Figure 28. Growth (g) potential of a 20 g *O. mykiss* between July 1-Aug 15 (2002), for current thermal conditions (black dotted line), post-restoration as planned by the Middle Fork IMW study (blue solid line), and under natural thermal potential (green dashed line) for the upper Middle Fork John Day River.

## Estimating Energy Availability and Carrying Capacity of Salmonids in a Stream Reach

While the growth potential model described above highlights the importance of temperature and prey availability, it completely ignores the importance of physical structure in streams (e.g., pools, riffles, gradient) in driving salmonid production. Quantifying physical structure is a large emphasis of habitat monitoring protocols such as CHaMP. In ISEMP we are attempting to incorporate the latest developments in fish foraging models to estimate energy intake and carry capacity, with the CHaMP protocol customized to provide data inputs for these model. We expect these model results to be used directly as input into life-cycle models that will likely be used in regional population assessments.

**Carrying Capacity Prediction Model**
Multiple CHaMP metrics may be used to predict the reach-level costs and benefits to fish, eventually allowing us to answer the question "What is the carrying capacity of a reach and how might it change with restoration activity?"

The mechanistic model we are using to represent how a fish makes a living in a reach incorporates how water flows through the reach (hydraulic model), how food is delivered throughout the reach (drift transport model), how fish capture drifting prey (foraging model) and expend energy in the process (water velocity) (Figure 29). The net rate of energy intake (NREI) of salmonids is the difference in the energy gained from foraging and energy lost through swimming. Thus, NREI can be converted into growth rates of salmonids and the model can map areas of a reach where fish have positive NREI (Step 6 of Figure 29). The number of foraging areas that have a positive NREI can serve as an estimate of carrying capacity of the reach (Step 7 of Figure 29).

We used CHaMP surveys to provide inputs to the models which includes: temperature, discharge, invertebrate drift, the digital elevation model (digital 3D map of the channel), and channel unit substrate type. We used fish collected in these reaches to validate how well the model might predict steelhead abundance in seven stream reaches. Although very preliminary, the model predicted the number of fish extremely well, and thus we are hopeful that once fully developed the model will be very informative in translating CHaMP style surveys to metrics that describe fish performance and abundance.

The model can also be used to estimate how changes to stream channel can translate into changes in NREI and carrying capacity much like the way DEMs can be used to evaluate changes in stream topography (see Figure 15). We conducted a CHaMP survey at a site within the Asotin IMW and then altered the DEM to reflect the expected changes due to the proposed action of wood additions. We can subtract the pre-treatment NREI surface from the post-treatment surface to create an NREI difference surface that intuitively explains how the restoration could potentially create more fish (Figure 30).

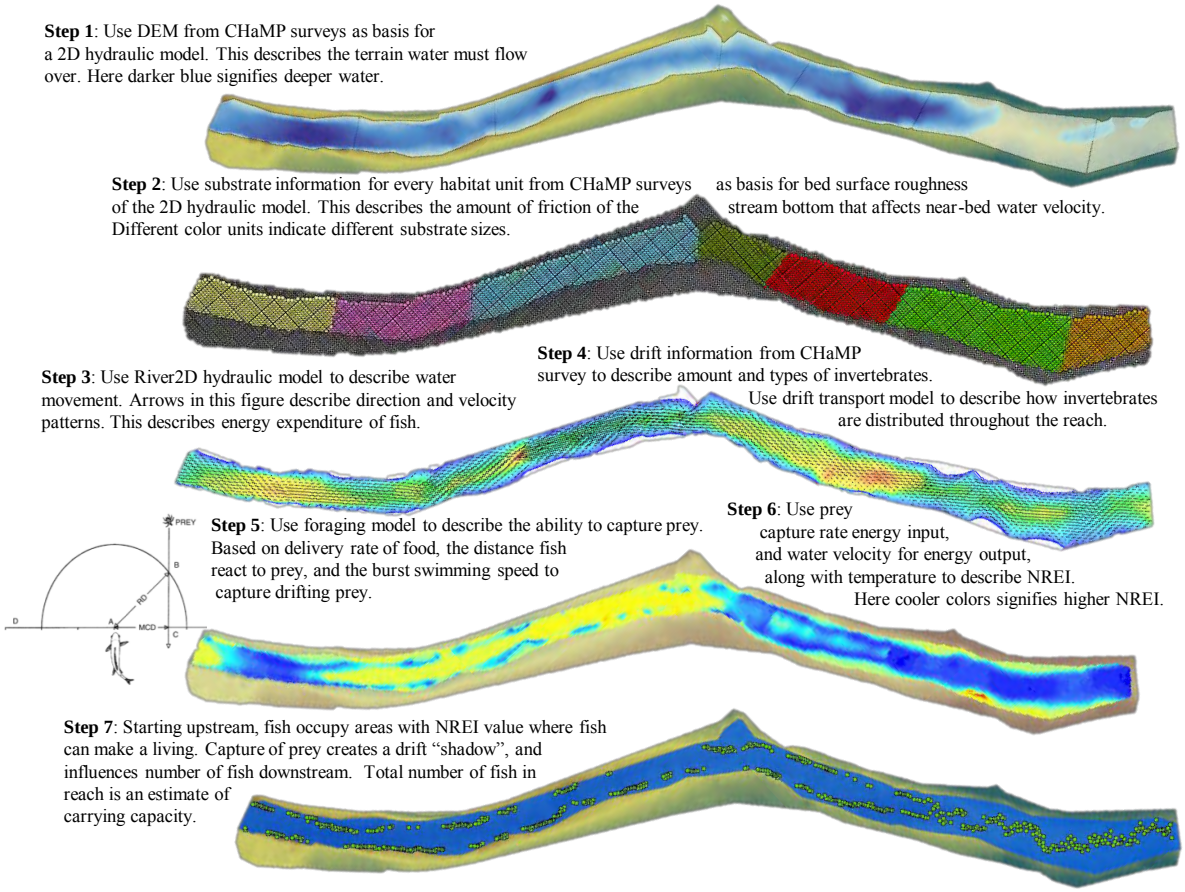



Figure 29. Estimating energy available (net rate of energy intake or NREI) and carrying capacity of juvenile steelhead in a stream reach.

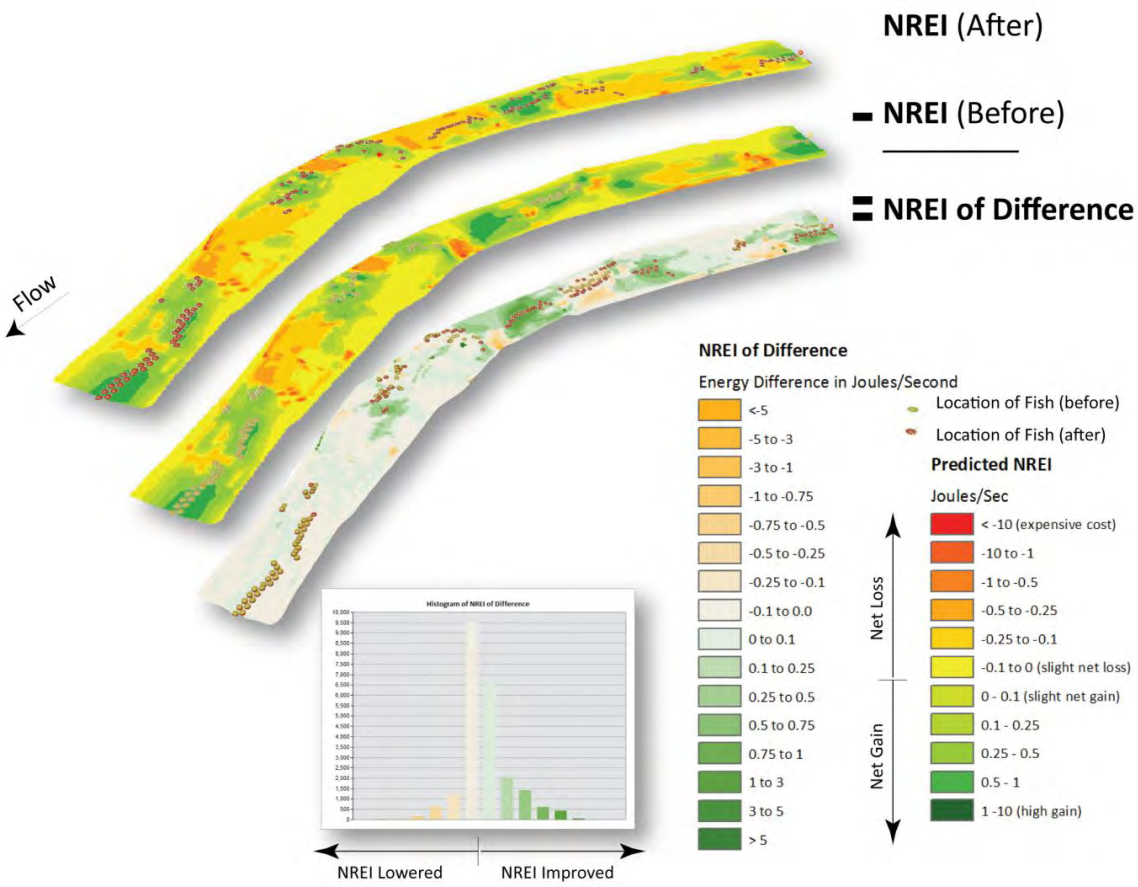



Figure 30. The energy available (net rate of energy intake or NREI; the colored surface) and abundance (dots represent placement of fish) pre-treatment (Before) and hypothetical post-treatment (creation of pools via wood additions) in a reach of the South Fork of the Asotin. If NREI (Before) is subtracted from NREI (After) for each pixel that has an XY coordinate, another surface is created that spatially describes the change in energy available and carrying capacity of the reach due to restoration.

# CHAPTER IV: CONCLUSIONS

## Next Steps for the FCRPS BO Habitat Strategy: Fill the Gaps

In light of the December 2011 commitment by policy makers to provide a forum that guides the development and implementation of an analytical framework, ISEMP reaffirms its commitments to assisting the policy makers with the underlying technical tasks within our purview. The next step, therefore, in the FCRPS BO habitat strategy is to convene and structure the forum and set to work on the development of the analytical framework. Other steps, however, that are outside of ISEMP's purview, are also necessary and have direct bearing on the success of the framework necessary to answer the key management questions.

**Analytical Framework**
The list of ISEMP decision support products is obviously lacking a structure. ISEMP provides a suite of technical tools but is relying on the participation of policy managers within the forum setting to determine the best way to employ these tools in decision making.

### *Action Prioritization*

One major piece that needs policy work, with technical support, is deciding "How are restoration actions prioritized?" Conceptually, it makes sense that a strategic approach to implementing restoration actions and allocating effort is more likely to achieve the desired outcomes than is the current opportunistic approach to action implementation. Putting this into practice will require 1) understanding technical nuances, 2) balancing short term gains with long term priorities, and 3) choosing between competing projects.

An example of a technical nuance that is not accounted for under the current approach to allocating restoration actions can be seen in ISEMP's research in the Lemhi. There, apparent declines in productivity due, presumably, to local density dependence effects suggests we are already encountering situations where addressing one limiting factor (e.g., reconnecting tributaries to the stream network) immediately elevates the next limiting factor (e.g., the need to increase habitat capacity for fish production) to stand in the way of recovery. Removing one limiting factor only to have it replaced by another is, ironically, an inherent, albeit frustrating, feature of "progress" on the road to recovery. This will become increasingly evident the more successful we become at implementing effective actions, and therefore, needs to be built into the analytical framework.

Short-term gains may not be as important in the effort to move toward recovery as are some long-term priorities but an explicit way to balance between the two is not built into the current habitat strategy. As explained in Chapter 2, the experimental approach of IMWs is ultimately critical to restoration effectiveness (because experimentation is necessary to achieve certainty) but IMWs are expensive investments and are rare opportunities. Meanwhile, the very real pressure to implement actions wherever possible can be devastating within IMW watersheds if that implementation disrupts, or even ruins, the complicated design of the IMW. The importance of learning lessons from IMWs that can be exported to many other watersheds may, at times, outweigh local benefits from delayed or foregone restoration actions.

Finally, choosing between competing projects is, generally, not currently done in a strategic fashion. The current habitat strategy does not adequately describe how to compare the relative merits of projects, say, on a tributary to the Methow versus a tributary to the Upper Grande Ronde. One project may be clearly more cost-effective but no clear decision-making mechanism now exists with which to objectively choose between projects. Fortunately, there are local examples of approaches that could inform improvements to the current habitat strategy. For example, the implementation strategy in the Upper Columbia, guided by the Upper Columbia Biological Strategy, has developed an approach for gaging the relative importance of projects within and among subbasins.

In summary, the analytical framework that we are advocating is not merely a tool useful for assigning credit for restoration actions or for quantifying the results. It is, more importantly, a decision-making tool for prioritizing which projects to implement and knowing when to stop. In the end, the answer to the fundamental management question may depend more on "how are restoration actions prioritized" than on "how are projects being monitored."

### *Using Monitoring Information*

Another critical gap in the current habitat strategy that would be filled with a robust analytical framework is deciding "How monitoring information is going to be used?" Scientists can easily provide more information than could ever be used: it is not merely a question of designing better studies. Decision makers also need to be more explicit about their decision points: what is the currency of the decision, what are the magnitudes of the desired effect, what are the trade-offs and risks of making the wrong decision. When concepts like these become clearer, it becomes much easier to define and explain how monitoring information will be used.

### *ISEMP's Role*

ISEMP's role to date stems from our original proposal: since then, we have consistently plugged along at the tasks we laid out for ourselves in 2003, within the context of a "pilot" approach. Now, as we are reaching a point where lessons-learned can serve other watersheds and be built into decision-making by the management community, some aspects of ISEMP's role may need to be reaffirmed while other aspects may need to be redefined.

The primary part of ISEMP's original mission that must be reaffirmed and completed is the development of spatially explicit fish-habitat relationships that link to fish productivity. To do this, the IMWs must run their course, and ISEMP must finalize development of PIT tag array work in order to explicitly quantify fish productivity, and complete research that will streamline the collection of habitat data within CHaMP.

ISEMP's role could mature in certain ways. ISEMP's mission of developing monitoring methods for export to users in the Columbia Basin logically supports working within the proposed forum to aid in the development of the analytical framework. ISEMP could also synthesize fish and habitat information at the regional scale, something that is not within ISEMP's current scope but that remains a gap in the regional context.

Some changes to ISEMP's role might be inappropriate for a variety of reasons. For example, ISEMP should not be asked to make decisions about where restoration actions should be performed or which actions to perform or other similar decisions that include value

judgments. There is a thin line between informing policy decisions with technical understanding and making policy decisions: policy makers need to be responsible for the latter. More importantly, though, ISEMP should not become a monolithic data collection enterprise. Co-managers have the purview and experience best suited for large-scale and disbursed data collection. Again, it is logical for ISEMP to discover and define and standardize the appropriate metrics and methods but it is up to others to eventually perform the monitoring.

## Next Steps for ISEMP: Incorporating Lessons Learned

### *IMW Designs*

All results presented in Chapter III are legitimate and based on actual analyses conducted by ISEMP. However, they are preliminary and more work is necessary to determine how generally and broadly applicable these results are. Like all adaptive management tools, future application and research will improve their utility. The following sections describe major areas where we will be working to refine these decision support products.

Habitat restoration has been ongoing in the Entiat River subbasin, WA, since the 1990s at a rate of about three projects per year, creating a timeline of up to 25 years for all the habitat restoration projects to be implemented. From 2005-2009, ISEMP had been estimating fish abundance indices using snorkel surveys using a Before-After-Control-Impact design (BACI, monitoring occurs at sites before and after treatment is implemented and at control sites) as recommended in the Upper Columbia Monitoring Strategy to determine what effects tributary habitat actions have on fish populations and habitat conditions. In this example, ISEMP evaluated a study design's ability to detect change and suggested an alternative design better suited to detecting the change of interest.

A power analysis to determine if, under the current implementation timeline and scope, monitoring would be able to detect a significant effect of habitat restoration actions on steelhead density found that there may be a relatively large treatment effect. However, the mixed responses (+ and -) and large confidence intervals in snorkel counts precluded conclusion of a significant effect. Modeling showed that it would take 262 habitat restoration sites to attain 80% power, significantly more than is planned for the Entiat River. It was concluded that the power to detect a change based on this relatively simple design would likely be low. This exemplifies the problems with detecting changes in fish populations without rigorous experimental designs and large sample sizes or long time series.

As a result of this analysis, ISEMP proposed an IMW approach to restoration in the Entiat. An IMW is a watershed-scale coordinated restoration effort with an associated effectiveness monitoring program implemented in an experimental fashion to maximize the ability to detect fish responses to changes in their habitat. Such an approach seeks to maximize contrast to increase the ability to detect an effect. In addition, intensive monitoring is used to identify mechanisms by which habitat manipulations impact fish, so that these strategies can be extrapolated to other systems. As such, an IMW is a powerful approach to answer cause-and-effect questions at the watershed or population scale relevant to management.

In the Entiat IMW ISEMP is utilizing a hybrid hierarchical/staircase statistical design to guide implementation to compare treatment and control sections within the Entiat River

subbasin. Under the IMW, habitat restoration actions are implemented in a spatially and temporally explicit manner to maximize the magnitude of the actions' effect, i.e., rather than treatments occurring at a rate of one or two a year spread across the entire 26 miles of the treatment area, actions are concentrated in a given area in a given year. This is the staircase aspect of the design. The hierarchy aspect of the design is based on temporal and spatial separation of actions, i.e., actions are implemented in different valley segments of the river over time, e.g., actions implemented in the upper river first and then in the lower river.

There are several advantages to using a staircase design. First, the staggering of the treatments over time allows for the distinction between the random effects of year and year by treatment interactions. This prevents random initial environmental condition (e.g., drought or high water year) from having an overriding effect on the ability of the experiment to detect true treatment effects. Second, by staggering treatments within the treatment area, treatment sections can be used as controls until they are treated, guarding against the loss of other control areas. Third, it is uncertain to which degree restoration actions may impact downstream reaches. A comparison of multiple reaches within a single watershed may be more powerful because of a greater number of replicates and the ability to accurately describe a reach versus a watershed or subbasin; however, these sites may not be independent from each other.

A nested hierarchical approach is appropriate when the scale of impact is unclear. Our monitoring design will directly address the issue of fish movement at the reach and watershed scale by comparing control and treatment sites within a hierarchy of progressively finer spatial scales consisting of watershed, valley segment, and reach scales. We will evaluate the degree of variability and statistical power associated with each scale. The latter two points will provide insight into the scale at which future restoration actions should be monitored. This hierarchical aspect of our statistical design will also lend itself to the testing and development of causal relationships. These relationships include fish-habitat relationships, relationships between instream characteristics, and relationships between landscapes, habitat, and fish, and thus require multi-scale information. This multi-scale approach will be robust and flexible enough to account for range of responses we are likely to observe.

## *Fish Monitoring Methodology and Tools*

One of the primary goals of ISEMP is the development and implementation of standardized methods for data collection, data management, and data reduction across the Columbia Basin. A standardized protocol that guides the distribution and implementation of remote site juvenile capture and tagging protocols has been developed and applied across all three ISEMP subbasins. This protocol is supported by an electronic data capture device (Juniper Systems Allegro Mx) and associated program developed specifically to support the protocol and populate a database. Data produced by remote site juvenile capture and tagging are reported to regional databases.

Fish-habitat relationships have traditionally been studied through direct observation of fish in their natural environments. This traditional "snorkeling" approach can be useful at fine scales or for determining the distribution of fish at larger scales but snorkeling is limited in many ways. In common practice, snorkeling is poor at generating populations estimates compared to other more rigorous methods like mark-recapture techniques. More importantly, however, snorkeling affords no opportunity to quantify growth or survival, both aspects of productivity

that must be understood to answer fundamental management questions. The ability to identify and track individual fish is necessary, hence the switch in ISEMP from snorkeling to PIT tagging. This is an example of ISEMP's evaluation of a method and the resulting method adjustment that is likely to improve data collection and meet analytical needs and objectives.

### *CHaMP*

One of the most important deliverables to date for ISEMP was the habitat monitoring protocol and data stream management tools that has become the Columbia Habitat Monitoring Program (CHaMP). The development of measures of stream habitat quality and quantity that were feasible, repeatable and information rich – in terms of predicting fish population processes – was an initial goal of ISEMP. Starting with a suite of off-the-shelf stream habitat monitoring protocols ISEMP and its partners explored these methods limitations and developed alternatives specifically targeted to the fundamental management question, " do stream habitat restoration actions benefit salmonid populations?. The development of the CHaMP methods and project implementation procedures is one of the best examples of ISEMP learning from its work and an excellent example of the project's ability to export products to the resource management community. The CHaMP program is in the midst of an aggressive adaptive development process. The recent CHaMP "lessons-learned" workshop produced many results that will feedback to CHaMP and ISEMP designs. For example, fish-habitat relationships uncovered by ISEMP's early habitat work in ISEMP's three pilot-watersheds will be refined and tested at more regional scales as CHaMP data is accumulated.

## Timelines for Existing Tasks

### *Entiat IMW*

The Entiat IMW has been underway since 2010, with the first round of habitat implementation projects scheduled for 2012. By 2013, process-based large-scale restoration actions will have been implemented in three reaches in the upper river (Figure 31) and fish and habitat effectiveness monitoring under the IMW will have been ongoing for 4 years: 3 years pre-treatment and 1 year post-treatment data. By 2017, large-scale restoration actions will have been implemented in the lower 7 miles of the river in 2014. In 2017, the last round of habitat restoration actions will be implemented in the upper river. By 2020 the remaining habitat restoration actions will be implemented in the lower river, resulting in the restoration of the lower 26 miles of the Entiat River in a 10-year time period using a rigorous experimental design with associated effectiveness monitoring.

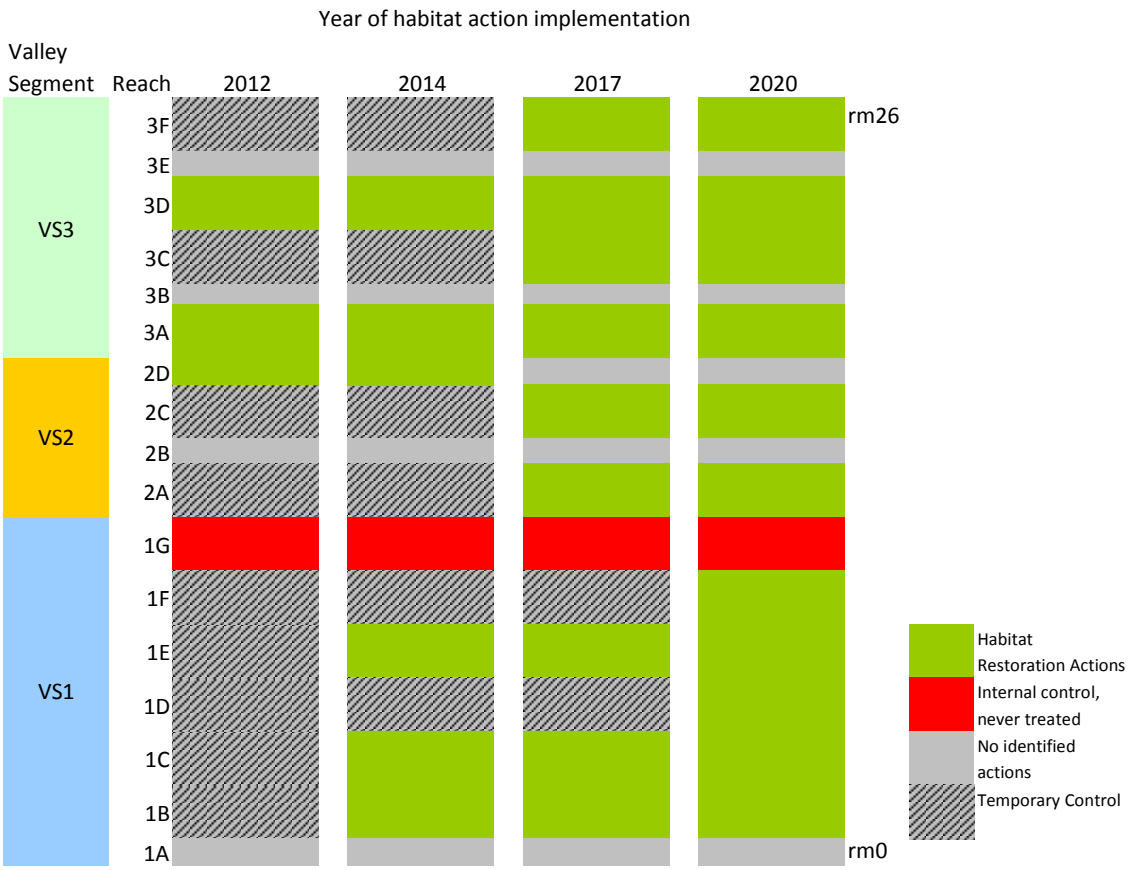



Figure 31. Schedule of habitat restoration actions in the Entiat IMW.

## Instream PIT Tag Detection Arrays Technology

ISEMP relies heavily on juvenile and adult PIT tagging and interrogation at instream PIT tag arrays. The development of instream PIT tag interrogation technology represents a significant advancement with regard to the estimation of juvenile and adult distribution and survival. However, the efficient use of this technology requires significant data management support to cost-effectively enable data retrieval, ensure data quality, and enable efficient and meaningful access to data.

PIT tag arrays are commonly located in remote areas with limited access, thus it is not cost-efficient or even possible, in some cases, to visit sites for the purpose of downloading interrogation data. Beginning in 2009, Biomark, Inc. and ISEMP began development of a standardized suite of PIT tag array infrastructure enabling reliable remote downloading of interrogation data and routine site diagnostics. These efforts also enabled the development of software that automatically parses downloaded PIT tag array data, reduces the data to required fields in PTAGIS, and uploads the data automatically to the PTAGIS database.

The data generated from a PIT tag array are similar to data collected by many other survey types. Generally, the data can be summarized as a unique tag code, date, time, location of interrogation, and various attributes. However, PIT tag arrays also produce diagnostic data such

as noise by antenna, enabling a variety of analyses such as estimation of instantaneous read range. Similarly, given the orientation of antennas in a single array, or orientation of multiple arrays, it is possible to calculate array efficiency and direction of fish movement. The volume of data produced by PIT tag arrays as well as the desire to conduct analyses aimed at estimating array efficiency and/or fish directionality requires data storage and query resources that are not currently well supported by regional databases. Similarly the analyses generated using PIT tag array interrogations require explicit knowledge regarding the quality of data (e.g., the ability to determine when arrays are not functioning properly).

The emphasis on tributary instream PIT tag arrays by Bonneville Power Administration and associated collaborating agencies led PTAGIS and PSMFC to recognize the need to provide a formal process to identify data storage and query needs across the Columbia River Basin. ISEMP is currently assisting PTAGIS and Pacific States Marine Fisheries Commission (PSMFC) staff in identifying data management needs for instream PIT tag arrays that are not supported by the current PTAGIS database. During the January 2011 PTAGIS PIT Tag Steering Committee annual meeting the Committee requested that scientists with experience in the development and use of these systems form an ad hoc subcommittee to provide recommendations that would allow PTAGIS to fully support the data storage needs specific to in-stream interrogation sites. Given the substantial PIT tag array infrastructure operated within ISEMP, ISEMP personnel were identified to lead the newly formed ad-hoc Instream PIT Tag Subcommittee. The subcommittee began monthly meetings starting in April 2011 and produced a formal suite of database requirements in the fall of 2011.

## Fish-Habitat Modeling

One of the major goals for ISEMP is to design an analytical framework that can take habitat data and make predictions about salmonid survival and abundance. Since this has never been done before, ISEMP has developed several different approaches to address this goal. Correlative models such as boosted regression trees generate predictions about fish from habitat conditions without making any assumptions about biologically realistic linkages between habitat and fish. The watershed production model is a biologically mechanistic description of how various habitat characteristics impact fish at a population level. Bioenergetics and growth models link inputs of temperature and food to individual fish growth, which can then be tied to energy expenditure to determine the carrying capacity of a particular area. All of these approaches can identify habitat factors that are limiting fish productivity, and predict the fish response to specific restoration actions. These distinct analytical frameworks have been developed, and further data collection will help to refine their results.

By 2013, we will have collected two years of CHaMP habitat data, together with fish abundance estimates, from across the Columbia basin. This spatial contrast will help inform all three analytical frameworks and we will have enough data to use those approaches to start making predictions as to what the most important habitat characteristics are for fish production and what the limiting factors may be in certain locations. By 2017, we will have six years of habitat and fish data, which will provide much better temporal contrast, which will allow us to separate the effect of particular habitat measures from annual effects such as the number of spawners or environmental conditions like precipitation.

## Watershed Models

The primary products for ISEMP implementation in the Salmon subbasin (Lemhi and South Fork Salmon River (SFSR)) are fully populated watershed models that serve as decision support tools with broad application across the Columbia River Basin. For both subbasins, the watershed models will be fully populated and developed in time to support the 2013 comprehensive check-in for the BiOp. Similarly for both steelhead and spring/summer Chinook salmon the watershed models will be sufficiently populated in time to inform the 2017 evaluation of the BiOp and to evaluate whether habitat restoration actions implemented in the Lemhi River were sufficient to achieve the 20% improvement in freshwater productivity identified in the BiOp.

# APPENDIX: ANALYSIS UNDERLYING ISEMP DECISION SUPPORT TOOLS

# CHAPTER 1: Habitat Status and Trends Monitoring

We all recognize that fish and habitat conditions are spatially and temporally variable, and that our ability to measure important aspects of fish populations and habitat in streams is not perfect. One of the underlying and often not explicitly stated objectives of any monitoring program is to describe this spatial and temporal variability and to evaluate how much uncertainty our measurements might introduce to these descriptions.

**Status:** In the ISEMP and CHaMP context, the phrase "habitat status and trends monitoring" generally refers to obtaining a snapshot of habitat conditions and patterns of change across stream networks. These networks may vary in size from those in small watersheds to those across the entire set of watersheds in the Columbia Basin. Status refers to a snapshot during particular time intervals, such as: what is the status of habitat in CHaMP or ISEMP watersheds during 2011's low flow summer season? Survey designs that incorporate randomization in the selection of monitoring sites, as has been incorporated into CHaMP and ISEMP habitat monitoring, allow inferences across the domain of interest from the sample of monitored sites. Frequency distributions are often used to summarize the set of data from which statistics such as the mean, median, various percentiles and expressions of variability are derived. Graphical approaches are used to display spatial patterns or similarities or differences among groups. For example, Figure 1.1 illustrates, through the use of "boxplots", one easily explained way of summarizing and comparing habitat conditions among three CHaMP watersheds based on the 2011 habitat surveys. Visualizing boxplots side by side allows approximate inferences about whether habitat differs among the watersheds (to be verified by appropriate statistical tests). For example, for the habitat attribute "fraction of sediment particles < 2 millimeters in diameter", there is no overlap between the John Day box and the Lemhi/UGR boxes indicating the John Day's distinctness from the other two. As well, the near overlap of the boxes for the Lemhi and UGR indicate similar fine sediment condition in these two watersheds. Part of a monitoring program's documentation includes data summaries such as these, along with the data files for each habitat metric for each watershed, for each sampling interval. Of general interest are interpretations of these summaries, such as relationships among watersheds, between fish and habitat condition, the identification of patterns in good or poor condition or the achievement of particular restoration criteria.

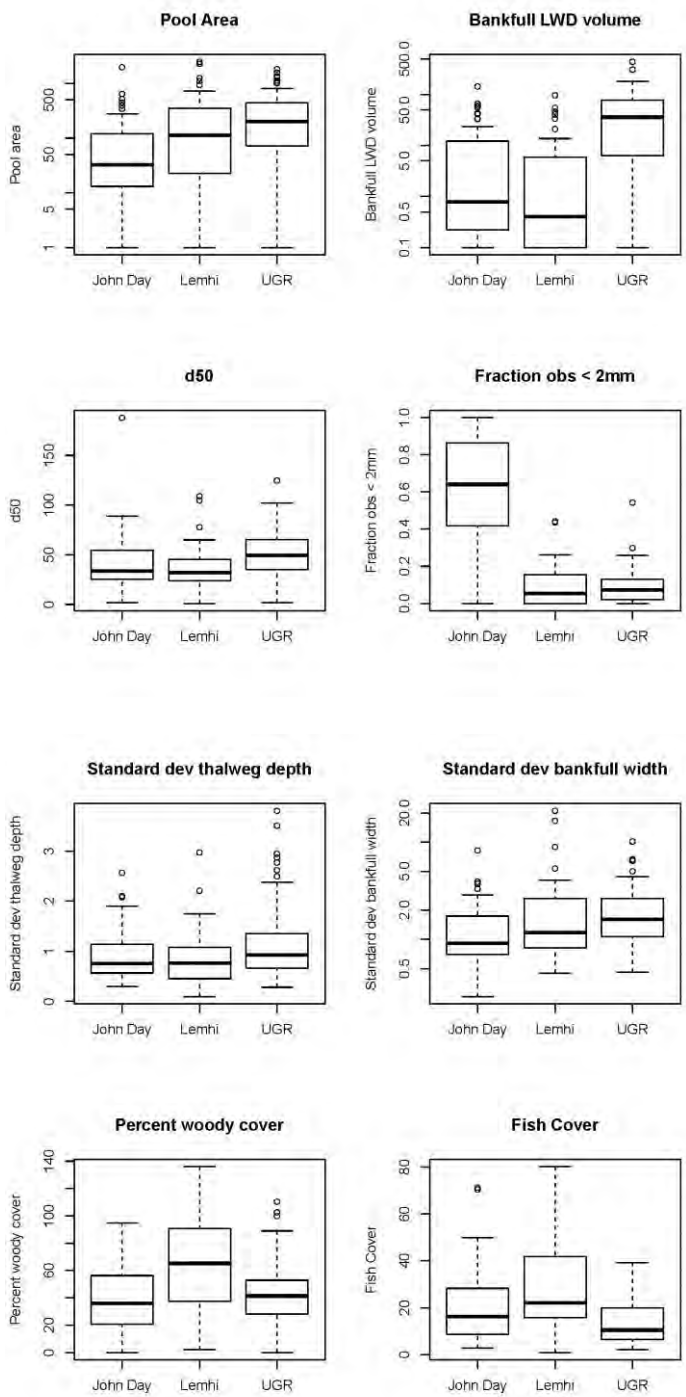



Figure1.1. Boxplots are convenient summaries of data distributions derived from sample surveys, illustrated here for a variety of habitat metrics in the John Day, Lemhi and Upper Grande Ronde (UGR) watersheds. The figures include: the median (dark horizontal bars), the range within which 50 % of the observations fall (the boxes enclose the interquartile range, the data falling between the 25th and 75 percentile of the frequency distribution), the range

within which 80 % of the observations fall (the data falling between the whiskers, demarking the 10th and 90th percentiles), and any outliers.

**Trend:** The trend part of the phrase "status and trends monitoring" belies substantial complexity in what we mean by trend. In general, we might think of "patterns of change over time" usually with respect to change across years. Urquhart (Urquhart, et al. 1998; Urquhart and Kincaid 1999) argues that any pattern of change with a consistent upward or downward component (i.e., not just a cyclical pattern with no underlying changes over time) can be evaluated or detected as a linear component: the complex pattern would be superimposed upon the underlying linear change. He and colleagues have evaluated monitoring designs aimed at balancing the needs for good status estimation (monitoring more sites is better) and trend detection (revisiting the same sites is better). For example, monitoring a set of sites every year is best for trend detection. But monitoring different sites every year is best for estimating the status of the resource. Urquhart and Kincaid (1999) conducted a variety of simulation studies that support designs consisting of a set of panels (a panel consists of a set of sites with the same temporal sampling pattern, e.g., an annual panel of sites (monitored each year), three panels of sites each monitored on a three year cycle. Although the power for an annual panel to detect linear change is most sensitive during the early years of a monitoring program, the power the non-annual panels to detect linear trend catches up with the annual panel design after sites have been sampled three times. ISEMP and CHaMP monitoring programs incorporate panel designs that include an annual panel and either a set of three year panels (CHaMP), or a mixture of an annual panel, a random panel, and panels on a three-year cycle.

Trend can be expressed as an underlying 'average' trend across all sites in a region: is habitat condition changing in the domain of interest? Or, trend might be expressed as the 'status' of site specific trends, i.e., for each site (after at least three visits to the site), a site specific linear regression of the metric of interest with respect to years can be fit. The distribution of these trends constitutes a "status" estimate over, for example, a nine year period. Figures 1.2 and 1.3 illustrate one example of the estimation of regional (Figure 1.2) and site specific (Figure 1.3) trends for one habitat attribute (bankfull depth) monitored in the Wenatchee watershed covering the years 2004 – 2009. CHaMP's 9-year monitoring design is intended to allow powerful trend estimation after the completion of 3 three year cycles (i.e., each site will have been sampled at least three times). Continued monitoring increases the power to detect subtle habitat trends.

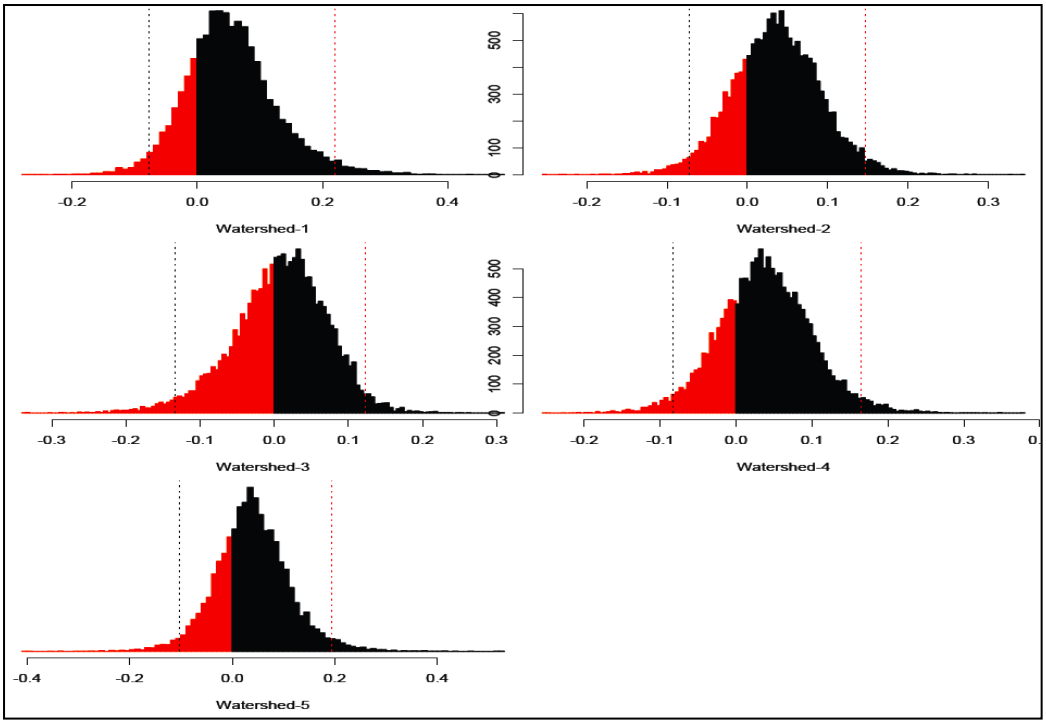



Figure 1.2. A Bayesian trend analysis evaluating average trend in each of five sub-watersheds in the Wenatchee basin reveal the likelihood that an average trend is either positive or negative. Color coding reveals the probability that a negative (red) or positive (black) trend is detectable in each of the watersheds. There is evidence for a positive trend in bankfull depth in four of the watersheds, but not for one of them (watershed-3), based on a visual inspection.

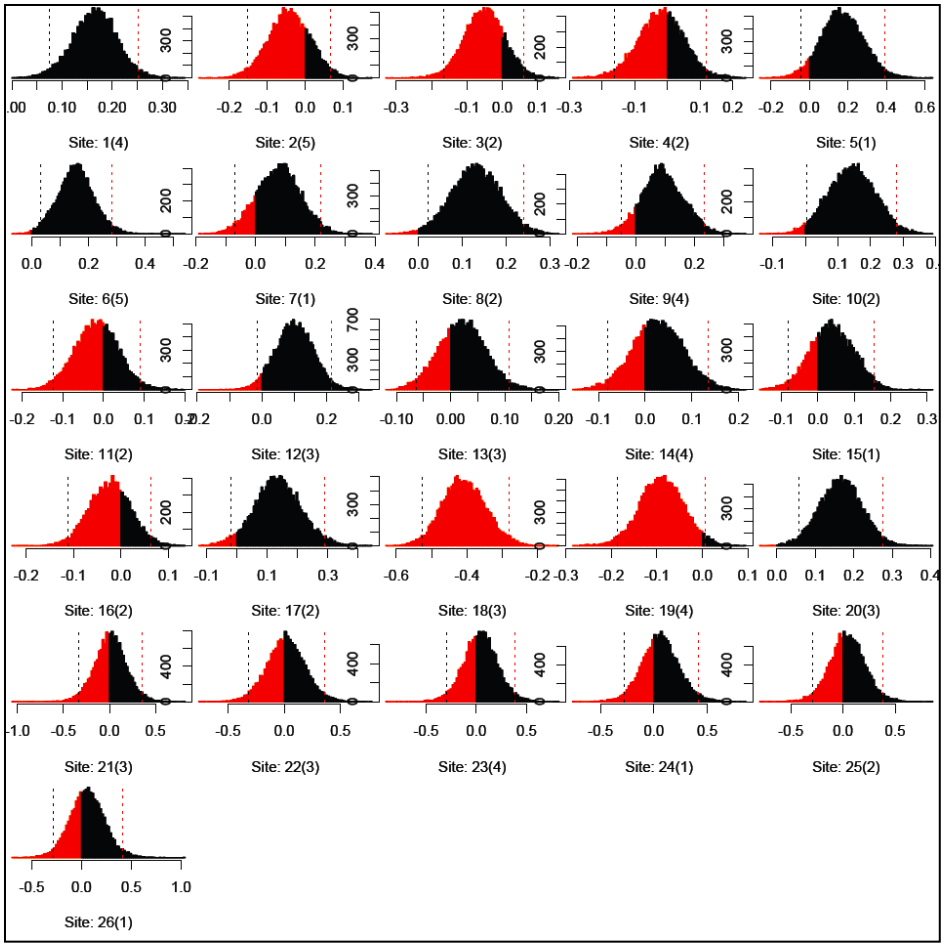



Figure1.3. A similar Bayesian trend analysis run on each of the sites comprising the subwatersheds in Figure 1.2 reveals the variation in estimated site specific trends. For example, there is strong evidence for a positive trend in several sites (distribution is mostly black), and strong evidence for only a couple of sites (distribution is mostly red).

**Variance decomposition:** In order to evaluate how well we can determine status and trends, we need a framework that describes important components of variation and survey designs that allow us to determine those components. Variation in the various attributes of interest is associated with from a variety of sources: spatial variation (differences among sites), temporal variation (that might be within or across years), or variation introduced during the measurement process (variation due to the repeatability of a particular protocol). Understanding the roles and magnitudes of the different components of variation allows us to estimate the uncertainties associated with characterizing status and trends as well as to adjust the designs to accommodate the most troublesome components. The framework that ISEMP and CHaMP use decomposes variability in a hierarchical fashion:

- **Spatial variation** describes the fundamental differences among sites, the unique "siteness".
- **Yearly temporal variation** consists of two parts. One part (**coherent temporal variation**) is the common variation across all sites as might be affected by regional

forcing (e.g., wet or dry years would influence the flow of all sites in a particular year; cold or warm years would influence stream temperatures in a common way; ocean conditions might yield low or high abundances of salmon across all sites). A second part (**interaction variation**) is the independent yearly variation each sites yearly pattern is subject to its local forcing.

- **Residual variation**: Extraneous variation introduced during the yearly sampling window might come from: a) temporal changes during the summer low flow sampling season, b) an imprecise sampling or measurement protocol, or c) crew to crew differences in the implementation of a particular protocol.

Properly designed surveys, like those adopted by ISEMP and CHaMP allow us to estimate these important components of variation and to estimate their influence on estimates of status and trends. The following two figures (Figures 1.4 and 1.5) illustrate variance decomposition for a variety of habitat metrics. The first case comes from the Wenatchee ISEMP monitoring project covering the years 2004 – 2010. The Wenatchee example illustrates the relative magnitude of the four components of variation described above. The second illustration comes from the first year's CHaMP monitoring in which habitat was monitoring in 9 watersheds. In this case, the focus is on the repeatability of protocols by different crews. A single year's monitoring such as CHaMP 2011 doesn't provide the data across years to estimate the two temporal components of variation. Figure captions describe interpretive highlights.

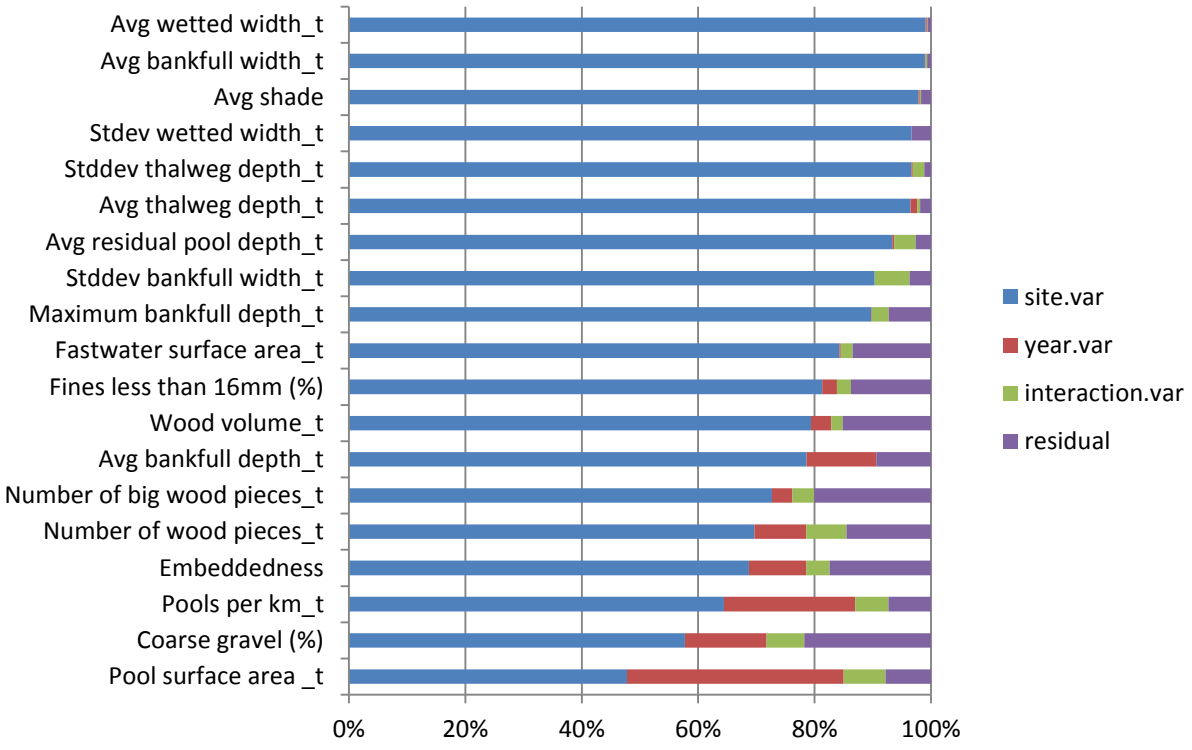


Figure 1.4. This graph illustrates the relative proportion of total variation that is attributable to site, concordant (year), interaction, and residual variation, as described in the text. Data come from the ISEMP habitat surveys in the Wenatchee watershed collected during the sampling seasons from 2004 – 2010. The attributes are ordered by the proportion attributable to site variance.

Graphs like these quickly illustrate several points. Site variance comprises 90 or more % of the total variance for 9 of the metrics indicative of a relatively clear "site" signal. These metrics provide an unambiguous description of status and would perform well in models (given that these metrics are important to the dependent variable in the models). At the other end of the scale, site variance for five of the metrics accounts for 75 or less % of total variance. Characterizing 'status' for these metrics will be less accurate than for those with higher signal:noise ratios, and these metrics might perform more poorly than others in modeling enterprises. For most of the metrics, the interaction component is relatively low, but six of the metrics have a significant "year" effect that is likely to impinge severely on trend detection capability. Notes: Avg denotes average; Stddev denotes standard deviation, a measure of variation; _t denotes that the metric was transformed to approximate a normal distribution. These graphs retain the attribute names given in the relevant database. The names are simplified in the Chapter 3 shortened versions of the graphs.

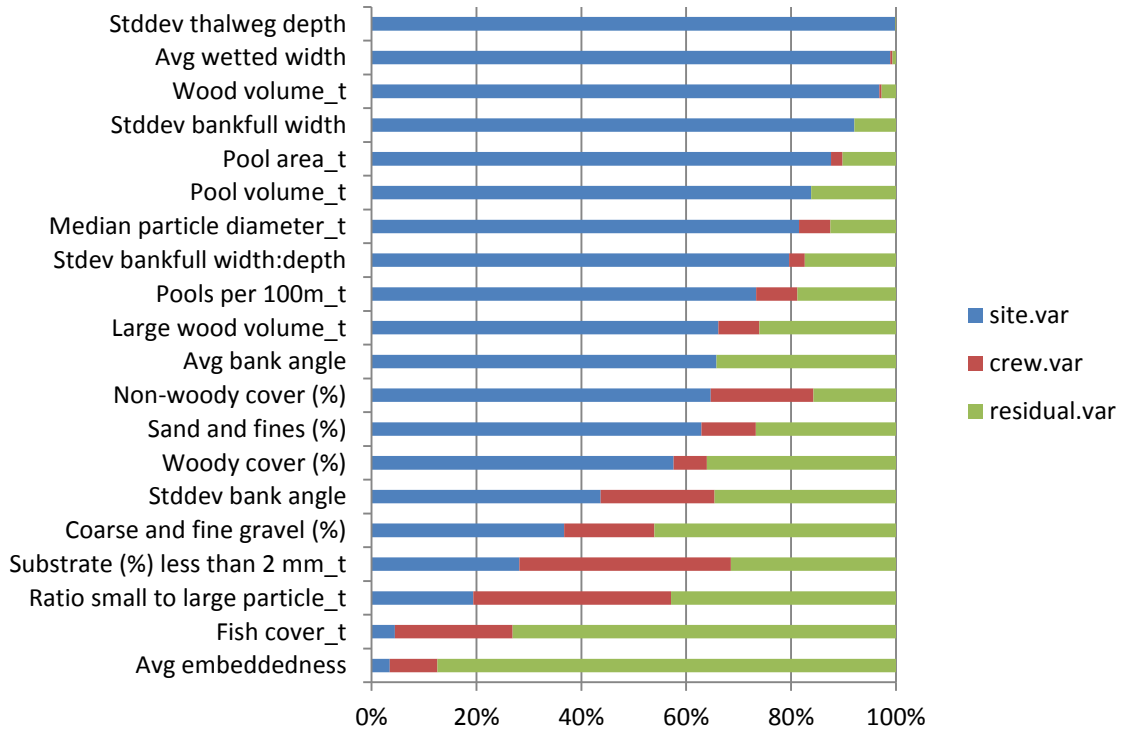



Figure 1.5. As part of the CHaMP surveys during 2011, a design to evaluate the performance of different crews at the same sites was developed in which several crews sampled the same six sites in the Upper Grande Ronde during a short time interval.

Three important variance components are summarized here: site variance across the six sites surveyed for this study, crew variance (what fraction of the total variance could be attributed to different crews applying the same protocol), and residual variance (which most likely covers the variance associated with one crew applying the same protocol repeatedly during a short time interval). This graph illustrates the difficulty in obtaining repeatable measures of several metrics as indicated by the relatively small site proportion of variance of the set at the bottom of the graph. Average embeddedness and Fish cover are especially difficult. In both these cases, most of the variation is residual indicating that the protocol is difficult to implement even by a single crew. For several of the metrics, additional crew training might reduce the noisiness, e.g., Substrate less than 2mm and the Ratio small to large particle metrics. A caveat is in order: this study covered six sites that were relatively close together in the Grande Ronde watershed. It is likely that site to site variability is relatively low among the six sites, possibly exaggerating the "noisiness" and "repeatability" of applying the sampling protocols compared with what might be seen across a broader range of habitat conditions. Future studies should cover a broader geographic coverage of sites. Notes: Avg denotes average; Stddev denotes standard deviation, a measure of variation; _t denotes that the metric was transformed to approximate a normal distribution. These graphs retain the attribute names given in the relevant database. The names are simplified in the Chapter 3 shortened versions of the graphs.

These components of variation affect our ability to estimate status and trends in different ways. Understanding their relative magnitudes allows us to adjust the monitoring designs or to incorporate "external" factors into the monitoring program, e.g., climatic or ocean conditions that might be forcing the coherent variation.

**Status**: We are often interested in describing the fundamental "siteness": to what extent are sites different from each other, unconfounded by extraneous variation. The more sites we sample, the better our description of the status of the resource. However, extraneous variability interferes with our ability to describe status. With respect to estimating status, extraneous variation primarily consists of interaction (a particular habitat metric might be highly variable from year to year making it difficult detect true differences among sites) and residual variation. Coherent temporal variation is generally small and interferes little with status estimation.

**Trend**: Designs that incorporate repeated measurements at the same sites are much more sensitive to detecting temporal patterns than are designs that visit different sites over time: site to site differences can have a major effect on trend detection. Revisiting sites (i.e. via the panel designs described above) "factors out" the effect of sites on trend detection. Trend detection is also sensitive to the other components of variation. The effect of the combination of interaction and residual variation on trend detection can be minimized by the number of sites incorporated into a survey. However, the coherent component of temporal variation is not amenable to design choices. In a sense, it is an "external" factor imposed on the domain. Accommodating its effect on trend detection involves identifying and monitoring the magnitude of the "external" forcing such that its magnitude can be incorporated into the trend detection models. Ocean and climatic conditions are common external forcing factors that affect all sites in a region in a common way.

ISEMP's and CHaMP's spatial and temporal designs are based on a firm research foundation that recognizes the need to determine the structure of variability and adapt monitoring designs as we understand the magnitude of these variance components and their influence on our ability to estimate status and trends. These designs have been used by several agencies for more

than a decade (Oregon Dept. of Fish and Wildlife's coastal coho monitoring program, US Forest Service's AREMP and PIBO monitoring projects, and the US EPA has been using these designs for more than a decade both regionally and has adopted the design approach for its national lakes, streams and rivers, near coastal, and wetlands monitoring programs).

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Substantial information about the design principles and applications can be found at: www.epa.gov/nheerl/arm and at www.salmonmonitoringadvisor.org

# CHAPTER 2: Decomposition of Lower Granite Dam Aggregate Spring/Summer Chinook Salmon and Steelhead into Tributary and Population Specific Escapement Using Instream Pit Tag Arrays

As proposed in the Salmon Subbasin Study Design (QCInc 2005), the adult spring/summer Chinook salmon and steelhead run-at-large past Lower Granite Dam (LGD) can be decomposed into population and/or tributary specific escapement estimates based on mark-recapture methods. Regional agencies and ISEMP operate an extensive network of PIT arrays (Figures 2-1 and 2-2). These arrays are intended to provide co-managers information on run-timing (A-run and B-run steelhead), tributary/population escapement estimates as well as age and sex composition required to meet subbasin plans and the information needs of the BiOp (NMFS 2008).

Assuming a known run size past LGD and a known tagging rate, we can estimate the total number of fish, tagged and untagged in any location PIT tags are detected with known efficiency (e.g. instream PIT tag arrays, weirs, dams, etc.). Unfortunately, estimating the run-at-large at LGD is complicated "fallback" (downstream passage of adults immediately following ladder ascension), passage through dam locks, and diel operation of fish counting facilities (technicians typically count migrating fish between 10 and 16 hours per day depending on time of year, and only count 50 minutes out of every hour). In addition, trapping and tagging rates at LGD are not typically constant, owing to the multiple production and research projects that rely on trapping at the facility. Lastly, the adult trap at LGD is subject to periods of closure for maintenance and during periods when high water temperatures endanger fish health. Each of these issues has the potential to bias subsequent PIT tag expansions to an unknown degree. In general, these sampling inconsistencies affect steelhead sampling to a much greater degree than spring/summer Chinook salmon.

Depending on the species and tagging rates, we have developed several distinct approaches to estimate tributary/population escapement. During periods of consistent tagging rates, lower temperatures, and consistent ladder count schedules (predominately the Chinook migration) we can use basic mark/recapture models. During periods of inconsistent tagging rates, count periods, and trap operations (Steelhead migration), more complex models were developed to estimate the total escapement of fish that pass upstream of LGD. Specific to this report, we evaluate the performance of a series of auto-regressive moving average models to estimate the impacts of sampling inconsistencies from ladder fish counting and trapping operations.

In addition to estimating total adult escapement, scales and tissues samples collected during adult PIT tagging at LGR enable estimates of age and sex ratio for the run-at-large, which can be parsed into population/tributary specific estimates as described below.

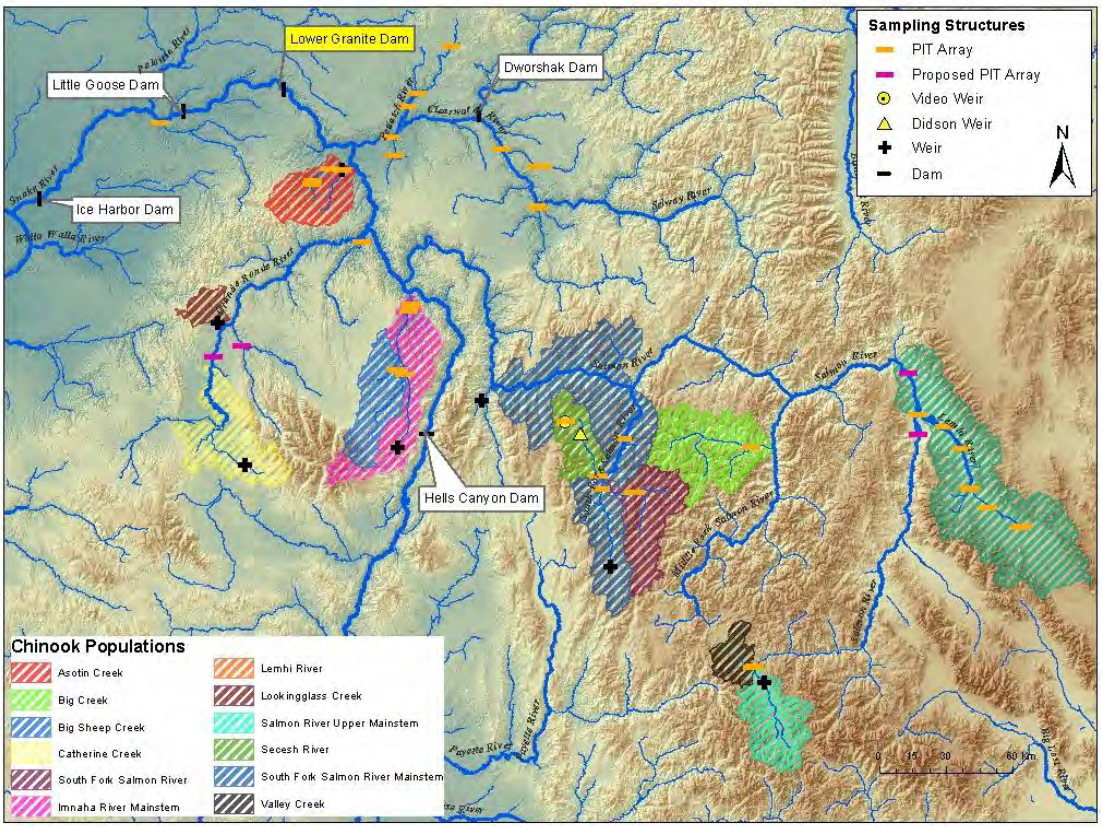


Figure 2.1. ISEMP and other state agency Snake River Basin instream PIT detection sites and PIT collection locations (weirs) relative to Interior Columbia Technical Recovery Team's population designations for -run Chinook salmon.

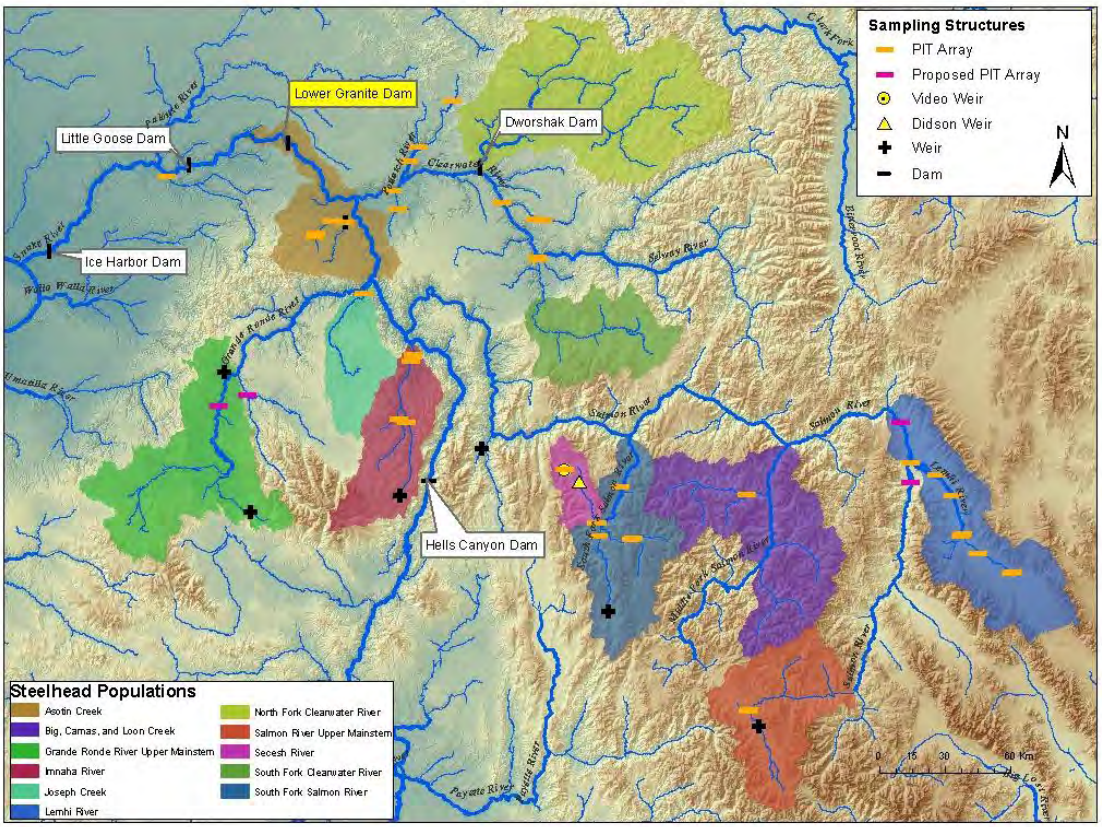


Figure 2-2. ISMEP and other state agency Snake River Basin instream PIT detection sites and PIT collection locations (weirs) relative to Interior Columbia Technical Recovery Team's population designations for summer-run steelhead.

As described above, the expansion of PIT tags passing an array requires an estimate of the total fraction of the migrating adult spring/summer Chinook salmon and steelhead that are PIT tagged. The tagging rate is defined as the percent of total escapement (total PIT tagged divided by total escapement) over LGD. Fallback, passage through locks, and straying (e.g., adults passing LGD that later migrate to downstream populations) can either be calculated from other studies and/or can be largely ignored if it can be safely assumed that these issues are equally realized for tagged and untagged adults. Unfortunately, during parts of the spring/summer Chinook salmon and steelhead migration, the tagging program is interrupted by sampling constraints at LGD. However, estimates of the number of returning adults can still be generated via window counts. In order to assign these counted (but not tagged) fish to the upstream areas requires an understanding of whether there are consistent differences in how fish distribute spatially as a function of run timing. It may be the case, for instance, that different local stocks return to the dam at distinct times during the season. If unaccounted for, these seasonal run differences may result in under- or over-estimating returns to upstream populations based on the timing of the interruption in tagging effort. Under optimal tagging conditions (i.e., consistent tagging rate), differential run-timing does not affect the resulting tributary population estimate, however, during some years the assumption of a constant tagging rate is unrealistic.

To investigate the effects of inconsistent trap operations, we employ a Bayesian patch-occupancy model to estimate the time-varying probability that fish PIT tagged at the dam escape to major tributary areas (for specific model details see QCI 2011). Additionally, we develop a Bayesian multivariate auto-regressive state space model for adult spawners passing the dam by date using both trap and window counts. Finally, we merge the results of both modeling exercises to generate estimates of total escapement to each of the monitored major tributary areas.

We estimate the number of adults that were missed during periods of trap closure using window counts. However, window counts have two estimation problems: 1) window counts only occur in 10-16 hour periods, "daytime" and 2) Snake Basin escapement includes unclipped (adipose intact) hatchery-origin adults which are incorrectly counted as natural origin adults. In order to correct window counts for nighttime passage (non-counted periods), we fit a third order polynomial regression to nighttime window count data available from 1997-2007 to estimate the proportion ($p$) of fish passing the dam at night as a function of the day of the year ($d$; 1-365):

$$\text{LOGIT}(p) \leftarrow B[1] + B[2]*d + B[3]*d^2 + B[4]*d^3$$

$$NWC \sim \text{poisson}(p, DWC)$$

The model uses the median posterior estimates of nightly fish passage to adjust the window counts (comprising an *a priori* adjustment to window count data). Day of year is used to correct for seasonal differences in window count durations and fish migration behavior.

In order to correct for the misclassification of unclipped hatchery-origin adults, we constructed a simple linear relationship between window counts of unclipped adults and the fraction of adults captured at the Lower Granite Dam trap confirmed to be natural-origin adults. Adjustments to the observed number of natural origin adults are made using a first order auto-regressive moving average model of the proportion ($p$) of unclipped adults that are natural origin as a function of the day of the year ($t$; 1-365):

$$p_t = p_{t-1} + e_t, \quad e_t \sim \text{norm}(acf*e_{t-1}, e_{var})$$

$$UCW_t \sim \text{poisson}(p_t, UCtot)$$

Thus, the estimation procedure utilizes daily counts of natural origin fish ($t_w$) based on daily trap operations (proportion of day; $p_t$) and daily window counts of natural origin adults ($w_w$). The model then employs an auto-regressive state-space moving average model to estimate the number of natural origin adults ($f$) passing Lower Granite Dam as a function of the day of the year ($t$; 1-365):

$$f_t = f_{t-1} + e_t, \qquad e_t \sim \text{norm}(acf * e_{t-1}, e_{var})$$

$$tw_t \sim \text{poisson}(p_t, f_t)$$

$$ww_t \sim \text{log-normal}(f_t, Obs.Err)$$

In order to evaluate the magnitude of bias that could be introduced during periods of trap closure and differential migration timing by upriver populations, the model includes both a "time-invariant" and "time-varying" component. The time-varying version models the probability ($p_{it}$) that a given fish passing LGR returns to a tributary of interest ($i$) using a second order polynomial function of the date of passage ($t$):

$$\text{LOGIT}(p_{i,t}) \text{ <- } B[1] + B[2]*t + B[3]*t^2$$

Once the models are used to generate a total escapement estimate and rate of adult tagging at Lower Granite Dam, the following simple expansion of estimated tags ($T$) can be used to estimate tributary specific escapement ($N$) based on PIT tag interrogations:

$$\hat{N}_a = \hat{t} \big/ tagrate_{LGD}$$

$$tagrate_{LGD} = pit \big/ totalrun$$

$$\hat{t}_{LGD} = t \Big/ \left(\frac{m}{\hat{T}}\right)$$

$$\hat{T} = \frac{(c+1)(m+1)}{(r+1)} - 1$$

Where the number of fish detected at Pit arrays ($m$) and subsequently detected at other upstream detection sites ($r$), can be used to estimate the total number of PIT tags that crossed the array.

Additionally, to determine the effect of population specific migration timing and bias introduced by LGD tagging operations, ladder trap maintenance, and window counting methods, we utilize a Bayesian patch-occupancy ("tributary") model that estimates the daily proportion of fish crossing LGD and assigns them a probability of migrating to a specific tributary.

***Age and Sex Structured Run reconstruction***

Using the escapement estimates generated as described in the previous section in conjunction with age information from scale samples at LGD and sex markers applied to tissue samples collected from PIT tagged adults at LGD, escapement can be further partitioned into sex and age as follows:

$$\hat{N}_{t,a,s} = \hat{N}_t * \hat{p}_{t,a,s}$$

$$\hat{p}_{t,a,s} = \sum_i f_{t,i,a,s} \Big/ n$$

Where:
$t$ = tributary
$N$ = escapement estimate in tributary t
$p$ = proportion of fish with age a and sex s,
$i$ = PIT tag in tributary t
$a$ = age of fish $i$
$s$ = sex of fish $i$
$n$ =number of fish with pit tags aged

Notably, obtaining tissue and scale samples while tagging fish at LGD precludes the need to sample fish later upon their arrival at tributaries, thus limiting the handling otherwise necessary to generate tributary specific abundance estimates by age and sex.

***Results***

Lower Granite Dam ladder trap PIT tagging operations began in August 2009. Cooperating agencies and groups including NOAA, IDFG, WDFW and QCI collaboratively sample and tag spring/summer Chinook salmon and summer-run steelhead at LGD. Two run years, 2010 and 2011 Chinook and 2009-2010 and 2010-2011 steelhead have been consistently tagged and the modeled results of escapement over Lower Granite Dam and yearly tagging rates are found in 2-1.

Table 2-1. Spring/summer Chinook salmon and summer-run steelhead escapement estimates over LGD, 95% confidence interval, number of PIT tagged fish by species, and PIT tagging rates by run year.

| Species | Run Year | Escapement over Lower Granite Dam | 95% CI | Tagging Rate | 95% CI | No. PIT Tagged |
|---|---|---|---|---|---|---|
| **Chinook** | 2010 | 26,465 | 24,650-27,929 | 0.044 | 0.042-0.047 | 1,177 |
| **Chinook** | 2011 | 26,972 | 25,889-28,173 | 0.103 | 0.099-0.107 | 2,786 |
| **Steelhead** | 2009 | 45,889 | 44,680-46,928 | 0.087 | 0.084-0.089 | 3,773 |
| **Steelhead** | 2010 | 48,639 | 47,409-49,690 | 0.099 | 0.097-0.102 | 4,638 |

Tagging rates during the spring/summer Chinook salmon run have remained fairly constant within the sampling season over the two years since adult tagging at LGD was initiated for ISEMP (4% and 10%, respectively). However, tagging rates for steelhead have varied from 4-15% depending on time of year and trap operations. Beginning in 2011, tagging rates have stabilized at 10% for both species.

***Spring/Summer Chinook Salmon Tributary Estimates***

The number of instream PIT tag detection locations has increased since 2009. The 2010 run-year Chinook salmon tributary escapement estimates are found in Table 2-2. Owing to logistical and permitting issues, the 2010 tagging rate was fixed at an estimated rate of 4.4%

(95% CI 4.2% - 4.7%), yielding 1,177 PIT tagged spring/summer Chinook salmon at the LGD ladder trap and an estimated total escapement of 26,465 naturally produced Chinook (95% CI 24,650 – 27,929) migrating past LGD. Even at a low tagging rate, the one independent estimate obtained from the Johnson Creek Weir (tributary to the East Fork South Fork Salmon River) aligned consistently with the estimate provided by the model.

Table 2-2. 2010 run-year wild spring/summer Chinook salmon estimated tributary escapement (95% confidence interval) for select Snake Basin detection locations (PTAGIS site abbreviations in parenthesis). The independent estimate for the East Fork SFSR was generated by adult capture at the weir operated by the Nez Perce Tribe (BPA project number 1996-043-00).

| Tributary | Estimate | 95% CI | Independent Estimate |
|---|---|---|---|
| **South Fork Salmon River (SFG)** | 7,005 | 6,655-7,355 | |
| **Secesh River (S. Fk. Salmon, ZEN)** | 1,308 | 1,165-1,451 | |
| **E. Fk. South Fork Salmon River (ESS)** | 1,026 | 1,015-1,038 | 1,032 |
| **Upper South Fork Salmon River (KRS)** | 3,450 | 2,731-4,169 | |
| **Big Creek (M. Fk. Salmon) (TAY)** | 285 | 150-411 | |
| **Lemhi River (LLR)** | 262 | 243-281 | |
| **Valley Creek (VC)** | 235 | 191-281 | |

Run-year 2011 was tagged at a significantly higher rate (10.3%, 95% CI 9.9% - 10.7%), with 2,786 fish PIT tagged and an estimated escapement of 26,972 Chinook (95%CI 25,889-28,173). Additionally, 15 additional PIT tag detection sites placed in the Snake Basin increased the number of available estimates of tributary abundance. However, for the purpose of this report, we summarized the major tributary populations (Table 2-3). Field data from weir sites were unavailable for this report, precluding independent estimates of escapement.

Table 2-3. Spring/summer Chinook salmon tributary escapement estimates and associated 95% confidence intervals for select Snake Basin detection locations for run year 2011 (PTAGIS site abbreviations in parenthesis).

| Tributary | Estimate | 95% CI |
|---|---|---|
| **Imnaha River (IR1)** | 2,421 | 2124 - 2716 |
| **South Fork Salmon River (SFG)** | 4,749 | 4326 - 5201 |
| **Secesh River (S. Fk. Salmon, ZEN)** | 779 | 745-791 |
| **EF South Fork Salmon River (ESS)** | 652 | 649-657 |
| **Big Creek (M. Fk. Salmon, TAY)** | 449 | 290 - 689 |
| **Lemhi River (LLR)** | 337 | 230 - 470 |
| **Valley Creek (VC)** | 460 | 380-560 |

### *Steelhead*

Steelhead tagging was subject to trap closures and multiple changes in trap rates at LGR over the 2009-2010 run year and to lesser degree for the 2010-2011 run year. For the 2009-2010 run, a total of 3,971 steelhead were tagged by ISEMP at LGD, resulting in an estimated tagging rate of 8.7% (95% CI 8.4% - 9.0%). Total steelhead escapement past LGR was estimated to be 45,889 natural origin adults (95% CI 44,680 – 46,928). Escapement estimates are presented in Table 2-4. Independent corroboration from estimates of steelhead escapement from four projects using weirs neighboring instream PIT tag arrays are provided as means to validate the approach.

Table 2-4. Tributary escapement estimates and associated 95% confidence intervals for run year 2009-2010 naturally produced steelhead at instream PIT tag arrays and independent estimates for select ISEMP and other Snake River basin instream PIT detection sites (PTAGIS site abbreviations in parenthesis). The independent estimate for Asotin Creek was generated by weir operated by the Washington Department of Fish and Wildlife, and the Rapid River weir, Fish Creek, and Sawtooth weirs operated the by the Idaho Department of Fish and Game.

| Tributary | Estimate | 95% CI | Independent Estimate |
|---|---|---|---|
| **Potlatch River (JUL)** | 784 | 621-992 | |
| **Fish Creek (Lochsa River, Weir)** | 246 | 129-434 | 205 |
| **Asotin Creek (AFC)** | 1687 | 1407-1963 | 1,500 |
| **Rapid River (Weir)** | 136 | 72-235 | 150 |
| **South Fork Salmon River (SFG)** | 1795 | 1527-2081 | |
| **Secesh River (S. Fk. Salmon, ZEN)** | 298 | 169-558 | |
| **Big Creek (M. Fk. Salmon, TAY)** | 753 | 431-1914 | |
| **Lemhi River (LLR)** | 630 | 455-928 | |
| **Valley Creek (VC)** | 237 | 155-411 | |
| **Upper Salmon (Sawtooth Weir)** | 138 | 76-226 | 115 |

Field crews tagged 4,638 naturally produced steelhead during run year 2010-2011 resulting in a 9.9% (95% CI 9.7% - 10.2%) tagging rate and a total escapement estimate of 48,639 (95% CI 47,409-49,690) naturally produced steelhead over LGD. Table 2-5 summarizes tributary run estimates above instream PIT tag arrays and an independent estimate generated at the Asotin Creek weir.

Table 2-5. Tributary escapement estimates and associated 95% confidence intervals for run year 2010-2011 naturally produced steelhead at instream PIT tag arrays and independent estimates for select ISEMP and other Snake River basin instream PIT tag detection sites (PTAGIS site abbreviations in parenthesis). The independent estimate for Asotin Creek was generated by a weir operated by the Washington Department of Fish and Wildlife, and Sawtooth weir operated the by the Idaho Department of Fish and Game.

| Tributary | Estimate | 95% CI | Independent Estimate |
|---|---|---|---|
| **Potlatch River (JUL)** | 739 | 443 - 1541 | |
| **Lapwai Creek (LAP)** | 455 | 262 - 1340 | |
| **Asotin Creek** | 973 | 778 - 1224 | 1,128 |
| **Joseph Creek (Grande Ronde River, JOC)** | 1,663 | 1420 - 1921 | |
| **Cow Creek (Imnaha River, COC)** | 161 | 94 - 250 | |
| **Imnaha River (IR1)** | 3,516 | 3167 - 3897 | |
| **South Fork Salmon River (SFG)** | 2,980 | 2654 - 3361 | |
| **Secesh River (S. Fk. Salmon, ZEN)** | 433 | 250-738 | |
| **Big Creek (M. Fk. Salmon, TAY)** | 745 | 562 - 960 | |
| **Lemhi River (LLR)** | 503 | 346 - 736 | |
| **Valley Creek (VC)** | 270 | 190 - 382 | |
| **Upper Salmon (Sawtooth Weir)** | 79 | 36 - 147 | 98 |

### *Discussion*

We have generated two years of escapement estimates for spring/summer Chinook salmon and steelhead by decomposing the estimated run-at-large over LGD into tributary and/or population specific escapement. As demonstrated by the paucity of locations that are available for independent validation, these escapement estimates reflect a much needed component with regard to estimating the effectiveness of mitigation actions on population growth rates - particularly for Snake River steelhead, owing to difficulties that accompany the operation of weirs during high-flow periods during their migration. Results from two years of PIT tagging adults at LGD and decomposing that run using instream PIYT tag demonstrates the potential for instream PIT tag arrays to provide efficient, cost-effective, and accurate estimates of tributary escapement.

### *Literature Cited*

NMFS. 2008. Endangered Species Act section 7(a)(2) consultation biological opinion and Magnuson-Stevens Fishery Conservation and Management Act essential fish habitat consultation: Consultation on remand for operation of the Federal Columbia River power system, 11 Bureau of Reclamation projects in the Columbia Basin and ESA section 10(a)(I)(A) permit for juvenile fish transportation program (revised and reissued pursuant to court order, NWF v. NMFS, Civ. No. CV 01-640-RE (D. Oregon). NOAA's National Marine Fisheries Service (NOAA Fisheries), Northwest Region, Seattle, WA. https://pcts.nmfs.noaa.gov/pls/pcts-pub/pcts_upload.summary_list_biop?p_id=27149

# CHAPTER 3: Watershed Production Model

Within the Salmon Subbasin, we have implemented a habitat and population status and trends monitoring project in the South Fork Salmon River (SFSR) watershed and habitat action effectiveness evaluation in the Lemhi River watershed. These initiatives are joined through the application of a watershed model (QCInc 2005) that views fish vital rates (survival/productivity, abundance, and condition) as a function of the quantity and quality of available habitat. These functions are constructed using both coarse (e.g., Geographic Information Systems (GIS)) and fine (e.g., reach) scale habitat measures (Figure 3-1). Once validated via the collection of empirical data within habitat classes, the model provides a statistical framework to assess the effects of different classes of habitat actions on life-stage specific vital rates (productivity/survival and condition) of anadromous and resident salmonids. Additionally, the model includes survival functions enabling the user to alter survival rates (juvenile to emigrant and emigrant to adult) as necessary to compensate for hatchery production.

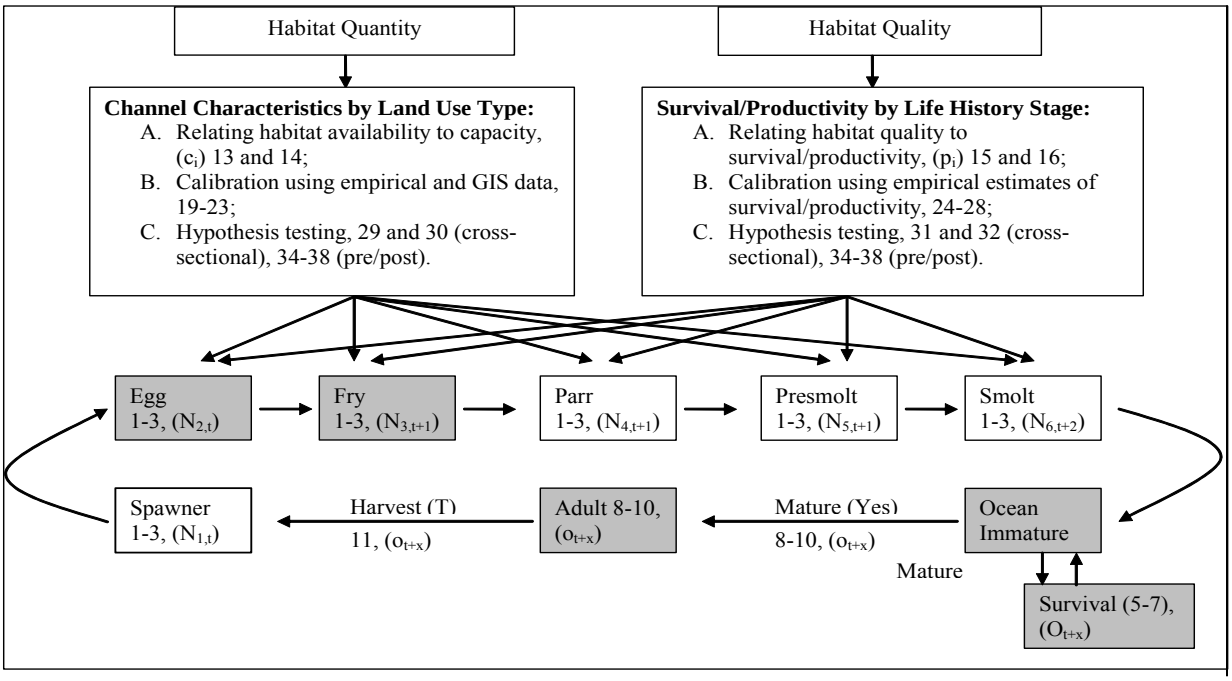


Figure 3-1. Schematic illustrating how the model develops relationships between habitat quantity (capacity) and quality (survival/productivity) and stage-based abundance, productivity, and survival. Grey boxes indicate those life stages for which metrics will be inferred, notation in parentheses refers to model parameters, and numbers within the boxes refer to equations in the Lemhi Study Design (QCInc 2005).

Basic model details are described below, greater detail can be found in the Salmon Subbasin ISEMP Proposal (QCInc 2005). Briefly, the watershed model utilizes a multi-stage Beverton Holt model (Mousalli and Hilborn 1986):

$$R_{t+1} = \frac{aS_t}{b + S_t} \longrightarrow N_{i+1,t+1} = \frac{N_{i,t}}{\frac{1}{p_{i,t}} + \frac{1}{c_{i,t}} N_{i,t}}$$

*Where:*
$N_{i,t}$ = number of fish at life stage (i), time (t)
$N_{i+1,\,t+1}$ = number of fish in next life-stage (i+1) and time (t+1)
$p_{i,t}$ = productivity, or maximum survival rate for life-stage (i)
$c_{i,t}$ = carrying capacity, or maximum survival to the next life-stage

Productivity is equivalent to the maximum survival from one life stage to the next. We assume that productivity is functionally related to habitat quality, itself an expression of multiple factors such as land use. To include this relationship we utilize a scalar to adjust survival based on habitat classification. For the purposes of this report, the scalar was maximized for habitat in the Upper Lemhi River, which is currently the primary spawning and rearing habitat utilized by spring/summer Chinook salmon.

$$p_i = S_i \longrightarrow p_i = S_i * E_l \longrightarrow p_{i,t} = S_i \times \frac{\sum_{q=1}^{n} [E_{i,q}] \times [L_{q,k}]_t}{\sum_{q=1}^{n} [L_{q,k}]_t}$$

*Where:*
$p_i$ = productivity (maximum survival from one life stage to the next)
$S_i$ = survival
$E$ = scalar
$t$ = temporal period (e.g., season, year, life stage, etc.)
$k$ = spatial context (e.g., tributary, subwatershed, watershed etc.)

The maximum number of fish surviving from one life stage to the next is a function of carrying capacity. In a habitat context this translates to the maximum number of fish of a specific life stage that can reside in a specific habitat type.

$$c_i = \sum_{j=1}^{n} [H_j] \times [D_{j,i}] \longrightarrow c_{k,i,t} = A_k \times \sum_{j=1}^{n} \left[ \left[ \sum_{q=1}^{n} [H_{j,q}] \times [L_{q,k}]_t \right] \times [D_{j,i}] \right]$$

*Where:*
$c_{ij}$ = maximum number of fish at life stage *i* in habitat type *j*
$H$ = habitat class (e.g., pool or reach type)
$D$ = fish density
$t$ = temporal periods (e.g. year, seasonal, etc.)

$k$ = spatial context (e.g. watershed, tributary, etc.)
$A$ = areal extent (or other spatial measure)
$L$ = Land use type (or other characteristic)

These relationships reduce as follows:

$$N_{i+1,t+1} = \frac{N_{i,t}}{\frac{1}{p_{i,t}} + \frac{1}{c_{i,t}} N_{i,t}} \longrightarrow N_{k,i+1_t} = \frac{N_{k,i_t}}{\frac{1}{Sr_i \times \frac{\sum_{q=1}^{n} [E_{i,q}] \times [L_{q,k}]_t}{\sum_{q=1}^{n} [L_{q,k}]_t}} + \frac{1}{A_k \times \sum_{j=1}^{n} \left[ \left[ \sum_{q=1}^{n} [M_{j,q}] \times [L_{q,k}]_t \right] \times [D_{j,i}] \right]} N_{k,i_t}}$$

### *Populating the Watershed Model*

The watershed model requires multiple years of adult escapement and juvenile abundance, survival, distribution, and growth data in order to generate capacity and freshwater productivity estimates. Tying these estimates to physical habitat at appropriate spatial scales (e.g., subwatersheds of the Lemhi River targeted for reconnection) similarly requires multiple years of habitat survey effort. The Salmon Subbasin ISEMP project initiated sampling in 2009, collecting the first adult return data in 2010. Thus, the watershed model will be sufficiently populated in 2013 for Brood Year 2010 juvenile production. Given the additional complexity of steelhead life-history, these initial model runs will be most applicable to spring/summer Chinook salmon, whereas complete steelhead information for brood year 2010 may not be available until as late as 2018 (Table 3-1). After 2013, an additional brood year of data will be added to the model each subsequent year.

Similar to freshwater productivity data, physical habitat data are cumulative. With each passing sample year, the density of points within subwatersheds/tributaries of interest increases. As sample density increases within the smallest spatial scales of interest (e.g., individual subwatersheds of the Lemhi), our ability to relate fish performance to habitat characteristics will improve as will our ability to identify differences in the distribution of key habitat attributes across subwatersheds/tributaries of interest. As illustrated in Table 3-1, two full rotating panels of GRTS-based habitat surveys will be completed in 2013, enabling the watershed model to evaluate the "restoration value" of alternative tributary reconnection scenarios.

Despite the fact that the data necessary to fully populate the watershed will be unavailable until 2013, we aggregated data into "reporting units" in order to demonstrate the utility of the watershed model. Reporting units represent biologically meaningful habitat groupings, but describe a much larger spatial scale than the individual tributaries that are the ultimate target of restoration. This aggregation is simply an approach to enable the application of the watershed model by creating groups of tributaries that yield sufficient data density to populate the model. As such, results of the watershed model presented in this report should be viewed as demonstration products.

The reporting units developed for this report can be summarized as:

- The lower mainstem Lemhi River extending from its confluence with the Salmon River upstream to the confluence of Hayden Creek;
- The upper mainstem Lemhi River extending from the confluence of with Hayden Creek to the origin of the Lemhi River at the confluence of Hawley Creek and Eighteenmile Creek;
- Hayden Creek;
- Tributaries to the lower mainstem Lemhi River;
- Tributaries to the upper mainstem Lemhi River.

Within reporting units, tributaries can be grouped into the following categories:

- Hayden Creek, and the lower and upper mainstem Lemhi represent habitat that was accessible to anadromous fish at the inception of the ISEMP project.
- High priority watersheds are those watersheds that are identified as having both high quality habitat and most likely to be cost-effectively reconnected to the mainstem Lemhi River.
- Moderate priority watersheds are those watersheds that exhibit greater habitat degradation and/or represent greater logistical difficulties with regard to their potential for reconnection.
- Low priority tributaries are those that are either heavily degraded and/or are logistically infeasible and/or cost-prohibitive with regard to their potential for reconnection.

Table 3-1. Relationship between sampling year, brood year, and parameterization of the watershed model.

| | | | | | Fish Surveys (carrying capacity and productivity estimates) | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Instream (Quantity/Quality) | | | | Measurement = Density and Survival | | | | | | | | | | | | | | |
| | Wading Surveys (Yearly Panel) | | | | Brood Year | | | | | | | | | | | | | | |
| **Sampling Year** | **LiDAR** | **1** | **2** | **3** | 2005 | 2006 | 2007 | 2008 | 2009 | 2010 | 2011 | 2012 | 2013 | 2014 | 2015 | 2016 | 2017 | 2018 | 2019 |
| **2009** | | 65 | | | SH Age 4<br>CH Age 3 | SH Age 3<br>CH Age 2 | SH Age 2<br>CH Age 1 | SH Age 1<br>CH Age 0 | SH Age 0 | | | | | | | | | | |
| **2010** | | | 37 | | SH Age 5<br>CH Age 4 | SH Age 4<br>CH Age 3 | SH Age 3<br>CH Age 2 | SH Age 2<br>CH Age 1 | SH Age 1<br>CH Age 0 | **SH/CH Spaw**<br>**SH Age 0** | | | | | | | | | |
| **2011** | | 55 | | | SH Age 6<br>CH Age 5 | SH Age 5<br>CH Age 4 | SH Age 4<br>CH Age 3 | SH Age 3<br>CH Age 2 | SH Age 2<br>CH Age 1 | **SH Age 1**<br>**CH Age 0** | SH/CH Spaw<br>SH Age 0 | | | | | | | | |
| **2012** | X | | 55 | | SH Age 7 | SH Age 6<br>CH Age 5 | SH Age 5<br>CH Age 4 | SH Age 4<br>CH Age 3 | SH Age 3<br>CH Age 2 | **SH Age 2**<br>**CH Age 1** | SH Age 1<br>CH Age 0 | SH/CH Spaw<br>SH Age 0 | | | | | | | |
| **2013** | | | | **55** | **SH Age 8** | **SH Age 7** | **SH Age 6**<br>**CH Age 5** | **SH Age 5**<br>**CH Age 4** | **SH Age 4**<br>**CH Age 3** | **SH Age 3**<br>**CH Age 2** | **SH Age 2**<br>**CH Age 1** | **SH Age 1**<br>**CH Age 0** | **SH/CH Spaw**<br>**SH Age 0** | | | | | | |
| **2014** | | 55 | | | | SH Age 8 | SH Age 7 | SH Age 6<br>CH Age 5 | SH Age 5<br>CH Age 4 | SH Age 4<br>CH Age 3 | SH Age 3<br>CH Age 2 | SH Age 2<br>CH Age 1 | SH Age 1<br>CH Age 0 | SH/CH Spaw<br>SH Age 0 | | | | | |
| **2015** | | | 55 | | | | SH Age 8 | SH Age 7 | SH Age 6<br>CH Age 5 | SH Age 5<br>CH Age 4 | SH Age 4<br>CH Age 3 | SH Age 3<br>CH Age 2 | SH Age 2<br>CH Age 1 | SH Age 1<br>CH Age 0 | SH/CH Spaw<br>SH Age 0 | | | | |
| **2016** | X | | | 55 | | | | SH Age 8 | SH Age 7 | SH Age 6<br>CH Age 5 | SH Age 5<br>CH Age 4 | SH Age 4<br>CH Age 3 | SH Age 3<br>CH Age 2 | SH Age 2<br>CH Age 1 | SH Age 1<br>CH Age 0 | SH/CH Spaw<br>SH Age 0 | | | |
| **2017** | | 55 | | | | | | | SH Age 8 | SH Age 7 | SH Age 6<br>CH Age 5 | SH Age 5<br>CH Age 4 | SH Age 4<br>CH Age 3 | SH Age 3<br>CH Age 2 | SH Age 2<br>CH Age 1 | SH Age 1<br>CH Age 0 | SH/CH Spaw<br>SH Age 0 | | |
| **2018** | | | 55 | | | | | | | SH Age 8 | SH Age 7 | SH Age 6<br>CH Age 5 | SH Age 5<br>CH Age 4 | SH Age 4<br>CH Age 3 | SH Age 3<br>CH Age 2 | SH Age 2<br>CH Age 1 | SH Age 1<br>CH Age 0 | SH/CH Spaw<br>SH Age 0 | |
| **2019** | | | | 55 | | | | | | | SH Age 8 | SH Age 7 | SH Age 6<br>CH Age 5 | SH Age 5<br>CH Age 4 | SH Age 4<br>CH Age 3 | SH Age 3<br>CH Age 2 | SH Age 2<br>CH Age 1 | SH Age 1<br>CH Age 0 | SH/CH Spaw<br>SH Age 0 |

### *Fish Data*

From a fish sampling perspective, the information needs of the watershed model include life-stage specific juvenile abundance, productivity/survival, growth/condition, and distribution as well as adult escapement across habitat classes and within treated and untreated stream reaches. Within the SFSR and Lemhi, we emphasized the use of existing sampling activities to satisfy information needs whenever possible; where necessary, additional sampling was implemented through ISEMP (Tables 3-2 through 3-4). Notably, much of the information supporting ISEMP analyses and many of the sampling activities upon which ISEMP relies require close collaboration with a number of key cooperating agencies, primarily the Nez Perce Tribe and the Idaho Department of Fish and Game. Additionally, ISEMP utilizes PIT tag detections and juvenile and adult abundance estimates from mainstem Snake and Columbia River hydropower facilities. In order to utilize the data provided by existing and proposed sampling, ISEMP has worked with collaborators to develop a standard set of protocols that define how sampling is conducted.

Briefly, rotary screw traps and PIT tag arrays are used in conjunction with adult and juvenile PIT tagging efforts to generate abundance, survival, and growth estimates at the reach and population spatial scales as appropriate for populations and subpopulations of steelhead and Chinook salmon in the SFSR and Lemhi watersheds. As described in Part B Chapter 2 of this report, adult escapement estimates for steelhead and spring/summer Chinook salmon are generated via adult tagging at Lower Granite Dam and subsequent interrogation of adults as they pass instream arrays. Juvenile abundance, survival and distribution estimates are generated by pairing juvenile tagging with interrogations and/or recaptures in remote surveys, at rotary screw traps, and at instream and mainstem PIT tag arrays.

For the purposes of this report, we focused primarily on data generated via remote juvenile capture and tagging surveys. These surveys are distributed across existing, high, and moderate priority tributaries of the Lemhi River Basin using GRTS. This sampling effort is distributed among three temporal components; unique sites, within-year repeat sites, and annual sites. Unique sites are sampled only one time, within-year repeat sites are sampled two or three times within a year, and annual repeat sites are visited at least once every year. This distribution of effort enables an evaluation of the repeatability of surveys and allows estimation of the change in abundance of juveniles over time in specific tributaries. Those estimates will be used to determine the amount of sampling effort (i.e., number of sites) required to generate reliable juvenile abundance estimates. Mark recapture at individual sites is used to generate abundance estimates (Figures 3-2 and 3-3), which can be expanded to generate total abundance for tributaries, subwatersheds, and the entirety of the Lemhi using standard GRTS expansions (Stevens and Olsen 2003). PIT tags deployed during these surveys enable estimates of growth, survival, and distribution by life stage via interrogation at instream PIT tag arrays and recapture at rotary screw traps and in subsequent remote juvenile capture and tagging surveys. Life stage and brood year of origin are obtained by ageing scales collected from all PIT tagged juveniles.

Table 3-2. Location, project, sponsor, and sampling activity providing adult and juvenile tagging (pt = PIT tag, bb = Bismark Brown Dye, em = external mark) that supports ISEMP adult and juvenile monitoring.

| Location | Project # | Project Title | Agency | Activity | Adult Tagging | | Juvenile Tagging | |
|---|---|---|---|---|---|---|---|---|
| | | | | | Sp/Su Chinook | Steelhead | Sp/Su Chinook | Steelhead |
| Lower Granite Dam | GSI Fast Track/2003-017-00 | GSI/ISEMP | IDFG/QCInc | Adult PIT Tagging/Tissue Sampling | pt | pt | | |
| Lower Granite Dam | USACE | USACE | USACE | Juvenile Trapping | | | pt | pt |
| Mainstem Salmon | 1987-127-02 | CSS | FPC | Mainstem Salmon Juvenile Trap | | | pt | pt |
| SFSR | 2003-017-00 | ISEMP | QCInc | Remote Juv Capture | | | pt | pt |
| SFSR | 2003-017-02 | ISEMP | NPT | Lower Secesh RST | | | pt, bb, em | pt |
| SFSR | 2003-017-02 | ISEMP | NPT | Mainstem lower SFSR RST | | | pt, bb, em | pt |
| SFSR | 1996-043-00 | JCAPE M&E | NPT | Johnson Creek RST/Adult Weir | | | pt, bb, em | pt |
| SFSR | 1989-098-00 | ISS | NPT | Lake Creek RST/Video Weir | | | pt, bb, em | pt |
| SFSR | 1989-098-00 | ISS | NPT | Upper Secesh River RST | | | pt, bb, em | pt |
| SFSR | 1989-098-00 | ISS | ISS | Mainstem upper SFSR RST | | | pt, bb, em | pt |
| Lemhi | 2003-017-00 | ISEMP | QCInc | Remote Juv Capture | | | pt | pt |
| Lemhi | 2003-017-00 | ISEMP/IMW | IDFG | mainstem lower Lemhi RST | | | pt, bb, em | pt |
| Lemhi | 1989-098-00 | ISEMP/IMW | IDFG | Mainstem upper Lemhi RST | | | pt, bb, em | pt |
| Lemhi | 1989-098-00 | ISEMP/IMW | IDFG | Hayden Creek RST | | | pt, bb, em | pt |

Table 3-3. Location, project, sponsor, and sampling activity providing adult and juvenile biological sampling (t = tissue sampling, s = scale sampling, l = length, w = weight, and o = origin) that support ISEMP analyses.

| Location | Project # | Project Title | Agency | Activity | Adult Biological Sampling | | Juvenile Biological Sampling | |
|---|---|---|---|---|---|---|---|---|
| | | | | | Sp/Su Chinook | Steelhead | Sp/Su Chinook | Steelhead |
| Lower Granite Dam | GSI Fast Track/2003-017-00 | GSI/ISEMP | IDFG/QCInc | Adult PIT Tagging | t, s, l, o | t, s, l, o | | |
| Lower Granite Dam | USACE | USACE | USACE | Juvenile Trapping | | | t, s, l, w, o | t, s, l, w, o |
| Mainstem Salmon | 1987-127-02 | CSS | FPC | Mainstem Salmon Juvenile Trap | | | l | l |
| SFSR | 2003-017-00 | ISEMP | QCInc | Remote Juv Capture | | | t, s, l, w, o | t, s, l, w, o |
| SFSR | 2003-017-02 | ISEMP | NPT | Lower Secesh RST | | | t, s, l, w, o | t, s, l, w, o |
| SFSR | 2003-017-02 | ISEMP | NPT | Mainstem lower SFSR RST | | | t, s, l, w, o | t, s, l, w, o |
| SFSR | 1996-043-00 | JCAPE M&E | NPT | Johnson Creek RST/Adult Weir | t, s, l, o | | t, s, l, w, o | t, s, l, w, o |
| SFSR | 1996-043-00 | JCAPE M&E | NPT | East Fork SFSR Carcass survey | t, s, l, o | | | |
| SFSR | 1989-098-00 | ISS | NPT | Secesh River RST | | | t, s, l, w, o | t, s, l, w, o |
| SFSR | 1989-098-00 | ISS | NPT | Secesh River Carcass Survey | t, s, l, o | | | |
| SFSR | 1989-098-00 | ISS | IDFG | Mainstem upper SFSR RST | | | t, s, l, w, o | t, s, l, w, o |
| SFSR | 1989-098-00 | ISS | IDFG | Mainstem SFSR Carcass Survey* | t, s, l, o | | | |
| SFSR | LSRCP | LSRCP | IDFG | Upper SFSR Adult Weir | t, o | | | |
| Lemhi | 2003-017-00 | ISEMP | QCInc | Remote Juv Capture | | | t, s, l, w, o | t, s, l, w, o |
| Lemhi | 2003-017-00 | ISEMP/IMW | IDFG | mainstem lower Lemhi RST | | | t, s, l, w, o | t, s, l, w, o |
| Lemhi | 2003-017-00 | ISEMP/IMW | IDFG | Hayden Creek RST | | | t, s, l, w, o | t, s, l, w, o |
| Lemhi | 1989-098-00 | ISS | IDFG | Mainstem upper Lemhi RST | | | t, s, l, w, o | t, s, l, w, o |
| Lemhi | 1989-098-00 | ISS | IDFG | Lemhi Carcass Survey | t, s, l, o | | | |

Table 3-4. Metrics provided by location, project, sponsor, and sampling activity (s = survival, g = growth, d = distribution, t = timing, c = condition, and o = origin) that support ISEMP analyses.

| Location | Project # | Project Title | Agency | Activity | Adult Recapture | | Juvenile Recapture | |
|---|---|---|---|---|---|---|---|---|
| | | | | | Sp/Su Chinook | Steelhead | Sp/Su Chinook | Steelhead |
| Lower Granite Dam | 2003-017-00 | GSI/ISEMP | IDFG/QCInc | Adult PIT Tagging | s, g, d, t, c, o | s, g, d, t, c, o | | |
| Lower Granite Dam | USACE | USACE | USACE | Juvenile Trapping | | | s, g, d, t, c, o | s, g, d, t, c, o |
| Mainstem Salmon | 1987-127-02 | CSS | FPC | Mainstem Salmon Juvenile Trap | s, d, t, c, o | s, d, t, c, o | s, d, t, c, o | s, d, t, c, o |
| SFSR | 2003-017-00 | ISEMP | QCInc | Remote Juv Capture | | | s, g, d, t, c, o | s, g, d, t, c, o |
| SFSR | 2003-017-00 | ISEMP | NPT | Lower Mainstem SFSR PIT Array | s, d, t | s, d, t | s, d, t | s, d, t |
| SFSR | 2003-017-00 | ISEMP | NPT | Secesh River PIT Array | s, d, t | s, d, t | s, d, t | s, d, t |
| SFSR | 2003-017-00 | ISEMP | NPT | East Fork SFSR PIT Array | s, d, t | s, d, t | s, d, t | s, d, t |
| SFSR | LSRCP | LSRCP | NPT | Mainstem Upper SFSR PIT Array | s, d, t | s, d, t | s, d, t | s, d, t |
| SFSR | 2003-017-02 | ISEMP | NPT | Secesh RST | | | s, g, d, t, c, o | s, g, d, t, c, o |
| SFSR | 2003-017-02 | ISEMP | NPT | Mainstem RST | | | s, g, d, t, c, o | s, g, d, t, c, o |
| SFSR | 1997-030-00 | CSAAM | NPT | Secesh River DIDSON | s, d, t | | | |
| SFSR | 1996-043-00 | JCAPE M&E | NPT | Johnson Creek RST/Adult Weir | s, g, d, t, c, o | | s, g, d, t, c, o | s, g, d, t, c, o |
| SFSR | 1996-043-00 | JCAPE M&E | NPT | East Fork SFSR Carcass survey | s, g, d, t, c, o | | s, g, d, t, c, o | s, g, d, t, c, o |
| SFSR | 1989-098-00 | ISS | NPT | Secesh River RST | | | s, g, d, t, c, o | s, g, d, t, c, o |
| SFSR | 1989-098-00 | ISS | NPT | Secesh River Carcass Survey | s, d, o | | | |
| SFSR | 1989-098-00 | ISS | IDFG | Mainstem upper SFSR RST | | | s, g, d, t, c, o | s, g, d, t, c, o |
| SFSR | 1989-098-00 | ISS | IDFG | Mainstem SFSR Carcass Survey* | s, d, o | | | |
| SFSR | LSRCP | LSRCP | IDFG | Upper SFSR Adult Weir | s, d, t, c, o | | | |
| Lemhi | 2003-017-00 | ISEMP | QCInc | Remote Juv Capture | | | s, g, d, t, c, o | s, g, d, t, c, o |
| Lemhi | 2003-017-00 | ISEMP/IMW | IDFG | mainstem lower Lemhi RST | | | s, g, d, t, c, o | s, g, d, t, c, o |
| Lemhi | 2003-017-00 | ISEMP/IMW | IDFG | Hayden Creek RST | | | s, g, d, t, c, o | s, g, d, t, c, o |
| Lemhi | 2003-017-00 | ISEMP | IDFG | Lower Mainstem Lemhi PIT Array | s, d, t | s, d, t | | |
| Lemhi | 2003-017-00 | ISEMP | IDFG | Upper Mainstem Lemhi PIT Array | s, d, t | s, d, t | | |
| Lemhi | 2003-017-00 | ISEMP | IDFG | Hayden Creek PIT Array | s, d, t | s, d, t | | |
| Lemhi | 2003-017-00 | ISEMP | IDFG | Big Timber PIT Array | s, d, t | s, d, t | | |
| Lemhi | 2003-017-00 | ISEMP | IDFG | Kenney Creek PIT Array | s, d, t | s, d, t | | |
| Lemhi | 2003-017-00 | ISEMP | IDFG | Canyon Creek PIT Array | s, d, t | s, d, t | | |
| Lemhi | 1989-098-00 | ISS | IDFG | Mainstem upper Lemhi RST | | | s, g, d, t, c, o | s, g, d, t, c, o |
| Lemhi | 1989-098-00 | ISS | IDFG | Lemhi Carcass Survey | s, d, o | | | |

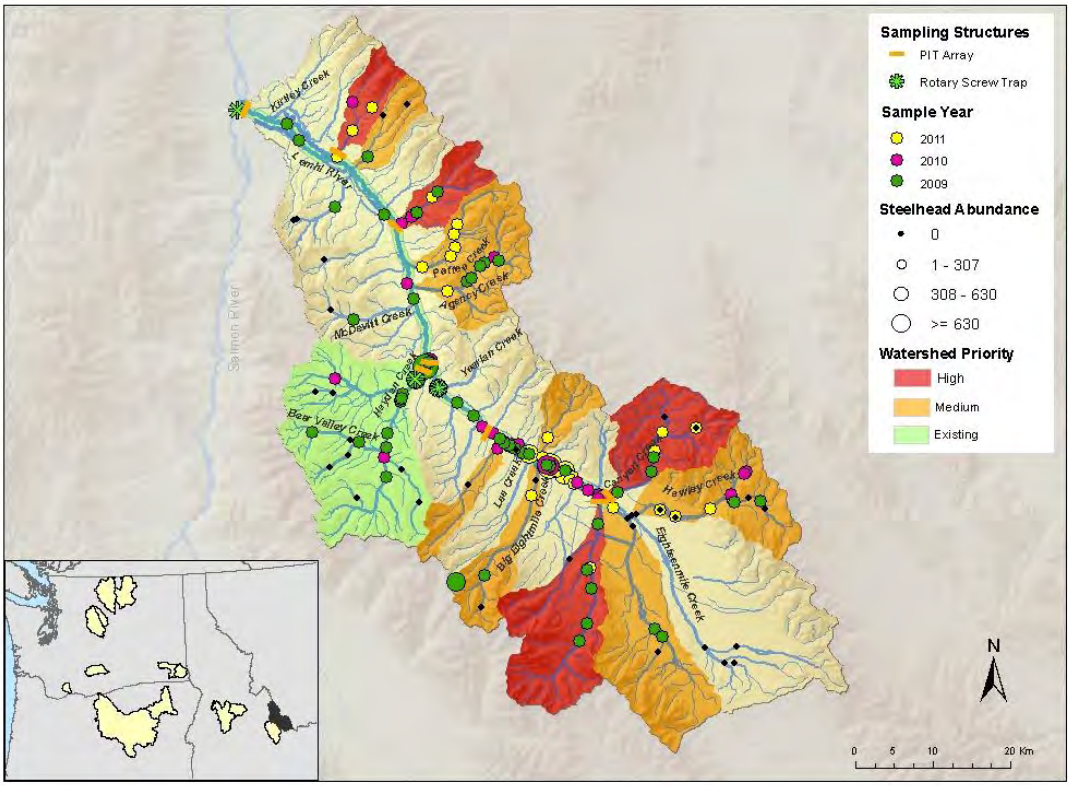


Figure 3-2. Abundance of steelhead at Lemhi River remote juvenile capture and tagging locations sampled in 2009, 2010, and 2011. Also shown is the distribution of sampling infrastructure. Subwatershed coloration identifies habitat available at the inception of ISEMP and high and moderate priority subwatersheds identified for reconnection.

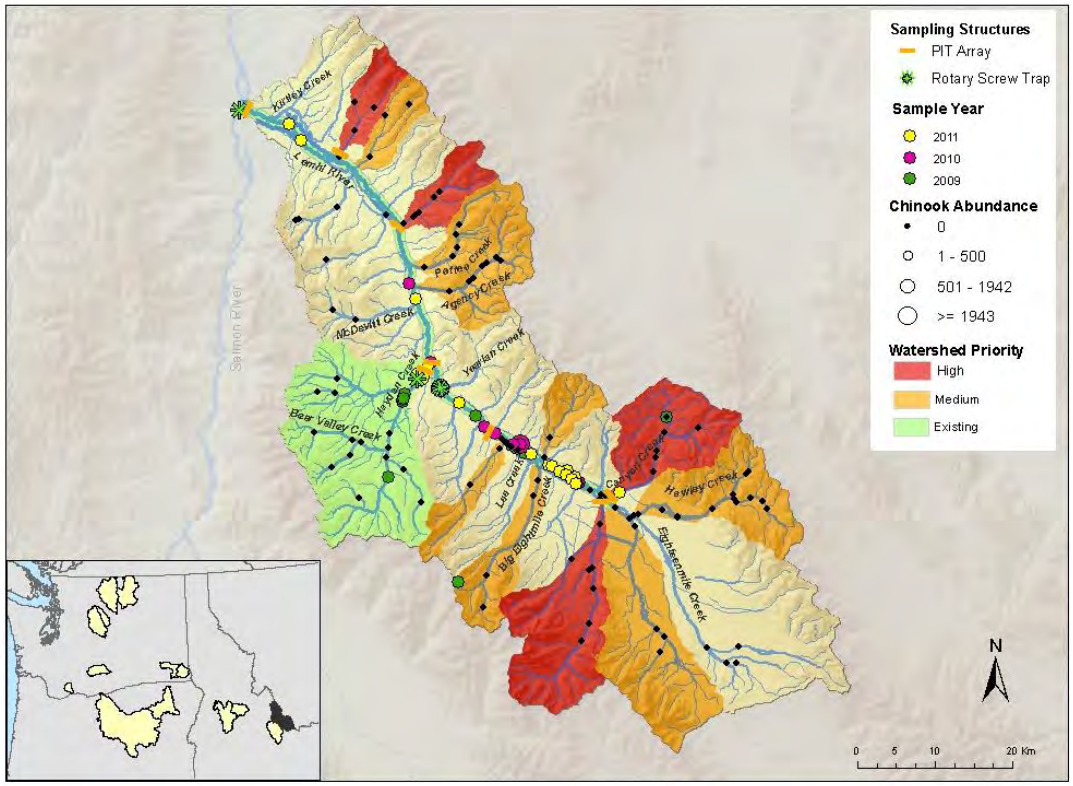


Figure 3-2. Abundance of spring/summer Chinook salmon at Lemhi River remote juvenile capture and tagging locations sampled in 2009, 2010, and 2011. Also shown is the distribution of sampling infrastructure. Subwatershed coloration identifies habitat available at the inception of ISEMP and high and moderate priority subwatersheds identified for reconnection.

### *Habitat Sampling*

Ground-based habitat survey effort is distributed across the Lemhi River Basin using the same GRTS design utilized by remote juvenile capture and tagging effort. Generally, the goal of ground-based habitat survey efforts is the characterization of habitat across tributaries that are currently connected to the Lemhi River and those that are targeted for potential reconnection (high and moderate priority tributaries). Sampling effort at sites identified by GRTS has three components; unique sites, within-year repeat sites, and annual sites. Unique sites are sampled only one time, within-year repeat sites are sampled two or three times within a year, and annual repeat sites are visited at least once every year. Data will be analyzed to evaluate the repeatability of survey attributes and variance among those attributes at sites within a watershed and to determine the amount of sampling effort (i.e., number of sites) required to characterize habitat within sub-watersheds given the resolution of the sampling approach.

For the purposes of this report, we utilized a subset of habitat attributes from 2009, 2010, and 2011 sampling efforts; including, fraction of total habitat composed of turbulent, non-turbulent, and pool, pool volume, d50, pool tail fines, and large woody debris. Standard GRTS

expansions were used to estimate total habitat available by tributary and further aggregated to reporting units (Figure 3-3).

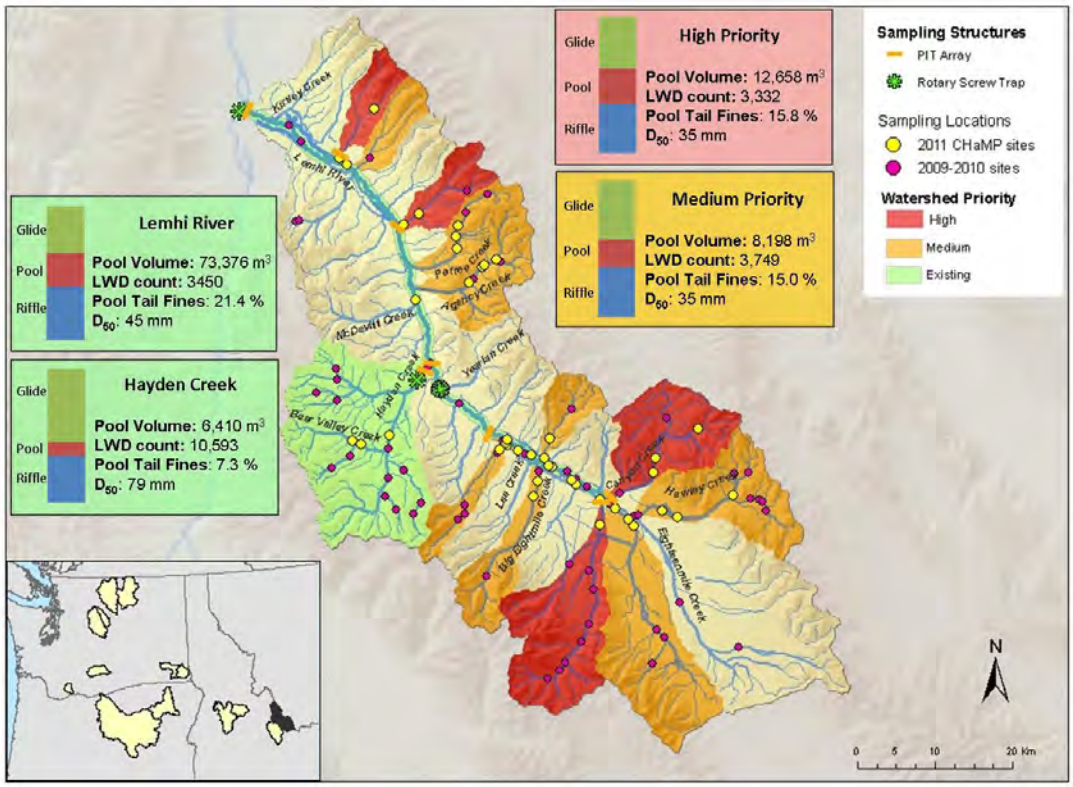


Figure 3-3. Location of 2009, 2010, and 2011 habitat surveys in the Lemhi River. Habitat indicators are summarized by habitat available at the inception of ISEMP (Lemhi mainstem and Hatden Creek) and high and moderate priority subwatersheds identified for reconnection.

### *Survival and Capacity*

As previously described, the ISEMP project was implemented in the Salmon Subbasin in 2009. As such, the information from the first complete brood year of juvenile production will be available in 2013. Given that we have less than a single brood year of juvenile production data, replication is insufficient to generate estimates of survival or maximum capacity using empirical data at this time. Life-stage specific survival estimates were obtained from Bjornn (1978, Table 3-5) based on work conducted in the Lemhi River from 1962 to 1975. Average maximum juvenile densities were obtained empirically from GRTS-based remote juvenile capture and tagging surveys (Table 3-6). For the purposes of this report, we assumed the maximum estimated fish density recorded by habitat type are a facsimile of the minimum expected carrying capacity.

Table 3-5. Life stage specific survival estimates from Bjornn (1978).

| Life Stage | Suvival |
|---|---|
| Spawner-Egg | 5000 |
| Egg-Fry | 0.46 |
| Fry-Parr | 0.84 |
| Parr-Presmolt | 0.89 |
| Presmolt-Smolt | 0.9 |

Table 3-6. Average maximum densities of juvenile spring/summer Chinook salmon based on remote site juvenile capture and tagging surveys.

| Juvenile Densities | Turbulent | Non Turbulent | Pools |
|---|---|---|---|
| Chinook | 0.40 | 0.81 | 2.84 |

***Provisional Results***

For the purposes of this report, we focused on reporting provisional model results for spring/summer Chinook salmon. Although incomplete, survival, abundance, distribution, and growth data for the first brood year of spring/summer Chinook salmon are only one year short of completion as opposed to steelhead which require up to four additional years of data collection. Nonetheless, we caution that the results presented in this report should be treated as a demonstration of the utility of the watershed model and should not be used to inform management decisions.

The distribution of habitat in reporting units can be summarized by pool, turbulent, and non-turbulent habitat (Table 3-7), which in conjunction with empirical observations of average maximum density information and empirical information on fish distribution and emigration rates can be used to generate the scalar term (Table 3-8).

Table 3-7. Distribution of pool, turbulent, and non-turbulent habitat by reporting unit.

| Reporting Unit | Pools | Turbulent | Non-Turbulent |
|---|---|---|---|
| Hayden | 10% | 35% | 55% |
| Upper Mainstem | 30% | 39% | 31% |
| Lower Mainstem | 21% | 42% | 37% |
| Upper Tributaries | 20% | 45% | 35% |
| Lower Tributaries | 21% | 47% | 32% |

Table 3-8. Scalar term as applied to reporting units.

| Reporting Unit | Hayden | Upper Mainstem | Lower Mainstem | Upper Tributaries | Lower Tributaries |
|---|---|---|---|---|---|
| Scaler | 0.7 | 1 | 0.4 | 0.7 | 0.7 |

Assuming that habitat quality and quantity in existing, high, and moderate priority tributaries reflect the mean values described by their reporting unit, using available area in those classes (Table 3-9) enables estimates of changes in productivity (Table 3-10) and capacity (Table 3-11) should they be reconnected.

Table 3-9. Square kilometers of spring/summer Chinook salmon habitat in currently connected (existing) tributaries and high priority tributaries and high and moderate priority tributaries.

| Priority Designation | Hayden | Upper Mainstem | Lower Mainstem | Upper Tributaries | Lower Tributaries | Total |
|---|---|---|---|---|---|---|
| Existing | 0.09 | 0.11 | 0.30 | 0.01 | 0.00 | 0.52 |
| High Priority | 0.09 | 0.11 | 0.30 | 0.06 | 0.00 | 0.57 |
| Moderate Priority | 0.09 | 0.11 | 0.30 | 0.07 | 0.02 | 0.60 |

Table 3-10. Percent change in spring/summer Chinook salmon productivity (smolts/female) estimated given the reconnection of high priority tributaries and high and moderate priority tributaries.

| Restoration Scenario | Percent Change |
|---|---|
| Existing Habitat | 0% |
| High Priority Reconnections | 11% |
| High and Moderate Priority Reconnections | 13% |

Table 3-11. Percent increase in capacity by life-stage for spring/summer Chinook salmon given the reconnection of high priority tributaries and high and moderate priority tributaries.

| Capacity | Fry | Parr | Smolt |
|---|---|---|---|
| Existing Habitat | 0.0% | 0.0% | 0.0% |
| High Priority Reconnections | 11.3% | 13.0% | 14.6% |
| High and Moderate Priority Reconnections | 16.3% | 18.8% | 21.0% |

The model yields a number of estimates that are useful in a management context, for the purposes of this section we focused on changes in juvenile production (smolts per female; Table 3-12) predicted following the reconnection of all high priority tributaries and all high and moderate priority tributaries. Anticipated changes in juvenile and adult abundance accompanying restoration alternatives are illustrated in Figures 3-4 and 3-5.

Table 3-12. Percent change in spring/summer Chinook salmon productivity (smolts/female) estimated given the reconnection of high priority tributaries and high and moderate priority tributaries.

| Restoration Scenario | Percent Change |
|---|---|
| Existing Habitat | 0% |
| High Priority Reconnections | 11% |
| High and Moderate Priority Reconnections | 13% |

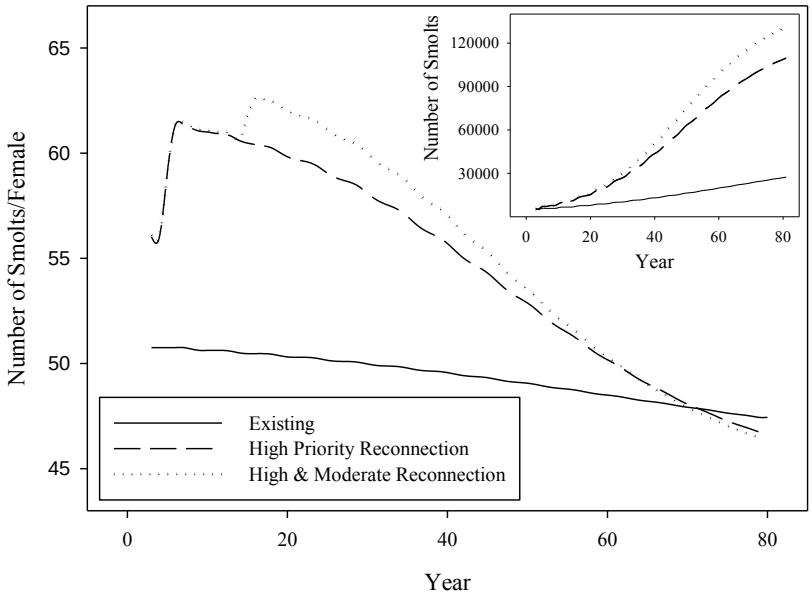


Figure 3-4. Number of smolts per female and total estimated smolt production (inset) given existing habitat, reconnection of high priority tributaries, and addition of high and moderate priority tributaries.

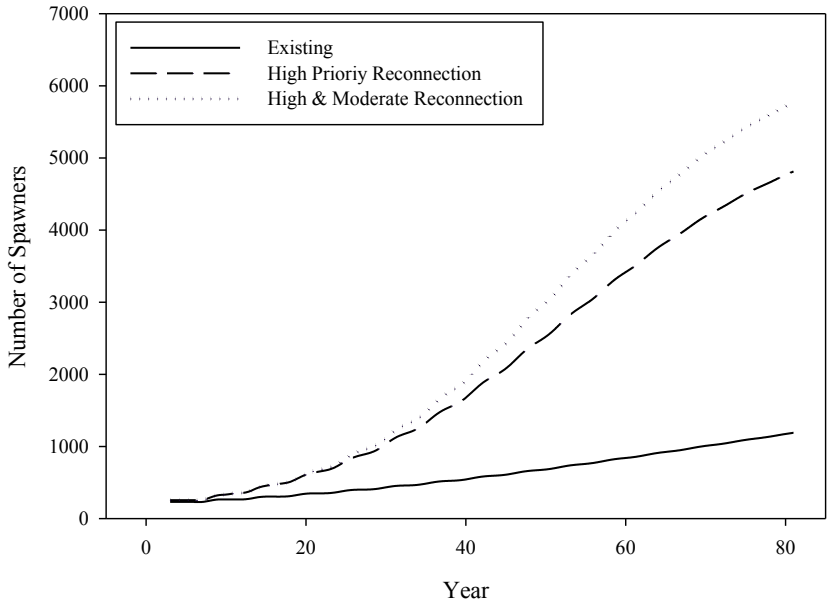


Figure 3-5. Number of spring/summer Chinook salmon adults returning to the Lemhi River given existing habitat, reconnection of high priority watersheds, and reconnection of high and moderate priority watersheds.

### *Management Application*

Although the data necessary to fully populate the watershed model will not be available until 2013, the preliminary results presented in this report illustrate the utility of the model approach for managers. In terms of policy and management, the watershed model provides several useful products:

1) It identifies factors that limit freshwater productivity at specific life-stages, enabling habitat restoration actions to better target problems and conversely to avoid habitat initiatives that are unlikely to address primary limiting factors;
2) It identifies the types and magnitude of habitat alteration most likely to improve freshwater productivity;
3) It provides a platform to evaluate alternative restoration actions to identify/prioritize actions most likely to cost-effectively improve freshwater productivity;
4) It translates habitat quantity and quality to fish abundance, namely identifying reasonable expectations for total production;
5) It identifies the types of monitoring most likely to detect changes in habitat conditions and freshwater productivity within a specified period of time;
6) It provides an analytical tool to quantitatively evaluate change in habitat conditions and freshwater productivity; and,
7) It can be used to predict adult escapement taking into account ocean conditions, harvest, and hatchery impacts.

Utilizing the provisional results described above, the relationship between total smolt production and number of smolts per female suggests that juvenile rearing habitat (at all life-stages) continues to limit total productivity across all three restoration scenarios. This is not a surprising result, but it suggests that improvements in habitat quality in addition to increased access to rearing areas may be necessary to achieve the 20% improvement in freshwater productivity identified for the Lemhi in the biological opinion. Unfortunately, habitat and fish data are not yet sufficient to precisely estimate the freshwater productivity benefit of reconnecting specific individual tributaries. By 2013, the model will enable evaluations of freshwater productivity benefits anticipated by the reconnection of individual tributaries. In turn, this will enable managers to prioritize those tributaries anticipated to yield the greatest benefit, potentially allowing the conserved restoration funding to be targeted towards improvements in habitat quality in exiting and reconnected areas (e.g., channel rehabilitation). Similarly, this provisional application of the watershed model illustrates its utility as an analytical tool, despite the fact that data are insufficient at this time to fully populate all parameters. Lastly, as adult escapement estimates accumulate we hope to observe sufficient contrast in ocean conditions and harvest to enable model predictions under various climate and management scenarios.

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# CHAPTER 4: Analyzing the Relationship Between Fish and Habitat in the Wenatchee Basin Using Boosted Regression Trees.

Author: Kevin E. See

### *Introduction and Methods*

Exploratory models provide a flexible framework to infer which of a plethora of habitat measures provide information about a fish response such as density or growth. Generally, these models fall into two categories: generalized linear models (GLMs) and classification and regression trees (CARTs). GLMs require assumptions about the statistical distribution of the data, which may dictate that some metrics be transformed to meet that assumption. They also require the assumption that the fish response to a particular habitat measurement is linear. Although this may be relaxed by assuming some parametric response curve (e.g. quadratic) instead, this requires forethought into the shape of the fish response curve for each habitat variable. CARTs do not require any assumptions about the distribution of the data, so no transformations are ever needed. They also naturally capture interactions between predictor metrics. Some tree-based methods can also easily identify non-linear relationships between habitat and fish metrics (Friedman & Meulman 2003). CARTs have recently been applied to a variety of ecological data (De'Ath 2007, Elith et al. 2008, Pittman et al. 2009, Knudby et al. 2010) to make predictions such as the probability of occupying particular sites or fish and coral diversity, biomass and abundance. For these reasons, we chose a CART-based method to analyze the relationship between juvenile fish densities and habitat metrics.

A CART model builds a decision tree by creating break points among the predictor variables that minimize the prediction error. The prediction error is calculated by building the tree using only a subset of the data, and then testing the predictions on the remainder of the data. The break points are invariant to monotone transformations of the variables, so no transformations are necessary. A single decision tree, while easy to interpret, is more prone to inaccuracy compared to other modeling approaches such as generalized linear regression (Hastie et al. 2009). One remedy to this problem is known as “boosting”, which consists of fitting an initial tree, then fitting a subsequent tree to the residuals of that tree, and so on. This stepwise approach focuses the subsequent trees on those data points that are not described well by the previous set of trees, dramatically improving the accuracy of the final tree ensemble, called a boosted regression tree (BRT) (Schapire 2002).

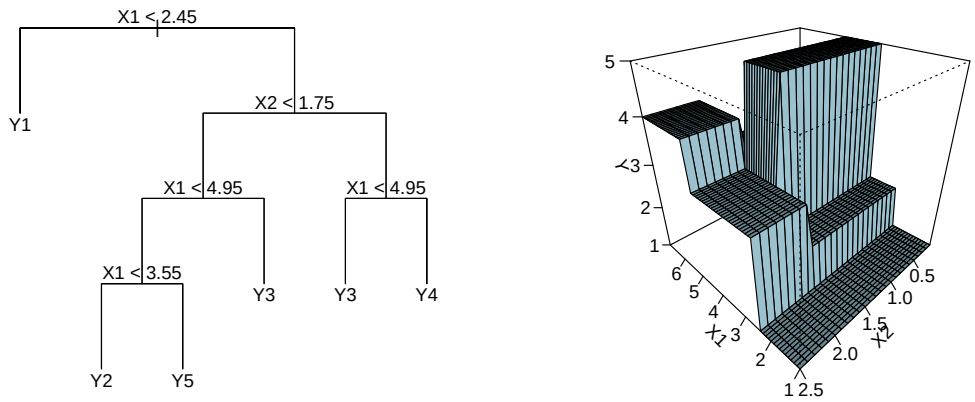


Figure 4.1. A single decision tree (left panel) based on a response variable, Y, and two predictor variables, X1 and X2. The panel on the right depicts the prediction surface.

The inputs to and results from a BRT are easily interpreted. Inputs are untransformed predictor variables of any type (numeric, binary, categorical, etc.) and a single untransformed response variable, again of any type. The results consist of several important components. First, the final ensemble of trees can be used to predict fish density from habitat metrics, or to predict the change in fish density if habitat metrics are altered to one degree or another. Second, a measure of the relative importance of each habitat metric is produced. This provides insight into which habitat metrics should be targeted by restoration work to have the greatest effect on salmon populations. Finally, partial dependence plots can be created, which graphically show the marginal effect on fish when one habitat metric is changed while holding the others at their mean values.

### *Results and Discussion*

For this analysis, we used observed juvenile fish densities and measured habitat characteristics that were collected in the Wenatchee subbasin from 2004 to 2010. We used BRT to determine which habitat metrics are most important in predicting fish densities. Figure 4.2 shows the relative importance of 15 habitat metrics identified from an original 23 metrics as most important for predicting the density of juvenile Chinook. They have been scaled to sum to 100, and listed with the most important metrics at the top, down to the least important. The most important, the year effect (which accounts for differences in spawner abundances as well as environmental conditions not included among the predictor variables) is about twice as important for predicting juvenile Chinook density as gradient or a measure of temperature. This highlights the importance of monitoring habitat and fish for more than one or two years in order to get a reliable picture of juvenile densities: densities in any one year could be very misleading.

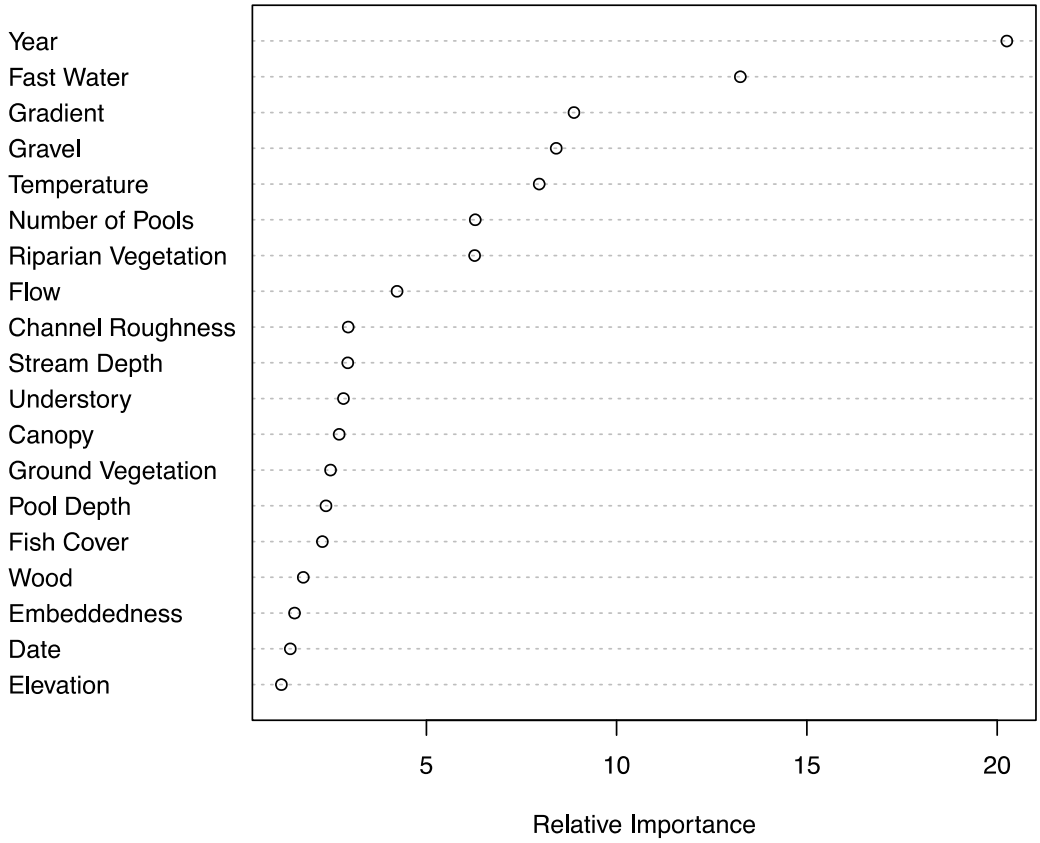


Figure 4.2. The relative importance of various habitat metrics in predicting the density of juvenile Chinook using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010 analyzed using a boosted regression tree approach. The relative influences have been scaled to sum to 100, and the habitat metrics are arranged from most important at the top to least important at the bottom.

Once a particular habitat measure has been identified as relatively important, the next question is what is the predicted relationship between that habitat measure and fish density? Partial dependence plots answer this by showing how predicted fish densities change as one habitat measure changes while all other characteristics of the habitat remain unchanged. Figure 4.3 shows the partial dependence plots for the six most important habitat variables for predicting juvenile Chinook abundance in the Wenatchee data set. Instead of the linear relationships assumed by GLMs, this analysis shows evidence for several thresholds where predicted densities increase or decrease significantly from one side of the threshold to the other, while remaining fairly constant otherwise. Such thresholds can be used to identify limiting factors, and provide quantifiable goals for habitat restoration work, to move the habitat conditions from one side of the threshold to the other.

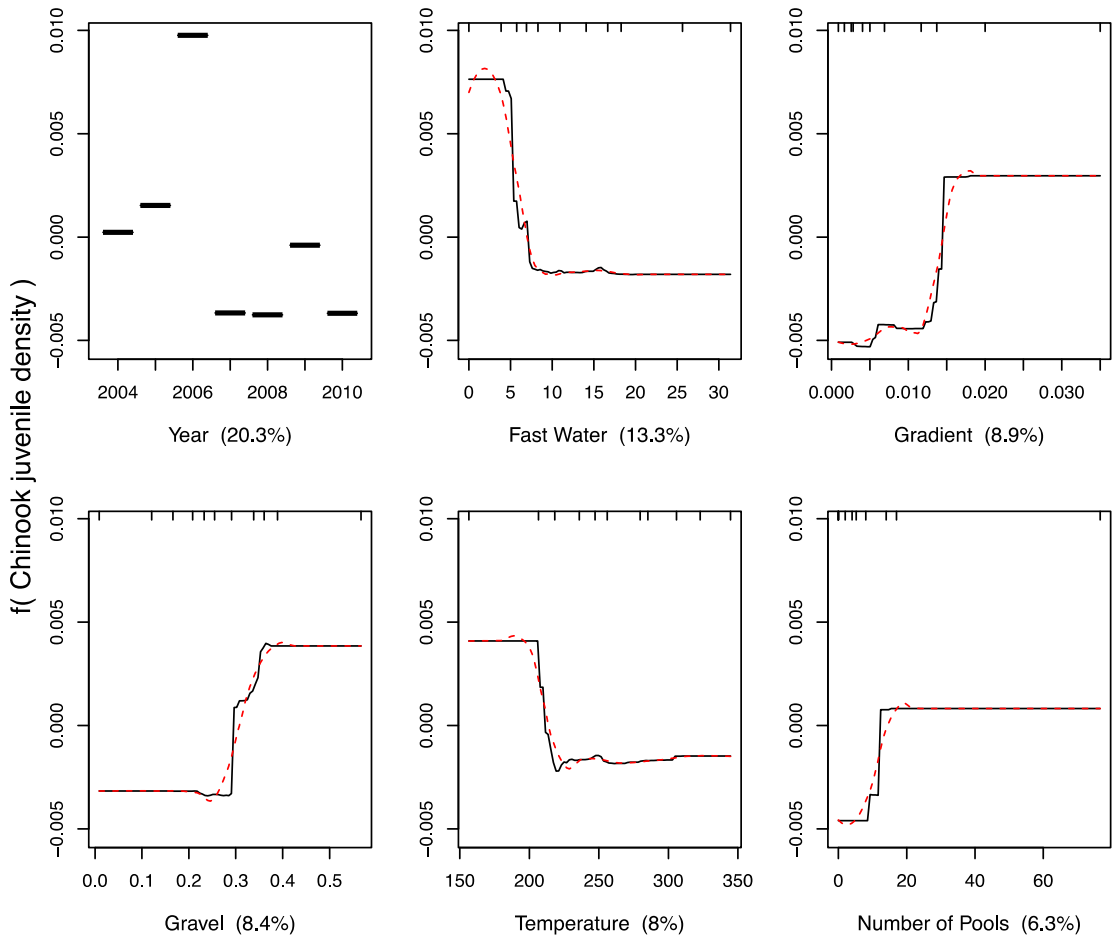


Figure 4.3. Partial dependence plots showing the marginal effect of the six most important habitat metrics identified from a BRT on juvenile Chinook densities using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010. The y-axis is a function of the predicted value of juvenile density, which has been centered on 0. Higher values on the y-axis correspond to higher expected juvenile densities, and vice-versa. The black line describes the predicted value for each value of the habitat metric, based on this dataset. The red dashed line is a smoothed version of the black line. Along the top of each plot, the tick marks show the deciles of the data for that habitat metric. For example, 90% of the site visits had less than 20 pools per river kilometer.

The results of this analysis confirm some known relationships between habitat and fish densities. Juvenile Chinook prefer pools (less fast water), require a certain amount of coarse gravel and have particular temperature preferences. One of the benefits of this type of CART analysis is the ability to identify the fact that for all of these relationships, habitat thresholds are apparent (Figure 4.3), which can guide restoration work. For example, predicted densities of juvenile Chinook are high for low values of fast water, decline steadily for mid-range values and level off at higher values. This implies that sites with low values of fast water are important for juvenile Chinook and that restoration actions should target sites with too much fast water area, i.e., restoration actions should create slow water refugia. Similarly, restoration actions that increase the percentage of coarse gravel from 30% to 40% should be effective, but increasing that percentage from 40% to 50% or from 15% to 25% may not have the same effectiveness

because neither of those actions shifts the amount of gravel across the threshold important to fish.

Different species have different habitat needs, which can be seen from the results of a similar BRT analysis on steelhead densities in the Wenatchee from 2004-2010. For steelhead, the metrics considered relatively important (Figure 4.4) and how those metrics relate to fish densities (Figure 4.5) are different compared to Chinook (Figures 4.2 and 4.3). These methods did a better job of predicting Chinook than steelhead densities (as measured by mean deviance, 0.001 vs. 0.008), perhaps due to Chinook having a more consistent life history. The ISEMP monitoring and this type of analysis is able to detect those differences between species, which should shift restoration actions and priorities, depending on the target species.

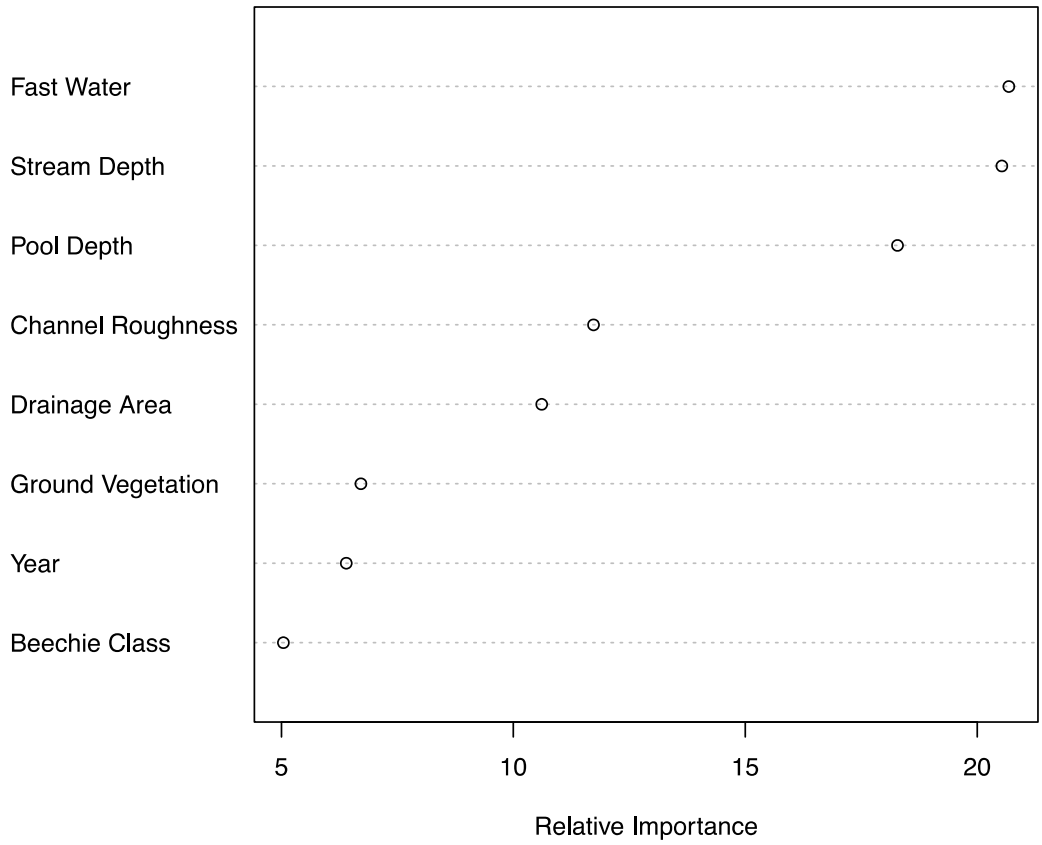


Figure 4.4. The relative importance of various habitat metrics in predicting the density of juvenile steelhead using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010 analyzed using a boosted regression tree approach. The relative influences have been scaled to sum to 100, and the habitat metrics are arranged from most important at the top to least important at the bottom.

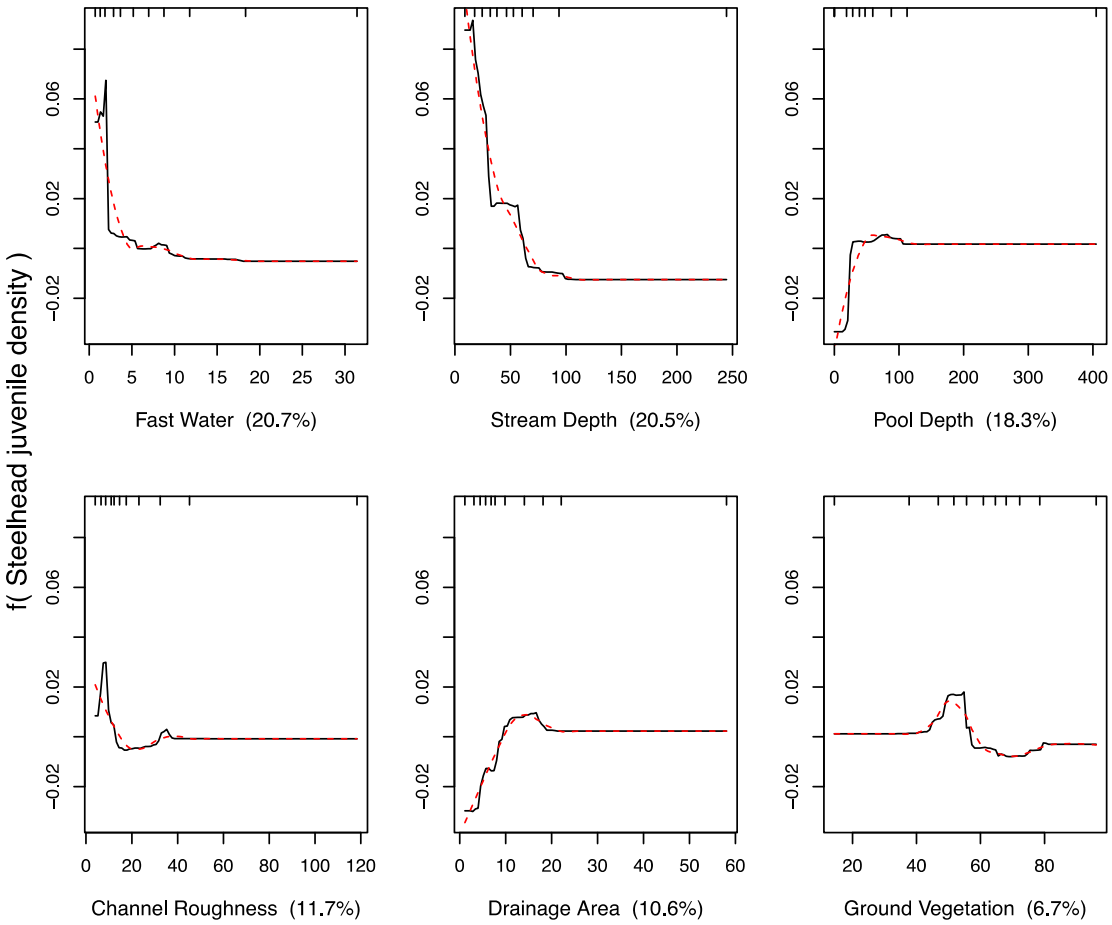


Figure 4.5 Partial dependence plots showing the marginal effect of the six most important habitat metrics identified from a BRT on juvenile steelhead densities using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010. The y-axis is a function of the predicted value of juvenile density, which has been centered on 0. Higher values on the y-axis correspond to higher expected juvenile densities, and vice-versa. The black line describes the predicted value for each value of the habitat metric, based on this dataset. The red dashed line is a smoothed version of the black line. Along the top of each plot, the tick marks show the deciles of the data for that habitat metric. For example, 90% of the site visits had less than 20 pools per river kilometer.

Before these results should be used in management decision-making, additional work needs to be done to more specifically define threshold levels and to confirm consistency outside of the Wenatchee subbasin. Although not presented here, we have also conducted BRT analyses on presence/absence data in the Wenatchee for Chinook and steelhead, steelhead growth rates in Bridge Creek, densities of Chinook and steelhead across all basins with 2011 Columbia Habitat Monitoring Program (CHaMP) data, and within the Salmon basin for the period 2009 – 2011. Although some similar habitat characteristics were identified as important across many of these data sets, there are enough differences to suggest that different habitat characteristics may be more important in some subbasins compared to others.

Our results demonstrate that this analytical framework can be used to answer questions such as what habitat characteristics should be targeted for restoration and how much restoration

is necessary. Given habitat and fish status and trend data, these methods can be used to help answer the question "Are habitat restoration actions effectively helping salmonid populations recover?"

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# CHAPTER 5: Evaluation of Riparian Fencing as a Restoration Tool in the John Day Basin

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### *Introduction*

Livestock grazing has been cited the most pervasive source of riparian and instream habitat degradation in the western U.S. (Elmore et al. 1994, Fleischner 1994), affecting ~80% of all western riparian and stream ecosystems (Platts 1982, Belsky and Uselman 1999). In the Pacific Northwest, grazing is presumed to have negatively impacted the quality of habitat for salmon and steelhead populations through changes to riparian vegetation and channel morphology. In an effort to mitigate these effects and aid the recovery of salmonid populations, land managers throughout the region have installed fences to exclude livestock from riparian areas and stream channels (Sarr 2002). Studies of channel response to grazing exclosures have produced mixed and often inconclusive results (Belsky and Uselman 1999; Sarr, 2002). Variable results have been attributed to a number of factors, including inadequate or incomparable study designs, inherent between and within-site variability, insufficient study replication, grazing within exclosures by small wildlife, prior grazing history, different recovery rates, and outside influences (Belsky and Uselman 1999). Furthermore, little is known about geomorphic adjustments to grazing exclusion at large spatial and temporal scales. Most research has evaluated geomorphic adjustment to exclusion from grazing at a single paired site (Knapp and Matthews 1996, Nagle and Clifton 2003). Only a few studies have evaluated response patterns between multiple sites at a broader spatial scale, such as a geographic region or watershed (Magilligan and McDowell 1997, Kauffman et al. 2002).

Over the past couple decades, funded by BPA, the Oregon Department of Fish and Wildlife (ODFW) has built exclosures over 200 miles of riparian corridors at 90 locations throughout the John Day River basin in an effort to mitigate the potential impacts on salmonid habitat associated with livestock grazing. In this study, we assess whether the grazing exclosures result in altered channel morphology and improved habitat conditions for a subset of streams in the John Day watershed of eastern Oregon.

Riparian exclosures are a very common passive restoration approach. However, changes to the riparian corridor and stream channel after exclosures are built can take a decade or more to occur, whereas decisions of whether to continue with this approach in order to provide necessary benefits to endangered populations is an immediate need. Therefore, ISEMP conducted a two year study to evaluate whether benefits of activities that have already been in place for up to 25 years can be observed to inform future restoration actions.

### *Methods*

In 2009, ISEMP sampled eight exclosures sites and eight control sites to evaluate geomorphic, riparian, and biological changes that may have occurred as a result of the release of grazing pressure. In 2010 ten additional paired sites were sampled. Treatment (fenced) and control (unfenced) pairs were evaluated and selected based on criteria that will minimize anthropogenic and confounding variables and increase the likelihood that differences in reaches

will be due to differences in land use (Kaufman et al. 2002). An extensive review of grazing literature aligned the study's site selection to be based on the following criteria:

- Streams with salmonids,
- The stream must be wadeable,
- Knowledge of the history of the exclosure (grazing records),
- Sites are contiguous ungrazed exclosures and unfenced grazed reaches adjacent to each other,
- Sites are not inclusive of or directly adjacent to mining operations or water diversion reservoirs
- Sites should have limited human implemented restoration (i.e. check dams, logs steps, in-channel structures, rip rap or mass vegetative bank stabilization),
- No significant tributaries should intervene between the treatment and control reaches,
- Channel reaches shall be as geomorphically similar as possible and shall not have major bedrock constraints. Geomorphic attributes shall include streams with similar valley confinement, valley slope, and elevation, and
- For the purposes of our study, landowner permission played a critical role in site selection. Exclosures in the John Day Basin are installed on both private and public lands. When exclosures are installed on private lands, the property is leased and maintained by the Oregon Department of Fish and Wildlife, but activities outside the realm of routine fence maintenance must be approved by the landowner.

The site selection criteria were evaluated with GIS, aerial photos, and verbal accounts from landowners. Watershed data was calculated with the use of ArcMap GIS software, utilizing a 10 meter digital elevation model. Since landscapes are dynamic systems and are prone to change, final site selection was made only after a site walk through.

We have surveyed 14 locations throughout the John Day Basin consisting of one exclosed site (treatment) and one grazed site (control). At each site geomorphic units are delineated, based upon stream bed morphology, and each unit is assigned a code that describes the habitat it provides for steelhead (e.g. riffle, pool, glide, and cascade). Widths, depths, and length of each unit were measured using tape measures and depth rods as well as total station surveys. At each site riparian, habitat, and fish population variables were collected at the reach and unit scales. The sites that were monitored ranged in exclosure age from 2 to 25 years allowing us to look at the effects of exclosure and exclosure age on our measured variables.

For all results, we created confidence intervals (CI) around the difference between control (open to grazing) from treatment (exclosure) across the age of exclosures to get an estimate of the time towards recovery. For field experiments, a CI of 90% is commonly used to assess statistical significance. For management decision (i.e. should riparian fencing projects continue), a less stringent significance level is helpful (not as rigorous as experiment but far more helpful than intuition), thus we also report 80% CI.

***Riparian Zone***

Vegetation surveys were be conducted along two greenline transects (Winward 2000, Coles-Ritchie et al. 2007, Heitke et al. 2007) along which dominant cover types and percent cover for each species was estimated. A wetland indicator status (as per NRCS plant database, region 9) was assigned to each species and was weighted by the percent cover estimate along the greenline (Coles-Ritchie et al 2007, Heitke et al 2007). The weighted values were averaged by transect and reach to define a wetland indicator value (Coles-Ritchie et al 2007, Heitke et al 2007) by transect and site that could be compared between exclosed and grazed sites.

Stream shading was measured using a Solar Pathfinder (http://www.solarpathfinder.com) at 20 meter intervals along the length of the stream (Bouwes et al. 2011). The Solar Pathfinder and associated software measures the amount of shading (Clarke et al. 2004, Zoellick and Cade 2006) from digital photos of a convex hemisphere. The amount of solar radiation input, occurring at any time and/or date can be extrapolated from this information using the associated software providing data that can be prepared between exclosed and grazed sites.

***Habitat monitoring***

The protocols used to evaluate these potential geomorphic changes including detailed geomorphic surveys using a total station, and methods based on the preliminary habitat monitoring protocol developed by ISEMP (Bouwes et al. 2011). Metrics that were used to compare channel differences included channel unit distribution, width, depth, substrate type, large wood, and bank attributes.

The stream reach was categorized by bedform units according to the ODFW's Aquatic Inventories protocol (Moore et al. 2008). The channel units were classified by major categories of pool, glide, riffle, rapid, cascade, and step. Each major unit was then subclassified, by a more specific channel unit type (i.e. plunge pool, rapid with boulder). The channel unit name and reference numbers were preserved throughout the monitoring.

Using the geomorphic unit data collected during site setup and total station surveys we are able to define the area within each site made up of riffles and pools to create a riffle pool ratio. This ratio can then be used to compare the habitat arrangements between exclosed and grazed sites.

The measurement of streambank and bed morphology was conducted along transects within each channel unit. Transects were laid out at the bottom of each channel unit and labeled as a percent of the unit length (i.e. bottom transect equals 0% middle equals 50%) with at two transects within each channel unit. If the channel unit was long or complex, additional transects were used to describe the full range of variability in the unit.

Overall bed substrate composition was visually estimated by channel unit using the size classes outlined by Peck et al. (2006; Table 11). These data were collected for comparison to more rigorous techniques used within the geomorphic monitoring. Wolman pebble counts (Wolman 1954, Schuett-Hames and Pleus 1996, Kondolf 1997) were conducted in riffles within each site (300 pebbles per site; Bouwes et al. 2011). With this data we can plot the substrate distribution by size class and compare the distributions of exclosed and grazed reaches. For this example the median grain size (D50) was used.

LWD that was 10 cm in diameter and at least 1 m long, whether by its self or within an aggregate, was measured (Heitke et al. 2007). Counts of aggregates with a total measurement of greater than 10 cm diameter and 1 meter length were also quantified as an additional source of cover and velocity refugia. Boulder counts were conducted by channel unit and were binned by size, defined as > 0.5 m (Moore et al. 2008), along with cobble of sufficient size to provide refugia from velocity for juvenile *O. mykiss* and young of the year (defined as 0.25 -0.49m).

Percent fish cover was estimated visually as per each of 7 variables. HOBO Pendant Temperature loggers were placed in the stream at the top and bottom of each reach to provide data that will describe the rate of change in water temperature along the length of the reach. Temperatures in all pool habitats were collected at multiple locations within the pool including: at each bank where bank material meets bed material, at the deepest point in the pool, and at the surface of the pool (Nielsen et al. 1994). To detect possible pockets that may provide thermal refugia to juvenile *O. mykiss* by way of conduction from bank and bed material, stratification, and hyporheic or groundwater inputs (Nielsen et al. 1994, Boyd and Kasper 2003).

Aquatic invertebrate sampling was conducted following the protocol outlined in Bouwes et al. (2011). This sampling includes benthic samples from each reach, to describe the invertebrate community, and drift samples, to describe the drift rate of aquatic invertebrates and terrestrial invertebrates that have entered the drift.

***Fish monitoring***

Steelhead populations were sampled in the early summer, using electroherding, in a two pass mark-recapture event during which all *O. mykiss* >70mm were PIT tagged (Bouwes 2010). This was conducted by channel unit (Muhlfeld et al. 2001) using electro-fishing equipment to herd fish a seine or multiple dip nets. O. mykiss that were ≥80 mm received a 12mm PIT tag which was used to track habitat use, growth, survival, and movement during subsequent fish sampling events. Additional data collected included fork length, weight, caudal fin clip (isotope analysis), and scales (growth rate).

A mobile PIT tag antenna was used in a resight event in mid-summer to determine if marked fish remained within the reaches and what habitat is being utilized. Sampling was conducted by channel unit (pool, riffle, etc.) and took place twice in each reach.

A third sampling event took place in the early fall during which we electro fished each site using three passes in an effort to recapture as many tagged fish as possible. A weight and length was collected on all *O. mykiss* that were caught to provide a growth rate, over the summer months, as well as data needed for survival estimates.

***Results***

The literature describes the direct impacts of cattle grazing in riparian zones as consumption and trampling of vegetation and streambanks resulting in decreased plant diversity, weaker streambanks, and increased fine sediment input to the stream. We found a statistically significant difference between riparian zones grazed (treatment) and exclosed (controls) in response in the wetland indicator values at sites > 7 years old (Figure 1). A statistically significant response was also found in stream shading at all of the exclosed sites that were surveyed (Figure 2).

With increased stream shading we expected the rate at which water temperature increases through a site to decrease and the difference in this rate would be greater in older sites. We were unable to detect a statistically significant difference the warming rate between exclosed and grazed sites at any exclosure age but we did notice that the older exclosures did show a slightly greater difference (Figure 3).

With the removal of livestock and increased vegetation, overland flow decreases and fine sediment inputs are expected to decrease. Over time fine sediment in the reach will move downstream leaving larger substrate exposed. We found that the median grain size (D50) towards larger grain sizes, but the difference was not statistically significant (Figure 4).

We were unable to find a statistically significant difference in the riffle: pool ratio in older exclosures but did document a statistically significant difference in exclosed sites < 11 years old (Figure 5). Some anecdotal evidence leads us to believe that fenced sites were often selected because they were more degraded to adjacent areas that landowners were willing to fence. This would result in the difference that is shown at the younger aged exclosures.

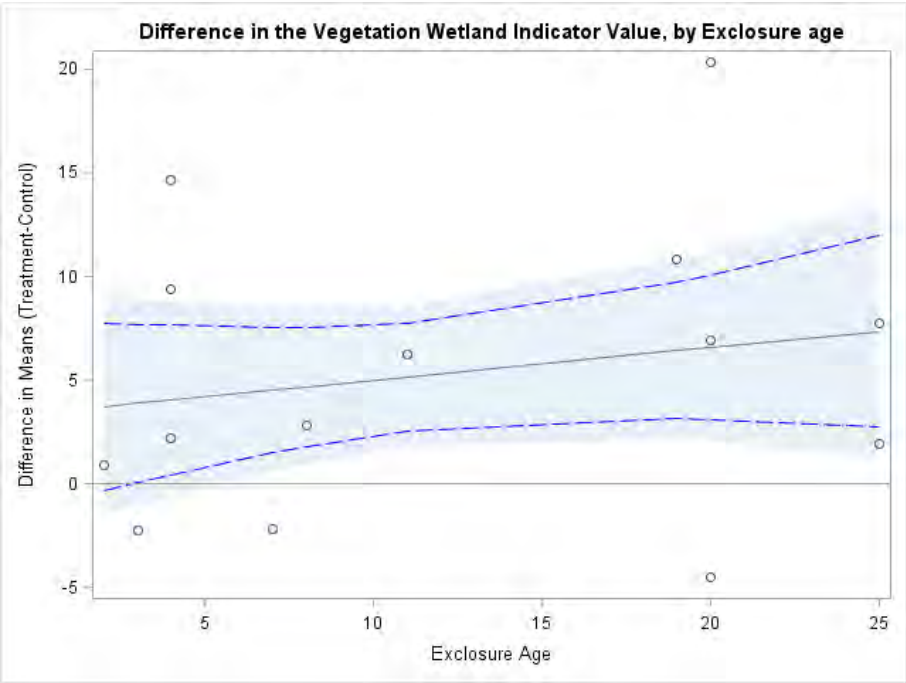


Figure 1. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, in wetland indicator values for the greenline plant communities. Statistically different values observed at exclosure sites > 6 years old.

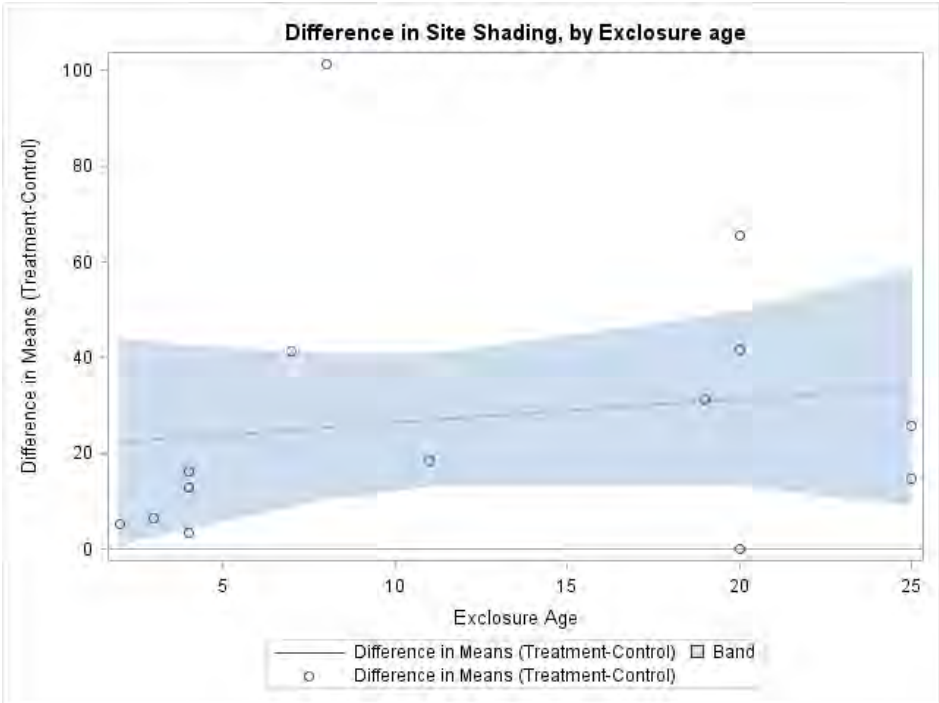


Figure 2. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs) , across different ages of exclosures, in shading, as measured using the Solar Pathfinder.

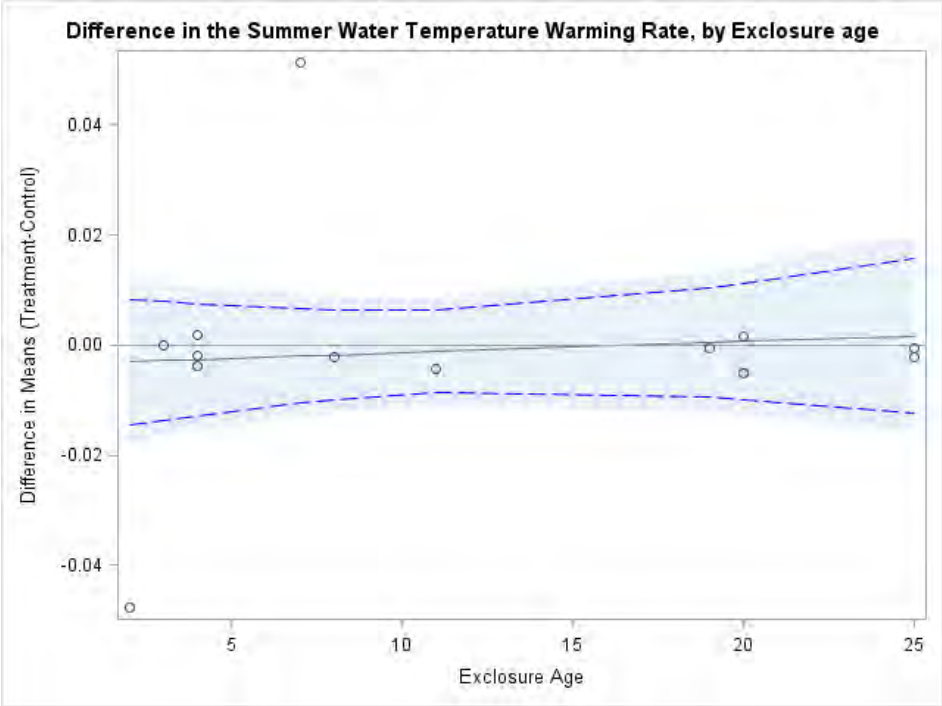


Figure 3. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, in rate of temperature change.

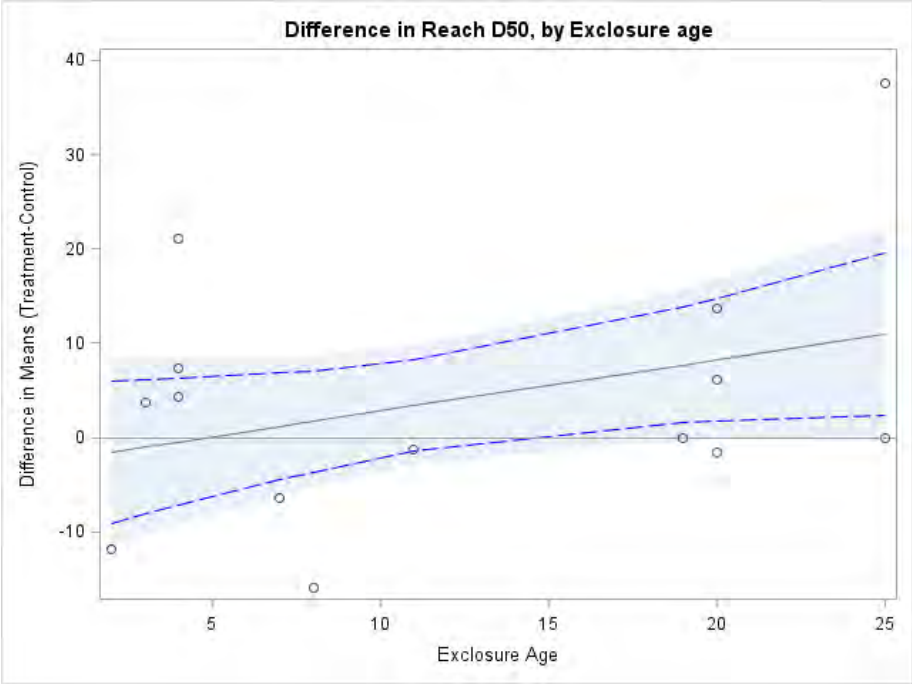


Figure 4. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, of the median grain size (D50).

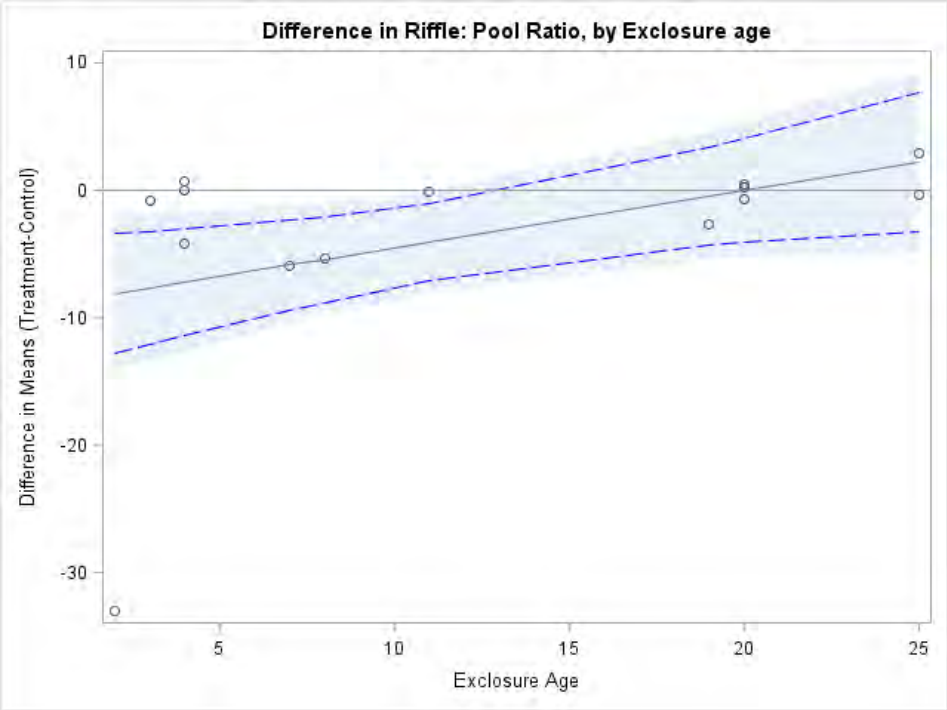


Figure 5. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, in the riffle: pool ratio to describe the difference in habitat arrangement.

Due to the size of age 0 fish and the gear that we used in our mark-recapture events the likelihood of capturing age 0 fish increased as the fish over the summer months as fish grew. By excluding age 0 fish from estimates of density and production we can account for the differences in capture likelihood (Platts and McHenry 1988, Knapp and Matthews 1996) between locations sampled earlier in the season versus later in the season. We found no statistically significant difference in steelhead biomass (g/m2) between treatment and control reaches (Figure 6).

Bayley and Li (2008) found that age 0 steelhead showed a strong preference for exclosed reaches at sites they monitored in the John Day Basin so we also looked at fish density using fish per meter squared. We found that density of age 0 steelhead (fish/m2) was significantly greater in exclosures greater than 21 years (Figure 7).

We were unable to detect a treatment effect on steelhead summer growth rates at any sites (Figure 8). Since summer growth rates were not statistically significant it is no surprise to find no statistically significant difference in summer fish production (Figure 9).

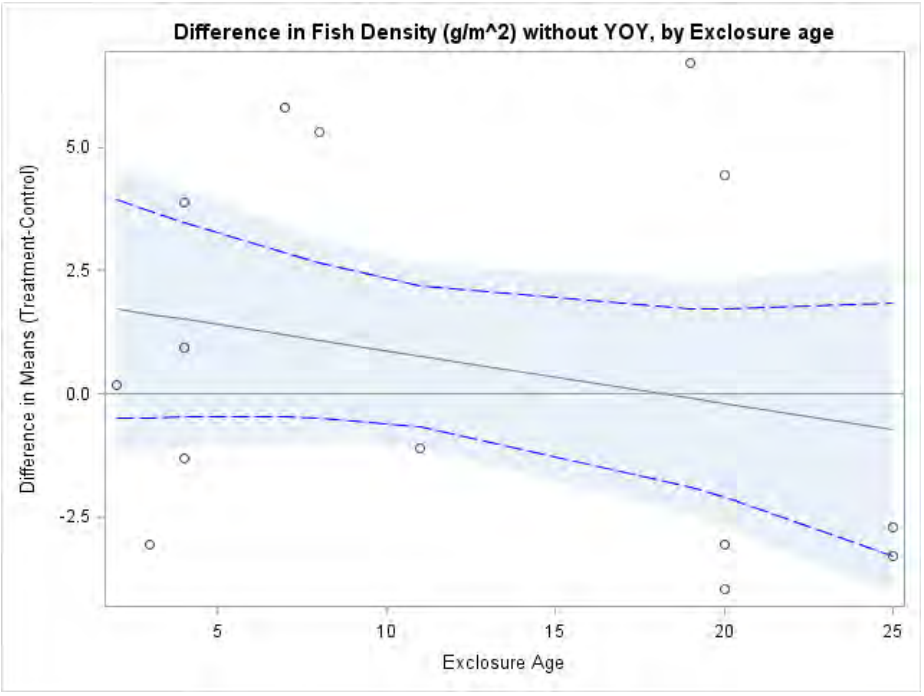


Figure 6. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, in fish biomass (g/m2) excluding age 0 steelhead.

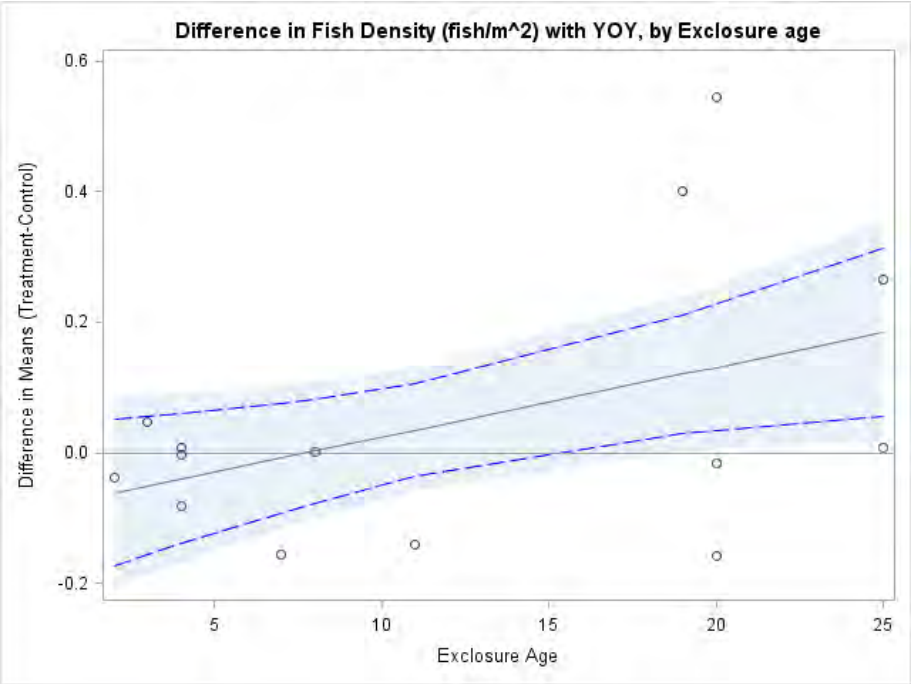


Figure 7. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, of fish density (fish/m2), including age 0 steelhead.

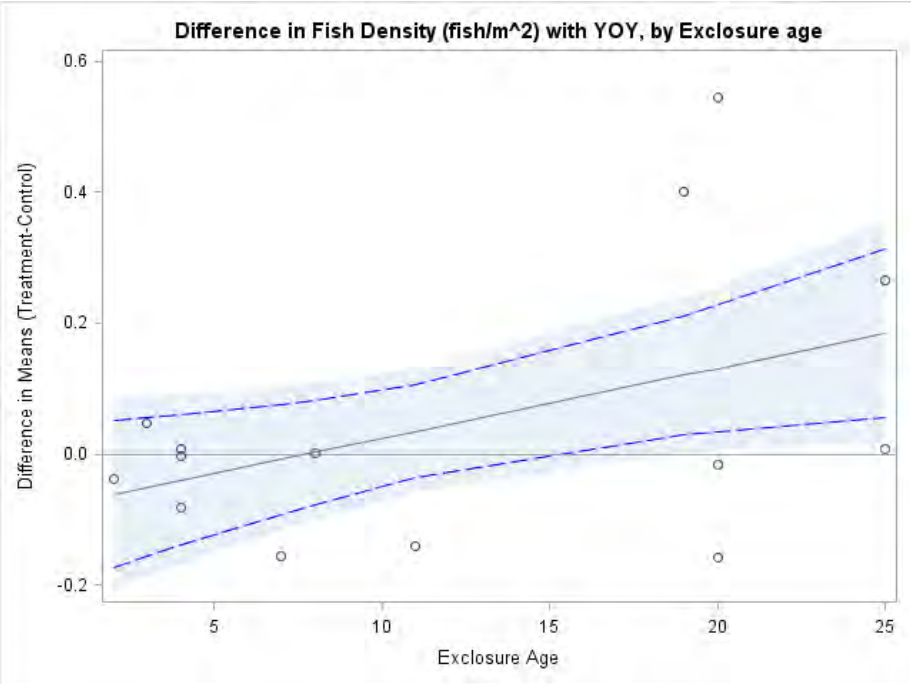


Figure 8. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, in steelhead summer growth rates, excluding age 0 steelhead.

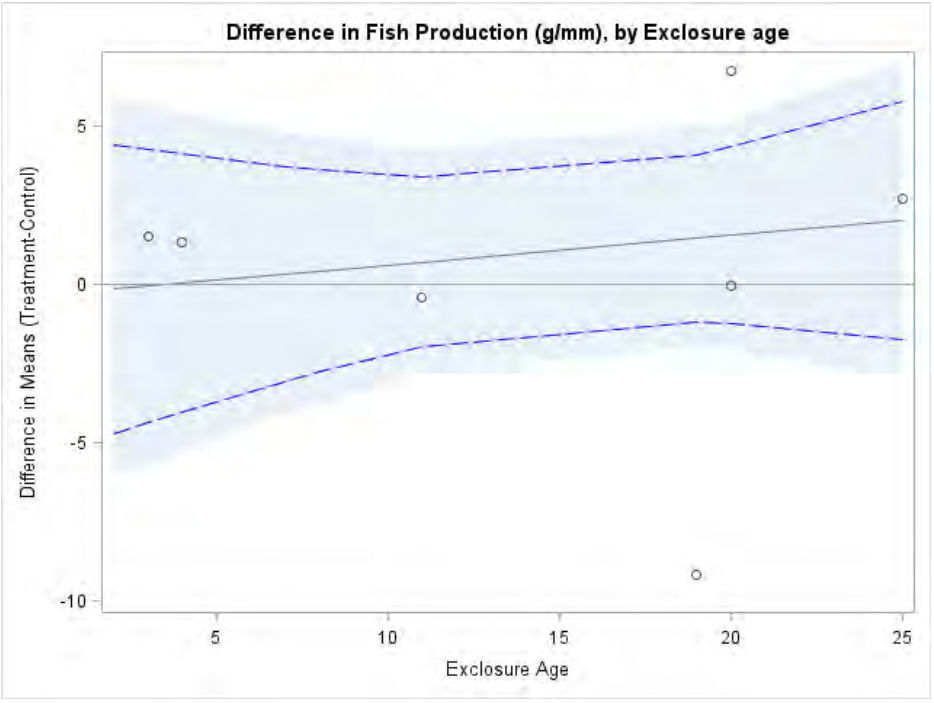


Figure 9. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, in fish production, excluding age 0 steelhead.

### *Discussion*

While we were able to detect changes to the riparian area due to exclosures, we were unable to detect associated responses to fish habitat or steelhead performance. Change occurring to riparian vegetation is expected to be the first response to the cessation of grazing, which then leads to changes in stream morphology. Fish are interacting directly with stream morphology and indirectly with riparian vegetation. From these results, we cannot infer whether grazing exclosures have elicited channel recovery (for a far more complete geomorphic evaluation that ISEMP conducted with this study see Salant and Schmidt 2011) or subsequent fish responses to grazing impacts in this basin. Explanations for the lack of response includes: the channels may not have been altered prior to the construction of exclosures; the history of grazing in the basin may have been so long-term, widespread, and/or intense that it altered channel conditions beyond the ability of the channel to adjust and recover (i.e., caused a regime shift into a new stable state); trends suggest some recovery, but more time may be required for changes in fish habitat and fish performance to occur; other sources of degradation may override the effects of grazing and grazing exclosures, such as the eradication of beaver; there is truly no benefit to fencing; or the benefits have occurred but we simply cannot tease them apart from environmental variability. A study design that included pre-project evaluation in both treatment and controls would have resolved some of these confounding explanations. Thus, a post-hoc study design, such as what we had to undertake, is not likely to be powerful enough to detect differences if they really do exist.

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# CHAPTER 6: Designing Watershed Scale Experiments within the Intensively Monitored Watershed Framework

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### *Introduction*

Dramatic declines in salmon and steelhead populations in the Pacific Northwest have been attributed to harvest, hatcheries, hydro development, and stream habitat destruction (Nehlsen et al. 1991, Jelks et al. 2008). Although these stressors are being addressed to varying degrees, stream habitat restoration is the primary approach for recovering steelhead and salmon populations within the Columbia Basin (BiOP 2008). For example, a billion dollars are spent annually in the US on stream restoration (Bernhardt et al. 2005) and almost 100 million dollars are spent annually on stream restoration for salmon and steelhead in the Pacific Northwest (NOAA 2007). However, past restoration efforts have rarely included effectiveness monitoring programs to determine if projects have increased salmon and steelhead freshwater production. Also, restoration efforts are often hampered by funding and political constraints (e.g., landowner cooperation and competing management objectives) and are rarely implemented over large contiguous areas with specific ecological and hydrological objectives (Katz et al. 2007, Fullerton et al. 2010). As such, despite the large expenditure on stream restoration, there is almost universal agreement for the need to better understand the linkages between restoration and population response which requires detailed implementation and effectiveness monitoring (Bernhardt et al. 2007, Katz et al. 2007).

### *Ecosystem Experiments*

Ecosystem experiments are arguably the most direct method available for detecting a population or environmental response to management (Carpenter et al. 1995). Ecosystem scale experiments have contributed greatly to our understanding of ecological processes within watersheds (Likens et al. 1970, Hartman and Miles 1996), and results from many of these studies have led to changes in management strategies (Likens et al. 1978; Wright et al. 1993; Hartman et al. 1996). Watersheds are well suited for ecosystem experiments because they define natural boundaries of climatic conditions, nutrient cycling, sediment and water routing, and species migration and movement. Whole watershed experiments will likely have a far greater chance of detecting a population level response because they are more likely to trigger a population response that can be detected above the considerable natural variability of natural systems (Roni et al. 2010a). Also watershed scale restoration is implemented at the scale that species are typically managed at, unlike small and isolated restoration actions that are often difficult to evaluate in terms of management success (Fullerton et al. 2010, Roni et al. 2010a).

However, there are limitations to watershed scale restoration actions and what can be learned from them when they are conducted in an experimental fashion. One of the most serious limitations of these large scale experiments is that they are very difficult, if not impossible, to replicate. Replication is a fundamental component of many scientific experiments (Green 1979), but finding replicate watersheds is often impractical for logistical reasons (e.g., budgetary limits,

land ownership, political boundaries, etc.) or ecologically infeasible (e.g., each watershed is likely to respond differently due to biological and geophysical differences).

Hence, historical evaluations of restoration, if conducted at all, have mostly been limited to site level evaluations. Site level evaluations have mostly produced equivocal results of their effectiveness because they have not accounted for other factors (Thompson 2006); have looked at local effects that may simply reflected a redistribution of individuals within a population rather than benefits to the population (Riley and Fausch 1995); are conducted at insufficient spatial and temporal scales to observe a population benefit; or have not used proper experimental approaches (Roni et al. 2010b).

However, there are some examples of restoration activities that have been implemented in an experimental setting that have provided data on fish responses (Cederholm et al. 1997, Solazzi et al. 2000). These examples provide information that Roni et al. (2010a) pointed out are what managers and funders of salmon habitat restoration are most interested in, namely:

- How many fish are created by restoration,
- How much habitat needs to be restored to significantly increase fish abundance, and
- How much habitat needs to be restored to achieve recovery of threatened and endangered populations.

Restoration projects that have been able to provide information on their effect on salmonid production have had a direct influence on the availability of fish habitat (i.e., instream structures, floodplain reconnection, or elimination of fish migration barriers), and have intensive habitat and fish monitoring pre and post project (Roni et al. 2010b). However, there is an urgent need for a more coordinated approach to understanding the effectiveness of restoration actions

***Intensively Monitored Watersheds***

One recent approach to evaluating restoration actions is the Intensively Monitored Watershed Program (Roni et al. 2002, Bilby et al. 2005, PNAMP 2005). Coordination at the regional scale has been initiated to develop a network of IMWs assessing a variety of actions, limiting factors, and watershed types. This coordination should lead to a better understanding of fish-habitat relationships and empirically based recommendations on how restoration should be prioritized and implemented as a recovery strategy. The goal of the IMW program is to improve our understanding of the relationship between fish and their habitat (Bilby et al. 2004; PNAMP 2005). Financial and logistical constraints make the IMW approach impractical for all restoration actions. Therefore, the IMW approach must be implemented in the framework of experimental management where the goals are to benefit the resource while maximizing learning so that the result can be extrapolated to other situations (Walters 1986 ). Generalization beyond a single system requires knowledge of mechanistic interactions or multiple ecosystem studies (Carpenter et. al. 1995). Directed research within an IMW might reveal the mechanisms by which the environment influences population performance of salmonids in a cost effective manner. In addition, the lessons learned from this network of IMWs will enable the region to implement further restoration with greater confidence without the rigorous effectiveness monitoring of the IMW approach.

### *Experimental Approaches*

#### *Past Experimental Approaches*

Multiple experimental designs exist to assess the impacts of stream restoration efforts. Most of these designs were developed to evaluate the impact of some human perturbation on a resource (Box and Tiao 1975, Green 1979, Stewart-Oaten and Bence 2001, Downes et al. 2002). The designs precisely address how the impact is assessed and proper statistical models have been developed to answer these specific questions (Downes et al. 2002). Using the improper statistical model, assumes a different design and question than may have been originally stated. Downes et al. (2002) suggest that it is incorrect to determine the proper statistical model for analysis after the data is collected. The experimental design is driven by the question and the statistical model is driven by the design. The statistical model requires sampling to occur in a certain fashion (e.g. random versus fixed assignments of treatments). The literature discussing these designs is confusing and often conflicting (e.g., Underwood 1994, Stewart-Oaten and Bence 2001).

The most common designs to evaluate the impacts of restoration actions is to apply a Before and After (BA) treatment comparison. In BA designs, samples are taken at various locations before and after a treatment. This occurs in the same reach or reaches impacted by restoration action, but in some situations are also measured in control areas, referred to as a before-after-treatment-control or BACI design. In most cases, the use of control(s) greatly increases the power of detecting impacts; however, poorly chosen controls sites can decrease the power of detecting an impact (Korman and Higgins 1997).

The most common statistical models used to assess the impact of a human action on an ecological process is the family of general linear models such as analysis of variance (ANOVA) models and time-series analyses. The ANOVA approaches are flexible, robust and powerful hypothesis testing procedures (Downes et al. 2002). Intervention analyses (IA) are another family of models that have been widely used to assess environmental impacts (Stewart-Oaten and Murdoch 1986, Carpenter et al. 1989). These models are based on timeseries analyses to estimate environmental impacts (Box and Tiao 1975). Intervention models use a covariates to filter out natural variability rather than control sites.

#### *Alternative Experimental Approaches*

A design that was first proposed by Walters et al. (1988) and referred to as a “staircase” design has been recommended as an alternative to standard BACI designs (Loughin 2006, Loughin et al. 2007). A staircase design involves a modification to the typical BACI design whereby treatments are staggered in time within the treatment area (i.e., temporal contrast). Instead of a single treatment being initiated and compared to a control through time, the treatments are staggered so that treatment replicates are established in different time periods (Loughin 2006). There are several advantages to using a staircase design. First, the staggering of the treatments over time allows for the distinction between the random effects of year and year x treatment interactions. This prevents random initial environmental condition (e.g. drought or high water year) from having an overriding effect on the ability of the experiment to detect true treatment effects. Loughin et al. (2007) demonstrated that standard long-term experiments “fail to model both random environmental effects and their interactions with the treatments” which can lead to misleading results. Second, by staggering treatments within the treatment area,

treatment sections can be used as controls until they are treated, guarding against loss of other control areas. Third, it is uncertain to the degree restoration may impact downstream reaches. A comparison of multiple reaches within a single watershed may be more powerful because of a greater number of replicates and the ability to accurately describe a reach versus a watershed or subbasin; however, these sites may not be independent from each other. The independence of control sites will depend on how far fish move within and between streams, and on the degree to which physical impacts from treated reaches propagate into the surrounding reaches. Finally, implementing the full suite of treatments over an extended period can be a benefit logistically and economically because large areas do not have to be treated all within one year.

Another alternative design is a nested hierarchical approach. Underwood (1994) suggests a nested hierarchical approach when the scale of impact is unclear (i.e., does restoration at the site level influence habitat or fish populations at the reach or stream scale). The hierarchical design provide insight into the scale at which future restoration actions should be monitored and can better identify and describe the casual mechanisms of fish responses restoration which often require multi-scale data.

### *Properties of Powerful and Robust Experimental Design*

ISEMPs review of experimental designs has identified a suit of experimental design properties that may increase the likelihood of ecosystem (watershed) experiments tasked with determining the effectiveness of restoration at increasing salmon and steelhead production and understanding the casual mechanisms. These properties can be grouped into four categories: contrasts, treatment size, treatment and control properties, and logistics.

In order to detect a signal due to a restoration action, distinct contrasts in either time or space must be created that can be distinguished from background natural variability (i.e., noise). Both biological and physical processes are highly heterogeneous throughout stream systems such as between valley, geomorphic reaches or channel units. Biological and physical processes also exhibit a wide temporal variability such as within and between days, seasons, and years. This noise can make detection of a signal (i.e., response to restoration) very difficult unless the effect is extremely large. Thus, the larger treatment effects are, the more likely noise can be separated from the true treatment effect. Another approach is to replicate treatments either across space to cover the heterogeneous environment, or place treatments in very homogeneous sections. The same approach could be used to distinguish the effects of time from treatment. However, replication across time and space is difficult with a large scale experiment. Therefore, an ideal experiment for testing stream restoration would incorporate both time and space contrasts with a large treatment effect. This requires an understanding of the current and historical conditions and a proper identification of the limiting factors within the study watershed (Roni et al. 2010b).

Ideally treatment and control sites should be similar to each other prior to restoration. The absolute difference in a variable (e.g., fish density) over time can be large as long as the fluctuations over time are consistent (i.e., synchronous; Downes et al. 2002). Control and treatment sites should also be independent so for example fish movement between sites should be minimal. A balance between independence and similarity between treatment and controls because as sites are located further apart they are more likely to be less similar in terms of biological and physical characteristics (Downes et al. 2002).

Watershed experiments by their very nature are expensive. In order to implement large scale restoration it may not be feasible with current funding levels for restoration. This may necessitate multiple treatments over several years.

### *Hybrid Hierarchical-Staircase Experimental Design*

An experimental design that has the properties listed above can be achieved by an hybrid design that combines temporal contract of the staircase design and the spatial contrast of the hierarchical design. ISEMP has been working with a statistician to assess the power of the hierarchical-staircase design compared to more traditional approaches to detect fish responses to restoration. ISMEP has also assisted a group of Intensively Monitored Watershed projects to implement this hybrid design because of all the apparent advantages. Below is a brief description of these IMWs and the statistical analyses performed.

### *ISEMP Intensively Monitored Watershed Projects: Experimental Designs*

ISEMP is currently assisting in the coordination and implementation of IMWs in Asotin Creek, Bridge Creek, Entiat River, Lemhi River and the Middle Fork John Day watersheds. The following sections describe the first three of these IMWs and the types of experimental designs beginning to be implemented and the rationale for the design selections.

### *Intensively Monitored Watershed descriptions*

#### *Asotin Creek*

Asotin Creek and its tributaries were selected as an IMW location in southeast Washington because of the wild population of steelhead present, strong agency and land owner support, the extensive amount of historical data, and the development of a model watershed plan. A limiting factors analysis indicated that riparian function was the most significant limiting factor in Asotin Creek. The limiting factors analysis also indicated that there are less large woody debris (LWD) and pools. The proposed restoration treatment is riparian fencing to exclude cattle, riparian planting to reestablish riparian vegetation, and addition of LWD to increase pool habitat. Riparian fencing and planting are expected to take a decade or more to have a significant effect; therefore, the addition of LWD is the main treatment that will be assessed in the Asotin IMW.

A hybrid hierarchical-staircase design is being implemented to compare treated and control sections within and between sub-watersheds in Asotin Creek. Treatments will be implemented in a staircase design after 4 years of pre-treatment monitoring (Figure 8). Three tributaries to Asotin Creek will be used as the treatment and control streams: Charley Creek, North Fork Creek, and South Fork Creek. Three treatment sections, each approximately 4 km in stream length, will be restored, one section in each creek (Figure 8). The expected results of the IMW are the restoration of 12 km of riparian habitat and ecological function in Asotin, an increase in LWD and pool habitat and pool and cover quality, an increase in overall residual pool depths, and an increase in average juvenile steelhead abundance and production. Other benefits of the IMW will include a greater understanding of the effects of LWD treatments on growth and survival of juvenile steelhead, the specific mechanisms for how LWD treatments influence geomorphic processes and fish habitat (which in-turn impacts fish population performance), and the movement of juvenile steelhead within and between subwatersheds.

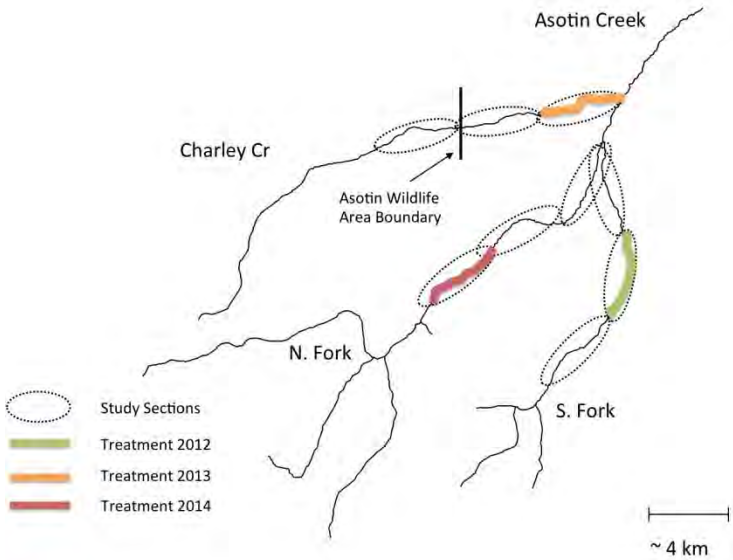



Figure 1. Experimental design and restoration schedule for the Asotin Creek IMW.

### *Bridge Creek*

The Bridge Creek IMW is described the next section of this Chapter. ISEMP has developed a hierarchical-staircase experimental design for the implementation of the proposed restoration action (Figure 2). Comparisons will be made pre- and post-treatment between restored treatment and non-restored control areas at the site, sub-watershed, and watershed scales (Figure 2). At the largest scale, the restored Bridge Creek watershed will be compared to a similar nearby watershed, Murderers Creek, a tributary to the South Fork John Day River, where ongoing monitoring of steelhead populations and physical habitat conditions is occurring. The Bridge Creek and Murderers Creek basins have similar climatic conditions and historic, land use (ranching), and downstream Columbia River, estuary and ocean conditions. Within the Bridge Creek watershed, changes in the mainstem will be compared to two unmanipulated tributaries, Bear Creek and Gable Creek. At the highest level of resolution, comparisons will be made between control and treatment sites of the mainstem of Bridge Creek.

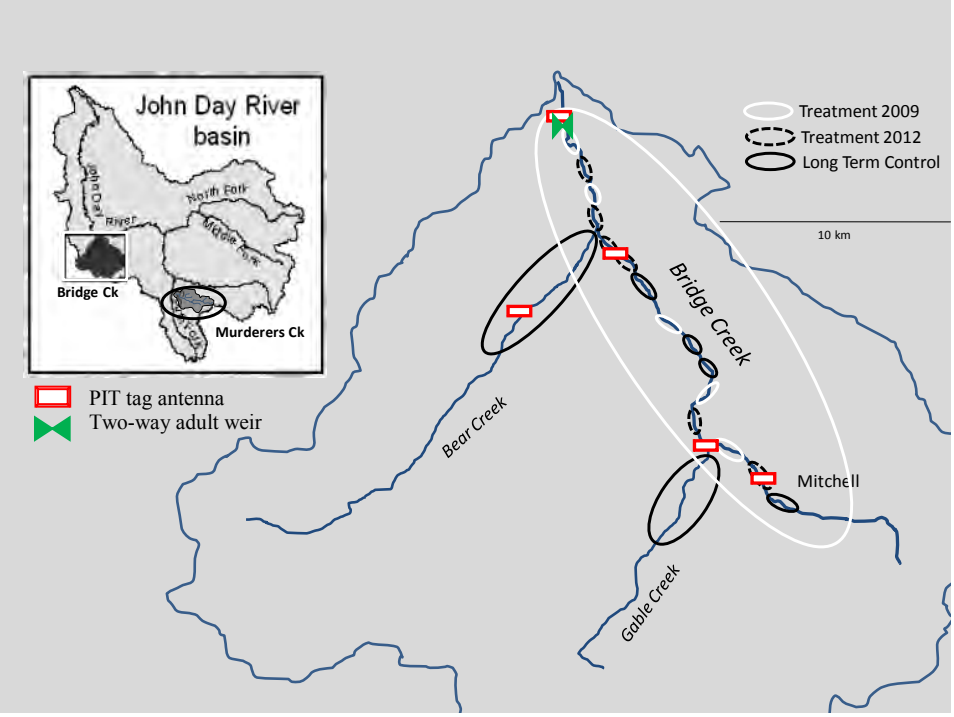



Figure 2. The Bridge Creek IMW experimental design. White, black-dashed, black-solid oval represent restoration units, subwatersheds, watersheds that will be treated in 2009, 2013, or act as long term controls, respectively. Gable Creek and Bear Creek will act as long term sub-watershed controls and Murderers Creek (inset map), will act as the watershed level control. Monitoring occurred before any beaver dam support structures (BDSS) were installed in 2009.

### *Entiat River*

The Entiat River subbasin represents an area of significant concern for the Upper Columbia region and BPA has identified the Entiat River subbasin as a priority for implementation funding. The focus of the Entiat IMW is on detecting changes to Chinook and steelhead freshwater productivity, growth and survival resulting from the implementation of restoration actions within the Entiat River subbasin. Within the Entiat River subbasin, the lack of mainstem habitat diversity was identified as the most important factor underlying depressed production. The second most important factor is a lack of food, especially in the upper middle Entiat, with channel stability, key habitat and flow playing distinctly lesser roles. In 13 of the 16 mainstem Entiat reaches supporting spring Chinook salmon, the most severely impacted life stage was either fry or parr.

ISEMP proposed that a hybrid hierarchical-staircase statistical design be implemented to compare treatment and control sections within the Entiat River subbasin. The hybrid hierarchical-stairstep experimental design uses the USBR's 2008 tributary assessment to divide the lower 26 miles of the Entiat mainstem into geomorphic reaches that can be treated in a spatially and temporally driven manner. The tributary assessment identifies three valley segments and 17 geomorphic reaches identified in the mainstem (Figure 3) that distinguish sections of river with unique physical characteristics and provides a context for customizing river restoration strategies based on specific characteristics of each reach (USBR 2009). Valley

segments were defined based on changes in the channel gradient and geologic features that control channel morphology.

These valley segments act as natural breaks that restoration actions may be evaluated within and perhaps provide information on the interaction of valley types and the ability of instream structures to provide benefits (i.e., slow meandering sections may respond differently than more confined higher gradient reaches). The EWPU determined that the primary means to address limiting factors in VS1 is implementation of active instream restoration actions that restore habitat complexity and diversity such as large pools and off-channel areas (CCCD 2006). Both rock and wood instream structures were considered appropriate within this area of the subbasin, although concerns about the stability of wood structures in the lower Entiat has limited their use in the past.

### *Developing Hierarchical-Staircase Statistical Models*

Statistical models have been developed for the Asotin and Entiat IMWs. Individual factors were identified for each experiment: Creek, Section within Creek (written "Section(Creek)"), Reach within Section of Creek (written "Reach(Section*Creek)"), Year, Season, and Years After Restoration (YAR). Among these, the factors Creek, Season, and YAR are considered as *fixed effects* factors. The factors Year, Section(Creek) and Reach(Section*Creek) are considered as *random effects* factors. This means that we consider these years to be representative of years to which the results may apply; they are a sample of possible years in which the experiment could have been run and are used to quantify the variability that might be seen across years in the future. Similarly, the sections and reaches actually used in the study are used to represent the entire creeks and to allow us to quantify the variability that is seen spatially within a stream.

The models that have been developed for all of the response measurements are based on experimental design principles. Experimental units were identified to which each factor or interaction of factors was assigned or observed. These are determined from the rectangles of various sizes and shapes that represent each factor in Figure 4 a and b. Then a model was derived containing terms corresponding to each different size of experimental unit. Fixed and random effects were identified as above, and also using the convention that interactions involving random effects are also random.

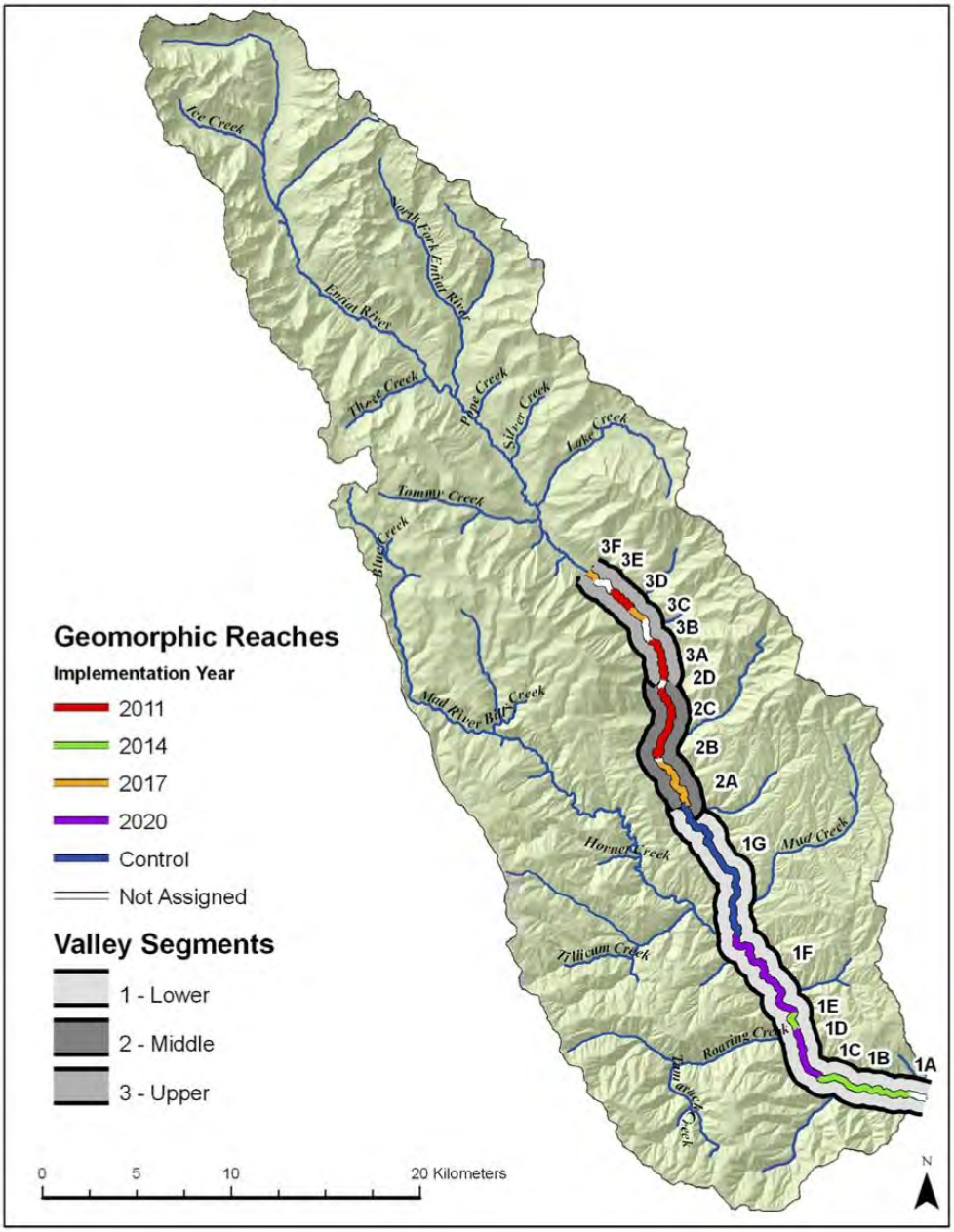
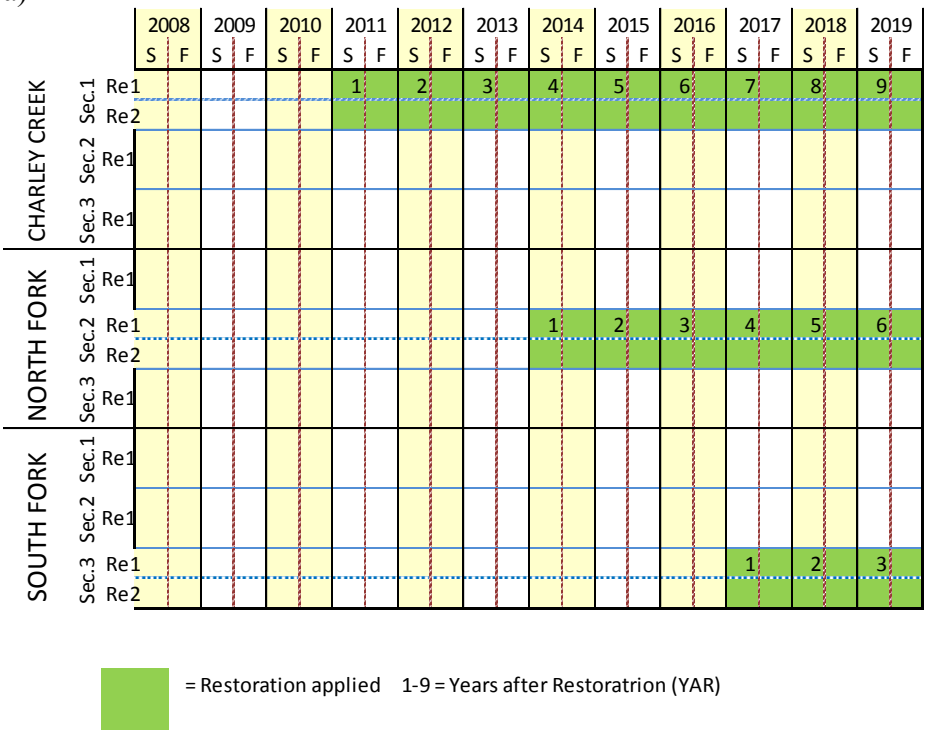



Figure 3. The Entiat River IMW experimental design. Treatments are stratified by valley types. Numbered letters represent reaches. Red reaches will be treated in 2011, green reaches in 2014, orange reach in 2017, and purple reach in 2020. The Mad River (large tributary coming in at the upstream section of 1F), will act as sub-watershed control and will not be treated.

a)




b)

| River | RAU | Site | 09 | 10 | 11 | 12 | 13 | 14 | 15 | 16 | 17 | 18 | 19 | 20 | 21 | 22 | 23 | 24 | 25 | 26 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| ENTIAT | 1B/1C | | 0 | 0 | 0 | 0 | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 |
| | 1D | | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 |
| | 1E | | 0 | 0 | 0 | 0 | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 |
| | 1F | | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 |
| | 1G | | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | 2A | | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 |
| | 2C | | 0 | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 | 15 | 16 |
| | 3A | | 0 | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 | 15 | 16 |
| | 3C | | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 |
| | 3D | | 0 | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 | 15 | 16 |
| | 3F | | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 |
| MAD | M1 | | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | M2 | | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| | M3 | | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

Figure 4. Schematic of the treatment and control reaches and location and timing of treatments used to develop statistical models for a) the Asotin Creek IMW and b) the Entiat River IMW.

Not all combinations of YAR and Creek are present in the experiment; indeed, these two factors are highly unbalanced. Main effects of YAR and Creek can therefore not be estimated separately without interference from each other's effects. Instead, the effects of YAR must be estimated using contrasts within the context of the YAR*Creek interaction. This puts restrictions

on the terms that can be included in the model. The resulting models can be expressed as an ANOVA table or as an equation (Table 1 a and b).

Table 1. Statistical models and factors for the a) Asotin Creek IMW and b) the Entiat River IMW.

a) Asotin Creek IMW

| Source | DF | Fixed or Random | Symbol | Subscript |
|---|---|---|---|---|
| Year | 11 | Random | q | H |
| Creek | 2 | Fixed | β | I |
| Year*Creek | 22 | Random | (qβ) | Hi |
| YAR*Creek | 9 | Fixed | (βτ) | Ij |
| Section(Creek) | 6 | Random | s | Ik |
| Year*YAR*Section(Creek) | 57 | Random | (qτs) | Hijk |
| Reach(Section*Creek) | 3 | Random | r | Ikl |
| Residual Error | 33 | Random | e | Hijkl |
| TOTAL | 143 | | | |

Model and definitions of effects are:

$y_{hijkl} = q_h+\beta_i+(q\beta)_{hi}+(\beta\tau)_{ij}+s_{ik}+(q\tau s)_{hijk}+r_{ikl}+e_{hijkl}$

$y_{hijkl}$ = Response in year h, creek i YAR j section k reach l

b) Entiat River IMW

| Source | DF | Fixed or Random | Symbol | Subscript |
|---|---|---|---|---|
| Year | 17 | Random | q | I |
| Stream | 1 | Fixed | γ | H |
| Year*Stream | 17 | Random | (q γ) | Hi |
| RAU(Stream) | 12 | Fixed | b | Hj |
| YAT*Stream | 16 | Fixed | (γ τ) | Hk |
| Year*RAU*YAT(Stream) | 188 | Random | (qbτ) | Hijk |
| Site(RAU*Stream) | 70 | Random | s | Hjl |
| Residual Error | 1190 | Random | e | Hijklm |

Model is:

$Y_{ijklm}= q_i+\beta_j+(\beta\tau)_{jk} +(q\beta\tau)_{ijk}+s_{jl}+ei_{ijklm}$

### *Testing the Power of the Hierarchical-Staircase Design*

Once statistical models were developed we used the Asotin Creek IMW to test the hybrid design using extensive computer simulations. We developed a computer model for watershed based on the Asotin Creek watershed to describe the spatial and temporal layout of the study. There were three streams, which were treated as independent of one another (responses on one stream were not affected by responses on another). Within each stream there were 3 sections; within each section there were 2 fish sites ("f-sites"). Within each fish site there were three habitat sites ("h-sites"). Thus there were a total of 3x3x2x3=54 locations within the watershed at which measurements could be taken. This spatial structure is observed for 12 years, so that a total of 648 potential observations could be created.

Because the variance components were estimated with (sometimes substantial) uncertainty, three different variability scenarios were considered. The first was used the estimated variance components, which represents our "best guess" as to the actual variability present. The second used variance components set to the lower limits of their respective confidence intervals, representing a "best case" for variability. The third used variance components set to the upper limits of their respective confidence intervals, representing a "worst case" for variability.

We compared the power of the tradition BACI design (referred to as "1-Site") to three different experimental designs were compared for assigning restoration treatments to units in the study. The first was the original design for the Asotin Creek IMW (referred to as the "planned"). In this design the three sections of one stream, Charley, are to be restored one-at-a-time in three-year intervals. The second design is the alternative design that is depicted in Figure 8. In this design, the staggering of treatment applications in three-year intervals continues, but the sections treated at different times are in different streams. This is called the "alt" design. The third design is the simple design that might be used by many researchers. It consists of a single treated section, restored at the midpoint of the experiment (i.e. after 6 years). Without loss of generality, the middle section of Charley was used as the treated section. This is called the "1-time" design.

Once the model was defined, pseudo-watersheds were generated by simulating pseudo-random data to represent the potential measurement at each of 54 h-sites across 12 years. Random effects were generated independently according to their respective variance components and stream means were added in. One pseudo-watershed consisted of 648 potential measurements. For each of the 12 combinations of response variable, variability scenario, and autocorrelation, 1000 pseudo-watersheds were simulated. This number allows Type I error rates of analyses conducted at the 5% level to be estimated to less than ±1.5% error with 95% certainty. Power estimates similarly can be estimated to at worst ±3.2% error with 95% certainty.

Here we provide the results of simulations using juvenile steelhead abundance measurements which are collected at the f-sites. We simulated the following sampling plans to investigate what relative differences in power could be obtained by different levels of sampling intensity.

For abundance, 5 different sampling plans on f-sites were considered:

a) "1-per-stream", in which one f-site is chosen at random from the middle section of each stream and measured in each year. This represents the barest minimum measurement that could take place in a BACI-type study, and is used only with the 1-time experimental design.
b) "1-per-section", in which one f-site is randomly chosen from each section of each stream and measured in each year. This represents a minimum sampling plan design in which all three designs can be run and compared.
c) "Planned", which consists of the same measurements as in 1-per-section, plus a second f-site in each section in Charley, the treated stream.

d) "Alternative", which follows the same spirit Planned, but matches the extra measured f-site with the treated sections from the Alt design.

e) "Full", in which 2 f-sites are measured in each section (twice as much measurement as in 1-per-section, 50% more than Planned).

Under the best case for variability, all designs and sampling plans have 100% detection of the 25% increase. Even under the estimated variability, all designs and sampling plans have at least 95% power to detect the treatment effect except the BACI combination, 1-per-stream sampling with a 1-time design, which has just over 70% power (Figures 5). Once measurements are made in each section, confidence interval lengths do not change much with additional subsampling within the sections. The alt design has the shortest intervals, while the 1-time design has the longest.

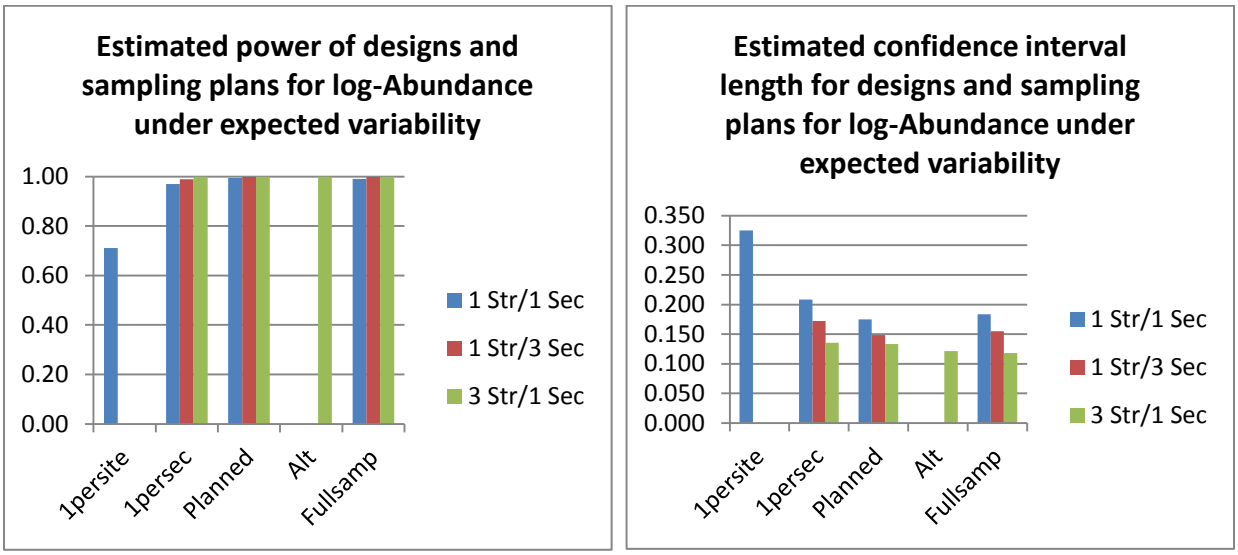



Figure 5. The estimated power of designs and sampling plans (left panel) and their associated estimated confidence intervals (right panel) for detecting a 25% change in juvenile steelhead abundance using Asotin Creek IMW historic data and ***best case*** estimates of variance.

Under the worst-case variability, greater differences among the methods begin to emerge (Figure 6). The 1-time and current designs have very similar powers and lengths regardless of the subsampling intensity. However, the alt design distinguishes itself in terms of both power and length of confidence interval. Powers range between 60-70%, compared to 25-35% for the other designs. Confidence interval lengths are roughly 2/3 those of the other designs.

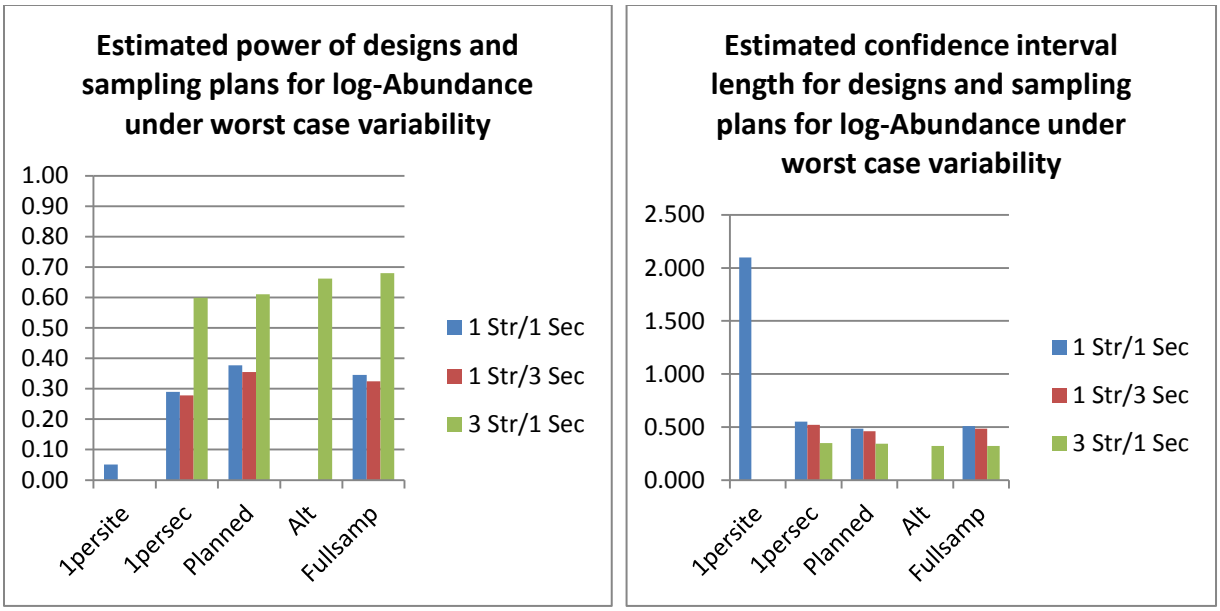



Figure 6. The estimated power of designs and sampling plans (left panel) and their associated estimated confidence intervals (right panel) for detecting a 25% change in juvenile steelhead abundance using Asotin Creek IMW historic data and ***worst case*** estimates of variance.

These results must be viewed as somewhat speculative. The process of estimating all of the needed variance components for the watershed was not straightforward. The historical data were sparse, particularly at the subsampling levels (F-site and H-site). Data sets tended to have either spatial or temporal components to them, so combined spatio-temporal random effects (year*unit interactions) may not be well estimated at all. Despite the use of upper endpoints of confidence intervals in formulating a "worst case", some variance components are guesses and therefore may be subject to far greater variability than presumed. Hence, caution should be applied in interpreting the values of the powers.

However, the principles that drive the *comparisons* among powers do not depend on the actual values of the variance components, but rather on their relative sizes. Fundamentally, treatments are applied to sections and subsequently measured in different years. The analyses include terms that account for any variability that occurs on a larger scale, and hence this variability does not affect the designs or sampling plans' relative powers of confidence interval lengths. Similarly, subsampling of F-sites and H-sites is predictably less effective than measuring more sections. The difference between the 1-per-stream and 1-per-section sampling plans for the 1-time design was very large. The difference between taking 1 measurement per site and full subsampling was not generally very large. However, this is largely due to the relative sizes of the variance components for the subsampling effects. It is conceivable that lower-level variability is much greater than assumed, in which case subsampling becomes an effective and relatively inexpensive means of improving precision. The only way to know this is to collect data on a finer scale than what is presently available.

Additional runs were performed under planned sampling, varying the ultimate treatment effect from a 5% increase to a 40% increase. This was intended to allow more detailed comparison of the current and alternative designs, specifically addressing the concern that a

multiple treatments applied in different sections of the same stream may synergize to generate a larger treatment effect in each treated section than would be observed by treating only one section of a stream. By looking at the power curves for the two designs, we can see how much synergy would need to take place in order to make the planned design favored over the alternative (i.e., treating one stream versus treating all streams; Figure 7).

Under estimated variability the curves are separated only for detecting changes of 20% or less. If synergy of multiple treatments in one stream accounts for the horizontal difference between the curves, then the designs are equivalent. Here we see that the horizontal difference is never more than 5%, so that the synergism does not have to be large for the planned design to have power that is favored over the alt design. However, the difference in confidence interval lengths is independent of the treatment effect, so the current design really needs to be *more* powerful than the alt design in order to make up for the fact that it produces longer, less useful intervals.

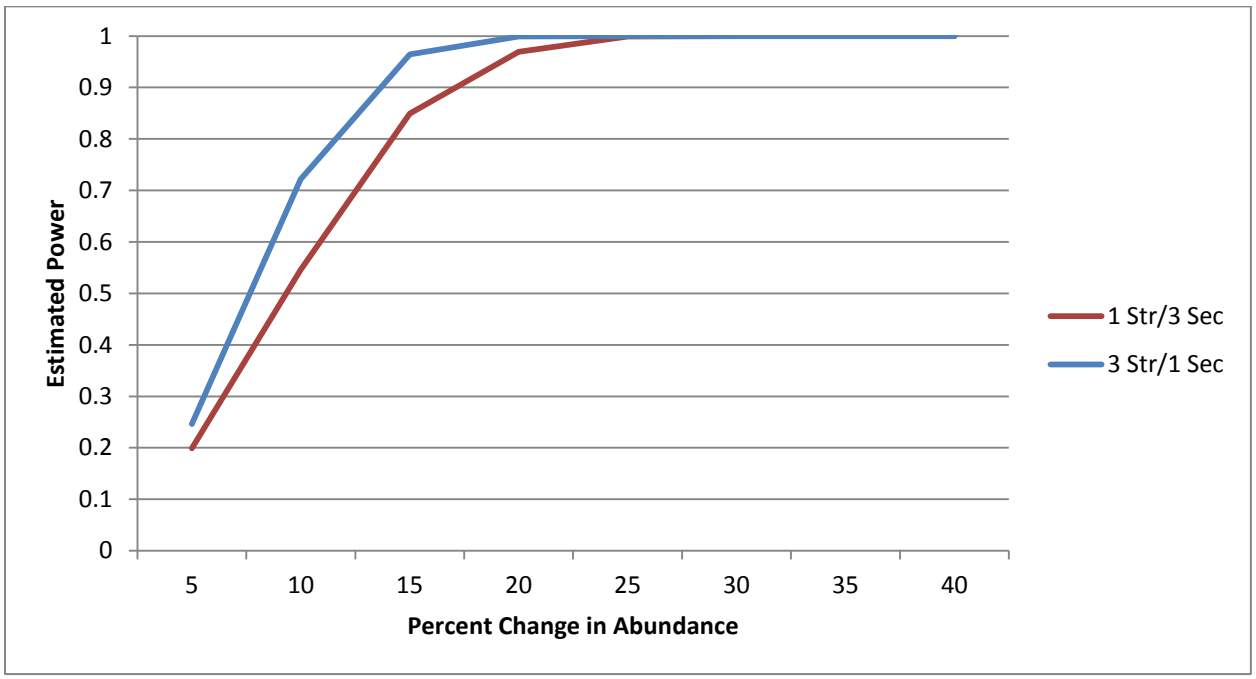



Figure 7. Power for varying alternatives for log-abundance under ***best case*** scenario variability.

In the worst case variability, the horizontal difference can be as much as 15% (Figure 8). There would need to be considerable synergy, making up a sizable portion of the total treatment effect, before the current design would be favored over the alt design.

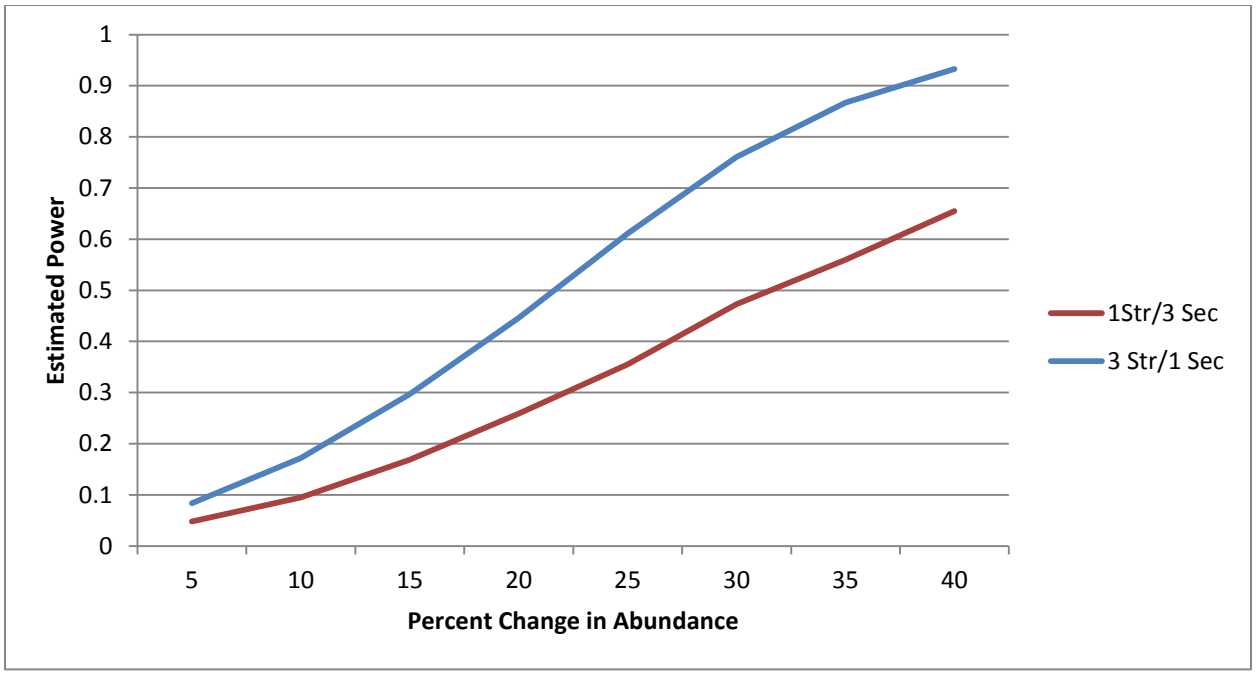



Figure 8. Power for varying alternatives for log-abundance under ***worst-case*** variability.

Whether such synergism exists is unknown. If it exists, its exact nature is also unknown. Four years of pre-treatment sampling in Asotin Creek suggest that juvenile movement between streams is rare which suggests improvements in habitat will have a very limited effect toward attracting fish from other streams. Therefore we can expect that there is little negative dependence from stream-to-stream.

Furthermore, pool creation is a local phenomenon. It is unlikely that a treatment applied to one section of a stream will produce pools several km away. Therefore, we can expect that there is no synergism in the pool effects, and comparative power curves like those above are unnecessary. If section-to-section wandering of fish is low, then we can expect that the treatment effects on abundance will be mostly independent from section to section, and any synergism is quite limited. We will be able to test this hypothesis more stringently when data arrive.

The relatively similar performance of the 1-time and planned designs for detecting differences in abundance is a bit disheartening, considering the extra effort that the planned design requires. The planned design suffers because comparison between treated and untreated sections cannot be made within the same stream. It is the variability of such sections that is the most important component of the error term for testing and forming confidence intervals for treatment effects. This increases the variability of treatment effects estimated later in the design, in particular those associated with times more than 6 years after treatment. So the advantages of multiple treated sections are diminished by the disadvantages of increased difficulty in separating treatment effects from inherent variability.

It should be noted that the alt design overcomes this issue by having treated sections spread among three streams, with untreated sections in the same streams. This creates a situation akin to blocking in that treatment comparisons against controls are made within stream rather than between streams, and therefore incur less variability in estimating effects. This explains the

improvement enjoyed by the alt design, both in terms of power and, crucially, confidence interval length. NOTE: this alternative design has now been adopted by the Asotin Creek IMW.

If it is believed that treatments applied to different sections of the same stream synergize to create a broader, more favorable environment for fish, then the application of multiple treatments to sections of the same stream has the potential to create an environment that is overwhelmingly favorable in a single stream. This can lead to a larger overall treatment effect which, if sufficiently larger, would be easier to detect than effects caused by other designs. If, on the other hand, the potential for synergy is viewed as minimal or nonexistent, then the clear favorite is the alt design, in which a section of each stream is eventually treated over time.

***Caveats***

- Watershed model is somewhat simple
- No Year*Treatment or 3-factor terms
- this makes BACI look better than it is!
- Some estimates based on different data collection methods
- Some variance components are total guesses
- Year interactions are critical!
- View values of power and CI width lightly
- Relative comparisons of designs likely fairly stable

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# CHAPTER 7: Bridge Creek Intensively Monitored Watershed Project

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### *Introduction*

As described in Chapter 3, Bridge Creek is an incised stream that we are trying to restore by encouraging beavers to build stable dams that will capture sediments, build up the stream bed, and reconnect the stream to the historic floodplain. The project takes a process-based restoration approach that involves installing a series of beaver dam support structures (BDSS) designed to mimic beaver dams and to assist beaver in the construction of stable dams that will create pool habitat for juvenile steelhead in the short term (Pollock 2009). A major goal is to have the beaver do the bulk of the restoration work while we facilitate the process of beaver colony establishment in a degraded stream system. As such, the project is not an "engineered" approach to stream restoration with a spatially fixed outcome. Providing some short term ( < 10 yr) assistance to set in motion natural processes by which the stream restores its natural dynamics is the expected outcome. In the long term, beaver dams will facilitate stream geomorphic changes that include sediment retention, stream bed aggradation, increased stream sinuosity, pool formation, increased stream length, reduced stream slope, reduced bed shear stress and a shift in the bed composition from cobble towards gravel (Pollock et al. 2007, Demmer and Beschta 2008). In both the short and long term, the beaver dams will raise water tables in the alluvial aquifer and thus help to greatly expand the amount of riparian forest and reduce stream temperatures (Lowry 1993, Pollock et al. 2007, Pollock et al. 2011). Previous work has shown this type of restoration approach to be successful in the John Day and elsewhere (reviewed in Pollock et al. 2003).

As described in the Experimental Designs for IMW (Chapter 6; Figure 2), because Bridge Creek is an IMW, emphasis is placed on detecting the benefits of stream restoration on fish habitat and fish performance. This requires an experimental design that creates contrast in space and time allowing separation of treatment signals from environmental variation. This also requires detailed monitoring to ensure we capture these signals. We are conducting monitoring to capture habitat, geomorphic, and fish responses to these treatments within the experimental design.

### *Methods*

Because the Bridge Creek IMW restoration efforts have not had enough time to influence fish responses, we focus the discussion here on habitat and geomorphic monitoring approach and preliminary responses. Habitat monitoring is being conducted at restored treatment and non-restored control areas at the site (Figure 1), sub-watershed, and watershed scales. We used an adaptation of protocols developed by the PACFISH/INFISH Biological Opinion (PIBO) Effectiveness Monitoring Program (Heitke et al. 2007) as well as the Oregon Department of Fish and Wildlife Aquatic Inventories protocol to describe a number of physical stream channel attributes**Error! Reference source not found.**. These survey methods have been implemented hroughout Bridge IMW study sites 3 years prior to restoration and one year after restoration. We

also implemented a draft protocol developed by ISEMP, CHaMP, to test its feasibility and refine the particular protocol elements.

Sampling of stream channel attributes using PIBO based surveys began in 2007 and are conducted annually throughout 20 study assessment units, for a total of 40 sample sites within the Bridge Creek IMW study watersheds. Each assessment unit has been broken into four habitat survey sites that are roughly 160 m in length. One of these sites is sampled annually, and sampling effort is distributed among sites according to a rotating panel design**Error! Reference ource not found.**. During the first year of sampling one site was randomly selected in each site as an annual site each to be sampled each year. An additional random site is also selected without replacement and sampled each year.

We have also been collecting high-resolution spatial documentation (via topographic data) of treatment and control conditions in Bridge Creek. Together, the aerial photographic and topographic data collected is intended to detect, monitor, and quantify geomorphic change within the ten monitoring reaches units along Bridge Creek (Figure 14).

Topographic surveys are used to acquire bathymetric data of the channel and topographic data of the riparian corridor and valley context. Bathymetry (topography beneath the water's surface) was collected using RTK GPS where possible, and an auto-tracking Total Station everywhere else. The bathymetric surveys were conducted to capture the major grade brakes and geomorphic units (e.g. pools, bars, etc.) within the channel. Additionally, the RTK GPS allowed for the creation of break lines between data points of linear features, such as water's edge, during data collection. Segregation of points and creation of break lines during data collection in the field greatly reduces the post processing time and errors in data interpretation during DEM development. Point spacing was semi-regular (1 point every 1-2 meters) feature-based morphologically stratified sampling scheme (Wheaton 2008). Point densities varied spatially with higher point densities (e.g. 2-3 points/m2) in topographically complex areas and lower point densities in topographically simple areas. This survey approach forms the basis upon which the CHaMP protocol was developed.

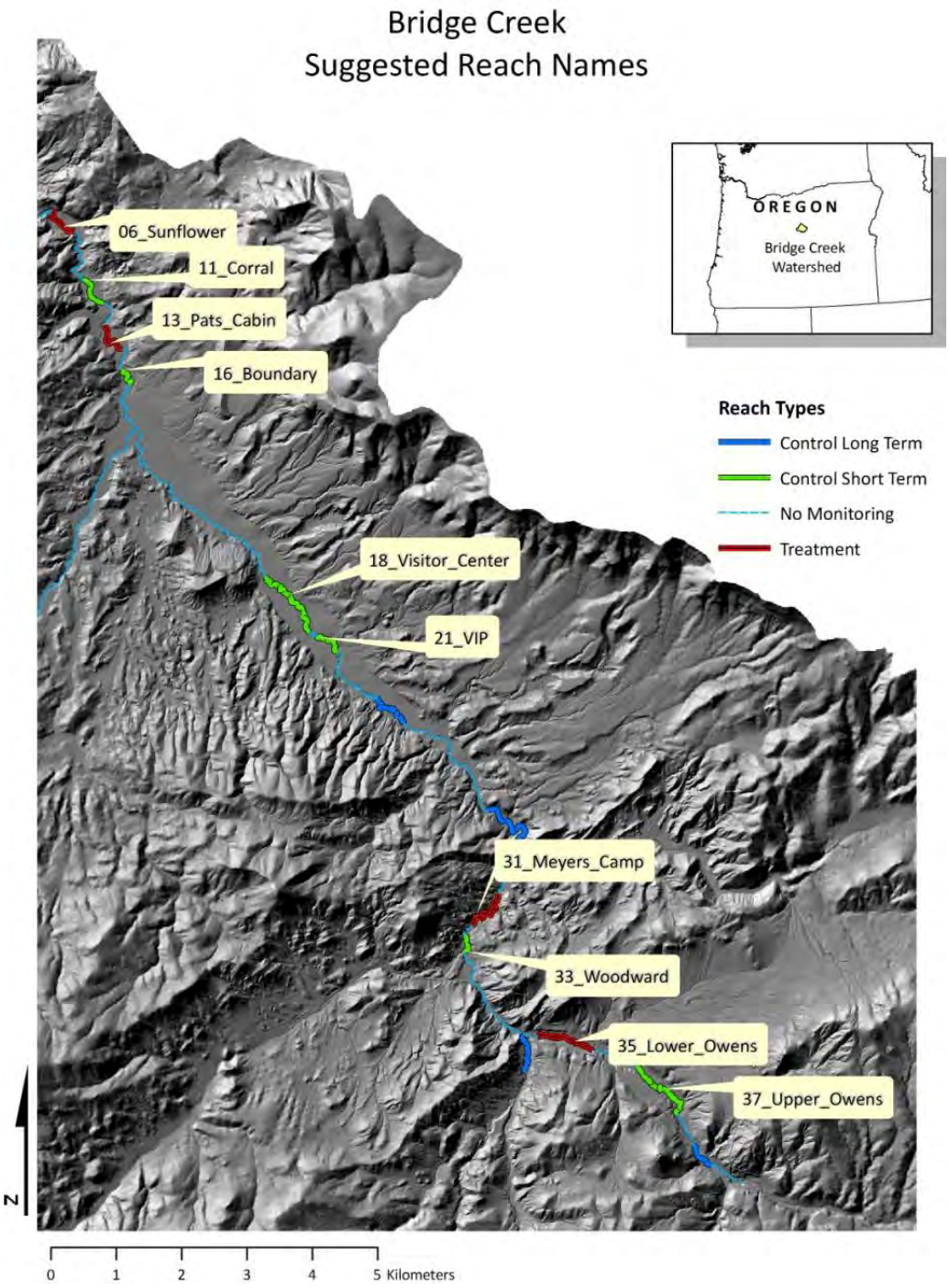



Figure 1. Location of restoration assessment units in Bridge Creek. Labeled are the treated (red) or soon to be treated units (green). Blue assessment units will act as long term controls.

### *Results*

One year after installation of the BDSS, 30% were colonized by beaver, beaver activity was present in all treatment reaches, and beaver had expanded into a treatment reach previously

unoccupied. In general, deposition occurred behind beaver dams and BDSSs, with scour pools forming downstream.

To make simple comparisons between treatment and control reaches, we used an intervention analysis to evaluate changes in stream channel metrics taken from our PIBO based surveys (Stewart-Oaten, 2001). This approach uses a t-test to test for significant difference between the average difference between treatment and control assessments units before (2007-2009) and after (2010) restoration implementation. For this analysis, we focused on a handful of metrics that describe channel morphology (average bankfull width), the quantity and characteristics of pool habitat (avg. residual pool depth, pool frequency, percent pool), and substrate composition (particle D50 and percent fine sediment) that were expected to change following restoration activities.

The intervention analysis suggests that a number of channel attributes are responding to the restoration activities on Bridge Creek, and that these changes are measurable using PIBO based sampling approaches (Table 1, Figure 2). All metrics describing the quantity and characteristics of pool habitat were found to have significantly increased at treatment sites following restoration implementation ($\alpha = 0.1$). There is also evidence that the bankfull width is increasing, and that the composition of channel substrates has decreased following restoration.

Table 2. Average difference (SE), and significance (p-value) between channel attribute metrics for treatment and control assessment units on Bridge Creek both pre and post restoration.

| | Treatment control difference | | |
|---|---|---|---|
| **Channel metric** | **Pre** | **Post** | **p** |
| Bankfull width | -0.48 (0.08) | 0.57 | 0.90 |
| Percent pool habitat | 1.2 (1.9) | 9.80 | 0.05 |
| Pool frequency | 0.76 (1.6) | 7.05 | 0.06 |
| Residual pool depth | -10.2 (2.1) | 10.67 | 0.01 |
| % particles < 6mm | -2.5 (4.1) | 6.38 | 0.58 |
| Particle D50 | 3.2 (4.7) | 4.96 | 0.75 |

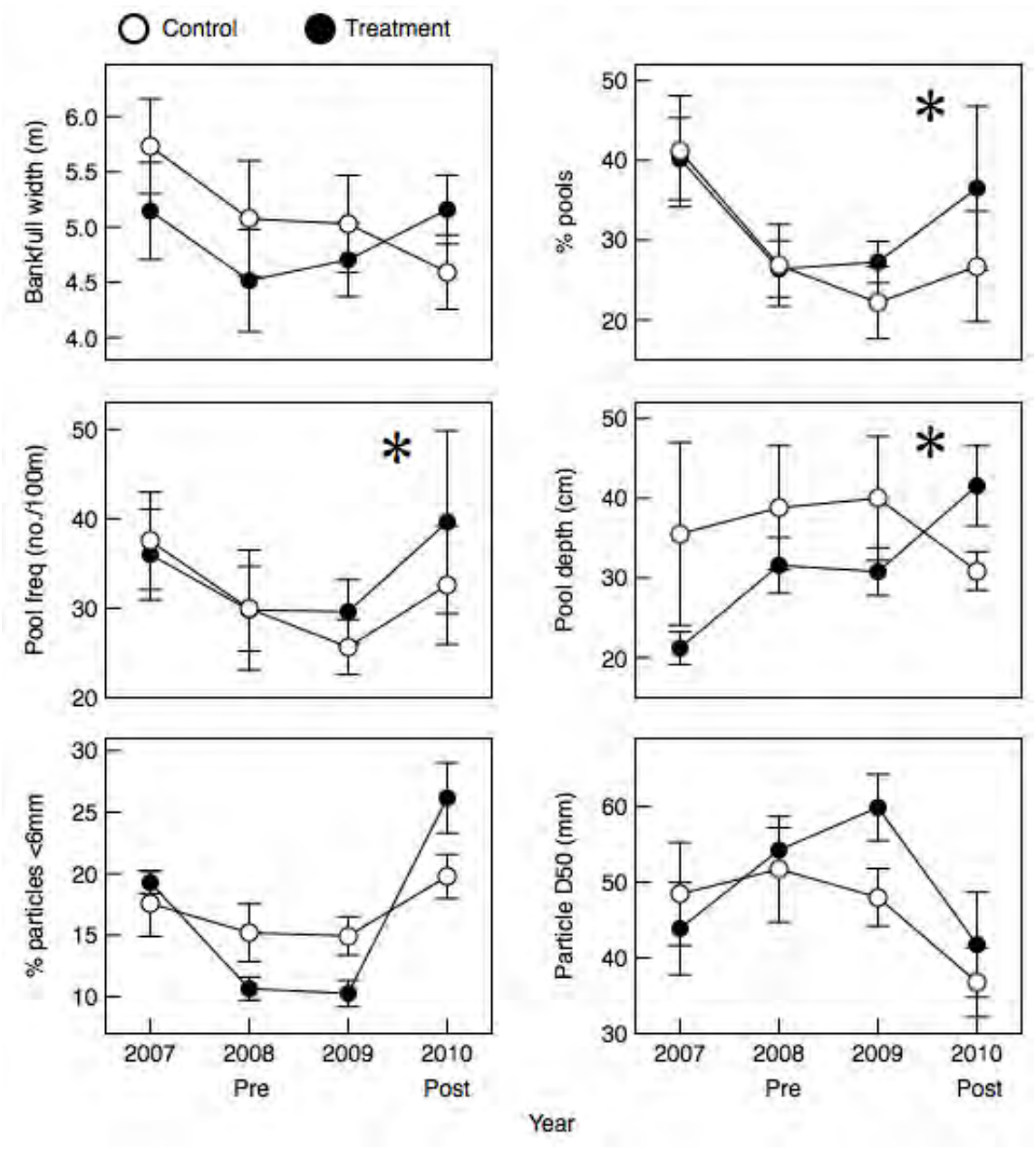



Figure 2.Average of channel attributes for treatment and control reaches across years, * indicates significant differences for pre and post-restoration.

Aerial photography was provided by AggieAir Flying Circus of the Utah Water Research Laboratory using unmanned aerial vehicles (UAV; Figure 3). Their surveys covered approximately a 25 km corridor with 300 m or greater width. Images were collected in April of 2010 and October of 2010. The mosaic image resolution is 0.10 meters provides significant detail (Figure 4).

A variety of geospatial outputs are created from the raw topographic data points acquired via RTK GPS, TS, and TLS (Figure 5). For example, DEMs can be used to show in-channel as well as floodplain topography; water depth maps overlaid on this can highlight the presence of

pools and bars in the channel as well pick up the beaver dam support structures. The UAV imagery can clearly show the BDSS structures and riparian vegetation responses. In Figures 6, we display products from the one of treatment reaches (these exist for all 10 sites labeled in Figure 1).

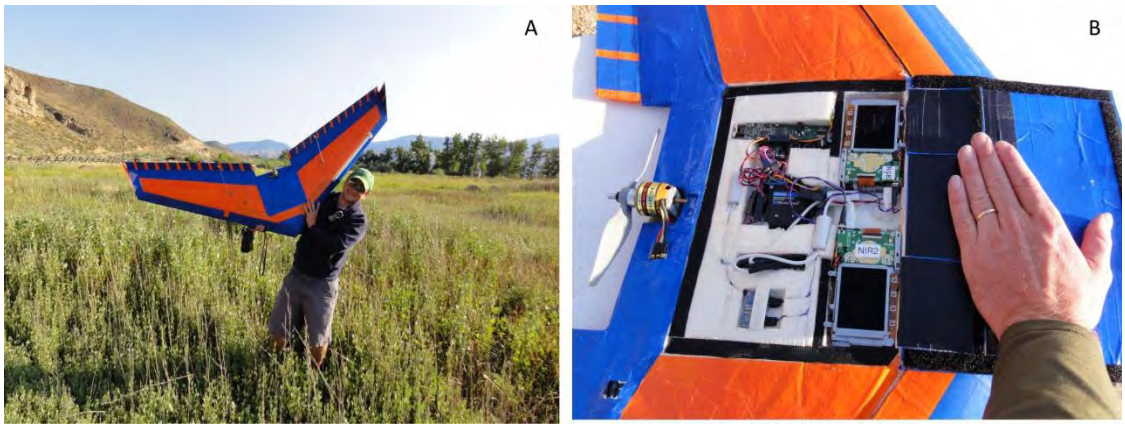



Figure 3. UAV Drone (A) equipped with RGB and NIR digital cameras (B).

Figure 432. Example of high resolution of UAV imagery at the Painted Hills National Monument visitor center.

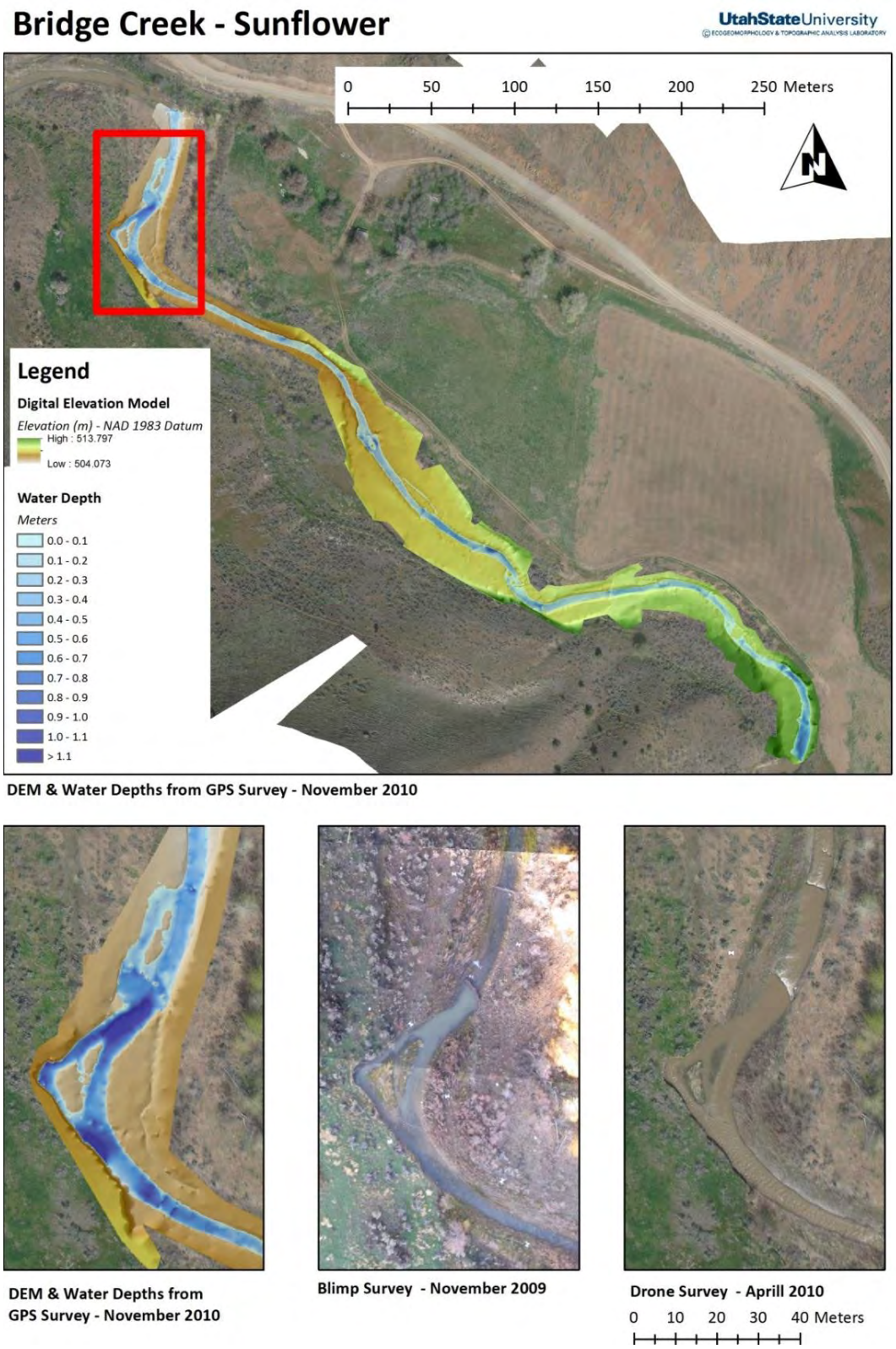



Figure 5. Examples of derived products from topographic and aerial surveys.

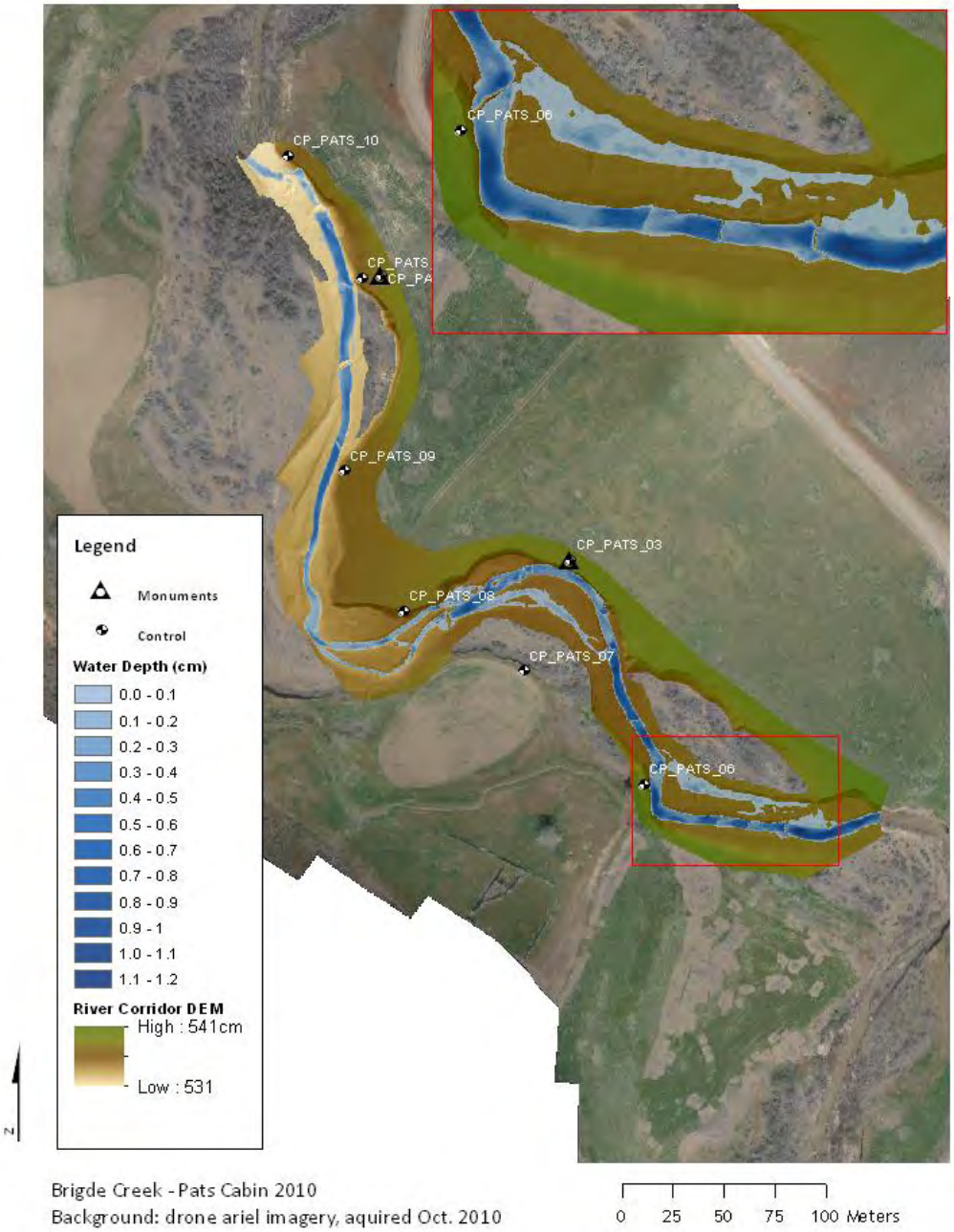



Figure 6. Pats Cabin treatment reach showing the digital elevation model, water depth maps, derived from topographic data. Also shown is the location of BDSS.

Digital elevation models derived from each survey are differenced to produce DEMs of difference (DoD). DoDs are used to estimate the net volumetric change in a reach through time (Figure 7**Error! Reference source not found.**). From a geomorphic perspective, these represent he change in storage terms (due to erosion and deposition) of a sediment budget. In Wheaton *et al.* (2010) methods are described for accounting for uncertainties in the individual DEMs, such that confidence can be developed in distinguishing changes due to geomorphic processes from changes due to noise. A fuzzy inference system was used to estimate the errors in each of the twenty DEMs between 2009 and 2010 at the ten sites on a cell-by-cell basis. Once those errors were established for each DEM, they were propagated through on a cell by cell basis using standard error propagation, to establish minimum levels of detection for meaningful change as calculated by the DoD. We used the Geomorphic Change Detection Software version 5 to do these analyses (http://gcd.joewheaton.org).

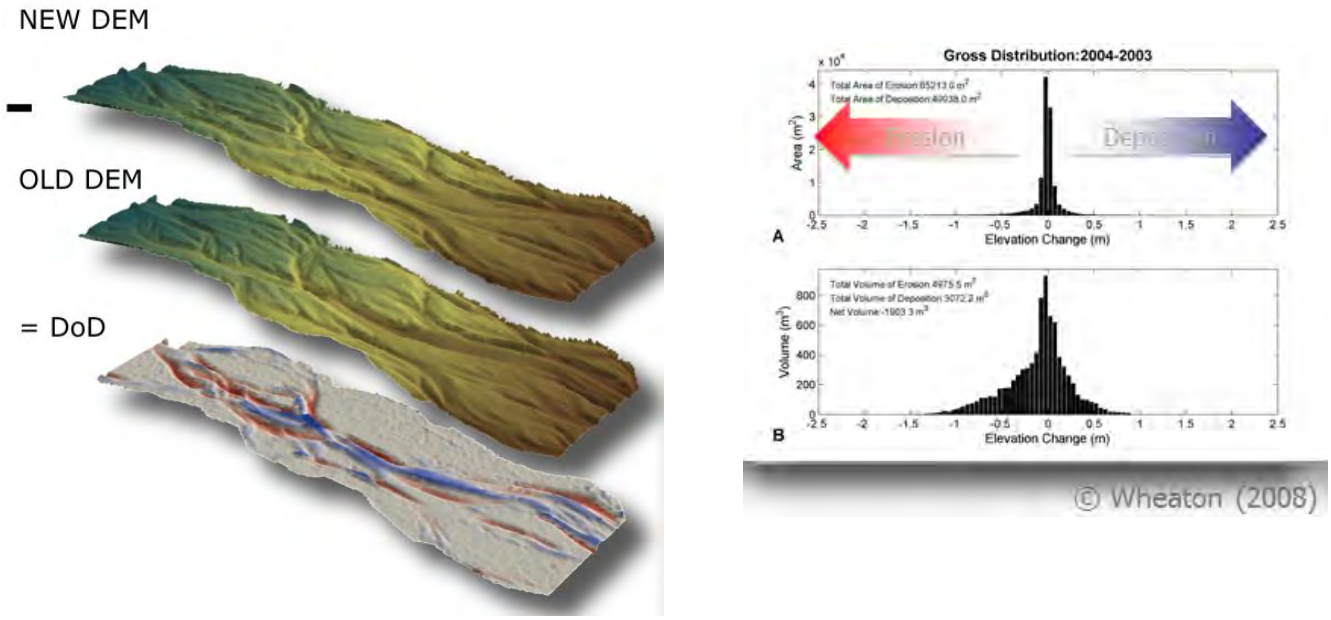



Figure 7. Concept of DEM differencing. For an X, Y pixel the old elevation (Z) is subtracted from the new elevation. A negative value (represented in red) indicates erosion, where a positive value (new elevation is higher than old; represented in blue) indicates deposition, and neutral change (represented as white). This is done for every X,Y pixel to create a surface (DoD), and a distribution of the actual elevational changes can be summed to create a sediment budget.

From each change detection analysis between 2009 and 2010, we calculate the total area of deposition, total area of erosion, the net volume difference, total volume of deposition, total volume of erosion, and total volume of difference (e.g. Figure 8). The net volume difference is simply the difference between erosion and deposition and indicates whether a reach is experiencing net aggradation (when positive) or degradation (when negative) or is in approximate equilibrium (roughly zero). We also plot elevation change distributions and the Thresholded DoDs (Figure 8**Error! Reference source not found.**). DEMs of difference clearly apture the general pattern of deposition, scour, deposition seen at most BDSS (Figure 9).

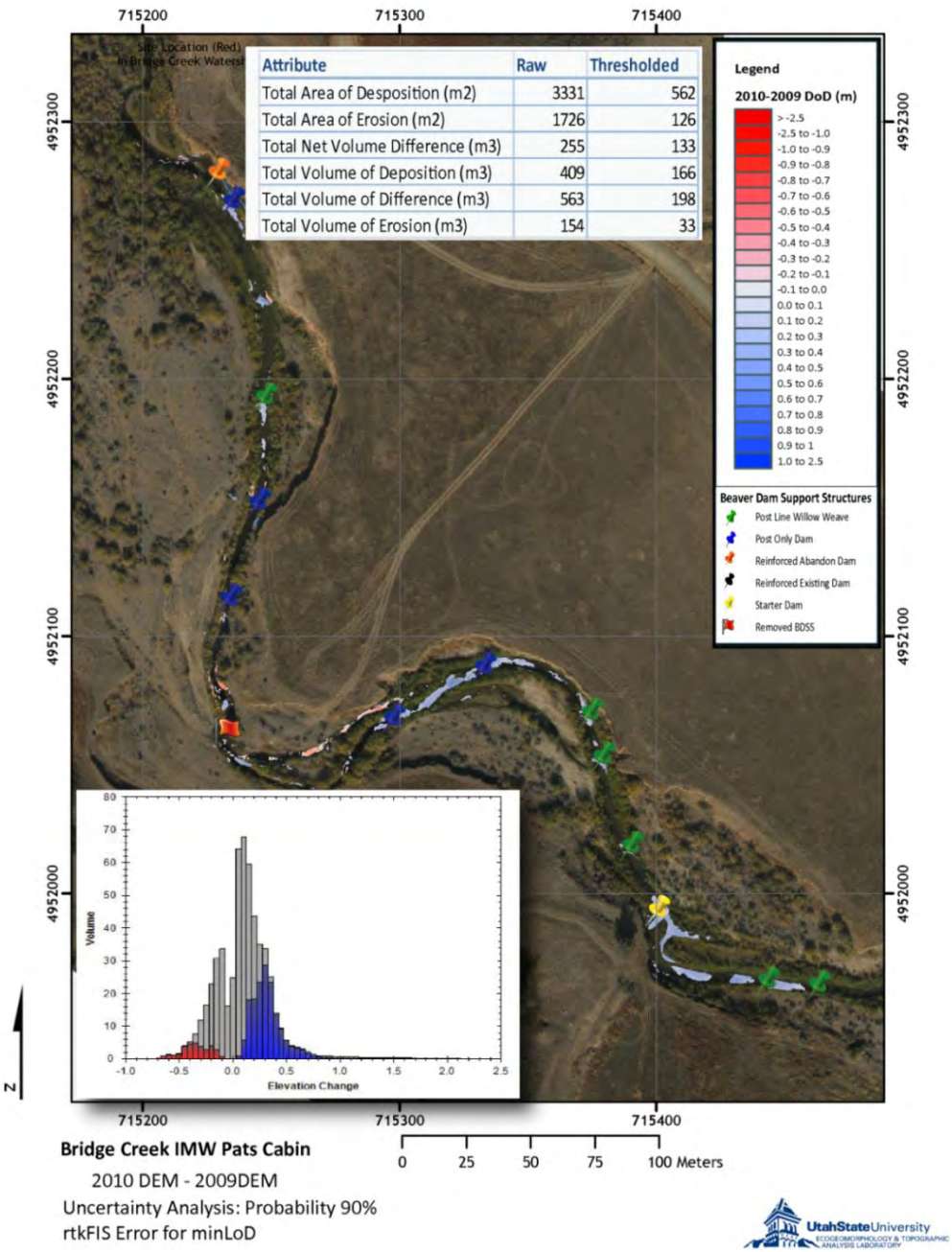



Figure 8. Pats Cabin treatment reach DEM of Difference calculation after applying an uncertainty analysis and thresholding to only include changes that have a 95% or greater probability of being real.

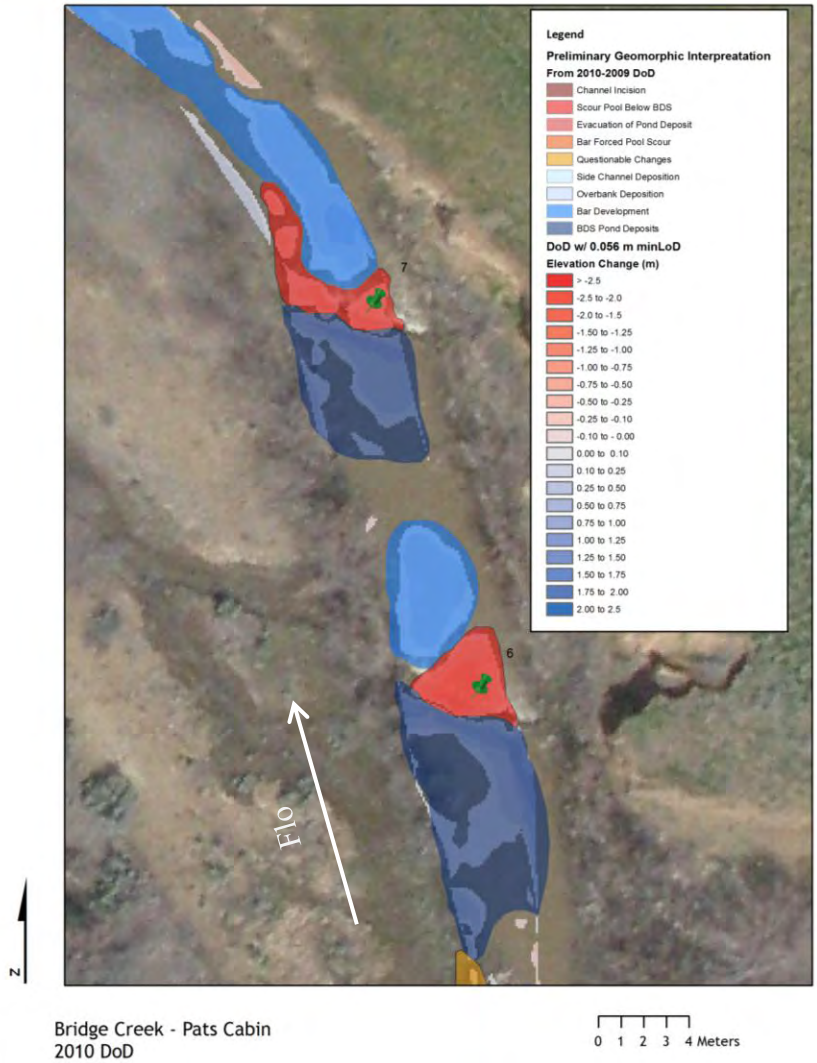
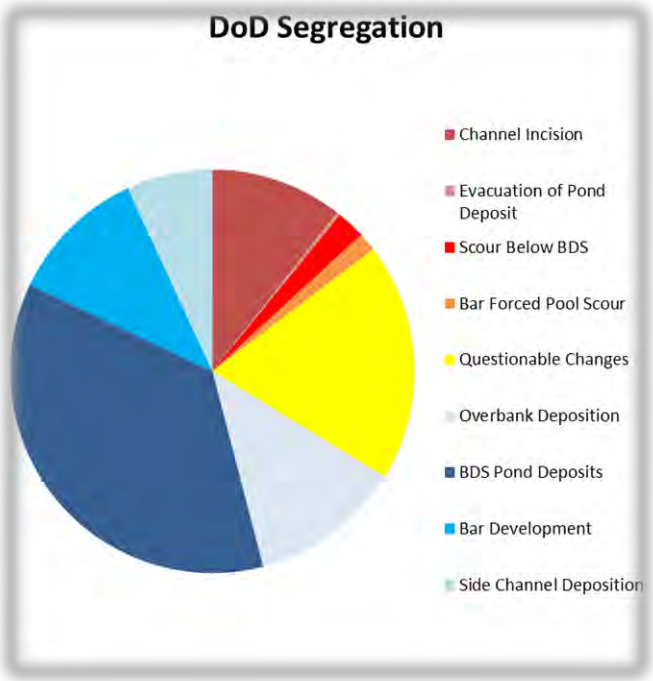
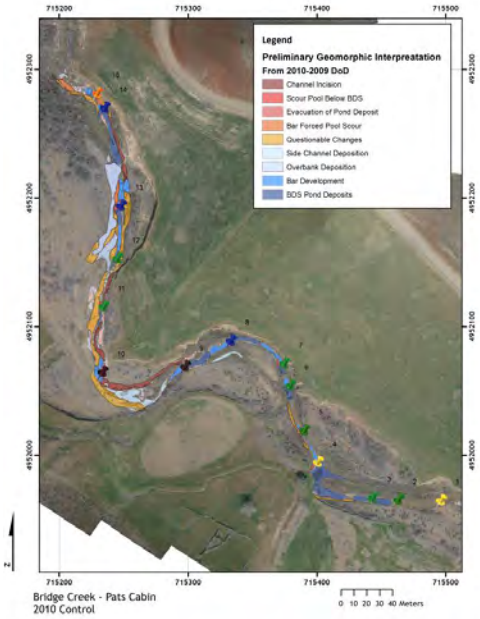



Figure 9. DEM of difference (post-restoration minus pre-restoration) from topographic surveys for a portion of treatment reach in Bridge Creek. Pushpins represent structure location. Blue color represents aggradation (deposition of sediments), and red represents erosion. General pattern was to have deposition behind structures, scour pool below structures, and deposition of the scour downstream from the pools.

As described in Wheaton (2008) and Wheaton et al. (2010), masking of the DoD budget (a.k.a. budget segregation) can be a very effective interpretation tool. The results of the geomorphic interpretation of the DoD results for Pats Cabin treatment reach in terms of specific mechanisms and/or processes of change is shown in Figure 10. There are many ways to segregate a budget, but as an example here, we show how the budget can be segregated in terms of the primary geomorphic responses in the reach. These processes include both those of concern (channel incision, evacuation of pond deposits) and those, which the restoration treatment is explicitly trying to encourage (e.g. BDSS pond deposits, bar development). We can also pull out those questionable changes, which may be in areas of sparse data, where we are not confident in




Erosion: 135 $m^3$ +/- 53

Deposition: 380 $m^3$ +/- 125

NET: + 245 $m^3$

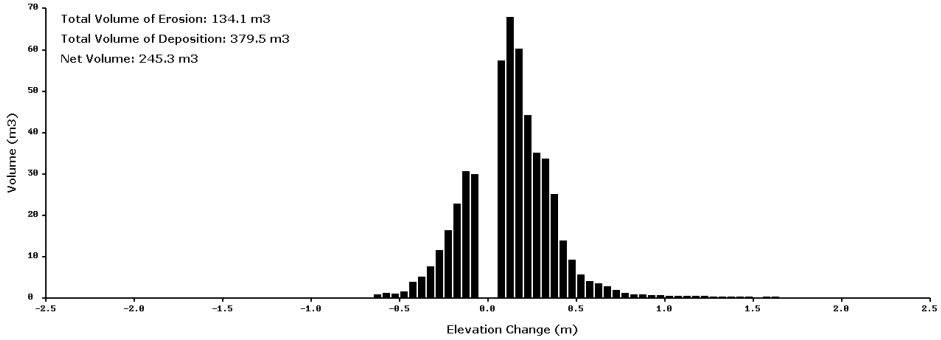



Figure 10. DEM of difference (post-restoration minus pre-restoration) from topographic surveys for a treatment reach in Bridge Creek. Pushpins represent structure location. Changes to the stream channels were identified and quantified.

the changes. For the example shown in Figure 10, we see that the majority of the volumetric change (in pie chart) is depositional (blue categories) and that nearly 40% of the total change is in the form of deposition in beaver ponds above BDSS (beaver dam support structures).

Table 2. Summary results of geomorphic change detection analysis from DoDs for 2009 to 2010 at all ten reaches. Simple refers to the unthresholded DoD. Propagated refers to a minimum level of detection uncertainty analysis based on propagated errors estimated for each DEM by a fuzzy inference system. 90% probability refers to a probabilistically Thresholded DoD based on the same propagated errors, but using a 90% confidence interval. 90% Probability w/ Bayesian, uses a Bayesian updating of the probabilities with a conditional probability estimated from a spatial coherence filter. NV refers to the net volume difference, VD is the volume of deposition, and VE is the volume of erosion (all in cubic meters).

| | | Simple | | | Propagated | | | 90% probability | | | 90% Probability w Bayesian | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | NV | VD | VE | NV | VD | VE | NV | VD | VE | NV | VD | VE |
| BC 04 | UPPER OWENS | 110 | 156 | 46 | 79 | 109 | 31 | 52 | 76 | 24 | 75 | 105 | 30 |
| BC06 | LOWER OWENS | 286 | 342 | 56 | 260 | 343 | 42 | 224 | 257 | 33 | 197 | 231 | 33 |
| BC07 | WOODWARD | 41 | 140 | 99 | 34 | 107 | 74 | 33 | 83 | 49 | 28 | 97 | 69 |
| BC08 | Meyers Camp | 46 | 94 | 49 | 45 | 78 | 34 | 39 | 63 | 24 | 34 | 62 | 29 |
| BC16 | VIP | -56 | 434 | 491 | -89 | 370 | 460 | -141 | 289 | 431 | -53 | 299 | 352 |
| BC18 | MONUMENT | 410 | 559 | 148 | 356 | 478 | 122 | 276 | 379 | 103 | 276 | 373 | 96 |
| BC22 | BOUNDARY | 65 | 101 | 36 | 64 | 93 | 29 | 62 | 83 | 21 | 47 | 70 | 23 |
| BC24 | PATS CABIN | 255 | 409 | 154 | 176 | 239 | 63 | 133 | 166 | 33 | 182 | 280 | 98 |
| BC26 | CORRAL | -229 | 212 | 440 | -173 | 139 | 311 | -137 | 110 | 246 | -159 | 140 | 299 |
| BC28 | Sunflower | 3 | 610 | 607 | 78 | 263 | 185 | 59 | 167 | 108 | 12 | 422 | 410 |
| | Totals/Column | 931 | 3057 | 2126 | 830 | 2219 | 1351 | 600 | 1673 | 1072 | 639 | 2079 | 1439 |

Table 2 summarizes preliminary results for Geomorphic Change Detection results. The "rtk" fuzzy inference system was utilized during calculations. Calculations for 90% probability with Bayesian updating utilized a five by five window at 60% to 100% values. For each reach the DoDs, elevation change distributions and summary tables are all available from the Editor upon request. Figure 11 shows the 90% probability data for erosion and deposition volumes graphically at each study reach to illustrate spatially where what changes have taken place and how they relate to each other in terms of relative magnitudes.

### *Discussion*

The results presented in this report describe the sampling methods and present the variation in physical habitat conditions within the Bridge Creek IMW study area prior to and following implementation of restoration actions. Based on this data we are able to draw a number of inferences regarding restoration monitoring designs, channel sampling approaches, and also how stream channel characteristics respond to the type of restoration being applied to Bridge Creek.

The staircase design (see Experimental Designs of IMW Chapter 3) being used to monitor the responses to restoration on the Bridge Creek IMW provided for 3 years of pre-restoration data prior to the first round of treatment implementation. Having three years of data has already allowed for a preliminary intervention analysis of the effects of restoration, even when annual variation in stream channel attributes is fluctuating in both treatment and control assessment units.

It should be noted that some of the annual variation observed in treatment and control reaches may be due to the PIBO protocols that have been applied to Bridge Creek. For example, all metrics describing pool habitat (% pools, pool frequency, pool depth) appear to fluctuate among years even before restoration implementation. This is likely due to how the protocol qualifies pools using the depth from the water surface. This criteria leads to a greater abundance of pools being counted during low flow years. As an example, 2007 and 2008 were particularly low water years for Bridge Creek, and this trend is easily observable in Figure. Future channel attribute monitoring on Bridge Creek using CHaMP protocols will be used to create a continuous survey of channel topography, and should be less influenced by water year.

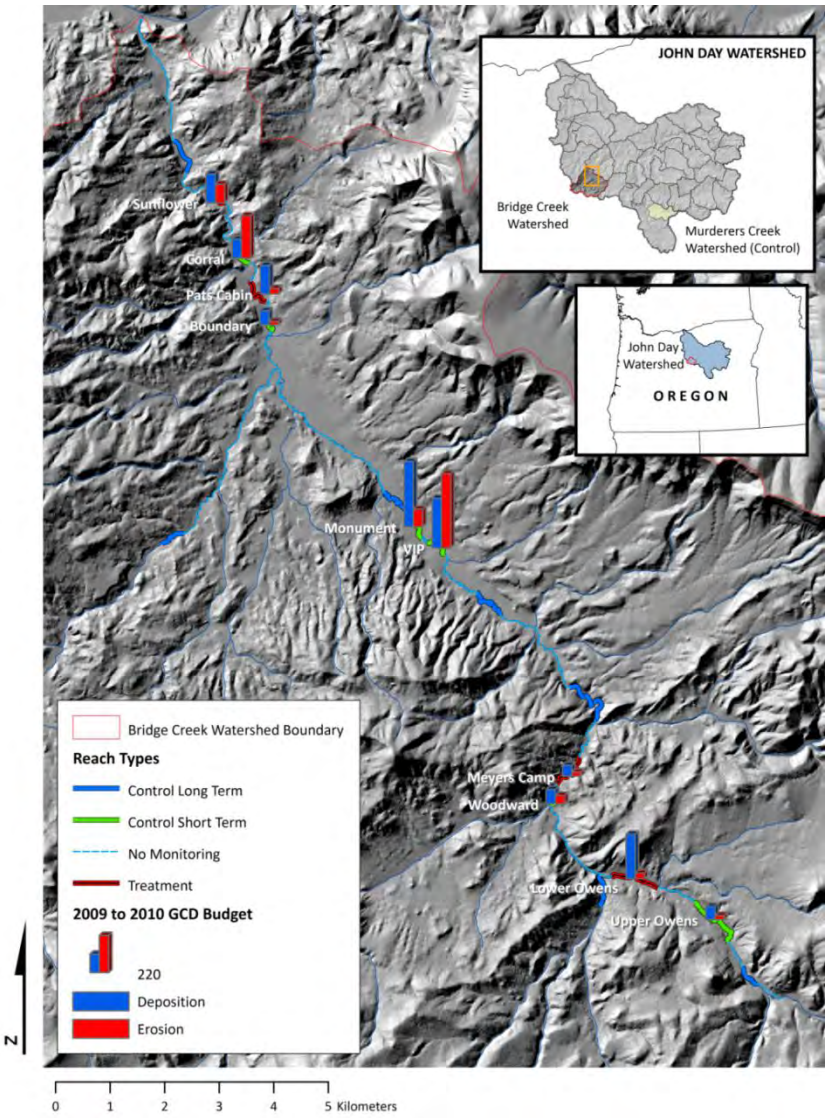



Figure 11. Graphical representation of relative magnitudes of erosion and deposition experienced at each of the 10 monitoring reaches between November 2009 and November 2010. Values based on geomorphic change detection using a fuzzy inference system and Thresholded to show only changes with a 90% or greater probability of being real.

This analysis also gives some insight as to how the channel is initially responding to the implementation of Beaver Dam Support structures (BDSS). Although not significant, the data does suggest that the average bankfull width of treated sections of channel has increased relative to control sections. This is likely due to the formation of beaver dams, and an aggradation of the channel onto the inset floodplain. BDSS have also significantly increased the percent of pool habitat, the number of pools, and the depth of pool habitat in treatment channel sections. This is not surprising, as in many cases BDSS create a beaver pond pool upstream of dams, and a scour just below dams. Some evidence of changes to the stream substrate are also apparent based on pebble count data. Although the particle D50 of treatment and control reaches have both decreased following restoration, there is some evidence that more fine sediment is being retained in treatment sections.

A strong to minor net depositional signal is recorded within the first year in all four treatment reaches. Of the six control reaches, four also show net depositional signals. Both Upper Owens and Boundary had two to three small beaver dams present, which blew out during the study period and experienced most of the net deposition in these areas. Similarly, Woodward had a couple of active beaver dams, where most of the deposition took place. By contrast, Monument is one of the few places in Bridge Creek with persistent long-term beaver dams that are major sites of net aggradation. Both VIP and Corral show strong net degradational signals. This is primarily associated with a major debris flow on Pats Cabin Creek that deposited a large volume of material in the Corral reach, which is subsequently being reworked, incised into and partially evacuated from the reach as the Creek carves out its old channel and a new side channel through this deposit. VIP shows a minor net degradation signal, which may be associated with the failure of two beaver dams. Figur

In Bridge Creek, the BDSS appear to be eliciting the response we expected, which is to cause net aggradation and reconnect the floodplain habitat. Whether this is a long term response is unknown at this point. Similar responses occur due to beaver dams without BDSS, as evident in these results, but these dams are short-lived and any aggradation is generally equally as short-lived. We have demonstrated a useful approach that can not only indicate whether a change has occurred but how those changes occurred. In addition to the aggradational response, we are observing an increase in channel complexity which we believe will be beneficial to fish. More time is required to determine whether the steelhead population will become more productive as result of these stream restoration efforts.

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# CHAPTER 8: Growth Potential Models

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### *Introduction*

The amount of heat and water delivered to a stream is determined by external factors such as valley topography, upland vegetation, precipitation, air temperature, wind speed, solar angle, cloud cover, relative humidity, phreatic groundwater temperature and discharge, and tributary temperature and flow (Poole and Berman 2001). Internal stream structure, such as channel slope, width, topography, and pattern, substrate, and vegetation influence how heat and water are distributed and exchanged between the channel, riparian and alluvial aquifer (Poole and Berman 2001). Together, external and internal factors, determine stream temperature. Human impacts, such as those that affect riparian and upland vegetation, water withdraws, dam operations, and channel modifications can influence all these processes. In fact, because of the sensitivity of temperature to human influences, this metric receives considerable attention in the TMDL process of the Clean Water Act (Boyd and Kasper 2002).

Physiological processes of organisms are generally temperature dependent. Temperature influences overall stream production, as well as salmonid growth. Growth is related to survival and production. These fish responses are often limited by temperature in several areas of the Columbia River Basin. In the John Day basin, temperature is thought to be problematic in several salmonid bearing streams and is the focus of several stream restoration projects. In particular, the Middle Fork Intensively Monitored Watershed study is currently implementing a large scale restoration effort, with several actions expected to address temperature issues (Bouwes 2011).

Because temperature is an integrative response across multiple external and internal stream factors, is sensitive to multiple human disturbances, and is crucial in influencing salmonid production, this metric is a research, monitoring, and evaluation focus of the ISEMP John Day Pilot project. We have developed a model to map potential fish growth across stream reaches of the John Day by combining models that estimate heat budgets based on physical inputs and bioenergetics models that use these heat budgets and invertebrate abundance information to estimate fish growth.

We use two temperature models to estimate growth rates at different reaches and streams throughout the John Day. The first temperature model was developed by ISEMP to estimate spatially and temporally continuous stream temperatures for the John Day River basin. Daily Land Surface Temperature [LST] measures from NASA's Moderate Resolution Imaging Spectroradiometer [MODIS] and *in situ* water temperature logger data, collected by various agencies and compiled by NOAA, are the covariates in the models. The spatial structure inherent in the LST datasets is leveraged such that models can be developed over large geographic regions across years. The temporally continuous nature of the models allows for the development of summary metrics that characterize the in-stream thermal regime for each stream reach such as; growing Season Thermal Inputs, counts of days above minima and maxima, and the timing of thermal milestones.

The Heat Source model (Boyd and Kasper 2002) used in the John Day TMDL process, uses habitat and landscape information to describe physical processes that define heat transfer and water transfer for the total heat budget for a reach. As is done in the TMDL process, current and historical estimates of temperature load as well as the impacts of different scenarios, such as the increase of the riparian canopy through a riparian fencing project or increased discharge by purchasing instream water rights, on stream temperature can be estimated with the Heat Source model (Figure 1). In fact, this model was used to estimate the impacts of the Middle Fork IMW on temperature (Crown 2010; Figure 2).

Under the Clean Water Act, biologically based critical thresholds have been established (Figure 1). However, these threshold are very crude and do not adequately describe the true impact of temperature to salmonids. Juvenile steelhead can exhibit negative growth under the all temperature regimes described in Figure 22. However, if enough food is available they can actually grow better under this temperature regime than cooler temperature regimes. Thus growth is an interaction between temperature and food.

The rate at which respiration and the consumption rates change as a function of temperature and body size has been determined for several fish species (Hanson et al. 1997). These processes have been summarized into bioenergetics models that allow for examination of factors affecting growth and consumption rates. Growth and temperature can be measured in the field, and consumption required in maintaining metabolism and obtaining the observed growth rates can be estimated with this model. ISEMP examined invertebrate information (drift and/or benthic samples) and growth rates of juvenile steelhead collected in the John Day to develop a relationship between prey density and percent of the maximum consumption rates of juvenile steelhead. This simple relationship could be used to estimate growth potential of different stream reaches that have temperature and invertebrate abundance information.

Incorporated with the Heat Source model, which describes temperature regimes under restored and current conditions, temperature, invertebrate, and fish density data could be used to estimate fish production . Restoration activities addressing these factors can then be prescribed for these reaches with anticipated impacts also described by these models.

### *Methods*

#### *ISEMP Temperature Model*

The spatial extent of these models includes all stream reaches in the John Day River basin in eastern Oregon, USA. The land surface area directly draining into each reach (Reach Contributing Area [RCA]) was identified and used as the working spatial resolution of analysis and prediction. Daily LST datasets for 2001-2009 at ~1 $km^2$ spatial resolution were downloaded from NASA. Cloud cover measurement gaps in LST were filled by developing individual $4^{th}$-order polynomial regression models for each 1 km pixel for each year. A zonal mean LST for each RCA was calculated. Site-specific daily mean and maximum water temperature was calculated and compiled from loggers deployed by various agencies for the same time period as the LST dataset. Parameter estimates from site-specific regression models were used to develop basin-wide predictions of water temperature within and across years.

A cross-correlation analysis was conducted on pre-whitened and differenced data to identify any significant time lags in the correlative relationships between LST and water

temperature. A zero lag had the highest cross-correlation coefficient, so LST and water temperature data were aligned temporally. Spectral analysis yielded no consistent frequency information beyond the obvious seasonal signal. Linear regression models using LST as the predictor variable and water temperature as the response variable were developed for 2001-2009 for RCAs with sufficient data within a year. Separate models for the first (Julian days 1-196) and second (Julian days 197-365/6) halves of the year were developed for any RCA with at least 60 days of water temperature data for the first and/or second halves of a given year. Separate models were developed for mean and maximum water temperature. Parameter estimates from models with adjusted $r^2$ >= 0.60 were used to calculate median model coefficients for each year. The median coefficients were used to estimate mean and maximum water temperature for every day from 1 January 2001 to 31 December 2009 for every stream reach in the John Day River basin.

***Heat Source***

Heat Source was use by the Oregon Department of Environmental Quality to evaluate the total maximum daily load of the heat budget for the Middle Fork John Day River (Crown 2010). This extensive modeling effort conducted by ODEQ was leveraged by ISEMP to provide temperature inputs into the growth potential model. Methods of field collection, model development and calibration for Heat Source Middle Fork evaluation can be found in Crown (2010). In general, Heat Source estimates heat budgets and mass transfer of water to estimate stream temperature. In the Middle Fork, stream temperature was estimated every 200 m of stream every 0.5 min. over the summer (Table 1). Information about valley topography, stream position and aspect, stream elevation and gradient, channel width, vegetation, wetted widths are summarized in a GIS platform and used as inputs to Heat Source to estimate solar inputs. In addition to this GIS-derived landscape information, other inputs to the model were also used to estimate stream temperature (Crown 2010). These include:

- Constant values that applies to the whole model corridor
    - Wind function coefficients
    - Deep alluvium temperature
- Parameters that vary by model node
    - Channel bottom width
    - Channel angle z
    - Manning's n
    - Sediment thermal conductivity
    - Sediment thermal diffusivity
    - Sediment/hyporheic zone thickness
    - Percent hyporheic exchange (Porosity)
- Parameters that apply to tributary inputs

- Flow
- Temperature

Table 1: Assumptions made for the different scenarios modeled by Heat Source for the Middle Fork total maximum daily load evaluation and the Intensively Monitored Watershed study (From Crown 2010).

| | |
|---|---|
| "Current" | Current Calibrated Condition (see **Appendix A** for details). Model results were produced every 0.5 min and 200 m. The model extent was 112.95 km. |
| "Restored Vegetation" | System Potential Vegetation and increased hyporheic exchange in meadow reaches (see **Section 2** and below for details). |
| "Restored Flow" | No points of diversion or ditch inputs and tributary flows adjusted to OWRD's estimates of natural flow (see **Section 3** for details) |
| "Restored Morphology" | Bankfull widths reduced by 10-50% while cross sectional area preserved (see **Section 4** for details). |
| "NTP" | Natural Thermal Potential: combining the inputs of system potential vegetation, natural stream flow, and range of reduced bankfull width estimates (see **Section 5** for details). Hyporheic flow restored in meadow reaches. No other temperature adjustments were made to tributary inputs. |
| "Pre-restoration" | Scenario estimating instream temperatures before major current restoration projects were started. |
| "Post-restoration" | Scenario estimating instream temperatures when major current restoration projects near natural thermal potential. |

Crown (2010) calibrated the Heat Source model for the Middle Fork using 2002 temperature information. Several scenarios were modeled to evaluate the impacts of they would have on stream temperature including natural thermal potential (NPT), which assumes historic riparian vegetation, no water withdraws, connected tributaries, etc (Table 1; Crown 2010). Different restoration scenarios were also modeled including full restoration (back to natural thermal potential scenario) of either vegetation, flows, or stream morphology. In relation to the Middle Fork IMW, NPT, current conditions, and post-restoration scenarios (Table 1; Figure 2).

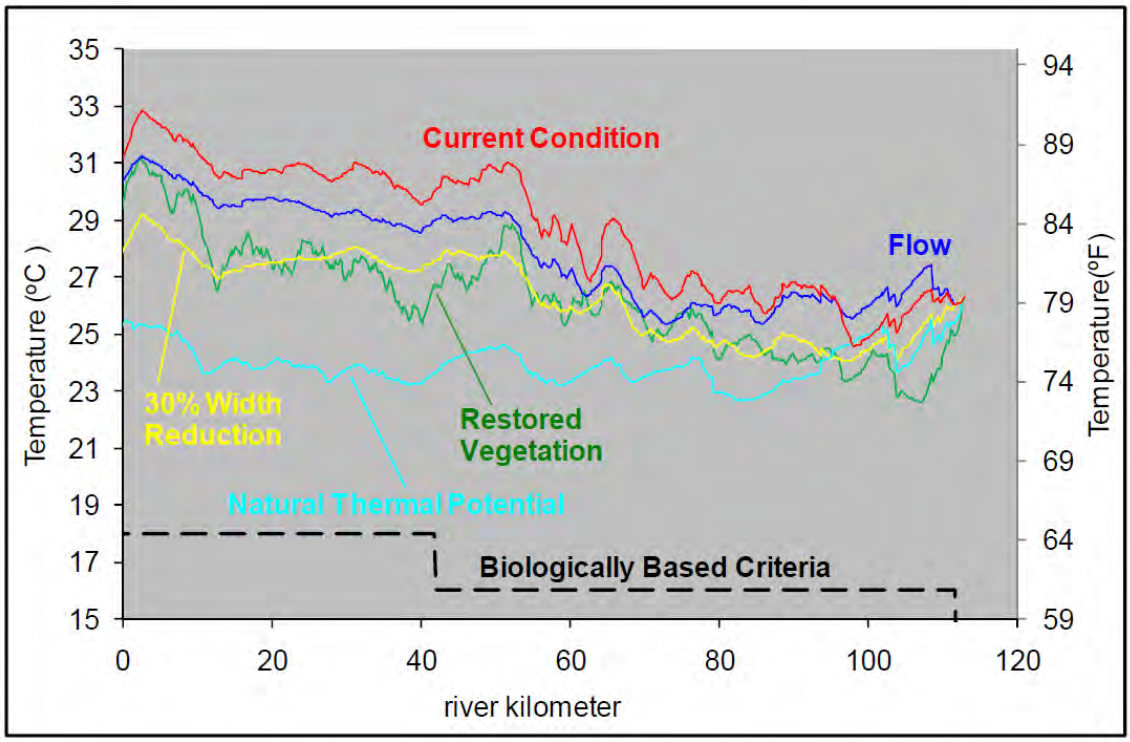


Figure 1. Longitudinal profile of 7 day daily average maximum stream temperatures for the Middle Fork John Day in summer 2002 as modeled by ODEQ (2010) using HeatSource, across different scenarios. Scenarios include current conditions (2002; red line), restored flows (no withdraws and reconnected tributaries; dark blue line), restored vegetation (historic unimpacted riparian vegetation; green line), and decreased stream width (stream width:depth ratio was assumed to be smaller pre- European settlement: yellow line), and natural thermal potential (all historic condition of riparian vegetation, flow, and morphology; light blue line).

### *Growth Potential Model*

The rate at which respiration and the maximum consumption rate changes as a function of temperature and body size has been determined for several fish species (Hanson et al. 1997). These processes have been summarized into bioenergetics models that allow for examination of factors affecting growth and consumption rates. The basic physiological processes affecting these rates exhibit little variability among individuals. Bioenergetics models use an energy balance equation to describe energy input (consumption) equal to energy output as:

$$(1)\ \text{consumption} = \text{growth} + (\text{respiration} + \text{wastes})$$

Respiration and waste can be further divided into more specific functions that have been well established in the laboratory (Hanson et al. 1999). Therefore growth and temperature can be measured in the field and consumption required to maintain metabolism and obtain the observed growth rates can be estimated with the bioenergetics model.

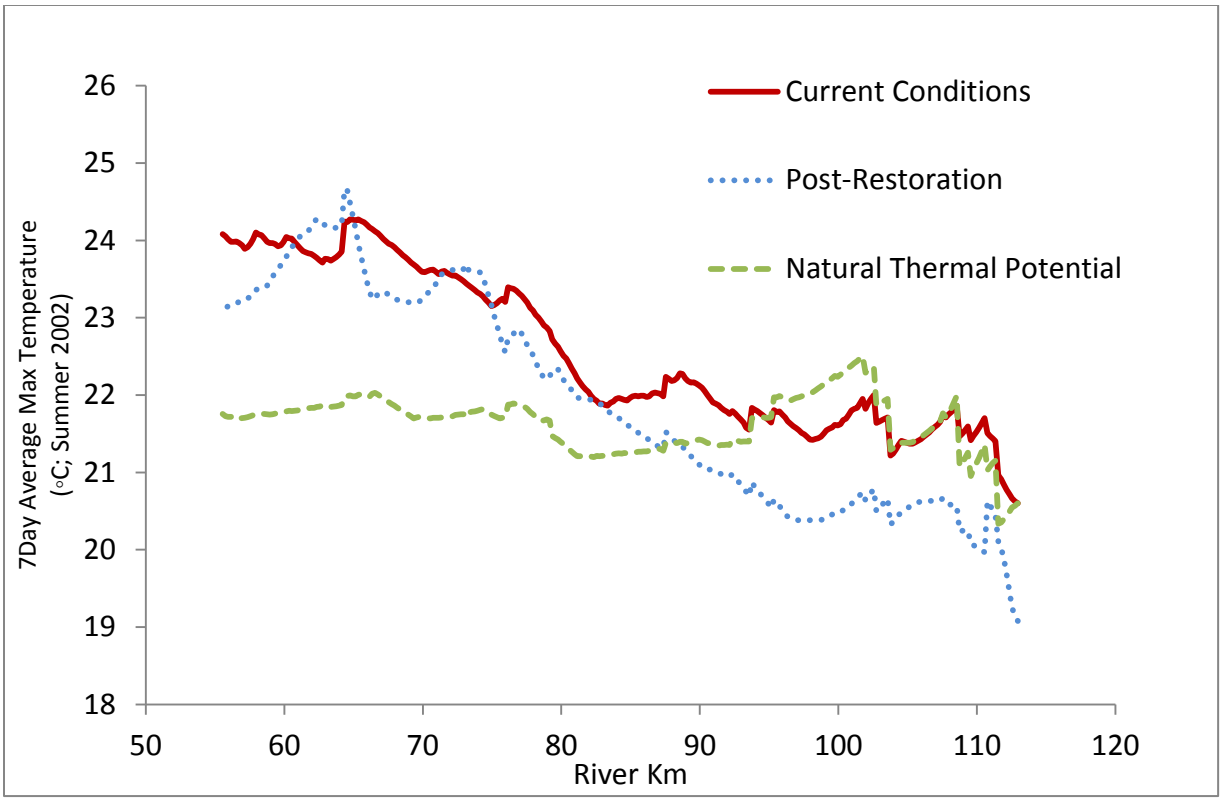


Figure 2. Longitudinal profile of average maximum weekly stream temperatures for the Middle Fork John Day in summer 2002 as modeled by ODEQ (2010) using Heat Source, across different scenarios. Scenarios include current conditions (2002; red solid line), and post-restoration implemented in the Middle Fork IMW (blue dotted line), and the temperature regime under "natural" conditions (Natural Thermal Potential; green dashed line).

The equation can also be rewritten as:

(2) growth = consumption – (respiration + wastes)

In this equation, if we can find a means to estimate consumption than we can estimate growth if we know fish's thermal experience (stream temperature) and size.

ISEMP has begun testing this approach using a suit of observations of juvenile salmonid growth rates, macroinvertebrate abundances, and stream temperatures collected as part of the Bridge Creek IMW monitoring project. Individual juvenile steelhead (*O. mykiss*) growth, drifting and benthic invertebrate samples, and stream temperatures were collected within 10 stream reaches chosen to encompass a range of physical habitat characteristics and temperature profiles. Steelhead growth and temperature measurements were used as inputs for bioenergetics simulations to estimate proportion of maximum juvenile steelhead consumption (P-values). Linear and non-linear regression analysis was used to determine if food abundance could explain variation in consumption, and determine a measure of invertebrates that may provide a best description of food availability. In this initial testing, measurements of the total biomass of terrestrial and aquatic invertebrates in the drift explained the greatest amount of variation in estimates of salmonid consumption along a non-linear type II predator feeding response curve (Figure 3). This relationship represents a first cut at the development of an accessible, yet

mechanistic relationship between macroinvertebrate sampling abundances and juvenile salmonid consumption. Thus, if we estimate steelhead consumption based on the total amount of drifting invertebrate biomass, using equation 2 we can estimate growth rates under a given temperature regime.

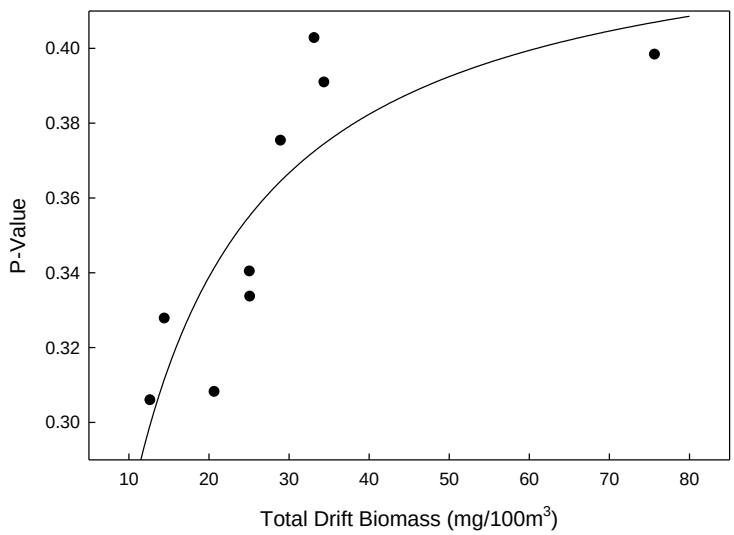


Figure 3: Non-linear regression of *O. mykiss* consumption (proportion of maximum consumption for given temperature regime and fish size or the P-value) and total drift biomass (mg/100m$^3$)

We used the equations described in the Wisconsin bioenergetics model for steelhead (Hanson et al. 1999). The physiological processes responsible for *O. mykiss* growth have been extensively evaluated in the laboratory and are well understood and documented (Rand et al. 1993). Some of these parameter settings have been revised and documented in Railsback and Rose (1999). We used the same parameter settings for the consumption, respiration, specific dynamic activity, egestion, and excretion equations as described in Railsback and Rose (1999). We also used the same energy densities for prey and red band trout (*O. mykiss*).

### *Model Application*

#### *Growth Potential in the Middle Fork John- An estimate of the influence of the Middle Fork Intensively Monitored Watershed Study on Steelhead growth.*

The National Oceanic and Atmospheric Administration (NOAA), in coordination with the Oregon Watershed Enhancement Board (OWEB), has funded an IMW in the upper Middle Fork of the John Day River basin, Oregon. The goals of the Middle Fork IMW are to improve adult and juvenile salmonid freshwater habitat in the upper Middle Fork John Day using a variety of restoration actions, to assess how restoration actions alter stream habitat conditions, and to understand the causal mechanisms between stream habitat restoration and changes in salmonids production at the watershed scale.

The Middle Fork IMW study area supports several species of fish including spring and fall Chinook salmon *Oncorhynchus tshawytscha*, summer steelhead *O. mykiss*, bull trout *Salvelinus confluentus*, Pacific lamprey *Lampetra tridentata*, and westslope cutthroat trout *O.*

*clarkii lewisi.* Spring Chinook salmon and summer steelhead are the predominate salmonids inhabiting the Middle Fork watershed. Both steelhead and bull trout are listed as threatened species. Spring Chinook salmon are not currently listed. Steelhead are the most widely distributed salmonid species occupying most tributaries and mainstem habitats. Chinook distribution is slightly more confined to mainstem habitats and larger tributaries compared to steelhead although juvenile Chinook often migrate into cool-water tributaries during warm summer periods. Both steelhead and Chinook will be the focus of fish monitoring for this IMW.

Limiting factors for both species are temperature, key habitat quantity, and sediment. Chinook spawning has been increasing over time but not smolt production and steelhead spawning has been decreasing. The limiting factors identified form the basis for the type of restoration planned by Working Group partners. Restoration actions have been divided into SIX separate categories: 1) channel reconfiguration and floodplain reconnection; 2) fish passage, 3) flow increase, 4) grazing/upland management, 5) instream habitat enhancement, and 6) riparian fencing and planting (Figure 4).

ISEMP has proposed four different experimental designs are proposed to determine the effects of restoration at different scales: watershed design, mainstem treatment control design, tributary design, and temperature modeling design at the watershed and reach scale (Bennett and Bouwes 2009). Here we describe the temperature modeling used to evaluate the potential of the MF IMW to influence steelhead production by improving juvenile steelhead growth.

Using these parameter inputs, we estimated the growth potential of every 200 m reach of the Middle Fork John between km of 20 g juvenile *O. mykiss* between July 1 to August 15, 2002 because this time period generally encompasses the warmest most stressful period of the growing season, and was also the time period ODEQ used HeatSource to model difference scenarios. Growth was estimated on a daily time step using average daily temperature. We estimated growth over the three temperature scenarios: current conditions, post-restoration, and natural thermal potential.

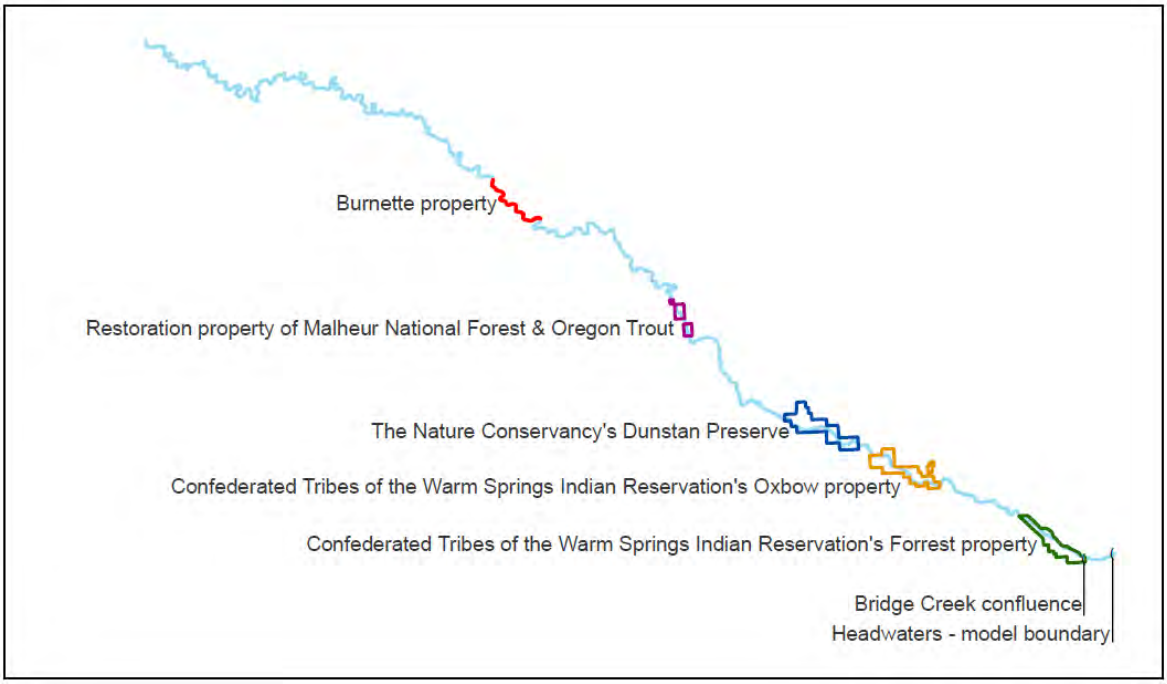


Figure 4: Location of restoration actions of the Middle Fork John Day Intensively Monitored Watershed study.

The proportion of the maximum consumption (the P-value) that was used for energy inputs was derived from the drift P-value relationship described above. Drift was collected at nine location in the Middle Fork John Day as collaborative effort between ISEMP and North Fork John Day Watershed Council. Two nets were used to estimate drift at each location. We total drift biomass to estimate P-value using the relationship in Figure 25. In general, drift values were relatively low but highly variable. Because drift values were highly variable, we used the average total drift biomass across all sites. The relationship estimated P-values lower than we have observed elsewhere throughout the John Day Basin, and the resulting P-value resulted in negative growth across all temperature regimes. The non-linear relationship quite sensitive to drift biomass, and at low values, low P-values are produced. There for we used a linear relationship based on the invertebrate/P-value information obtained through ISEMP invertebrate study. Although linear relationship does not fit the values quite as well, the relationship is still significant and less sensitive to low drift values. Thus, we used this to produce a P-value of 0.32 for the Middle Fork John Day.

***Validation***

To evaluate whether the model is estimating growth appropriately, we predicted the expected growth in 2011 and compared this to observed growth based on ODFW fish surveys for that summer. ODFW sampled sites at river km 77, 92, 94, 99, 108, 112, twice over the summer (mid-July to early August and in early October). Fish captured on first event were PIT tagged and recaptured fish were scanned for tags. Differences between weights from first and subsequent events of recaptured fish were used to estimate growth.

Temperature was estimated over this time period using the ISEMP temperature model for each of the sites where fish sampling occurred. Again, the only drift information we had

available was from the efforts described above. For the same reasons previously described we used a P-value of 0.32. We used these input variables and the above input parameters to conduct our modeled estimates of growth rates for juvenile steelhead at these six sites.

In addition to using the Middle Fork site to validate the model, we also used reaches elsewhere in the John Day, including Bridge Creek a tributary to the Middle Fork John Day. Here we used the drift invertebrate information collected from CHaMP and the relationship in Figure 3 to create reach specific P-values. Temperature was estimated using the ISEMP Temperature model. Growth was modeled for the average size fish located at each reach over the time period between recaptures (approximately mid-July- Oct 1).

***Results***

Growth was generally under predicted in the Middle Fork John Day (Figure 5). We calibrated the model to more accurately reflect Middle Fork productivity. We increased P-value to 0.4 which produced fish growth surrounding the 1:1 line. Because drift and growth were not collected in the same year we did not continue calibration beyond this change.

In general, the stream restoration planned by the Middle Fork IMW resulted in lower predicted temperatures than current temperatures, throughout the project area. In fact, restoration efforts were able to bring down temperatures lower than NPT conditions in the upper half of the project area, but the lower project area remain warmer than NPT after restoration (Figure 6). The results of the Heat Source modeling are discussed in more detail in Crown (2010).

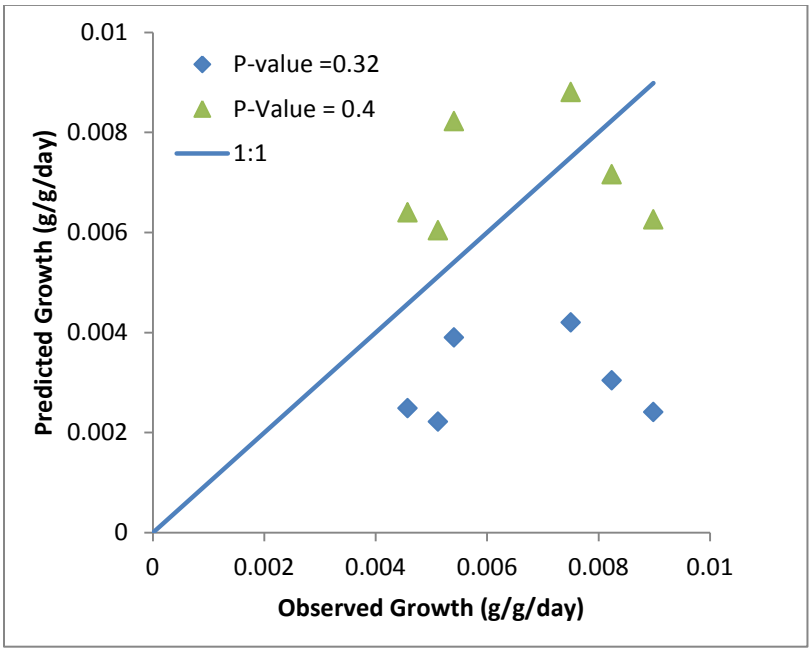


Figure 5. Observed versus predicted growth (normalized by grams of growth per gram of fish per day) at six sites in the upper Middle Fork John Day River. If we were able to predict actual growth, the points would fall on the 1:1 line. Blue diamonds are original P-value of 0.32, and green triangles are growth modeled with a P-value of 0.4.

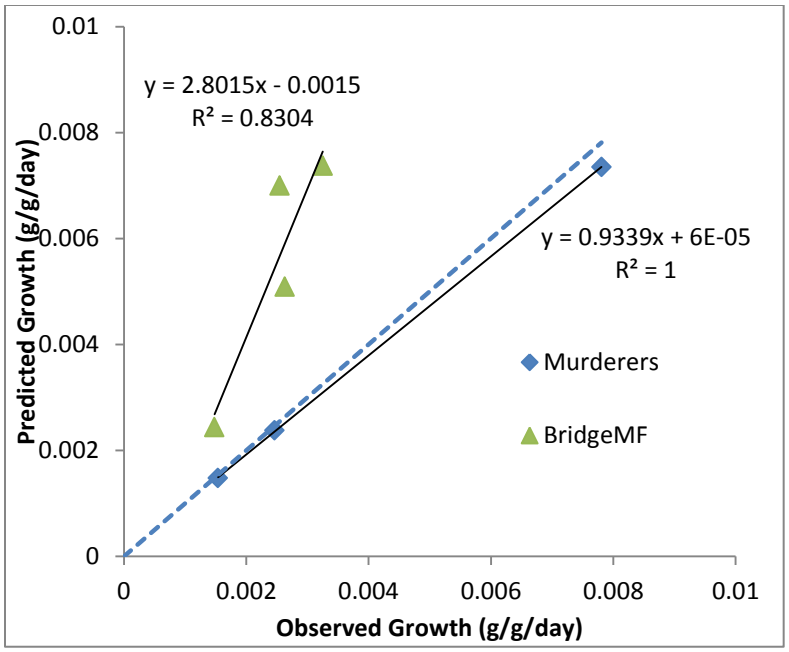


Figure 6. Observed versus predicted growth (normalized by grams of growth per gram of fish per day) at four sites on Bridge of the Middle Fork (green triangles), and 3 sites on Murderers Creek (blue diamonds). If we were able to predict actual growth, the points would fall on the 1:1 line (dotted blue line).

Differences in growth in this model exercise are driven by temperature alone, since a common P-value was used throughout the Middle Fork John Day. The relationship between maximum weekly temperature and growth is generally negative (Figure 7). The model suggests that a fish starting off at 20 g is likely to be 3 g smaller, lower in the river than at the top of the river. Because fish are growing about 10 g during this time period, this represents a 30% reduction in growth.

Modeled temperature scenarios suggest positive growth occurs in the upper half of the project area under all scenarios (Figure 8). In the lower half of the project area, fish under current thermal conditions, and conditions after restoration, will still demonstrate negative growth during the warmest 1.5 month of the year in lower part of the project area, but should exhibit positive growth under NTP.

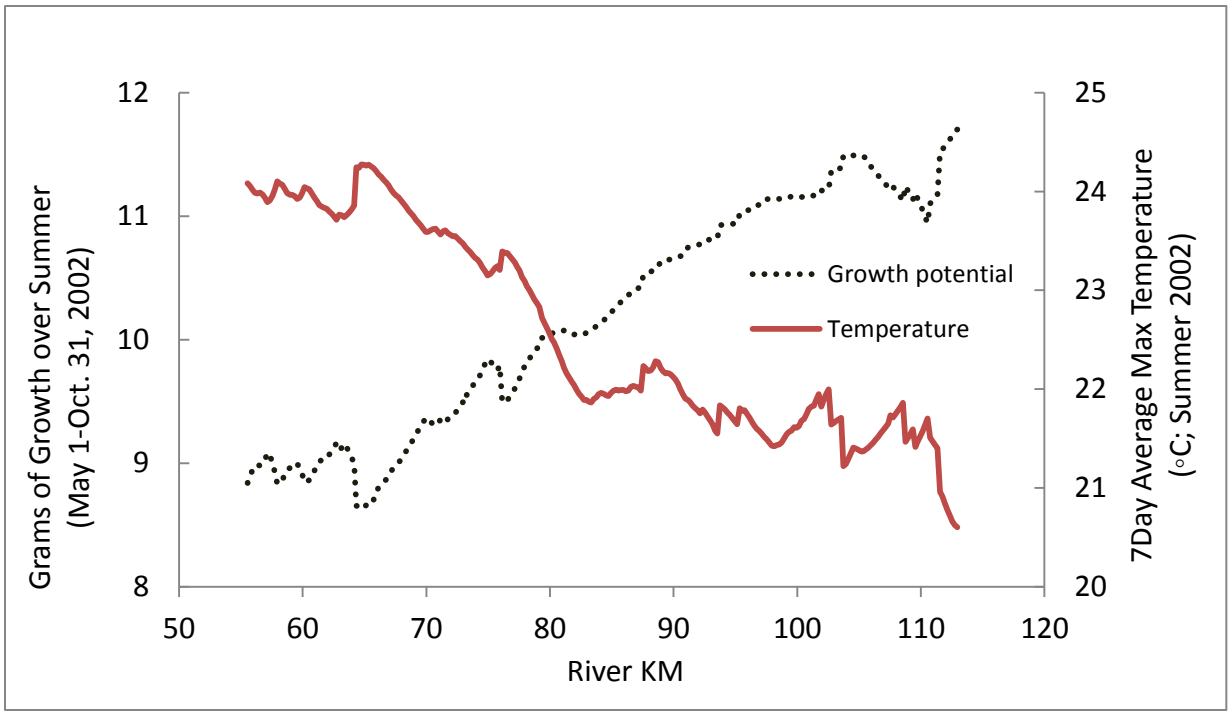


Figure 7. Relationship between temperature profile as modeled from Heat Source in 2002 (red solid line) and the modeled growth potential of 20 g juvenile steelhead during May 1-Oct. 31, 2002 (black dotted line) along the upper Middle Fork John Day River.

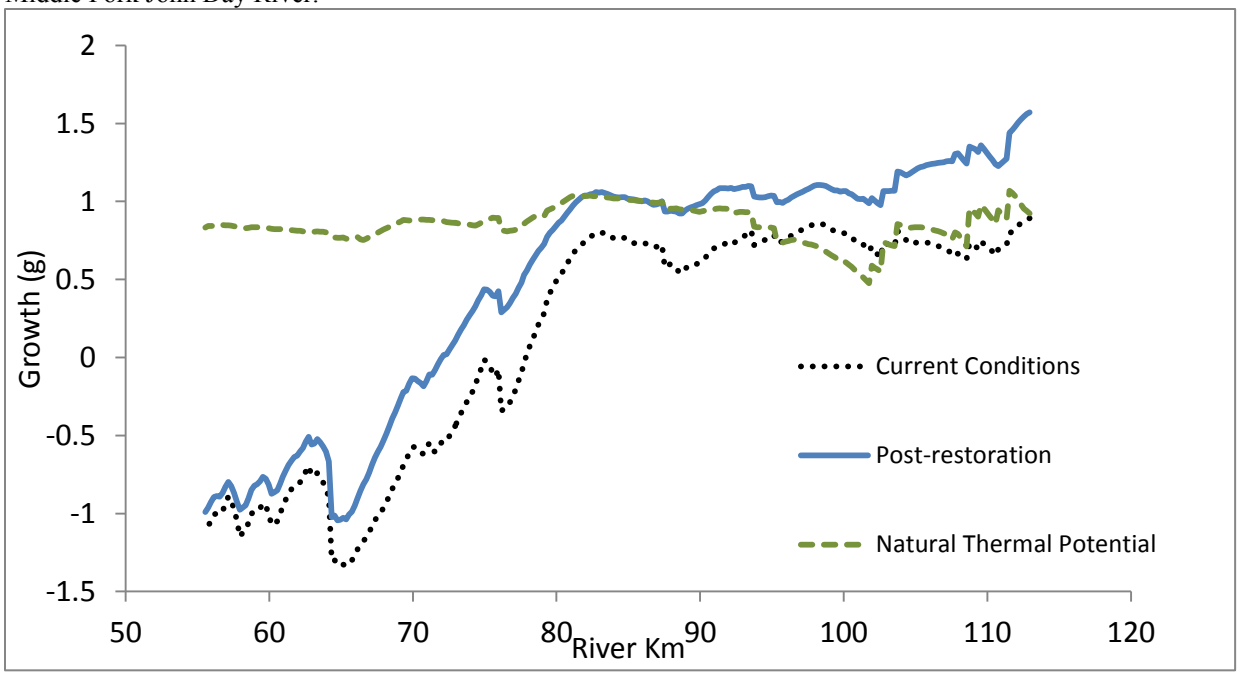


Figure 8. Growth (g) potential of a 20 g *O. mykiss* between July 1-Aug 15 (2002), for current thermal conditions (black dotted line), post-restoration as planned by the Middle Fork IMW study (blue solid line), and under natural thermal potential (green dashed line) for the upper Middle Fork John Day River.

### *Discussion*

The growth model accurately predicted growth on the stream that it was partially developed in (Murderers Creek); the relationship (Figure 3) was developed 5 yrs earlier than the data used to validate the model. The model predicted growth in MF Bridge Creek precisely but not accurately. In the Middle Fork John Day, the model did not predict growth accurately or precisely. This may be due to the high variability in drift estimates observed in this relatively large river. Drift was estimated using two nets in MF Bridge Creek and Murderers Creek, and during the model development process. This same protocol was used in the Middle Fork John Day even though the river is 4 times greater in width. Larger streams may require more drift nets to estimate drift more accurately and precisely. In addition, drift and fish sampling occurred in different years, and thus the drift we used to estimate growth may have been different than the year growth actually was observed.

Once calibrated the model produced results in the Middle Fork that was properly scaled to the amount of invertebrate biomass available for steelhead. Thus, we believe the relative influence of stream temperature between scenarios was properly portrayed. What is clear is that the large range of temperatures observed along the longitudinal gradient of the Middle Fork John Day has a substantial impact on growth rates of juvenile salmonids, especially as growth approaches near lethal temperatures in the lower reaches. Stream restoration will mitigate for some of these impacts; however furthers studies will be required to determine how this translate into recruitment to later life stages.

We believe the combination of different temperature and growth potential models have the ability to help synthesis the multiple effects of land use and stream restoration on the integrative metric of temperature and apply impacts to salmonids. The approach is fairly simple and does not require much data input and may be a powerful means to evaluate and plan restoration as well as provide information to life-cycle models used to assess the status of these populations.

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# CHAPTER 9*:* Estimating Energy Availability and Carrying Capacity of Salmonids in a Stream Reach

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### *Introduction*

While the growth potential model described in Chapter 8 highlights the importance of temperature and prey availability, it completely ignores physical structure in streams (e.g. pools, riffles, gradient) in driving salmonid production. Quantifying physical structure is a large emphasis of habitat monitoring protocols such as CHaMP. A recent modeling approach incorporates components of foraging theory, physiology, distribution of individuals, and explicit spatial descriptions of streambeds, and offers great promise to further our understanding of fish-habitat relationships (Hayes et al. 2007). This approach begins with a spatially explicit, three-dimensional representation of the streambed. Hydraulic models use this streambed representation to generate spatially explicit depth and velocity estimates. A model of drifting food items uses hydraulic model output to predict spatially explicit food distribution, while a mechanistic foraging model predicts which drifting food items are ingested by foraging fish in the modeled stream area. Using energy consumed (food ingested) and energy spent (metabolism and swimming costs), the approach calculates net rate of energy input as the difference of these two quantities. The distribution of NREI can also be used to estimate abundance of fish in a reach.

In ISEMP we are attempting to incorporate this latest development in fish foraging models to estimate energy intake and carry capacity, with the CHaMP protocol customized to provide data inputs for these model. We expect these model results to be used directly as input into life-cycle models that will likely be used in regional population assessments.

### *Methods*

The mechanistic model we are using to represent how a fish makes a living in a reach incorporates how water flows through the reach (hydraulic model), how food is delivered throughout the reach (drift transport model), how fish capture drifting prey (foraging model) and expend energy in the process (water velocity) (Figure 1). The net rate of energy intake (NREI) of salmonids is the difference in the energy gained from foraging and energy lost through swimming. The NREI there can be converted into growth rates of salmonids and the model can map areas of a reach where fish have positive NREI (Step 6 of Figure 1). The number of foraging areas that have a positive NREI can serve as an estimate of carrying capacity of the reach (Step 7 of Figure 6).

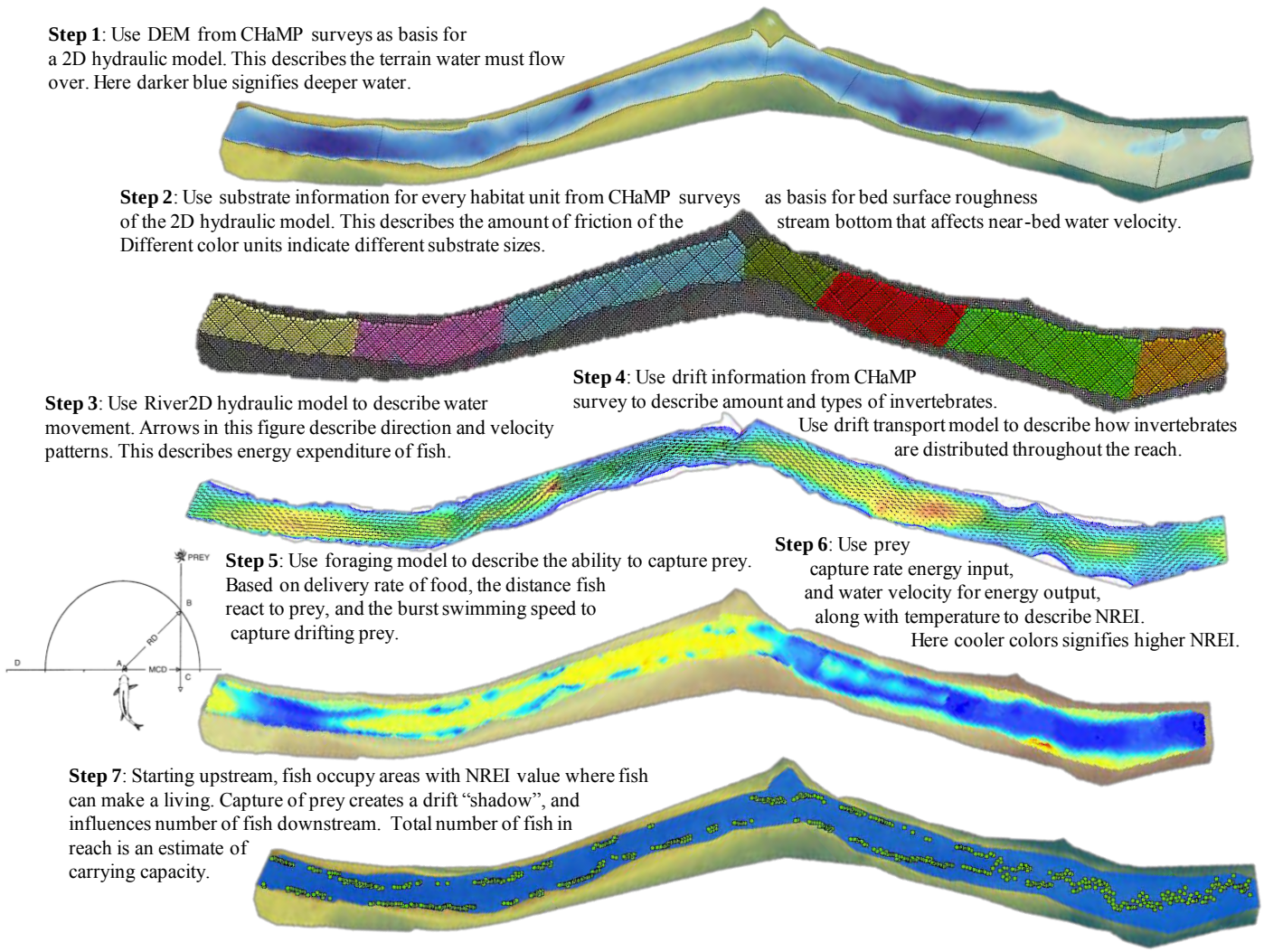


Figure 2. Estimating energy available (net rate of energy intake or NREI) and carrying capacity of juvenile steelhead in a stream reach.

### *Hydraulic Model*

The stream hydraulic model describes flow through a stream reach and provides spatially explicit information regarding stream depths and longitudinal, lateral, and vertical variations in velocity. Field derived inputs used to parameterize the hydraulic model include a Digital Elevation Model (DEM), stream substrate roughness estimates, and a discharge measurement which were collected using methods outlined in the Columbia Habitat Monitoring Program (CHaMP) protocol (Bouwes et al. 2011). Digital Elevation Models were generated from topographic surveys (x, y, z coordinates) of the streambed using Total Station surveying equipment. The topographic surveys also included the delineation of wetted channel unit boundaries based on the classifications of Hawkins et al. (1993). Within each channel unit, the proportion of substrate in each of six size classes ranging from fines (0 to 6mm) to boulders (250 to 4000 mm) was approximated based on ocular estimates. From these estimates, the midpoint of the dominant substrate class in each wetted channel unit was used as an effective roughness height, while roughness outside the wetted channel was set to 0.5m. Depth and velocity measurements were taken at a single cross section using a Global Water Flow Probe and used to calculate discharge.

Field derived measurements were used as inputs to the River2D and Streamtubes programs (Steffler et al. 2003) to facilitate flow modeling. The River 2D model is a depth averaged hydraulic model that uses topographic information (DEMs), roughness estimates, discharge, and water surface elevation to simulate depths and two-dimensional velocities in a modeled stream reach (Figure 1 Steps 1-3). Two-dimensional results from the River 2D model are then converted to 2.5 dimensions using the Streamtubes model. The Streamtubes model divides flows both laterally and vertically along evenly spaced cross sections (0.25m.,a reasonable foraging spacing for 150mm steelhead), creating a three-dimensional array of cells, each of which contain an equal portion of the total discharge (Figure 1 Steps 1-3). This step enables the creation of depth-dependent velocity differences that are used to calculate the downstream transport of invertebrate drift and the energetic swimming costs of drift-feeding salmonids.

### *Drift Transport Model*

The invertebrate drift transport model uses the three-dimensional cell arrays at each cross section generated from the Streamtubes model combined with field collected measures of invertebrate drift to model spatial variation in drift within the stream reach (Figure 1 Steps 4). Drifting invertebrates were collected at each site using methods described in the CHaMP protocol (Bouwes et al. 2011). These samples were sorted and weighed by Rhithron Associates Inc. and drift concentrations were summarized based on total biomass per volume of the sample. The drift transport model uses flow information from the hydraulic model, initial drift concentrations in the furthest upstream cross section, and field-measured settling velocities for each modeled insect taxa to determine concentrations in each subsequent downstream cross section cell. Drift concentrations in the first cross section were initialized by distributing densities of invertebrates evenly throughout all cells based on the individual weight of 3-6mm size class *Ephemeroptera* (mayflies), a common taxa found at all sample sites. Settling velocities of drifting *Ephemeroptera* were based on size-specific settling rates determined

experimentally in the field. To incorporate a mechanism of invertebrate entry downstream, drift concentrations were reset to match initial concentrations at modeled streambed velocities exceeding a 0.2 m/s threshold. From these inputs, the drift transport model predicts the lateral and vertical dispersion of invertebrate drift to determine the spatial variation of invertebrate drift density available to salmonids throughout the sample reach.

***Foraging Model***

The foraging model incorporates information derived from the hydraulic flow and drift transport models to calculate the gross rate of energy intake and the energetic costs of swimming to predict the net rate of energy intake for drift-feeding salmonids. This is accomplished by first establishing foraging volumes at each three-dimensional cell in the hydraulic model's cross-section arrays, which serve as foraging focal points. Foraging volumes are calculated using the foraging model of Hayes et al. (2007). This model assimilates velocity estimates and the reaction distance of fish in relation to fish and prey sizes to produce a foraging volume in which fish at each focal point would be expected to efficiently capture prey (Figure 1 Step 5). Based on foraging volumes at each focal cell, the foraging model then integrates results from the drift transport model along with prey capture rates and the energy content of prey to calculate the gross rate of energy intake at each focal point within the stream.

The foraging model uses estimates of species and size specific energy expenditure for a given velocity at each cell in an array to calculate the energetic cost of maintaining a constant position at a given temperature (Figure 1 Step 6). Parameters for 150mm steelhead, based on equations from Fish Bioenergetics 3.0 (Hansen 1997) and Railsback and Rose (1999) were combined with stream temperatures estimates and used as inputs to the model. The net rate of energy intake (NREI) is then calculated at user defined intervals across each cross section by subtracting the energetic cost of swimming from the gross rate of energy intake. NREI values are graphed to display the distribution of values within a stream reach where positive values represent favorable conditions at a given position in a stream (expected fish growth) and negative values represent unfavorable conditions (Figure 1 Steps 5-6). To estimate carrying capacity, the highest NREI value on each cross section is compared to a user-defined NREI threshold and locations meeting or exceeding the NREI threshold receive a fish. Fish are placed at upstream cross sections first and downstream drift predictions are then augmented to reflect consumption of drifting invertebrates by fish placed at upstream cross sections. Placement proceeds downstream until the last cross section has been evaluated for fish placement (Figure 1 Steps 7).

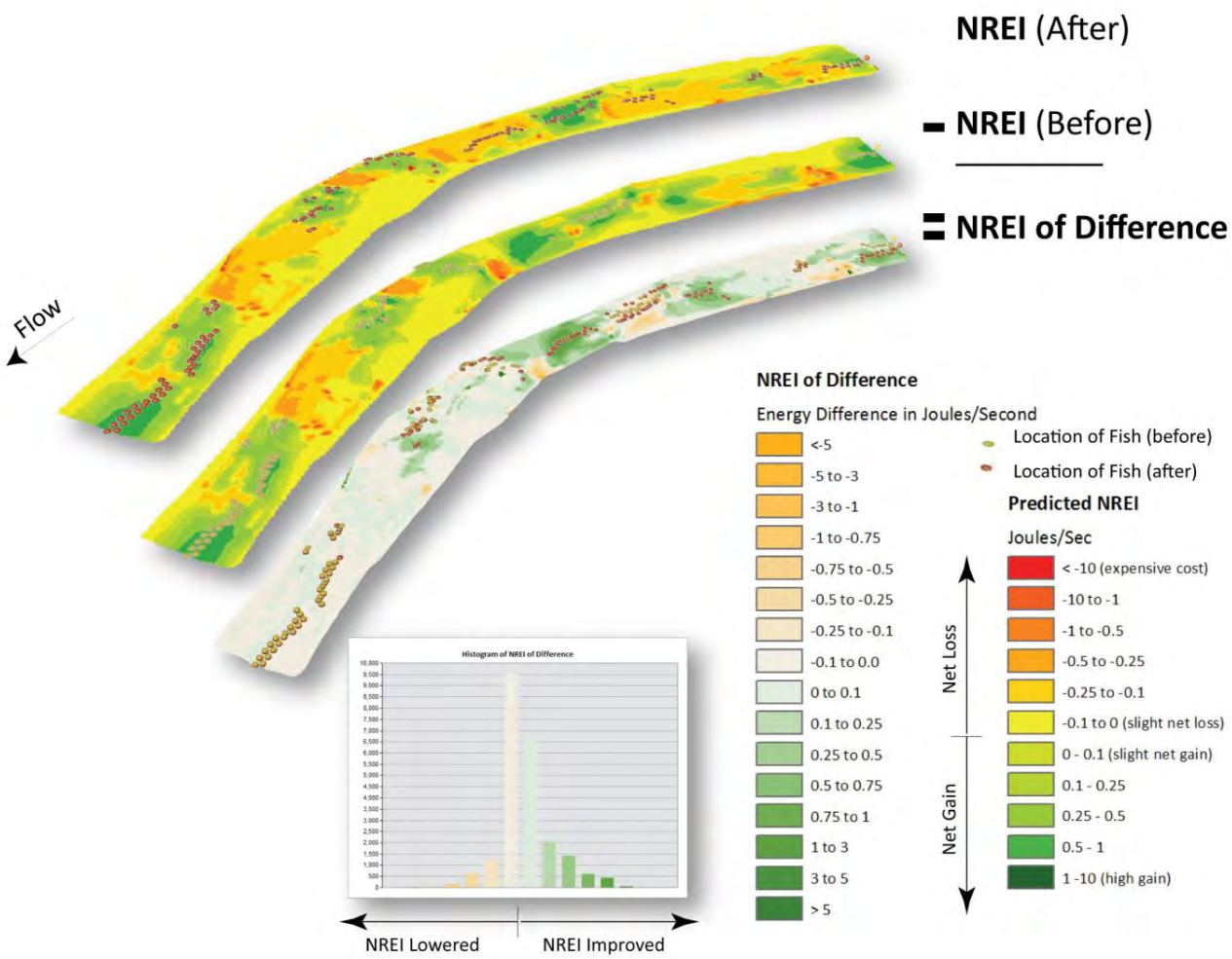


Figure 2. The energy available (net rate of energy intake or NREI; the colored surface) and abundance (dots represent placement of fish) pre-treatment (Before) and hypothetical post-treatment (creation of pools via wood additions) in a reach of the South Fork of the Asotin. If NREI (Before) is subtracted from NREI (After) for each pixel that has an XY coordinate, another surface is created that spatially describes the change in energy available and carrying capacity of the reach due to restoration.

The model can also be used to estimate how changes to stream channel can translate in to changes in NREI and carrying capacity much like the way DEMs can be used to evaluate changes in stream topography (see Chapter 7 Figure 8). We conducted a CHaMP survey at a site within the Asotin IMW and then altered the DEM to reflect the expected changes due to the proposed action of wood additions. We can subtract the pre-treatment NREI surface from the post-treatment surface to create an NREI difference surface that intuitively explains how the restoration could potentially create more fish (Figure 2).

### *Validation*

We also used CHaMP survey information in the John Day (seven sites) and Asotin (one site) to estimate NREI and carrying capacity. We compared the carrying capacity calculated from the model to observed fish numbers. Because steelhead are unlikely to be at carrying

capacity we expected a bias towards over prediction, but still a precise estimate of abundance. The model performed extraordinarily well, with predictions following the pattern expected (Figure 3).

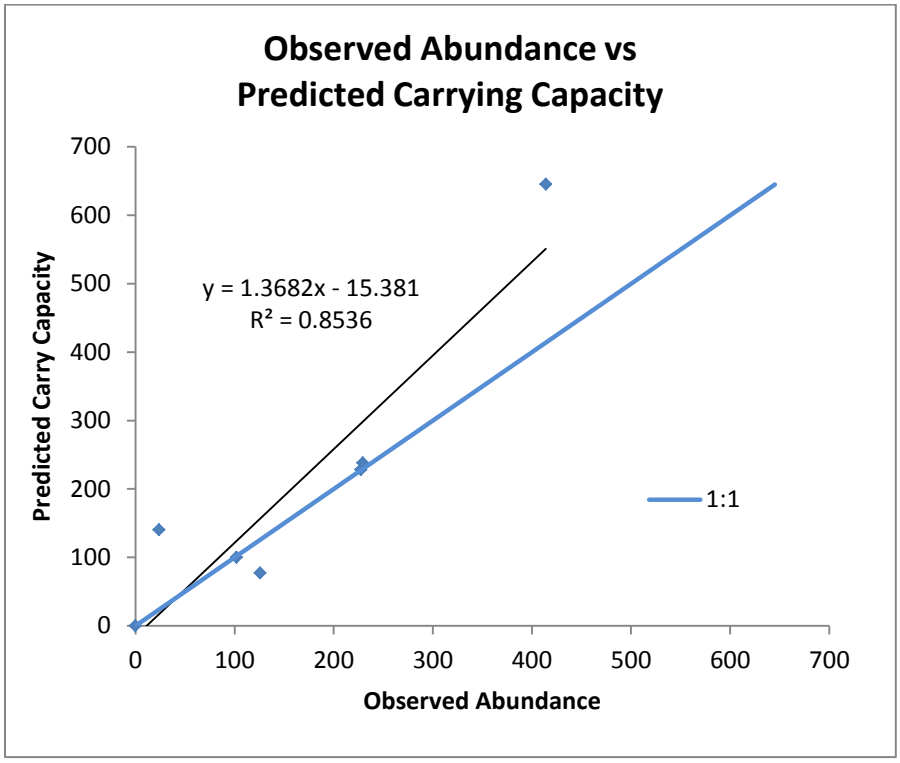


Figure 3: Observed versus predicted abundance of fish across reaches in 8 different streams . If we were able to predict actual abundance, the points would fall on the 1:1 line.

### *Discussion*

ISEMP has just recently begun to test this model to predict growth, abundance and production of a reach. The model has not been calibrated and several large simplifying assumptions were made to complete these analyses for this report. Still the model performed remarkably well, so we remain optimistic that further development will produce a product that synthesizes several metrics collected from CHaMP and describes what they mean to salmonids. The application of this approach can be many fold from evaluating limiting factors, assessing the benefits of stream restoration, and production of accurate information to be used in other analytical frameworks.

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