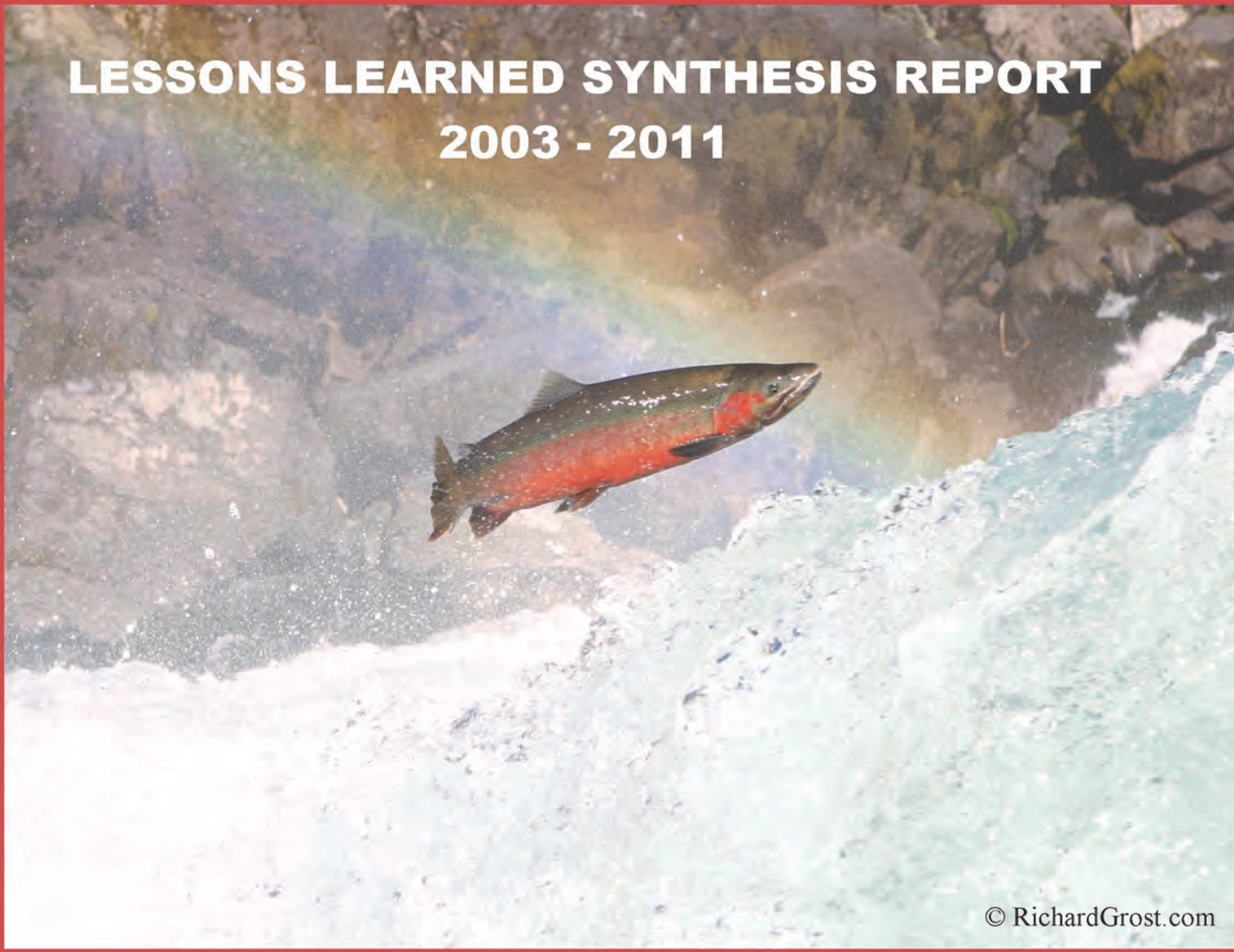
ISEMP

Integrated Status and Effectiveness Monitoring Program



July 6, 2012

Bonneville Power Administration
Project 2003-017-00

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This document is a summary of the lessons learned from work conducted by the Integrated Status and Effectiveness Monitoring Program (ISEMP) from 2003-2011 in the Columbia River Basin.

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LIST OF ACRONYMS

AMIP	Adaptive Management Implementation Plan
AREMP	Aquatic and Riparian Effectiveness Monitoring Program
BPA	Bonneville Power Administration
CHaMP	Columbia Habitat Monitoring Program
CRITFC	Columbia Inter-Tribal Fish Commission
DEM	Digital Elevation Model
DEQ	Department of Environmental Quality
DPS	Distinct Population Segment
EMAP	Environmental Monitoring and Assessment Program (USEPA)
ESU	Evolutionarily Significant Unit
GRTS	Generalized Random-Tessellation Stratified
IDFG	Idaho Department of Fish and Game
IMW	Intensively Monitored Watershed
ISW	Intensively Surveyed Watershed
ISEMP	Integrated Status and Effectiveness Monitoring Program
ISRP	Independent Science Review Panel
ISS	BPS's Idaho Supplementation Studies
LSRCP	USFWS's Lower Snake River Compensation Plan
MERR	Monitoring, Evaluation, Research, and Reporting plan (NPCC)
NOAA	National Oceanic and Atmospheric Administration
NWFSC	Northwest Fisheries Science Center (NOAA Fisheries)
NPCC	Northwest Power and Conservation Council
NPT	Nez Perce Tribe
NREI	Net Rate of Energy Intake
ODFW	Oregon Department of Fish and Wildlife
PNAMP	Pacific Northwest Aquatic Monitoring Partnership
QA	Quality Assurance
QC	Quality Control
RBT	River Bathymetry Toolkit
RST	Rotary Screw Trap
RMRS	USFS's Rocky Mountain Research Station
SRSRB	Snake River Salmon Recovery Board (Washington)
USBOR	US Bureau of Reclamation
USGS	US Geologic Society
USEPA	US Environmental Protection Agency
VSP	Viable Salmonid Population
WDFW	Washington Department of Fish and Wildlife
YN	Yakama Nation

EXECUTIVE SUMMARY: INTRODUCTION

The Integrated Status and Effectiveness Project (ISEMP) was created nearly 10 years ago to systematically answer questions such as "what is the best way to measure stream habitat?" and "what is the best way to measure salmonid populations?". These questions are related to the management that underpins the proposed tributary habitat-based, off-site mitigation strategy of the Federal Columbia River Power System Biological Opinion (FCRPS BiOp).

Quantifying the effect of habitat condition on fish populations is a required component of a population management strategy based on the conservation and rehabilitation of stream habitat. Linking fish population status and health to habitat condition can be done in two ways: measuring all aspects of stream habitat, habitat change and fish population condition at all possible locations, or developing key indicators of relevant habitat features and fish population responses in a spatially representative fashion to support a mechanistic, predictive framework. For reasons of efficiency, and to maximize the utility of the knowledge the project generates, ISEMP has adopted the latter tactic – to develop quantitative tools that relate habitat condition to fish populations in a framework that supports habitat and population management decision making.

Connecting habitat quality and quantity to fish population processes quantitatively allows the evaluation of habitat management actions for their potential impact on the abundance and productivity of listed salmonids in the Columbia River basin. The evaluation of management actions can be predictive, used in an adaptive management framework to forecast, plan and prioritize projects, and it can be extrapolative, used to quantify the potential impact of ongoing actions not included in an explicit evaluation or monitoring design. In either case, management decisions underlying the design and evaluation of the FCRPS BiOp habitat strategy need to be based on a documented, scientifically rigorous rule-set that links habitat condition with fish population response. In order to be most effective, the technical detail of how fish populations respond to habitat conditions must be translated into tools to support decision-making, interpretation by broad audiences, and use by technical and non-technical elements of the co-manager community. ISEMP's primary goal is to generate the decision support products that form the foundation of the FCRPS BiOp habitat strategy.

Research and monitoring conducted under the ISEMP project falls into three discrete, but related, categories:

Status and trends: monitoring data on fish and habitat to track and evaluate fish-habitat relationships at the Evolutionary Significant Unit (ESU), subbasin, and population levels.

Action Effectiveness: evaluating the effect of habitat actions (both project level, i.e., type of project, and watershed level, i.e., cumulative projects in a given area) on fish populations.

Analytical Framework: providing the context for monitoring data to address fish-habitat relationships, limiting factors, and whether management actions and restoration has led to changes in fish and their habitat.

ISEMP has developed fish and habitat status and trends monitoring efforts in the Wenatchee, John Day, South Fork Salmon and Lemhi subbasins (Figure ES1), and in 2011 initiated the Columbia Habitat Monitoring Program (CHaMP) to further develop standardized fish and habitat monitoring. As part of the ISEMP design for developing fish-habitat relationships, several forms of intensive monitoring have been employed: Intensively Monitored Watersheds (IMWs) that involve intensive monitoring and

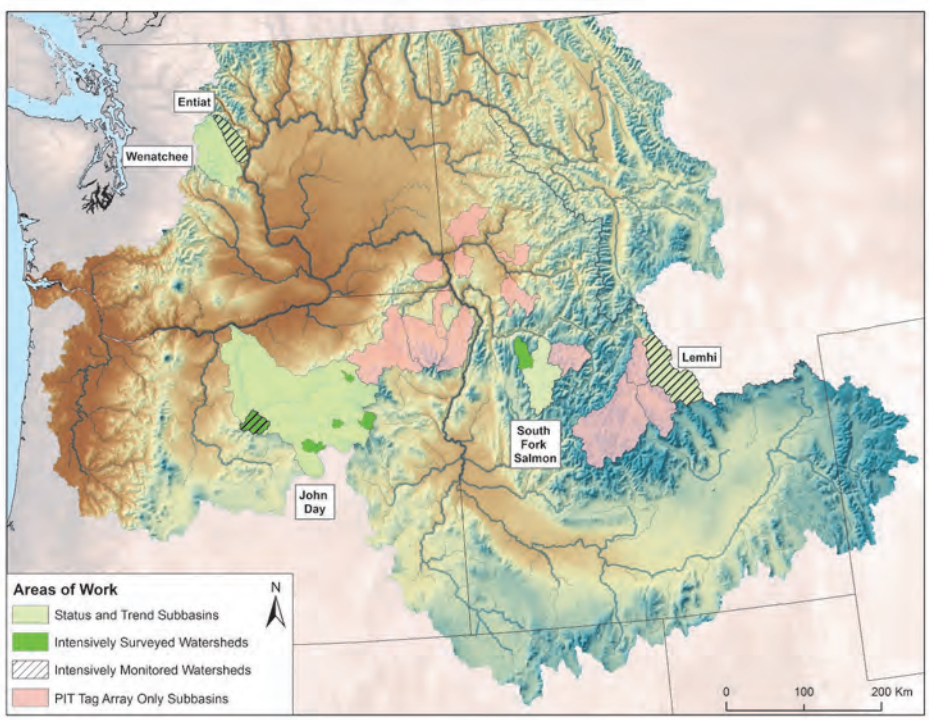


Figure ES1. Location of the ISEMP subbasins and associated Intensively Monitored Watersheds (IMWs) and Intensively Surveyed Watersheds (ISWs), and subbasins where monitoring is conducted using PIT tag detection arrays.

habitat manipulation, and Intensively Surveyed Watersheds (ISWs) that are reference areas for the habitat manipulation watersheds and metrics, indicator, and survey design development testbeds (Figure ES1). By coordinating fish data collection with habitat data collection from CHaMP, ISEMP is able to link habitat condition to fish populations, and ultimately, changing habitat conditions to change in fish population status. In addition, ISEMP is developing methods for adult and juvenile salmonid population estimation based on PIT tag detections – these efforts are underway in all of the ISEMP status and trends locations, as well as a suite of subbasins in the Snake River.

Implementation Timeline

ISEMP was initiated as a pilot project focused on monitoring program development in the Wenatchee River basin through the collection of stream habitat and juvenile salmonid population data (2004 – present). The project was given the additional responsibility of developing restoration project effectiveness monitoring and evaluation methods, which were first piloted in the Entiat River basin starting in 2006. Proposed work for the John Day and Salmon River basins was designed and reviewed (ISRP and others) during the initial phase of the project, with full implementation beginning in 2009 across these watersheds. In 2010, through the Fast-Track process, ISEMP was asked to develop a network of in-stream PIT tag detection arrays that linked the fish and habitat monitoring programs in key FCRPS BiOp population watersheds. In 2010 ISEMP also developed a stream habitat monitoring program, the Columbia Habitat Monitoring Program (CHaMP), which was initiated as a separate project (2011-006) in 10 watersheds during 2011. ISEMP currently implements three IMWs (Entiat, Bridge Creek, Lemhi), three population and habitat status and trends monitoring watersheds (Wenatchee, John Day and South Fork Salmon) and a network of approximately 50 in-stream PIT tag detection sites (Figure ES2).

Objectives, Priorities and Products

ISEMP has the overarching objective of developing management decision support tools from quantitative relationships of stream habitat quality and quantity's impact on anadromous salmonid population abundance and productivity in the Columbia River basin. ISEMP's data collection, management and analysis task and the resulting products can be organized into seven focal areas – three that are primarily data collection (ISEMP data streams) and four that are primarily data processing (ISEMP data management, monitoring guidance, and decision support tools).

Data streams

- Large-scale experimental manipulations of stream habitat condition to test mechanistic linkage between stream habitat and fish population processes.
- Broad-scale juvenile and adult fish population monitoring aligned with existing, ongoing habitat monitoring, to form the basis for extrapolating mechanistic fish-habitat relationships beyond experimental watersheds.
- Landscape context data collection and analysis to support extrapolating mechanistic fish-habitat relationships beyond experimental watersheds.

Data processing

- Survey and sampling designs to support population-scale inference of fish-habitat relationships.
- Data management to support population-scale inference of fish-habitat relationships.
- Spatio-temporal analysis of fish-habitat relationships to develop a quantitative rule set that links abundance and productivity to habitat quality and quantity.
- Watershed production model to evaluate the impact of management action scenarios for key populations and habitat action tactics.

Each of the ISEMP focal areas has a number of components, for example, each ISEMP watershed has independent fish and habitat data collection elements, and there are numerous ongoing fish-habitat relationship development elements. However, all of the tasks and their output products are coordinated to develop a key set of management support products. The timeline of product development is shown in Figure ES2 and Table ES1. This product list is made up primarily of short-term products that are scoped and will be completed in the next 3 years. Products are continuously generated to meet ISEMP and co-manager needs (e.g., steelhead redd study, PIT database). More detail on the products is provided in Chapter 11 of the Appendix.

How to Use This Report

This report focuses on the achievements of ISEMP since its inception in 2003 relevant to answering key management questions in both the science and policy arenas. We describe tools that allow the resource management community to address these questions using scientific raw materials to make decisions, and we present frameworks for thinking about and interpreting status, trends and effectiveness monitoring data, and mechanisms to support management action design and implementation. To help different audiences get the most relevant information efficiently we have broken the report into three components – an Executive Summary (a high-level overview, pp. xiv – xxxv), a series of Lessons Learned vignettes (take-home messages, pp. 1 – 96), and technical reports s chapters in the appendix (pp. 101 – 195).

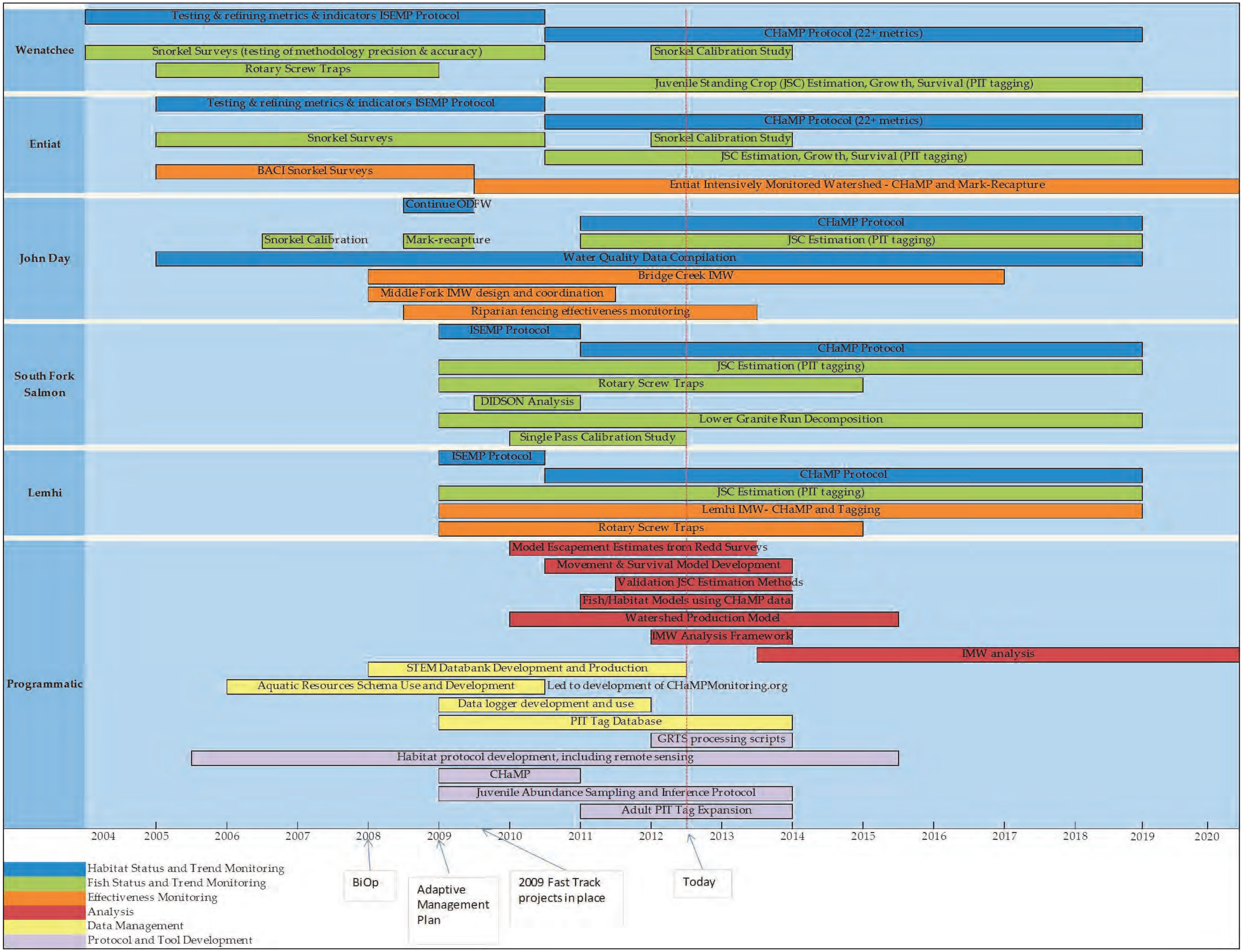


Figure ES2. The timeline of implementation for status and trend and effectiveness monitoring activities in the three pilot subbasins under ISEMP, with associated analysis, data management, and protocol and tool development.

Table ES1. A timeline showing products already completed and those scoped out for the future. Products represent those generated from the data streams from each of the three subbasins and the data processing that is associated with those data.

Data Stream/Data Processing	Product	Expected Date of Completion
Large-scale experimental manipulations of stream habitat condition to test mechanistic linkage between stream habitat and fish population processes	Effectiveness Monitoring of Riparian Fencing	June 2013
	Lemhi Intensively Monitored Watershed study	October 2017
	Bridge Creek Intensively Monitored Watershed study	December 2017
	Entiat River Intensively Monitored Watershed study	December 2020
Broad-scale juvenile and adult fish population monitoring aligned with ongoing habitat monitoring to form the basis for extrapolating mechanistic fish-habitat relationships beyond experimental watersheds	Estimating adult abundance from redds	Decemberr 2012
	Recommendations for steelhead redd surveys in the Wenatchee	*December 2012*
	Guidance on the use of steelhead index spawning ground counts.	*December 2012*
	Use of redd surveys manuscript	*June 2013*
	Recommendations for Chinook redd surveys in the Snake	*March 2014*
	Approaches to the uncertainty around downstream migrant trap data	December 2012
	Lower Granite Dam and IPTDS escapement estimates	Draft October 2010, final March 2013
	Steelhead life history patterns in the Wenatchee and Entiat River subbasins	June 2013
	Calibrating Snorkel Counts to Fish Abundance Estimates	November 2013
	Juvenile salmonid summer rearing (standing crop) population estimation	December 2013
	Juvenile Survival Models	December 2013
	Basin-wide estimates of juvenile abundance and juveniles per spawners	December 2013
	Precision, Bias, Reliability, and Cost-Comparison of RST Versus Instream PIT tag detection (IPTD) Based Juvenile Abundance and Survival	December 2014
Landscape context data collection and analysis to support extrapolating mechanistic fish-habitat relationships beyond experimental watersheds	Geomorphic framework (River Styles) for ISEMP watersheds	December 2014
	Integrate River Styles framework into the ISEMP Watershed Production model	June 2015
Survey and sampling designs to support population-scale inference of fish-habitat relationships	Determine strata for organizing standard stream habitat survey designs.	April 2010
	Design of observation error study.	June 2011
	Standardize fish sampling methods across ISEMP	June 2011
	Power analysis of habitat data.	April 2011
	PIT tag array detection analysis with DIDSON imaging sonar.	June 2011
	Evaluate channel unit fish sampling.	June 2012
	Set necessary sample size for calibrating single pass fish surveys.	June 2012
	Decision tree to determine fish sampling design.	December 2012
	Standardized GRTS processing scripts for fish and habitat data.	December 2013
Data management to support population-scale inference of fish-habitat relationships	Protocol documentation and requirements	January 2008
	STEM Databank	January 2008-present
	Measurement storage and retrieval	*January 2008*
	Metric storage and retrieval	*October 2012*
	Metadata linkage to MonitoringMethods.org	*December 2012*
	Metadata and metric linkage to ISEMP's fish database	*April 2013*
	Data distribution to DART	*August 2013*
	Metadata linkage to cbfish.org	*June 2013*
	Ongoing water quality compilation for John Day Basin	January 2009 & updates
	Data quality guidelines	May 2009
	Automated export of data between ISEMP and PTAGIS repositories	June 2009
	Aquatic Resources Schema (ARS)	September 2009
	IPTDS Data Management.	October 2009
	Uniform GIS layers compiled and clipped to PNW	January 2011, updated September 2012
	PITA inventory	February 2011, update September 2012
	IMW locations	June 2011, update September 2012
	Regional PIT Tag Data Queries.	January 2012
	Data Management lessons learned paper	July 2012
	Data flow paper	August 2012
	PIT tag Database.	December 2013
Spatio-temporal analysis of fish-habitat relationships to develop quantitative rule set that links abundance and productivity to habitat quality and quantity	Fish-habitat relationship modeling	December 2014
	Preliminary analysis on habitat data	*April 2010*
	Mechanistic models	*June 2013*
	Final analysis	*December 2014*
	Incorporate relationships in the watershed production model	*December 2014*
Watershed production models to evaluate the impact of management action scenarios for key populations and habitat action tactics	R-Code Watershed Model with Flexible Input and Parameterization.	December 2012
	Watershed Production Model for the Lemhi and Sesech Subbasins.	December 2012
	Watershed Production Model for Wenatchee and Entiat Subbasins.	July 2013
	Watershed Production Model for the John Day Subbasin.	December 2013
	Reduced Watershed Model	June 2015

EXECUTIVE SUMMARY: SAMPLING DESIGNS

Guidance on Sample Design and Sample Size for Habitat Status and Trends Monitoring

There are two central questions when designing a habitat or fish monitoring program is how much of the landscape must be sampled to accurately capture all of the natural variability and detect change due to natural or human factors, and what is the best way to sample in time and space to determine status and trends. Analysis of habitat data collected by ISEMP has yielded the following guidelines, which were incorporated into the design of CHaMP, a standardized fish habitat monitoring protocol implemented in 10 watersheds across the Columbia River Basin starting in 2011.

Sampling Design

- The GRTS (Generalized Random-Tessellation Stratified) algorithm incorporates randomization into selecting sampling locations and provides a representative sample of habitat conditions.
- A split rotating panel (a number of annual sites plus a number of sites sampled on a rotation, for example, every 5 years) best accomplishes the objectives of monitoring habitat status and trend. Status is best monitored by using as many sites as possible representing the broadest geographical distribution of metrics and indicators, and trends is best monitored by repeated sampling of the same sites over time.
- The ability to quantify and understand the components of variation is critical to knowing whether status and trend data are meaningful and usable by managers. Variation is introduced into status and trend monitoring data by:
  - ◊ Natural variation in fish and habitat conditions over space and time.
  - ◊ Measurement error among crews, and
  - ◊ By features of the landscape such as geomorphic valley classification.
- A split rotating panel design allows for sources of variation to be identified and quantified, allowing managers to assess whether the information contained in metrics and indicators are useful for describing fish-habitat relationships (Figure ES3).
- The power of a sampling design can be increased if it accounts for known sources of variation in the landscape by stratifying sites based on, for example, differences in geomorphology and elevation. Valley type, Strahler order, and ownership, were found to partition the variance of habitat metrics quite well.

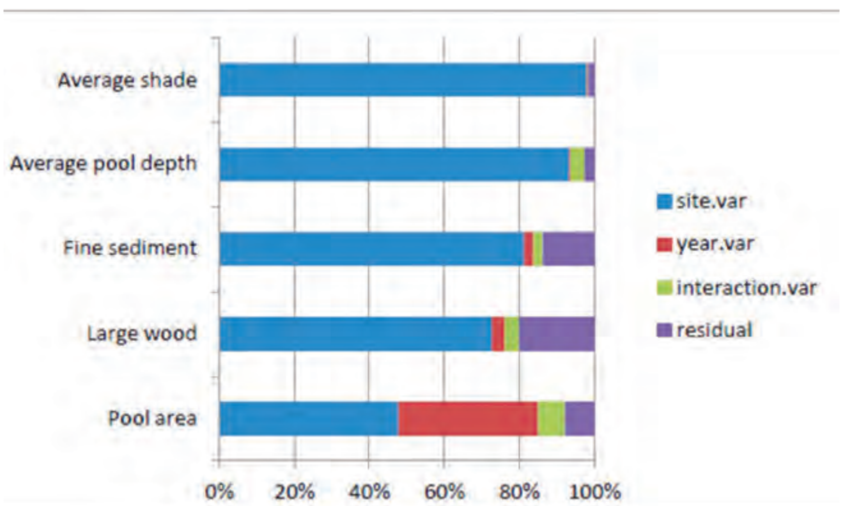


Figure ES3. The relative proportion of total variation that is attributable to site, year, the interaction between site and year, and residual variation for five habitat metrics collected in the Wenatchee 2004 – 2010.

Sample Size

- Knowing how many sites to visit, and establishing useful guidelines on how to choose sites can greatly improve the description of watershed habitat. More precise estimates of the habitat indicators important to fish will maximize the potential signal of fish – habitat relationships, making it easier to detect a signal and improve the accuracy of analyses.
- Analysis revealed that precision improved as sample size increased, but with diminishing returns (Figure ES4). Once the sample size grew larger than 45, there was very little improvement in precision. This pattern holds true for the mean value within a watershed, or for measures of variability across the watershed, such as standard deviation or coefficient of variation. It was also consistent across different habitat metrics.

These results were used to inform the survey design of CHaMP: 45 sites over 3 years per watershed (annual panel of 15 sites, 3 rotating panels of 10 sites sampled every 3 years) and stratified sites based on a geomorphological classification.

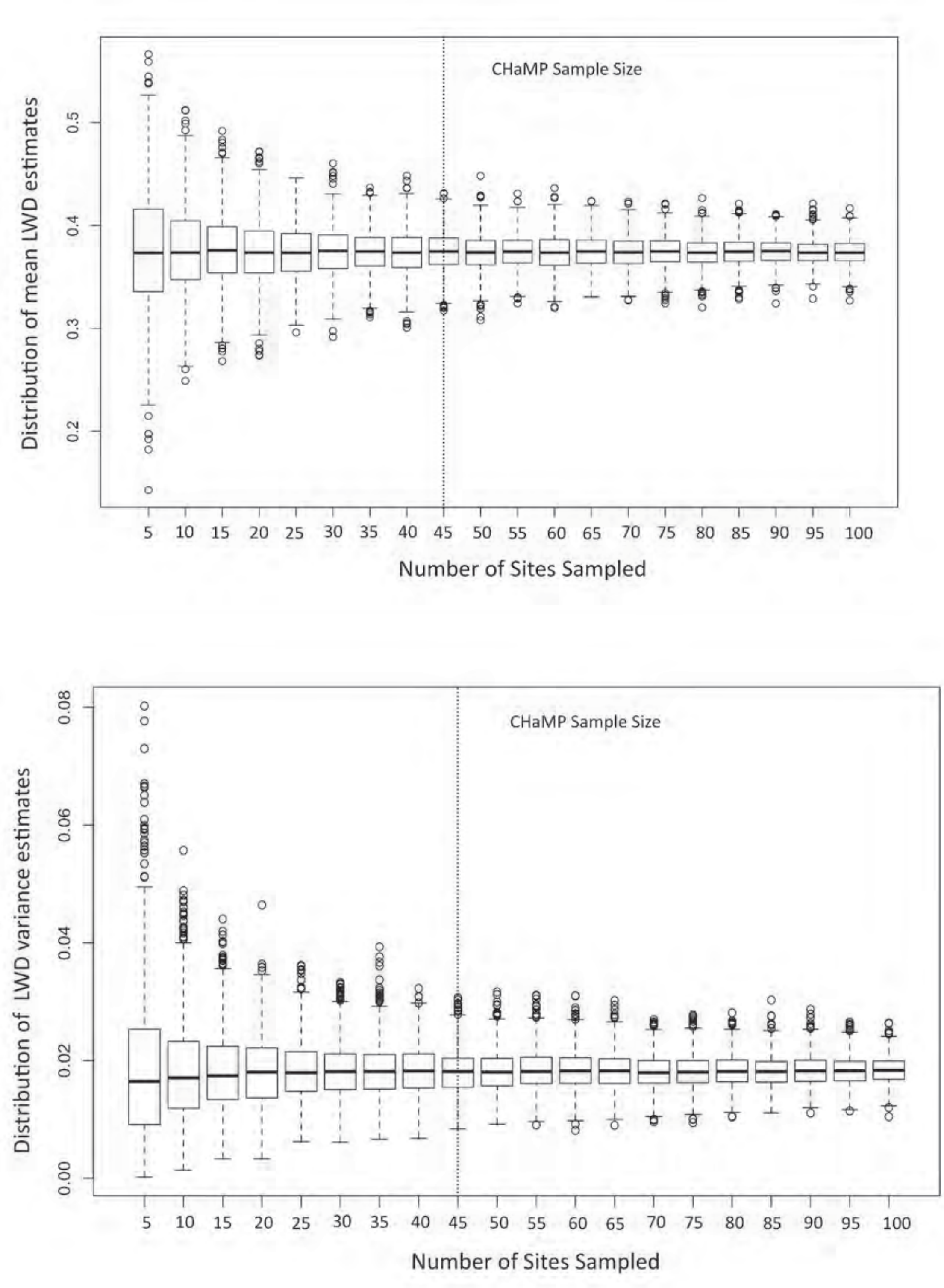


Figure ES4. Box plot displaying the distribution of mean estimates of log(X+1) transformed Large Woody Debris volume/stream km (top panel) and the distribution of variance estimates of log(X+1) transformed Large Woody Debris volume/stream km (bottom panel) for the Wenatchee subbasin based on varying site sample sizes (5-100 sites; X axis). The dashed line indicates the annual site sample size for CHaMP.

Developing Rules for the Inclusion of Metrics in Monitoring Protocols

To be useful to managers, current monitoring activities must be able to see differences in key habitat indictors 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 indicators that can consistently quantify spatial and temporal patterns that arise from natural variation and human impacts – only then are these indicators useful for management purposes.

Metrics and indicators are the units of information most useful and relevant to making inferences and decisions about the management of salmon habitat, and are the common language among data collectors, scientists, and natural resource decision makers, even those involved in different monitoring programs.

- **Metrics** are values resulting from the reduction or processing of measurements at a site or over a unit of time or space (i.e., metrics are site-scale values for the sampling period).
- An **indicator** is the value resulting from the processing of metrics across sites or across time and is a population-scale value for the sampling period.

ISEMP developed a rule set to evaluate which metrics and indicators should be included in a fish habitat monitoring protocol that is specifically designed to evaluate the features of stream habitat critical to juvenile salmonid growth and survival from egg to smolt life stages.

A measurement and related methodology was included in the CHaMP protocol if it would be used to calculate a metric that met each of the following three rules:

- **Information Content**: Habitat metrics and indicators must provide information directly related to salmonid productivity, including survival and growth, as documented by peer reviewed literature, modeling, or existing data analysis, or information that helps interpret processes by which fish habitat is influenced.
- **Data Form**: Habitat metrics and indicators must provide statistical information with robust data quality. The data generated for a prospective metric must be repeatable, detect heterogeneity, and have adequate properties for modeling/ statistics (e.g., variance distributions must meet statistical assumptions for modeling or testing).
- **Feasibility**: Habitat metrics and indicators need to be generated by field tools or software that are readily implementable as of the time field testing in fall 2010 (i.e., does not rely on future technological advances). Feasibility is also bounded by the need to fit all survey work within a three-person-day field survey at 80-90 percent of all sites likely to be encountered.

Below is a list of the indicators that ISEMP included in the CHaMP protocol.

- Average Alkalinity
- Average Conductivity
- Average pH
- Growth Potential
- Percent Below Summer Temperature Threshold
- Percent Above Winter Temperature Threshold
- Velocity Heterogeneity
- Embeddedness of Fast water Cobble
- Pool Frequency
- Channel Complexity
- Channel Score
- Residual Pool Volume
- Pool Tail Fines
- Total Drift Biomass
- Bank Angle
- LWD Volume
- Fish Cover
- Channel Unit Volume
- Channel Unit Complexity
- Riffle Particle Size (D_{16}, D_{50}, D_{84})
- Riparian Structure
- Solar Input

EXECUTIVE SUMMARY: STATUS AND TRENDS OF FISH POPULATIONS

Monitoring Adult Escapement

Indicators used to evaluate the effectiveness of actions proposed in the BiOp rely on estimates of adult escapement. In the Columbia River basin, redd counts are commonly used as an "index" of abundance, but are accompanied by substantial uncertainty and questions about the statistical reliability of many commonly used redd count protocols. There is a need to evaluate the precision and potential bias of redd counts as an index of escapement.

- ISEMP initiated a mark/recapture program that PIT tags a known, representative fraction of natural origin adult steelhead and spring/summer Chinook salmon as they pass Lower Granite Dam that are subsequently detected in upstream tributaries at instream PIT tag arrays. This allows us to generate estimates of adult escapement for steelhead and spring/summer Chinook salmon with accompanying estimates of uncertainty in the South Fork Salmon and Lemhi Rivers subbasins.

- The decomposition of the Lower Granite Dam runs-at-large of steelhead (Table ES2) and spring/summer Chinook salmon into tributary, population, and MPG specific escapement estimates is a reliable, precise and efficient alternative to continuous operation of multiple weirs.

- Adult capture and PIT tagging at Lower Granite Dam has not been accompanied by any direct mortality to date, suggesting that handling stress may be minimal at this location relative to upstream weirs.

- There is the potential to expand PIT

Table ES2. Steelhead run year, Major Population Group (MPG), population, subpopulation fraction of population sampled, escapement estimate, coefficient of variation (CV), and independent estimate (if available) monitored by ISEMP PIT tag arrays in the Snake River Basin. Shaded rows identify opportunistic independent estimates of escapement, primarily comprising locations where PIT tag wands are utilized to interrogate PIT tags.

Run-Year	MPG	Population	Subpopulation	PIT Tag Decomposition: Fraction Sampled[1]	PIT Tag Decomposition: Escapement	PIT Tag Decomposition: CV	PIT Tag Decomposition: 95% CI	Independent Estimate: Escapement	Independent Estimate: CV	Independent Estimate: 95% CI
2009-2010	Lower Salmon	Asotin Creek[2]		95%	1,687	8.5%	± 280	1,500	N/A	N/A
2009-2010	Salmon River	South Fork		90%	1,497	9.1%	± 268			
2009-2010		Secesh		100%	298	22.1%	± 129			
2009-2010		Middle Fork	Big Creek	100%	753	21.8%	± 322			
2009-2010		Upper Salmon	Valley Creek	100%	237	17.7%	± 82			
2009-2010		Upper Salmon	Pahsimeroi River[2]	100%	138	22.9%	± 62	115	N/A	N/A
2009-2010		Lemhi River		95%	630	14.2%	± 175			
2009-2010		Little Salmon	Rapid River[2]	95%	136	24.0%	± 64	150	Census	Census
2009-2010	Clearwater River	Lochsa River	Fish Creek[2]	100%	246	58.1%	± 117	205		164-255
2010-2011	Lower Salmon	Asotin Creek[2]		95%	890	10.0%	± 175	1,128	2.0%	± 44
2010-2011	Grande Ronde	Joseph Creek[3]		100%	1,627	1.4%	± 45	1,698	22.4%	± 744
2010-2011	Imnaha River	Imnaha River		100%	3,298	1.5%	± 97			
2010-2011		Imnaha River	Cow Creek	100%	147	1.4%	± 4			
2010-2011		Imnaha River	Big Sheep Creek	100%	765	2.2%	± 33			
2010-2011	Salmon River	South Fork		90%	2,540	1.9%	± 93			
2010-2011		Secesh		100%	397	3.1%	± 24			
2010-2011		Middle Fork	Big Creek	100%	687	1.6%	± 22			
2010-2011		Upper Salmon	Valley Creek	100%	232	1.5%	± 7			
2010-2011		Lemhi River		95%	428	1.7%	± 14			

[1]Fraction sampled refers to the fraction of spawning believed to occur above PIT tag arrays.
[2]Weirs that capture and enumerate steelhead and use handheld wands to identify PIT tags, but do not have PIT tag arrays.
[3]Locations with weirs that capture and enumerate steelhead and use handheld wands to identify PIT tags and also have neighboring PIT tag arrays.
[4]Independent estimate generated from a video weir paired with a single PIT tag array.

tagging at Lower Granite Dam to include hatchery origin adults, which would enable estimates of hatchery fraction in populations targeted for supplementation and enable estimates of stray rates into non-target populations that are monitored by PIT tag arrays.

- ISEMP is working with co-managers in the Upper Columbia to test the run decomposition approach, starting in 2012. Instream PIT tag detection arrays in the Wenatchee River and Entiat River subbasins will provide PIT tag data with which to test the escapement estimates into each tributary.

ISEMP is also working with co-managers in the Upper Columbia to estimate steelhead redd survey observer efficiency, identify and quantify sources of error that affect the uncertainty, and to standardize and improve steelhead redd surveys.

- The correct identification of steelhead redds in the Wenatchee subbasin was higher in the tributaries of the Wenatchee River than the mainstem itself, possibly related to the attributes of the tributaries (e.g., redd density, stream depth and width, and channel complexity).

- The suite of factors most important in predicting the proportion of redds correctly identified included experience on a specific reach, water clarity, density of redds, channel complexity, discharge, stream depth and stream width (Figure ES5).

- A similar analysis is taking place to explain the number of redds falsely identified. When combined with estimates of observer efficiency, this will lead to an estimate of the total number of redds throughout a season, with appropriate uncertainty bounds.

- In FY2012, ISEMP will collaborate with ODFW to use adult escape estimates to calculate freshwater production (i.e., juveniles per adult), spawner per spawner, and other potential metrics. This information will be combined with habitat information to develop spawner/habitat relationships, and the watershed production model will be used as a framework to synthesize life-cycle information.

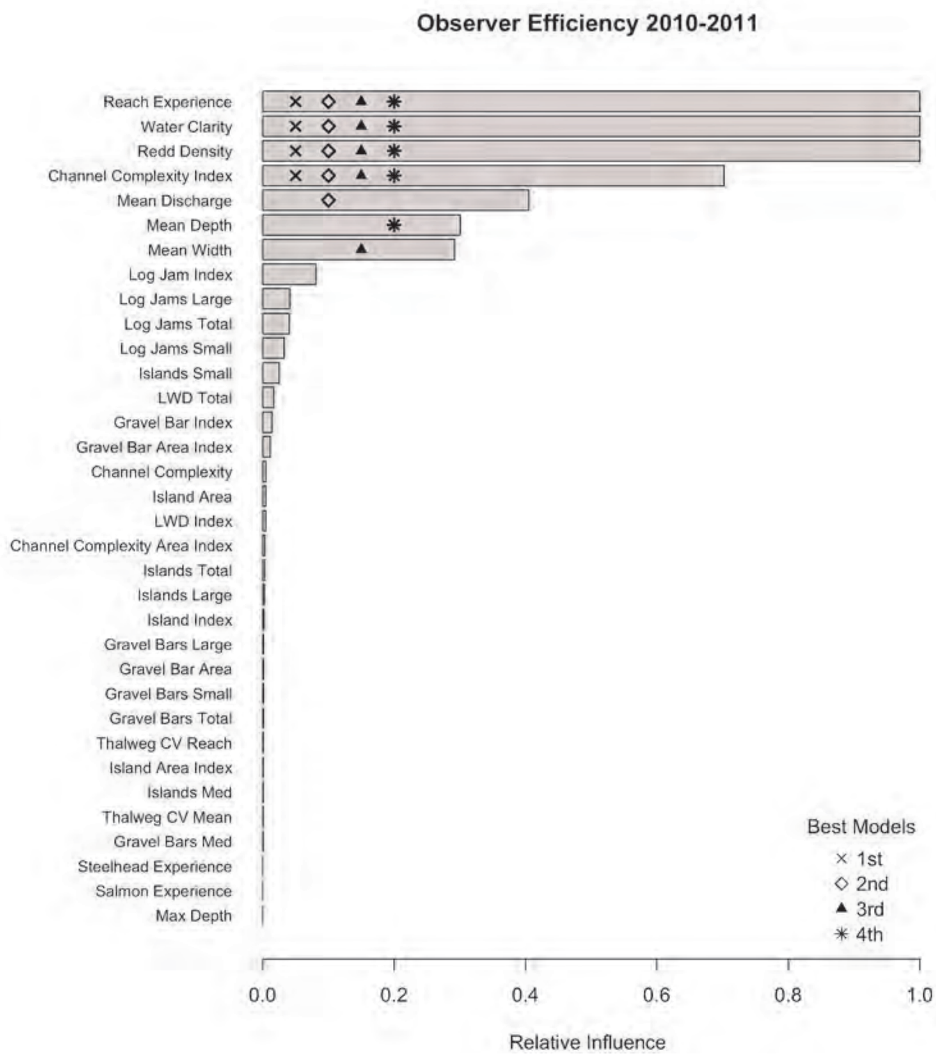


Figure ES5. The relative influence of each possible variable in predicting the proportion of visible redds that were observed. The predictors that were included in each of the four best models are marked.

Monitoring Juvenile Salmonid Standing Crop and Emigration

The BiOp identifies habitat restoration as a mitigation action to offset mortality imposed by the hydrosystem. Survival improvements accruing from habitat restoration actions are measured as improvements in egg to smolt survival but there remains a paucity of information directed at estimating the distribution, abundance, and survival of juvenile anadromous salmonids prior to their emigration to the hydrosystem. Infrastructure such as rotary screw traps generate an estimate of juveniles that survived to emigrate, but yield no information on these metrics prior to emigration or which (if any) habitat restoration actions contributed to the production of emigrants.

ISEMP is testing multiple survey types to estimate the standing crop and total emigrants of juvenile salmonids. Screw traps are used to generate estimates of total migration by life-stage from tributaries and populations, and field crews sample specific sites to generate site, tributary or population scale standing crop (total number of fish in a watershed at a specific time) of juveniles.

Standing Crop

- Employing a probabilistically based juvenile sampling effort utilizing GRTS allows for site-specific abundance estimates to be aggregated to estimate abundance at any spatial scale included in the GRTS sample frame, up to and including populations and MPGs (Figure ES6).

- Employing a standardized fish sampling protocol to capture and PIT tag juvenile anadromous salmonids ensures data can be "rolled up" across multiple spatial scales, for example, across watersheds and subbasins.

- PIT tagging, as opposed to direct observation surveys such as snorkeling, means survival and growth estimates can be generated from PIT tagged juveniles and can be used to represent the population at multiple spatial scales.

- Utilizing the same GRTS design for both juvenile sampling and CHaMP habitat sampling supports the development of relationships between fish and habitat attributes. These relationships allow the identification of habitat features that are conducive to fish, enabling an assessment of the realized and anticipated effectiveness of habitat actions.

- Site-based abundance estimates can enable an evaluation of where juveniles reside within a watershed relative to habitat restoration actions.

- When fish have been representatively tagged within these units of interest, survival of tagged juveniles indicates whether life-stage specific survival is improved in areas that have been restored.

- Abundance, survival, and growth estimates enable strong statements about the overall effectiveness of habitat restoration actions to increase fish production as opposed to a simple redistribution of fish to restored habitat.

Survey Types

Many different survey types (e.g., snorkeling, electrofishing, e-herding, snerding, seining) are used across the Columbia River Basin, and survey types can change even within a monitoring program as program objectives change.

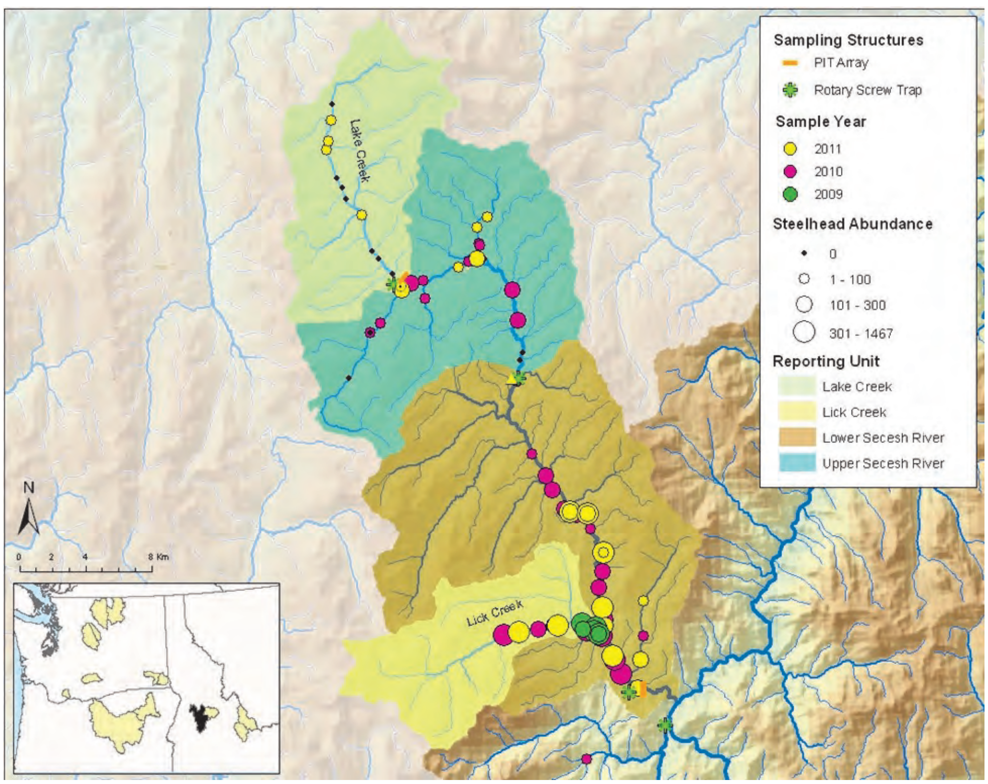


Figure ES6. Location of juvenile sampling infrastructure and distribution and abundance of juvenile steelhead obtained via remote juvenile surveys in the Secesh River. Note that reporting units identify spawning habitat (upper Secesh River, Lake Creek, and Lick Creek) and habitat used primarily for rearing or serving as a migration corridor (lower Secesh River).

ISEMP has implemented side-by-side comparisons of various juvenile fish monitoring survey types to develop cross-walks between methodologies. This is key for translating past time series of relative fish abundance to current fish monitoring protocols used by ISEMP and co-managers, and ensures that long-term datasets are not lost as survey types change or vary across the basin.

- Comparing the detection efficiency of snorkel surveys (Figure ES7) and single-pass electrofishing (Figure ES8) with a mark-recapture effort revealed a strong significant relationship between snorkel estimates and mark-recapture estimates.
- Snorkel survey efficiency was low, with about 12% of the total abundance observed by snorkelers.
- Snorkel surveys are precise, but not accurate, that is, a consistent bias is observed and reflected in the low detection efficiency of snorkel surveys. However, because this bias is consistent, these abundance estimates can be corrected for through the cross-walk relationships developed by ISEMP.
- A similar relationship was observed between mark-recapture and one-pass electrofishing.

Emigration Estimates

Rotary screw traps are used throughout the Columbia Basin to estimate total out-migration (emigration) of juvenile Chinook salmon and steelhead from a tributary. This information is used to estimate total juvenile production from a tributary or population, smolt-to-adult return rates, egg-to-smolt survival, and to study life history characteristics. The temporal and spatial extents of the estimate are usually dependent on logistics associated with access and environmental conditions such as high flows and ice. Traps are usually place in locations that are accessible for field crews and sometimes do not estimate the total population or sub-population of interest. Although the goal is to collect all juvenile migrants across the year, the operation of screw traps are often interrupted during the winter when the rivers freeze, high flows when it is too dangerous to operate, and by budgetary constraints when only a portion of the week is sampled. Because of these logistical constraints, the percentage of a population or tributary juvenile emigration estimated will vary between years.

The emigration estimate is derived by

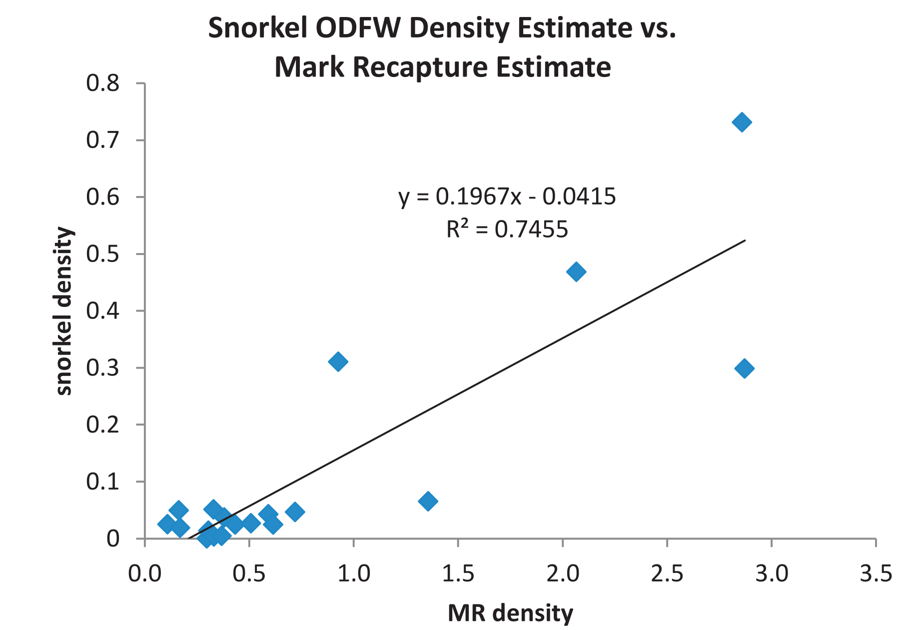


Figure ES7. The number of juvenile salmonids over an 100 m reach (expressed as no./m^2) based on mark-recapture methods were compared to the number observed snorkeling pool habitat.

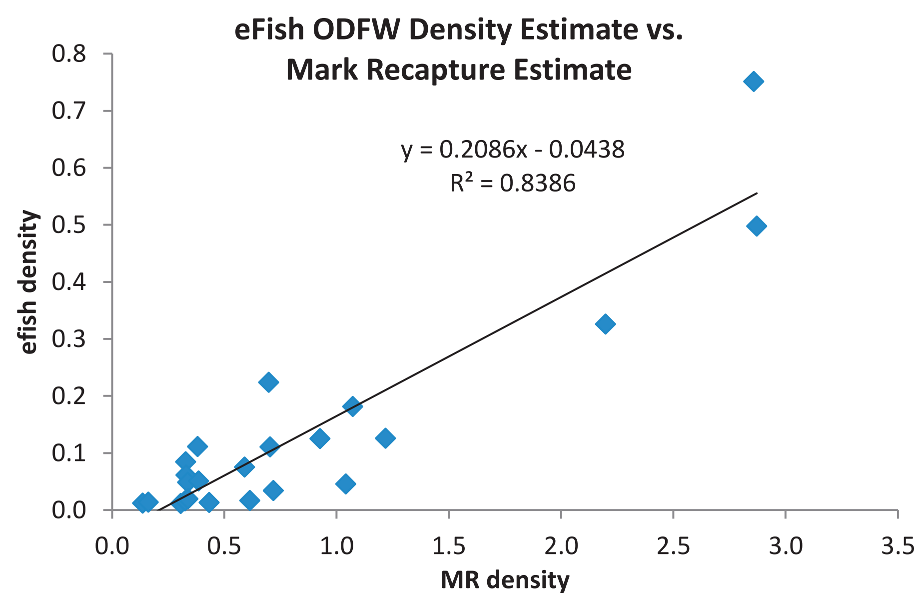


Figure ES8. The number of juvenile salmonids over an 100m reach (expressed as no./m^2) based on mark-recapture methods were compared to the number observed electroshocking pool habitat..

expanding the number of fish captured during a day by the trap efficiency, which is calculated by releasing a known number of fish upstream of the trap and calculating the "efficiency" of subsequent collection of these fish that pass the trap. However, screw trap estimates are often fraught with high levels of estimation error (e.g., trap efficiencies vary daily and seasonally in response to many factors, especially stream flow), this error is often not well reported, and managers may not realize the level of imprecision in these estimates. Additionally, methods to reduce the error in estimation can be expensive and ineffective.

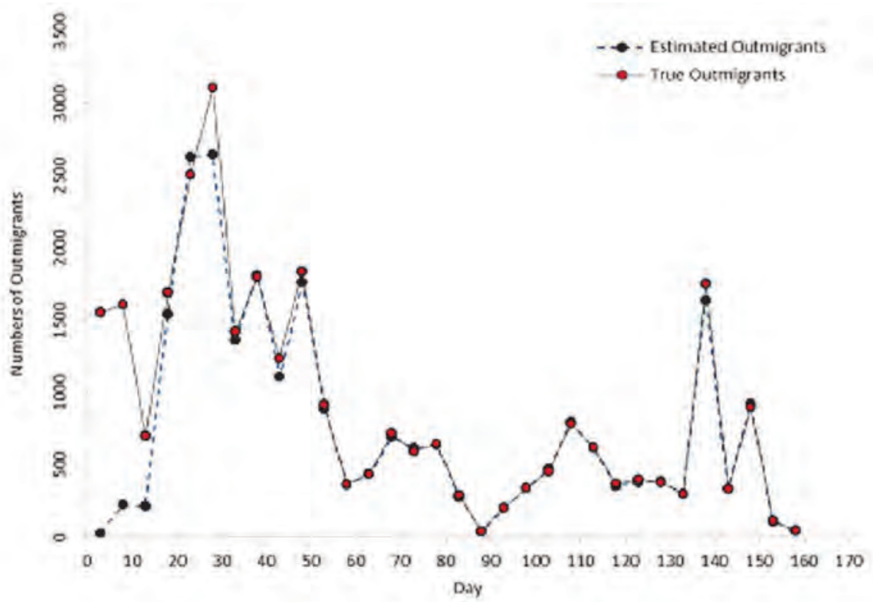
ISEMP has been conducting a series of investigations to highlight the importance of these generally overlooked weaknesses and to suggest improvements that will reduce sampling costs while improving the value of these critical estimates of fish emigration.

- Results suggest that estimating downstream migrant abundance using screw traps and mark-recapture methods *can* provide accurate estimates of abundance, but generic sampling designs for allocating mark-recapture effort (timing and amount) should be used with caution.

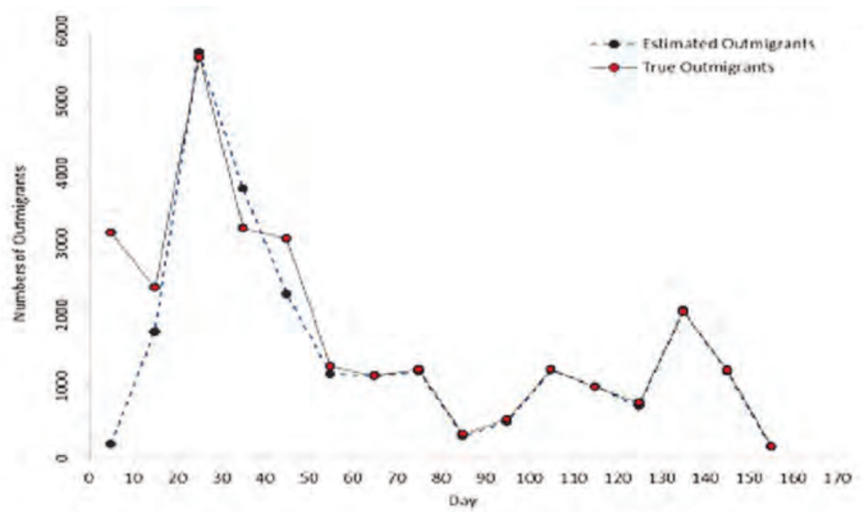
- Abundance estimates can be significantly biased as a result of violations in mark-recapture assumptions when these assumptions are not addressed or when violations of assumptions go undetected.

- Allocating more effort to trap efficiency trials, either by conducting more trials or supplementing the numbers of fish used in these trials, may not reduce the bias in abundance estimates (Figure ES9).

- For trapping environments that have significant variability in flows or trap efficiencies over the trapping season, allocating mark-recapture effort disproportionally through the season can address sampling challenges with least cost. Proportional allocation of effort may not be sufficient effort for challenging portions of the migrant run (high flows, low trap efficiencies and high number of migrants) and may be expending too much effort during times of stable flows and high trap efficiencies.



% Bias	Coverage
-15.0	35.5



% Bias	Coverage
-14.0	57.4

Figure ES9. Scenarios showing the effect of two trap efficiency tests that differ in the length of strata and number of trap efficiency estimates used for emigrant abundance estimations. Top panel shows shorter strata length (5 days) and more trap efficiency estimates (32) compared with lower panel that shows longer strata length and low effort (10 days/16 efficiency estimates).

EXECUTIVE SUMMARY: EFFECTIVENESS MONITORING

Intensively Monitored Watersheds:: Large-Scale Restoration Experiments

Intensively Monitored Watersheds (IMWs) are designed using principles of ecosystem-scale experiments, and are a powerful approach for detecting a population or environmental response to management actions.

- ISEMP has implemented IMWs to conduct restoration effectiveness monitoring in an experimental framework to demonstrate the utility of such designs, and because they are the mostly likely way we will be able to observe **population-level** benefits.

- ISEMP is using the IMW approach in three watersheds to test the effectiveness of the restoration at improving fish habitat and increasing productivity of salmon and steelhead.

- Some results are available for habitat responses to restoration actions (Bridge Creek) but it is too soon to report fish population-level responses.

Lemhi IMW

- Freshwater productivity in the Lemhi River watershed is thought to be limited by the availability of high quality juvenile rearing habitat.

- The primary goal is to test the effectiveness of reconnecting numerous small tributaries to the mainstem Lemhi River, and evaluate channel modifications, riparian fencing, diversion removals and screening, and side-channel development.

- The Lemhi IMW is being implemented in a staircase design, where connection of high priority watersheds occurred first, with order of subsequent reconnections depending on results of the prior treatment.

- ISEMP is using the Watershed Production Model to provide a landscape and life-cycle context, and synthesize how restoration is expected to result in tributary and/or reach-scale alterations and changes in Chinook and steelhead vital rates.

Bridge Creek IMW

- 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 leads to reduced base flows, increased summer stream temperatures, and a loss of spawning and rearing habitat.

- A series of instream beaver dams support structures (vertical wood post driven into the stream bottom) designed to assist beaver in the construction of stable, longer lasting dams are aimed at causing aggradation of the incised stream trench to restore floodplain connectivity.

- The primary change detection metric to describe the ability of beaver dams to restore floodplain connectivity is the aggradation (or deposition) rate. We are documenting aggradation by creating digital 3D maps of the channel (DEMs). The DEM of Difference is the change in bed elevation before and after implementation of restoration actions. A hierarchical-staircase statistical design is being implemented to compare treatment and control sections.

- One year after installation of the support structures (2009) 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 (Figure ES10).

Entiat IMW

- Primary restoration action to be tested is active instream modifications via engineered structures that increase habitat complexity and diversity by creating large pools and off-channel areas.

- A hierarchical-staircase statistical design is being implemented to compare treatment and control sections within the Entiat River. The lower 26 miles of the Entiat mainstem is divided into geomorphic reaches that can be treated in a spatially and temporally driven manner. Treatment and control sections will be represented in each geomorphic reach type, and each geomorphic reach will be implemented in staggered manner through time.

- First round of habitat restoration actions to be implemented in 2012. Pre-project implementation monitoring ongoing since 2010.

- Concurrent estimates of population size and individual growth and movement for Chinook and steelhead at the reach scale is ongoing to complement the population-scale effectiveness monitoring.

  ◊ Fish were enumerated at treated (a series of four engineered log jams and five rock barbs which have formed pools within the treated reach) and untreated reaches to determine if 1) fish growth and movement would show density

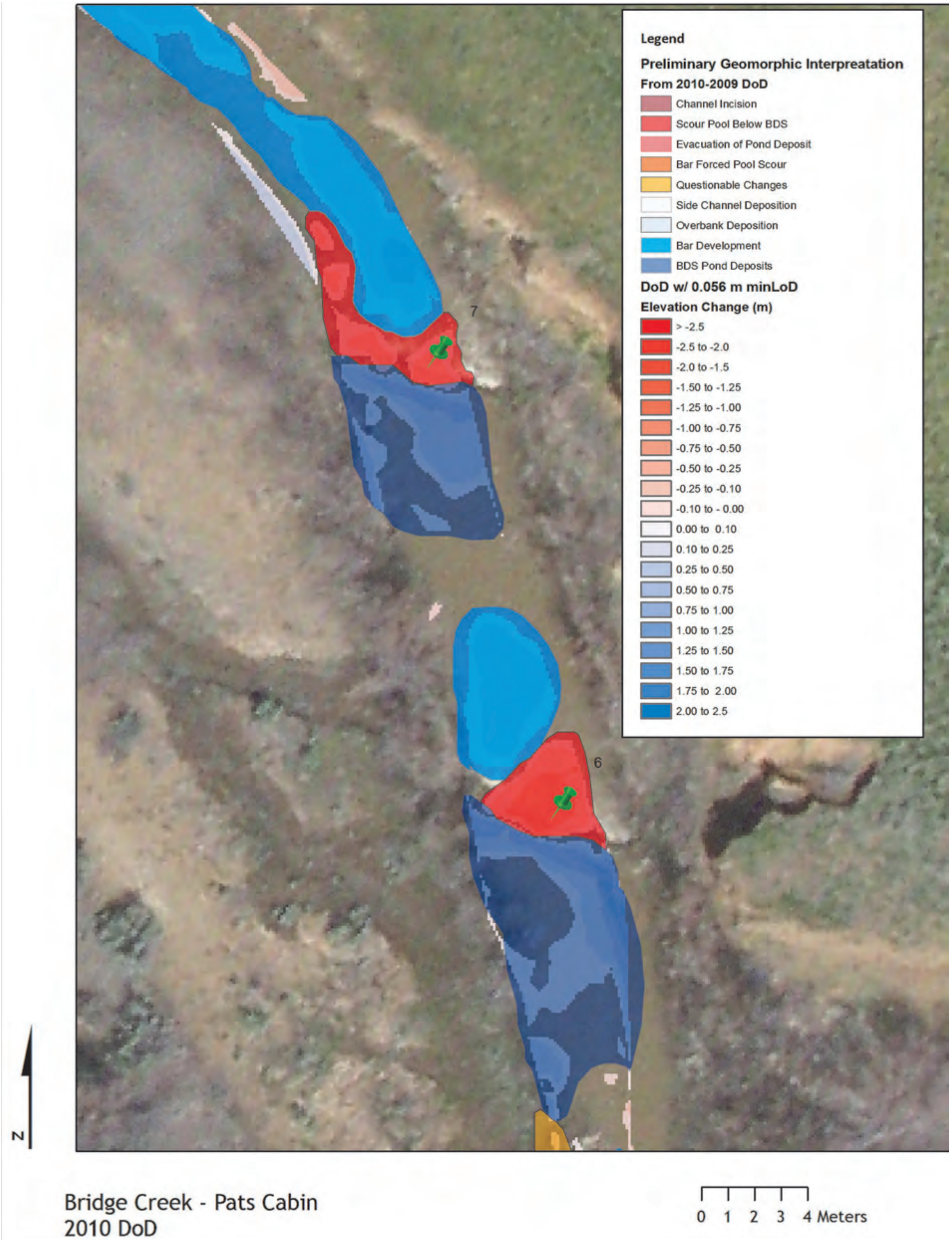


Figure ES10. DEM of difference (post-restoration minus pre-restoration) from topographic surveys for a portion of treatment reach in Bridge Creek. Pushpins represent structure location. Stream flow is from top to bottom. Blue color represents aggradation (deposition of sediments), and red represents erosion. General pattern was to have deposition behind structures, scour pool below structures, and deposition of the scour downstream from the pools.

dependence and 2) density dependence would differ between the treated and control reaches.

◊ Greater densities of juveniles were seen at treated sites but timing of sampling is important: both Chinook and steelhead had higher median density at structure pools during early summer, but later in the summer fish density was not strongly associated with structure pools (Figure ES11). This likely reflects the sub-yearling Chinook parr migration toward over-wintering habitat downstream and overall highly variable habitat selection patterns by steelhead.

◊ The elevated density of juvenile Chinook in treated microhabitats appears to be associated with a strong response to the increased water depth created by structures.

◊ Both Chinook and steelhead exhibited habitat affinity with pools treated with structures compared with untreated microhabitats (Figure ES12).

◊ Steelhead, despite being at lower density at structures than at untreated sites, had higher growth rates at structures, suggesting that density might not be the only indicator of fish response to restoration.

The Importance of a Study Design Pre-Implementation

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. Over the past two decades BPA has funded ODFW to build 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.

Changes to the riparian corridor and stream channel after exclosures are built can take decades or more to occur, whereas deciding whether to continue with this approach in order to provide necessary benefits to endangered populations is an immediate need. No pre-

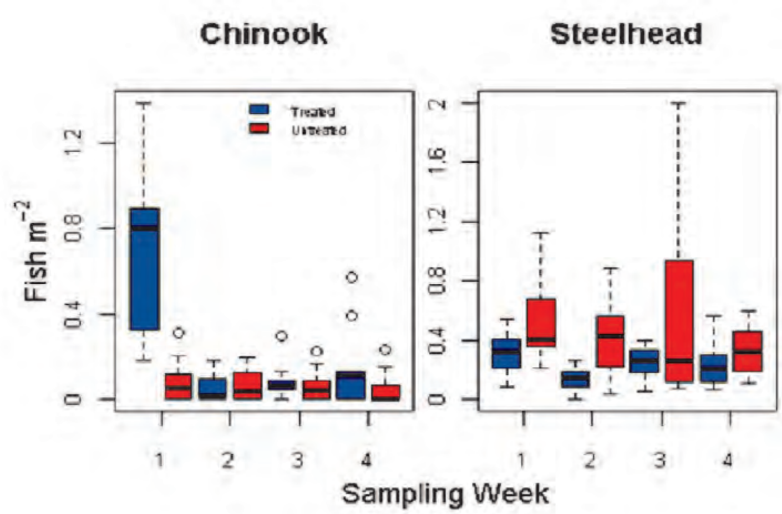


Figure ES11. Density of juvenile Chinook and steelhead in treated and untreated microhabitats in the Entiat River, August—September 2010.

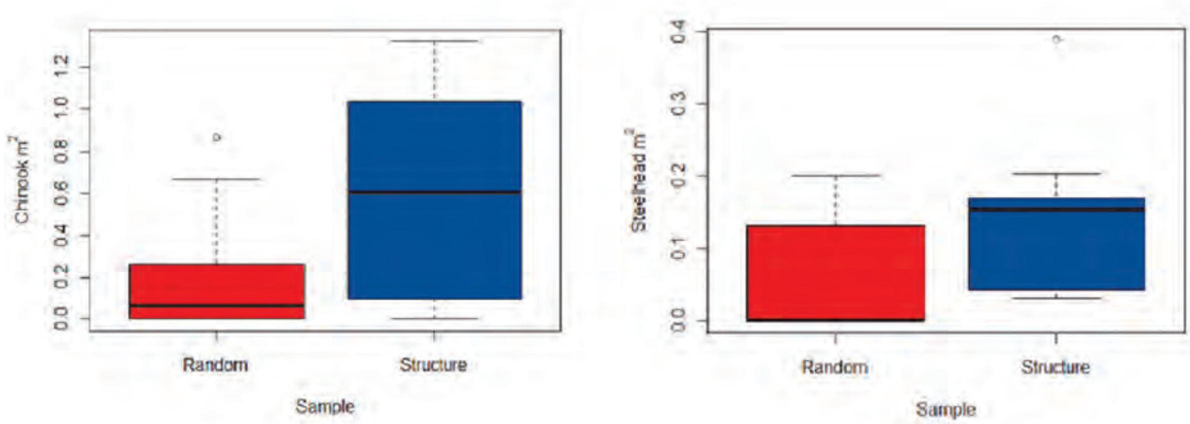


Figure ES12. Early summer density of juveniles Chinook (left panel) and steelhead (right panel) within a treated reach, either at a structure (blue box) or in randomly selected habitats within the same reach (red box).

project monitoring was implemented so a post-hoc study design was necessitated.

ISEMP conducted a two year study assessing whether grazing exclosures resulted in altered channel morphology and improved habitat conditions for a subset of streams in the John Day watershed 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.

- We were able to detect changes to the riparian area due to exclosures (Figure ES13) we were unable to detect associated responses to fish habitat (e.g., Figure ES14).
- From these results, we cannot infer whether grazing exclosures have elicited channel recovery or subsequent fish responses to grazing impacts in this basin.
- Explanations for the lack of response may include, but are not limited to:
    - ◊ 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. A post-hoc study design is not likely to be powerful enough to detect differences if they really do exist.

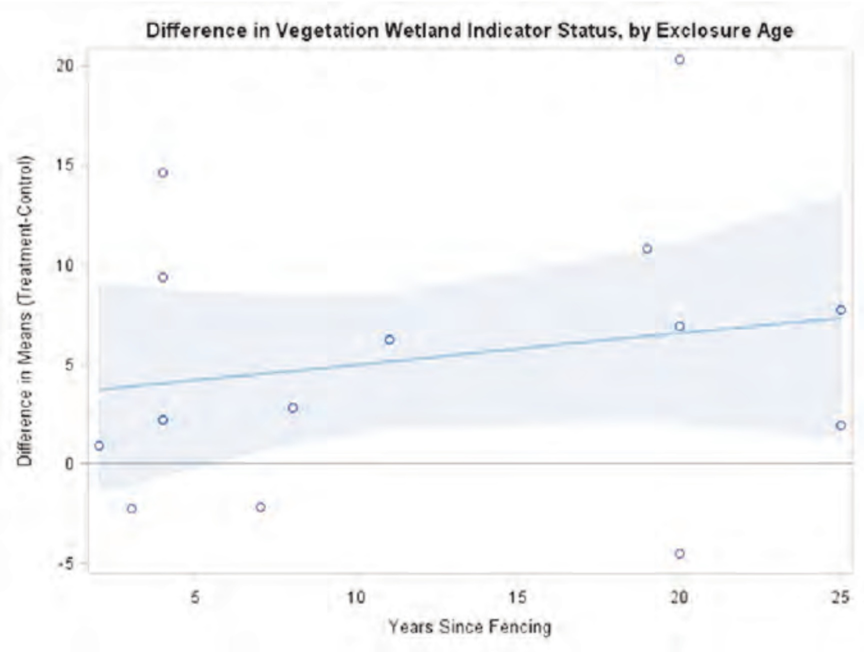


Figure ES13: Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, in wetland indicator values for the greenline plant communities. Statistically different values observed at exclosure sites > 6 years old.

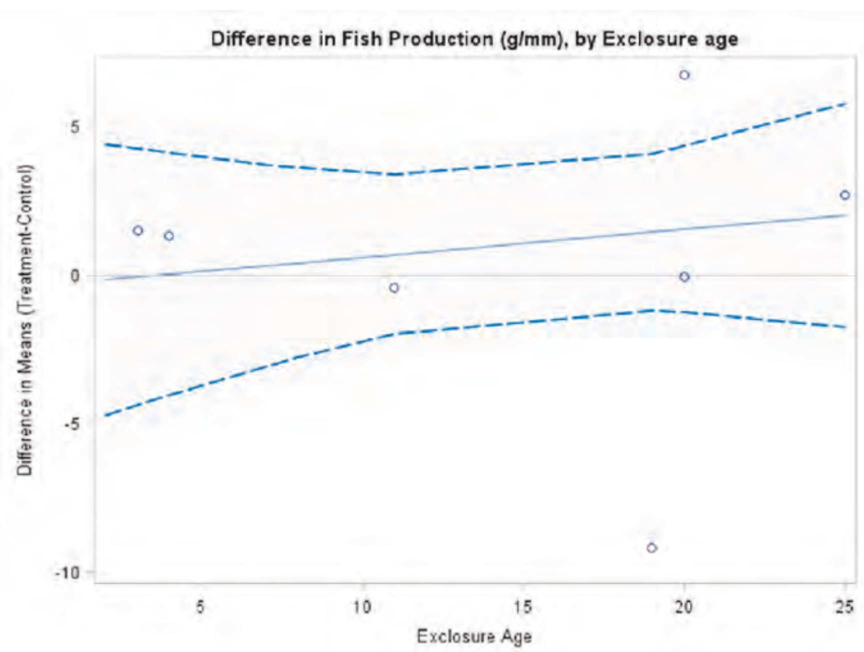


Figure ES14. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs, and 80% CIs on the dashed lines), across different ages of exclosures, in fish production, excluding age 0 steelhead.

EXECUTIVE SUMMARY: ANALYTICAL FRAMEWORK

Determining Metrics Useful for Change Detection

One of ISEMP's primary goals is to evaluate which metrics and indicators are useful in determining status and trends in stream habitat.

- ISEMP employed a Bayesian hierarchical model using posterior distributions of regression parameters to look for spatial and temporal patterns in the metric data. This model is a powerful tool for exploring data and allows for a graphical inspection of the data in an intuitive manner.
- The hierarchical aspect of the model enables a elucidation of patterns at different scales, such as the watershed and subbasin scales, while the Bayesian approach allows us to reveal the shape of the distribution o f the parameters.
- Results of this process were incorporated into CHaMP.

Describing Habitat-Juvenile Salmonid Abundance Relationships using Wenatchee ISEMP Data

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 using a boosted regression tree approach. This approach can make predictions about one measurement, such as fish density, based on a variety of predictor variables, including continuous measurements and classifications from habitat metrics, and can be used to assess the relative importance of the predictor variables and incorporate non-linear relationships between habitat and fish. For example:

- A year effect was the most important metric for predicting the density of juvenile Chinook, which was about twice as important for predicting juvenile Chinook density as gradient or a measure of temperature (Figure ES15, left panel).
- 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 because of brood year strength and migratory characteristics of juvenile salmonids.
- Steelhead respond to a different set of habitat metrics than Chinook. Four habitat metrics (Figure ES15 right panel) explain 82% of the variance in steelhead density at different sites. Steelhead are generally

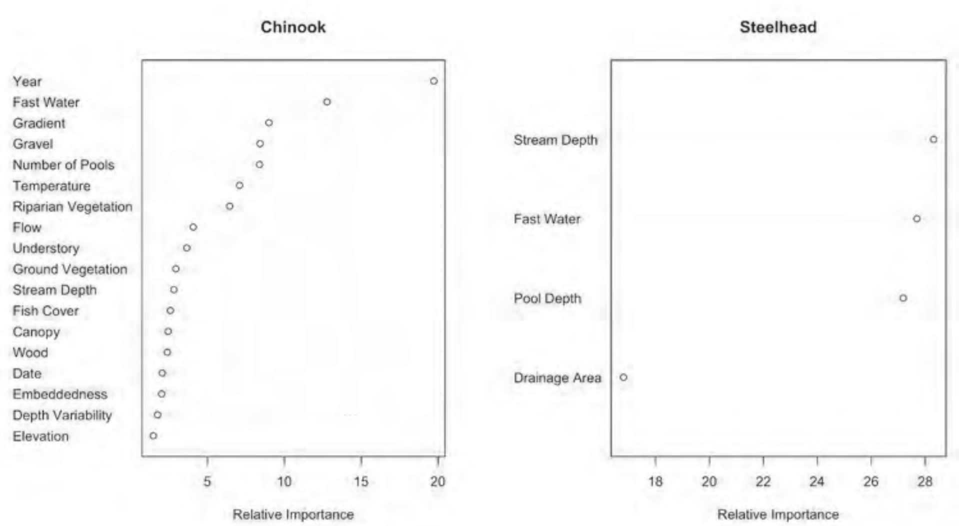


Figure ES15. The relative importance of various habitat metrics in predicting the density of juvenile Chinook (left panel) and steelhead (right panel) using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010 analyzed using a boosted regression tree approach.

found in higher densities in shallower streams with more slow water and deep pools.

- Some of these metrics also impact Chinook, such as the percent of fast water, but the relationship between the habitat metric and fish density is different for each species.

By isolating the effect of each habitat characteristic on fish density while all other characteristics of the habitat remain constant we looked at the expected effect on fish density as each habitat metric changes. We created partial dependence plots for the most important habitat variables for predicting juvenile abundance in the Wenatchee dataset revealed by the boosted regression tree. Rather than linear relationships several thresholds become apparent that can be used to identify limiting factors and provide quantifiable goals for habitat restoration work.

- Thresholds, suggesting limiting factors, are revealed when we look at the relationships between the most relevant habitat metrics and salmonid density (Figure ES16).

- Restoration actions need to account for the target species since different species have different habitat needs. Steelhead respond to a different set of habitat metrics than Chinook and at different thresholds that are consistent with differences between the species.

- 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 "What habitat actions are most effective?"

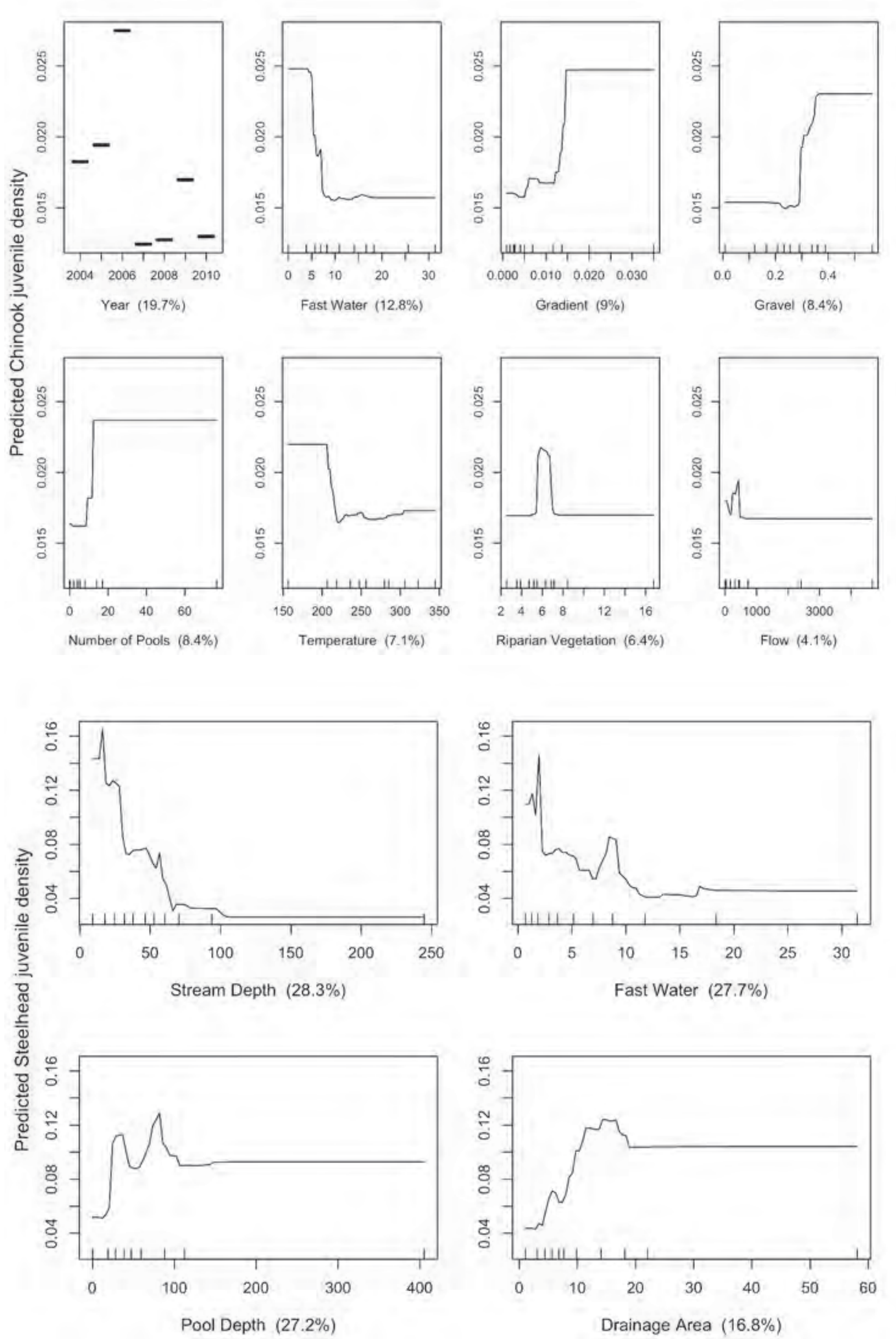


Figure ES16. Partial dependence plots showing the effect of the eight most important habitat metrics on juvenile Chinook densities (top panel) and four most important metrics for steelhead densities (lower panel) identified using a boosted regression tree approach using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010. (Percentages show relative importance from Figure ES14).

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

Managers need to be able to predict where habitat conditions are expected to be good or poor to efficiently guide habitat restoration planning. Developing spatially explicit models of expected habitat condition will allow us to create maps that show spatial patterns in expected good or poor habitat condition. These maps will facilitate targeting 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. Here we present a landscape classification that organizes watersheds (6th field HUCs) into classes with common natural features and classes with common "disturbance" features. This 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.

- Maps can be created that 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 ES17).

Using habitat monitoring data and assigning each sampling location a disturbance score (Best, Good, Moderate, and Poor) and a geomorphic valley type (Mountain and Floodplain/Constrained) illustrates 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.

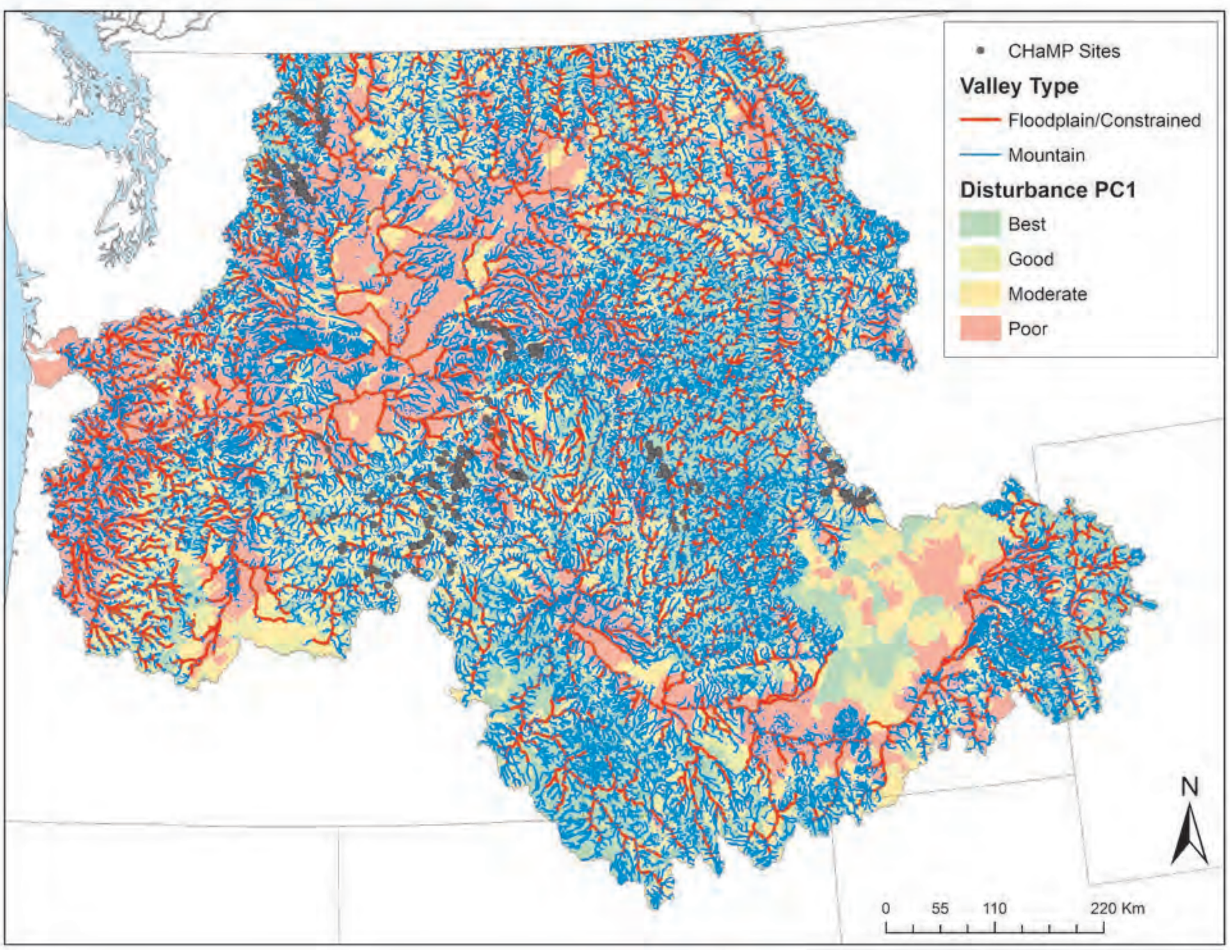


Figure ES17. A map illustrating where the probability of finding poor habitat condition is likely to be high and therefore where habitat restoration might be concentrated in the Upper Columbia. The stream network classified into two geomorphic groups: Mountain and Floodplain/Constrained) because patterns of disturbance and recovery goals could differ.

Evaluating Temperature Impairment and Intrinsic Potential

Summer stream temperature is thought to limit salmonid productivity. In parts of the interior Columbia River basin, summer stream temperatures are naturally higher than those tolerated by cold water fishes, but in other parts of the basin, human activity has resulted in elevated stream temperatures. The intersection of natural and man-made temperature regimes means 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.
- These maps can be liked with salmonid habitat intrinsic potential (IP), to predict the spatial locations (stream reach), degree of impairment (risk score), and relative priority for mitigation actions (risk score x IP score; Figure ES18).
- 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.

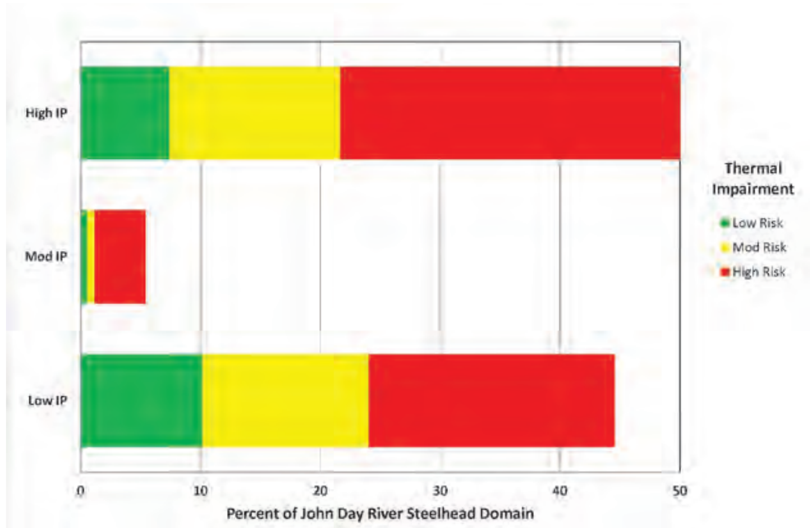


Figure ES18. Relative proportion of the John Day River basin steelhead domain of Low, Moderate and High Intrinsic Potential (IP) falling in Low/Moderate/High summer thermal impairment conditions.

Salmonid Production in a Life – Cycle Context

There is an absence of life cycle models to estimate habitat quality and freshwater survival benefits for anadromous salmonids.

- ISEMP is developing a watershed model in the Lemhi subbasin that is based on the premise that juvenile distribution, abundance, and survival are functions of habitat quantity and quality.
- This model provides several useful products for managers: and policy makers
  - ◊ Identifies limiting factors
  - ◊ Identifies project types:
  - ◊ Compares project types
  - ◊ Relates habitat improvements to survival improvements
  - ◊ Identifies appropriate research monitoring and evaluation
  - ◊ Evaluates changes in habitat and fish
  - ◊ Predicts adult returns
- The model assumes that locations supporting higher juvenile abundance and survival are indicative of good habitat, allows us to identify habitat features that are indicative of high quality habitat at each life stage, and which habitat attributes limit egg to smolt survival.
- Currently the model is based on data collection activities in the South Fork Salmon River and Lemhi River so it is initially most applicable to those locations.
- Model will be fully populated in 2013, allowing development of empirically based relationships and identification of minimum data requirements. This will enable a more generalized version of the model to be cost-effectively deployed across the Columbia River Basin in 2014.
- Transferability of the model to other watersheds will be assessed by testing the sensitivity of model results to differing data types with a range of uncertainty utilizing information collected by ISEMP in the John Day and the Wenatchee and Entiat in the Upper Columbia basin.

EXECUTIVE SUMMARY: DATA MANAGEMENT

ISEMP has developed several data management products to facilitate data analyses, data storage, and data retrieval within and across ISEMP pilot subbasins. Lessons learned include:

- Field protocols for collecting data are essential for maximizing the utility of data collected within a program.
  - ◊ Field protocols should include information on the associated metadata
    - * Metadata includes the how, what, when, where, and why of data collection: 1) study objectives and design; 2) protocols; 3) measurement details (definitions, units and species codes or size classes); 4) data processing procedures (data quality, summary, and metric calculations); and 5) intended analyses.
  - * Review field protocols annually to determine if field methods were accurate, precise, and contributed to the metrics of interest.
  - * Be diligent in tracking and recording annual changes in a format that is accessible, easy to maintain, and is stored closely with the data to guard against separation of data from metadata.
- The goal to develop a centralized standardized data storage tool (STEM Databank) highlighted many inconsistencies with data collection and management across the Upper Columbia, where the effort was piloted.
  - ◊ Organizations utilize multiple protocols and data storage structures, which sometimes changed annually within the same organization. This presented a challenge for documenting and storing data across years and agencies.
  - ◊ Consequently, the STEM Databank was developed as a highly normalized structure that would allow storage and retrieval of data from multiple protocols. This required an immense overhead of metadata, which was often limited in provided data and took several years to document and align with source data.
  - ◊ ISEMP designed a global-schema approach to managing data from disparate sources (the Aquatic Resources Schema), which was used from 2006-2010 to facilitate data entry, metadata documentation, terminology, and import formats for the STEM Databank. However, the structural complexity required to manage the diverse incoming datasets distracted from the feasibility of implementing it effectively across all ISEMP pilot subbasins.
  - ◊ The Aquatic Resources Schema demonstrated the utility and benefits of storing detailed metadata with raw data and the utility of a global schema for fisheries data. These concepts have continued to influence the development of other ISEMP data management tools.
- The increasing reliance on PIT tags to estimate adult escapement and juvenile distribution, abundance, and survival is complicated by the lack of standardized data capture and local database utilities.
- ISEMP has developed a data management system for remote juvenile capture and PIT tagging that incorporates hand-held data loggers for field data collection, a project level data storage standard, data QA/QC modules, and data transfer applications.
  - ◊ Electronic data-capture devices minimize data loss and data entry during surveys.
  - ◊ Using a data logger to electronically capture information requires an underlying standardized field collection protocol, so the implementation of a common data logger program across projects simultaneously increases standardization.
  - ◊ Electronic data capture reduces the labor and accompanying potential for transcription errors that accompany the transfer of data from field forms to electronic format.
  - ◊ The use of a data logger significantly reduces the QA/QC burden after the field season, limits data losses resulting from corrupted or incomplete surveys, and expedites data reduction and reporting.
- The development of instream PIT tag detection sites (IPTDS) technology represents a significant advancement with regard to the estimation of juvenile and adult distribution, abundance, and survival.
  - ◊ ISEMP has developed a data management system that allows efficient and real-time access to IPTDS site data, site diagnostics, and data storage and information transfer to regional data systems (Figure ES19).
  - ◊ ISEMP, in coordination with Biomark, has developed a standardized suite of PIT tag array infrastructure enabling reliable remote downloading of interrogation data and routine site diagnostics, and has developed software that automatically parses downloaded PIT

tag interrogation data, reduces the data to required fields for regional databases such as PTAGIS, and uploads the data automatically on a daily basis to PTAGIS. This process both automates data QA/QC and provides detection data, in near real-time, to the region.

◊ ISEMP developed the Instream PIT Tag Detection Database to efficiently access and store the large quantity of interrogation and diagnostic information, and also developed an automated upload system (LNDRefactor) to ensure data is transferred to PTAGIS in a timely fashion.

◊ ISEMP developed a real-time monitoring system of conditions at each IPTDS using software that allows a site data steward to visually monitor sites using a web interface. This interface allows the data steward to determine whether a site is functioning and if any alerts are present.

◊ Several ISEMP cooperators in the Upper Columbia and Snake River have adopted the IPTDS tool.

◊ ISEMP is currently assisting PTAGIS and Pacific States Marine Fisheries Commission (PSMFC) staff in identifying data management needs for IPTDS that are not supported by the current PTAGIS database.

- Although rules regarding QAQC are generally well known, they are not often practiced in a consistent fashion and ISEMP has learned that:

  - It is critical for field data collectors to review data soon after data collection;
  - Quality assurance checks performed by field crews should conform to a programmatic standard;
  - Quality assurance checks should be replicated within the program (e.g., performed locally and by a central quality assurance manager; and
  - Quality standards should be reviewed annually and prior to each analysis.

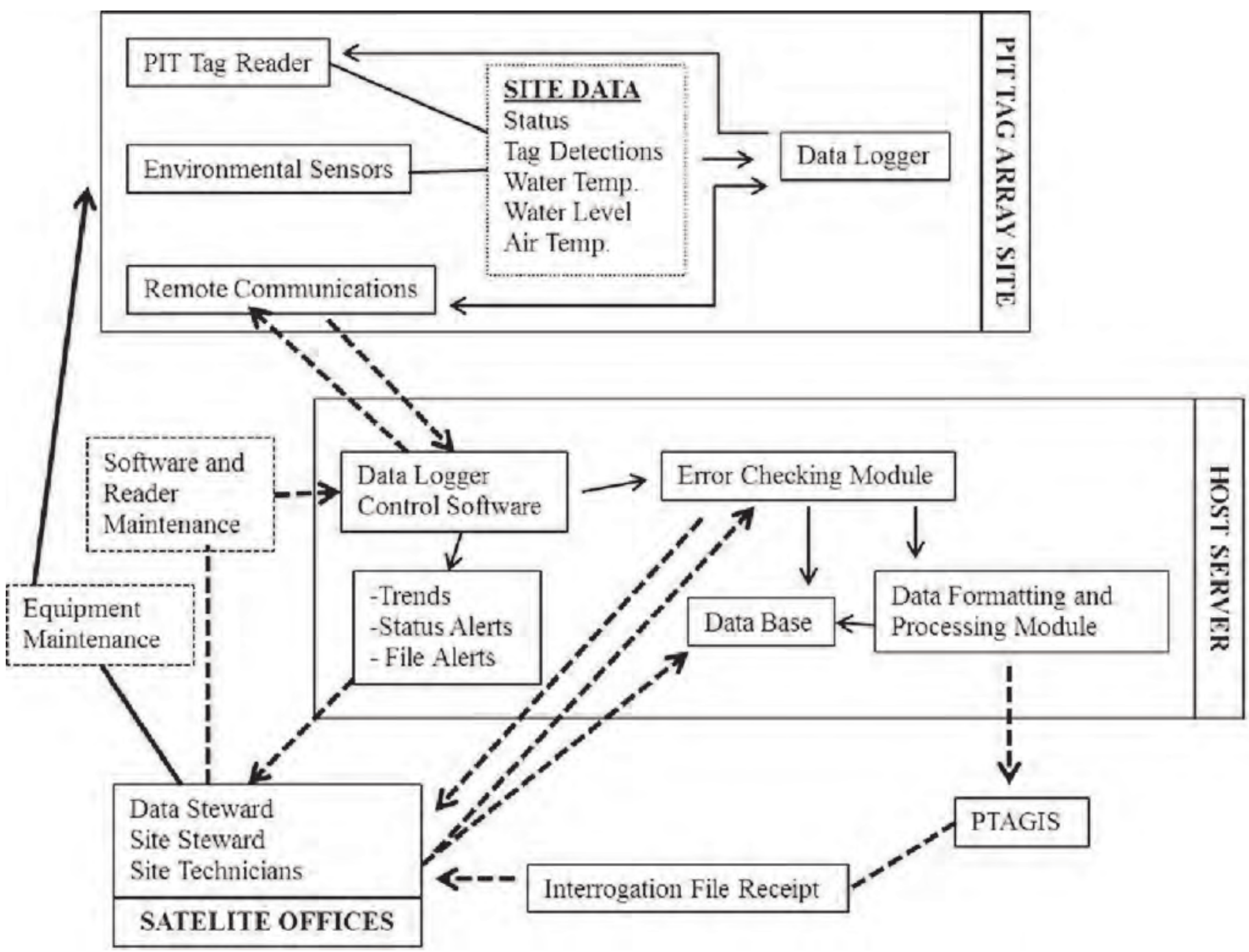


Figure ES19. ISEMP Instream PIT Detection Site Data Management System.

I. INTRODUCTION

Nearly 10 years ago, in developing the implementation plan for the 2000 Federal Columbia River Power System Biological Opinion (FCRPS BiOp), staff from National Oceanic Atmospheric Administration (NOAA), Bonneville Power Administration (BPA), U.S. Bureau of Reclamation (BOR), and U. S. Army Corps of Engineers recognized seemingly intractable key management questions that underpinned the proposed tributary habitat-based, off-site mitigation strategy. They 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?" as the foundations from which to build (Jordan et al. 2003). The Integrated Status and Effectiveness Project (ISEMP) was the result. In this report we will focus on the achievements of ISEMP since its inception in 2003 relevant to answering key management questions in both the science and policy arenas.

The scientific elements of these questions include how to measure habitat, fish populations, and the effects of habitat restoration on fish populations, as well as identifying and answering scientific uncertainties surrounding these measures, such as what aspects of fish habitat influence fish population changes. The policy elements are about defining "performance" and "effectiveness," providing a scale for "cost-effectiveness" and deciding what level of effort is enough to make efficient policy decisions. 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 questions on the role of tributary habitat management in salmonid population impact mitigation are decades old; directed research projects have been underway for almost the same time period, and yet definitive answers to the questions remain elusive. In this report, we describe tools that allow the resource management community to address these questions using scientific raw materials to make decisions. We also present frameworks for thinking about and interpreting status, trends and effectiveness monitoring data, and mechanisms to support management action design and implementation.

BiOp Context for ISEMP

Research and monitoring conducted under ISEMP are designed to address FCRPS BiOp requirements to evaluate the effectiveness of, and priorities for, tributary habitat protection and restoration projects, such as instream flow enhancement, screening irrigation diversion, restoring riparian areas and connectivity, and removing passage barriers. These habitat actions are being implemented as biological strategies under both the Northwest Power and Conservation Council's (NPCC) subbasin plans and Endangered Species Act (ESA) recovery plans endorsed by the National Marine Fisheries Service (NMFS), states, tribes, and local partners. Since habitat conservation was first included in the 2000 BiOp, BPA has spent more than $517 million dollars on salmon and steelhead habitat actions (FY2005 – 2011), and has invested even more in habitat conservation for resident fish and wildlife. For example, between 2005 and 2010, the Federal Action Agencies have:

•Acquired instream water to conserve or protect over 250,000 acre-feet of water in salmon and steelhead streams,

•Installed 220 fish screens on irrigation diversions,

•Improved more than 5,700 acres of riparian habitat,

•Improved 161 miles of spawning and rearing stream habitat, and

•Opened more than 1,350 miles of tributaries for salmon and steelhead spawning and rearing habitat.

These projects are currently targeted at priority populations and key limiting factors, and are prioritized and evaluated with the assistance of the BPA and USBOR Expert Panel Process.

Key Management Questions

- What are the tributary habitat limiting factors or threats preventing the achievement of desired tributary habitat performance objectives?
- What are the relationships between tributary habitat actions and fish survival or productivity increases, and what actions are most effective?
- Which actions are most cost-effective at addressing identified habitat impairments?

ISEMP's Mission

ISEMP's mission 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. The resource management community in the Columbia River basin is keenly interested in the process by which such an analytical framework is developed as well as the resulting "decision support products". The ISRP, in commenting on ISEMP's proposed work plan during the Council's Categorical Review process, expressed this interest and their uncertainty as to how the work would be accomplished. Three of their comments are as follows:

"We are still not sure how habitat status and trend monitoring data will be related to (integrated with) status and trends of fish population data within CHaMP watersheds to evaluate the effectiveness of specific restoration strategies or general restora-

tion effectiveness in a geographic area (e.g., are the co-managers in a given subbasin successful in restoring stream habitat in their area?)."

"It was unclear which entity or entities will be responsible for conducting fish status and trends monitoring at CHaMP sites, what kinds of fish data would be collected (e.g., site/ reach-specific abundance sampling or fish in- fish out), and what kinds of analytical methods will be used to relate fish status and trends to habitat status and trends."

"We believe that the description of life stages influenced by various habitat measurements could be more refined. Where possible, illuminate how some restoration actions are influencing VSP parameters."

There is no single best way to build this important decision support tool. 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).

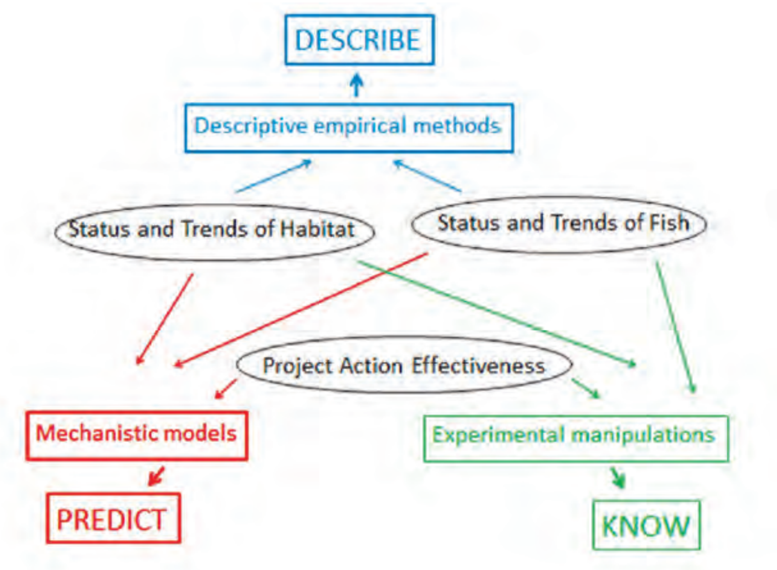


Figure 1. The three methods used by ISEMP to connect fish and habitat data that are the basis of the analytical framework proposed to managers for use as a decision-making tool to answer management questions.

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

[1]**Reasonable and prudent alternative language from the 2008 FCRPS Biological Opinion**

RPA 50.4 -- Fund status and trend monitoring as a component of the pilot studies in the Wenatchee, Methow, and Entiat River basins in the Upper Columbia River, the Lemhi and South Fork Salmon river basins, and the John Day River Basin to further advance the methods and information needed for assessing the status of fish populations.

RPA 50.5 -- Provide additional status monitoring to ensure a majority of Snake River B-Run steelhead populations are being monitored for population productivity and abundance.

RPA 56.1 -- Implement research in select areas of the pilot study basins (Wenatchee, Methow and Entiat river basins in the Upper Columbia River, the Lemhi and South Fork Salmon river basins, and the John Day River Basin) to quantify the relationships between habitat conditions and fish productivity (limiting factors) to improve the development and parameterization of models used in the planning and implementation of habitat projects. These studies will be coordinated with the influence of hatchery programs in these habitat areas.

RPA 56.2 -- Implement habitat status and trend monitoring as a component of the pilot studies in the Wenatchee, Methow and Entiat River basins in the Upper Columbia River, the Lemhi and South Fork Salmon River basins, and the John Day River Basin.

RPA 56.3 -- Facilitate and participate in an ongoing collaboration process to develop a regional strategy for limited habitat status and trend monitoring for key ESA fish populations. This monitoring strategy will be coordinated with the status monitoring needs and strategies being developed for hydropower, habitat, hatchery, harvest, and estuary/ocean.

RPA 57.1 -- Action effectiveness pilot studies in the Entiat River Basin to study treatments to improve channel complexity and fish productivity.

RPA 57.2 -- Pilot study in the Lemhi River Basin to study treatments to reduce entrainment and provide better fish passage flow conditions.

RPA 57.3 -- Action effectiveness pilot studies in Bridge Creek of the John Day River Basin to study treatments of channel incision and its effects on passage, channel complexity, and consequentially fish productivity.

RPA 57.4 -- Project and watershed level assessments of habitat, habitat restoration and fish productivity in the Wenatchee, Methow, and John Day basins.

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.

Experimental manipulation is 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.

Given the constraints of descriptive and predictive methods, it might seem that the ideal approach would 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, for example, experiments can be expensive and can only be performed in rare cases where technical and social conditions in a watershed are suitable.

In reality, a combination of all three methods is 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, 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.

Linking habitat and fish can be done within and across geographic areas because ISEMP is working at multiple scales (site, reach, watershed, subbasin) in each of its focal areas (Upper- and Mid-Columbia, and Snake; Figure 2). Table 1 illustrates the fish metrics ISEMP generates and the resulting capacity to link with stream habitat data at multiple scales based on the grain (spatial-temporal resolution) and extent (spatial-temporal coverage) of the fish and habitat data for correlation models.

Correlation models are one of two indirect methods that ISEMP has undertaken to relate fish and habitat. To directly address the dependence of fish population processes on stream habitat quality and quantity ISEMP is implementing IMWs. The Entiat, Lemhi River and Bridge Creek (lower John Day River) IMWs all involve a habitat action implementation design and a fish and habitat monitoring design to optimize the potential of quantifying the impact of habitat actions on the local fish population.

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 BiOp and the EF&W Program.

Tasks and Objectives

Like all of the RME being conducted under the BiOp and the Council Program, ISEMP tasks, activities and results fall into certain discrete, but related categories:

- **Status and trends:** monitoring data on fish and habitat to track and evaluate fish-habitat relationships at the ESU, subbasin, and population levels.

- **Action Effectiveness:** evaluating the effect of habitat actions (both project level, i.e., type of project, and watershed level, i.e., cumulative projects in a given area) on fish populations.

- **Analytical Framework:** providing the context for monitoring data to address fish-habitat relationships, limiting factors, and whether management actions and restoration has led to changes in fish and their habitat.

In its original manifestation, ISEMP was a project consisting of subbasin-scale pilot programs that focused on fundamental components: 1) how to develop status and trend monitoring efforts for anadromous salmonids and their habitat in the Wenatchee and upper Grande Ronde River basins, and 2) how to support restoration action planning. These tasks were developed specifically to be distinct from much of the ongoing status and trend monitoring in the Columbia River basin, as they were to focus on the explicit development and testing of the sampling protocols and methodologies required for generating habitat and population monitoring data of known spatio-temporal resolution, accuracy and precision. The primary utility of the information generated was for annual assessment of status and trend for these fishes and their habitat. Additionally, the program supported restoration action planning and assessment by serving as the baseline information used for action siting, and the baseline against which actions' biological impact could be measured.

Table 1. Outline of the correlation-based modeling ISEMP can do relating tributary habitat characteristics to fish metrics. Four fish metrics are given in the first column. Spatial and temporal grain of fish metrics and the methods for metrics generation is shown in column three by subbasin. The final two columns show the spatial and temporal grain of the correlation models that are possible across ISEMP and potentially at finer spatial-temporal resolution within specific subbasins.

		Metric generation	Model scale	
Metric	Subbasin	Spatial and temporal grain: methods	ISEMP-wide	Subbasin
Juvenile Growth	Wenatchee	Annual at the watershed grain: PIT tag recapture based growth metric	Annual at the watershed grain	John Day, Entiat - seasonal, site grain
	Entiat	Annual at the watershed grain, and seasonally at site grain: PIT tag recapture based growth metric		
	John Day	Annual at the watershed grain, and seasonally at site grain: PIT tag recapture based growth metric		
	Lemhi/Secesh	Annual at the watershed grain: PIT tag recapture based growth metric		
Juvenile Survival	Wenatchee	Annual at the watershed grain: PIT tag mark-recapture based survival metric	Annual at the watershed grain	John Day, Entiat - seasonal, site grain
	Entiat	Annual at the watershed grain, and seasonally at site grain: PIT tag mark-recapture based survival metric		
	John Day	Annual at the watershed grain, and seasonally at site grain: PIT tag mark-recapture based survival metric		
	Lemhi/Secesh	Annual at the watershed grain: PIT tag mark-recapture based survival metric		
Juvenile Abundance	Wenatchee	Annual at the watershed grain: by GRTS and RST based metric; Summer at the site grain: by mark-recapture, ratio, and GRTS based metric: summer at the channel unit grain: by depletion and one-pass based metric	Annual at the watershed grain	N/A
	Entiat	Annual at the watershed grain: by GRTS and RST based metric; seasonal at the site grain; by IMW design		
	John Day	Annual at the watershed grain: by GRTS based metric; summer at the site grain: by mark-recapture and one-pass based metric		
	Lemhi/Secesh	Annual at the watershed grain: by GRTS and RST based metric; summer at the site grain: by GRTS based metric		
Productivity	Wenatchee	Annual at the subbasin grain: by juvenile to adult and RST to redd based metric	Trends at the subbasin grain	N/A
	Entiat	Annual at the subbasin grain: by juvenile to adult and RST to redd based metric		
	John Day	Annual at the watershed grain: by juvenile to adult based metric annual; Annual at the subbasin grain: by RST to redd based metric		
	Lemhi/Secesh	Annual at the watershed grain: by juvenile to adult based metric		

During the first several years of implementation, ISEMP's focus shifted in response to Independent Science Review Panel (ISRP)/NPCC feedback and to meet additional programmatic needs of the EF&W program. In particular, the geographic extent was modified to include the John Day River basin in Oregon and the Salmon River basin in Idaho, rather than Oregon's upper Grande Ronde River basin (Figure 2). To more specifically address issues concerning evaluating watershed level population responses to habitat management strategies, ISEMP was given the task of developing IMWs in each of its focal areas (Upper and Mid-Columbia, and Salmon River)[2].

[2]The three study plans were reviewed by the ISRP - Review of the ISEMP John Day Study Plan, ISRP 2007-8; Review of Salmon Subbasin Pilot Projects Monitoring and Evaluation Plan, ISRP 2006-1; Review of revised mainstem/systemwide proposals for Research, Monitoring, and Evaluation, ISRP 2003-6.

Currently, ISEMP has nine programmatic objectives, five within each of its focal areas and three that are shared across the project as a whole. At the subbasin-scale ISEMP has the following objectives:

- **Programmatic coordination, design, planning and implementation -** Establish contacts and coordinate current monitoring and evaluation activities

- **Indicators and metric development and testing -** Establish causal relationships between ecological processes that control fish production, and develop metrics and indices to better capture these mechanisms.

- **Protocol development, refinement and testing -** Determine the accuracy and precision of information we need collected through different protocols.

- **Sampling design development and testing -** Given what we learned about the accuracy and precision of different protocols, develop a sampling design.

- **Effectiveness and status and trend monitoring experimental design and implementation -** Determine the effectiveness of restoration actions through effectiveness monitoring or an experimental management framework, such as the Intensively Monitored Watershed studies.

At the project level, ISEMP has the following objectives:

- **Evaluation tools development and testing –** Develop monitoring data analysis tools and tools to evaluate monitoring programs and approaches.

- **Data management tools development and testing –** Develop databases, data communication templates, data and information output tools, and populate databases with current and historic monitoring data.

- **Columbia River basin-wide stream habitat status and trends monitoring –** Develop a standardized, programmatic approach to habitat data collection to support habitat resource assessments and the evaluation of population-scale habitat – fish relationships.

Implementation Timeline

ISEMP is designed to build on and add to the existing body of data on the benefits of tributary habitat protection and restoration. ISEMP work has been undertaken at hundreds of field locations developing fish and habitat status and trends monitoring efforts in the Wenatchee (began 2004), Entiat (2005), John Day (2006), South Fork Salmon and Lemhi River basins (2009, Figure 3), and in 2011 included the Columbia Habitat Monitoring Program (CHaMP) to assist in further developing fish and habitat monitoring. IMWs were implemented in 2009-2010, along with Intensively Surveyed Watersheds (ISWs, reference areas for the habitat manipulation watersheds

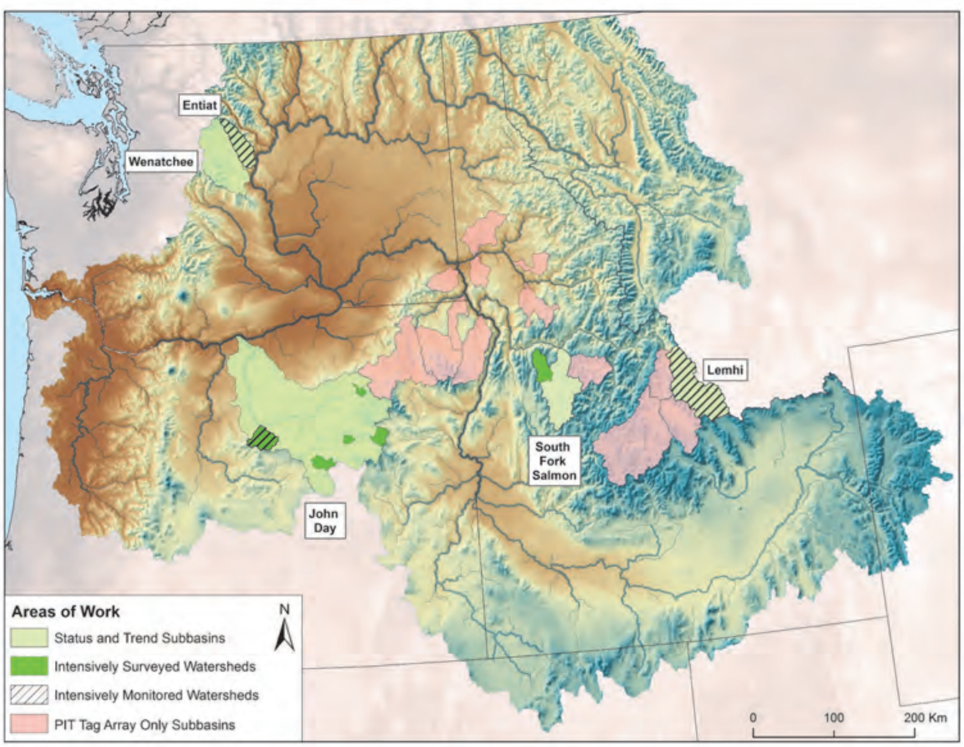


Figure 2. Location of the ISEMP subbasins and associated Intensively Monitored Watersheds (IMWs) and Intensively Surveyed Watersheds (ISWs), and subbasins where monitoring is conducted using PIT tag detection arrays.

and metrics, indicator, and survey design development test-beds). By coordinating fish data collection with habitat data collection from CHaMP, ISEMP is able to link habitat condition to fish populations, and ultimately, changing habitat conditions to change in fish population status.

In addition, ISEMP is developing methods for adult and juvenile salmonid population estimation based on PIT tag detections – these efforts are underway in all of the ISEMP status and trends locations, as well as a suite of subbasins in the Snake River. As a result of all of ISEMP's actions, the project generates adult and juvenile salmonid population estimates for status, trends and action effectiveness monitoring for 28 populations (Tables 2 and 3).

Partnerships and Collaborators

ISEMP's work is critically dependent on a vast network of collaborators and cooperators. NPCC's Draft Columbia River Basin Monitoring, Evaluation, Research and Reporting (MERR) Plan emphasizes collaborative monitoring as a mechanism for achieving more efficient and effective programs and to foster the development and use of standardized methods (NPCC 2010). ISEMP depends on collaboration at the project management level through engagement with local and regional groups such as the Pacific Northwest Aquatic Monitoring Partnership and the Upper Columbia Salmon Recovery Board's Regional Technical Team, as well as ISEMP specific co-manager groups such as the Salmon Basin Research, Monitoring and Evaluation Technical Oversight Committee. For on the ground data collection, ISEMP has built off, and is dependent upon, the ongoing collaborations with monitoring programs run by Washington Department of Fish and Wildlife (WDFW), Oregon Department of Fish and Wildlife (ODFW), Idaho Department of Fish and Game (IDFG), the Yakima Nation, the Nez Perce Tribe and the U.S. Fish and Wildlife Service (USFWS). As the chapters of this lessons learned report show, ISEMP data analysis tasks are also collaborative efforts with technical staff from Columbia River Intertribal Fish Commission, WDFW, ODFW, IDFG, the Nez Perce Tribe, Utah State University and the USFS Rocky Mountain Research Station. The project relies on a broad and diverse partnership with state, tribal and federal resource managers to be successful at tool and method development for the co-manager community.

ISEMP Products

The evaluation of management actions can be predictive, used in an adaptive management framework to forecast, plan and prioritize projects, and it can also be extrapolative, used to quantify the potential impact of ongoing actions not included in an explicit evaluation or monitoring design. In either case, management decisions underlying the design and evaluation of the FCRPS BiOp habitat strategy need to be based on a documented, scientifically rigorous rule-set that links habitat condition with fish population response. In order to be most effective, the technical detail of how fish populations respond to habitat conditions must be translated into tools to support: decision-making, interpretation by broad audiences, and use by technical and non-technical elements of the co-manager community.

Connecting habitat quality and quantity to fish population processes quantitatively allows the evaluation of habitat management actions for their potential impact on salmonid population abundance and productivity. ISEMP is developing management decision support tools from quantitative relationships of stream habitat quality and quantity's impact on anadromous salmonid population abundance and productivity in the Columbia River basin. However, while data analysis and interpretation is the ultimate goal of ISEMP, the vast majority of project resources are targeted to the data streams: defining, collecting, storing and curating field data are the ISEMP tasks that make it possible to develop robust fish-habitat relationships.

ISEMP's primary goal is to generate the decision support products that form the foundation of the FCRPS BiOp habitat strategy. Figure 3 and Table 4 show these products in more detail, organized into seven focal areas – three that are primarily data collection (ISEMP data streams) and four that are primarily data processing (ISEMP data management, monitoring guidance, and decision support tools). Each product has an expected delivery date – most products are in the form of software (analysis and data management tools) or technical reports (manuscripts for peer review publication or agency technical reports). A more complete listing with product descriptions is presented in Chapter 11 of the Appendix.

In this report, we show how ISEMP is building the analytical framework necessary to evaluate the habitat strategy's relevance, extent, timing and contrast. ISEMP is working the co-manager community and the FCRPS BiOp policy group to shape this framework to generate pragmatic answers to the key management questions. Overall, this report is meant to reinforce past and ongoing RME actions and the resource management community's expectations of the role habitat conservation will play in recovering fish populations in the Columbia River basin. The sections that follow will show: what have we learned so far about fish status and trends at the subbasin, and population levels; what have we learned so far about habitat action effectiveness at the watershed level; what have we learned so far about limiting factors for particular subbasins or populations, and what we have learned about developing explicit linkages between tributary habitat quantity and quality and anadromous salmonid population processes.

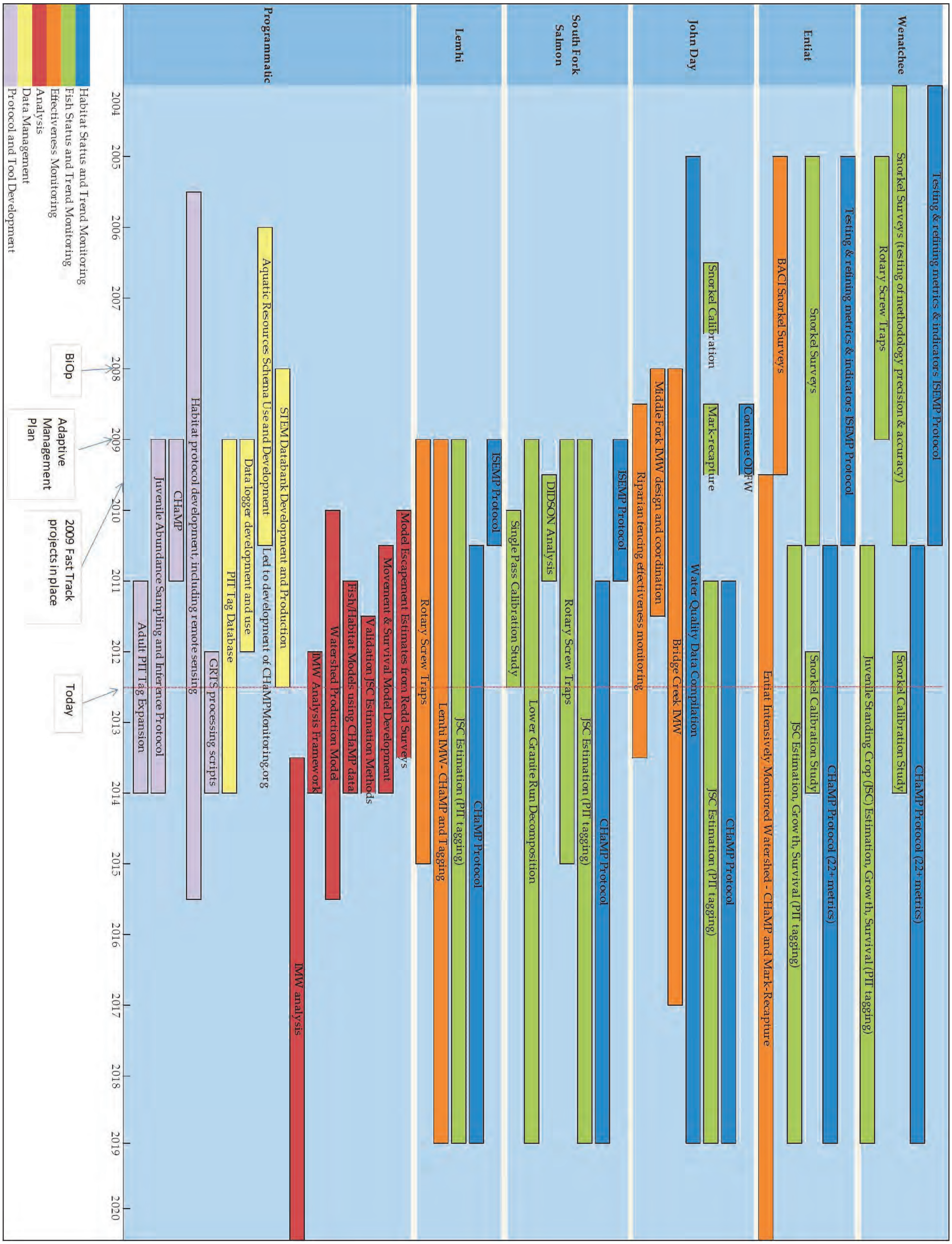


Figure 3. The timeline of implementation for status and trend and effectiveness monitoring activities in the three pilot subbasins under ISEMP, with associated analysis, data management, and protocol and tool development.

Table 2. Data availability for steelhead in the Columbia River Basin. Blue highlights indicate data are collected by ISEMP. This data summary illustrates the general relationship between ISEMP project areas and other ongoing Columbia Basin M&E programs, is collated from multiple sources, and is not meant to quantify activities in the Columbia Basin.

			Adult Escapement				Juvenile Abundance		
Extant ESU	Extant MPG	Population	Redd Count	Weir	Mark/ Re-capture	Uncertainty Measure?	Standing Crop	Emigration	Uncertainty Measure?
Middle Columbia	Cascade	White Salmon						Trap	
		Klickitat						Trap	
		15 Mile Creek	Yes						
		Deschutes West	Yes	Yes				Trap	
		Deschutes East	Yes					Trap	
		Crooked River							
		Rock Creek							
	John Day	JD Lower Mainstem	Yes	No	No	Yes	Yes	No	No
		NF John Day	Yes	No	No	Yes	Yes	Seine	Yes
		MF John Day	Yes	No	No	Yes	Yes	Trap	Yes
		SF John Day	Yes	No	No	Yes	Yes	Trap	Yes
		JD Upper Mainstem	Yes	No	No	Yes	Yes	Trap	Yes
	Walla Walla/ Umatilla	Willow Creek							
		Umatilla		Dam					
		Walla Walla		Dam					
		Touchet	Yes	Yes					
	Yakima	Satus Creek	Yes	Dam	Radio Tag				
		Toppenish Creek	Yes	Dam	Radio Tag				
		Naches	Yes	Dam	Radio Tag				
		Yakima Upper Mainstem	Yes	Dam	Radio Tag				
Snake River	Lower Snake	Tucannon River						Trap	
		Asotin Creek		Yes	Yes	Yes		Trap	
	Grande Ronde	GR Lower Mainstem							
		Joseph Creek	Yes	Yes	Yes	Yes			
		Wallowa River	Yes		2012	Yes			
		GR Upper Mainstem	Yes		2012	Yes		Trap	Yes
	Clearwater River	Lower Mainstem Clearwater							
		NF Clearwater							
		Lolo Creek			Yes	Yes		Trap	Yes
		Lochsa River							
		Selway River							
		SF Clearwater			Yes	Yes			
	Salmon River	Little Salmon (LS)/Rapid River		LS		LS			
		Chamberlain Creek							
		SF Salmon River			Yes	Yes			
		Secesh River			Yes	Yes	Yes	Yes	Yes
		Panther Creek							
		Big Creek (BC)/Camas/Loon			BC	BC		Yes	Yes
		MF Salmon Upper Mainstem							
		NF Salmon River							
		Lemhi River			Yes	Yes	Yes	Yes	Yes
		Pahsimeroi River			Yes	Yes		Yes	Yes
		EF Salmon River							
		Salmon River Upper Mainstem			2012	Yes			
	Imnaha River	Imnaha River	Yes		Yes	Yes		Trap	Yes
Upper Columbia	Wenatchee/ Methow	Crab Creek	Rare	No	No	No			
		Wenatchee	Yes	Yes	Floy Tag	No	Yes	Trap	Yes
		Entiat	Yes	No	No	No	Yes	Trap	Yes
		Methow	Yes	Yes	Floy Tag	No	Yes	Trap	Yes
		Okanogan	Yes	Omak	2013	No	Yes	Trap	Yes

Table 3. Data availability for Chinook in the Columbia River Basin. Data sources highlighted in blue are collected by ISEMP. Blue highlights indicate data are collected by ISEMP. This data summary illustrates the general relationship between ISEMP project areas and other ongoing Columbia Basin M&E programs, is collated from multiple sources, and is not meant to quantify activities in the Columbia Basin.

			Adult Escapement				Juvenile Abundance		
Extant ESU	Extant MPG	Population	Redd Count	Weir	Mark/ Re-capture	Uncertainty Measure?	Standing Crop	Emigration	Uncertainty Measure?
Snake River	Lower Snake	Tucannon	Yes	Yes		Yes		Trap	
		Asotin	Yes	Yes		Yes		Trap	
	Grande Ronde /Imnaha	Wenaha	Yes	Yes				Trap	
		Lostine	Yes	Yes	Yes	Yes		Trap	
		Minam	Yes		Yes	Yes		Trap	Yes
		Catherine Creek	Yes	Yes	Yes	Yes		Trap	Yes
		Grande Ronde Upper Mainstem	Yes	Yes	Yes	Yes		Trap	Yes
		Imnaha Mainstem	Yes	Yes	Yes	Yes			
		Big Sheep Creek			Yes	Yes			
	SF Salmon River	SF Salmon Maistem	Yes		Yes	Yes			
		Secesh	Yes	Yes	Yes	Yes	Yes	Trap	Yes
		EF SF Salmon River	Yes	Yes	Yes	Yes		Trap	Yes
	MF Salmon River	Chamberlain Creek	Yes						
		MF Below Indian Creek							
		MF Above Indian Creek							
		Big Creek	Yes		Yes	Yes		Trap	Yes
		Camas Creek	Yes						
		Loon Creek	Yes						
		Sulphur Creek	Yes						
		Bear Valley Creek	Yes						
		Marsh Creek	Yes					Trap	
	Upper Salmon River	NF Salmon River							
		Lemhi River	Yes		Yes	Yes	Yes	Trap	Yes
		Mainstem Salmon Below Redfish							
		Pahsimeroi	Yes	Yes		Yes		Trap	Yes
		EF Salmon River	Yes	Yes		Yes		Trap	Yes
		Yankee Fork	Yes		Yes	Yes		Trap	Yes
		Valley Creek	Yes		Yes	Yes			
		Salmon River Above Redfish Lake	Yes		2012	2012		Trap	Yes
		Panther Creek							
		Little Salmon River						Trap	Yes
	Dry Clearwater	Lapwai/Big Canyon	Yes		Yes	Yes			
		Potlatch	Yes		Yes	Yes			
		Lawyer Creek							
		Upper SF Clearwater			Yes	Yes			
	Wet Clearwater	Lower NF Clearwater							
		Upper NF Clearwater							
		Lolo Creek	Yes		Yes	Yes		Yes	Yes
		Lochsa River	Yes						
		Meadow Creek	Yes						
		Moose Creek	Yes						
		Upper Selway River	Yes						
Upper Columbia	Wenatchee /Methow	Wenatchee	Yes	Yes	PIT TAGS	No	Yes	Trap	Yes
		Entiat	Yes	No	No	No	Yes	Trap	Yes
		Methow	Yes	Yes	PIT Tag	No	Yes	Trap	Yes
		Okanogan							

Table 4. A timeline showing products already completed and those scoped out for the future. Products represent those generated from the data streams from each of the three subbasins and the data processing that is associated with those data.

Data Stream/Data Processing	Product	Expected Date of Completion
Large-scale experimental manipulations of stream habitat condition to test mechanistic linkage between stream habitat and fish population processes	Effectiveness Monitoring of Riparian Fencing	June 2013
	Lemhi Intensively Monitored Watershed study	October 2017
	Bridge Creek Intensively Monitored Watershed study	December 2017
	Entiat River Intensively Monitored Watershed study	December 2020
Broad-scale juvenile and adult fish population monitoring aligned with ongoing habitat monitoring to form the basis for extrapolating mechanistic fish-habitat relationships beyond experimental watersheds	Estimating adult abundance from redds	Decemberr 2012
	Recommendations for steelhead redd surveys in the Wenatchee	*December 2012*
	Guidance on the use of steelhead index spawning ground counts.	*December 2012*
	Use of redd surveys manuscript	*June 2013*
	Recommendations for Chinook redd surveys in the Snake	*March 2014*
	Approaches to the uncertainty around downstream migrant trap data	December 2012
	Lower Granite Dam and IPTDS escapement estimates	Draft October 2010, final March 2013
	Steelhead life history patterns in the Wenatchee and Entiat River subbasins	June 2013
	Calibrating Snorkel Counts to Fish Abundance Estimates	November 2013
	Juvenile salmonid summer rearing (standing crop) population estimation	December 2013
	Juvenile Survival Models	December 2013
	Basin-wide estimates of juvenile abundance and juveniles per spawners	December 2013
	Precision, Bias, Reliability, and Cost-Comparison of RST Versus Instream PIT tag detection (IPTD) Based Juvenile Abundance and Survival	December 2014
Landscape context data collection and analysis to support extrapolating mechanistic fish-habitat relationships beyond experimental watersheds	Geomorphic framework (River Styles) for ISEMP watersheds	December 2014
	Integrate River Styles framework into the ISEMP Watershed Production model	June 2015
Survey and sampling designs to support population-scale inference of fish-habitat relationships	Determine strata for organizing standard stream habitat survey designs.	April 2010
	Design of observation error study.	June 2011
	Standardize fish sampling methods across ISEMP	June 2011
	Power analysis of habitat data.	April 2011
	PIT tag array detection analysis with DIDSON imaging sonar.	June 2011
	Evaluate channel unit fish sampling.	June 2012
	Set necessary sample size for calibrating single pass fish surveys.	June 2012
	Decision tree to determine fish sampling design.	December 2012
	Standardized GRTS processing scripts for fish and habitat data.	December 2013
Data management to support population-scale inference of fish-habitat relationships	Protocol documentation and requirements	January 2008
	STEM Databank	January 2008-present
	Measurement storage and retrieval	*January 2008*
	Metric storage and retrieval	*October 2012*
	Metadata linkage to MonitoringMethods.org	*December 2012*
	Metadata and metric linkage to ISEMP's fish database	*April 2013*
	Data distribution to DART	*August 2013*
	Metadata linkage to cbfish.org	*June 2013*
	Ongoing water quality compilation for John Day Basin	January 2009 & updates
	Data quality guidelines	May 2009
	Automated export of data between ISEMP and PTAGIS repositories	June 2009
	Aquatic Resources Schema (ARS)	September 2009
	IPTDS Data Management.	October 2009
	Uniform GIS layers compiled and clipped to PNW	January 2011, updated September 2012
	PITA inventory	February 2011, update September 2012
	IMW locations	June 2011, update September 2012
	Regional PIT Tag Data Queries.	January 2012
	Data Management lessons learned paper	July 2012
	Data flow paper	August 2012
	PIT tag Database.	December 2013
Spatio-temporal analysis of fish-habitat relationships to develop quantitative rule set that links abundance and productivity to habitat quality and quantity	Fish-habitat relationship modeling	December 2014
	Preliminary analysis on habitat data	*April 2010*
	Mechanistic models	*June 2013*
	Final analysis	*December 2014*
	Incorporate relationships in the watershed production model	*December 2014*
Watershed production models to evaluate the impact of management action scenarios for key populations and habitat action tactics	R-Code Watershed Model with Flexible Input and Parameterization.	December 2012
	Watershed Production Model for the Lemhi and Sesech Subbasins.	December 2012
	Watershed Production Model for Wenatchee and Entiat Subbasins.	July 2013
	Watershed Production Model for the John Day Subbasin.	December 2013
	Reduced Watershed Model	June 2015

II. LESSONS LEARNED ABOUT SAMPLING DESIGNS

Guidance on Sample Design and Sample Size for Habitat Status and Trends Monitoring

Central questions when designing a habitat or fish monitoring program are how much of the landscape must be sampled to accurately capture all of the natural variability and detect change due to natural or human factors, and what is the best way to sample in time and space to determine status and trends. The lessons learned from analysis of the Wenatchee habitat data on these questions were used in the development of the CHaMP habitat monitoring protocol.

Basic GRTS Design

Before the implementation of CHaMP in 2011, the primary objective of ISEMP's spatial and temporal study design in the Upper Columbia, Salmon and John Day was to characterize the status and trends of selected habitat indicators relevant to the survival and growth of key salmonid populations at two spatial scales: across and within the pilot watersheds. ISEMP adopted the use of the GRTS (Generalized Random-Tessellation Stratified (Stevens and Olsen 2004)) algorithm to select spatially-balanced sampling locations within each watershed. GRTS can provide a representative sample of habitat conditions, incorporating randomization in the selection of locations where habitat conditions are to be measured.

This sample design for habitat, fish abundance, and macroinvertebrate surveys was developed to describe current status and to detect trends for a suite of indicators within the target population. While status is best monitored by using as many sites as possible representing the broadest geographical distribution of metrics and indicators, trends are monitored by repeated sampling of the same sites over time. Thus ISEMP implemented a split rotating panel in the Wenatchee subbasin in 2005, consisting of both annual and rotating sites: the trend panel consisted of 25 sites that were surveyed annually, while the status panel consisted of a different rotating panel of 25 sites sampled every year, and revisited every five years (Table 5). Five such rotating panels of 25 sites each were selected from the GRTS site list. This meant that GRTS selected a total of 150 sites (6 panels x 25 sites per panel = 150 sites). The annual panel and that year's rotating panel were sampled annually.

When monitoring began in the Wenatchee subbasin in the Upper Columbia in 2004 it was piloted at 25 sites. In 2005 the 5-year split rotating panel design was implemented and the number of sites was increased to 50 in the Wenatchee and monitoring was implemented at 25 GRTS sites in the Entiat subbasin. In 2009 in the Wenatchee River subbasin effort was directed at visiting the 25 annual panel sites 3 times in the one season to measure crew and temporal variability (Figure 4).

Value of a Panel Design

Fish and habitat conditions vary over space and time and this variation is what we hope to capture in the determination of status and trends. This variation is also affected by measurement error among crews and by features of the landscape such as geomorphic valley classification. The ability to quantify and understand the components of variation is critical to knowing whether status and trend data are meaningful and usable by managers.

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

Table 5. Annual panel and rotating panel design for status/trend monitoring within a given status/trend monitoring zone (e.g., Wenatchee subbasin).

Panel	Year																			
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
1	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■
2	■					■					■					■				
3		■					■					■					■			
4			■					■					■					■		
5				■					■					■					■	
6					■					■					■					■

* Shading indicates the years in which sites within each panel are sampled. For example, sites in panel 1 are visited every year, while sites in panel 2 are visited only in years 1, 6, 11, and 16, assuming a 20-year sampling frame.

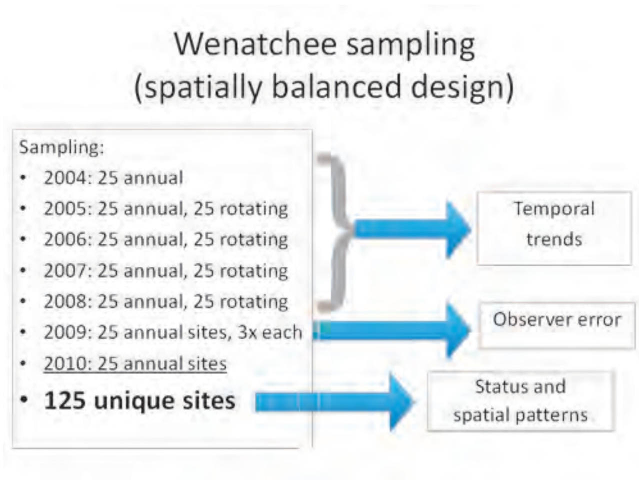


Figure 4. The number of sites sampled in the Wenatchee River sub-basin from 2004-2010 using a spatially balanced design.

components. The framework that ISEMP (and now CHaMP) use 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 those 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 (Figure 5).

As can be seen in Figure 5, 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 slightly larger 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.

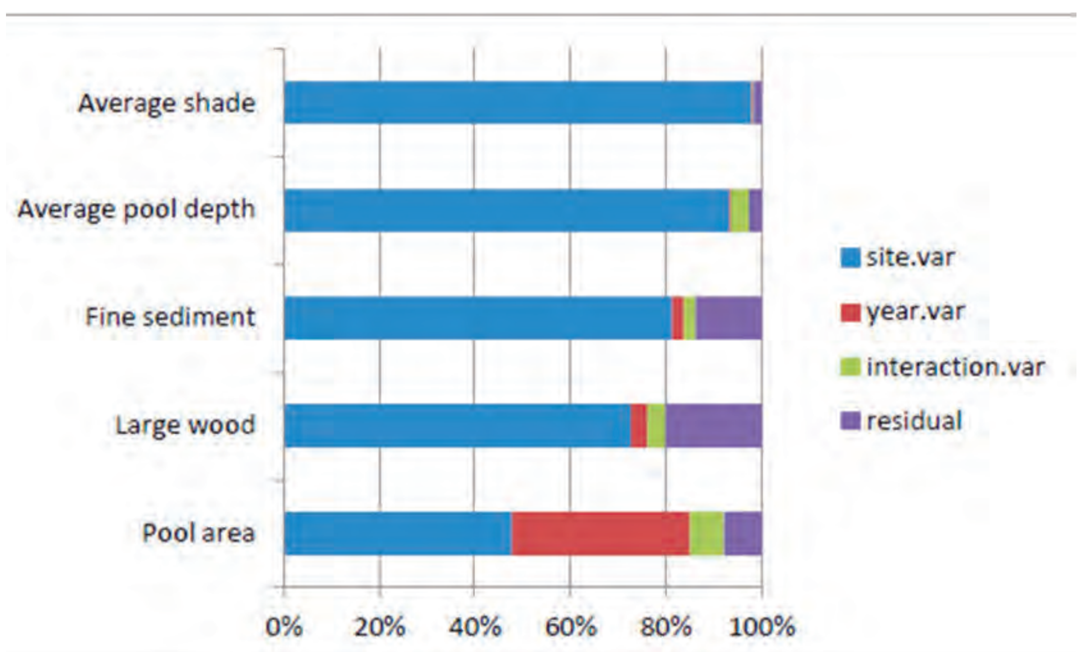


Figure 5. The relative proportion of total variation that is attributable to site, year, the interaction between site and year, and residual variation for five habitat metrics collected in the Wenatchee 2004 – 2010.

Figure 6. Sources of variation in habitat metrics collected in the Wenatchee subbasin with low and high numbers of within year repeat visits. Upper panel is low number of within-year revisits (n=61), lower panel high number of within-year revisits (n=15).

A variance decomposition analysis of the Upper Columbia data highlights the value of panel designs, in particular the repeated visits of within and across years, since independent site and year terms are better resolved, allowing for a test of the metrics and protocols' sensitivity to detect spatial and temporal patterns (Figure 6). The value of within year repeat visits for certain metrics is very apparent in a comparison of habitat metrics. For example, the PoolCount and

PercentFinesLT006mm metric residual variance is relatively high with a low number of revisits. Increasing the number of revisits allows the decomposition of a significant site*year term from a seemingly large residual term. Therefore, a within year repeat visit program is required to make best use of these metrics.

By using the panel design the information contained in all these metrics can be defined in terms of what metric and indicators are useful for describing habitat relationships. Habitat metrics on their own are not that important, it is the indicators and/or suite of information collected that provides the most information.

Sample Size

Characterizing the habitat of an entire watershed based on some number of site visits has to account for capturing the range of natural variability in the landscape in order to be able to detect any trends, and describe the status of habitat at any one point in time. Monitoring programs are faced with the challenge to design a monitoring program that balances capturing the status and trend of habitat with a limited amount of effort that can be deployed in the field using a limited budget. Since we are unable to saturate the landscape with sampling sites, what guidance is there for project designers when it comes to building a sampling design and choosing the number of sites to visit? Knowing how many sites to visit, and establishing useful guidelines on how to choose sites can greatly improve the description of watershed habitat. More precise estimates of the habitat indicators important to fish will maximize the potential signal of fish – habitat relationships, making it easier to detect a signal and improve the accuracy of analyses.

In order to address the question of how many habitat samples to take within a watershed, and how those samples should be scattered across the landscape to obtain more precise estimates of habitat indicators at the watershed scale, ISEMP looked at the spatial distribution of sites for habitat status and trend data collected in the Wenatchee subbasin from 2004 – 2010. For several habitat metrics deemed important to fish (e.g., large woody debris and fish cover), this dataset was subsampled multiple times with a range of sample sizes from five to 100. For each sample size, 1,000 subsamples were taken, and the mean and standard deviation of the habitat metric were estimated from each subsample. Sites with more than one sample (either within or across years) were only allowed to occur once within each subsample. Those estimates were then compared to determine the relationship between larger sample sizes and the precision of habitat indicators.

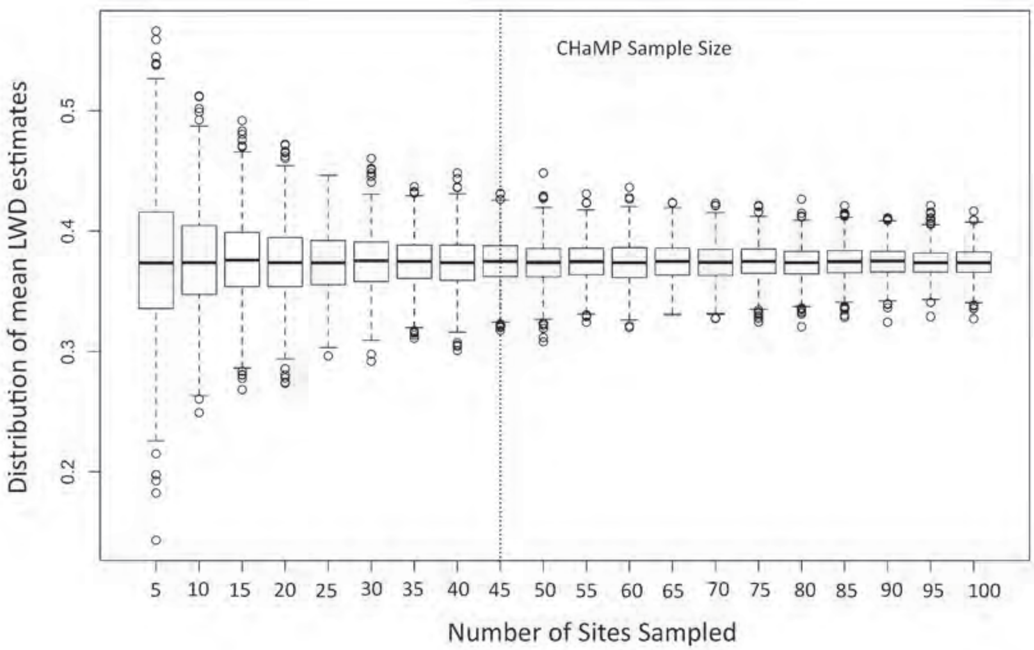
As expected, precision improved as sample size increased, but with diminishing returns (Figures 7 and 8). Once the sample size grew larger than 45, there was very little improvement in precision. This pattern holds true for the mean value within a watershed, or for measures of variability across the watershed, such as standard deviation or coefficient of variation. It was also consistent across different habitat metrics. These results were used to inform the survey design of CHaMP, which planned for a total of 45 sites over 3 years within each watershed.

Stratifying Sampling

Sampling designs can be made more powerful if they account for known sources of variation, such as differences in geomorphology and elevation in the landscape. ISEMP analyzed the Wenatchee and Lemhi status and trend dataset to determine how the variability of habitat metrics was explained based on several landscape characteristics. Several of them, including valley type, Strahler order, and ownership, partitioned the variance of habitat metrics quite well (Figures 9-11), while others such as watershed, did not (Figure 12). When fixed effects due to Strahler order and ownership were accounted for in the sample size analysis, the precision of fish cover (as measured by CV) improved by more than 20% (Figure 13). Because these characteristics are important in an analysis, they are also important to incorporate into a survey design to ensure appropriate contrast across these key factors. Once again, ISEMP utilized these results when developing the CHaMP survey design. This multivariate classification of habitat metrics indicated that a three part classification framework using a valley class geomorphic framework (based on work by Tim Beechie, Northwest Fisheries Science Center (Seattle, WA)) provided an acceptable level of site distinction that could be used as a stratification framework. As demonstrated by Figure 14, the CHaMP sampling design is more efficient and powerful through stratifying sample sites using a valley class geomorphic framework where the sites were allocated into three strata: Source, Transport, and Depositional. For three metrics, the valley class stratification accounted for 40% or more, and for eight metrics, 20% or more of the spatial variation. An analysis of variance indicates that there is a significant effect of the classification ($p < 0.05$) for most of the attributes, even though the proportion attributable to valley class might be low. The significance likely arises from the large sample size available for testing, allowing for detection of small differences in the mean between valley class types.

CHaMP

Based on what was learned from the ISEMP data collection using GRTS based sample design, CHaMP's basic design selected 45 sites to be sampled over a 9 year period, organized into four panels: an annual panel (15 sites to be monitored each year), and three panels each on a 3 year cycle with one panel starting in year 1 (10 sites), a second in year 2 (10 sites), and a third in year 3 (10 sites). After 3 years, all sites will have been sampled at least once. After 9 years, all sites will have been sampled for at least 3 years, which allows for an estimate of trend at all 45 sites.



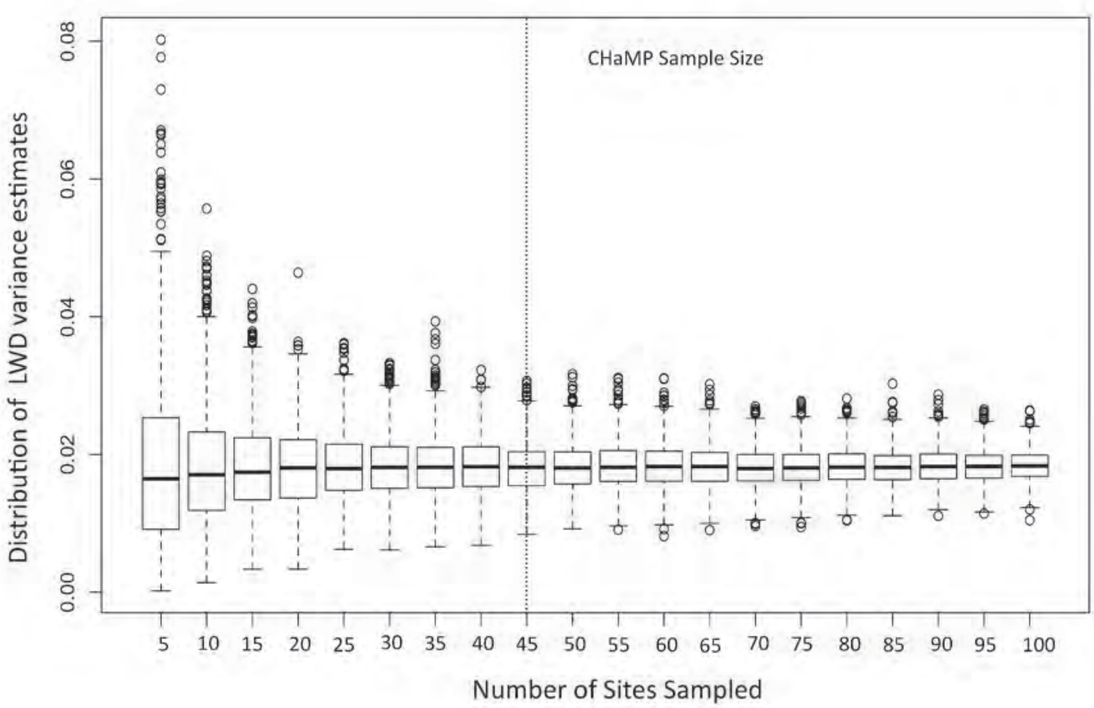


Figure 7. Box plot displaying the distribution of mean estimates of log(X+1) transformed Large Woody Debris volume/stream km (top panel) and the distribution of variance estimates of log(X+1) transformed Large Woody Debris volume/stream km (bottom panel) for the Wenatchee subbasin based on varying site sample sizes (5-100 sites; X axis). The dashed line indicates the annual site sample size for CHaMP.

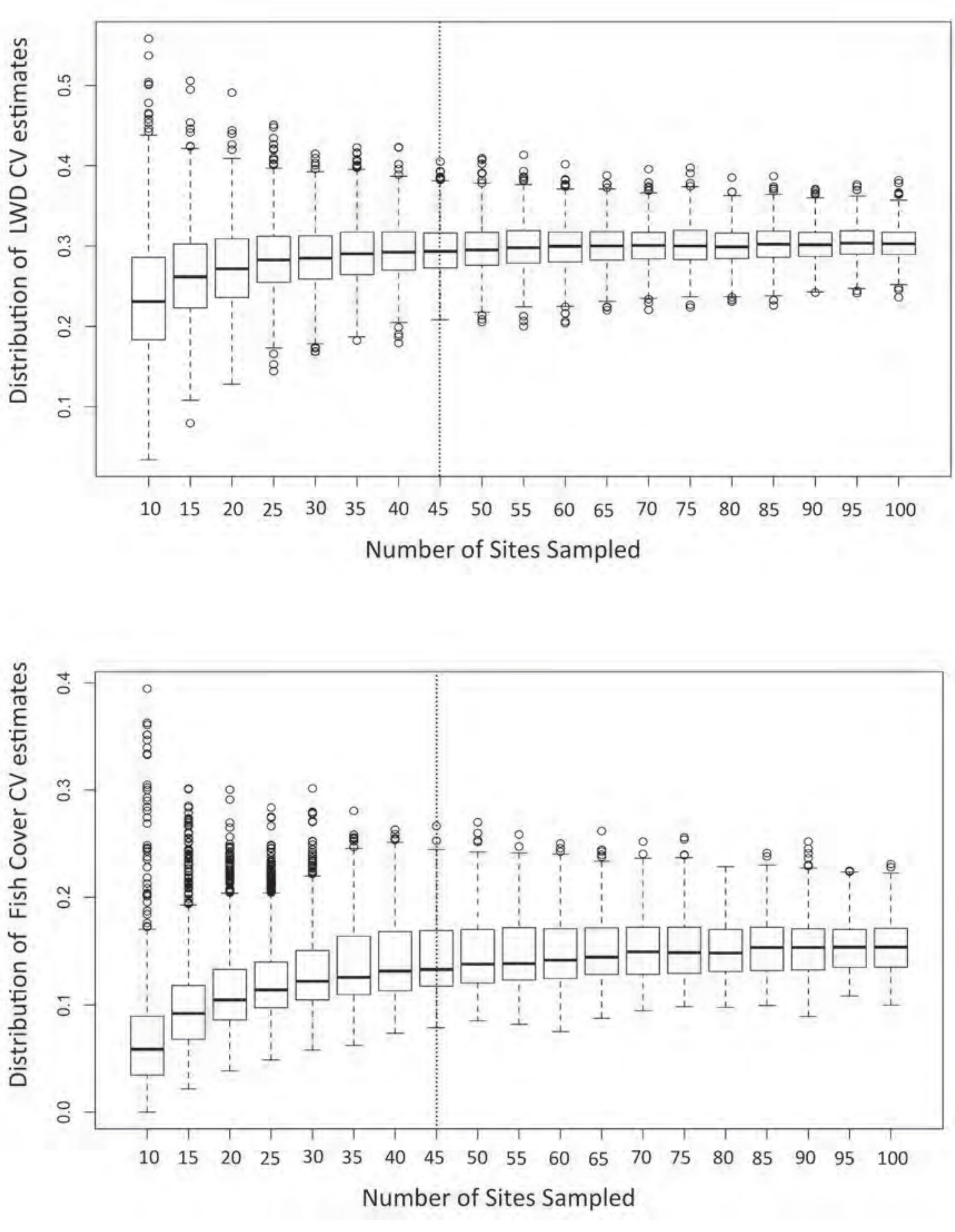


Figure 8. Box plot displaying the distribution of CV estimates of log(X+1) transformed Large Woody Debris volume/stream km for the Wenatchee subbasin (top panel) and distribution of CV estimates of square root transformed fish cover % by site estimates for the Wenatchee subbasin based on varying site sample sizes (5-100 sites; X axis). The dashed line indicates the annual site sample size for CHaMP.

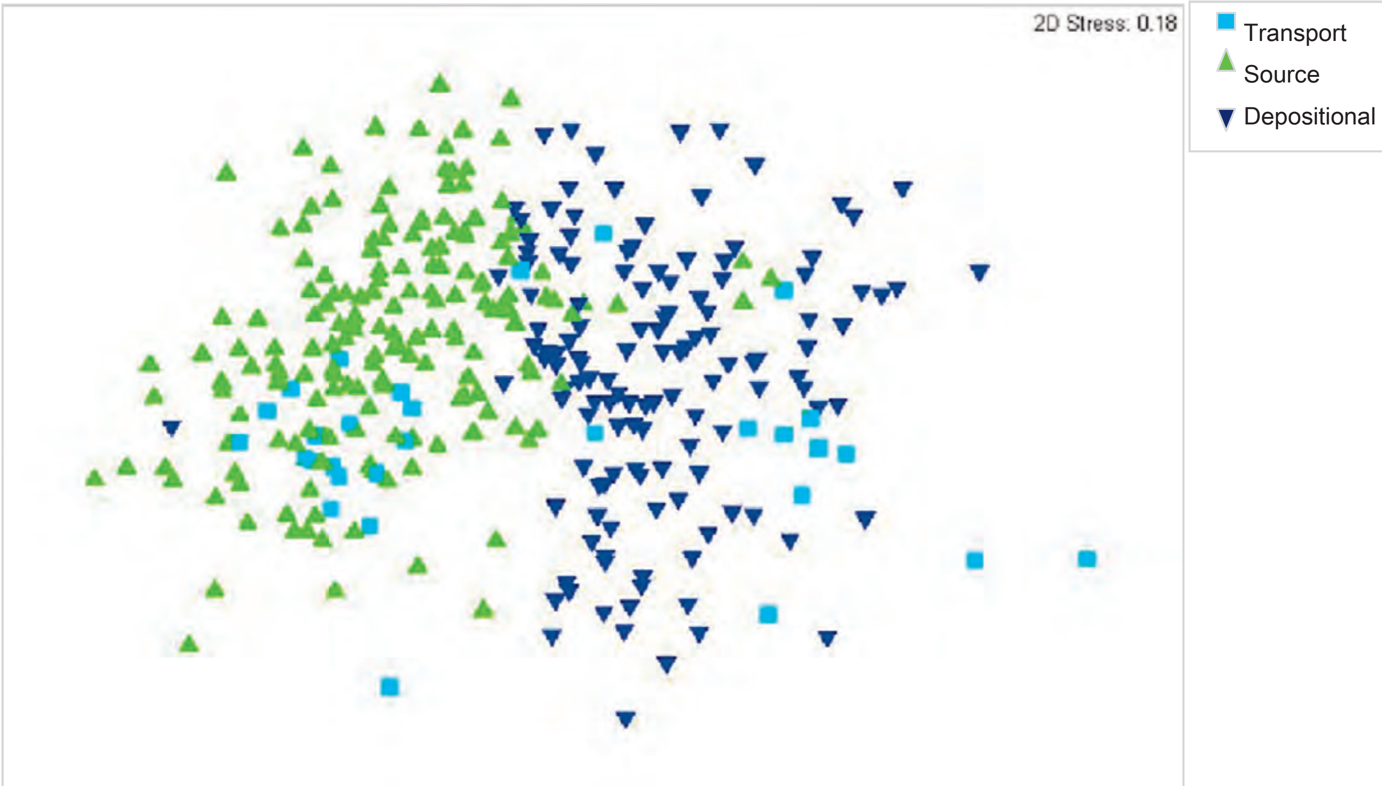


Figure 9. A multidimensional scaling plot using thirty habitat metrics in the Wenatchee subbasin. Each point represents one site. Sites closer to each other on the plot are considered more similar to each other. The colors and symbols correspond to three valley types.

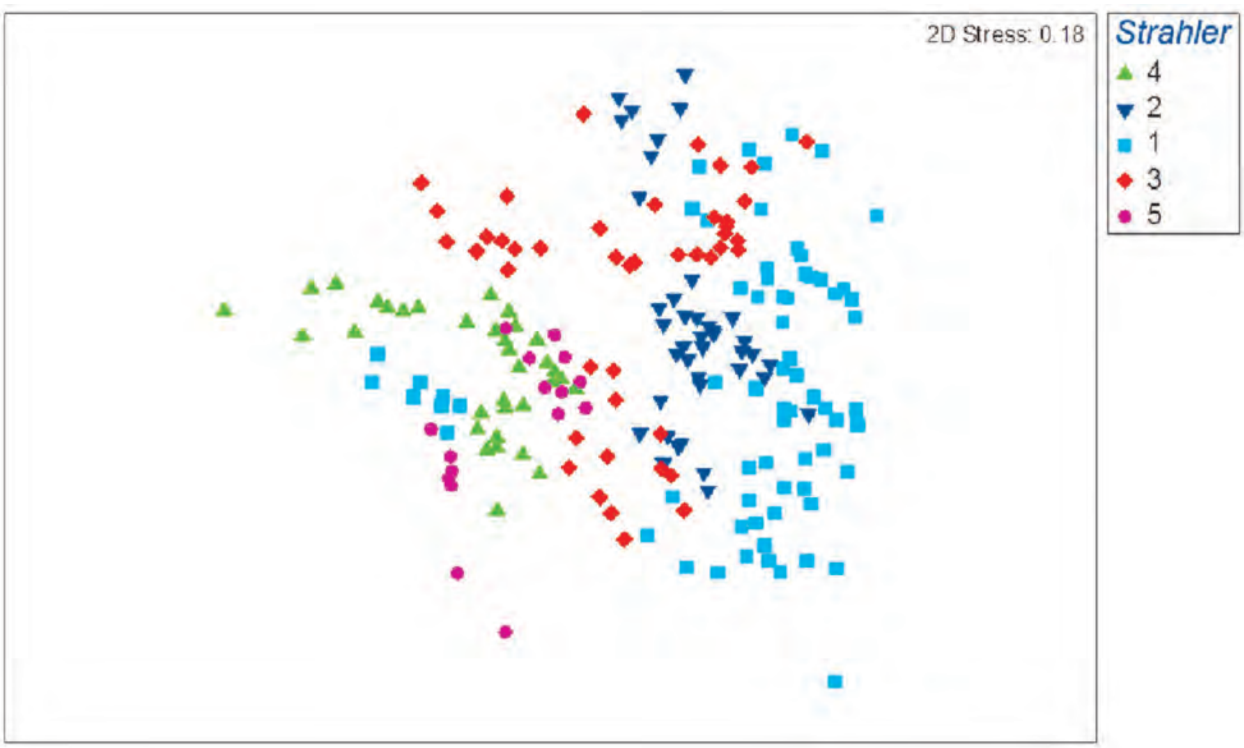


Figure 10. A multidimensional scaling plot using thirty habitat metrics in the Wenatchee subbasin. Each point represents one site. Sites closer to each other on the plot are considered more similar to each other. The colors and symbols correspond to Strahler order.

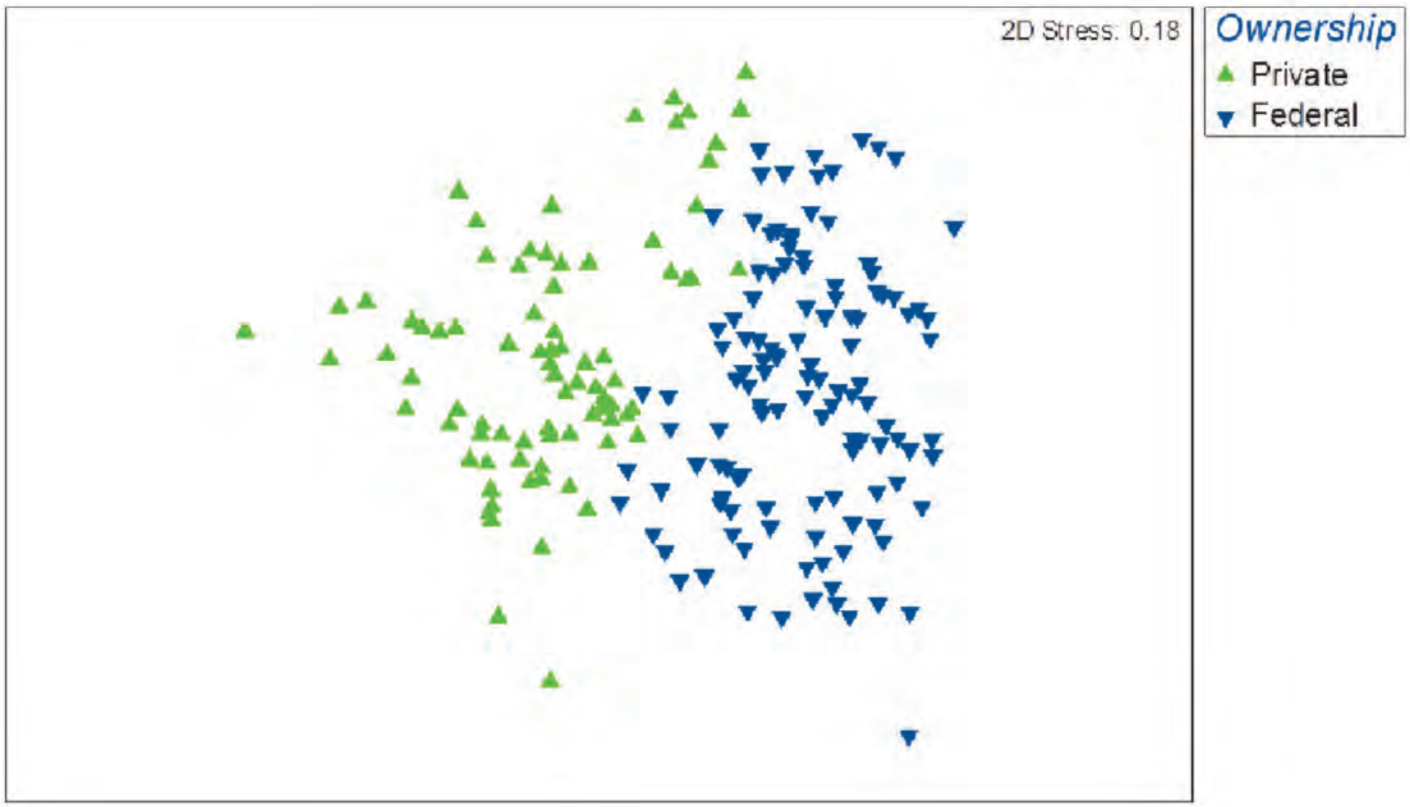


Figure 11. A multidimensional scaling plot using thirty habitat metrics in the Wenatchee subbasin. Each point represents one site. Sites closer to each other on the plot are considered more similar to each other. The colors and symbols correspond to two ownership classifications.

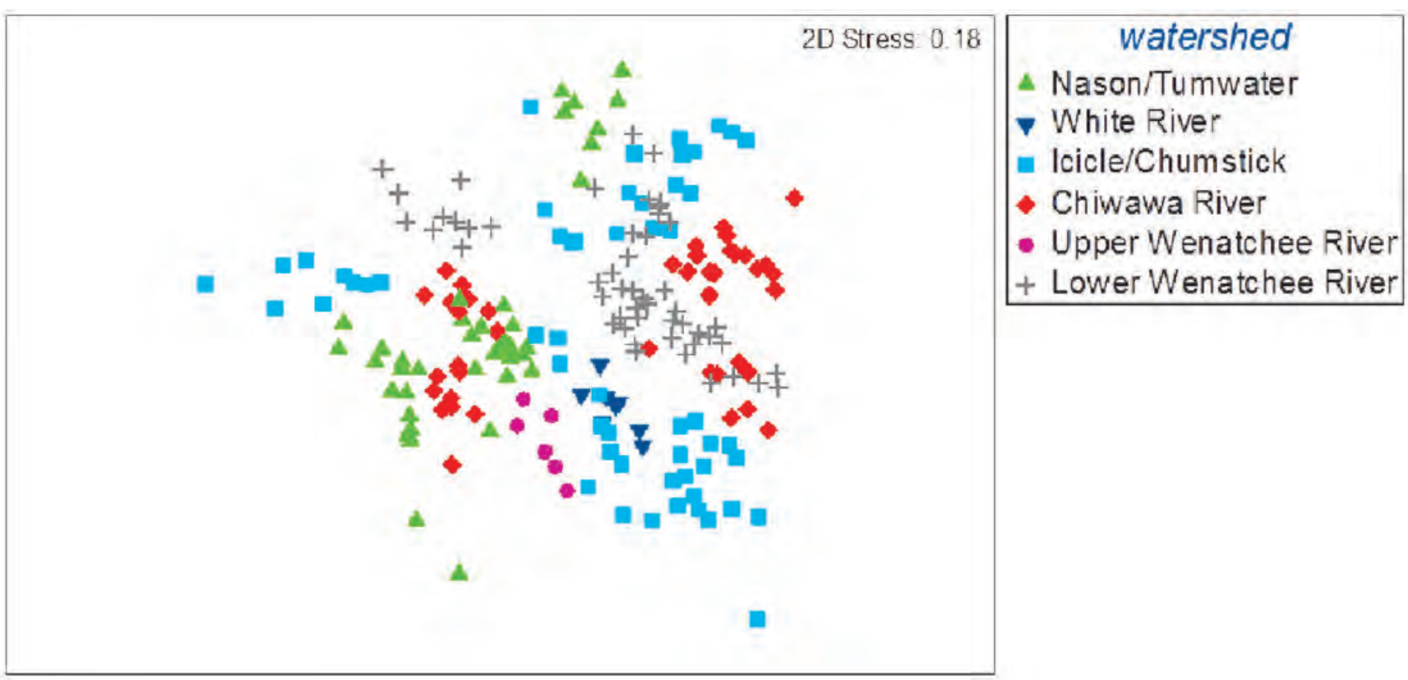


Figure 12. A multidimensional scaling plot using thirty habitat metrics in the Wenatchee subbasin. Each point represents one site. Sites closer to each other on the plot are considered more similar to each other. The colors and symbols correspond to different watersheds within the subbasin.

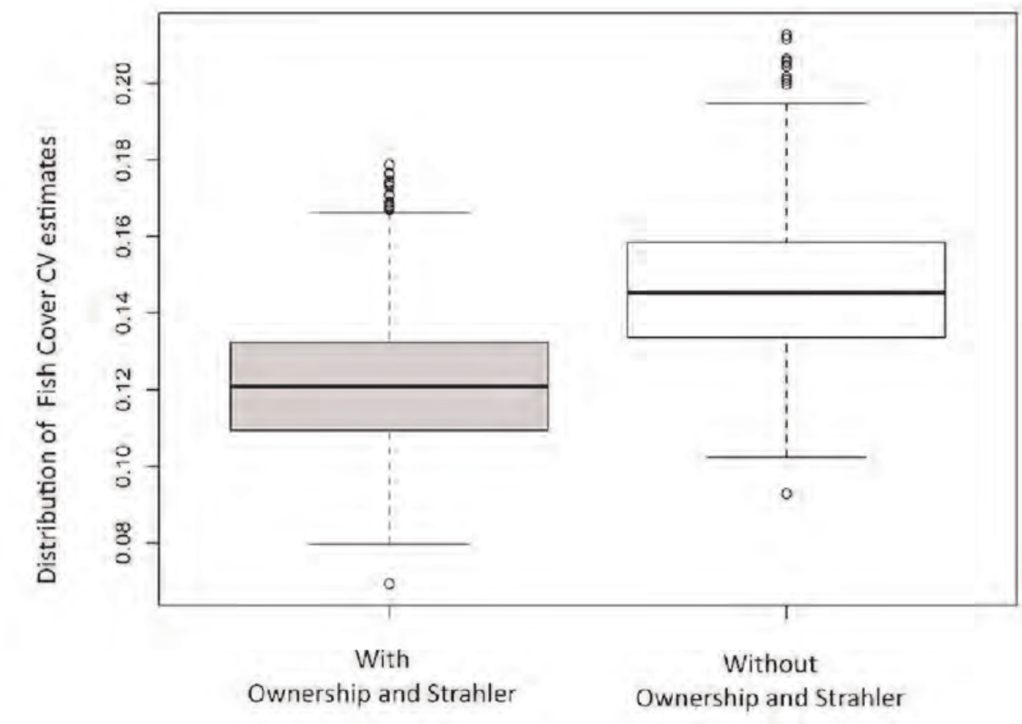


Figure 13. Coefficient of variation estimates for fish cover (FC) without accounting for fixed effects of Strahler order and ownership site classifications (white box), and coefficient of variation estimates when both fixed effects are accounted for (grey box). All model runs included fixed effects of the sampling year.

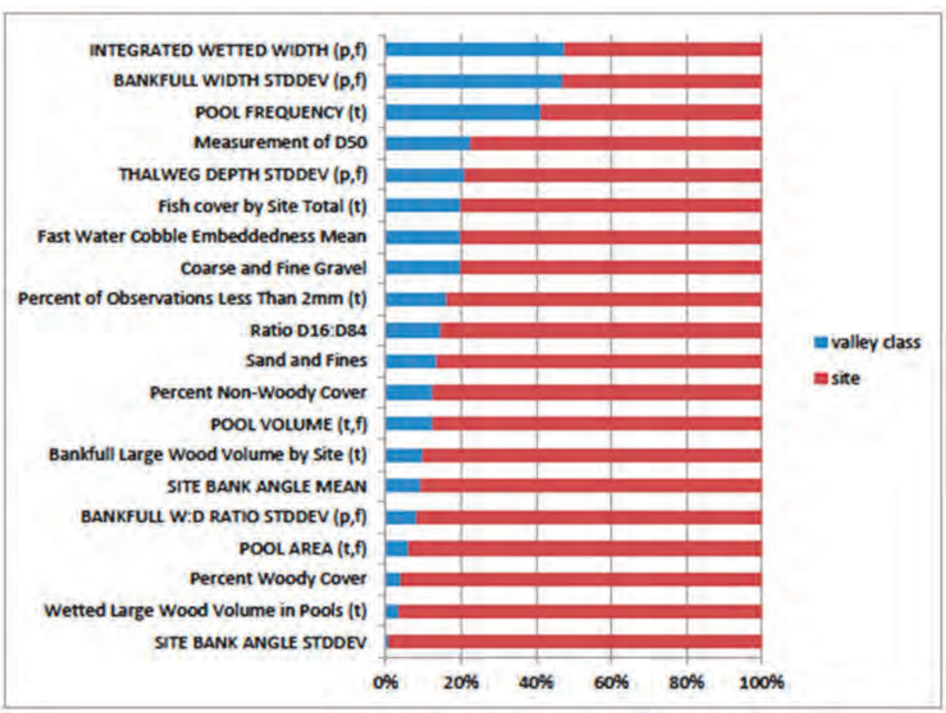


Figure 14. The relative proportion of site to site variation that is associated with the classification of sites into valley class (i.e., effectiveness of stratification).

Developing Rules for the Inclusion of Metrics in Monitoring Protocols

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. 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.

To be useful to managers, current monitoring activities must be able to see differences in key habitat indictors 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 indicators that can consistently quantify spatial and temporal patterns that arise from natural variation and human impacts – only then are these indicators useful for management purposes.

Measurement, Metrics and Indicators

Metrics and indicators are the units of information most useful and relevant to making inferences and decisions about the management of salmon habitat (NCEAS 2010) and are the common language among data collectors, scientists, and natural resource decision makers, even those involved in different monitoring programs. The nomenclature of **measurements, metrics, and indicators** that we use is derived from Stevens and Urquhart (2000) for numerical quantities as they pass in steps from data collected in the field into final processed parameter estimates, although there may be some overlap because some things called measurements can also be metrics or even indicators in the same study.

- A **measurement** is a value resulting from a field data collection event taken at a particular time and place. The field data collection protocols are described in the response design.
- **Metrics** are values resulting from the reduction or processing of measurements at a site or over a unit of time or space (i.e., metrics are site-scale values for the sampling period). The process of developing the metrics is also described in the response design.
- An **indicator** is the value resulting from the processing of metrics across sites or across time and are population-scale values for the sampling period. The methods for calculating the indicators are described in the inference design.

Metric and Indicator Inclusion Rule Set

ISEMP has developed a rule set to evaluate which metrics and indicators should be included in a fish habitat monitoring protocol, in this case the CHaMP protocol. The habitat quality and quantity indicators in the CHaMP protocol have been designed specifically to evaluate the features of stream habitat critical to juvenile salmonid growth and survival from egg to smolt life stages.

The methods in the CHaMP protocol were developed in a step-wise process that included literature reviews, field testing, and data analysis founded on the results of work conducted over the last two decades by groups like ISEMP (in pilot Columbia River subbasins since 2003); the U.S. Forest Service (e.g., the PIBO habitat sampling program begun in 1998, AREMP since 2001, and protocol comparison studies such as Roper et al (2002, 2008, 2010)); EPA's EMAP program (started in 1990); the Oregon Department of Fish and Wildlife (habitat programs since 1998); and work by other agencies like the Washington Department of Ecology and Oregon Department of Environmental Quality. An assessment of the applicability of commonly used attributes in stream habitat monitoring protocols (Bouwes et al. 2010) reviewed fish habitat requirements in the context of stream habitat attributes and geomorphic processes, assessed whether existing habitat protocols provided information that relates the quality of stream habitat to fish production, and developed a draft habitat monitoring protocol for projects that support the FCRPS and salmonid recovery planning.

The methods and approaches described in Bouwes et al (2010) were field tested by ISEMP during the summer of 2010 in the Bridge and Asotin IMWs and Lemhi River, and further revisions based on the results of field evaluations in the fall of 2010 in the South Fork Salmon River and Bridge and Asotin IMWs. At the same time, data collected by ISEMP in the Wenatchee and Entiat subbasins since 2004 were analyzed. Results from both field testing and data analyses were used to further refine the list of metrics and indicators included in CHaMP.

Lastly, ISEMP evaluated all possible metrics and indicators envisioned during the CHaMP protocol development process, or that were documented in other protocols, so that a measurement and related methodology was included in the CHaMP protocol if, and only if, it would be used to calculate a metric that met each of the following three rules:

1) **Information Content**: Habitat metrics and indicators must provide information directly related to salmonid productivity, including survival and growth, as documented by peer reviewed literature, modeling, or existing data analysis.
2) **Data Form**: Habitat metrics and indicators must provide statistical information with robust data quality. The data generated for a prospective metric must be repeatable, detect heterogeneity, and have adequate properties for modeling/ statistics (e.g., variance distributions

must meet statistical assumptions for modeling or testing).

3) **Feasibility**: Habitat metrics and indicators need to be generated by field tools or software that are readily implementable as of the time field testing in fall 2010 (i.e., does not rely on future technological advances). Feasibility is also bounded by the need to fit all survey work within a three-person-day field survey at 80-90 percent of all sites likely to be encountered.

Table 6 shows some examples of the final set of 14 CHaMP indicators, the metrics from which they are derived, as well as a brief outline of the inference design underlying each indicator. Table 7 includes the full list of metrics that were included in the CHaMP habitat protocol, and Table 8 shows the indicators and metrics that were not included in the CHaMP protocol. For example, bank stability was found to have low information content and poor data format, and fish cover (by bryophytes, macrophytes, filamentous algae, and small woody debris), was not included as individual metrics because these elements of fish cover have been found to have low information content and poor data format, although the CHaMP protocol rolls some of these metrics into larger fish cover categories. Dissolved oxygen was also determined to have low feasibility since it is inappropriate as a point measure and is too expensive to sample continuously at all sites.

Table 6. Three examples from the 22 indicators used by CHaMP and the inference design underlying each indicator. This table is a subset of Table 6.

Indicator	Units	Inference Domain	Inference Design	Inference Methods	Metrics	Indicator Generation Process	Software	Fish Response Category	Life Stage
Embeddedness of Fast Water Cobble	Percent	Valley type nested in survey frame	Mean, variance over inference domain, annually	Design-based	Average of site embeddedness measurements	Estimated annually for valley type nested in the survey frame with sampling design-based algorithm for riffle cobble embeddedness.	SP Survey	Survival	Eggs/Alevin
Pool Frequency	Count per meter	Valley type nested in survey frame	Mean, variance over inference domain, annually	Design-based	Site measurement of pool frequency	Estimated annually for valley type nested in the survey frame with sampling design-based algorithm for pool frequency.	SP Survey, River Bathymetry Toolkit	Growth	Parr to smolt
Channel Unit Complexity	Index	Valley type nested in survey frame	Mean, variance over inference domain, annually	Design-based	Site measurements of channel unit volume, LWD, and substrate	Estimated annually for valley type nested in the survey frame with sampling design-based algorithm for residual pool depth, pool tail fines and wood volume. A multivariate measure of channel unit complexity, similar to DSM approach applied by AREMP and PIBO to habitat metrics to capture complexity.	SP Survey, River Bathymetry Toolkit	Growth	Parr to smolt

Table 7. Indicators that ISEMP recommends including in a habitat status and trends monitoring protocol.

Indicator	Units	Metrics	Fish Response Category	Life Stage
Average Alkalinity	Milli-equivalent per liter	Site measurement of alkalinity	Survival	Parr to smolt
Average Conductivity	Micro-Siemens per meter	Site measurement of conductivity	Survival	Parr to smolt
Average pH	pH	Site measurement of pH	Survival	Parr to smolt
Growth Potential	Degree grams	Site measurement of drift biomass and temperature	Growth	Parr to smolt
Percent Below Summer Temperature Threshold	Percent	Year-round temperature logger data from sites	Growth	Parr to smolt
Percent Above Winter Temperature Threshold	Percent	Year-round temperature logger data from sites	Growth	Parr to smolt
Velocity Heterogeneity	Index	Modeled velocity heterogeneity at a site	Growth	Parr to smolt
Embeddedness of Fast water Cobble	Percent	Average of site embeddedness measurements	Survival	Eggs/Alevin
Pool Frequency	Count per meter	Site measurement of pool frequency	Growth	Parr to smolt
Channel Complexity	Index	Site measurements of depth, width, and thalweg sinuosity	Growth	Parr to smolt
Channel Score	Index	Site measurements of channel unit volume, LWD, and substrate	Growth	Parr to smolt
Residual Pool Volume	Cubic meter	Site measurement of residual pool volume	Growth	Parr to smolt
Pool Tail Fines	Percent	Site measurement of pool tail fines	Survival	Eggs/Alevin
Total Drift Biomass	Gram per square meter	Site measurement of total drift biomass	Growth	Parr to smolt
Bank Angle	Percent	Site measurement of bank angle	Growth	Parr to smolt
LWD Volume	Cubic meter	Site measurement of LWD Volume	Growth	Parr to smolt
Fish Cover	Percent cover	Site measurement of fish cover	Survival	Parr to smolt
Channel Unit Volume	Cubic meter	Site measurement of volume (DEM, photos, site map) and channel unit type	Growth	Parr to smolt
Channel Unit Complexity	Index	Site measurements of channel unit volume, LWD, and substrate	Growth	Parr to smolt
Riffle Particle Size (D_{16}, D_{50}, D_{84})	Millimeter	Site measurement of D_{50}, D_{16}, D_{84}	Survival	Eggs/Alevin
Riparian Structure	Kilometer by type	Site measurement of riparian structure	Growth	Parr to smolt
Solar Input	Degree day	Site measurement of solar input	Growth	Parr to smolt

Table 8. Indicators that ISEMP would not recommend including in a habitat status and trends monitoring protocol.

Indicator/Metric	Reason Not Included In CHaMP	Reference
Bank Stability	Low information content, poor data format	See CHaMP protocol Appendix D (Bouwes et al. 2012) for details.
Fish Cover by Bryophytes, Macrophytes, Filamentous Algae, Small Woody Debris	These elements of fish cover have been found to have low information content and poor data format as individual metrics. CHaMP rolls some of these metrics into larger fish cover categories. See Section 8.2.	CHaMP protocol Appendix D conclusions were modified by ISEMP analysis of Upper Columbia habitat data.
Benthic Macroinvertebrate metrics	Derived metrics are indirectly related to fish (it is informative but for water quality)	See CHaMP protocol Appendix D for details.
Dissolved Oxygen	Low feasibility (inappropriate as a point measure and too expensive to sample continuously at all sites.)	--
Nutrients - Chemistry	Low information content, low feasibility	See CHaMP protocol Appendix D for details.
Turbidity	Low information content, poor data format, low feasibility	See CHaMP protocol Appendix D for details.
Rosgen Channel Classification	CHaMP will use Montgomery and Buffington (1993) to classify channels. While all channel classification systems have Low Information Content, Montgomery and Buffington (1993) fits within the hierarchical geomorphological approach of CHaMP better than the Rosgen(1994) channel classification system.	--
Pesticides	Low feasibility	--
Heavy Metals	Low feasibility	--

III. MONITORING STATUS AND TRENDS OF FISH POPULATIONS

Monitoring Adult Escapement

Ultimately, evaluations of the effectiveness of actions proposed in the BiOp are gauged by the …"survival prong (24-year extinction risk) and the recovery potential prong (average returns-per-spawner, median population growth rate, and abundance trend) of the jeopardy standard" (NOAA 2008). Each of these indicators (24-year extinction risk, returns-per-spawner, median population growth rate, and abundance trend) as defined in the BiOp rely on estimates of adult escapement. In the case of returns-per-spawner, age structure is required to identify the brood year of origin. Estimating the number of adults entering a population can be challenging. In the Columbia River basin, redd counts are commonly used as an "index" of abundance, but are accompanied by substantial uncertainty contributed by both sampling error (failure to detect redds and/or counting false redds) and expansion error (i.e., assumptions about the number of adults represented by a given redd). Although standardized redd count data exist (e.g., for spring/summer Chinook salmon across much of the Snake Basin (Hassemer 1993)), there remain questions about the statistical reliability of many commonly used redd count protocols (Parsons and Skalski 2009), underscoring the need to evaluate

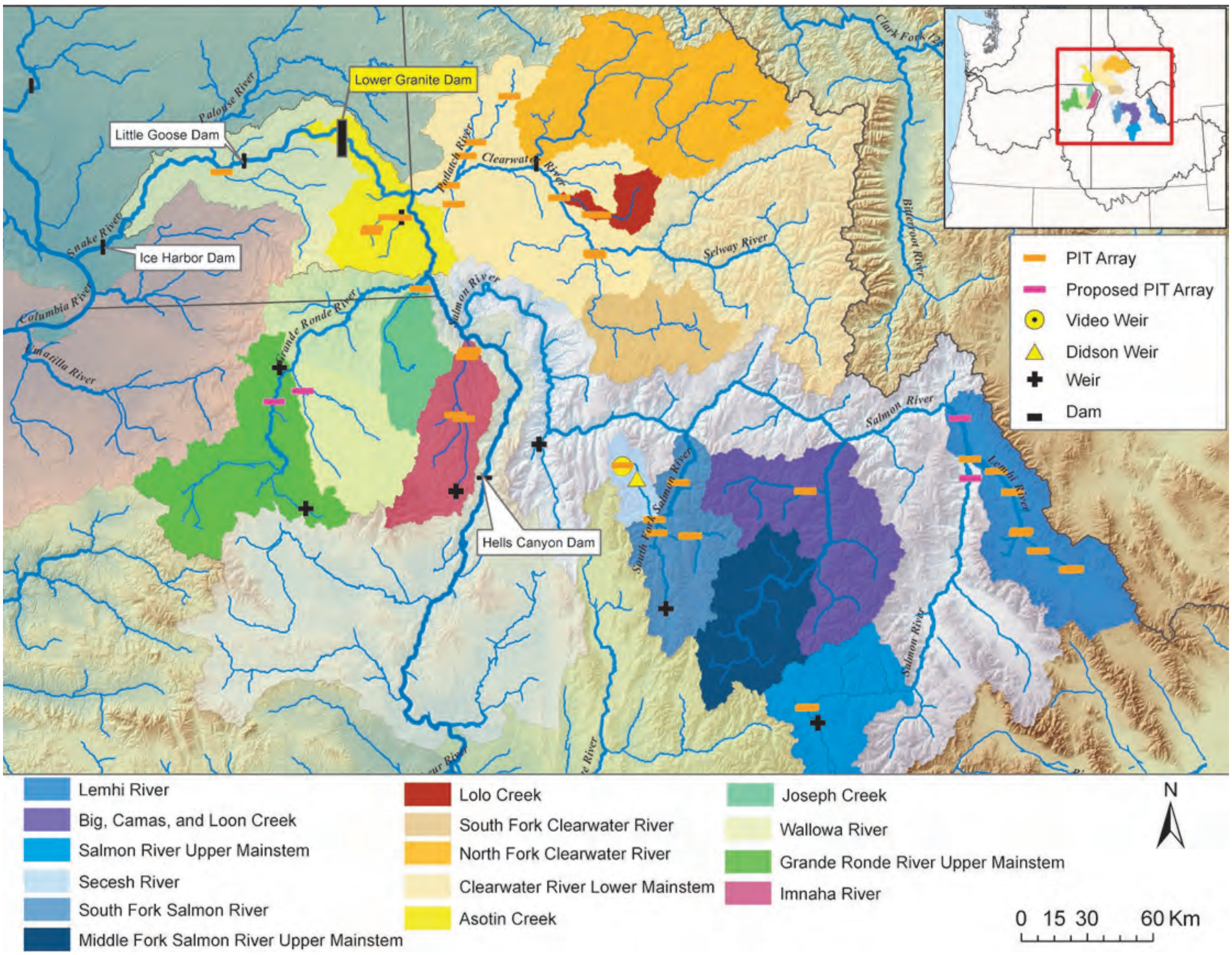


Figure 15. Location of PIT tag arrays operated by ISEMP relative to the population boundaries of Snake River steelhead populations for which escapement estimates are generated by Lower Granite Dam run decomposition.

the precision and potential bias of redd counts as an index of escapement (Crawford and Rumsey 2011).

Salmon River Subbasin: Estimating Adult Salmonid Escapement Using Instream PIT Tag Arrays (Lower Granite Dam Run Decomposition)

Redd surveys are not effective for steelhead across the majority of the Snake Basin, owing to the fact that high spring flows preclude observation. As a result, information on the distribution and abundance of A and B-run steelhead is a critical uncertainty, leading to the development of RPA 50.5, which requires "additional status monitoring to ensure a majority of Snake River B-run steelhead are being monitored for population productivity and abundance" (http://www.cbfish.org/FcrpsBiOp.mvc/Summary/50/5).

In the Salmon subbasin ISEMP was tasked with developing status and trends monitoring programs and habitat restoration action effectiveness monitoring programs for spring/summer Chinook salmon and steelhead residing in the South Fork Salmon River (SFSR) and Lemhi River. Each of the three populations of spring/summer Chinook salmon in the SFSR and the spring/summer Chinook salmon population in the Lemhi had existing programs that generated adult abundance estimates, using either redd counts, direct observations, or mark/recapture. Neither of the two populations of steelhead in the SFSR nor the steelhead population in the Lemhi had existing escapement estimates. The ISEMP watershed model (described in Appendix—Chapter 3) requires adult escapement estimates for steelhead and spring/summer Chinook salmon with accompanying estimates of uncertainty. To generate these estimates ISEMP initiated a mark/recapture program that PIT

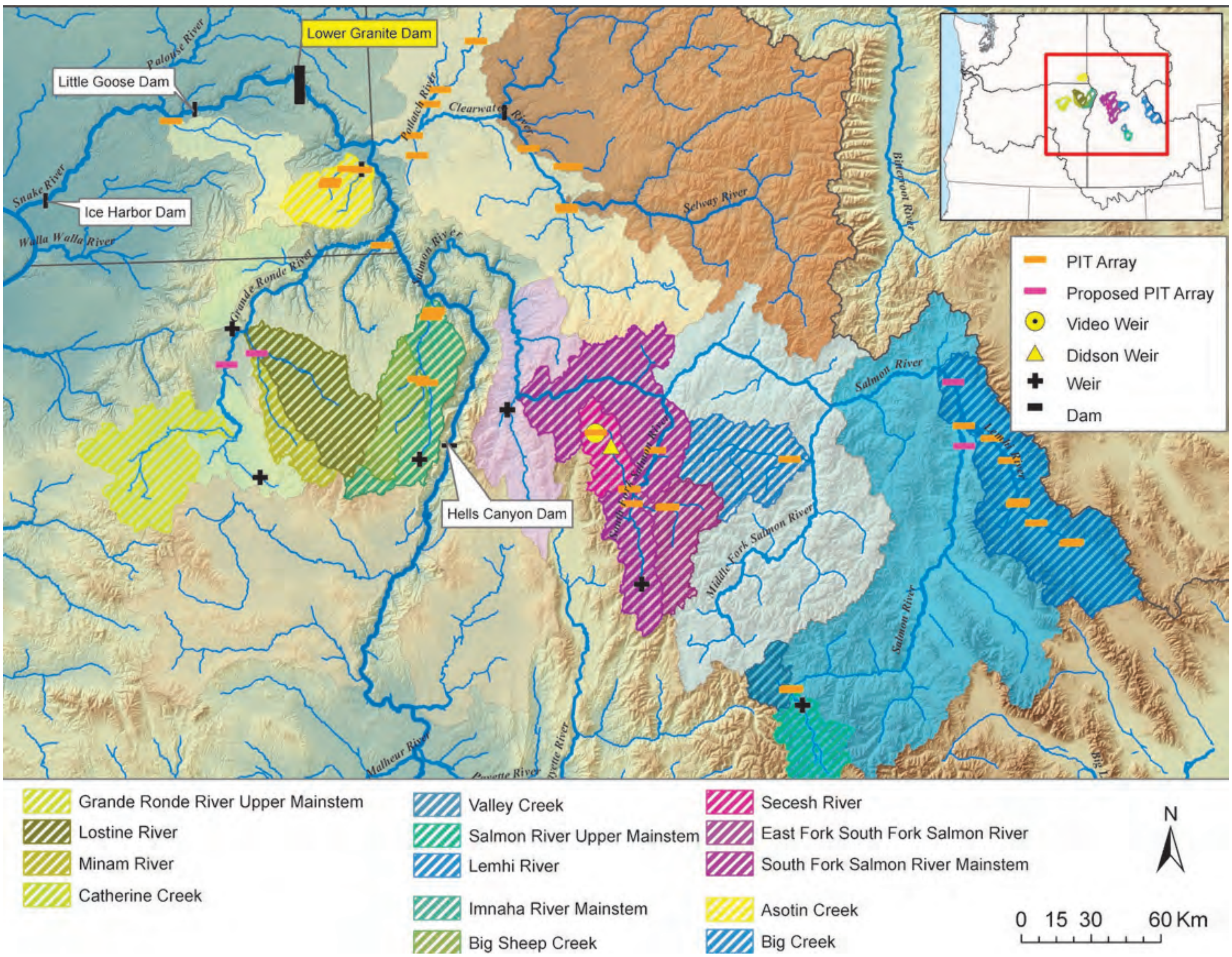


Figure 16. Location of PIT tag arrays operated by ISEMP relative to the population boundaries of Snake River spring/summer Chinook salmon populations for which escapement estimates are generated by Lower Granite Dam run decomposition.

Table 9. Steelhead run year, Major Population Group (MPG), population, subpopulation fraction of population sampled, escapement estimate, coefficient of variation (CV), and independent estimate (if available) monitored by ISEMP PIT tag arrays in the Snake River Basin. Shaded rows identify opportunistic independent estimates of escapement, primarily comprising locations where PIT tag wands are utilized to interrogate PIT tags.

Run-Year	MPG	Population	Subpopulation	PIT Tag Decomposition				Independent Estimate		
				Fraction Sampled[1]	Escapement	CV	95% CI	Escapement	CV	95% CI
2009-2010	Lower Salmon	Asotin Creek[2]		95%	1,687	8.5%	± 280	1,500	N/A	N/A
2009-2010	Salmon River	South Fork		90%	1,497	9.1%	± 268			
2009-2010		Secesh		100%	298	22.1%	± 129			
2009-2010		Middle Fork	Big Creek	100%	753	21.8%	± 322			
2009-2010		Upper Salmon	Valley Creek	100%	237	17.7%	± 82			
2009-2010		Upper Salmon	Pahsimeroi River[2]	100%	138	22.9%	± 62	115	N/A	N/A
2009-2010		Lemhi River		95%	630	14.2%	± 175			
2009-2010		Little Salmon	Rapid River[2]	95%	136	24.0%	± 64	150	Census	Census
2009-2010	Clearwater River	Lochsa River	Fish Creek[2]	100%	246	58.1%	± 117	205		164-255
2010-2011	Lower Salmon	Asotin Creek[2]		95%	890	10.0%	± 175	1,128	2.0%	± 44
2010-2011	Grande Ronde	Joseph Creek[3]		100%	1,627	1.4%	± 45	1,698	22.4%	± 744
2010-2011	Imnaha River	Imnaha River		100%	3,298	1.5%	± 97			
2010-2011		Imnaha River	Cow Creek	100%	147	1.4%	± 4			
2010-2011		Imnaha River	Big Sheep Creek	100%	765	2.2%	± 33			
2010-2011	Salmon River	South Fork		90%	2,540	1.9%	± 93			
2010-2011		Secesh		100%	397	3.1%	± 24			
2010-2011		Middle Fork	Big Creek	100%	687	1.6%	± 22			
2010-2011		Upper Salmon	Valley Creek	100%	232	1.5%	± 7			
2010-2011		Lemhi River		95%	428	1.7%	± 14			

[1]Fraction sampled refers to the fraction of spawning believed to occur above PIT tag arrays.
[2]Weirs that capture and enumerate steelhead and use handheld wands to identify PIT tags, but do not have PIT tag arrays.
[3]Locations with weirs that capture and enumerate steelhead and use handheld wands to identify PIT tags and also have neighboring PIT tag arrays.
[4]Independent estimate generated from a video weir paired with a single PIT tag array.

tags a known, representative fraction of natural origin adult steelhead and spring/summer Chinook salmon as they pass Lower Granite Dam and uses subsequent detections of PIT tagged adults at instream PIT tag arrays as recaptures. The resulting data provides an estimate of total natural origin adult escapement with an estimate of precision. Based on the initial success of this method, additional PIT tag arrays were funded under ISEMP through the "fast-track" proposal (Figures 15 and 16).

Notably, ISEMP PIT tagging at Lower Granite Dam is coordinated with BPA Project 2010-026-00, which uses genetic techniques to assign natural origin adult spring/summer Chinook salmon and steelhead to a reporting group of origin. Approximately half of the 4,000 natural origin spring/summer Chinook salmon and 4,000 natural origin steelhead targeted for ISEMP PIT tagging are genotyped by this project. The coordination of these two sampling efforts both reduces total fish handling and enables a side-by-side comparison of the efficacy of the two methods for generating population, major population group (MPG), and distinct population segment (DPS) adult escapement estimates. Genetic analysis of the samples enables estimates of gender, allowing the resulting estimates of escapement to be partitioned into male and female components. Additionally, the two projects share the cost of aging scales, allowing estimation of escapement by age, which is necessary to calculate returns-per-spawner as described in the BiOp.

Additional information describing the statistical methods underlying the PIT tag based run decomposition are

[2]The Collaborative Anadromous Workgroup includes tribal, state, and federal agencies with Columbia River Basin Management jurisdiction. Tables were compiled by this group to provide the implementation strategies guiding the development of a monitoring strategy for salmon and steelhead focused on Viable Salmonid Population parameters, tributary habitat effectiveness and hatchery effectiveness monitoring across the Columbia Basin. The November 2009 draft of the document describing this coordinated anadromous monitoring strategy is located at www.nwcouncil.org/dropbox.

Table 10. Spring/summer Chinook salmon run year, Major Population Group (MPG), population, fraction of population sampled, escapement estimate, coefficient of variation (CV), and independent estimate (if available) monitored by ISEMP PIT tag arrays in the Snake River Basin.

Run-Year	MPG	Population	PIT Tag Decomposition				Independent Estimate		
			Fraction Sampled[1]	Escapement	CV	95% CI	Escapement	CV	95% CI
2010	South Fork Salmon	Mainstem	100%	4,671	3.7%	± 340			
2010		Secesh	100%	1,308	5.6%	± 143			
2010		East Fork	100%	1,026	0.5%	± 11	1,032	N/A	N/A
2010	Middle Fork	Big Creek	100%	285	24.2%	± 135			
2010	Upper Salmon	Valley Creek	100%	235	9.6%	± 44			
2010		Lemhi	95%	262	3.7%	± 19			
2011	Grande Ronde/	Imnaha	100%	2,421	6.3%	± 297			
2011	South Fork Salmon	Mainstem	100%	3,318	6.5%	± 423			
2011		Secesh	100%	779	2.2%	± 34			
2011		*Lake Creek*[2]	100%	198	N/A	N/A	203	10.6%	± 42
2011		East Fork	100%	652	0.2%	± 3			
2011	Middle Fork	Big Creek	100%	449	18.1%	± 159			
2011	Upper Salmon	Valley Creek	100%	460	8.9%	± 80			
2011		Lemhi	95%	337	16.2%	± 107			

[1]Fraction sampled refers to the fraction of total spawning believed to occur above the PIT tag array.

available in Chapter 2 of the Appendix.

Tables 9 and 10 report the escapement estimates for steelhead and spring/summer Chinook salmon using the simple mark/recapture methods described above. These escapement estimates are intended to assist in satisfying the intent of RPA 50, which requires abundance monitoring for at least one population per MPG for spring/summer Chinook salmon and abundance monitoring for a majority of B-run steelhead populations. To be clear, the number and placement of ISEMP PIT tag arrays are insufficient to fully address the intent of RPA 50. Rather, the placement of PIT tag arrays funded through ISEMP was motivated by information needs of the ISEMP watershed model and, through the "fast-track" proposal process to fill gaps in information identified by the Collaborative Anadromous workgroup. Also reported are locations (e.g., Fish Creek) that allow a comparison of escapement estimates generated via PIT tagging against independent estimates generated by counts of adults and/or mark recapture. These comparisons are provided as a means to assess the efficacy of the ISEMP PIT tagging approach to generate the adult abundance estimates required by the BiOp.

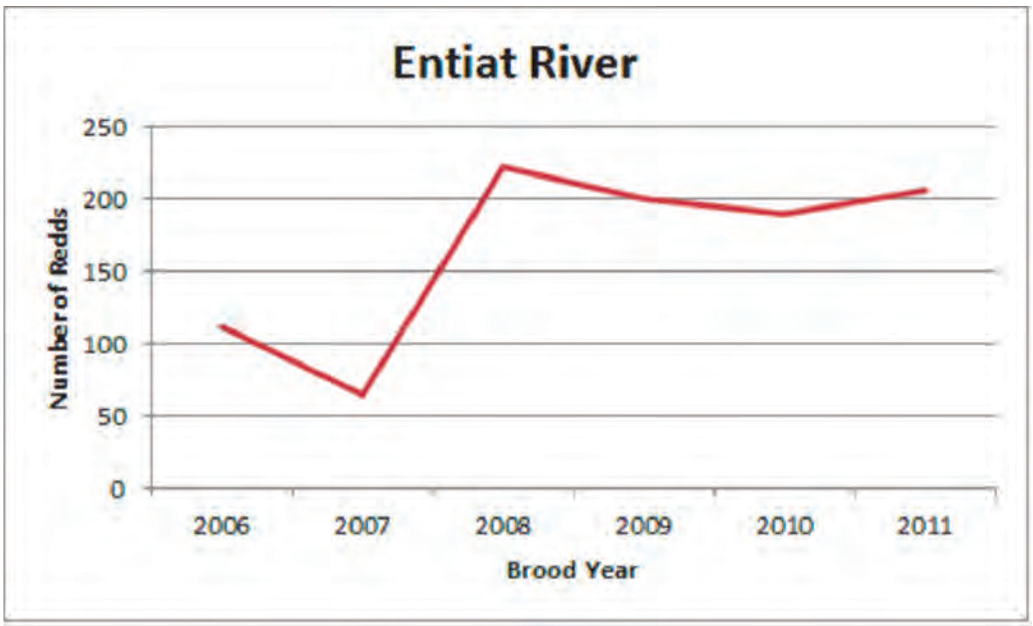
As demonstrated above, the decomposition of the Lower Granite Dam runs-at-large of steelhead and spring/summer Chinook salmon into tributary, population, and MPG specific escapement estimates is a reliable, precise and efficient alternative to continuous operation of multiple weirs. Additionally, adult capture and PIT tagging at Lower Granite Dam has not been accompanied by any direct mortality to date, suggesting that handling stress may be minimal at this location relative to upstream weirs. Finally, there is the potential to expand PIT tagging at Lower Granite Dam to include hatchery origin adults. Pursuing that option would enable estimates of hatchery fraction in populations targeted for supplementation and enable estimates of stray rates into non-target populations that are monitored by PIT tag arrays.

Upper Columbia Basin: Steelhead Redd Surveys in the Entiat River Watershed

In the Upper Columbia steelhead redd surveys are used to track the annual spawning success of adults returning to the Wenatchee and Entiat Rivers (Figure 17), map the distribution of steelhead redds (allowing for the evaluation of historic spawning areas and habitat restoration actions) and are used when calculating annual estimates of juvenile productivity. Surveys are conducted by raft and on foot on a weekly basis as weather and stream conditions permit, although some differences exist among watersheds. In the Wenatchee River and its major tributaries the WDFW surveys non-random index areas, defined as major spawning area(s) for each stream. The USFS-Entiat Ranger District used index surveys through 2011 in the Mad River (the main tributary to the Entiat River) but will implement a census starting in 2012. The USFWS conducts a census from river mile 34 to the mouth of the Entiat mainstem.

As discussed before, there are many issues with redd surveys and ISEMP is working to improve traditional redd surveys and develop alternative methods for estimating adult escapement.

Managers in the Upper Columbia are working with ISEMP to test the run decomposition approach that ISEMP has developed in the Salmon subbasin, starting in 2012. Nine instream PIT tag detection arrays in the Wenatchee River subbasin and six in the Entiat River subbasin (Figure 18), designed and installed in cooperation with the WDFW, NOAA-



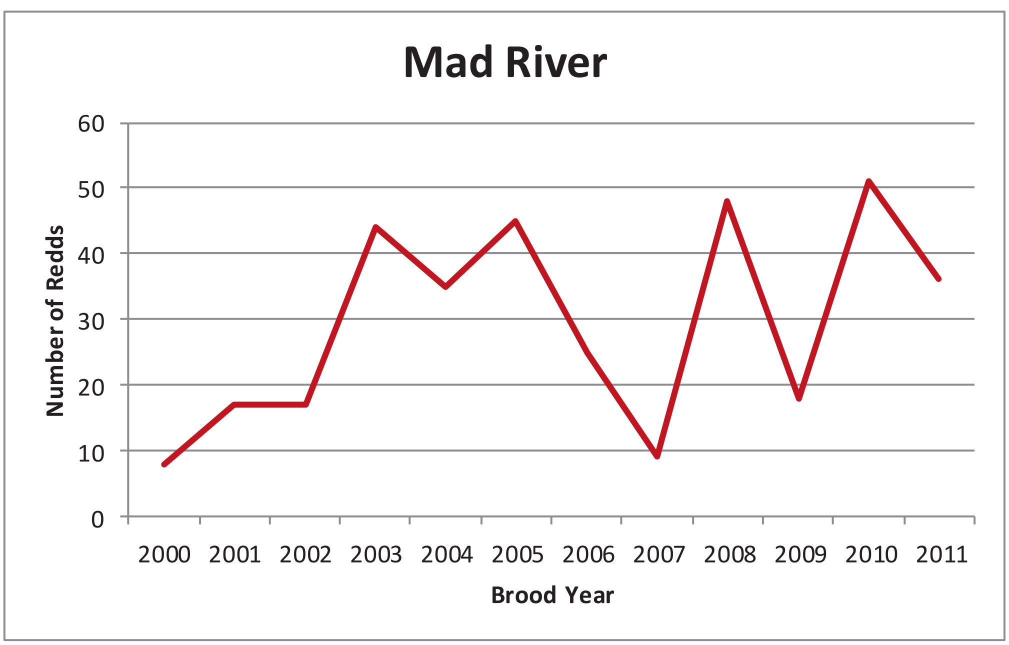


Figure 17. Steelhead redd survey data from the Entiat and Mad Rivers. A weekly census is conducted on the mainstem Entiat by U.S. Fish and Wildlife Service. The U.S. Forest Service Entiat Ranger District conducted weekly index surveys through 2011.

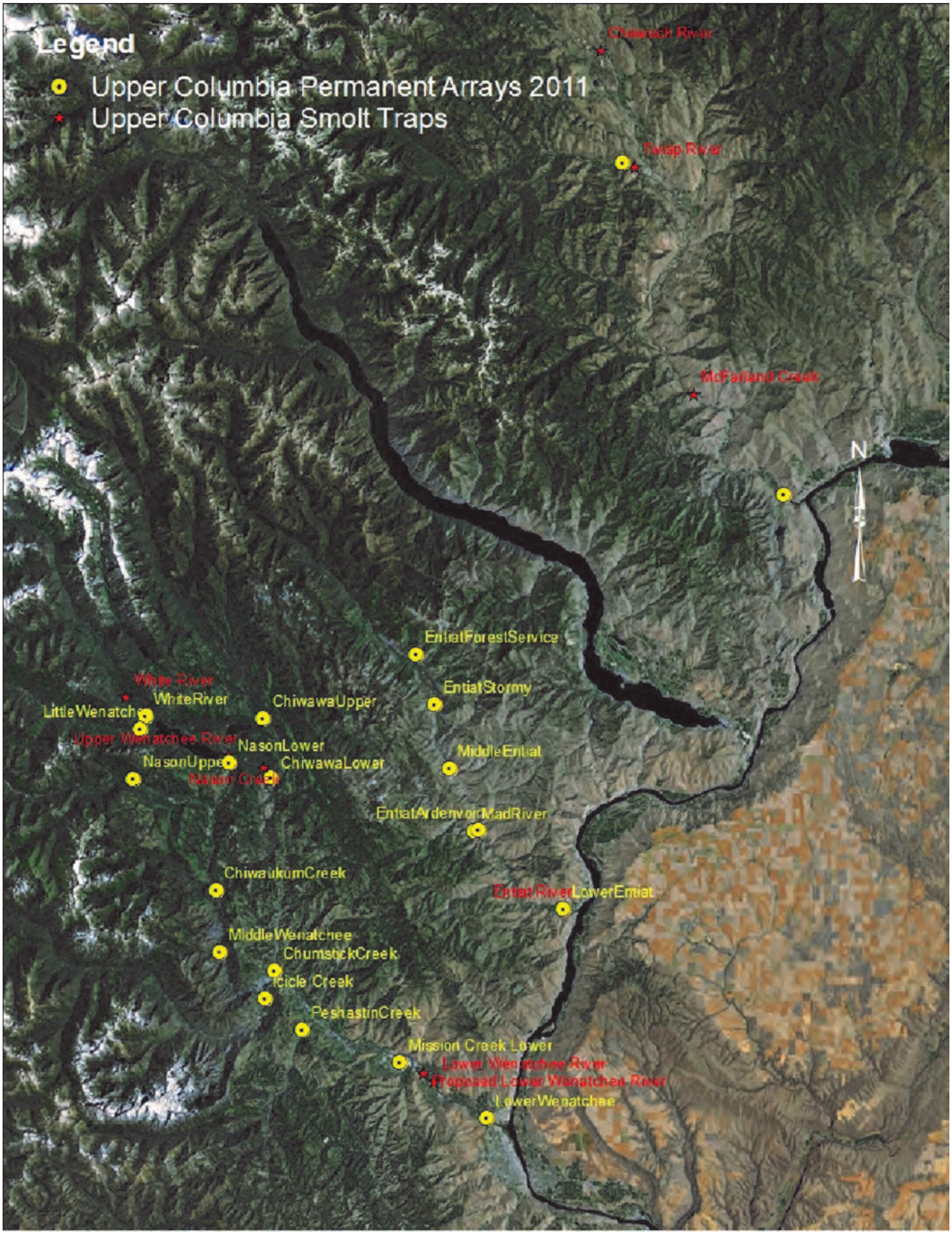


Figure 18. Location of instream PIT tag detection arrays and rotary screw traps in the Wenatchee and Entiat Rivers. Map courtesy of the WDFW.

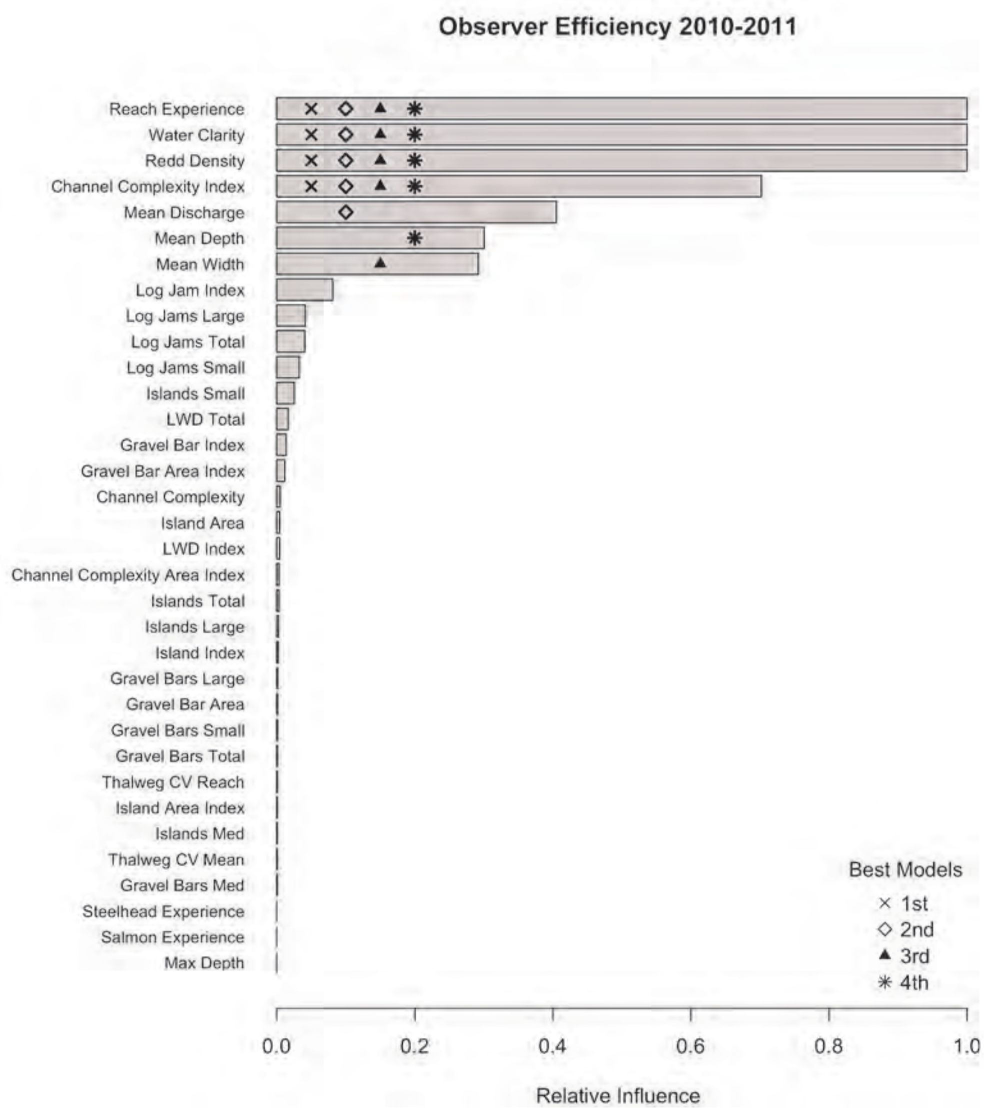


Figure 19. The relative influence of each possible variable in predicting the proportion of visible redds that were observed. The predictors that were included in each of the four best models are marked.

Fisheries and ISEMP, will provide PIT tag data with which to test the escapement estimates into each tributary.

ISEMP has also been working with co-managers in the Upper Columbia to estimate steelhead redd survey observer efficiency and not only identify, but also quantify sources of error that affect the uncertainty contributed by both sampling error (failure to detect redds and/or counting false redds) and expansion error (i.e., assumptions about the number of adults represented by a given redd) and to standardize and improve steelhead redd surveys.

From 2010 through 2012 ISEMP has worked with WDFW to conduct a steelhead observer efficiency study on the Wenatchee River (generally following the methods described in Thurow and McGrath (2010)) to determine what factors drive the efficiency of steelhead redd counts (e.g., water clarity, habitat complexity, experience) using naïve versus "true" redd surveys.

The correct identification of steelhead redds in the Wenatchee subbasin was higher in the tributaries of the Wenatchee River than the mainstem itself, which could be related to the attributes of the tributaries (e.g., redd density, stream depth and width, and channel complexity) (Murdoch and Herring 2011). There was a wide range of individual observer efficiencies but no significant correlation was found between the proportion of redds correctly identified and experience conducting salmonid spawning ground surveys (rs = 0.04), or experience conducting steelhead spawning ground surveys (rs = 0.14) or experience conducting steelhead spawning ground surveys on a specific reach (rs = 0.2). However, the suite of factors that were most important in predicting the proportion of redds correctly identified included experience on a specific reach, water clarity, density of redds, channel complexity, discharge, stream depth and stream width (Figure 19). The best model (i.e., with the lowest AIC score) was able to explain 73% of the variance in omission rates (Figure 20).

A similar analysis is taking place to explain the number of redds falsely identified. When combined with estimates of observer efficiency, this will lead to an estimate of the total number of redds throughout a season, with appropriate uncertainty bounds.

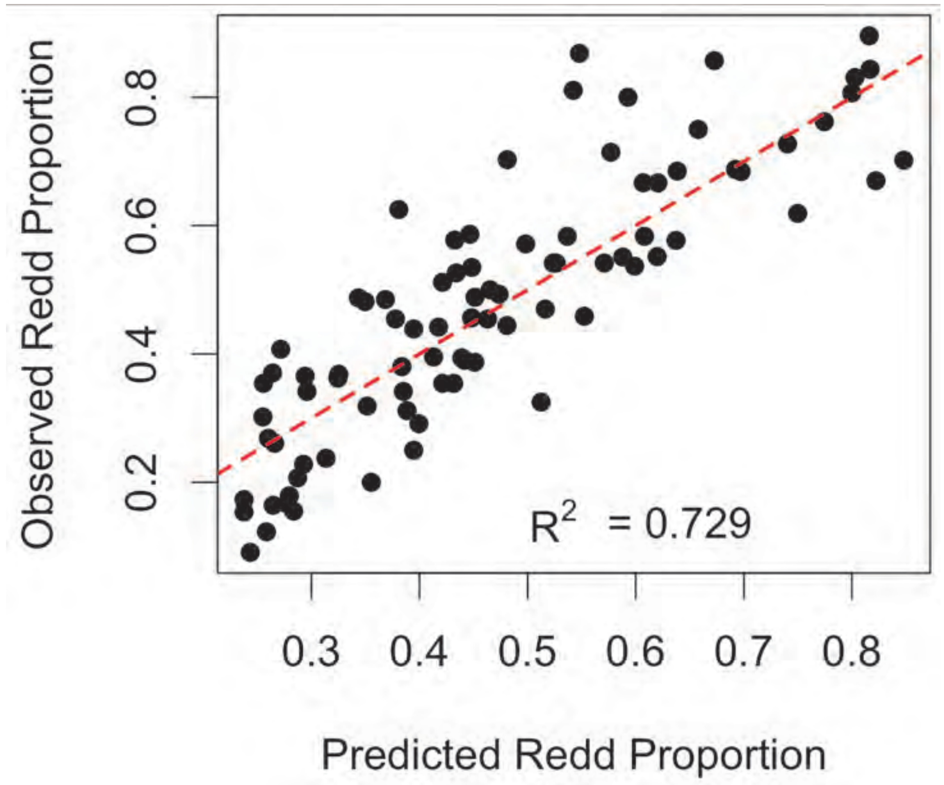


Figure 20. Comparing predicted observer efficiency rates to the observed rates using the best model, as shown in Figure 18.

John Day Basin Steelhead Redd Surveys

The first years of ISEMP in the John Day basin focused solely on learning about, reviewing, and summarizing ongoing monitoring efforts and data (Bouwes 2006). From this summarization, ISEMP determined where they could be complementary to existing monitoring programs. At the scale of the John Day basin, the most extensive fish and fish habitat monitoring designed to fulfill BiOp requirements at the time was being conducted by ODFW. Thus, ISEMP has collaborated with ODFW on status and trends monitoring of listed salmonids in the John Day basin. We have added to these efforts and have also developed effectiveness monitoring programs (Appendices: Chapter 5, Evaluation of Riparian Fencing as a Restoration Tool in the John Day Basin, Chapter 7 Bridge Creek Intensively Monitored Watershed Project).

ODFW has conducted steelhead redd surveys throughout the John Day Basin since 2004 (Figure 21). Here we provide a brief summary of the survey approach and results from Banks et al. (2011). The sampling domain was defined as perennial streams accessible to anadromous salmonids as determined by biologists from ODFW and other entities (Figure 21). Sites were selected under the GRTS sampling design. Fifty sites are selected each year and assigned to rotating panels with 17 of these sites sampled every year, 16 sites sampled once every 4 years on a staggered basis, and 17 new sites each year. Four additional sites are surveyed for the South Fork John Day population.

Steelhead redd surveys are based on standard ODFW methods (Susac and Jacobs 1999; Jacobs et al. 2000; Jacobs et al. 2001). Redds are counted at sites approximately 2 km in length, and revisited as many as six times with approximately two-week intervals between successive surveys.

Overall linear redd density is estimated by the number of unique redds observed at all sites divided by the distance surveyed. The total number of redds occurring throughout the basin is estimated by expanding the redd density to the total km of river in the sample universe (4,322 km). Total steelhead escapement is estimated by multiplying the

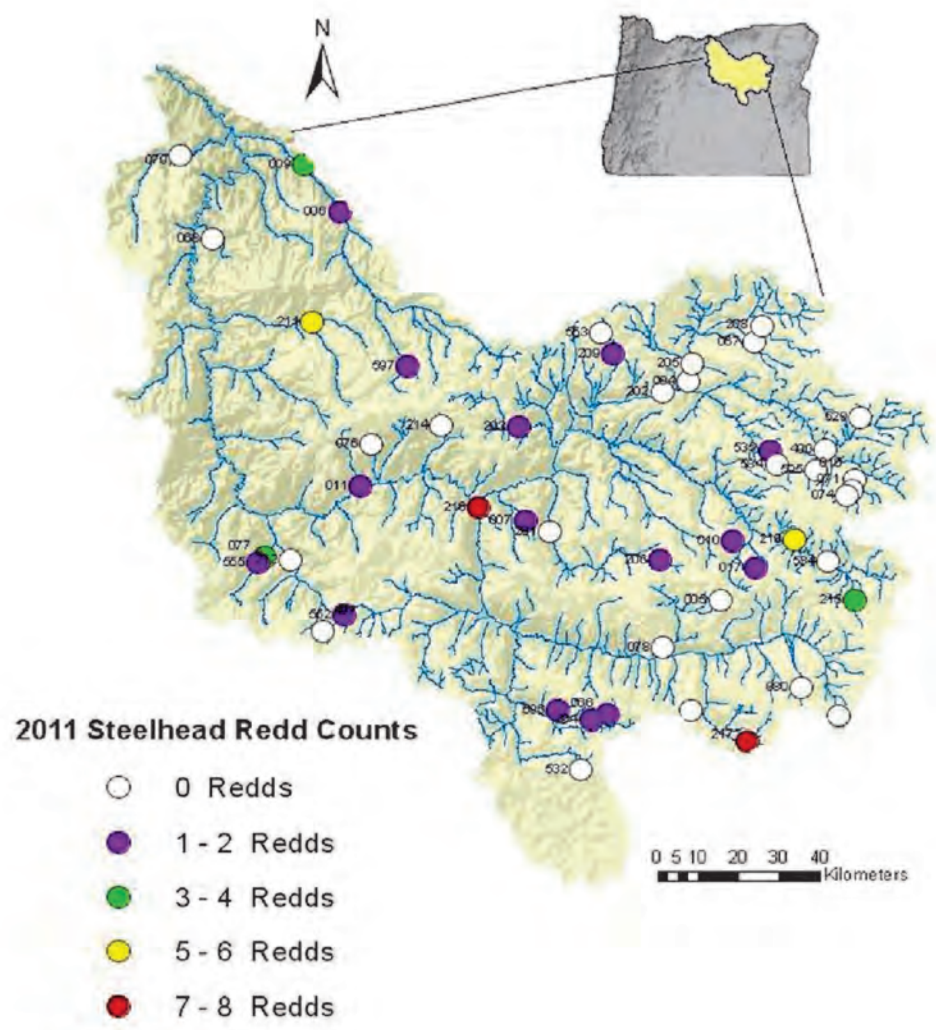


Figure 21. Distribution and number of summer steelhead redds observed in the John Day River basin during spawning surveys conducted in the spring of 2011 (From Banks et al. 2011).

total redds by an empirical steelhead/redd estimate (e.g. 4.75 fish/redd in 2011). A locally weighted neighborhood variance estimator (Stevens and Olsen 2004), incorporating the pair-wise dependency of all points and the spatially constrained nature of the design, is used to estimate a 95% confidence interval for the escapement estimate. Estimates of steelhead adult escapement conducted by ODFW are found in Figure 22 and Table 11 (Banks et al. 2011).

In FY2012, ISEMP will collaborate with ODFW to use adult escape estimates to calculate freshwater production (i.e., juveniles per adult), spawner per spawner, and other potential metrics. In addition, this information will be combined with habitat information to develop spawner/habitat relationships. Finally we plan to use the watershed production model as a framework to synthesize life-cycle information.

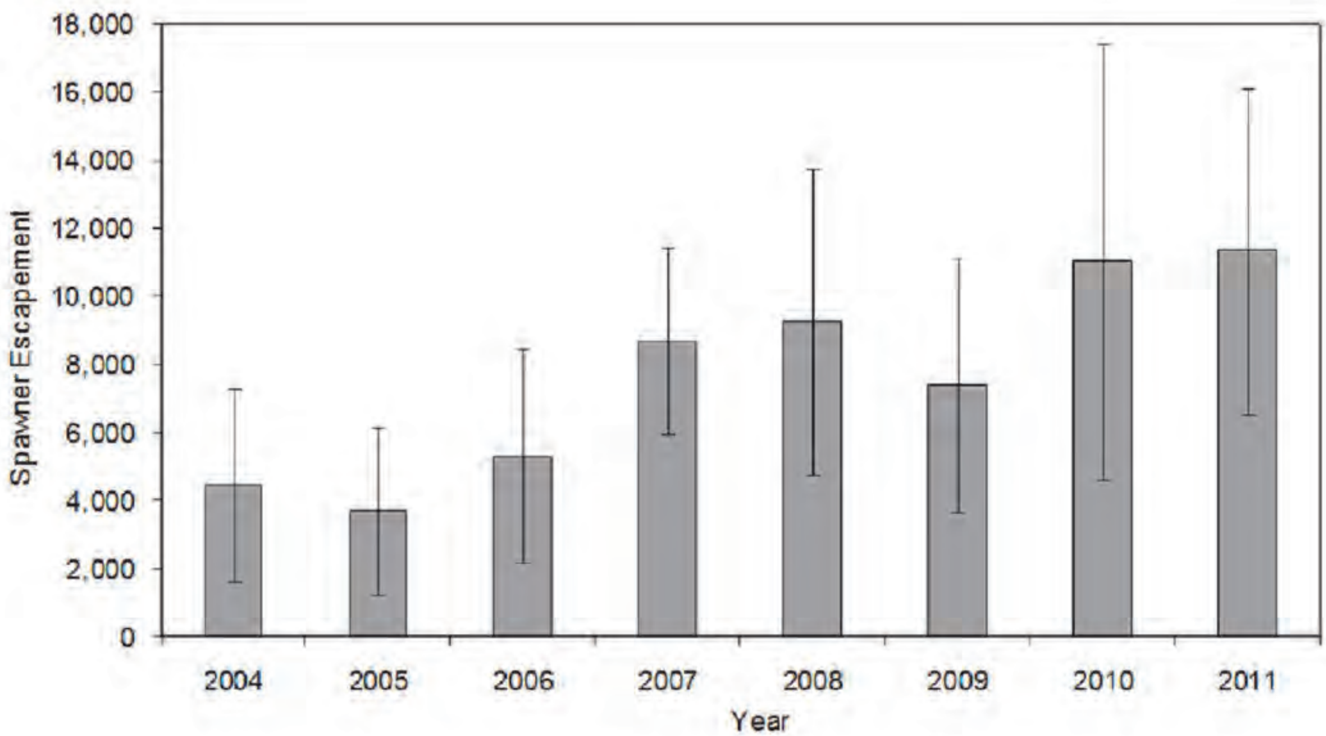


Figure 22. Annual adult steelhead spawner escapement estimates for the John Day River basin from 2004 to 2011. Error bars indicate 95% confidence intervals (From Banks et al. 2011).

Table 11. Distance surveyed, number of unique redds observed, redd density (redds/km), estimated total number of redds, fish per redd estimate from Deer Creek (Grande Ronde River basin), and spawning escapement estimate with 95% C.I. for the John Day River basin from 2004 to 2011 (From Banks et al. 2011).

Year	km	Redds	Density	Total Redds	Fish/Redd	Escapement	95% Lower	95% Upper
2004	94.7	66	0.7	3,071	1.46	4,484	1,657	7,310
2005	101.2	39	0.39	1,681	2.2	3,698	1,261	6,137
2006	90.5	67	0.74	3,202	1.66	5,315	2,189	8,441
2007	99.6	181	1.82	7,758	1.12	8,689	5,939	11,439
2008	105	56	0.53	2,277	4.07	9,260	4,742	13,775
2009	98.6	44	0.45	1,934	3.81	7,368	3,642	11,099
2010	96.9	155	1.6	6,914	1.59	11,027	4,628	17,434
2011	96.0	53	0.55	2,386	4.75	11,334	6,565	16,103

We are also using steelhead adult abundance, in part, to evaluate the effectiveness of stream restoration in the Bridge Creek IMW (see Appendix- Chapter 7). As is common in several watersheds, steelhead often create redds at the onset of high flows when water is very turbid water and redds are difficult and, at times impossible, to see. This is especially true in Bridge Creek and to ameliorate for this we have placed a 2-way weir low in the watershed to capture and tag adults going upstream and recapture kelts coming downstream to estimate adult spawners via a mark-recapture estimate (Figure 23). We continue to conduct redd counts in Bridge Creek to maintain the historic data stream and to evaluate the discrepancy between the two approaches. Preliminary findings suggest that redd counts greatly underestimate the number of adult spawners, as counted passing over the weir in Bridge Creek.

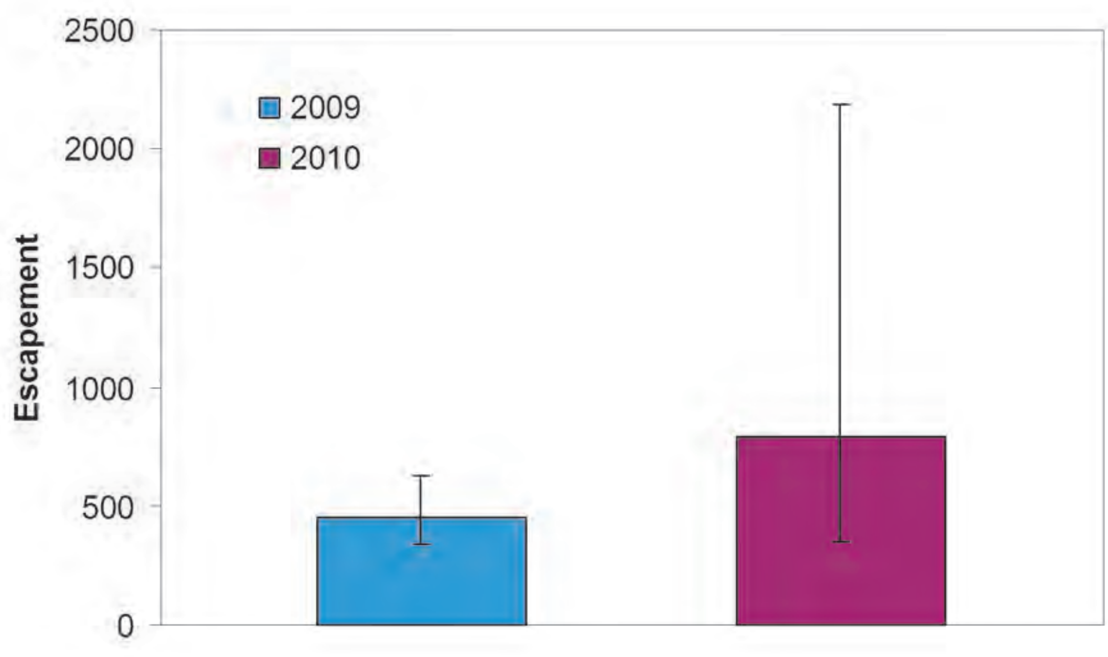


Figure 23. Escapement estimate for all steelhead (hatchery and wild) in Bridge Creek based on mark-recapture estimation. Error bars are 95% Confidence Intervals. 2009 escapement estimate was 449, with 19% of the run being adipose-marked hatchery fish. 2010 escapement estimate was 792 with 41% of the run comprised of adipose-marked hatchery fish.

Monitoring Juvenile Salmonid Standing Crop and Emigration

The BiOp identifies habitat restoration as one of the offsite mitigation actions to offset mortality imposed by the hydrosystem. Survival improvements accruing from habitat restoration actions are measured as improvements in egg to smolt survival (USACE, BPA, USBOR 2007; Attachment C-1). In general, an assessment of changes in egg to smolt survival is sufficient to determine whether habitat restoration actions successfully improved survival, but is insufficient to identify why or how those actions were or were not successful.

As described in Chapter 2 of the Appendix, the ISEMP watershed model describes juvenile abundance and survival as a function of habitat quantity and quality, unique to the requirements of each life stage. Viewed in that context, the habitat conditions that limit egg to smolt survival can occur at any life stage; prior to emergence (e.g., as a function of fine sediment), from fry to parr (e.g., as a function of habitat complexity), from parr to presmolt (e.g., as a function of temperature), and from presmolt to smolt (e.g., as a function of limited overwintering habitat, such as large pools). Programs aimed at identifying why and how habitat restoration worked or failed to work require much more information than programs directed to simply address whether restoration yielded an improvement in egg to smolt survival. To answer these questions monitoring programs such as ISEMP must generate life-stage-specific estimates of juvenile abundance, distribution, and survival. As described in Appendix—Chapter 2, this information enables an evaluation of the veracity of identified limiting factors, effectiveness of restoration actions intended to address those limiting factors, and overall effectiveness of the employed actions at improving survival. This allows managers to select the most effective habitat restoration actions and determine the aggressiveness of implementation necessary to achieve survival goals.

A primary challenge faced by programs tasked with habitat effectiveness is a paucity of information directed at estimating the distribution, abundance, and survival of juvenile anadromous salmonids prior to their emigration to the hydrosystem. Infrastructure such as rotary screw traps generate an estimate of juveniles that survived to emigrate, but yield no information on these metrics prior to emigration or which (if any) habitat restoration actions contributed to the production of emigrants.

ISEMP uses multiple survey types to estimate the standing crop and total migrants of juvenile salmonids: rotary screw traps are used to generate estimates of total migration by life-stage from tributaries and populations and field crews sample specific sites to generate site, tributary or population scale standing crop (total number of fish in a watershed at a specific time) of juveniles.

Standing Crop

ISEMP employs a probabilistically based juvenile sampling effort utilizing GRTS (Stevens and Olsen 2004) in each of the subbasins. GRTS distributes sampling effort in a manner that represents both space and time across tributaries of varying sizes that comprise the totality of tributary habitat utilized by anadromous salmonids prior to emigration to the mainstem. At GRTS sites, ISEMP crews employ a standardized fish sampling protocol to capture and PIT tag juvenile anadromous salmonids. These efforts generate estimates of juvenile abundance at that site, allowing tagged individuals to represent a known fraction of the untagged population. Since these efforts are deployed using GRTS, site-specific abundance estimates can be aggregated to estimate abundance at any spatial scale included in the GRTS sample frame – up to and including populations and MPGs. Similarly, survival and growth estimates generated from PIT tagged juveniles can be used to represent the population at any of those spatial scales. Ultimately, this enables contrasts in abundance, distribution, and survival among stream reaches subject to habitat restoration versus reaches that are untreated. Aggregated to the population scale, this allows an estimate of these metrics resulting from habitat restoration.

Spatially and temporally balanced sampling of this nature has generally been implemented to monitor habitat, and in some cases to conduct redd surveys. In ISEMP, we utilize the same GRTS design for both juvenile sampling and habitat sampling which supports the development of relationships between fish and habitat attributes. These relationships allow the identification of habitat features that are conducive to fish, enabling an assessment of the realized and anticipated effectiveness of habitat actions.

Salmon Subbasin

This sampling effort allows us to state where fish are (Figures 24-27) by age, and estimate their survival as a function of where they reared prior to leaving the watershed in the Salmon subbasin. By comparing the survival of fish that reared in restored and unrestored locations we can estimate the value of habitat actions and scale the survival value of that action against the population as a whole.

Ultimately, the site-based abundance estimates can be aggregated to enable an evaluation of where juveniles reside within a watershed relative to habitat restoration actions (Tables 12-15). Similarly, because fish have been representatively tagged within these units of interest, survival of tagged juveniles indicates whether life-stage specific survival is improved in areas that have been restored. Taken together, the abundance and survival estimates enable strong statements about the overall effectiveness

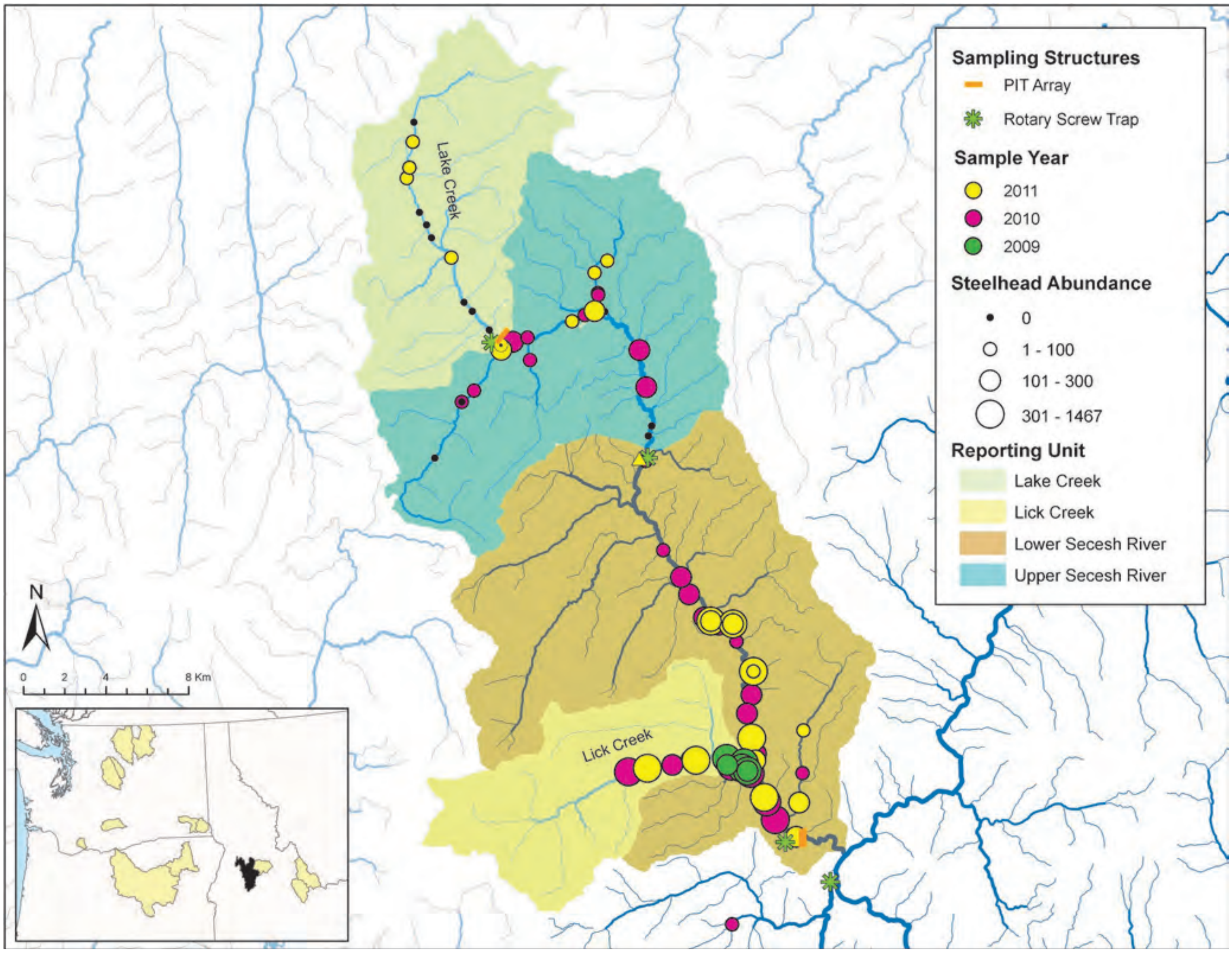


Figure 24. Location of juvenile sampling infrastructure and distribution and abundance of juvenile steelhead obtained via remote juvenile surveys in the Secesh River. Note that reporting units identify spawning habitat (upper Secesh River, Lake Creek, and Lick Creek) and habitat used primarily for rearing or serving as a migration corridor (lower Secesh River).

of habitat restoration actions, and identify whether those actions ultimately lead to a greater number of fish as opposed to a simple redistribution of fish to restored habitat.

Table 12. Abundance of juvenile steelhead by reporting unit in the Secesh River.

Year	Priority	Sites	Estimate	Perecentage of Population by Category	CV
2010	Population	43	46,484		11%
	Upper Secesh	14	7,140	15%	35%
	Lower Secesh	17	28,284	61%	10%
	Lick Creek	6	10,915	23%	28%
	Lake Creek	6	145	0.3%	85%
2011	Population	24	66,355		18%
	Upper Secesh	8	6,909	10%	33%
	Lower Secesh	9	45,958	69%	24%
	Lick Creek	2	9,843	15%	32%
	Lake Creek	5	3,646	5%	38%

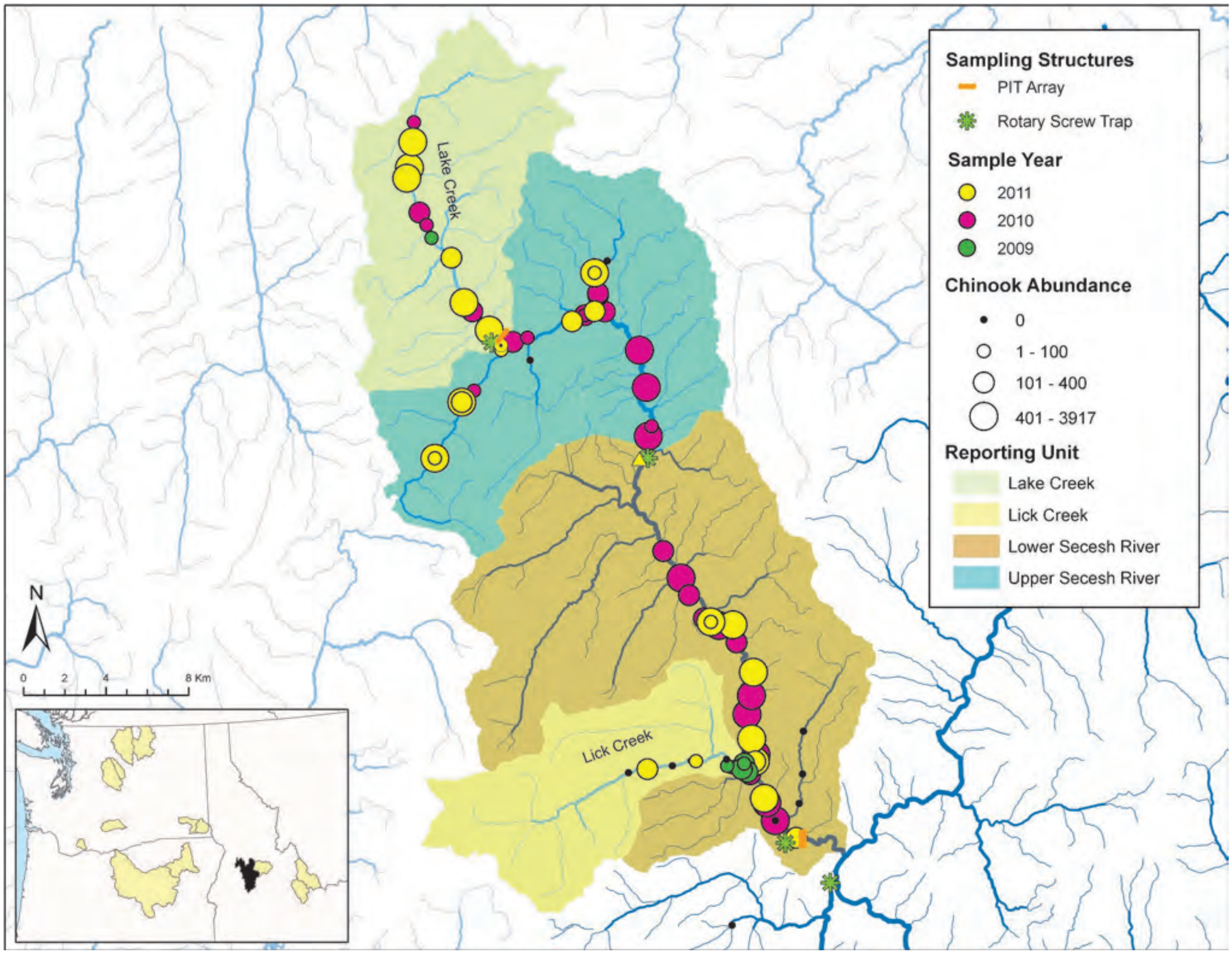


Figure 25. Location of juvenile sampling infrastructure and distribution and abundance of juvenile spring/summer Chinook salmon obtained via remote juvenile surveys in the Secesh River. Note that reporting units identify spawning habitat (upper Secesh River, Lake Creek, and Lick Creek) and habitat used primarily for rearing or serving as a migration corridor (lower Secesh River).

Table 13. Abundance of juvenile spring/summer Chinook salmon by reporting unit in the Secesh River.

Year	Priority	Sites	Estimate	Perecentage of Population by Category	CV
2010	Population	43	118,196		12%
	Upper Secesh	14	24,309	21%	26%
	Lower Secesh	17	76,962	65%	17%
	Lick Creek	6	727	1%	43%
	Lake Creek	6	16,199	14%	15%
2011	Population	24	329,783		19%
	Upper Secesh	8	57,162	17%	58%
	Lower Secesh	9	126,284	38%	29%
	Lick Creek	2	2,142	1%	25%
	Lake Creek	5	144,196	44%	28%

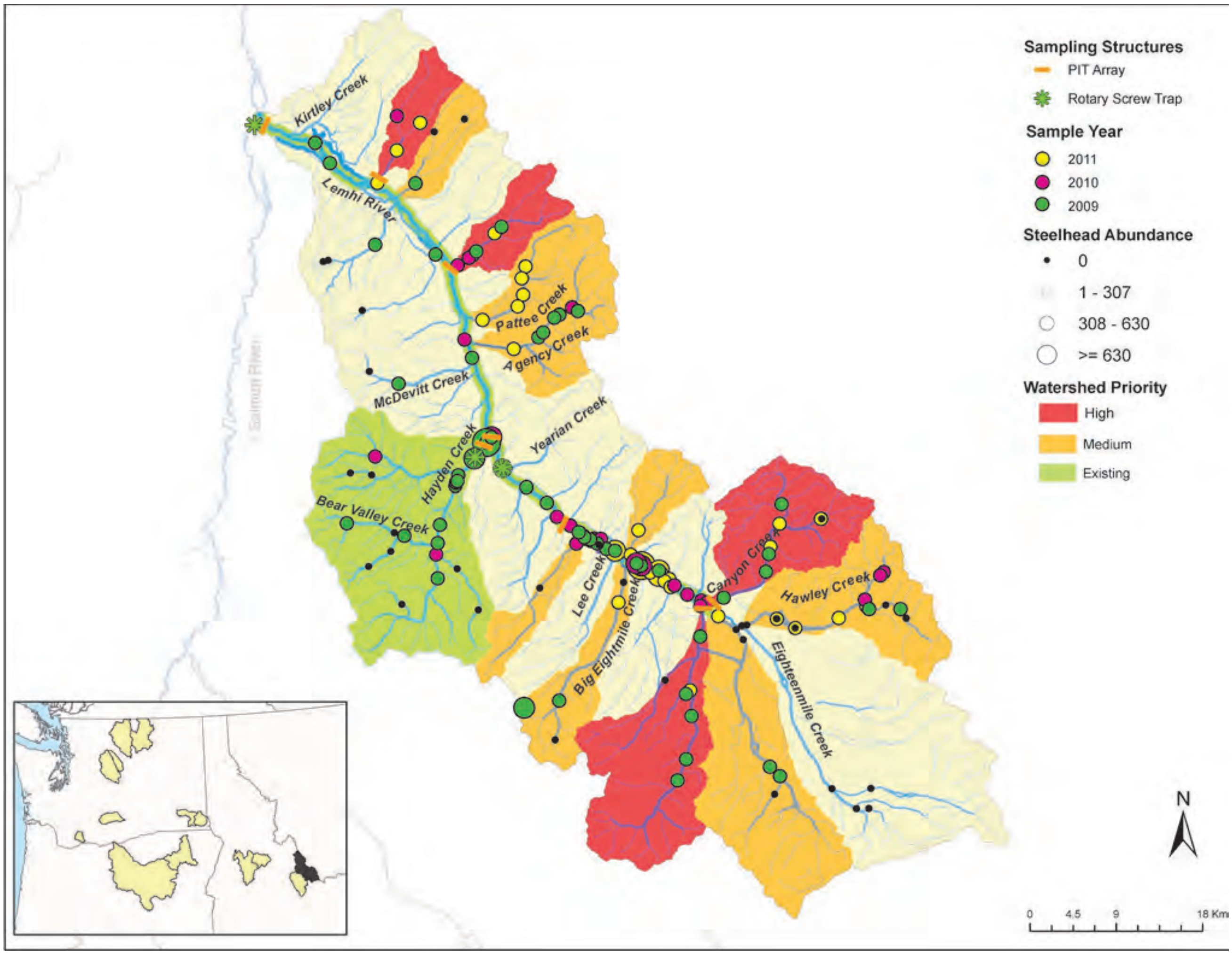


Figure 26. Location of juvenile sampling infrastructure and distribution and abundance of juvenile steelhead obtained via remote juvenile surveys in the Lemhi River. Note that watershed priority refers to habitat that was hydraulically connected prior to the implementation of habitat actions (mainstem Lemhi River and Hayden Creek), high priority tributaries (those reconnected in Phase I of the Lemhi Conservation Plan), and low priority tributaries (those tributaries that will either be reconnected in Phase II of the Lemhi Conservation plan).

Table 14. Abundance of juvenile steelhead in existing, reconnected (High Priority), and currently disconnected (medium priority) habitat in the Lemhi River.

Year	Priority	Sites	Estimate	Perecentage of Population by Category	CV
2010	Population	41	83,765		15%
	Existing	23	49,847	60%	25%
	High	8	19,864	24%	11%
	Medium	10	14,055	17%	19%
2011	Population	57	158,078		12%
	Existing	13	105,989	67%	18%
	High	26	37,223	24%	12%
	Medium	18	14,866	9%	15%

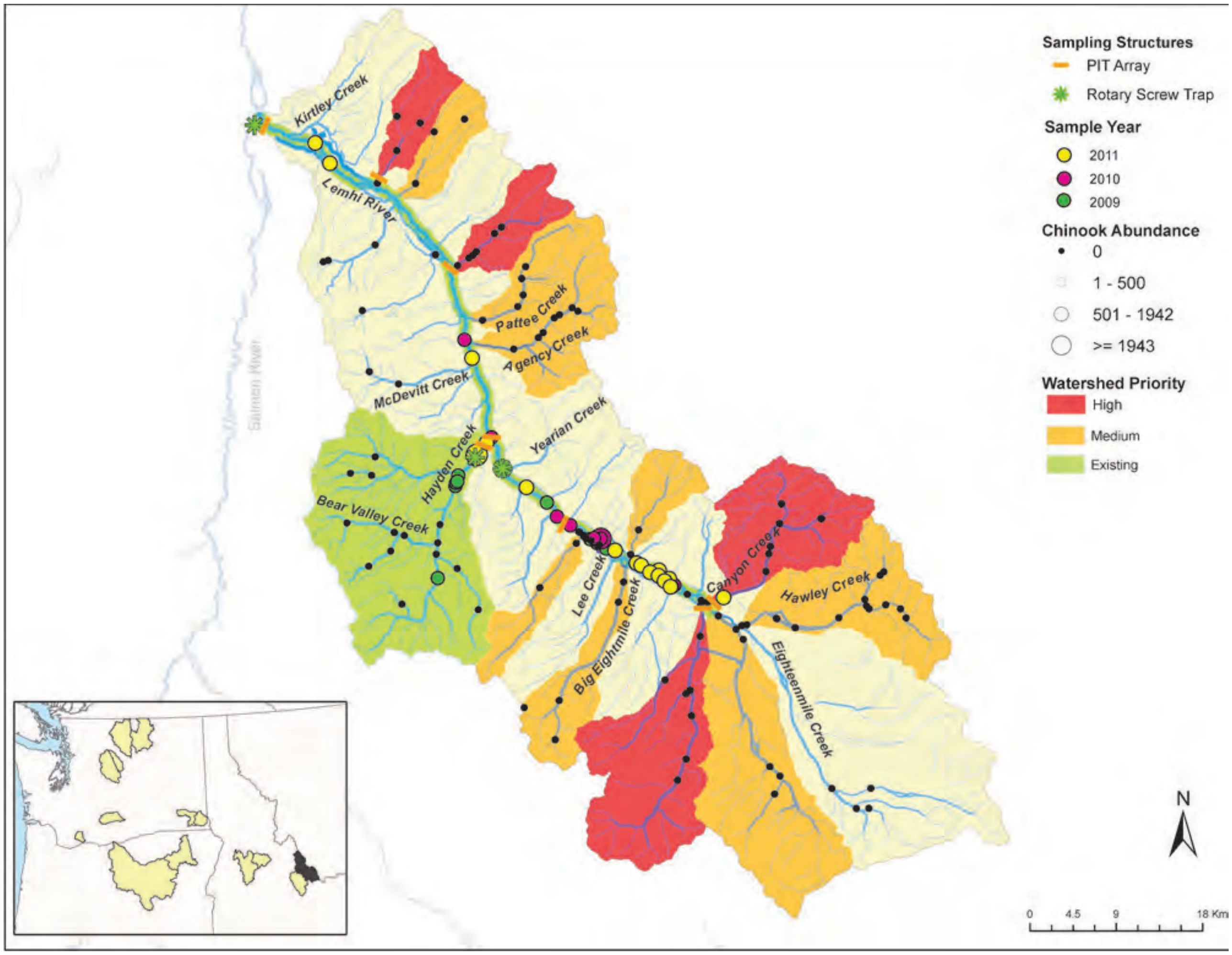


Figure 27. Location of juvenile sampling infrastructure and distribution and abundance of juvenile spring/summer Chinook salmon obtained via remote juvenile surveys in the Lemhi River. Note that watershed priority refers to habitat that was hydraulically connected prior to the implementation of habitat actions (mainstem Lemhi River and Hayden Creek), high priority tributaries (those reconnected in Phase I of the Lemhi Conservation Plan), and low priority tributaries (those tributaries that will either be reconnected in Phase II of the Lemhi Conservation plan).

Table 15. Abundance of juvenile spring/summer Chinook salmon in existing, reconnected (High Priority), and currently disconnected (medium priority) habitat in the Lemhi River.

Year	Priority	Sites	Estimate	Percentage of Population by Category	CV
2010	Population	38	113,152		44%
	Existing	20	113,152	100%	44%
	High	8	0	0%	
	Medium	10	0	0%	
2011	Population	57	139,145		37%
	Existing	13	138,220	100%	38%
	High	26	925	1%	36%
	Medium	18	0	0%	

Upper Columbia

Figures 29 through 32 show the distribution and average density across years for juvenile spring/summer Chinook salmon and steelhead as observed using snorkel surveys at 25 GRTS habitat sites from 2005 –2010 in both the Wenatchee and Entiat River subbasins. Figures 33 – 36 show the annual estimate of the standing crop of juvenile Chinook and steelhead in each assessment unit in the Wenatchee River subbasin and the Entiat River subbasin based on the GRTS design.

It should be noted that this data is a combination of snorkeling and e-fishing data and is from the annual panel sites only (in the Wenatchee the rotating panel sites were sampled from 2005 - 2008 and in the Entiat there was only an annual panel of 25 sites). Also, in 2011, ISEMP implemented a tagging study at the CHaMP habitat sites to replace snorkel surveys as part of ISEMP standardizing its fish monitoring protocols across the three subbasins and to allow us to garner more information from our monitoring efforts. For example, by PIT tagging fish we not only are able to generate an abundance estimate, either through a mark-recapture event or a single pass depletion estimate with electrofishing, we also get information on length and weight from having the fish in hand, and survival and growth data if the fish are subsequently recaptured (survival information from passing over instream PIT tag arrays and recaptures at rotary screw traps, growth information from rotary screw trap recaptures and possibly recaptures under the GRTS fish monitoring study).

Due to these different methodologies used to collect fish densities 2011 is not displayed in Figures 29 through 32, and in the graphs showing the annual estimate of density by assessment unit from 2005-2011 (Figures 33– 36) the 2011 data is not connected. However, ISEMP has conducted a study in the John Day to calibrate snorkel one-pass electrofishing surveys to absolute abundance, where a snorkel crew sampled a site and counted salmonids and then conducted a mark-recapture study over the next two days to get an abundance estimate. A strong significant relationship between snorkel estimates and MR estimates was observed (see Figures 38 and 39 for results), which allowed for an estimate of the ratio of what proportion of fish were counted in snorkel surveys. Also, the connection between one-pass e-fishing and abundance estimates has been investigated in the Salmon basin and show a good correlation (Figure 28). Similar studies in the Upper Columbia would allow us to translate the 2004-2010 snorkel and electrofishing counts into abundance estimates, making them compatible with data collected under the tagging study from 2011 and in future years.

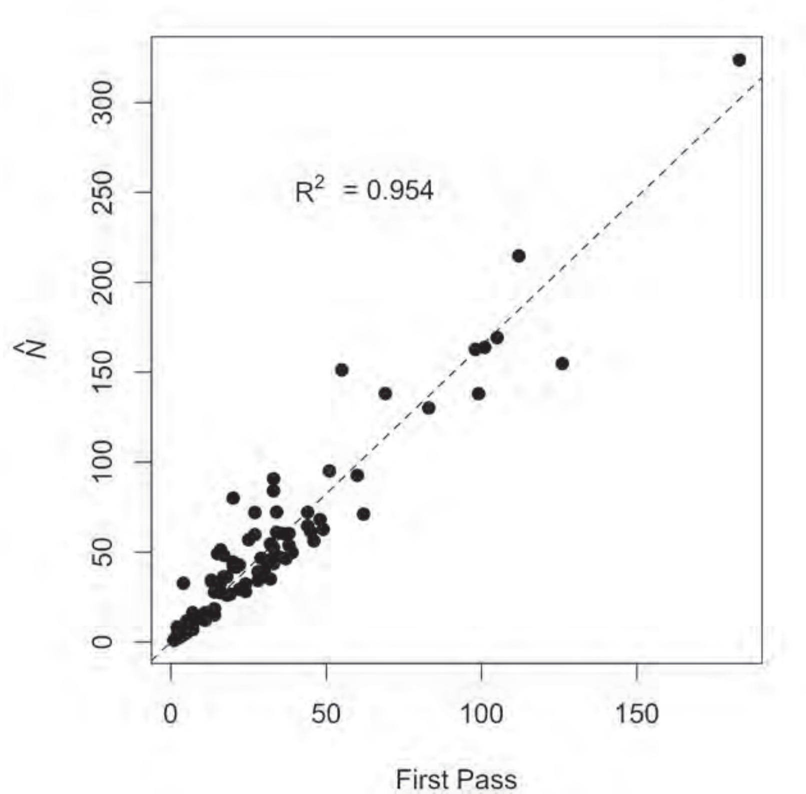


Figure 28. Relationship between one-pass electrofishing estimates and abundance estimates in the Salmon River subbasin.

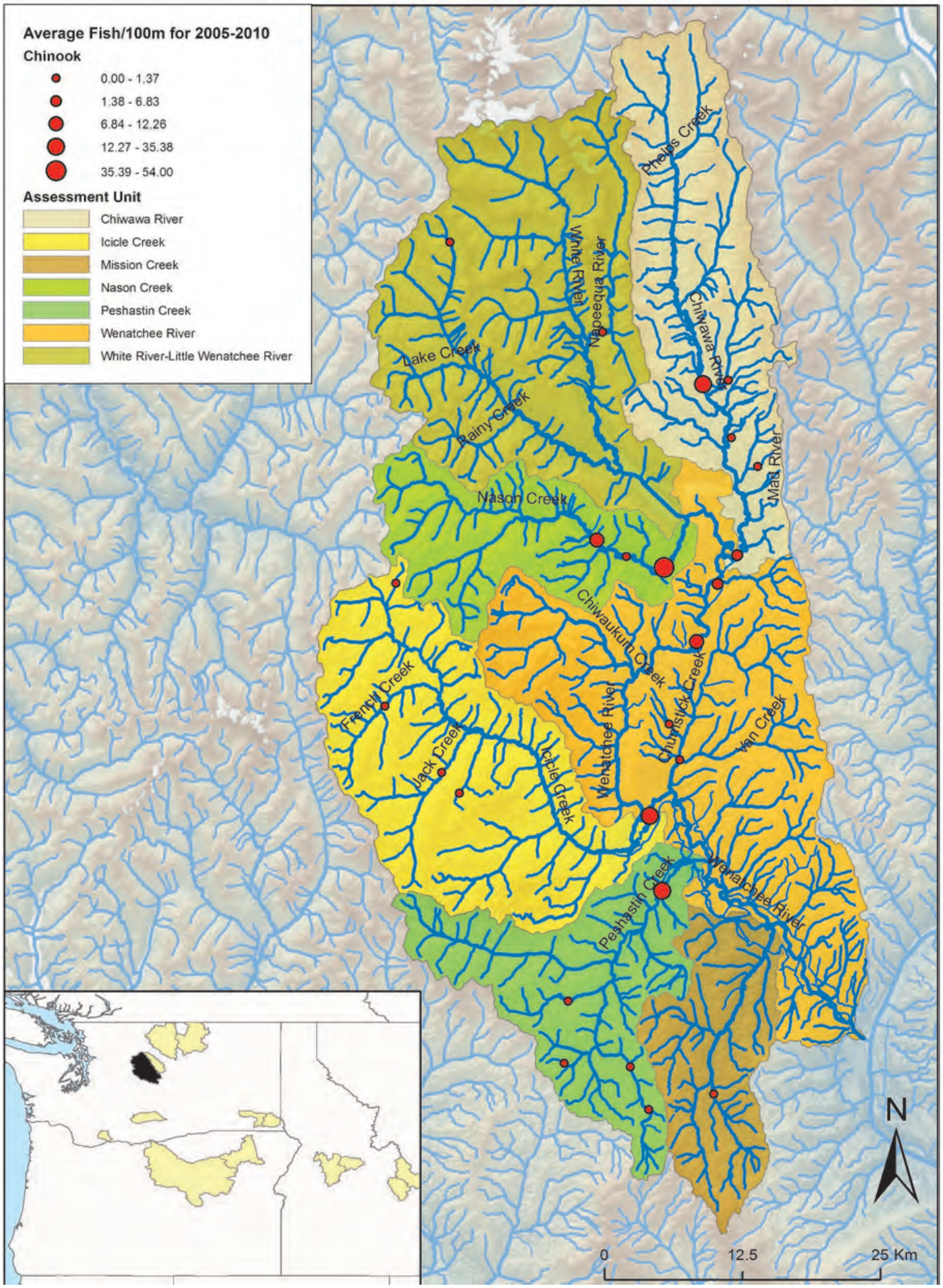


Figure 29. Distribution and abundance of juvenile spring/summer Chinook salmon obtained via remote juvenile surveys in the Wenatchee River subbasin, Upper Columbia 2005—2010.

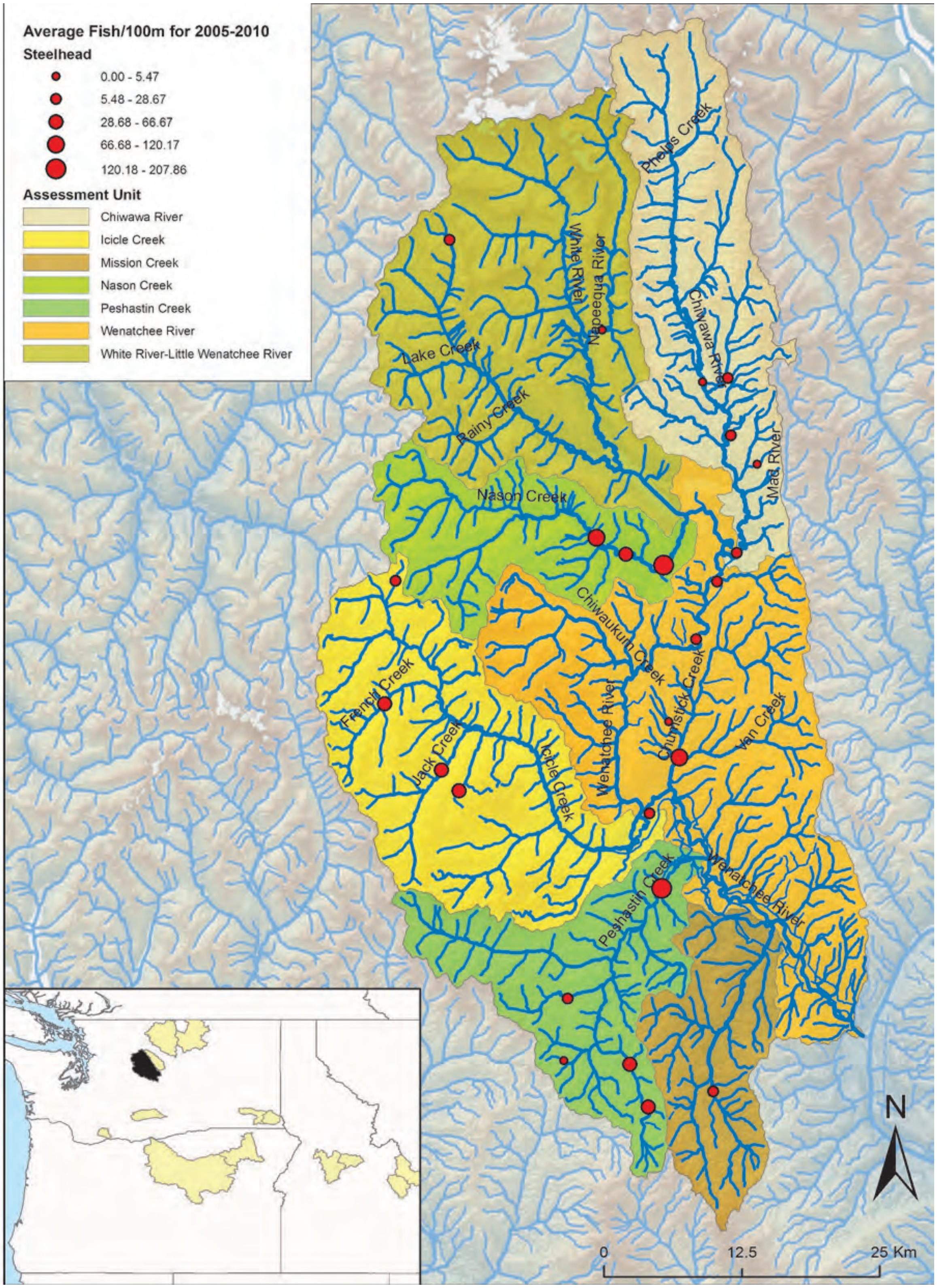


Figure 30. Distribution and abundance of juvenile steelhead obtained via remote juvenile surveys in the Wenatchee River subbasin, Upper Columbia 2005—2010.

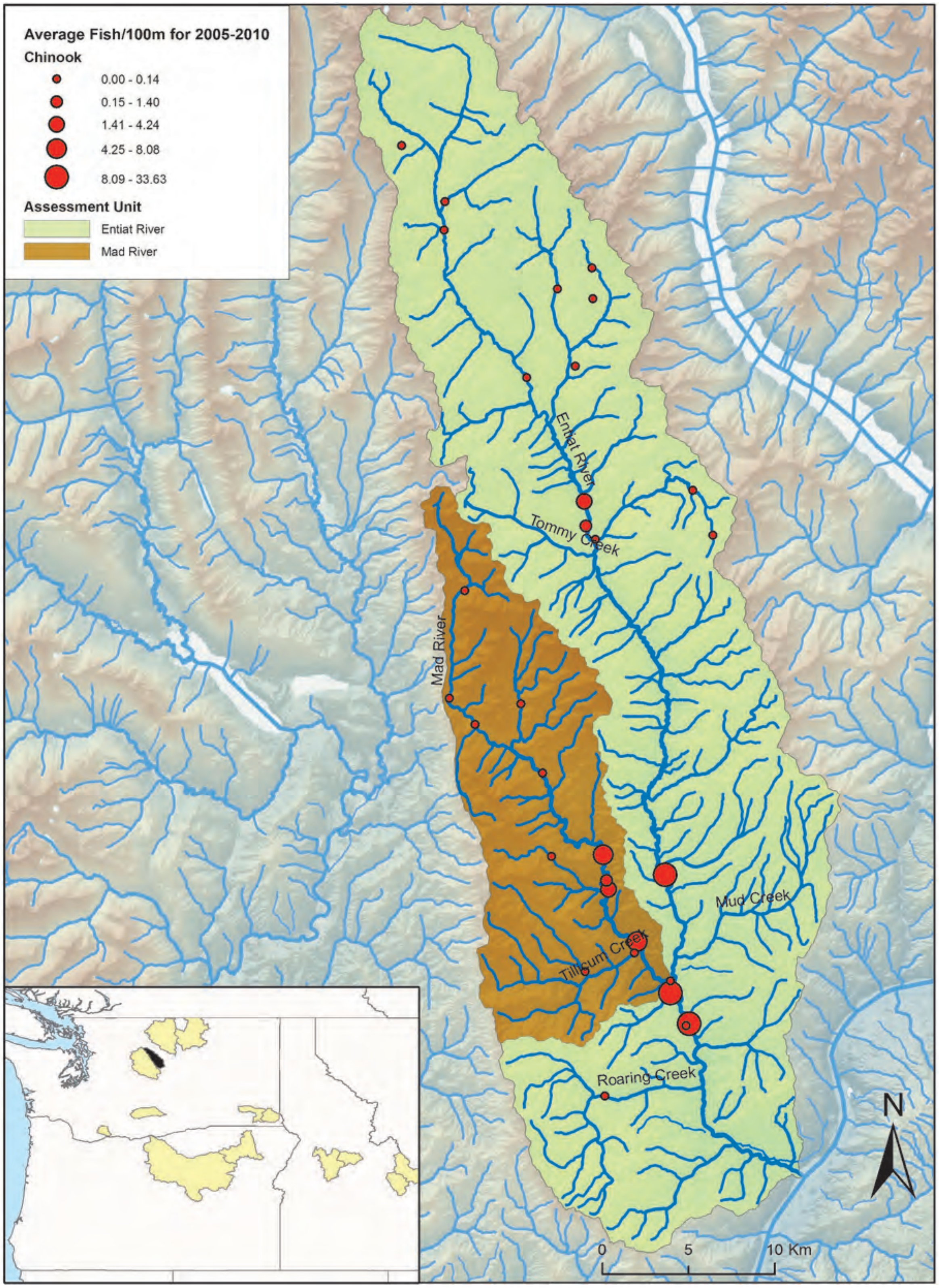


Figure 31. Distribution and abundance of juvenile spring/summer Chinook salmon obtained via remote juvenile surveys in the Entiat River subbasin, Upper Columbia 2005—2010.

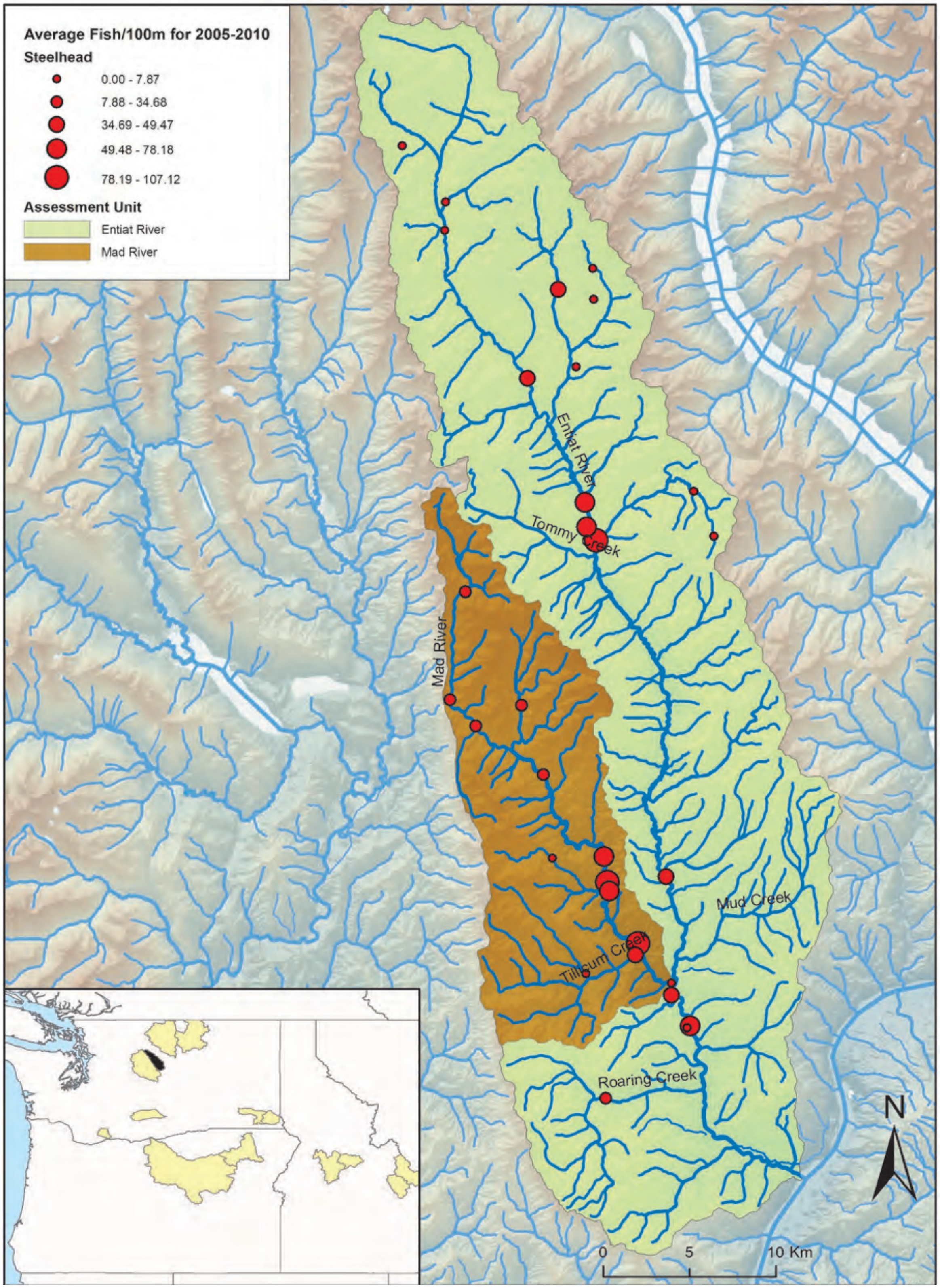


Figure 32. Distribution and abundance of juvenile steelhead obtained via remote juvenile surveys in the Entiat River subbasin, Upper Columbia 2005—2010.

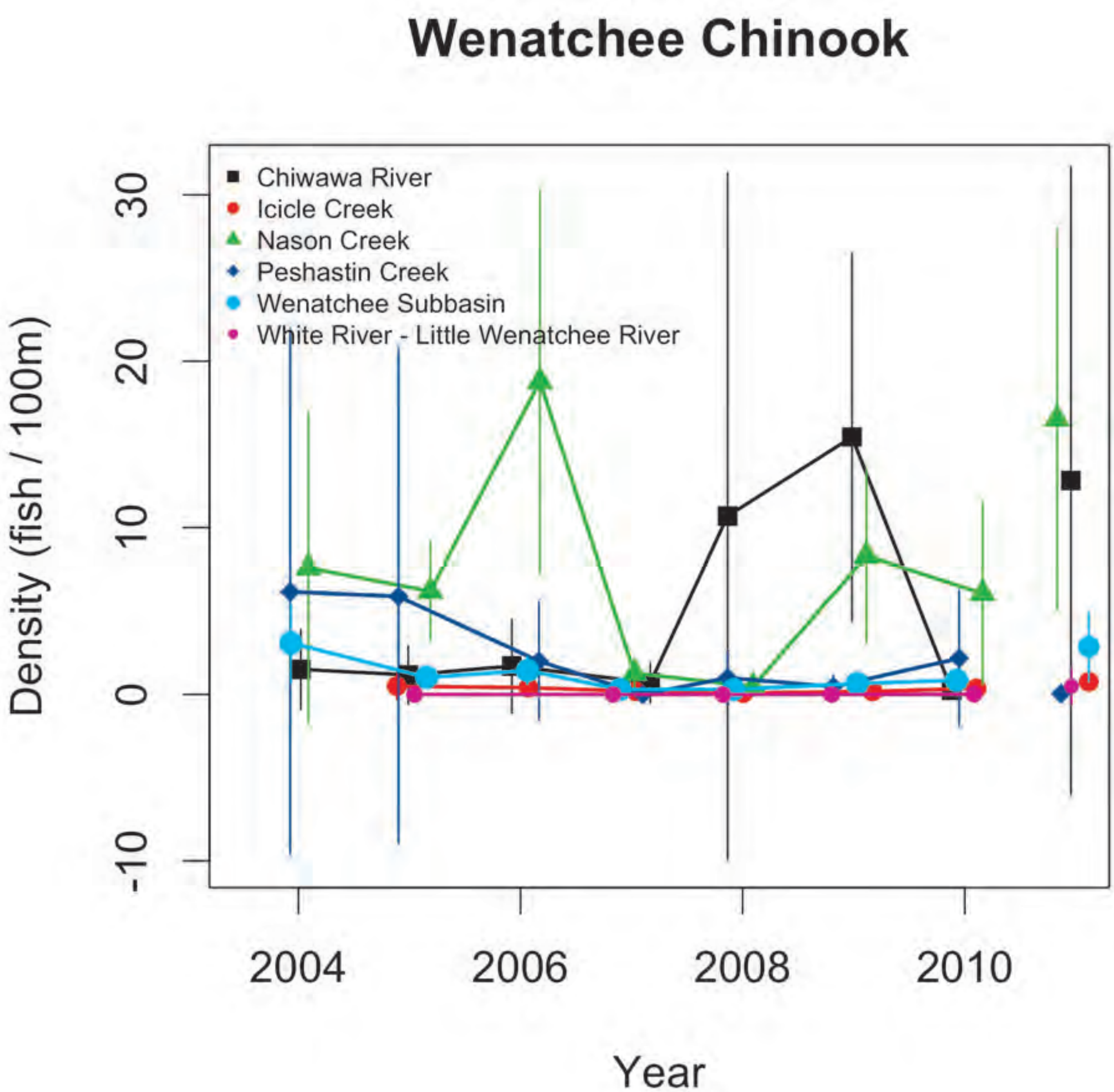


Figure 33. Density of juvenile Chinook standing crop in the Wenatchee River subbasin assessment units estimated from snorkel and electrofishing surveys (2005—2010) and mark-recapture/single pass electrofishing surveys (2011) from 25 annually sampled GRTS sites. Error bars represent plus and minus one standard error.

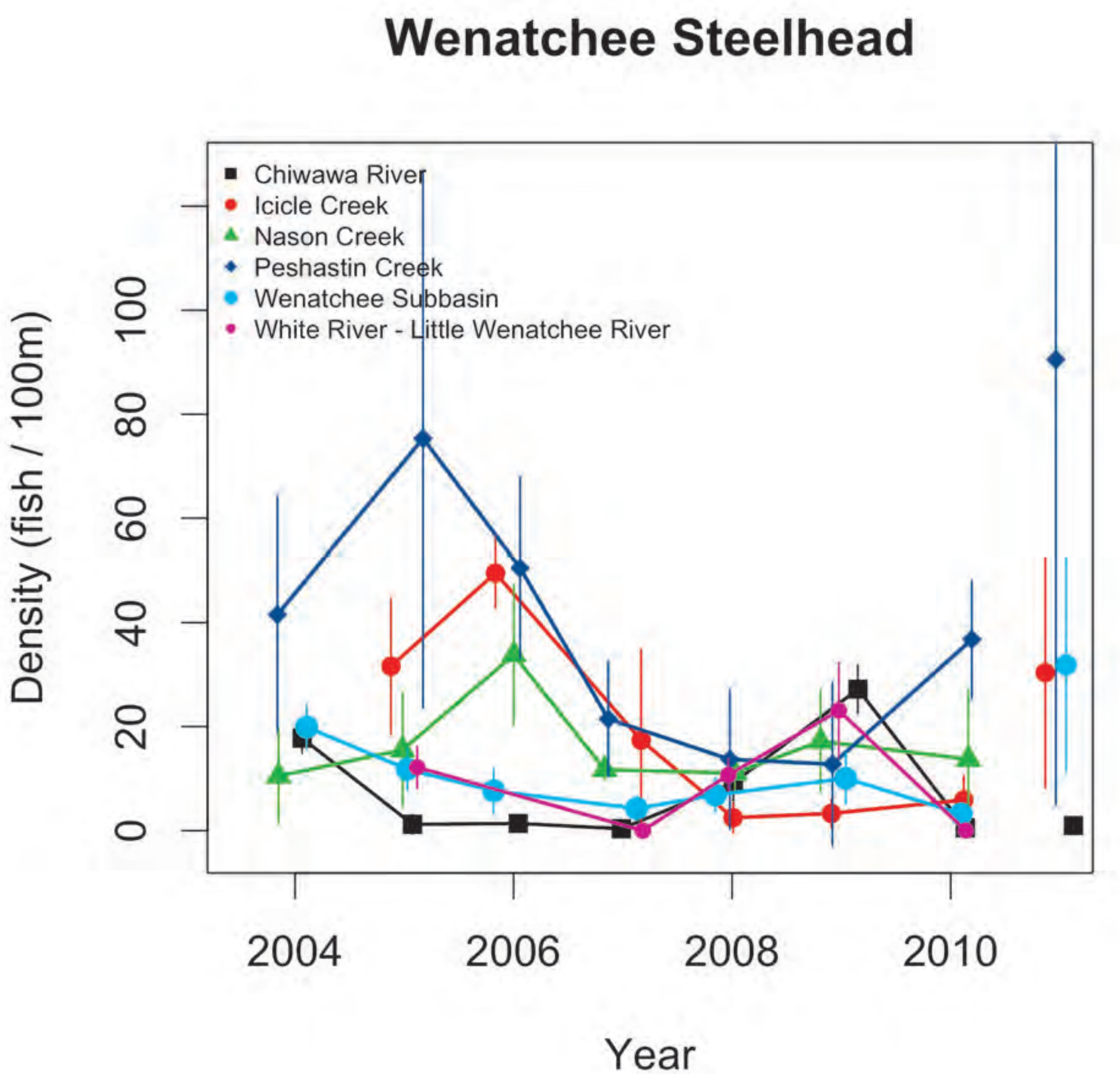


Figure 34. Density of juvenile steelhead standing crop in the Wenatchee River subbasin assessment units estimated from snorkel and electrofishing surveys (2005—2010) and mark-recapture/single pass electrofishing surveys (2011) from 25 annually sampled GRTS sites. Error bars represent plus and minus one standard error.

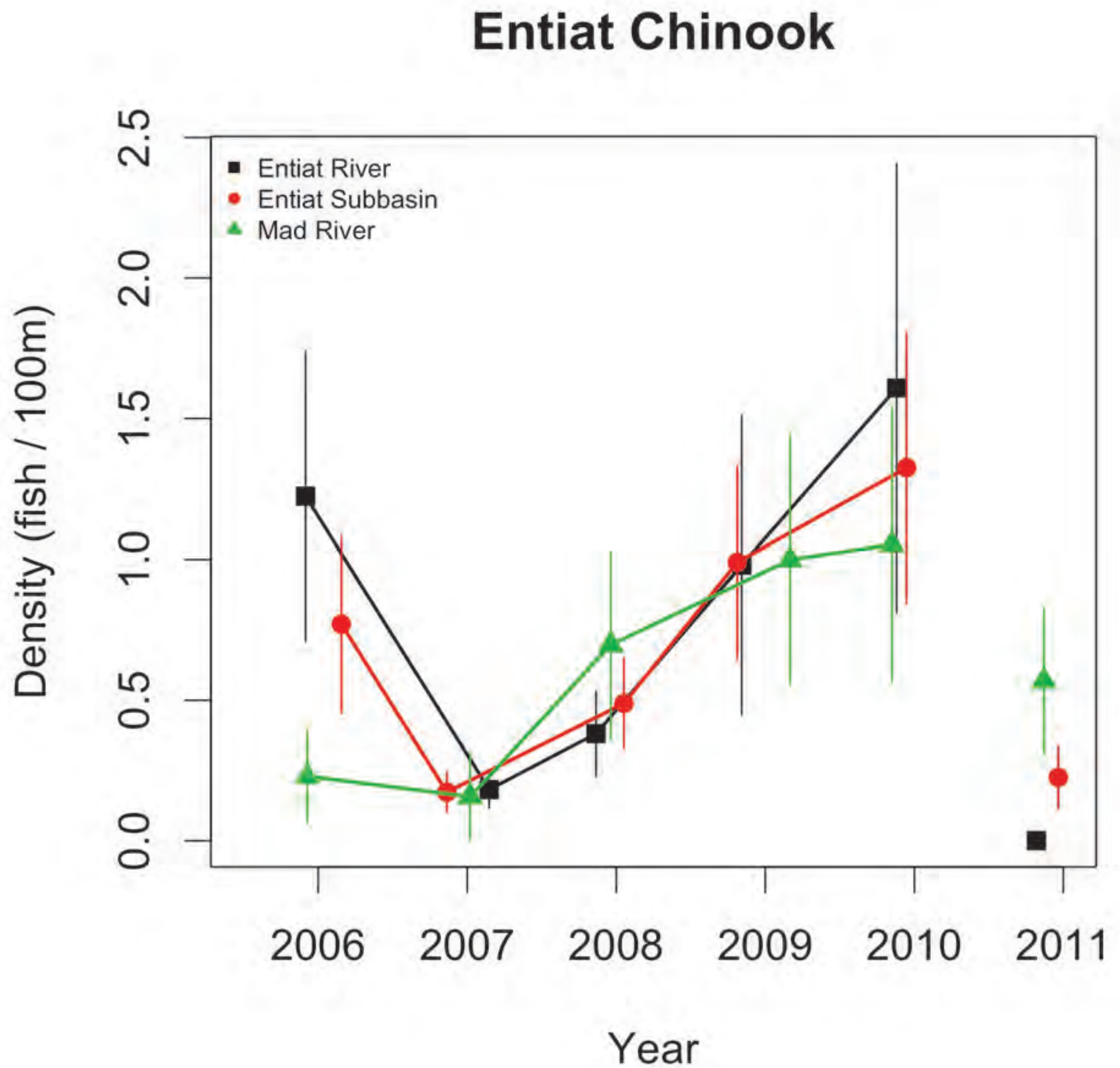


Figure 35. Density of juvenile Chinook standing crop in the Entiat River subbasin estimated from snorkel and electrofishing surveys (2005—2010) and mark-recapture/single pass electrofishing surveys (2011) from 25 annually sampled GRTS sites. Error bars represent plus and minus one standard error.

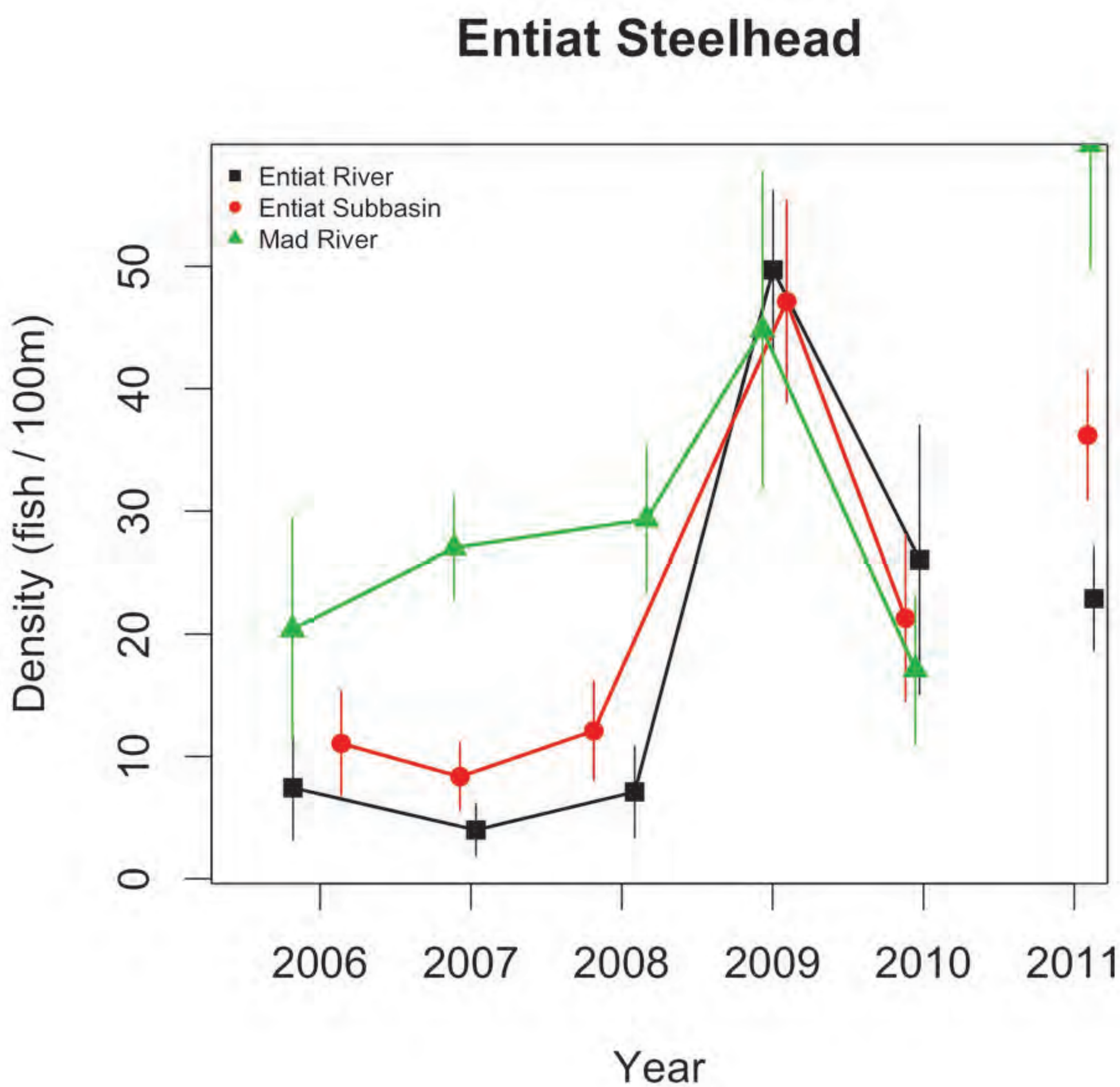


Figure 36. Density of juvenile steelhead standing crop in the Entiat River subbasin estimated from snorkel and electrofishing surveys (2005—2010) and mark-recapture/single pass electrofishing surveys (2011) from 25 annually sampled GRTS sites. Error bars represent plus and minus one standard error.

John Day Basin

In the John Day Basin, ISEMP is working closely with ODFW to provide standing crop estimates of juvenile summer steelhead at several spatial scales. Historically ODFW completed significant surveys comprising the major source of juvenile distribution and abundance information in the John Day. ISEMP has worked within both this historical and current infrastructure to test alternative approaches to monitoring and development of relevant indicators (Figure 37), and in 2011, ODFW incorporated some of the findings of ISEMP and adopted the CHaMP protocol to assess fish habitat. The challenge is to preserve the time series of past collection efforts while moving to adopt current strategies that integrates with other programs to address broader scale objectives such as those presented by ESA. Below we describe how ISEMP and ODFW plan to translate past information collected through ODFW's juvenile fish and fish habitat monitoring program so that the metrics generated are consistent with current monitoring efforts undertaken by both programs.

ODFW conducted basin-wide status and trend monitoring of juvenile steelhead and their habitat from 2004-2007 at 146 sites. The same survey design described earlier for the John Day steelhead redd surveys, was used to select sites to collect relative fish abundance and fish habitat information. The ODFW fish survey protocol was designed to assess species composition and distribution. These surveys generally consisted of a single pass by a trained snorkeler, except when water depths were judged to be too shallow for snorkeling when a single electrofishing pass was used.

During the summer of 2007, ISEMP assessed the detection efficiency of the single-pass electrofishing and snorkel surveys utilized by the ODFW survey design. This effort consisted of the collection of a mark-recapture (MR) population estimate of salmonid abundance by ISEMP technicians within reaches surveyed by ODFW because this approach has been shown to be accurate and precise (Rosenberger and Dunham 2006). ISEMP used these estimates to define the 'true' abundance of a reach to compare to ODFW's snorkel and electrofishing survey. A strong significant relationship between snorkel estimates and MR estimates was observed (Figure 38); however, efficiency was low with about 12% of the total abundance observed by snorkelers. A similar relationship was observed between MR and one-pass electrofishing (Figure 39).

The results from this study suggests that protocols used by ODFW to estimate juvenile abundance in the John Day are not very accurate but appear to be precise; that is, a consistent bias is observed

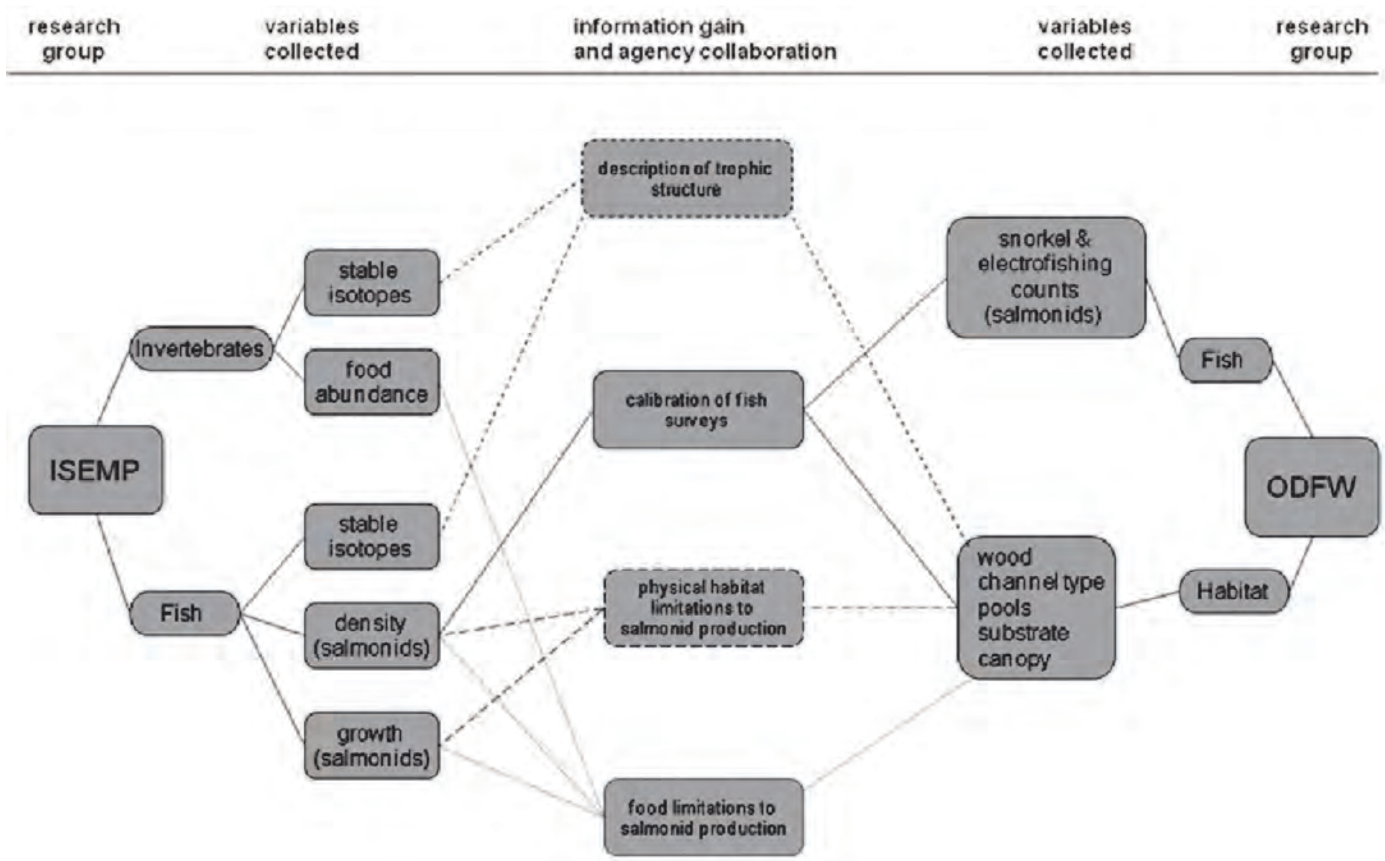


Figure 37: Integration of information collected in 2007 by ODFW and ISEMP.

and reflected in the low detection efficiency of the ODFW surveys. However, because this bias is consistent, these abundance estimates can be corrected for through the relationships developed from this study to derive absolute abundance. This is the key for translating the past time series of relative fish abundance to current fish monitoring protocols used by both ISEMP and ODFW in the John Day Basin, that now estimate absolute fish abundance.

ODFW did not receive funding for their fish and fish habitat monitoring program from 2008-2010. In 2009, ISEMP conducted these surveys at 30 sites. The surveys differed from ODFW's past surveys in that fish were sampled using a MR approach based on the calibration study, and the ODFW habitat protocol was expanded to include other habitat protocols to assess the ability to create cross-walks between different approaches. Some of the results from these efforts were used in the development of the CHaMP protocol to ensure it was consistent with past surveys. Now CHaMP is currently implemented by ODFW and ISEMP in the John Day, and several of the metric generated by the historic and current protocols are directly transferable.

In 2011, both ISEMP and ODFW implemented common fish and fish habitat protocols under a hierarchical survey design that was implemented to not only address status and trends of salmonids and their habitat but also provide potential information for the development of fish-habitat relationships, limiting factor analyses, prioritization and planning of restoration and management, and pre-project information for effectiveness monitoring programs. In addition, the strategy also provides data to help determine the appropriate sampling scales for monitoring, protocol precision, accuracy and efficiency, protocol comparisons, and development of protocol cross-walks.

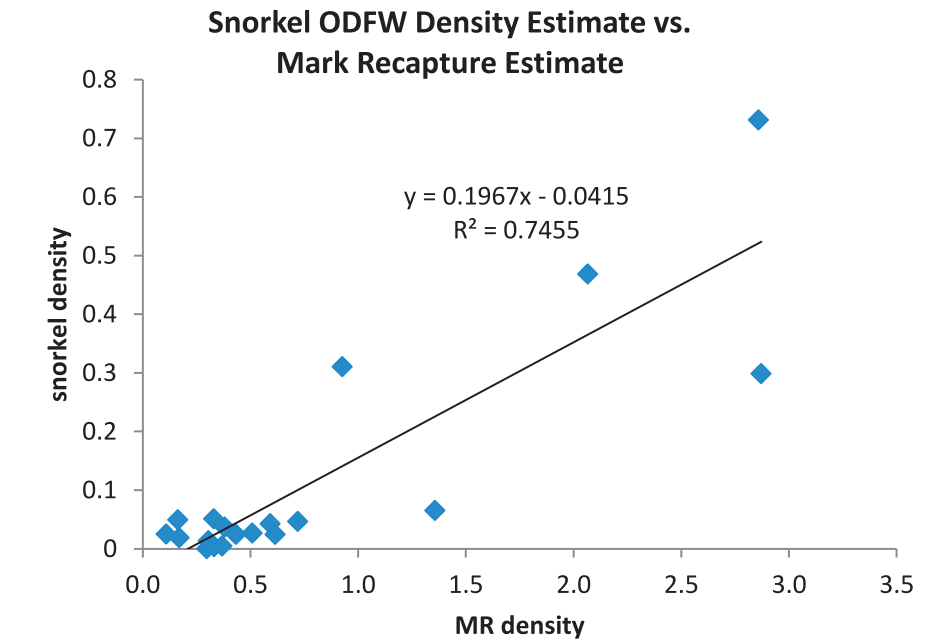


Figure 38. The number of juvenile salmonids over an 100 m reach (expressed as no./m²) based on mark-recapture methods were compared to the number observed snorkeling pool habitat.

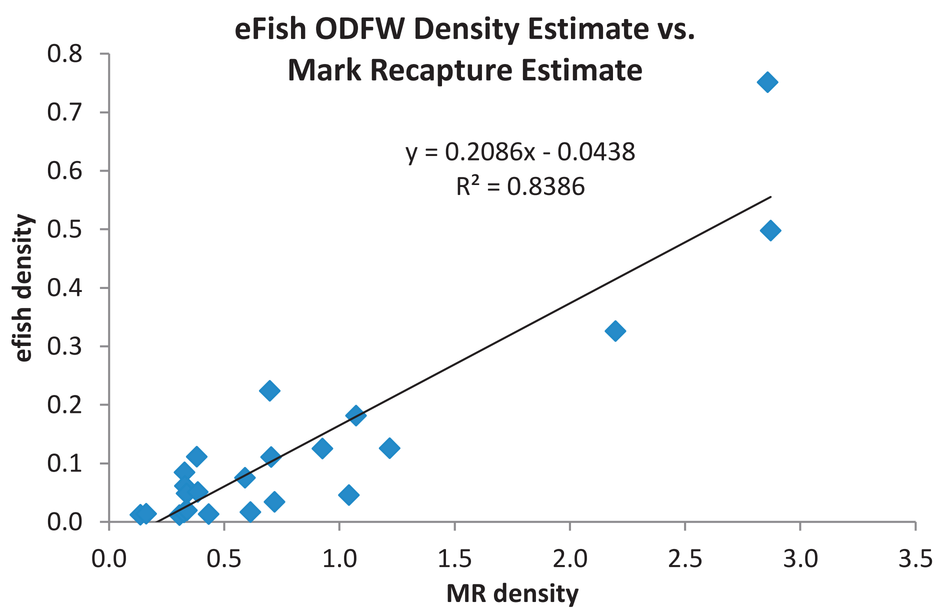


Figure 39. The number of juvenile salmonids over an 100m reach (expressed as no./m²) based on mark-recapture methods were compared to the number observed electroshocking pool habitat.

ODFW and ISEMP are conducting juvenile fish and habitat surveys between and within watersheds stratified by populations throughout the John Day basins. ODFW is conducting status and trend basin-wide mark-recapture fish and CHaMP habitat surveys at 40 sites using the CHaMP GRTS survey approach (Figure 40). ISEMP will monitor selected watersheds at a higher resolution also using a rotating panel design, with Murderers and Bridge Creek watersheds sampled annually (as part of the Bridge Creek IMW), and 2 watersheds sampled every year for 3 years. After 3 years, we will move to the next set of watersheds, for a total of 6 unique watersheds beyond Bridge and Murderers IMW over a 9 year period. In each watershed, site surveys and "continuous" surveys for fish and habitat will be conducted. Nine site surveys using the CHaMP protocol for fish habitat, and a mark-recapture effort for salmonids, will occur in each of the watersheds. The selection of these watersheds and in particular the sites within these watersheds are statistically robust in that they can be rolled into John Day basin-wide site survey summaries with the appropriate weightings.

From the information collected both previously and recently, ISEMP, in collaboration with ODFW, will estimate a time-series of basin-wide juvenile steelhead standing crop. Combined with ODFW estimates of adult abundance, we can begin to estimate freshwater production (juveniles/adult) through time to evaluate how these populations are responding to changes in management and stream restoration throughout the John Day Basin. In addition, because fish and fish habitat data is paired, we can use this information to establish fish-habitat relationships. This will also inform relationships used in the Watershed production model. Finally, we hope that through this hierarchical assessment we can establish both the scale and the factors that most influence steelhead abundance. This may allow for much more efficient monitoring designs that can detect how changes in landuse and restoration influence fish habitat and ultimately fish populations.

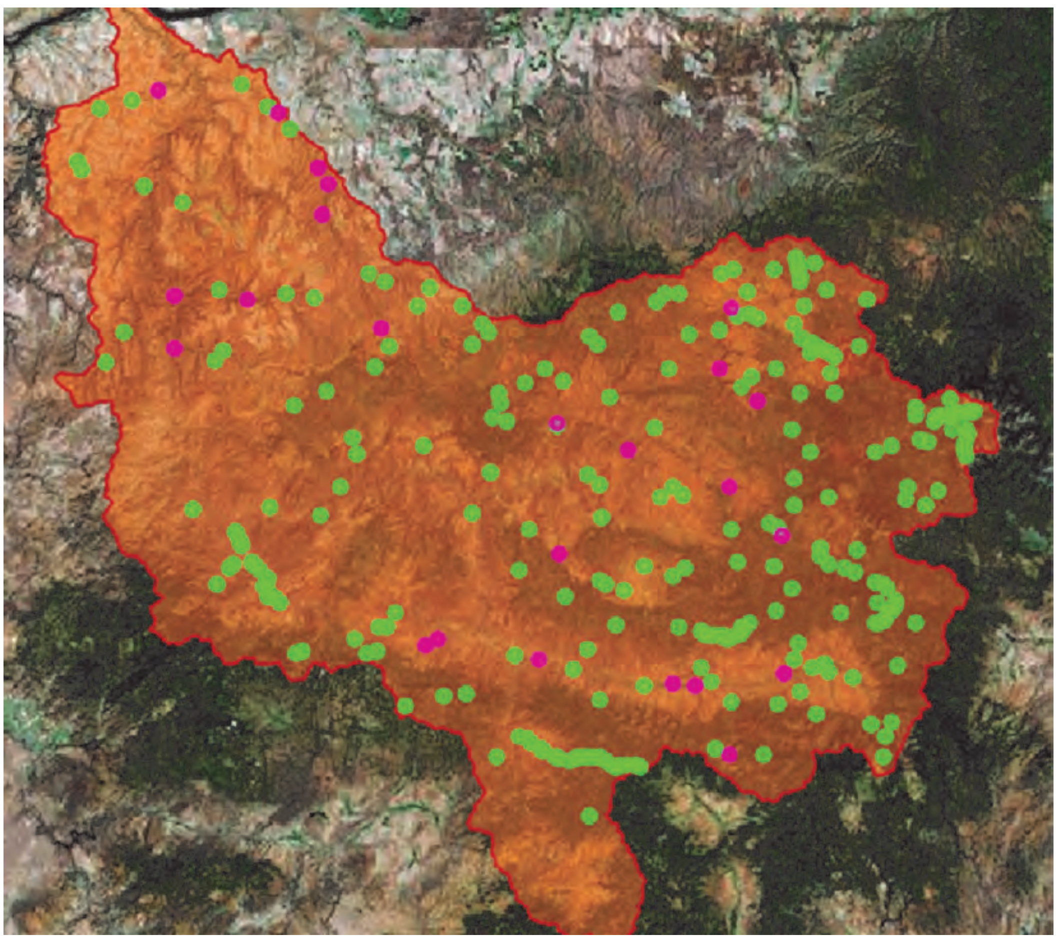


Figure 40. The distribution of fish surveys site throughout the John Day Basin where ODFW and ISEMP are using mark-recapture electrofishing surveys to estimate juvenile steelhead abundance. CHaMP surveys will also be conducted at several of these sites. Pink circles represent sites sampled on an annual basis, whereas green circles represent sites visited every 3 years. High density sites represent higher intensity surveys conducted.

Emigration Estimates

Rotary screw traps are used throughout the Columbia Basin to estimate total out-migration (emigration) of juvenile Chinook salmon and steelhead from a tributary. This information is used to estimate total juvenile production from a tributary or population, smolt-to-adult return rates, egg-to-smolt survival, and to study life history characteristics. The temporal and spatial extents of the estimate are usually dependent on logistics associated with access and environmental conditions such as high flows and ice. Traps are usually place in locations that are accessible for field crews and sometimes do not estimate the total population or sub-population of interest. Although the goal is to collect all juvenile migrants across the year, the operation of rotary screw traps are often interrupted during the winter when the rivers freeze, high flows when it is too dangerous to operate, and by budgetary constraints when only a portion of the week is sampled. Because of these logistical constraints, the percentage of a population or tributary juvenile emigration estimated will vary between years.

Generically, the emigration estimate is derived by expanding the number of fish captured during a day by the trap efficiency. Trap efficiency is calculated by releasing a known number of fish upstream of the screw trap and calculating the "efficiency" of subsequent collection of these fish that pass the trap. Although, each agency or subbasin has a different model to calculate the total emigration estimate, the field methods are very similar. Depending on species, the emigration or out-migration estimate can also be partitioned into juvenile age of migration or life-stage.

Rotary Screw Trap Estimates from the Salmon Subbasin

Salmon Basin ISEMP directly funds three screw traps in the Lemhi and South Fork Salmon River. The traps in the Lemhi River are focused on developing a single emigration estimate for the Lemhi Steelhead and spring/summer Chinook populations and one trap in the Hayden Creek, the control tributary for the Lemhi River restoration project. The South Fork Salmon River is monitored using the Secesh River trap, which estimates a single population of steelhead and one population of spring/summer Chinook salmon.

Emigration estimates are calculated using the mark recapture program developed by Steinhorst et al (2004). Efficiency estimates were stratified into homogeneous periods to calculate abundance. Because of the sample size, Chinook salmon abundance was estimated for parr,

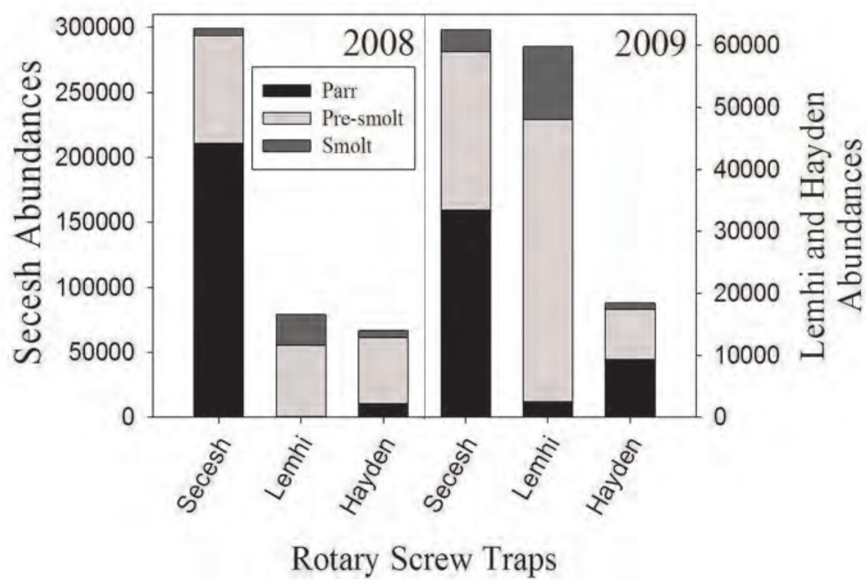
Table 16. Spring/summer Chinook salmon brood year total emigration estimates by populations and trap for the Secesh, Lemhi and Hayden Creek, Idaho.

Trap	Percentage of Population	Brood Year	Chinook Abundance	CV	Lower 95% CI	Upper 95% CI
Secesh	100%	2008	299,890	8.1%	265,455	347,526
Lemhi	100%	2008	16,298	8.2%	14,126	18,909
Hayden	40%	2008	14,731	4.4%	11,368	16,012
Secesh	100%	2009	289,659	7.3%	260,072	331,154
Lemhi	100%	2009	57,301	8.1%	49,763	66,387
Hayden	40%	2009	18,430	6.4%	16,330	20,731

Table 17. Steelhead migration year total emigration estimates by populations and trap for the Secesh, Lemhi, and Hayden Creek, Idaho.

Trap	Percentage of Population	Migration Year	Steelhead Abundance	CV	Lower 95% CI	Upper 95% CI
Secesh	98%	2009	16,964	59.0%	9,377	36,591
Lemhi	95%	2009	19,530	11.9%	15,439	24,088
Hayden	30%	2009	17,580	16.7%	13,282	23,325
Secesh	98%	2010	13,599	35.4%	8,656	23,039
Lemhi	95%	2010	37,170	28.5%	26,540	57,938
Hayden	30%	2010	8,733	20.1%	6,604	12,175
Secesh	98%	2011	18,371	22.9%	12,839	26,619
Lemhi	95%	2011	27,202	19.7%	20,029	37,703
Hayden	30%	2011	7,332	28.7%	4,879	11,455

presmolt, and smolt by brood year. Steelhead age at migration past the rotary screw trap varies significantly and coupled with a low capture/recapture rate prevents accurate brood year classification and the resulting steelhead estimates were calculated by migration year. Spring/summer Chinook salmon brood year total emigration estimates by populations and trap are shown in Table 16. Figure 41 illustrates the juvenile Chinook (top panel) and steelhead (bottom panel) life-stage specific estimates associated with each trapping location. Population and trap steelhead juvenile emigration migration year estimates are shown in Table 17.



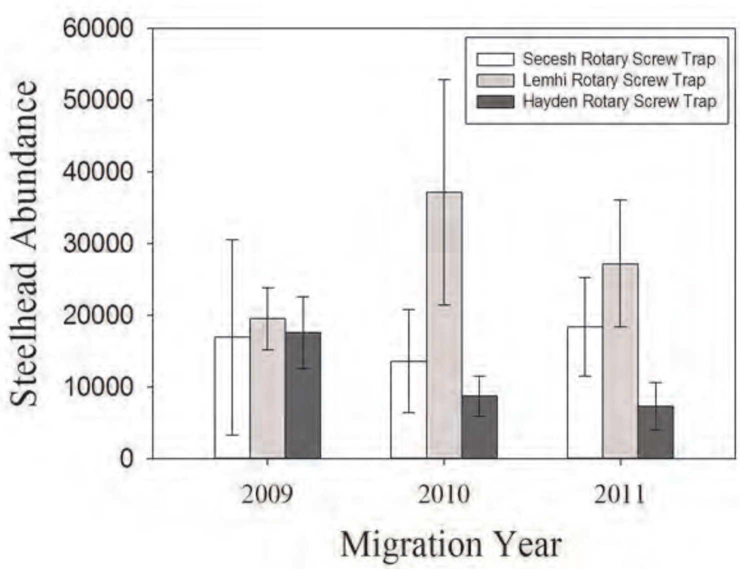


Figure 41. Chinook brood year (top panel) and steelhead migration year (bottom panel) abundance estimates from rotary screw traps operated in the Secesh, Lemhi and Hayden Creeks in the Salmon River subbasin.

Rotary Screw Trap Estimates from the Upper Columbia Subbasin

ISEMP provided funds in the past, or currently fully funds, two screw traps in the Wenatchee/Entiat Rivers. In cooperation with the WDFW, ISEMP assisted in funding operations of the Monitor Bridge screw trap (rkm 9.6) on the Wenatchee from 2005-2009 in order to expand its window of operation and gain a better estimate of subbasin production (the trap generally operates between February and August). ISEMP funds the operation of the Entiat screw trap, located at rkm 0.8, which is operated by the U.S. Fish and Wildlife Service Mid-Columbia Fisheries Resource Office. Specific to the Upper Columbia, estimates are focused on the smolt life-stage.

Estimates of smolt emigration were estimated utilizing several methods, depending on the trap and/or species. The Wenatchee screw trap results are calculated using a regression model which relates flow to trapping efficiency. Depending on the flow the trap is located in different locations across the river's cross-section with independent trials of Chinook juvenile releases used to calculate trap efficiency. The subsequent estimates were based on the knowledge that at any given flow or trap position, a specific proportion of the fish would be captured. For steelhead, mark-recapture methods were used to estimate emigration. When too few efficiency trials for a given position or species were conducted, efficiency trials from previous years were incorporated into the regression model. The flow-based estimation model

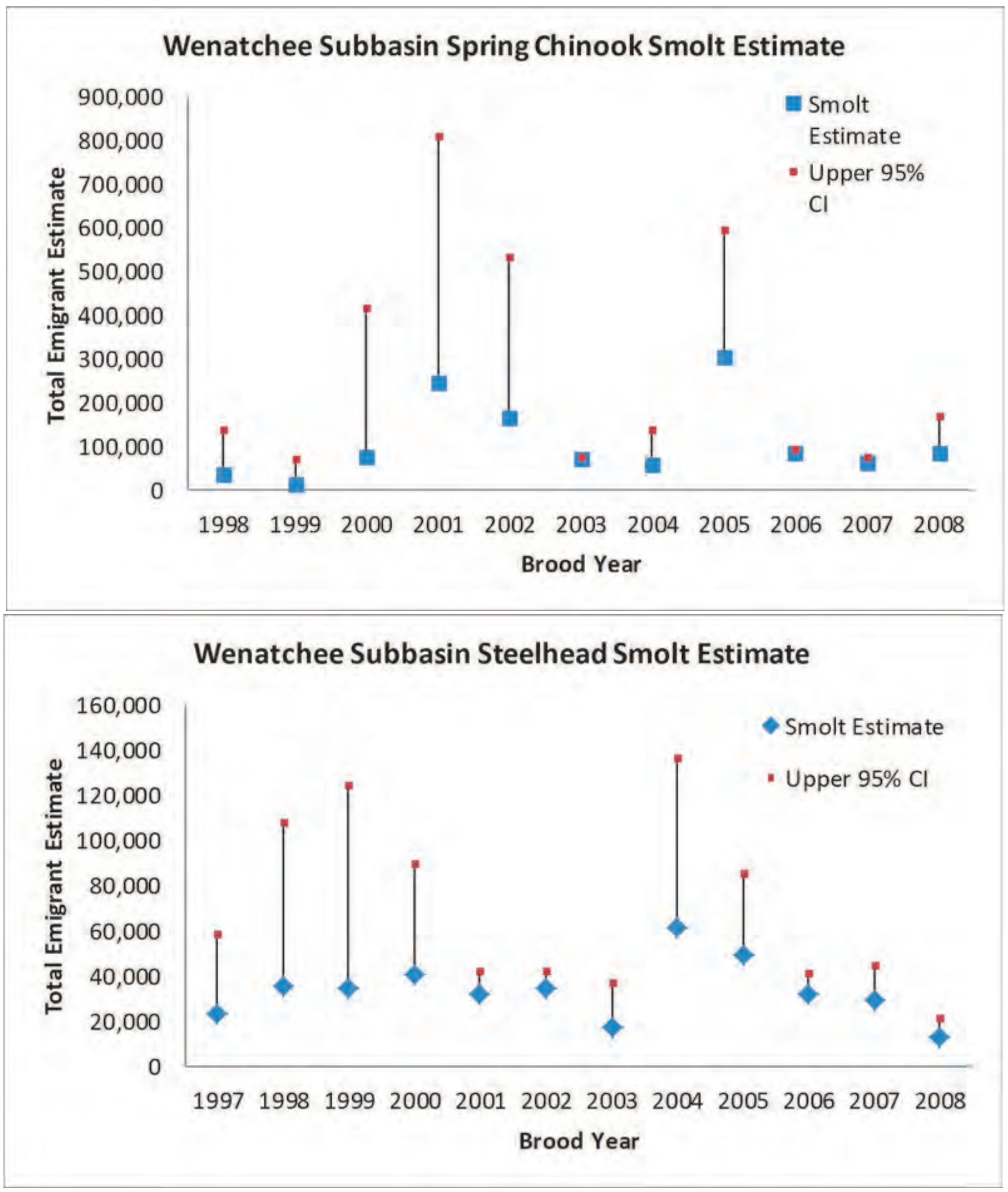


Figure 42. Spring Chinook (top panel) and steelhead (bottom panel) estimates of smolts outmigrating from the Wenatchee subbasin caught in a rotary screw trap at the Monitor Bridge on the Wenatchee River 1997—2008. Error bars are 95% confidence intervals. Lower CI not shown as it crosses 0. (Data provided by the Washington Department of Fish and Wildlife).

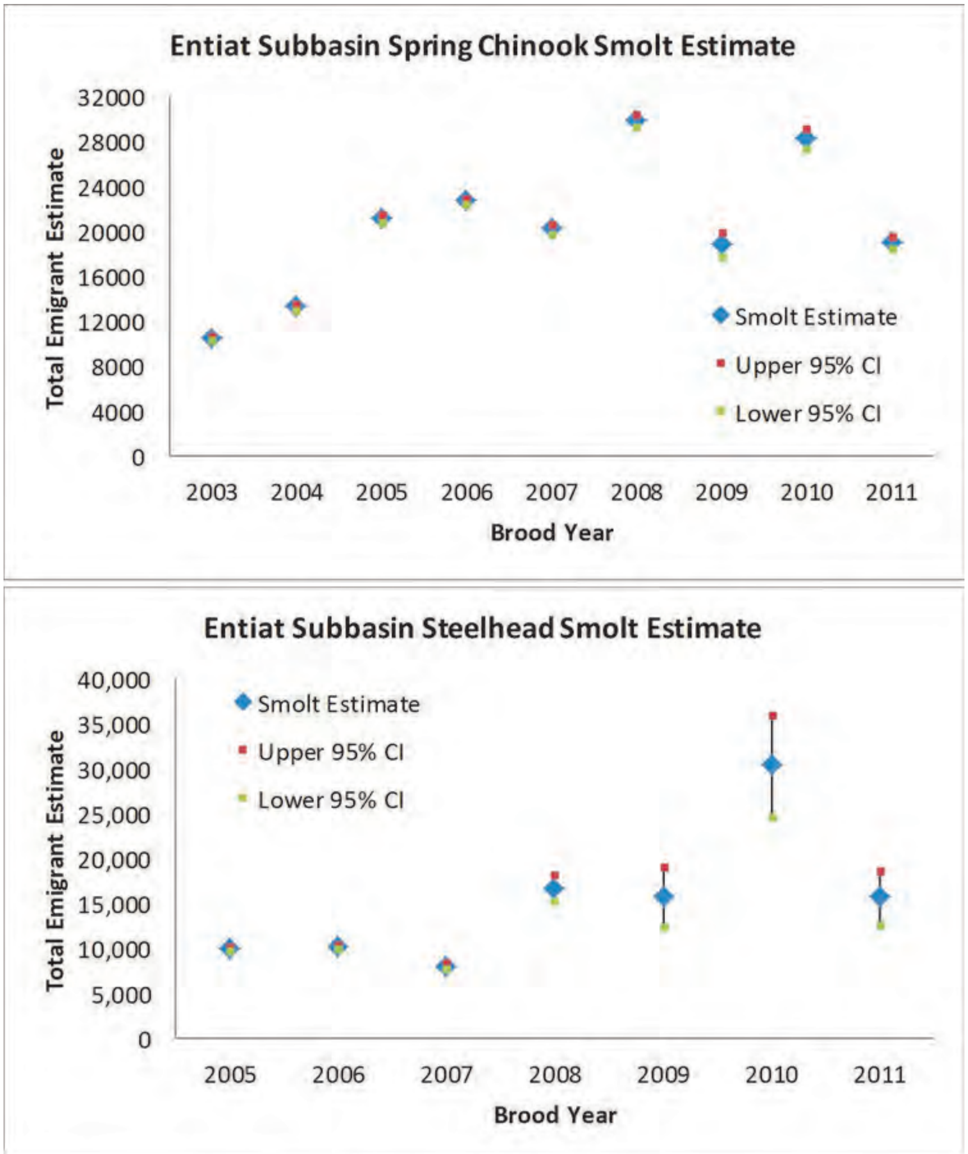


Figure 43. Spring Chinook (top panel) and steelhead (bottom panel) estimates of smolts outmigrating from the Entiat River subbasin caught in a rotary screw trap at the mouth of the Entiat River 2005—2011. Error bars are 95% confidence intervals. (Data provided by USFWS Mid-Columbia Fishery Resource Office).

results for spring Chinook and mark-recapture models for steelhead are shown in Figure 42.

Similarly to the Wenatchee screw trap, estimates of natural juvenile salmon emigration from the Entiat watershed were derived for wild yearling spring Chinook, wild subyearling spring Chinook and wild steelhead. Emigration estimates were calculated using the flow-based regression analysis of the relationship between trap efficiency (dependent variable) and flow (independent variable). Results are shown in Figure 43.

It should be noted that the USFWS comments in its 2011 annual report that "Calculations of production estimates using rotary screw traps are standardized between monitoring agencies within the Upper Columbia basin to increase the consistency and usefulness of these annual estimates. A common consensus among researchers in the Upper Columbia is that a fundamental problem exists with the equation used to estimate variance of point estimates. Our current calculations may not adequately account for all variables that influence the confidence intervals associated with our estimates. Although we feel our estimates are accurate and applicable to resource management needs we will continue to proactively review the parameters included in these calculations in order to improve methodology." (Desgroseillier et al. 2011).

Assessing the Performance of Estimating Abundance from Rotary Screw Trap Data

BiOp performance metrics are based on “fish in/fish out” metrics where “fish out” is usually a measure of emigrating salmon or steelhead enumerated at rotary screw traps. Screw trap estimates of emigration are generated at many locations throughout the Columbia Basin. However, these critical estimates are often fraught with high levels of estimation error, this error is often not well reported, and managers may not realize the level of imprecision in these estimates. Additionally, methods to reduce the error in estimation can be expensive and ineffective. ISEMP has been conducting a series of investigations to highlight the importance of these generally overlooked weaknesses and to suggest improvements that will reduce sampling costs while improving the value of these critical estimates of fish emigration. The following section describes the results from the first of these three analyses. All three analyses will be finalized and reported by late spring 2012.

Interpreting the accuracy of rotary screw trap abundance estimates can be difficult as there is usually no easy way to independently verify migrant abundance by other means (although ISEMP is developing the use of PIT tags and PIT tag detection arrays as an alternative), and many uncertainties affect estimates of juvenile migrant abundance. Screw traps do not capture all emigrating fish and the proportion of fish that are captured is unknown. This “trap efficiency” must be estimated in order to generate total capture estimates; unfortunately, trap efficiencies vary daily and seasonally in response to many factors including, especially, stream flow.

Estimates of trap efficiency require many assumptions. However, many (if not most) trapping programs lack sufficient information to judge how violations of assumptions affect their estimates and confidence interval coverage. For example, even during times of similar environmental conditions, trap efficiencies can vary on a daily basis (Figure 44). Unless trapping programs are marking and releasing fish daily, they do not have the opportunity to identify and account for daily variability in trap efficiencies in their mark-recapture designs. Such daily trap efficiency tests can be expensive and unnecessary at time when daily variability is low.

Another challenge to generating useful estimates of the number of emigrating salmonids is that, while several

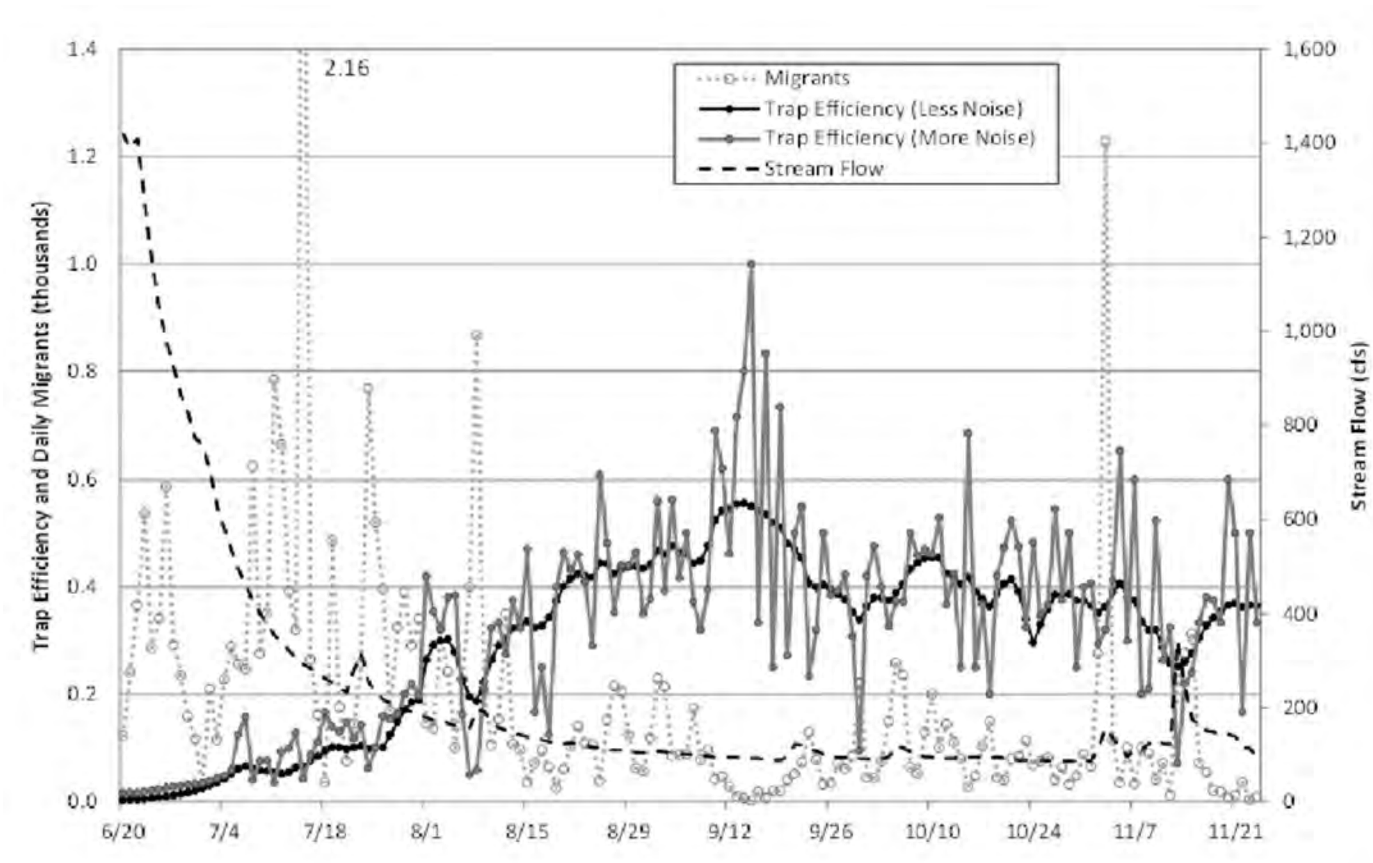


Figure 44. Johnson Creek known universes of daily migrants, trap efficiency and stream flow.

statistical methods exist to help make these estimates, the performance of these methods can also be significantly affected by the types of assumptions made. Multiple mark-recapture methods (e.g., Bayesian Time Stratified Population Analysis, Stratified Mark-Recapture (with several different methods for specifying strata); Modeled Trap Efficiency (based on flow); Bayesian Time Stratified Population Analysis; and Pooled Peterson) can be employed, but their relative performance for estimating abundance, their bias, and whether the confidence intervals generated accurately capture true abundance, has been unknown.

In conjunction with cooperators, ISEMP has explored some of these sources of variability, by evaluating these emigration models and their underlying assumptions, including:

1. The relative performance of common and emerging abundance estimation methods;

2. The influence of increased mark-recapture effort on estimates;

3. The effect that daily variability has upon estimates; and

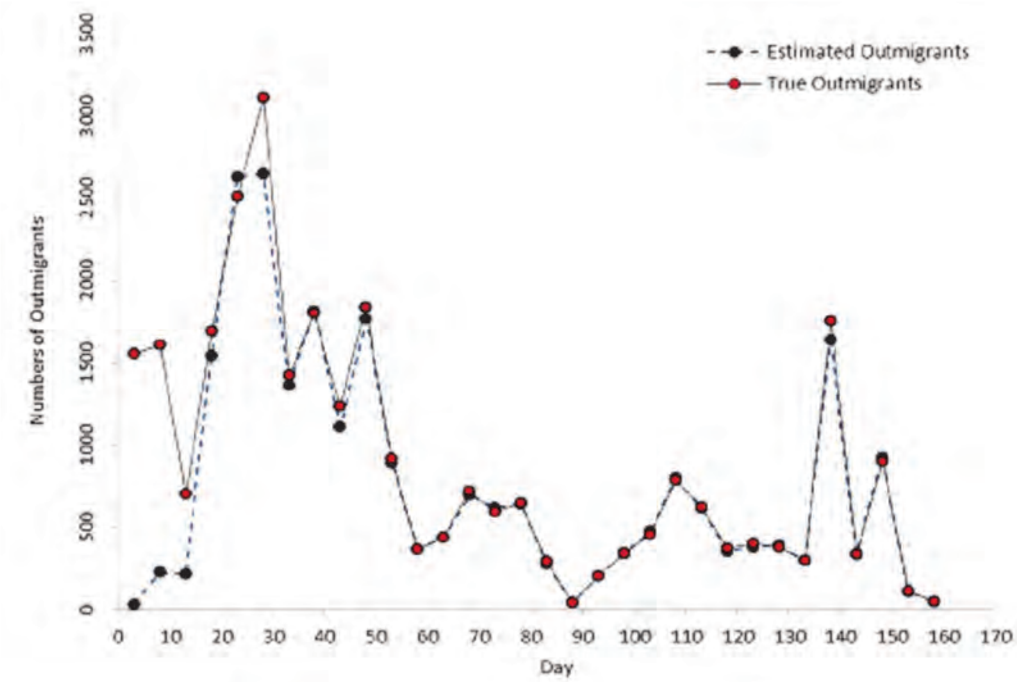
4. The potential benefits of daily marking and release of fish where daily variability is higher.

We used a simulation approach to explore abundance estimator performance in two rivers (Entiat River and Johnson Creek, a tributary of the South Fork Salmon River) with different physical characteristics, migrant abundances, and trap efficiencies where migrant trapping programs are established. Daily migrants and trap efficiencies were generated and the bias and confidence interval coverage (“coverage” describes the percentage of simulations where the confidence interval for the estimated abundance actually includes the known level of abundance:. Coverage values greater than 95 percent would be excellent and values greater than 80 are acceptable, while lower values reflect decreasing utility) for estimators were compared. Two strata scenarios and different numbers of trap efficiency trials (including those with and without the use of additional fish to supplement trap efficiency trial release numbers) were also used in simulations to determine their influence on abundance estimates.

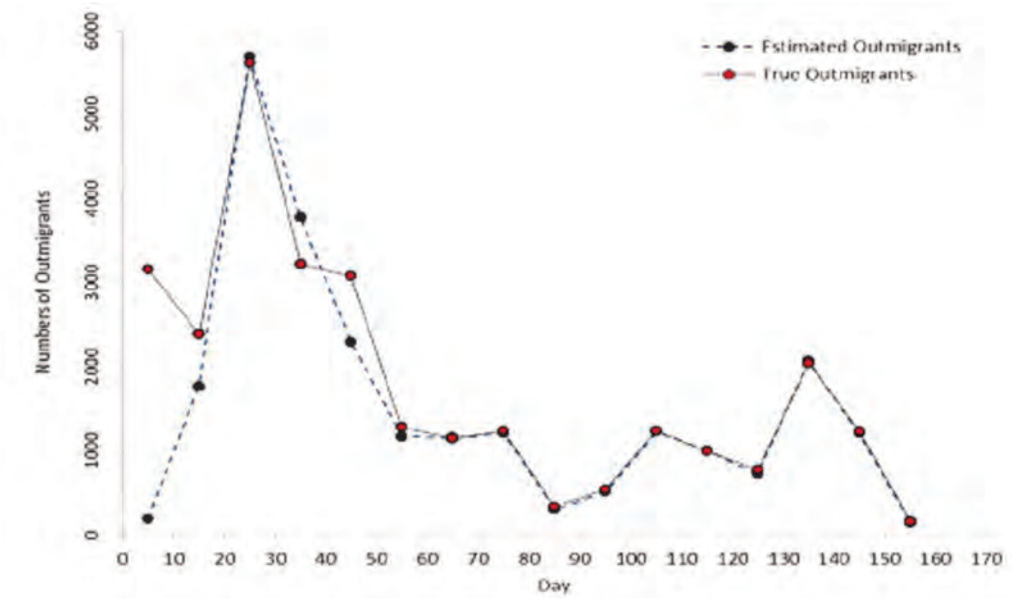
Our results suggest that estimating downstream migrant abundance using screw traps and mark-recapture methods *can* provide accurate estimates of abundance enabling the generation of metrics such as freshwater production and the tracking of trends over time, but generic sampling designs for allocating mark-recapture effort (timing and amount) should be used with caution. Abundance estimates can be significantly biased as a result of violations in mark-recapture assumptions when these assumptions are not addressed or when violations of assumptions go undetected. Furthermore, allocating more effort to trap efficiency trials, either by conducting more trials or supplementing the numbers of fish used in these trials, may not reduce the bias in abundance estimates. This was especially apparent when exploring the test universes that had relatively high daily variability in trap efficiencies.

Some of these points are illustrated in the following series of figures. Figure 45 illustrates two scenarios that differ in the length of strata used for emigrant abundance estimations. One might suspect that more trap efficiency estimates (32 versus 16) resulting from shorter strata length (5 days versus 10 days) would improve abundance estimation. However, the high bias and poor confidence interval coverage when strata length is large and effort is low (10 days/16 strata) actually gets worse when strata size is made smaller and effort is increased (5 days/32 strata). Fortunately, this counter intuitive result can be improved in some scenarios where creative trap efficiency trial designs are used. For example, Figure 46 shows one possible alternative design (a mix of short and long strata and mixed use of supplementation) that significantly reduces the bias (from -14.0 percent and -15.0 percent to -2.6 percent) and increases the confidence interval coverage from 35.5 and 57.4 percent to levels near 95 percent.

Note that this alternative design is not a prescription for any particular trapping program but is merely one example of how this simulation analysis shows practical ways to reduce effort (one trap efficiency trial per 10 days of sampling in the later portion of Figure 45 top panel versus one trap efficiency trial per 5 days of sampling in Figure 45 bottom panel). Other alternative designs can be identified with similar benefits and, indeed, will be discussed in the final report on these analyses. These ISEMP results are meant to illustrate the points that smart and creative allocation of effort can significantly improve estimates of abundance of emigrating salmonids



% Bias	Coverage
-15.0	35.5



% Bias	Coverage
-14.0	57.4

Figure 45. Scenarios showing the effect of two trap efficiency tests that differ in the length of strata and number of trap efficiency estimates used for emigrant abundance estimations. Top panel shows shorter strata length (5 days) and more trap efficiency estimates (32) compared with lower panel that shows longer strata length and low effort (10 days/16 efficiency estimates).

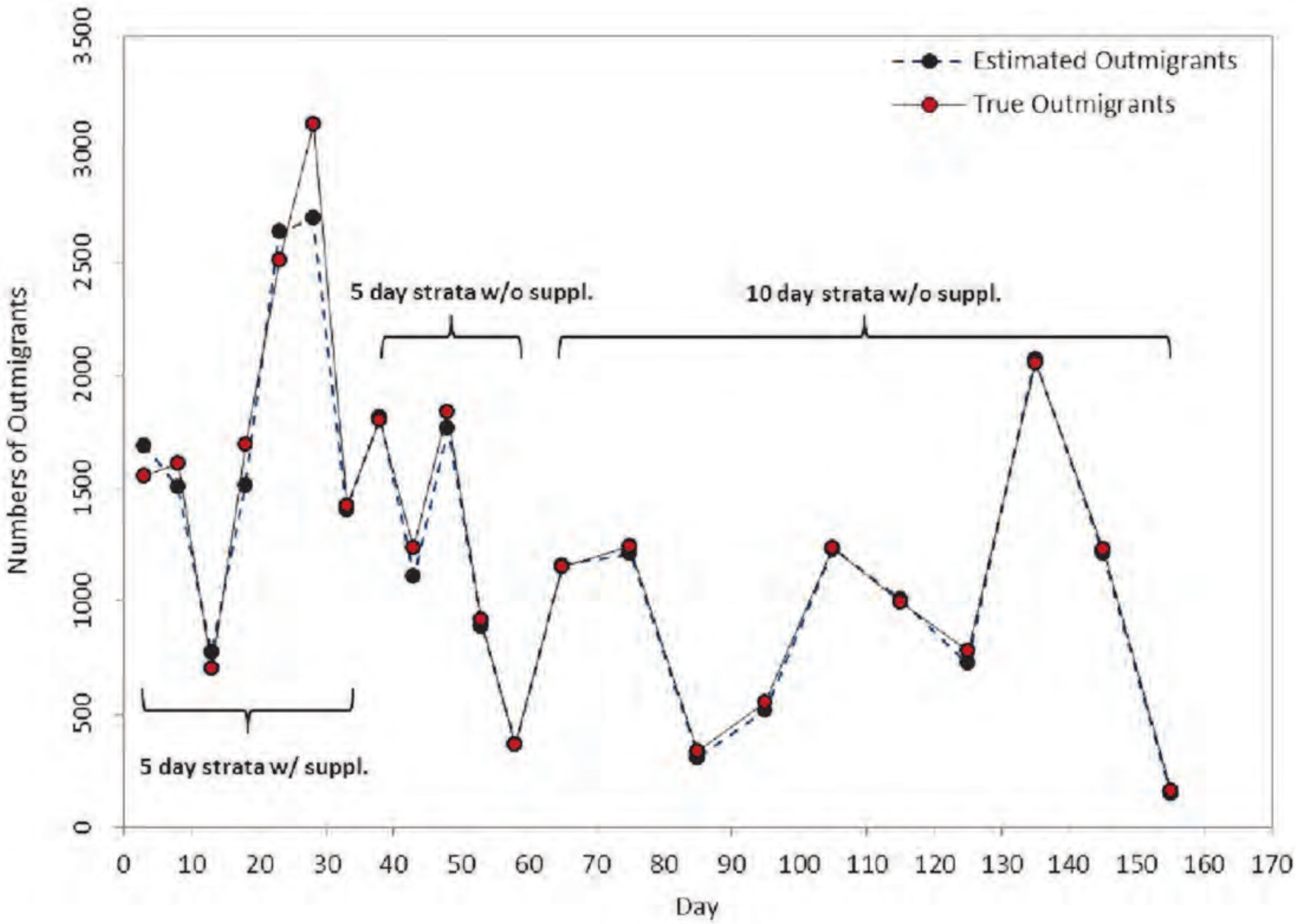


Figure 46. Combining various stratified mark-recapture methods to estimate abundance for the Johnson Creek smoothed universe. Days 1-35, 5 day strata with supplementation; days 35-60, 5 day strata without supplementation; days 60-160, 10 day strata without supplementation.

Rotary Screw Trap Estimates from the John Day Basin

As with the basin-wide adult steelhead survey, ISEMP John Day depends on ODFW to evaluate the basin-wide production of juvenile steelhead outmigrants (reported in DeHart et al. (2012)). ODFW has been operating rotary screw traps and conducting seining surveys for this purpose since 2004. Generally, screw traps have been operated at three locations; two screw traps located in the Upper Mainstem John Day River, referred to as the Mainstem, one trap on the South Fork John Day River, and one on the Middle Fork John Day River, and seining occurs when possible at the Mainstem John Day River (rkm 274-296, Figure 47). In some years, a trap has been attempted in the North Fork John Day River as well. The Mainstem, South Fork, and Middle Fork traps are all located downstream of the majority of known spring Chinook and steelhead spawning habitat. Some summer rearing and spawning does occur in Bridge Creek (Bouwes et al. 2010) and likely occurs in other tributaries downstream of our collection sites.

Trapping occurs all year except in July and August and traps are fished four days a week and checked daily. ODFW assumed that all fish captured were migrants and captured juvenile spring Chinook and steelhead migrants are PIT tagged.

Trapping efficiency was estimated separately for each fish species at each screw trap site by the proportion of marked fish released upstream subsequently recaptured in the trap (Thedinga et al. 1994). Trap efficiency estimates are used to stratify the trapping data into homogeneous periods. A Bailey estimator is used to estimate migrant abundance (Steinhorst et al. 2004) for each strata. Abundances estimated within strata are expanded for days when the traps were not operated through the assumption that the estimated mean daily number of migrants during each sampling period also migrated on each day that the trap was not operated.

Estimates of the number of outmigrants shown in Figures 48-50 are from the ODFW 2011 annual report (Dehart et al. 2012). This information can then be used to estimate freshwater production expressed as smolts per redd (Figure 51). PIT tagged fish can also be used to estimate the John Day dam/Bonneville dam smolt-to-adult return rate (Figure 52). ISEMP will use this information to parameterize the Watershed Production Model for the John Day Basin.

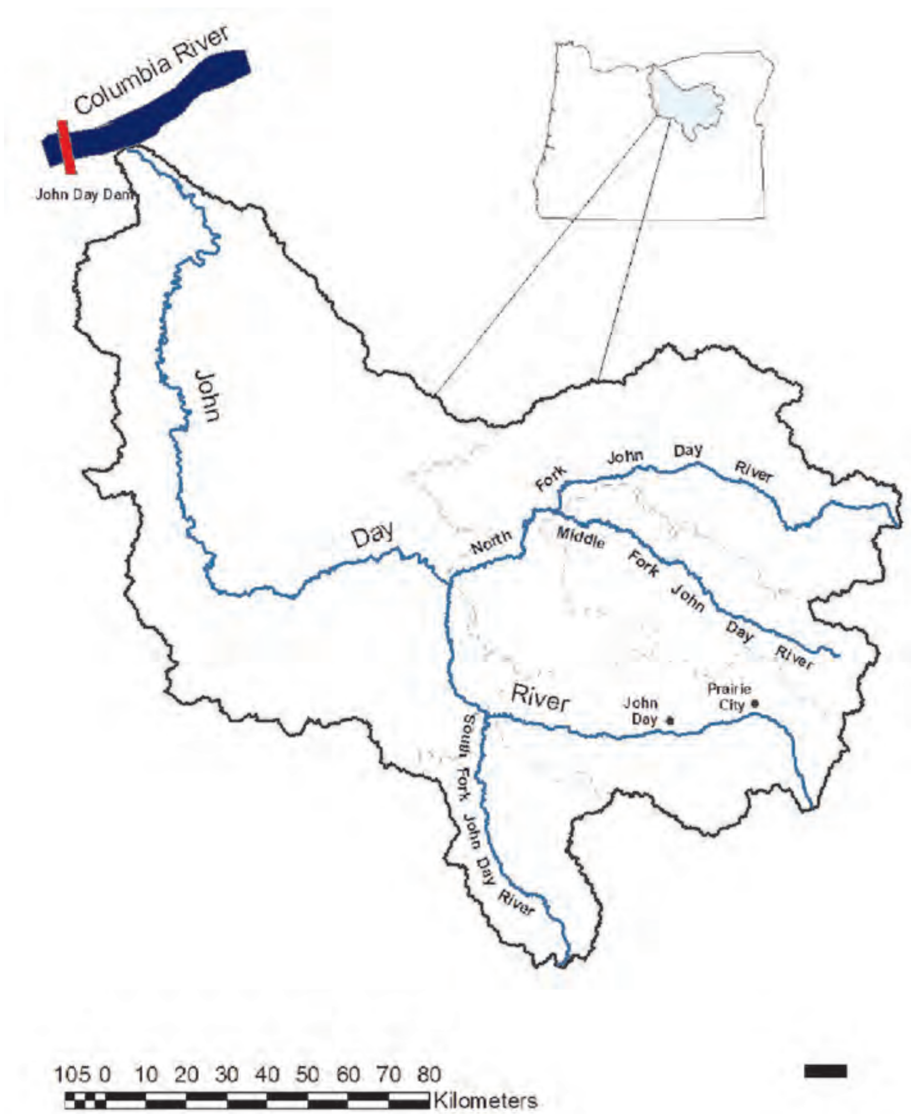


Figure 47. Map of John Day River basin. Dashed line denotes watershed boundary. Arrows indicate approximate locations of rotary screw traps and the circle indicates our Mainstem seining reach between Kimberly and Spray, OR.

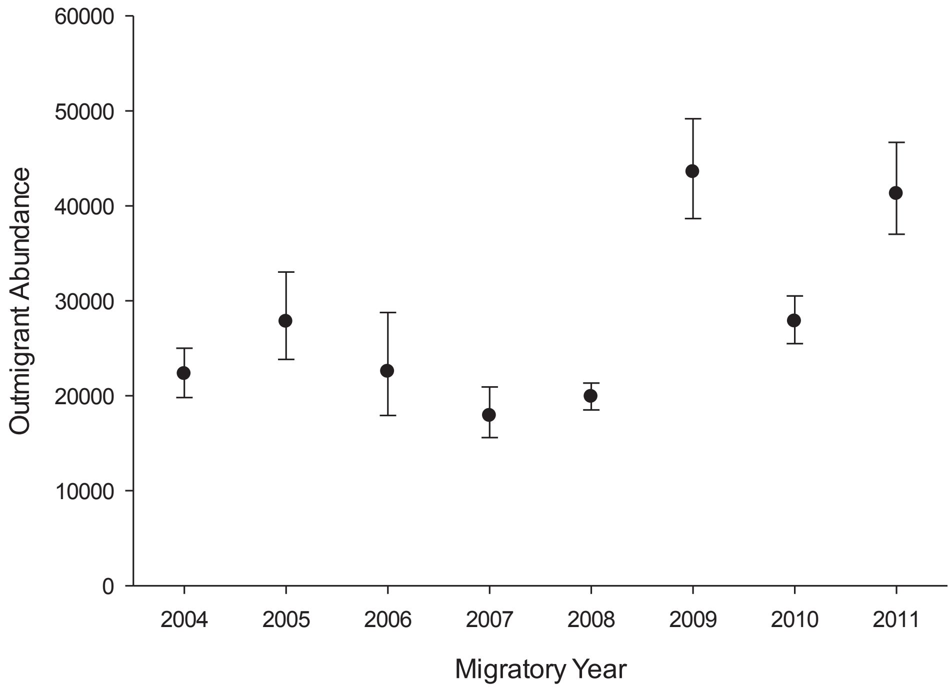


Figure 48. South Fork trap summer steelhead abundance estimate by migratory year. Error bars are 95% confidence intervals. Reported in DeHart et al. (2012).

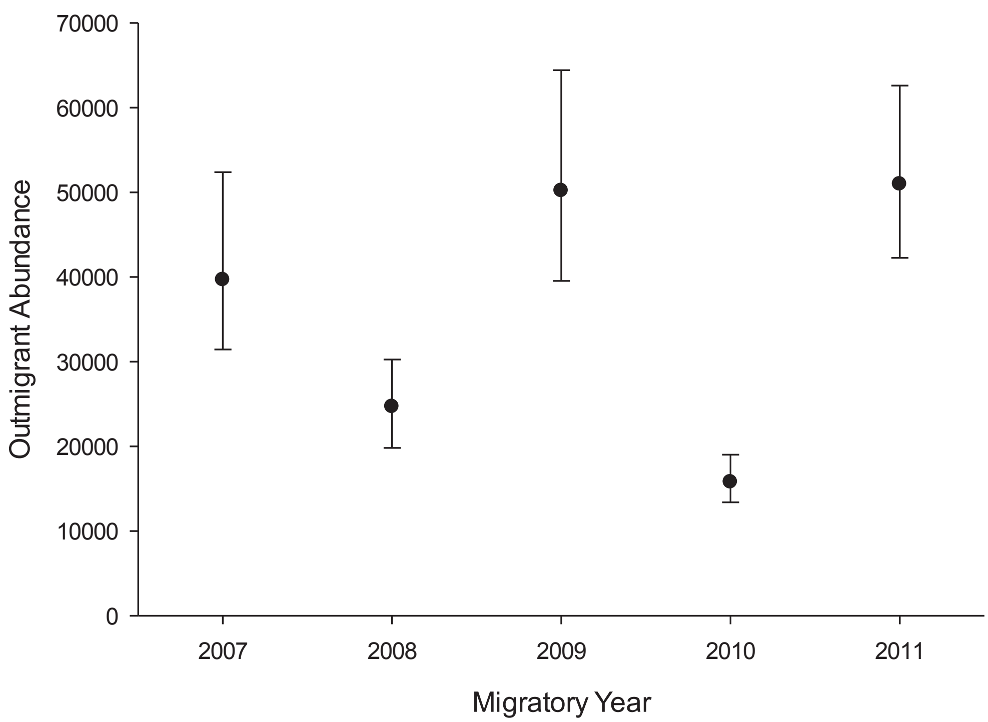


Figure 49. Mainstem trap summer steelhead abundance estimates by migratory year. Error bars are 95% confidence intervals. Reported in DeHart et al. (2012).

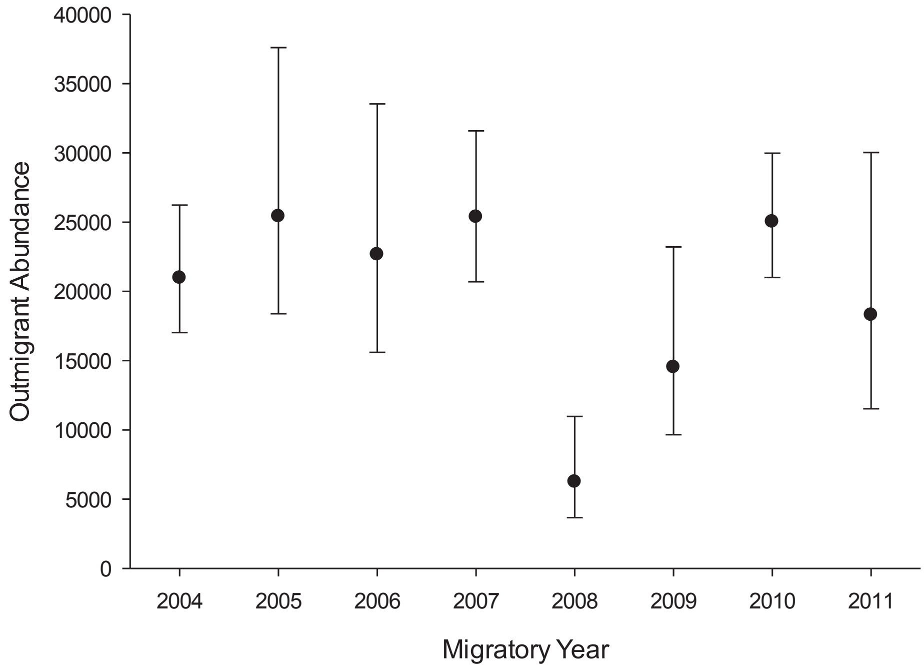


Figure 50. Middle Fork trap summer steelhead abundance estimates by migratory year. Error bars are 95% confidence intervals. Reported in DeHart et al. (2012).

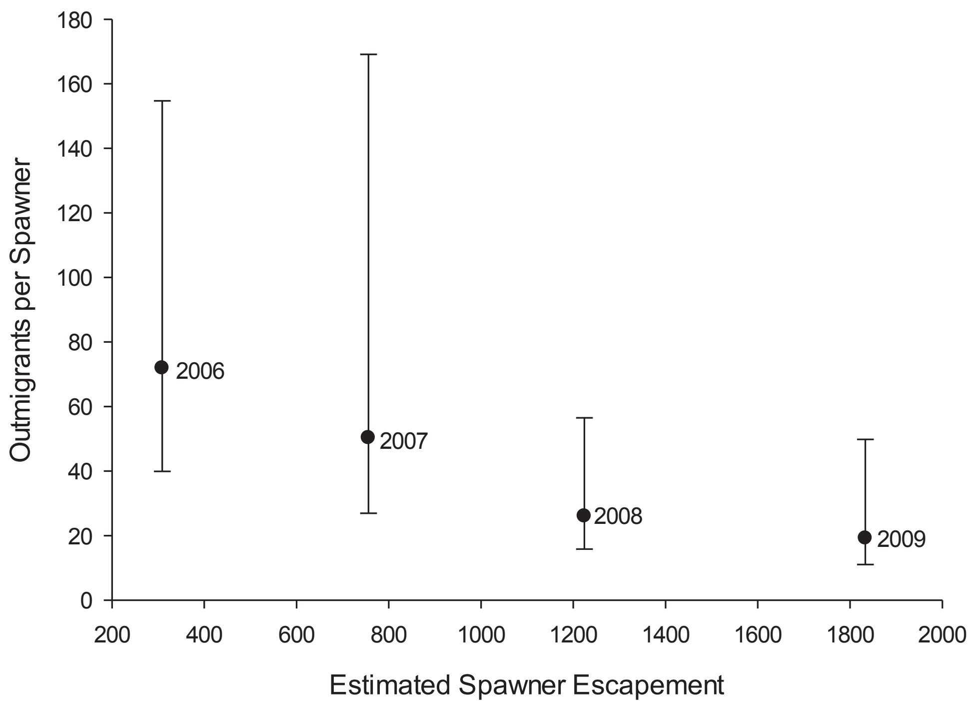


Figure 51. Estimated emigrants per spawner production from the South Fork John Day River steelhead population for the 2006 through 2009 brood years. The 2009 brood year is incomplete, but currently includes the majority of anticipated smolts. Error bars are 95% Confidence Intervals. Reported in DeHart et al. (2012).

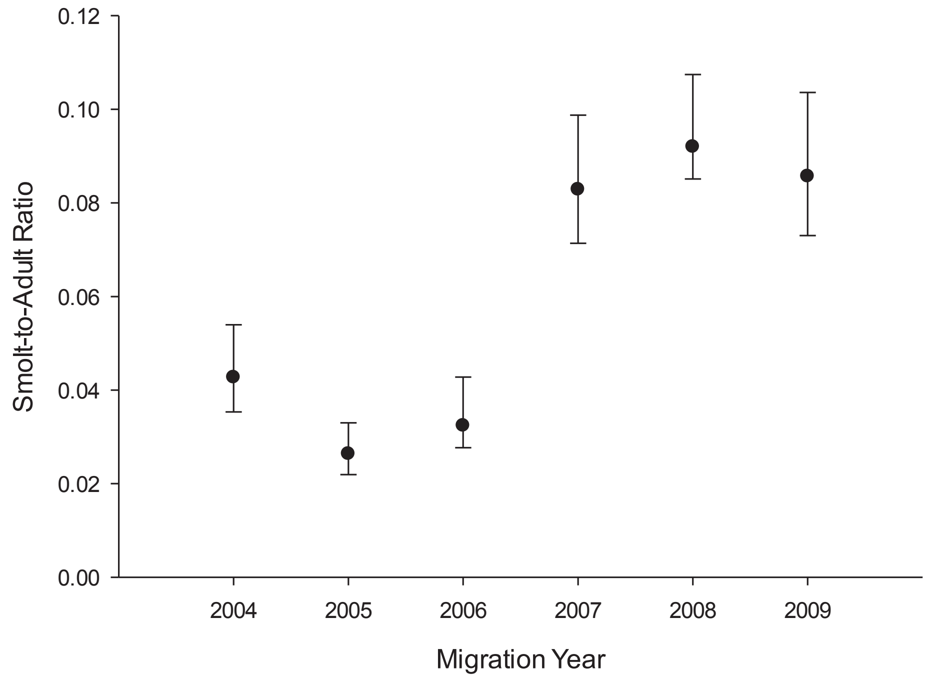


Figure 52. Trends in smolt-to-adult ratio (SAR) of juvenile summer steelhead tagged with Passive Integrated Transponder tags in the John Day River basin during migration years 2004 through 2009. SAR is estimated from smolt migration past John Day Dam to adult detection at Bonneville Dam. Error bars are 95% Confidence Intervals. Reported in DeHart et al. (2012).

IV. EFFECTIVENESS MONITORING OF STREAM RESTORATION

Improving spawning and rearing habitat for listed salmon and steelhead through stream restoration is a key mitigation strategy of the BiOp. Over a billion dollars are spent annually in the U.S. 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 resulted in changes in habitat and or increased salmon and steelhead freshwater production. Despite the large expenditure on stream restoration, there is almost universal agreement for the need to better understand the linkages between restoration and the population response, which requires detailed implementation and effectiveness monitoring (Bernhardt et al. 2007, Katz et al. 2007).

As discussed in the introduction, evaluation of restoration effectiveness relies on a robust experimental design and pre-project monitoring implementation. Without a strict study design, the actions may not affect a large enough area to elicit a population change, or may not address the limiting factor, and thus result in low power to detect differences. ISEMP has implemented IMWs to conduct restoration effectiveness monitoring in an experimental framework to demonstrate the utility of such designs, and because they are the mostly likely way we will be able to observe population-level benefits.

IMWs are designed using principles of ecosystem-scale experiments, arguably the most direct method available for detecting a population or environmental response to management actions (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). 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). Generally these experiments are implemented in the form of a large perturbation to restore habitat after anthropogenic practices with large negative impacts (e.g., logging). IMWs, however, use restoration as the experimental manipulation, thereby potentially providing benefits to salmonid populations while maximizing our ability to learn from these recovery efforts.

While IMWs have several strengths, they do have limitations in the number of locations they can realistically be implemented, can take a several years to decades to demonstrate benefits of restoration, and do not leverage projects that already been implemented. ISEMP has thus also conducted project-scale effectiveness monitoring to assess benefits of past designs and activities. Below we provide a brief description of both of these approaches within ISEMP.

Intensively Monitored Watersheds: Large-Scale Restoration Experiments

In order to detect a signal due to a restoration action, distinct contrasts in both time and 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 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 the treatment effects are, the more likely noise can be separated from the true treatment effect.

Another approach is to replicate treatments 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. IMWs as a means for testing stream restoration must incorporate both time and space contrasts and create large treatment effects to overcome the lack of replication. The proper context of the current and historical conditions and a proper identification of the limiting factors within the study watershed are also necessary.

ISEMP is using the IMW approach in three watersheds to implement restoration in an experimental framework to test the effectiveness of the restoration at improving fish habitat and increasing productivity of salmon and steelhead. Below we provide a brief description of each ISEMP IMW, and examples of approaches used to provide context, synthesis of the complexities of ecosystem responses, and the identification of mechanisms by which fish and habitat respond to restoration in order to improve our ability apply successful restoration elsewhere.

Lemhi IMW

ISEMP is conducting a Lemhi IMW in the Salmon River Basin with the primary goal of testing the effectiveness of reconnecting numerous small tributaries to the mainstem Lemhi River. While tributary reconnections are the major restoration focus, the Lemhi IMW also evaluates additional habitat actions including channel modifications, riparian fencing, diversion removals and screening, and side-channel development. Generally, freshwater productivity in the Lemhi River watershed is thought to be limited by the availability of high quality juvenile rearing habitat. Although spawning habitat is not currently believed to limit production, the spatial distribution of spawning, particularly for stream-type Chinook salmon, is limited to a relatively small section of the upper mainstem Lemhi and Hayden Creek. Thus, successful tributary reconnection may increase the geographic distribution of spawning thereby decreasing the risk of brood year failure that could accompany a catastrophic event in the primary spawning habitat. Additionally, the lack of access to tributary habitat limits access to potential thermal refugia that could improve survival and condition of migrating adults and rearing juveniles. The physical benefits of this action include: enabling access to historically available spawning and rearing habitat; decreased mainstem Lemhi River water temperatures owing to greater cool water tributary influence; and greater access to thermal refugia for juveniles and adults.

The Lemhi IMW is being implemented in a staircase design, where connection of high priority watersheds will (actually have) occur first, then depending on results of the first treatment, medium priority, and so on (see Chapter III Figures 24 and 25). The monitoring program is designed to assess status and trends as well as effectiveness of the tributary reconnections (for more detail see Standing Crop/Salmon Basin and Figure 24 and 25, Chapter III). The CHaMP protocol and bathymetric LiDAR are used to describe habitat at survey sites and throughout the Lemhi drainage, respectively. Mark-recapture of juvenile steelhead and Chinook also occur at each site. Rotary screw trap and PIT tag antennas are used to enumerate outmigrating smolts and adults respectively (see Chapter III).

In order to provide a landscape and life-cycle context, and synthesize how restoration is expected to result in tributary and/or reach scale alterations and changes in Chinook and steelhead vital rates (survival/productivity, abundance, and condition), ISEMP is employing the Watershed Production Model (see Salmonid Production in a Life-Cycle Context, Chapter V and for more details see Chapter 3 of the Appendices). The model will be used in adaptive management framework to explicitly state hypothesis, expectations, and triggers to determine stage two implementation strategies.

Bridge Creek IMW

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 leads to reduced base flows, increased summer stream temperatures, and a loss of spawning and rearing habitat. This is the situation with instream and floodplain habitat within Bridge Creek in the John Day Basin.

Beavers in Bridge Creek build dams that aggrade the stream channel (deposition of sediments behind beaver dams that raise the stream bed), but the lack of large wood results in unstable dams with a short lifespan and the loss of stored sediments. Restoration is aimed at causing 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 53) designed to assist beaver in the construction of stable longer lasting dams (see Appendix-Chapter 7).

Bridge Creek is an ISEMP IMW where stream restoration is implemented in a hierarchical-staircase design to create contrasts in time and space at multiple scales to detect a population level change in steelhead growth, survival, abundance, and production (Figure 54- See Chapter 6 and 7 in the Appendices for more detail). The first step of the staircase design was implemented in 2009

Figure 53. 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.

where 84 structures were installed in four treatment reaches, leaving six reaches that will act as controls until they are treated in 2013. The CHaMP protocol, aerial photos and aerial- and ground-based LiDAR are used to detect changes in habitat and riparian vegetation. Mark-recapture of juvenile steelhead also occur at each site in spring, fall, and winter. PIT tag antennas are used to enumerate out-migrating smolts and aide in the estimate of juvenile and adult survival. Adults are enumerated using a two-way weir and redd surveys (see Chapter III).

Here we provide an example of how we document precise geomorphic changes to help understand the mechanism by which fish habitat and floodplain connectivity results from the restoration strategy. This change detection approach has also been adopted by CHaMP to document trends in fish habitat. The primary change detection metric to describe aggradation is the DEM of difference, or the difference of digital 3D maps of the channel constructed before and after implementation of restoration. The DEM of difference is the change in stream bed elevation within the stream channel (Figure 55). 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 zero difference suggests no change (represented as white). This is done for every point to create a surface, and a distribution of the actual changes in elevation. The distribution of changes can be summed to describe a net degradation, aggradation, or no change to the reach.

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.

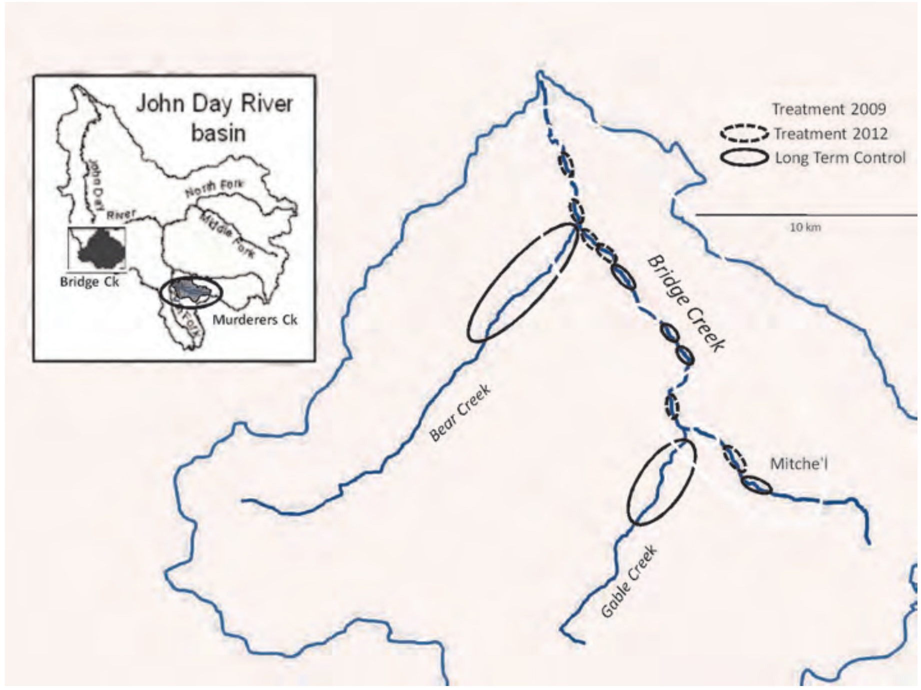


Figure 54. The Bridge Creek IMW experimental and monitoring design. White, black-dashed, black-solid oval represent restoration units, subwatersheds, watersheds that will be treated in 2009, 2012, or act as long term controls, respectively. Four habitat survey sites are monitored within restoration units (inset box) in a rotating panel design (inset table). All four sites make up one fish survey site. Aerial LiDAR and photos cover entire experimental watershed, and ground-based LiDAR, RTK GPS, and total station surveys are conducted across the entire restoration unit.

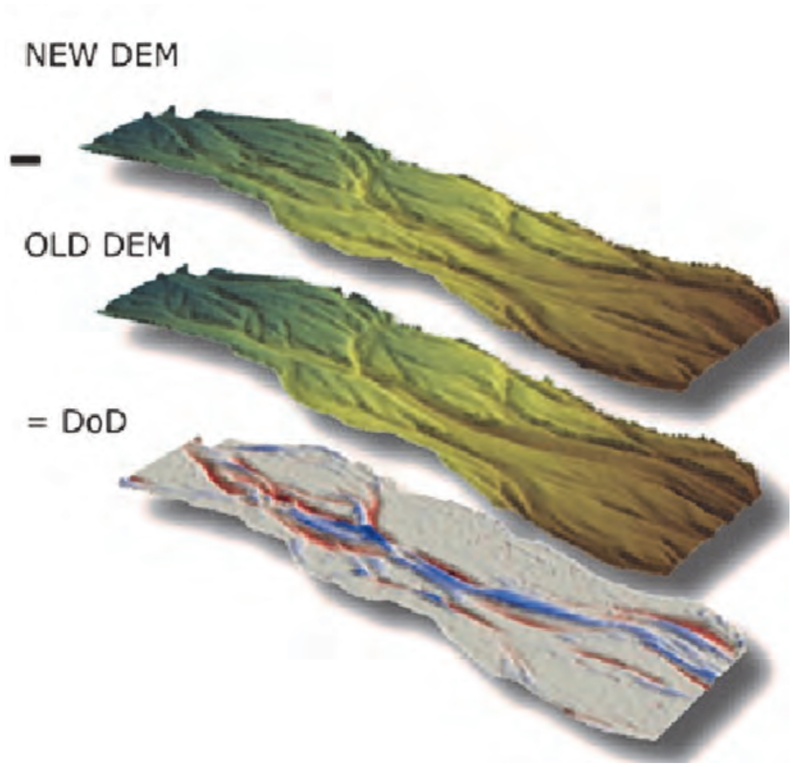


Figure 55. Concept of DEM differencing.

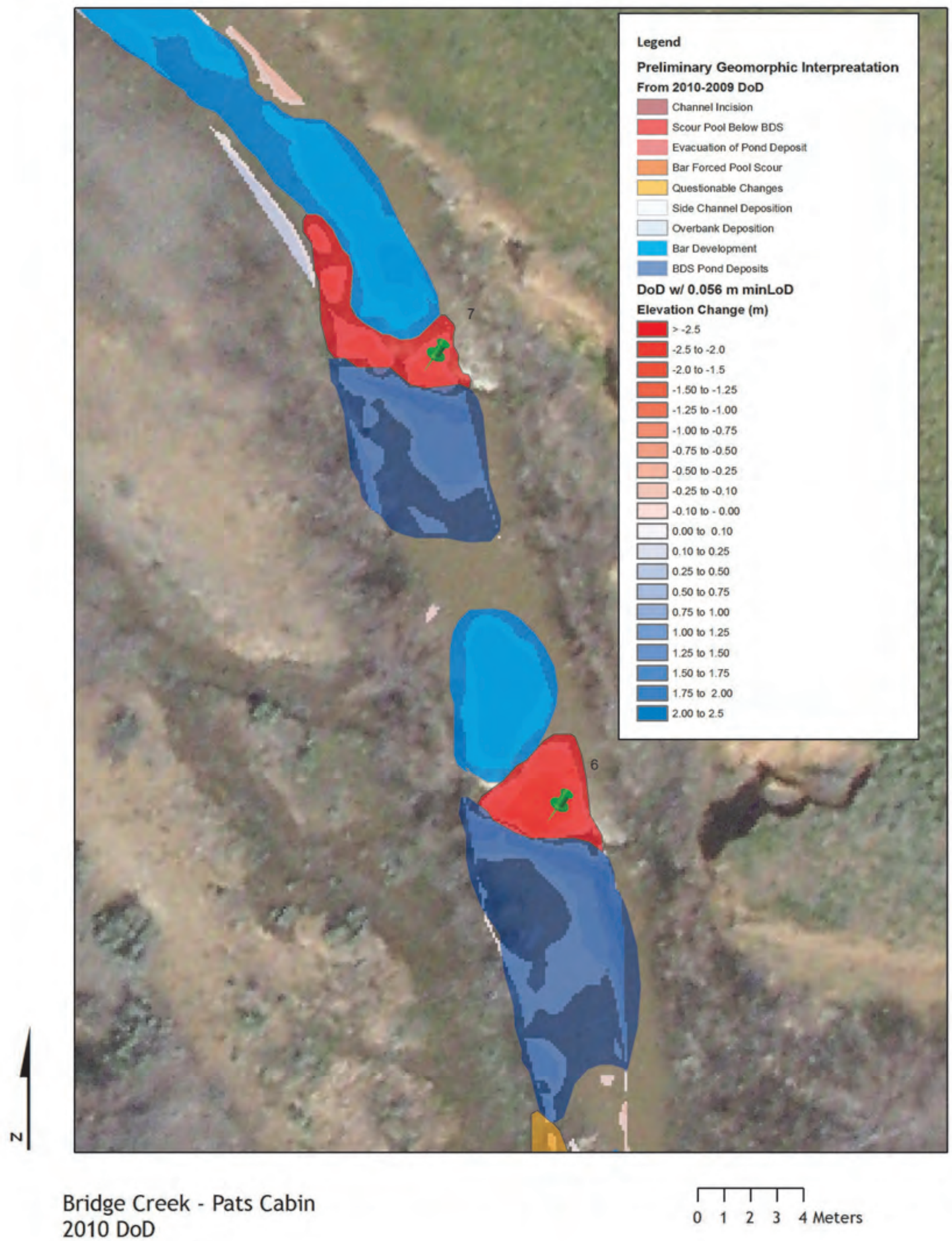


Figure 56. DEM of difference (post-restoration minus pre-restoration) from topographic surveys for a portion of treatment reach in Bridge Creek. Pushpins represent structure location. Blue color represents aggradation (deposition of sediments), and red represents erosion. General pattern was to have deposition behind structures, scour pool below structures, and deposition of the scour downstream from the pools.

DEMs of difference capture this general pattern clearly (Figure 56). 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 56).

Entiat IMW

In the Upper Columbia the Entiat River is an IMW to determine the effectiveness of restoration at improving Chinook and steelhead freshwater productivity. The primary restoration action to be tested is active instream modifications via engineered structures that increase habitat complexity and diversity by creating large pools and off-channel areas. ISEMP proposed that a hierarchical-staircase statistical design be implemented to compare treatment and control sections within the Entiat River. The hybrid hierarchical-staircase experimental design uses a 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 (Figure 57). Treatment and control sections will be represented in each geomorphic reach type, and each geomorphic reach will be implemented in staggered manner through time (for more detail see Appendix-Chapter 6). The status and trend monitoring program implemented in the Entiat will also address effectiveness monitoring (see Chapter III). Habitat and fish monitoring sites occur in each section and the CHaMP protocol will be used to evaluate changes in habitat. Mark-recapture of juvenile steelhead and Chinook will be used to capture fish metrics in each reach. A census of redds will be used throughout the study area and a screw trap will be used to enumerate adults and out-migrating smolts, respectively (see Chapter III).

To help understand why these habitat restoration actions have an effects on populations of Chinook salmon and steelhead, ISEMP has also been working with researchers from the Pacific Northwest Research Station USDA FS to estimate population size and individual growth and movement for Chinook and steelhead at the reach scale to complement the larger-scale effectiveness monitoring.

Snorkeling and a combination of snorkeling and seining were used to enumerate fish multiple times at treated (a series of four engineered log jams and five rock barbs have formed pools and therefore added microhabitat scale variation in rearing habitat within the treated reach) and untreated reaches to determine if 1) fish growth and movement would show density dependence and 2) density dependence would differ between the treated and control reaches.

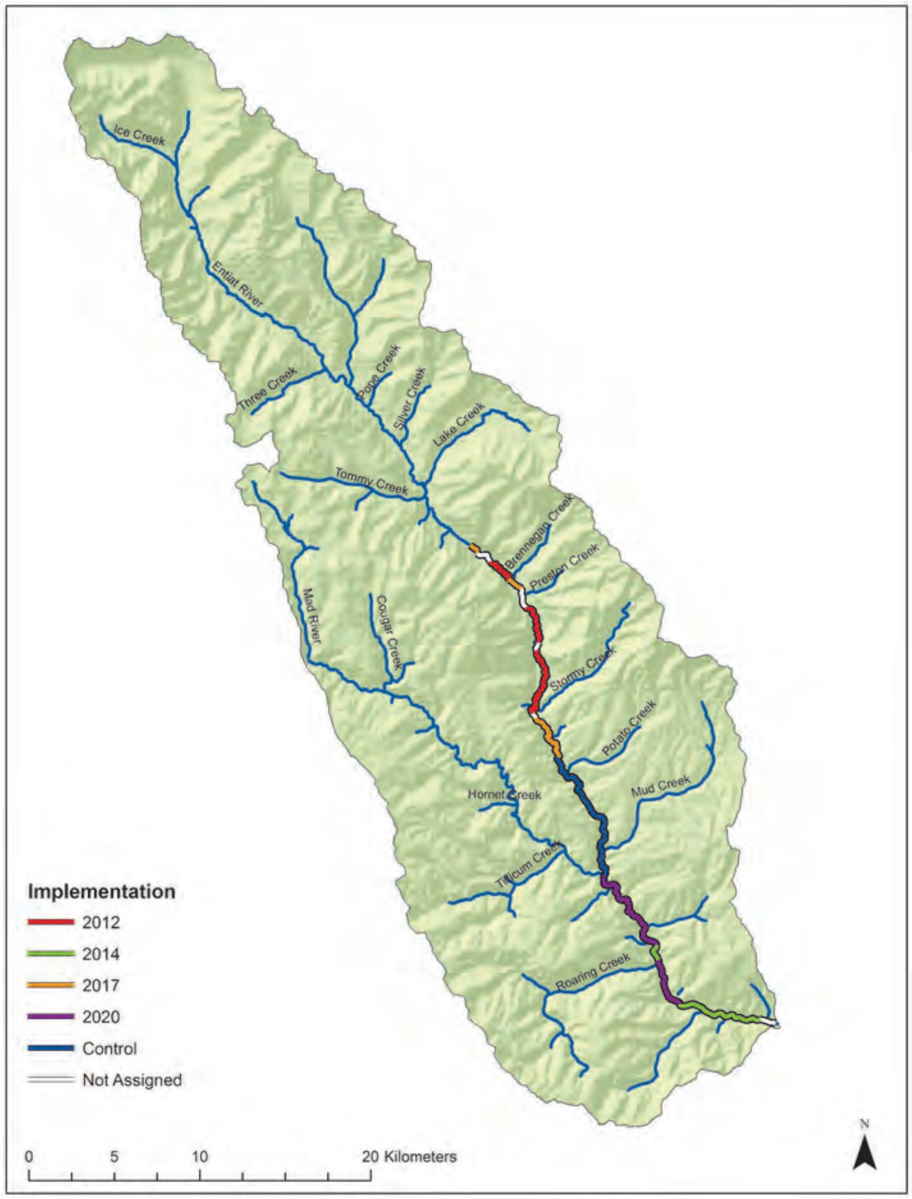


Figure 57. The Entiat River IMW experimental design. Treatments are stratified by valley types. Numbered letters represent reaches. Red reaches will be treated in 2012, green reaches in 2014, orange reach in 2017, and purple reach in 2020. The Mad River (large tributary coming in at the upstream section of 1F), will act as sub-watershed control and will not be treated.

A number observations regarding increased fish density are evident from three seasons of monitoring in the lower Entiat River. Figure 58 shows the results of fish counts within a treated reach either at the structures specifically (blue boxes) or at randomly selected microhabitats within the same reach (red). Both Chinook and steelhead had higher median density at structure pools during the first week of sampling (mid-August 2011). However, later in the summer fish density is not strongly associated with structure pools (Figure 59). This likely reflects the sub-yearling Chinook parr migration toward over-wintering habitat downstream and overall highly variable habitat selection patterns by steelhead.

The elevated density of juvenile Chinook in treated microhabitats appears to be associated with a strong response to the increased water depth created by structures. To determine whether the observation of higher density at microhabitats with structures was truly a benefit to fish or whether this was an artifact of fish movement, we examined behavior and growth in pools with or without treatments.

In a short term (24 hr) mark-recapture study, both Chinook and steelhead exhibited habitat affinity in that they tended to be recaptured in the same pools, where they were marked, more frequently when those pools were treated with structures compared with untreated microhabitats (Figure 60).

During 2010, we measured growth across the short season during which we were able to sample, mark and recapture both steelhead and Chinook. Due to the pattern explained above, in which Chinook density declines substantially during late August and early September, we obtained too few recaptures to identify any difference in growth among Chinook. However, steelhead, despite being at lower density at structures than at untreated sites, had higher growth rates at structures, suggesting that density might not be the only indicator of fish response to restoration.

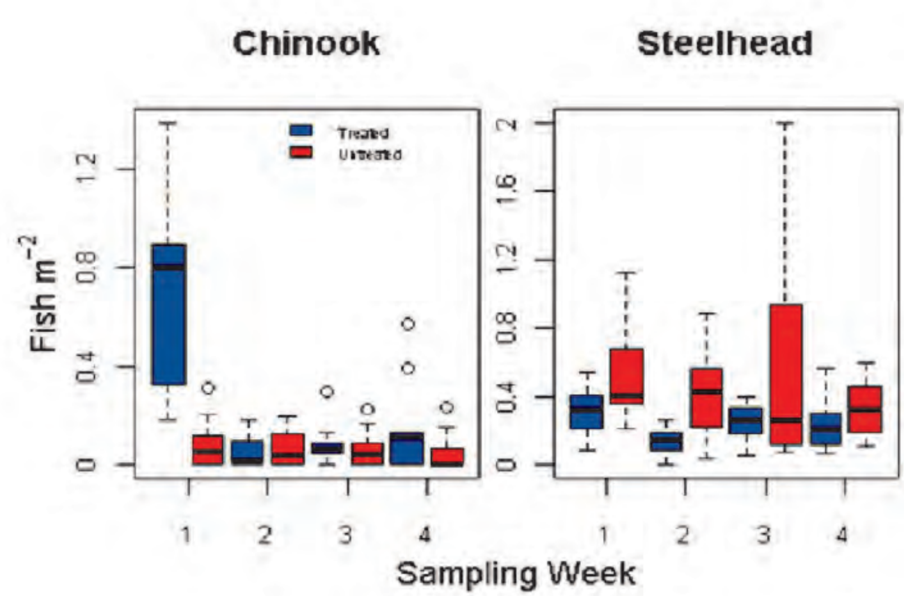


Figure 58. Density of juvenile Chinook and steelhead in treated and untreated microhabitats in the Entiat River, August—September 2010.

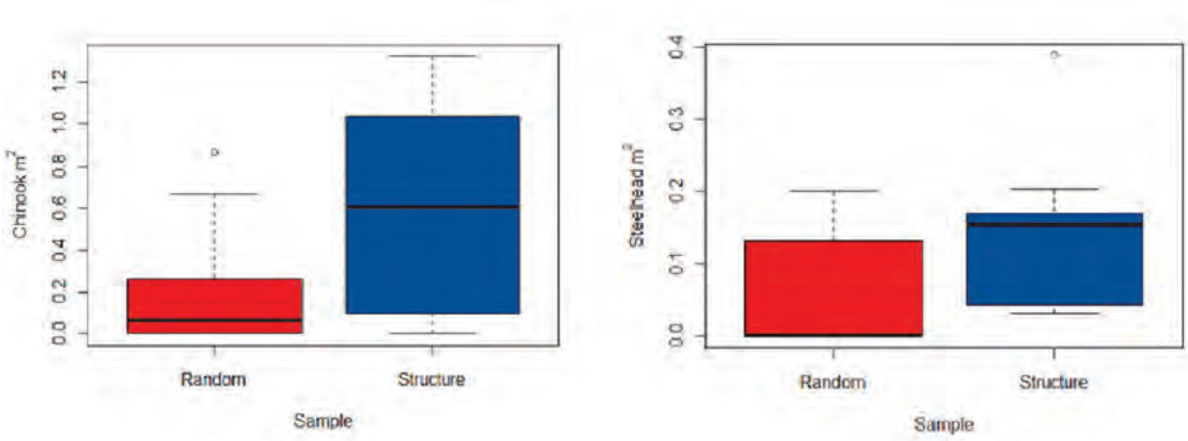


Figure 59. Early summer density of juveniles Chinook (left panel) and steelhead (right panel) within a treated reach, either at a structure (blue box) or in randomly selected habitats within the same reach (red box).

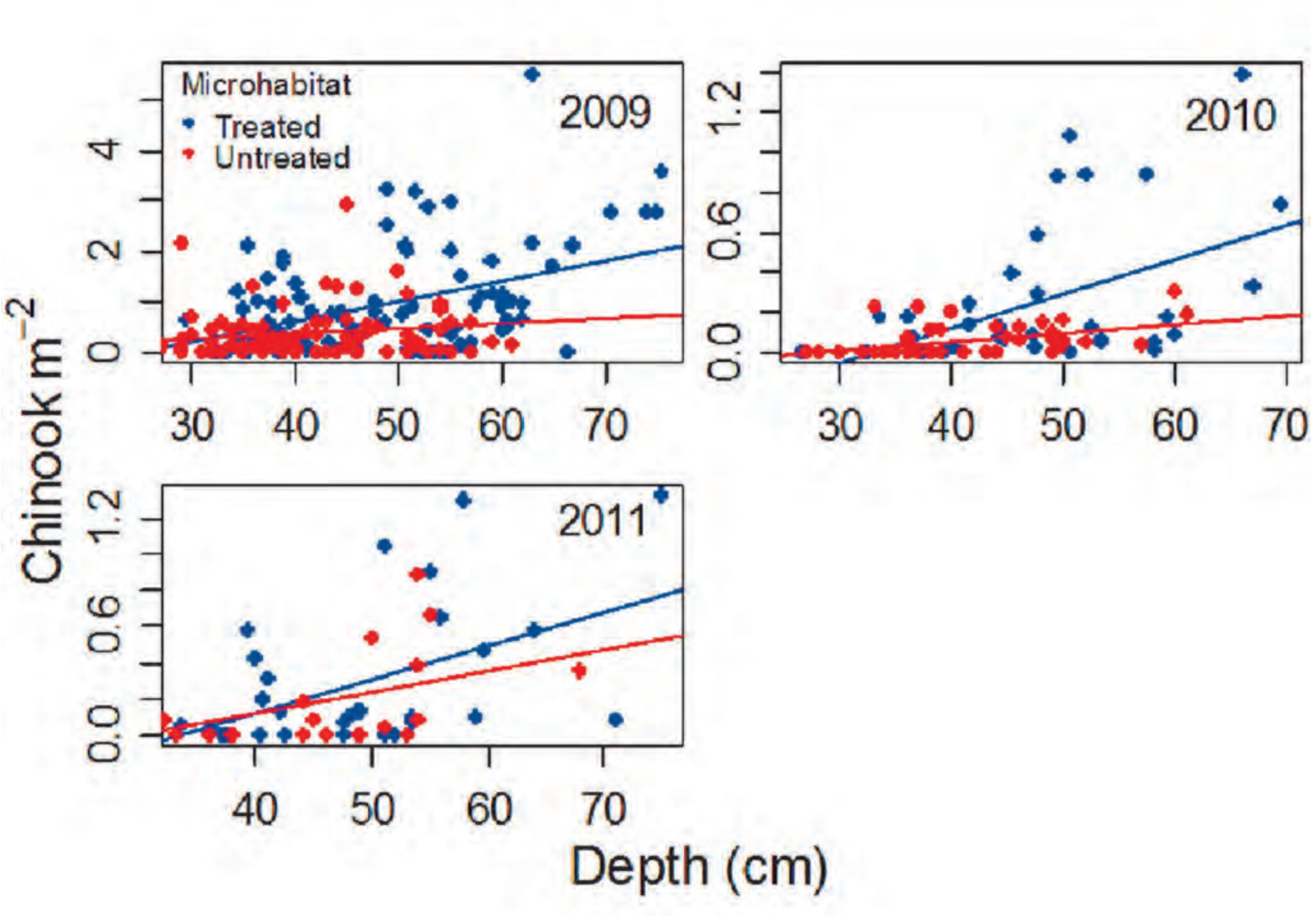


Figure 60. Density of juvenile Chinook in treated and untreated microhabitat and its correlation with water depth in the Entiat River.

The Importance of a Study Design Pre-Implementation

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). In fact, over the past two decades, funded by BPA, 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. Riparian exclosures are a very common passive restoration approach. However, changes to the riparian corridor and stream channel after exclosures are built can take decades 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. No pre-project monitoring was implemented so a post-hoc study design was necessitated. 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, assessing whether the grazing exclosures resulted in altered channel morphology and improved habitat conditions for a subset of streams in the John Day watershed of eastern Oregon.

In 2009-2010 ISEMP surveyed 14 locations throughout the John Day Basin consisting of one exclosed site (treatment) and one grazed site (control). To determine if exclosures resulted in changes, at each site, we conducted riparian, habitat and fish surveys (For more details see Chapter 5 in the Appendix).

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, but fish are interacting directly with stream morphology and indirectly with riparian vegetation. While we were able to detect changes to the riparian area due to exclosures (e.g., Figure 61), we were unable to detect associated response in steelhead performance (e.g., Figure 62). From these results, we cannot infer whether grazing exclosures have elicited channel recovery (for a 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. A post-hoc study design is not likely to be powerful enough to detect differences if they really do exist.

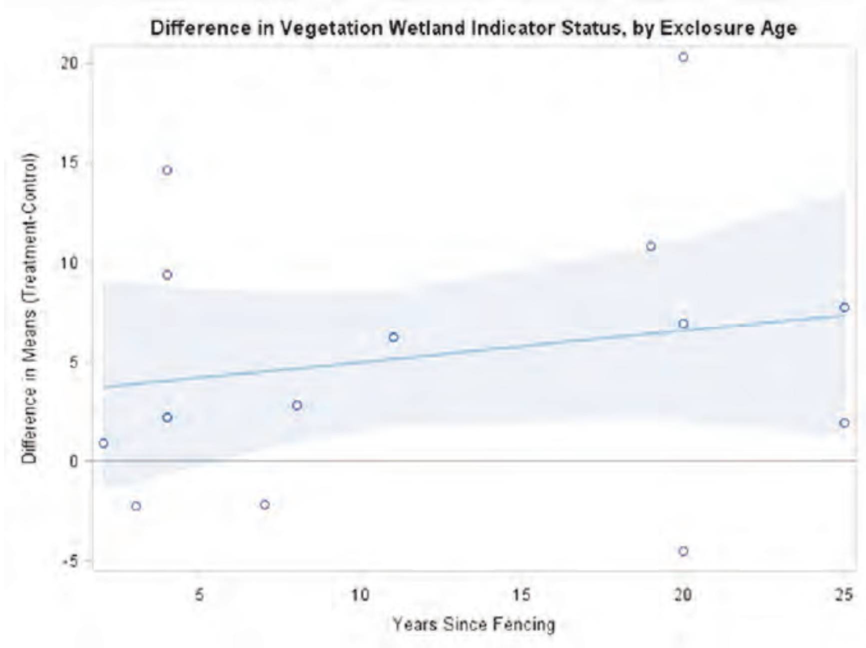


Figure 61. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, in wetland indicator values for the greenline plant communities. Statistically different values observed at exclosure sites > 6 years old.

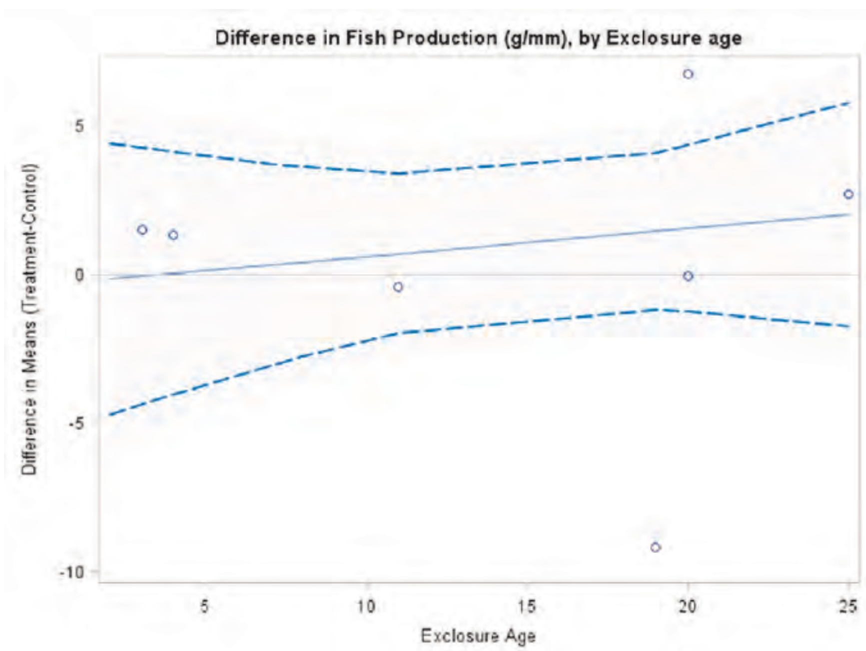


Figure 62. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs, and 80% CIs on the dashed lines), across different ages of exclosures, in fish production, excluding age 0 steelhead.

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V. ANALYTICAL FRAMEWORK

Determining Metrics Useful for Change Detection

The monitoring programs described previously provide data upon which to evaluate current fish population structure and habitat quality and availability, changes to these populations and their habitat, and whether actions undertaken are responsible for the changes. In addition, we have to acknowledge that our understanding of fish and their interaction with their environment is far from complete, and thus we need to discover relationships between fish and their habitat, and use this and the best available science to identify limiting factors. However, to turn raw data into interpretable results requires extensive summarization, analyses, and synthesis. Further, analyses are required to provide context to the data and results. In ISEMP, we are developing an analytical framework that:

- Provides descriptive and empirical relationships to help validate expectation and generate hypotheses;
- Uses mechanistic models that synthesizes multiple metrics into hypotheses of how the world works that if validated can allow to assess multiple scenarios; and
- Evaluates whether restoration experiments are providing their expected benefits.

Here we provide a few examples of this framework which are described in detail in the appendices. ISEMP is developing several additional methods to determine trends. As habitat actions are employed, the ability to detect a change if one occurs, and tie it to the specific action is extremely important.

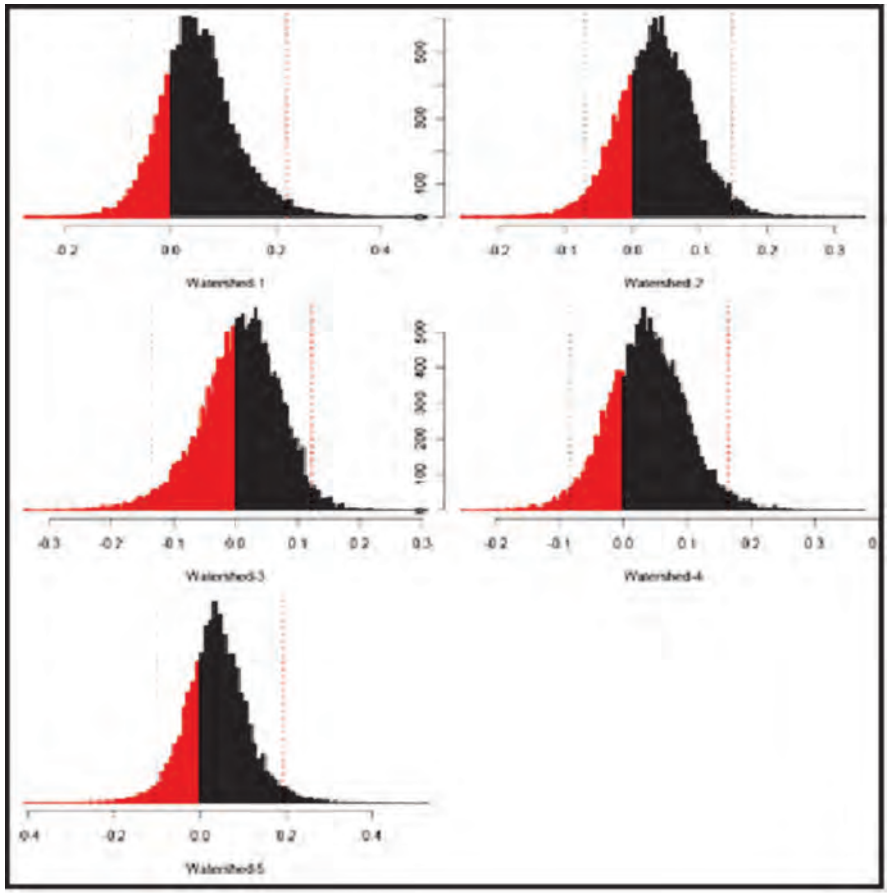


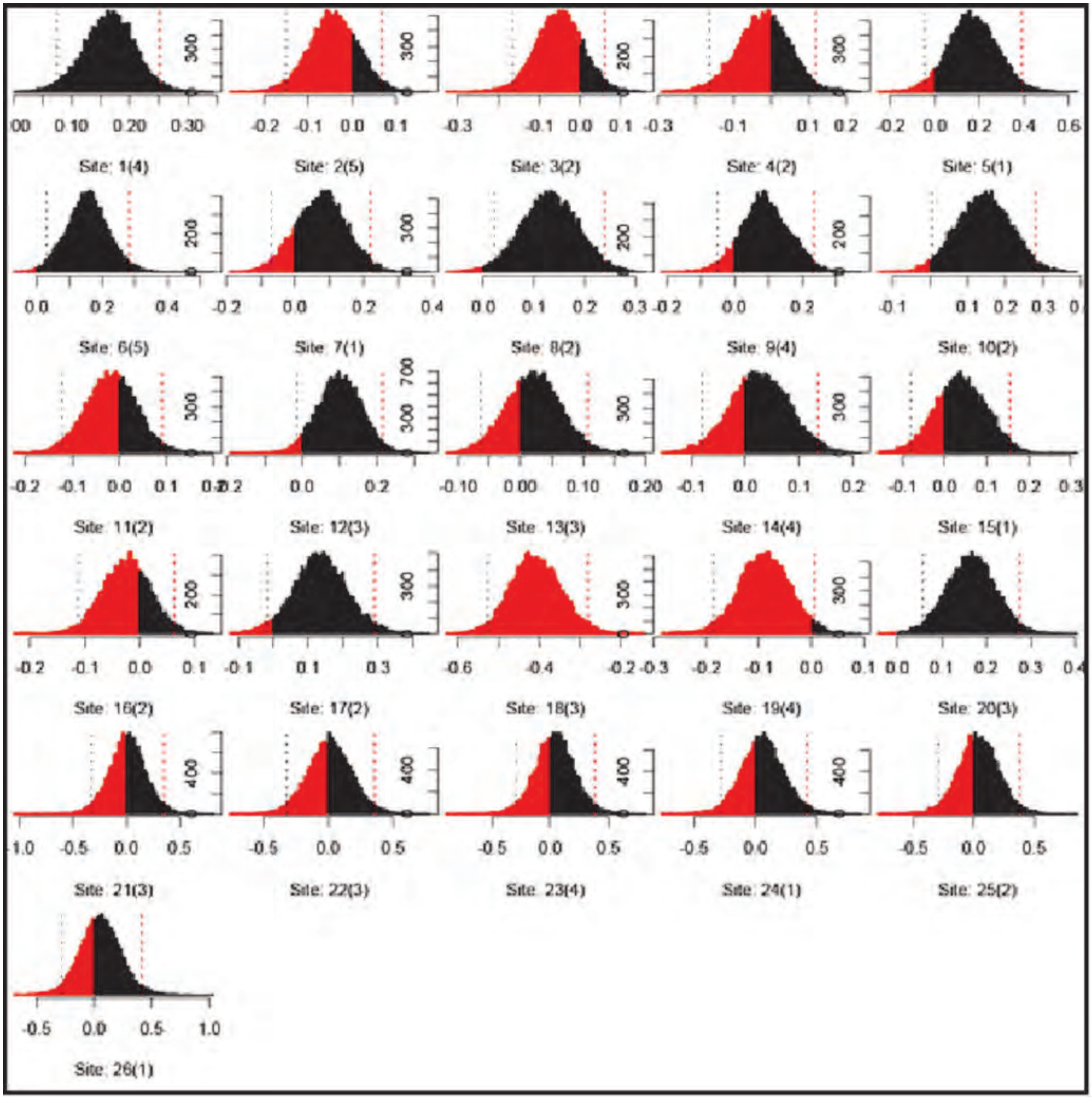
Figure 63. Trend in bankfull depth in 5 subwatersheds and monitoring reaches (panel this page) and at individual monitoring sites (panel facing page) in the Wenatchee River subbasin over the period 2004 - 2009. Color coding reveals the probability that a negative (red) or positive (black) trend is detectable. Those sites with nearly all black or red indicate a high probability of either a positive or negative trend, respectively.

Bayesian Hierarchical Model

As discussed before, one objective of ISEMP is to evaluate which metrics and indicators are useful in determining trends in stream habitat. Not all metrics are suitable for detecting change over time and space for a number of reasons. For example, a metric may not be useful to detect change across the domain of interest because of high sampling error due to variability in crew sampling because the metric is based on qualitative measures (e.g., bankfull height) rather than quantitative measures.

ISEMP employed a Bayesian hierarchical model using posterior distributions of regression parameters to look for spatial and temporal patterns in the metric data. This model is a powerful tool for exploring data and allows for a graphical inspection of the data in an intuitive manner.

The hierarchical aspect of the model enables a elucidation of patterns at



different scales, such as the watershed and subbasin scales, while the Bayesian approach allows us to reveal the shape of the distribution o f the parameters.

An example is shown in Figure 63 for trends in bankfull depth in five subwatersheds and at individual monitoring sites in the Wenatchee River subbasin. From a visual inspection it appears that the bankfull depth metric has a high probability of detecting a trend, either positive (black), negative (red) or no trend detected (equal black and red shading, such as in Site 22(3) and Site 22 (4)).

Describing Habitat-Juvenile Salmonid Abundance Relationships using Wenatchee ISEMP Data

We have described how ISEMP is using a Bayesian hierarchical model approach to identify those metrics or indicators that best detect changes in stream habitat within tributaries and across river basins. However, these metrics must able to predict how changes in habitat directly effect fish populations. 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.

Figure 64 shows the relative importance of 18 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 broader scale environmental conditions not included among the predictor variables) is about twice as important for predicting juvenile Chinook density as gradient or the number of pools.

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 because of brood year strength and migra-

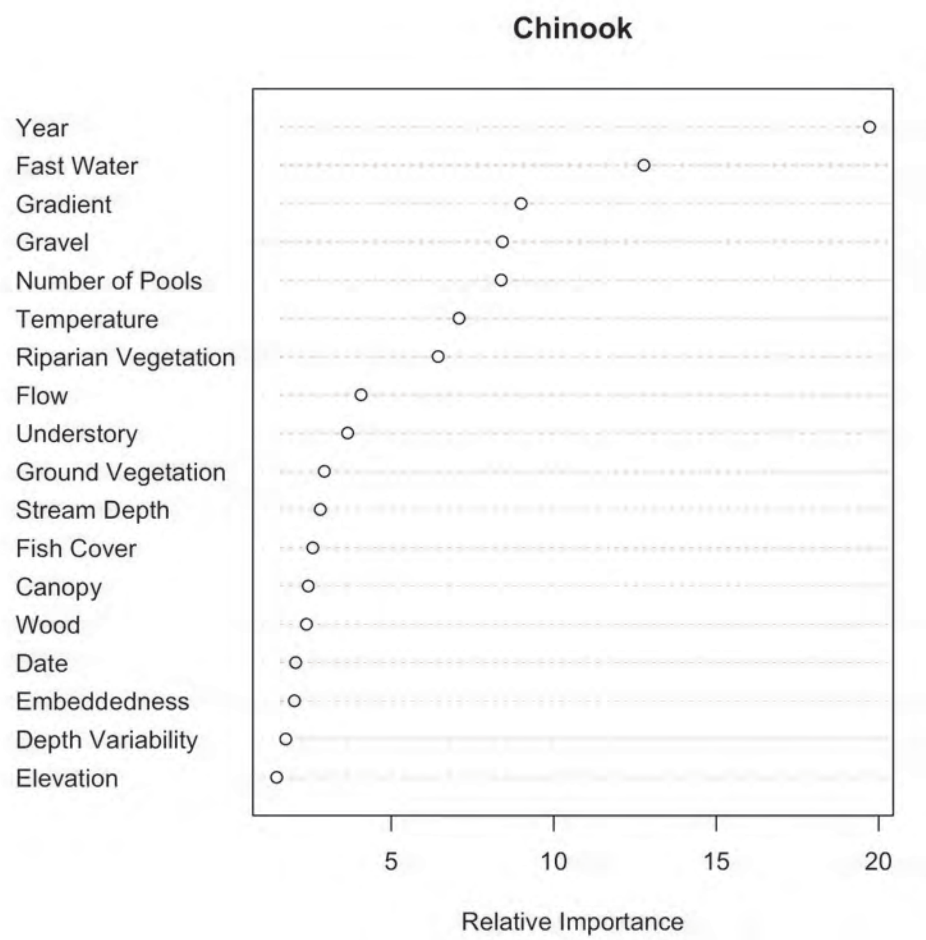


Figure 64. The relative importance of various habitat metrics in predicting the density of juvenile Chinook using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010 analyzed using a boosted regression tree approach.

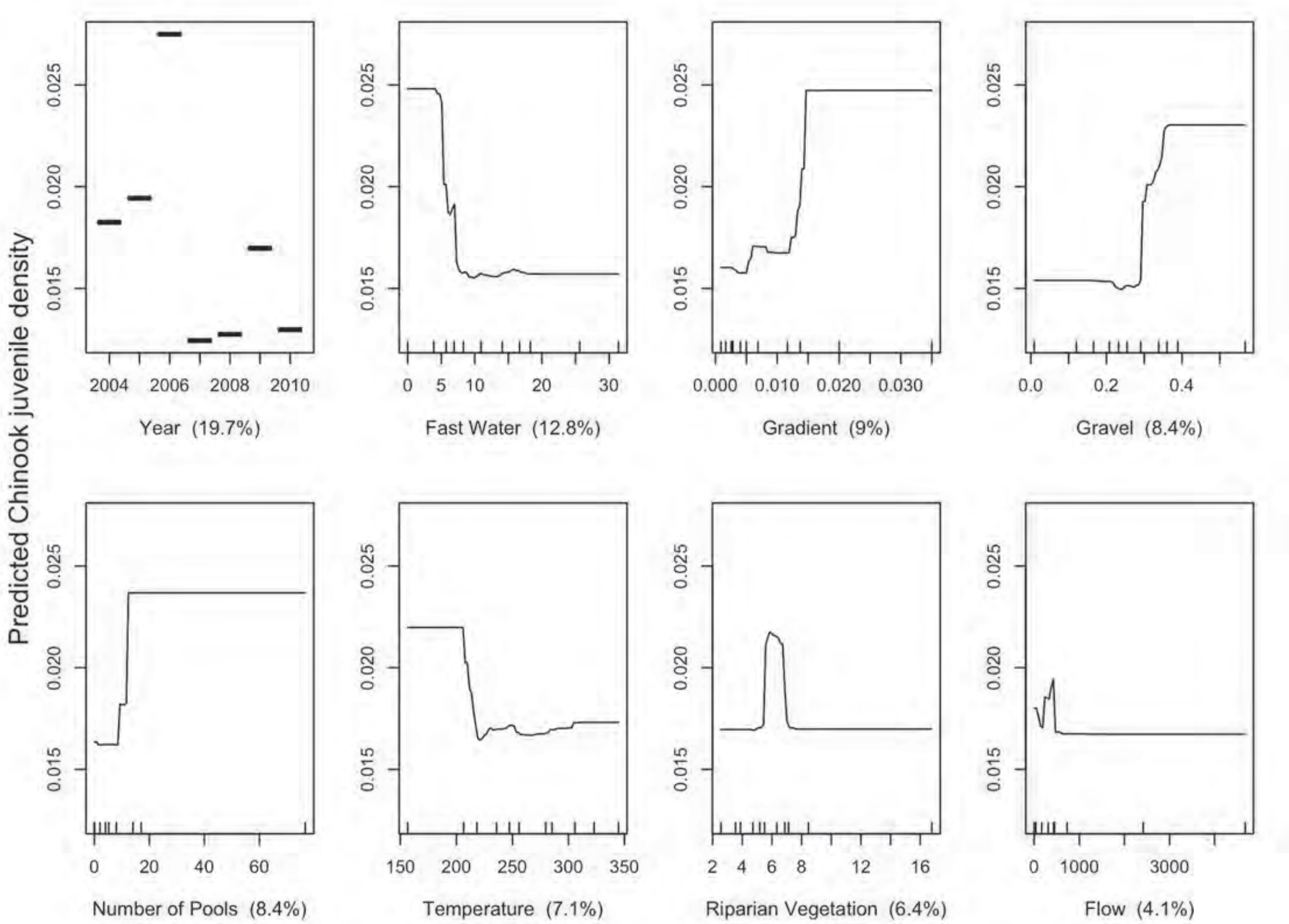


Figure 65. Partial dependence plots showing the marginal effect of the eight most important habitat metrics identified from a BRT on juvenile Chinook densities using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010. The y-axis is the predicted value of juvenile density. Along the bottom 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.

tory characteristics of juvenile salmonids.

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 65) several thresholds become apparent that can be used to suggest limiting factors and 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 predicted Chinook density (Figure 65). Chinook density jumps from a low to a high value once the percent coarse gravel crosses a threshold near 30%. This demonstrates how the amount of gravel at a site could be a factor limiting the density of juvenile Chinook.

Steelhead have fewer strong correlations with this suite of habitat metrics, making it more difficult to predict their densities from a single habitat measure, as seen in Figure 66.

This type of analysis is correlative in nature, providing guidance as to what types of habitat metrics are best at predicting fish densities, and suggesting what the shape of the fish – habitat relationship might be. These results can be used to generate more specific hypotheses about what habitat characteristics are

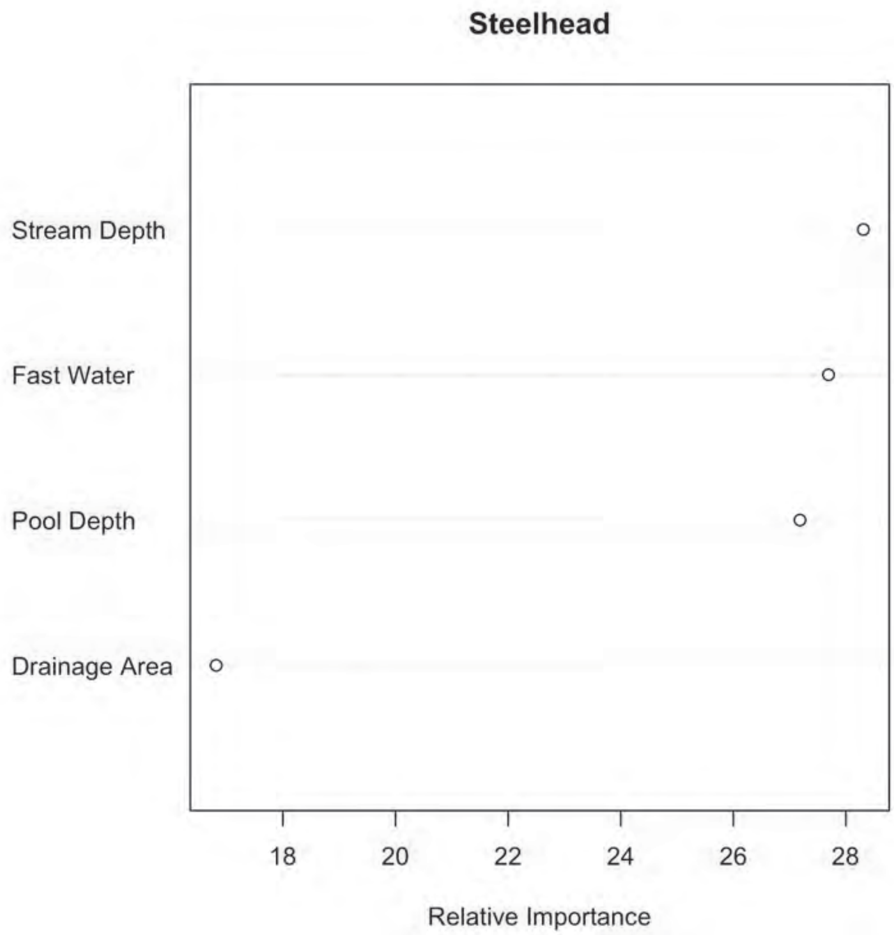


Figure 66. The relative importance of the four most important habitat metrics in predicting the density of juvenile steelhead using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010.

limiting factors for various salmonid populations.

Figures 65 and 67 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 "What habitat actions are most effective?"

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 66 and 67 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. These four habitat metrics explain 82% of the variance in steelhead density at different sites. Steelhead are generally found in higher densities in shallower streams with more slow water and deep pools. Some of these metrics also impact Chinook, such as the percent of fast water, but the relationship between the habitat metric and fish density is different for each species.

Figures 68 and 69 show the predicted densities of juvenile Chinook based on the amount of fast water and the percentage of coarse gravel respectively, and Figure 70 shows the predicted densities of steelhead based on average stream depth. When compared with the observed densities, the predictions based on a single habitat metric match the observed data fairly well. The predictions

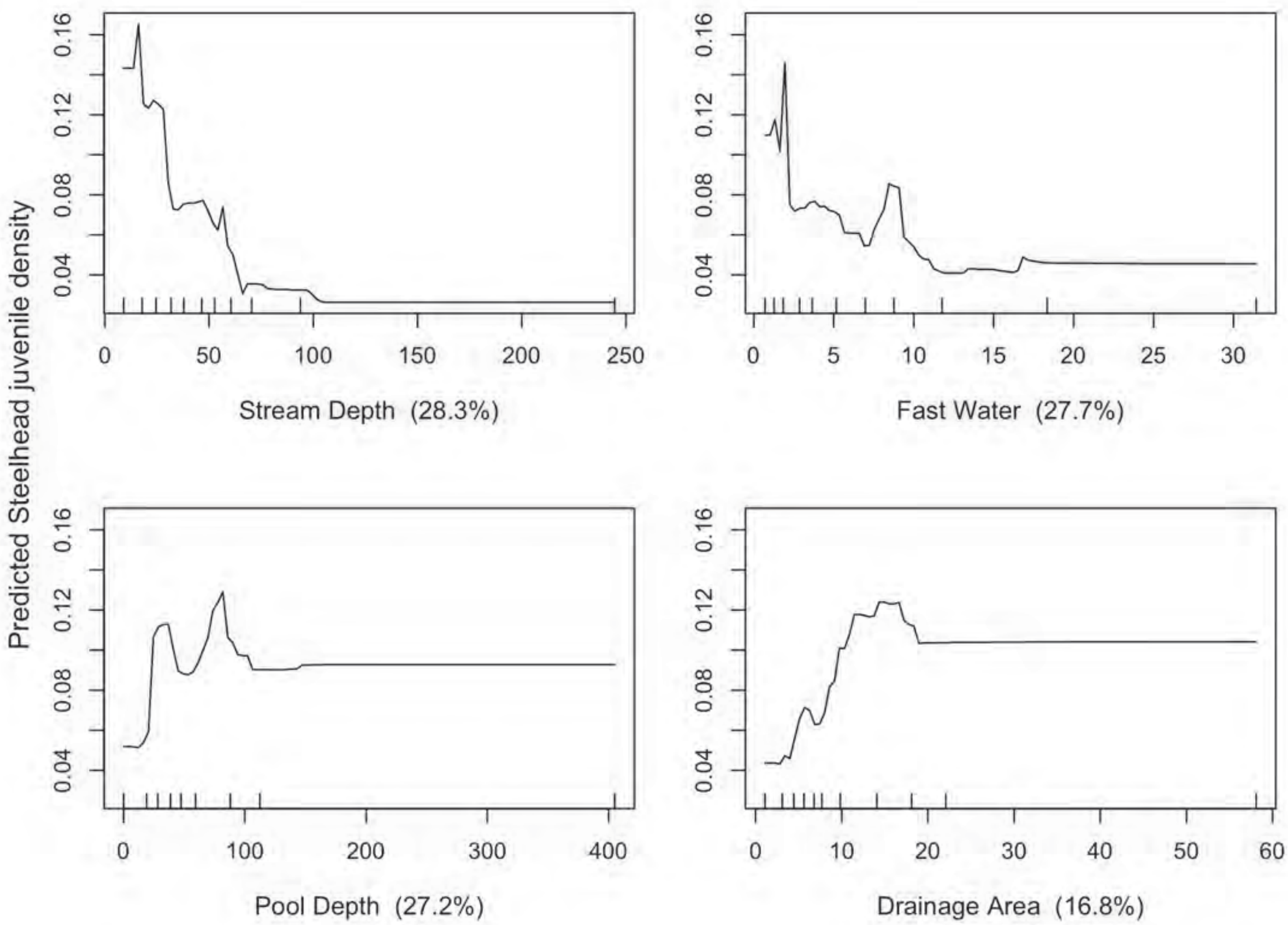


Figure 67. Partial dependence plots showing the marginal effect of the four most important habitat metrics identified from a BRT on juvenile steelhead densities using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010. The y-axis is the predicted value of juvenile density. Along the bottom of each plot, the tick marks show the deciles of the data for that habitat metric.

based on the entire suite of habitat metrics match even more closely.

A similar analysis has been undertaken across multiple basins using habitat data gathered from CHaMP. This work in the Wenatchee has highlighted the need to account for year effects, and although at this time there is only one year of CHaMP data, the preliminary results demonstrate this method can be applied to data from multiple populations.

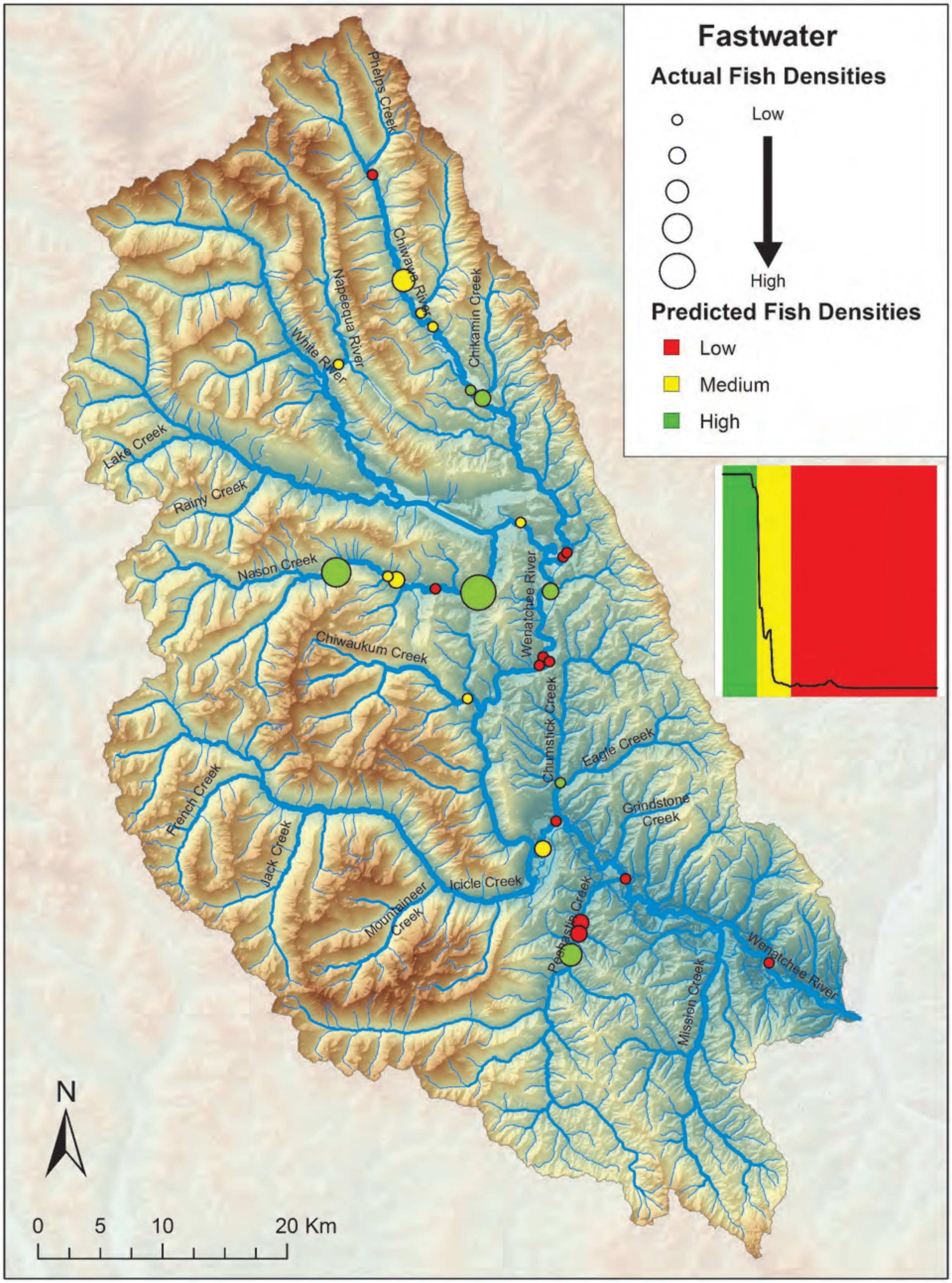


Figure 68. Observed juvenile Chinook densities, averaged across years, and the predicted densities, based on the amount of fast water at a site. The inset plot shows the relationship between fast water (x-axis) and predicted fish density (y-axis). Less fast water predicts high fish densities (green), more fast water predicts low fish densities (red).

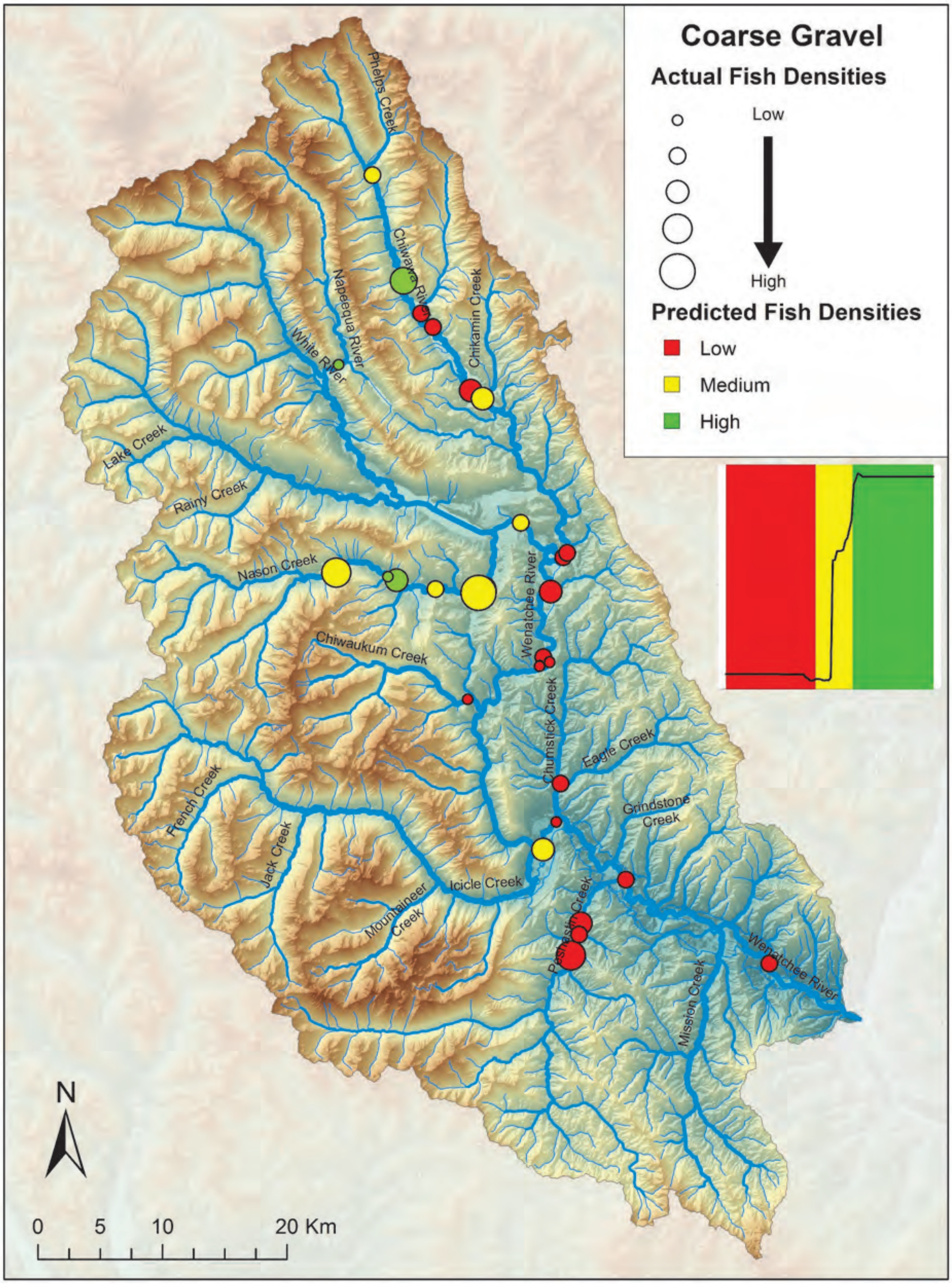


Figure 69. Observed juvenile Chinook densities, averaged across years, and the predicted densities, based on the percentage of coarse gravel at a site. Larger circles correspond to higher observed densities. The color corresponds to the predicted density. The small inset plot shows the relationship between 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).

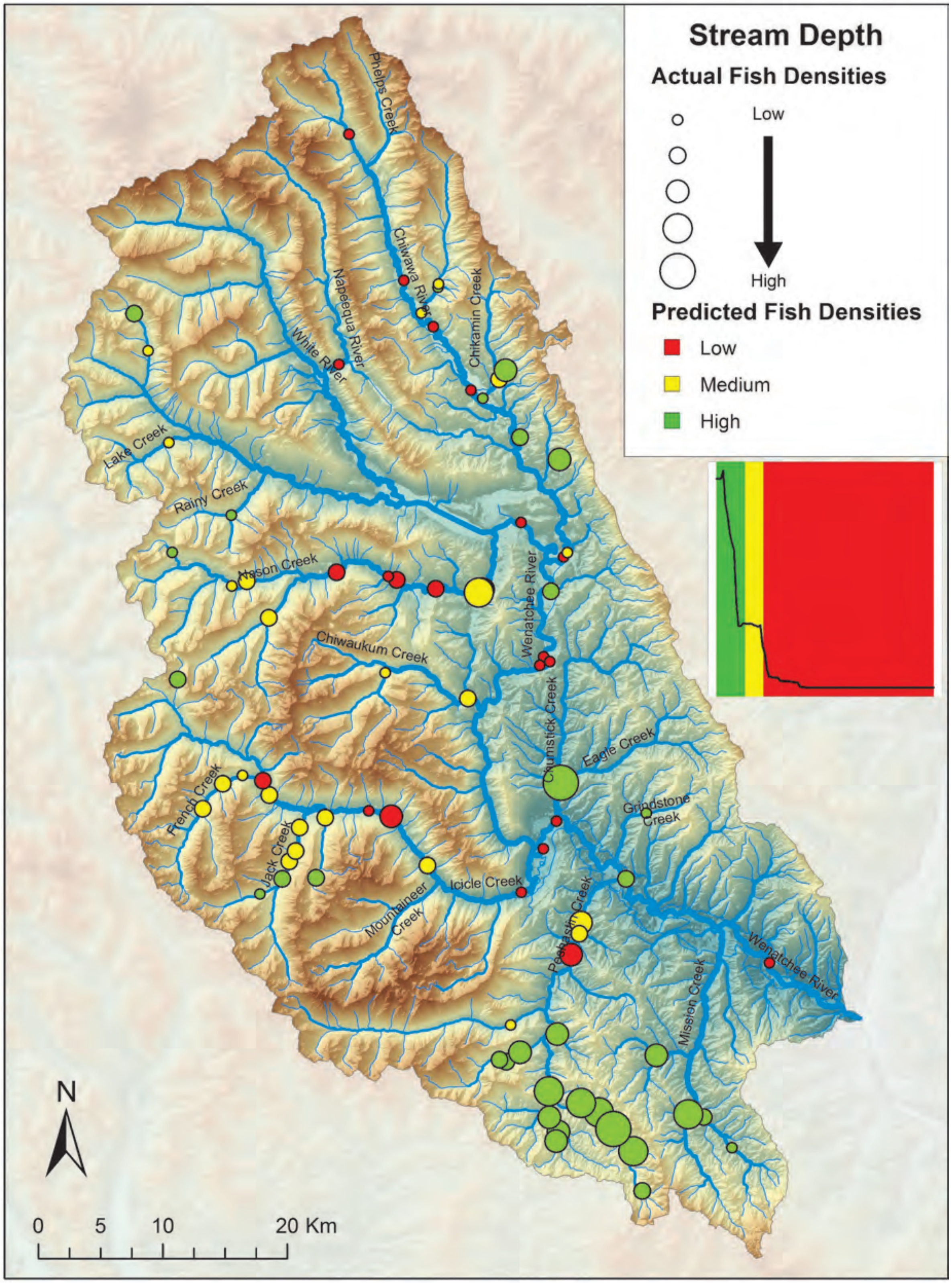


Figure 70. Observed juvenile steelhead densities, averaged across years, and the predicted densities, based on the average stream depth at a site. Larger circles correspond to higher observed densities. The color corresponds to the predicted density. The small inset plot shows the relationship between 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).

Classifying 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. 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 (Figure 71). There is a clear gradient in habitat condition as one progresses from sites classified as best toward those sites classified as poor.

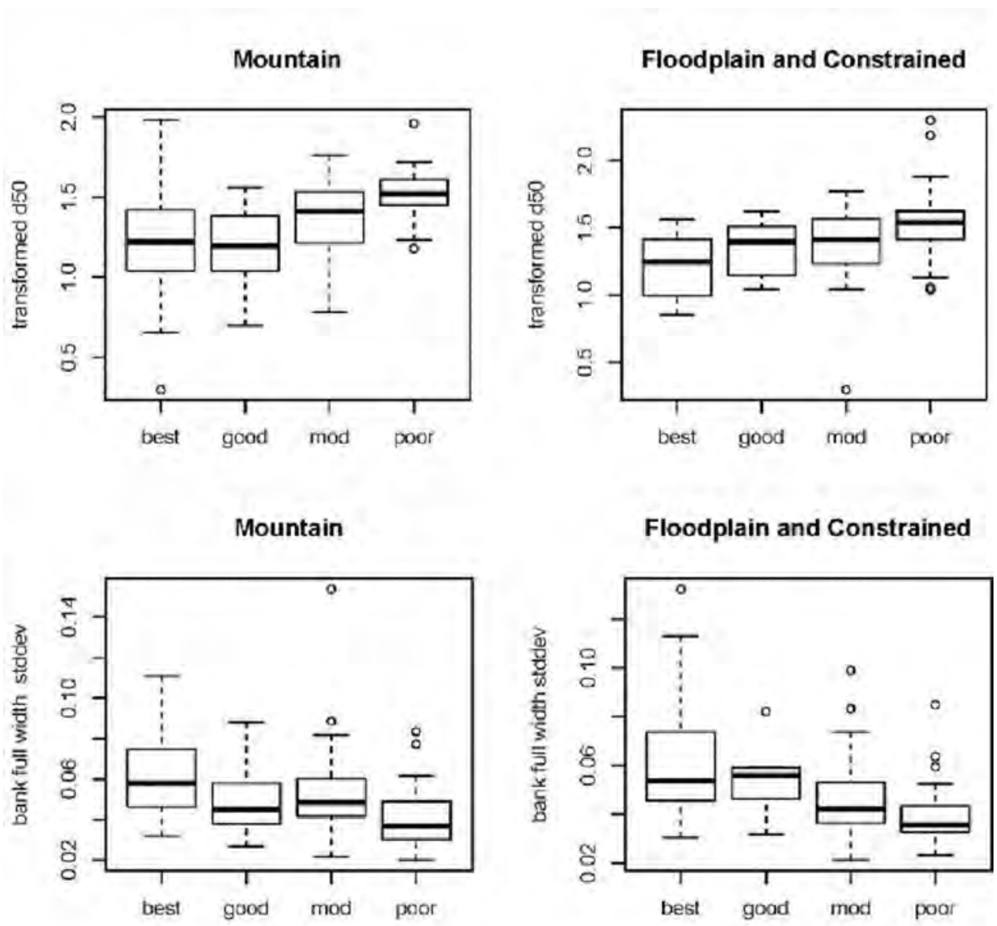


Figure 71. The relationship between two measures of habitat condition (d50, a measure of fine sediment, and standard deviation of bankfull width) and disturbance gradient for two classes of streams across four disturbance classes.

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 72). 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.

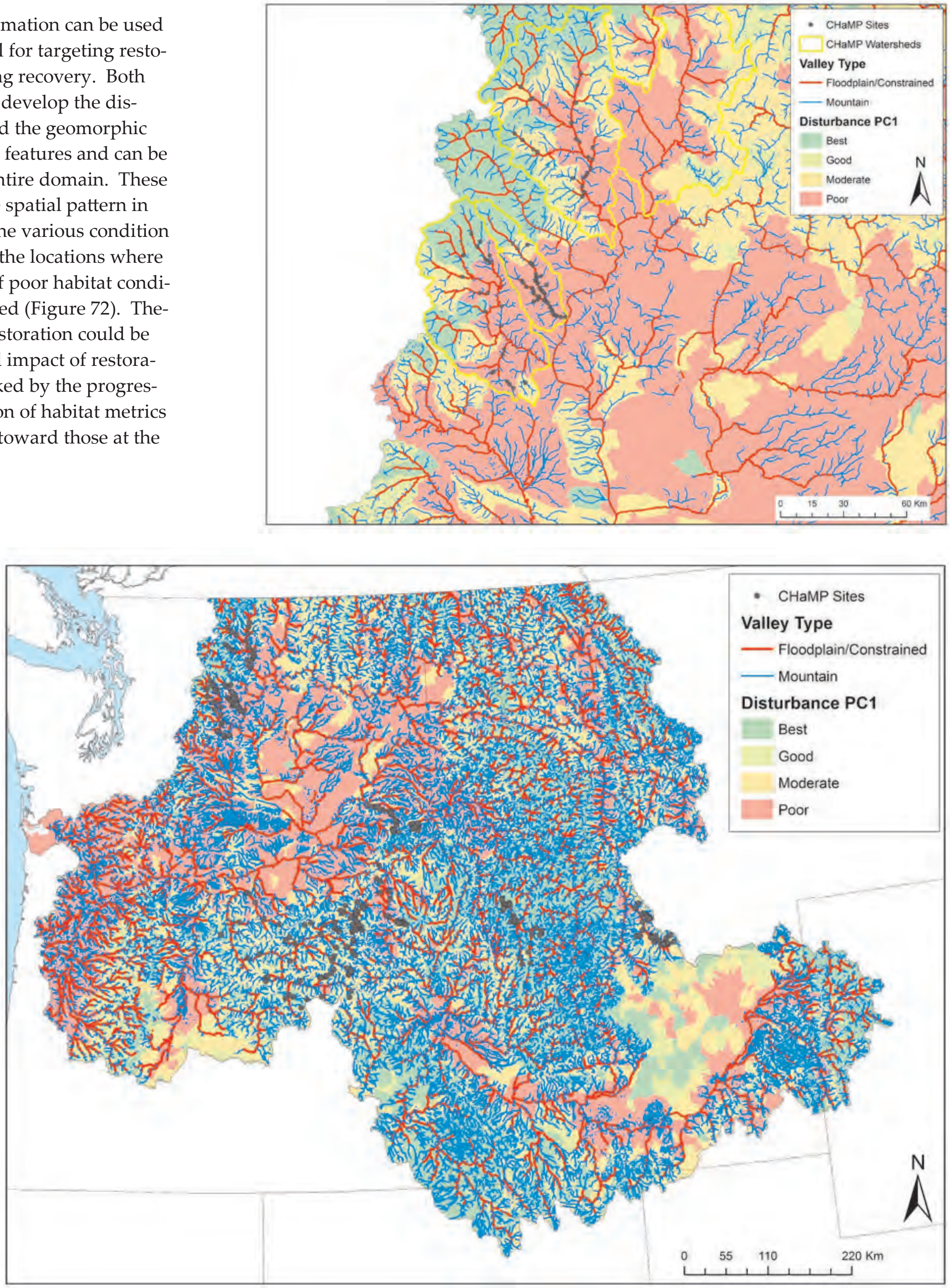


Figure 72. Maps illustrating where the probability of finding poor habitat condition is likely to be high and therefore where habitat restoration might be concentrated. The stream network classified into two geomorphic groups: Mountain and Floodplain/Constrained) because patterns of disturbance and recovery goals could differ. The lower panel is a closer look at the Upper Columbia portion of the upper panel.

Evaluating Temperature Impairment and Intrinsic Potential

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 73). 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 74). 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.

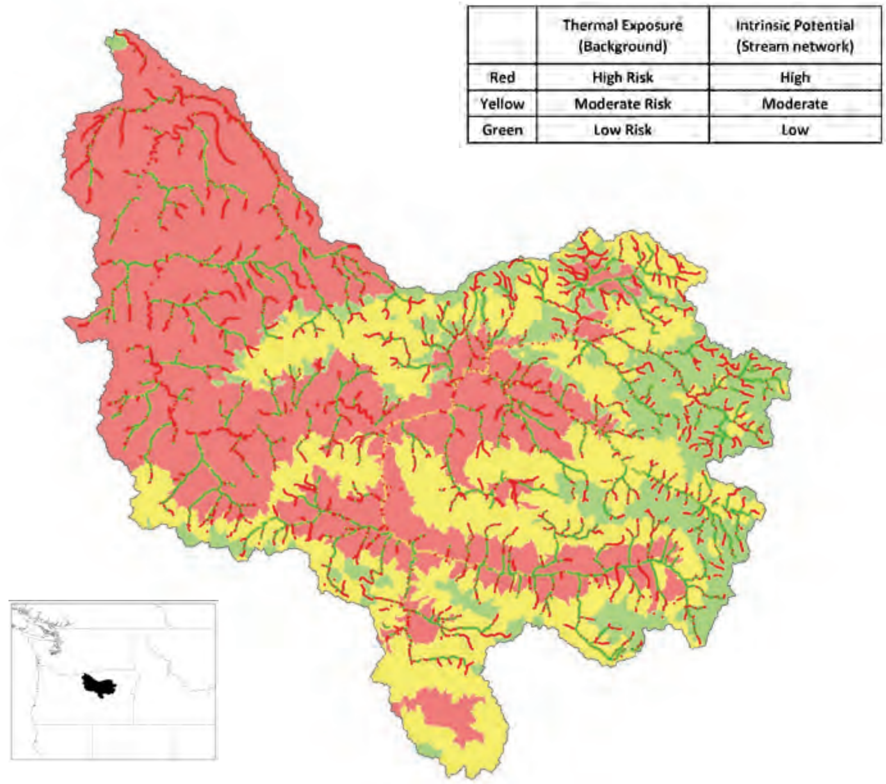


Figure 73. John Day River basin summer thermal impairment risk (background color) and Intrinsic Potential rating (stream color).

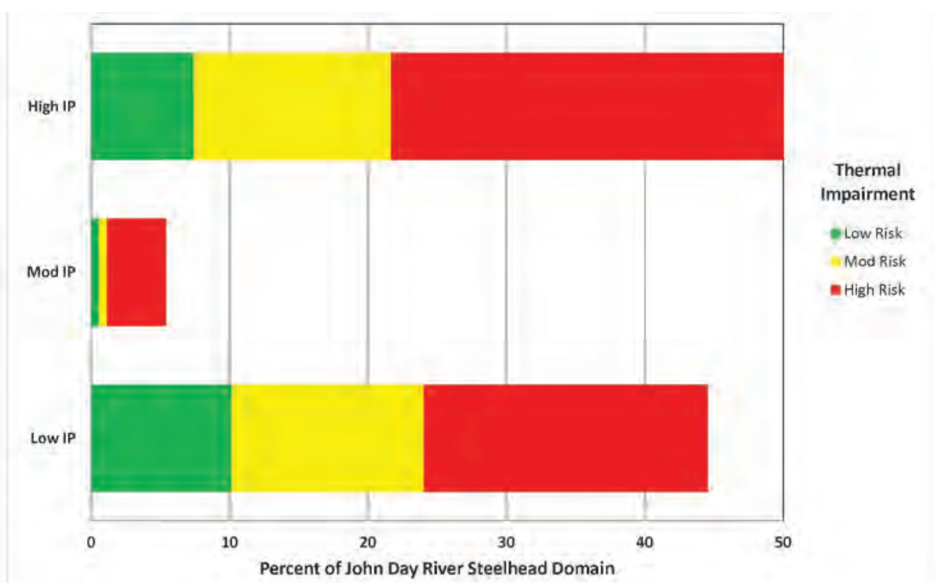


Figure 74. Relative proportion of the John Day River basin steelhead domain of Low, Moderate and High Intrinsic Potential (IP) falling in Low/Moderate/High summer thermal impairment conditions.

Salmonid Production in a Life-Cycle Context

The Comprehensive Analysis of the Federal Columbia River Power System and Mainstem Effects of Upper Snake and Other Tributary Actions (2007; Attachment C-1-31, page 854) notes an "absence" of life cycle models to estimate habitat quality and freshwater survival benefits for anadromous salmonids. The authors noted that life-cycle models were precluded from the document owing to a "*…lack of time, resources, and data to populate and run the models*." In the absence of such models, a simple logic path was developed wherein habitat restoration actions target habitat factors that directly limit "*…the freshwater survival or productivity of a population*." Notably, the authors list a number of assumptions that accompany this logic path:

- Limiting factors are known for each population.
- Habitat actions directly affect habitat variables that limit the population.
- Habitat variables can be combined to describe local habitat conditions.
- Local habitat conditions can be combined to describe overall habitat quality for the entire population.
- Changes in overall habitat quality are directly linked to changes in freshwater survival.

Utilizing these assumptions and the input of local biologists, estimates of expected survival improvements from the egg to smolt life history stage were generated for a number of Columbia River Basin anadromous populations.

Watershed Production Model

The watershed model described in this section is being developed in the Lemhi subbasin and is based on the premise that juvenile distribution, abundance, and survival are functions of habitat quantity and quality (see Appendix—Chapter 3). The model utilizes age and sex structured adult escapement (described in Section IV), spatially and temporally balanced age structured juvenile abundance and survival (described in Section IV), and spatially and temporally balanced habitat survey data (utilizing the CHaMP protocol). These data are combined in a life-stage specific Beverton-Holt production model where habitat quality is gauged by juvenile survival and distribution (i.e., we assume that locations supporting higher juvenile abundance and survival are indicative of good habitat). Since juvenile abundance and survival are age-structured, we can identify habitat features that are indicative of high quality habitat at each life stage. The underlying Beverton-Holt relationship in turn identifies which life-stages, and hence which habitat attributes, limit egg to smolt survival.

In its current form, the model is entirely based on empirical data; that is, functional relationships between habitat and survival are based purely on observations of fish distribution, abundance, and survival generated by data collection activities in the South Fork Salmon River (SFSR) and Lemhi River. Initially, this means that the model is most applicable to those locations. Once the model is fully populated in 2013, we intend to identify which fish and habitat relationships are "exportable" to other subbasins and identify minimum data requirements to effectively utilize the model as we recognize that the data collection efforts in the SFSR and Lemhi River are too costly for large-scale implementation across the Columbia Basin. Ultimately, developing empirically based relationships and identifying minimum data requirements will enable a more generalized version of the model to be cost-effectively deployed across the Columbia River Basin. Transferability of the model to other watersheds will be assessed by testing the sensitivity of model results to differing data types with a range of uncertainty utilizing information collected by ISEMP in the John Day and the Wenatchee and Entiat in the Upper Columbia Basin. The model is currently programmed in Visual Basic and a more complete version of the program will be available in R statistical language in 2012. Model reduction will commence in 2013, and we anticipate the development of an exportable model in 2014.

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

Identifies Limiting Factors: It quantitatively identifies life-stage specific habitat impairments or limiting factors, enabling habitat restoration actions to better target problems and conversely to avoid habitat initiatives that are unlikely to address primary limiting factors. For example, in the Lemhi River freshwater productivity is believed to suffer from a lack of high quality juvenile rearing habitat. Using observed and modeled juvenile survival, the model can be used to estimate total capacity of available habitat to test the assumption that habitat availability is limiting population growth.

Identifies Project Types: It identifies the types and magnitude of habitat alteration most likely to improve freshwater productivity. For example, the Lemhi River is targeted for habitat restoration actions that add additional tributary habitat (tributary reconnections) and improve the quality of existing habitat (e.g., channel realignment). The model can be used to predict the effects of each type of action on habitat capacity and ultimately freshwater survival.

Compares Project Types: It provides a platform to evaluate alternative restoration actions to identify/prioritize actions most likely to cost-effectively improve freshwater productivity.

Relates Habitat Improvements to Survival Improvements: The model provides an analytical framework that readily combines data from habitat and fish surveys providing an empirical basis as the foundation for modeled predictions.

Identifies Appropriate Research Monitoring and Evaluation: It identifies the types of monitoring most likely to detect changes in habitat conditions and freshwater productivity within a specified period of time.

Evaluates Changes in Habitat and Fish: It provides an analytical tool to quantitatively evaluate change in habitat conditions and freshwater productivity using a number of statistical frameworks including Before-After-Control-Impact designs.

Predicts Adult Returns: It can be used to predict adult escapement by taking into account ocean conditions, harvest, and hatchery impacts. The model places changes in egg to smolt survival in the context of age-structured adult returns, the ultimate metric by which the effectiveness of BiOp actions will be evaluated.

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 round. Regional management agencies identified the loss of tributary habitat as a factor limiting the productivity of spring/summer Chinook salmon and steelhead. Following this finding, significant BPA funding has been allocated towards projects aimed at "reconnecting" tributary habitat historically important for Chinook salmon and steelhead production and improving habitat conditions in existing habitat. The large spatial scale and aggressiveness of these tributary reconnections makes the Lemhi River an ideal case-study for ISEMP. The 2008 BiOp estimated that Lemhi River habitat restoration actions are anticipated to achieve a 3% and 7% increase in egg to smolt survival for steelhead and spring/summer Chinook salmon, respectively. It is unlikely that funding and logistics will enable the reconnection of all Lemhi River tributaries, so managers must choose which tributaries should be the focus of restoration efforts and are there alternative or additional habitat restoration actions that could prove as effective or more effective at achieving the egg to smolt survival improvements identified in the BiOp?

Initially, managers identified "high priority" watersheds as primary targets for "Phase I" restoration efforts. This prioritization was based on existing information describing habitat conditions in concert with the logistical feasibility of obtaining successful tributary reconnections; for example, the 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 I, and identifying whether additional tributary reconnects will be necessary to achieve the freshwater productivity improvements necessary to achieve the goals identified in the BiOp.

The ISEMP watershed model gives policy and management decision-makers a tool that provides a consistent and quantitative 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. The application of this tool will also enable managers to identify why habitat restoration investments to date have or have not delivered the anticipated benefits in freshwater 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 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 required the use of values from literature.

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 egg to smolt survival (Table 18) 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 75 and 76.

Table 18. Percent change in spring/summer Chinook salmon egg to smolt survival under scenarios including 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%

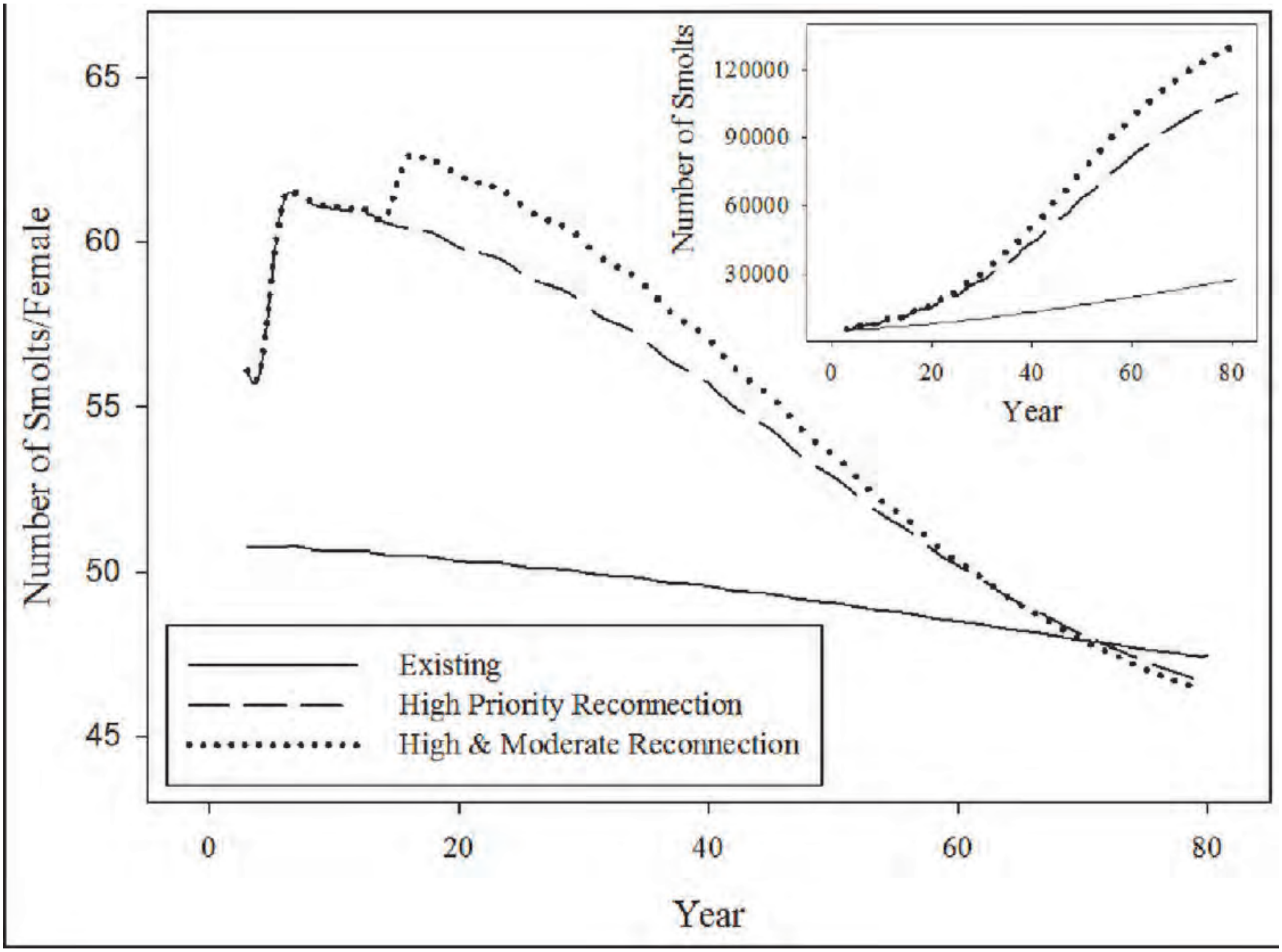


Figure 75. Number of spring/summer Chinook salmon smolts per female and total estimated smolt production (inset) given existing habitat, reconnection of high priority tributaries, and addition of high and moderate priority tributaries.

The provisional model results described above illustrate the utility of the watershed model as a tool for evaluating the outcomes of habitat restoration using the BiOp metrics of egg to smolt survival and adult abundance. This simple summary also demonstrates the value of the model for testing assumptions that guide habitat restoration. First, the model tests the assumption that freshwater survival is limited by habitat quantity and quality. Second the model tests the value of restoration scenarios. Third, the model predicts whether additional habitat restoration actions may be necessary to achieve the targeted improvement in freshwater survival. Finally, the provisional results demonstrate a clear link between habitat restoration and freshwater survival, a key assumption underlying the value of habitat restoration as an offsite mitigation tool. As importantly, the model places changes in freshwater survival into the context of future adult escapement. This component is particularly important given that initial increases in egg to smolt survival are predicted to decrease over time as a result of density dependence, although total smolt production remains much higher than the "existing" scenario (Figure 75). From the perspective of jeopardy, habitat restoration is predicted to stimulate a substantial increase in total adult abundance (Figure 76) despite the short-lived nature of egg to smolt survival improvements.

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 the required survival improvements. These results will be available to support the 2013 BiOp comprehensive check-in and 2017 BiOp evaluation. As importantly, model development can shift towards the goal of identifying the functional relationships and minimum data requirements that support the application of the model in less "data-rich" watersheds, providing a standardized tool for the evaluation of habitat actions across the Columbia Basin.

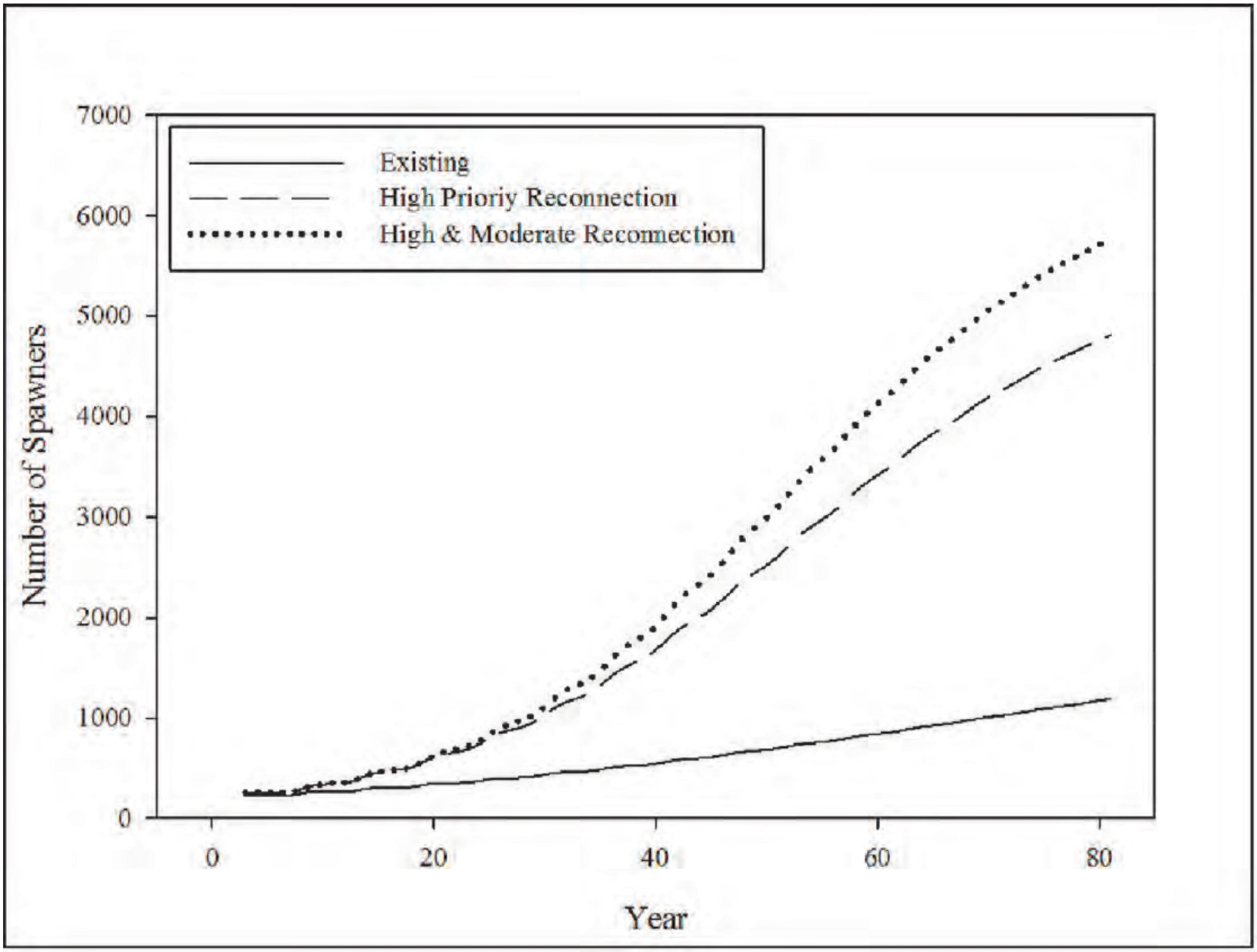


Figure 76. Number of spring/summer Chinook salmon adults returning to the Lemhi River given existing habitat, reconnection of high priority watersheds, and reconnection of high and moderate priority watersheds.

VI. LESSONS LEARNED IN DATA MANAGEMENT

ISEMP has developed several data management products to facilitate data analyses, data storage, and data retrieval within and across ISEMP pilot subbasins. Data management products to date include:

- Detailed documentation of field protocols for collecting data;
- The STEM Databank data repository;
- A standardized data logger application for storing remote tagging information that facilitates quality assurance and export of tag data to PTAGIS, and
- A packaged data capture, storage, quality assurance, and PTAGIS export system for data generated by PIT arrays, and general quality assurance guidelines to facilitate data handling and the production of clean and accurate data.

During the development process of these products we have tested and handled a variety of situations and issues, including the importance and utility of a data dictionary, issues and safeguards for protocol drift, data quality and standards, data flow and transfer, and the benefits/limitations of utilizing multiple protocols within ISEMP. We continue to evolve our strategies for handling these topics as we educate ourselves and collaborators in data handling practices, improve our data handling strategies, and refine our program objectives. Here we briefly introduce the primary data management products and their utility.

Field protocols

The development of field protocols is essential to maximizing the utility of data collected within a program. Protocols are often 'tweaked', which is seen as essential to accommodate program-specific needs; however, these tweaks are not often well documented. As a result, in 2008 all ISEMP-related projects were required to have field protocols that included study designs, field methods, training requirements, and associated metadata (Oakley et al. 2003). Metadata includes the how, what, when, where, and why of data collection. The more contextual information that is known about a dataset the more useful it is for future studies, and documenting metadata also helps maintain the integrity of the original dataset so that observations made in the past are not distorted to reflect current day standards or methods (Beier et al. 2007). Critical metadata includes: 1) study objectives and design; 2) protocols; 3) measurement details (definitions, units and species codes or size classes); 4) data processing procedures (data quality, summary, and metric calculations); and 5) intended analyses (Oakley et al. 2003, Ellison et al. 2006, Bowers et al. 2008).

We adopted the practice of annually reviewing field protocols to determine if field methods were accurate, precise, and contributed to the metrics of interest. Annual review of protocols are essential to maintaining a balance between feasibility of data capture, value of the data, and minimizing change, which facilitates data analysis over long periods of time. This requires careful consideration and definition of the desired metrics and managing protocol drift. ISEMP's best example of this documentation resulted in the explicit documentation of the CHaMP field protocol (Bouwes et al 2012). One of the most important lessons learned to date has been the diligence needed to track and record annual changes in a format that is accessible, easy to maintain, and stored closely with the data. This guards against separation of data from metadata (including the protocol) which can lead to data misuse.

In the testing and development of new monitoring methods for fish and habitat, we intentionally utilized multiple protocols and metrics to capture the data of interest. Although this may seem a counterintuitive strategy for attaining a shared programmatic goal, it has been an essential part of our learning and development process for robust protocols. For example, by utilizing five similar-yet-different- habitat protocols adapted from PIBO, EMAP, AREMP, and standard geomorphology data capture methods for the first 4 years of ISEMP, we were able to test the data utility and feasibility of these protocols. As a result of our previous learning, we were able to develop the CHaMP habitat protocol in 2011 that incorporated technological advances with tested methodologies to generate a robust protocol for habitat monitoring.

STEM Databank data repository

The Status Trend and Effectiveness Monitoring (STEM) Databank (https://www.webapps.nwfsc.noaa.gov/apex_stem/f?p=168:2:662373028707546), was created to store and distribute biological and physical aquatic ecosystem data collected or compiled through ISEMP efforts. STEM houses fish, instream habitat, fine sediment, and water quality data from various state, federal, tribal, and contracted organizations from 2004 - 2010. The majority of data within the STEM Databank is from the Upper Columbia (Wenatchee and Entiat) as data collection for ISEMP first began in the Upper Columbia pilot subbasin and these data were used as template data for development of the database.

Over the course of developing the STEM Databank, we noted that organizations commonly utilize multiple protocols and data storage structures. In some cases both of these items changed annually within the same organization. This presented a challenge for documenting and storing data across years and agencies, especially when there was limited metadata and documentation of data quality accompanying the data. As a result, the STEM Databank was developed as a highly normalized structure that would allow storage and retrieval of data from multiple protocols. Although this was an effective strategy for storing highly diverse data, it requires an immense overhead of metadata, which was often limited in provided data and took several years to document and align with source data. To moderate differences among organization-specific protocols, data storage formats and terminology, ISEMP began utilizing a standardized schema in 2007. This schema, the Aquatic Resources Schema, was used from 2006-2010 to facilitate data entry, metadata documentation, terminology, and import formats for the STEM Databank (Rentmeester 2008). Utilizing the Aquatic Resources Schema was a global-schema approach to managing data from disparate sources, but the structural complexity required to manage the diverse incoming datasets distracted from the feasibility of implementing it effectively across all ISEMP pilot subbasins. However, the Aquatic Resources Schema demonstrated the utility and benefits of storing detailed metadata with raw data and the utility of a global schema for fisheries data. As we discovered, migrating multiple datasets to a single, consistent and defined database structure can be time consuming, but vastly improved the efficiency of long-term data retrieval and use. These concepts have continued to influence the development of other ISEMP data management tools.

Remote tagging data logger application

ISEMP relies heavily on the use of PIT tags to estimate adult escapement and juvenile distribution, abundance, and survival. One of the challenges of utilizing PIT tag data is the lack of standardized data capture and local database utilities. Here we describe advances in ISEMP tool development aimed at addressing these shortcomings for both juvenile PIT tagging surveys and instream PIT tag arrays.

It is important to note that the development of data capture and data management systems is predicated upon there being an underlying protocol. Thus, the application of these tools not only streamlines data collection, QA/QC, and transfer to local and regional databases, they simultaneously enforce the protocols, in turn leading to greater regional standardization. These systems therefore ensure that local and regional monitoring programs provide consistent metrics and indicators for management agencies and scientists, ultimately improving coordination and decision making.

Remote Juvenile Capture and PIT-tagging Utilities

ISEMP has developed a data management system for remote juvenile capture and PIT tagging that incorporates hand-held data loggers for field data collection, a project level data storage standard, data QA/QC modules, and data transfer applications. By enforcing required fields and real-time error checking the electronic data-capture devices minimize data loss and data entry during surveys. As with any survey method, developing a data logger to electronically capture information requires an underlying standardized field collection protocol. Thus, the implementation of a common data logger program across projects simultaneously increases standardization. In addition to improved standardization, electronic data capture reduces the labor and accompanying potential for transcription errors that accompany the transfer of data from field forms to electronic format. Similarly, by enforcing common fields and implementing real-time QA/QC, the use of the data logger significantly reduces the QA/QC burden after the field season, limits data losses resulting from corrupted or incomplete surveys, and expedites data reduction and reporting.

Juvenile remote capture and PIT tagging efforts are utilized in the South Fork Salmon and Lemhi Rivers in Idaho, the Wenatchee and Entiat Rivers in Washington, and the John day Basin in Oregon to generate juvenile abundance, survival, and distribution estimates. In order to efficiently manage data collection and administration, ISEMP has developed a standardized protocol that is supported by electronic data capture devices and an associated database program developed specifically to support the field protocol. Data produced by these utilities for the remote site juvenile capture and tagging are reported to regional databases such as PTAGIS and allows effective and automated transfer to state-level data management systems such as the Idaho Department of Fish and Game Idaho Fish and Wildlife Information System (IFWIS) and BioSamples Database.

Several applications have been developed that collect data using different survey methodologies, including remote juvenile capture surveys and mobile PIT tagging applications. The remote juvenile capture surveys rely on a hand-held data logger (Juniper Systems Allegro CX) that uses a Windows Mobile application to store both metadata and fish sampling information for all survey collection methods (e.g., seining, electrofishing, and angling). The PIT Tagging application interfaces with commonly used tag readers (e.g., Destron FS2001 and Allfex Readers) enabling automated storage of PIT tag codes. Additionally, this application supports automated input from digital scales to record fish weight and GPS devices to record location information (Bluetooth/USB GPS antennas). The application stores the total number of each species captured, fraction tagged, site location, and applies a unique identifier to each sample that links PIT tag codes with accompanying biological samples taken from the individual (scales and tissue samples). Subsequent analysis of these samples enables age and gender to be assigned to the PIT tagged individual.

Recent developments in mobile PIT tag detection technology enable the use of hand-held wands and GPS systems to identify individual fish locations and micro-habitat use. Field technicians use the hand-held electronic data capture device to control both a GPS antenna and PIT tag wand. When a PIT tagged fish is detected, the mobile application automatically adds the GPS coordinates and allows the technician to identify the micro-habitat unit and other additional metadata, such as survey type. Combining the spatial data with individual PIT tag interrogations allows the development of fish survival and distribution models that were previously either impossible to collect sufficient data for or exceptionally costly to implement.

Instream PIT Tag Array Data Management

ISEMP relies heavily on juvenile and adult PIT tagging and interrogation at instream PIT tag detection sites (IPTDS). The development of IPTDS technology represents a significant advancement with regard to the estimation of juvenile and adult distribution, abundance, 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. ISEMP has developed a data management system that allows efficient and real-time access to IPTDS site data, site diagnostics, and data storage and information transfer to regional data systems (Figure 77).

PIT tags allow metric development at both the local and regional scales, but using PIT tags to describe fish population survival and abundance metrics requires a suite of metadata associated with the PIT detections to efficiently and appropriately use the information. By forcing data standards and required fields, we can ensure data quality is consistent across basin-wide datasets and monitoring projects. The complexity of data, and large volumes of information (e.g., 860,000 tag detections and 147 million data values were generated at ISEMP Snake River Basin IPTDS in 2011) required a new system to manage, store and transfer the PIT tag data to PTAGIS, and other users.

The remote locations often selected for IPTDS installations also present a challenge. Given that IPTDS systems are often located in areas with limited seasonal access, it is not cost-efficient or even possible, in some cases, to visit sites for the purpose of downloading interrogation data. Beginning in 2009, ISEMP, in coordination with Biomark, undertook the development of a standardized suite of PIT tag array infrastructure enabling reliable remote downloading of interrogation data and routine site diagnostics. Following that effort, ISEMP developed software that automatically parses downloaded PIT tag interrogation data, reduces the data to required fields for regional databases such as PTAGIS, and uploads the data automatically on a daily basis to PTAGIS. This process both automates data QA/QC and provides detection data, in near real-time, to the region.

Generally, the data generated by IPTDS can be summarized as a unique tag code, date, time, location of interrogation, and various attributes. However, unlike many surveys types, IPTDS are generally assumed to function continuously. Thus, it is critical to know the reliability of IPTDS (e.g., whether there were interruptions due to power outages, etc.). The standardized suite of IPTDS infrastructure at ISEMP sites produce diagnostic data that enable a researcher to determine the reliability of the system. Additionally, diagnostic information such as noise reports for each antenna, enable a variety of analyses such as estimation of instantaneous read range. Lastly, the system stores information on site architecture (e.g., the orientation of antennas in a single array, or orientation of multiple arrays), allowing researchers to assess the statistical approaches most useful to generate efficiency estimates required for programs seeking to utilize IPTDS interrogation data to estimate fish survival and abundance. The volume of data produced by IPTDS, the need for real-time monitoring of IPTDS function, and the desire to conduct analyses aimed at estimating array efficiency and/or directionality of fish movement requires data storage and query resources that are not currently well supported by regional databases.

PIT array data capture and export

In order to efficiently access and store the large quantity of interrogation and diagnostic information ISEMP developed the Instream PIT Tag Detection Database. A server automatically accesses field data via cell phone or satellite modem and stores the information in a SQL database. IPTDS "site stewards" can access the SQL database as necessary to inform users about changes in site architecture and infrastructure or interruptions IPTDS operation that might affect how the data can be used and analyzed.

The sheer volume of interrogation and diagnostic data produced by IPTDS led ISEMP to develop an automated upload system (LNDRefactor) to ensure data was being transferred to PTAGIS in a timely fashion. The program accesses the SQL database, formats the tag data into the PTAGIS data standard, and automatically emails the detection data to PTAGIS on a daily basis (Figure 78). Additionally, LNDRefactor performs automated QA/QC and alerts the site steward of data quality problems. The program has additional functionality enabling the data steward to configure the PTAGIS upload times, site types, and personnel alert lists.

Given the relationship between IPTDS site reliability and data quality, ISEMP developed a real-time monitoring system to alert site stewards to site malfunctions. Conditions at each IPTDS are monitored in real-time on an hourly basis using Loggernet Software from Campbell Scientific, Ltd. The software allows a site data steward to visually monitor sites using a web interface (Figure 79). This interface allows the data steward to determine whether a site is functioning and if any alerts are present. Additionally, if a site malfunctions, or starts to function sub-optimally,

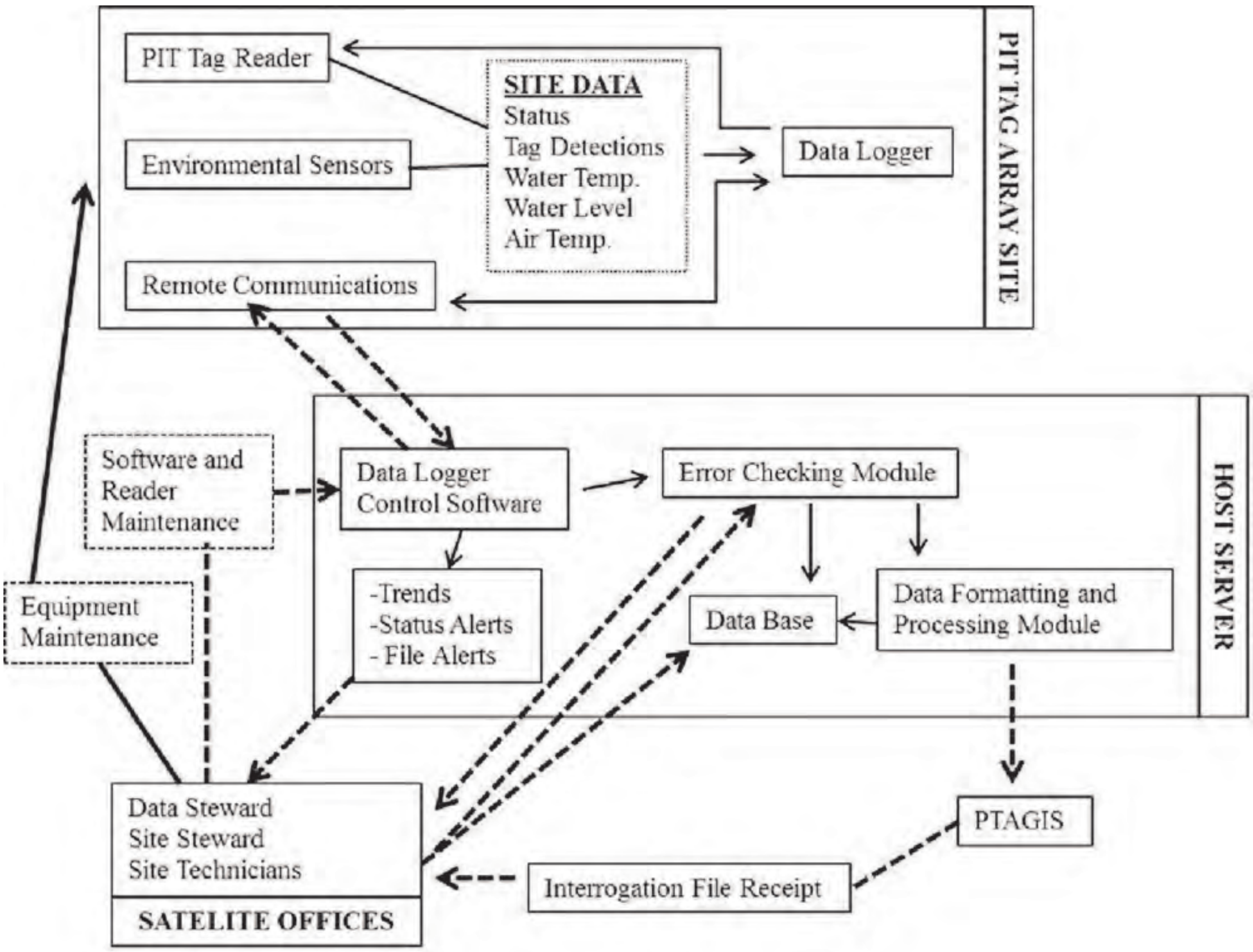


Figure 77. ISEMP Instream PIT Detection Site Data Management System.

an alert is emailed to the site data steward and technical point of contact.

Detailed data associated with each IPTDS can also be accessed using the web interface, allowing the site steward to identify trends in environmental conditions, or potential equipment malfunctions to ensure the site is functioning reliably and to minimize down-time due to equipment failures (Figure 80). By using these tools, we can minimize data loss and ensure data integrity. The data steward also has the ability to log in remotely to control the IPTDS from the office.

Since developing the IPTDS data management system, several ISEMP cooperators in the Upper Columbia and Snake River have adopted this tool. Adoption of the IPTDS data management system has been highly successful in generating a regional data standard, enabled a vast reduction in the labor necessary to reliably operate IPTDS, and ensuring that data of known quality are available to analysts.

The emphasis on tributary IPTDS 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 IPTDS 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 IPTDS. Given the substantial IPTDS 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 will produce a formal suite of database requirements in the summer of 2012. This process is also being coordinated with the development of the regional PIT tag plan.

Data Flow

A central concept in the development of ISEMP's data repositories and capture tools has been to facilitate data flow. The PIT tagging and array data systems heavily rely on electronic data capture (data loggers) and transmission, minimizing errors in transcription of PIT tagging codes, spatial data locations, and general data entry. Microsoft Access databases utilizing the Aquatic Resources Schema facilitated data entry from field forms directly to a database, which facilitated standard terminology and data formatting of field data. Similarly, the CHaMP data management system relies heavily on data loggers and total stations for data capture, and processing tools in both desktop GIS and CHaMPmonitoring.org environments were designed to streamline and accommodate the needs of field crews while maintaining a high standard of data quality. Field crews are often time-limited in the field, and it's critical to meet their data capture needs as accurately and quickly as possible to maximize the investment in field activities.

Tool development requirements

In the course of ISEMP's development of data management tools for data capture and handling, we have addressed difficulties in maintaining and supporting such tools. The development of production-level tools for data capture and handling requires advanced planning and professional programming expertise. In several instances ISEMP has been under field pressures to develop, test, and implement data capture tools simultaneously. This equates to high-risk scenarios in data capture, high stress for programmers, and frustration for field crews utilizing tools with limited pre-field season testing. As a result of these early lessons in tool development, ISEMP now requires early review and change submissions for protocols

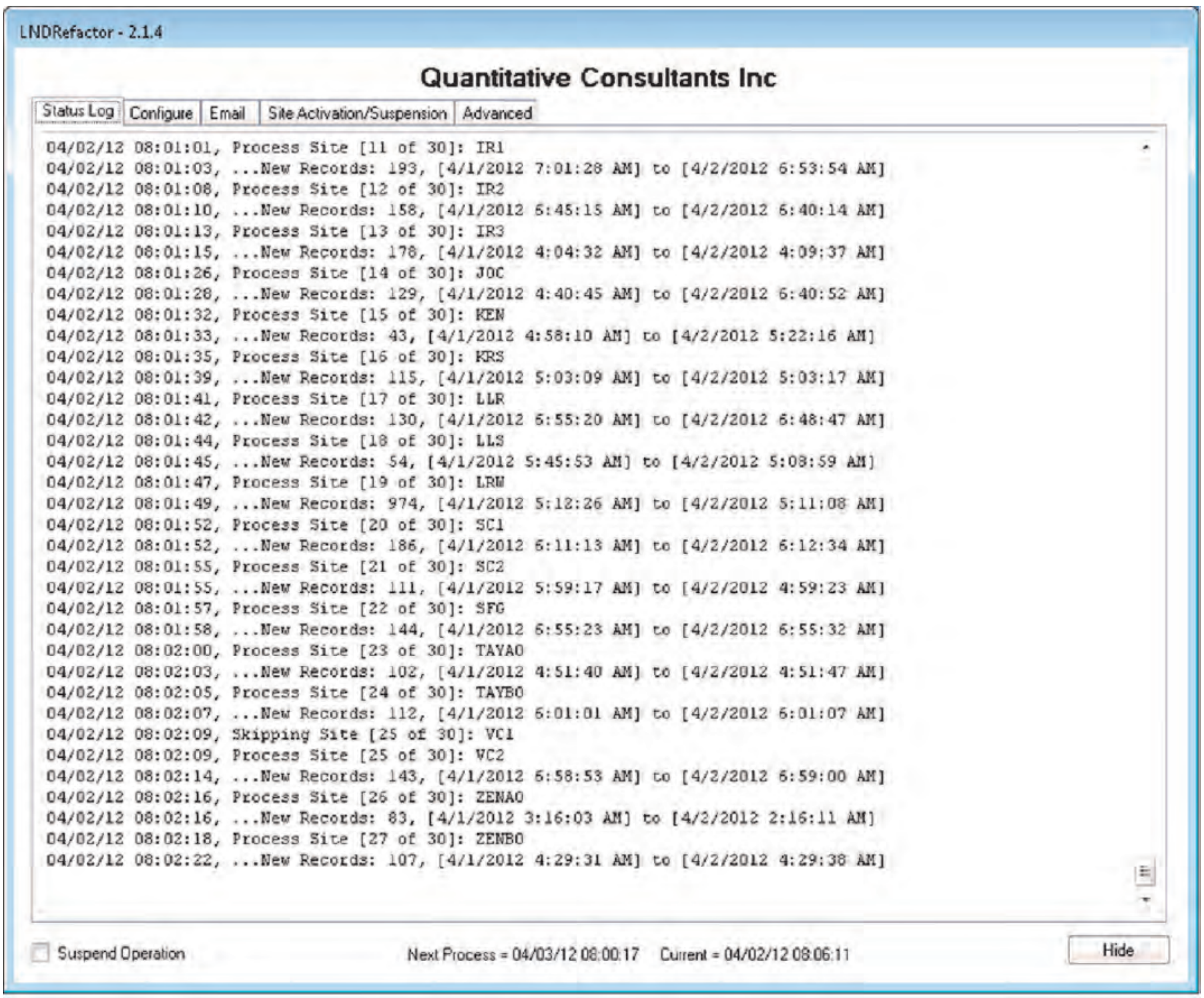


Figure 78. Screen shot of the LNDRefactor program for formatting and automatically transferring detection data to PTAGIS.

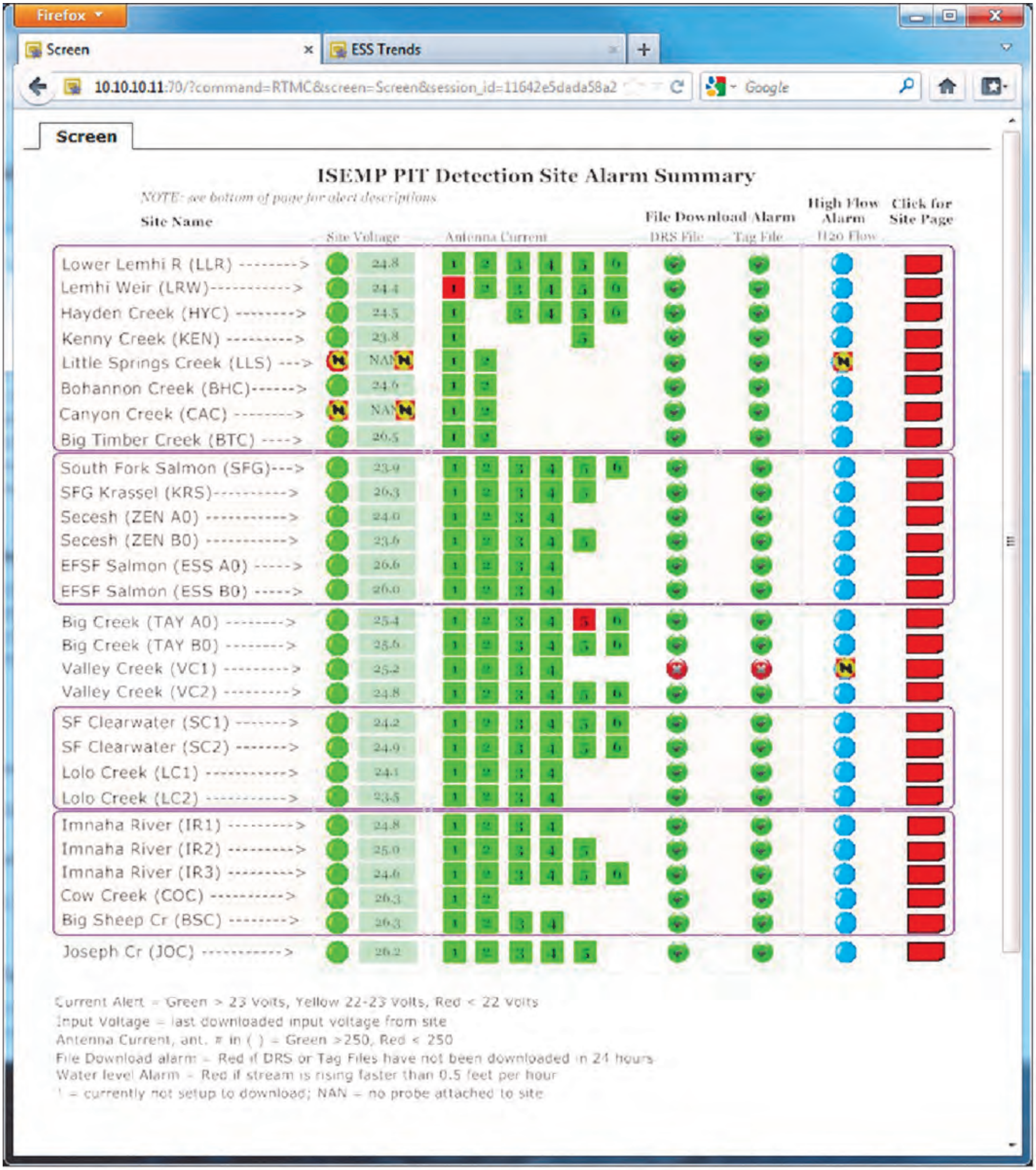


Figure 79. Internet web-interface for ISEMP's Snake River Basin instream PIT tag detection sites monitoring showing current site status, data transfer conditions and current alerts.

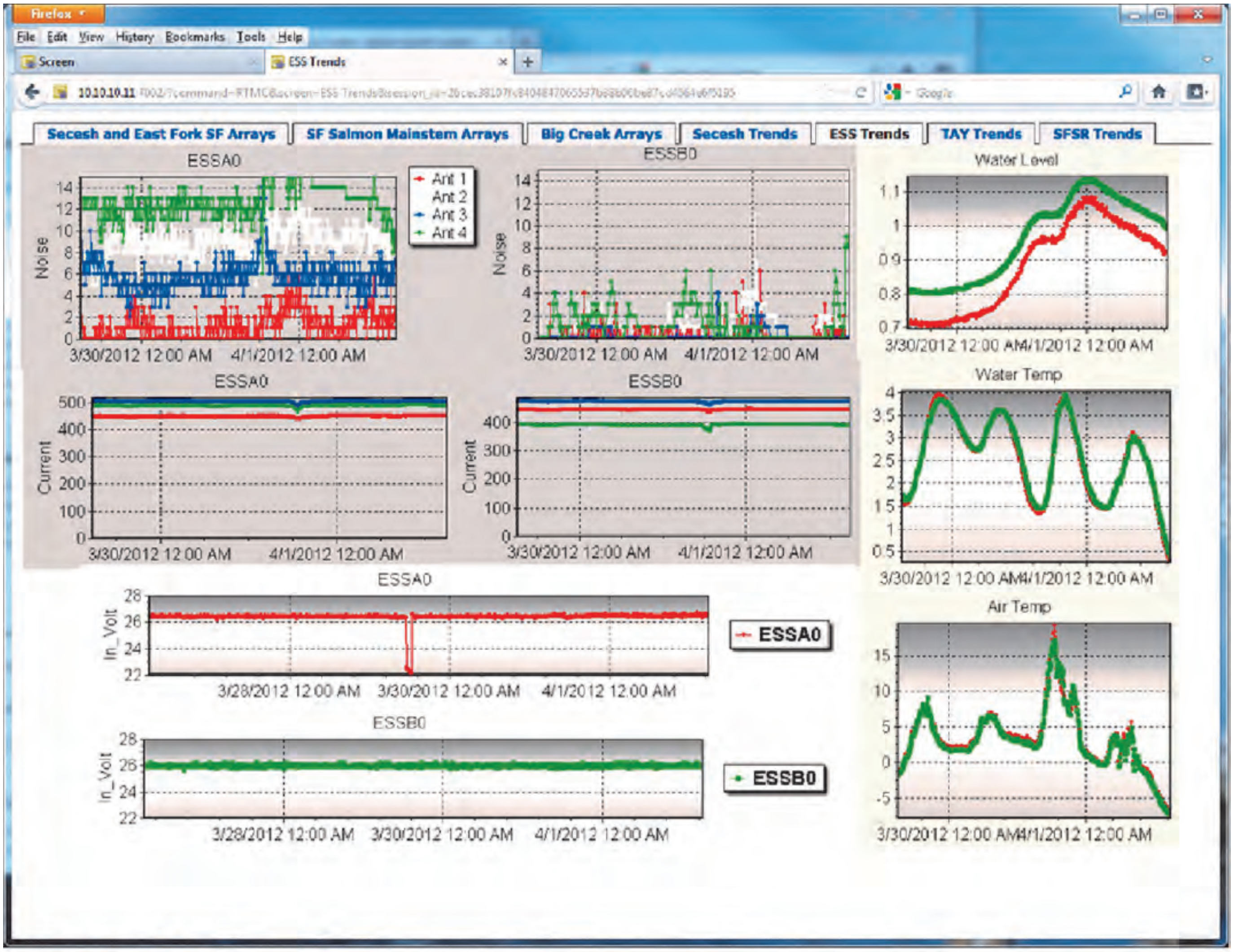


Figure 80. Example of specific instream PIT tag detection site environmental conditions, equipment functions and diagnostic data, and input power levels for the East Fork South Fork Salmon River detection site.

affecting data capture and handling tools. Although this is met with some consternation from field biologists, as protocol review and update deadlines are in early spring, these deadlines allow sufficient time for programming and development of robust data capture tools.

Quality assurance guidelines

ISEMP data quality has greatly improved over the years as our data management tools, standards, documentation and data handling have improved. ISEMP has focused much of its energy on developing dataset-specific checks for completeness, referential integrity, and accuracy. Through both data compilation exercises within ISEMP (e.g., collation of water temperature data throughout the John Day Basin) and the handling of ISEMP-generated datasets, we have learned that:

1) It is critical for field data collectors to review data soon after data collection;

2) Quality assurance checks performed by field crews should conform to a programmatic standard;

3) Quality assurance checks should be replicated within the program (e.g., performed locally and by a central quality assurance manager; and

4) Quality standards should be reviewed annually and prior to each analysis.

Although these rules are generally well known, they are not often practiced in a consistent fashion. Our data quality guidelines include a general outline of data quality checks for different data handlers within the program and customized data quality queries developed for field personnel for habitat and fish datasets. We continue to integrate data quality checks into our data management systems to ensure data quality checks are performed during data entry and review. The PIT tagging, PIT array, and CHaMP habitat data systems integrate data quality and completeness checks into the data management system, which is a direct result from our learned experiences in handling data prior to the development of these systems.

In summary, ISEMP's lessons in data management practices have improved data flow, quality and documentation of ISEMP's aquatic monitoring data. We now readily acknowledge and surface nuances in data storage that have developed from differences in terminology, artifacts from agency templates, and lack of required uniformity in data storage and work to amend these issues in an upfront manner. We strive to include data management concepts and personnel in both data capture, processing and analysis conversations to anticipate and reduce issues that may surface during the course of handling data for ISEMP. Supporting our program's methodology and analytical procedures has evolved over the lifespan of ISEMP and we continue to adapt our data management tools to take advantage of technological advances, new developments in procedures and needs, and workflow.

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VIII. APPENDICES

CHAPTER 1: Habitat Status and Trends Monitoring

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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.

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, left hand panel) and site specific (Figure 1.2 right hand panel) 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.

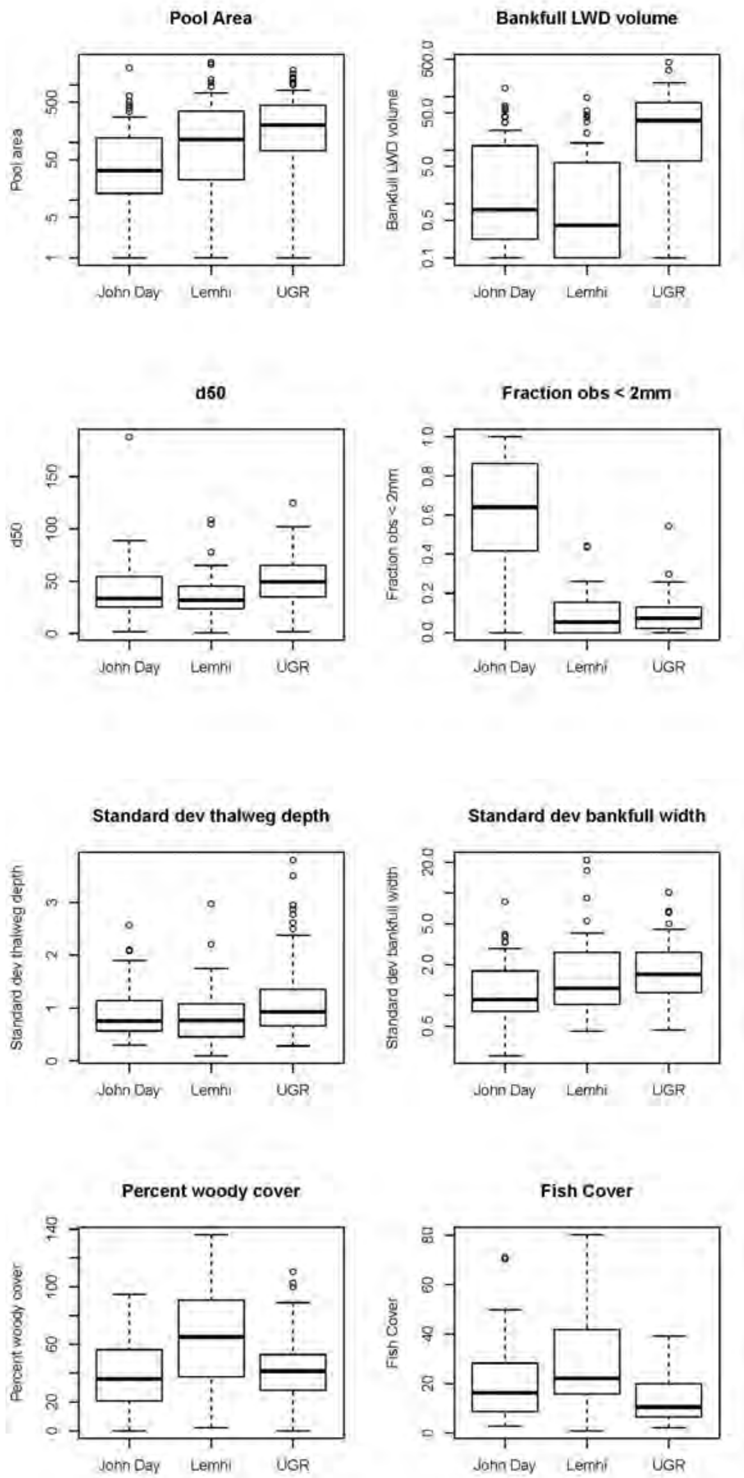


Figure 1.1. A Bayesian trend analysis evaluating average trend in each of five sub-watersheds in the Wenatchee basin reveal the likelihood that an average trend is either positive or negative. Color coding reveals the probability that a negative (red) or positive (black) trend is detectable in each of 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.

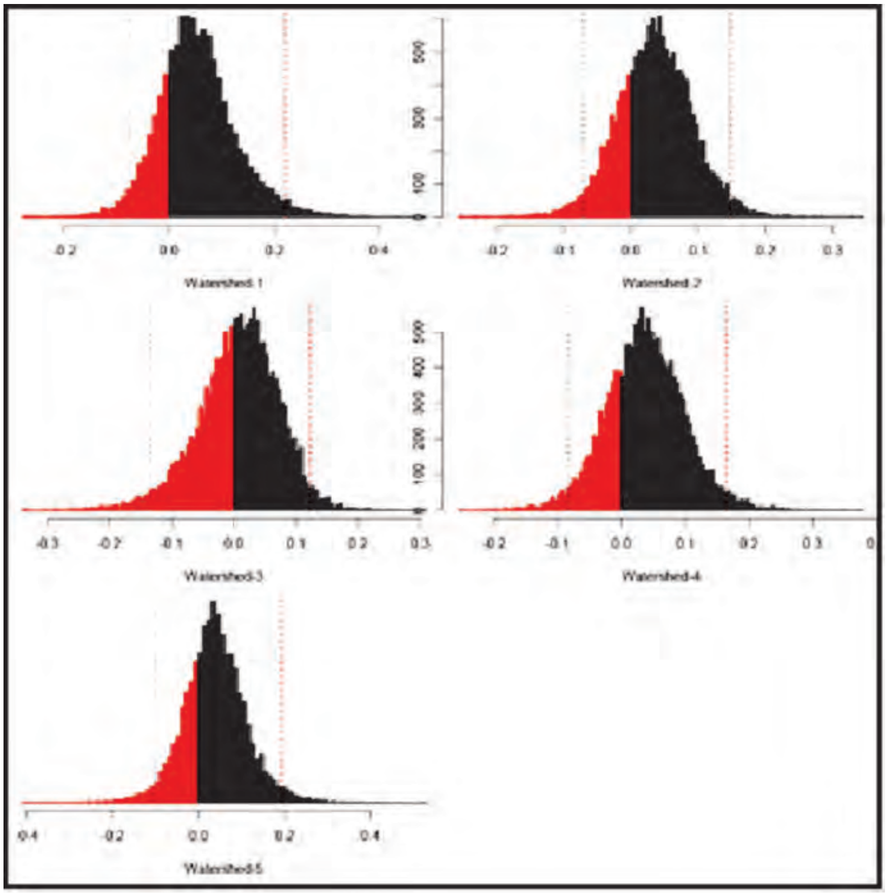
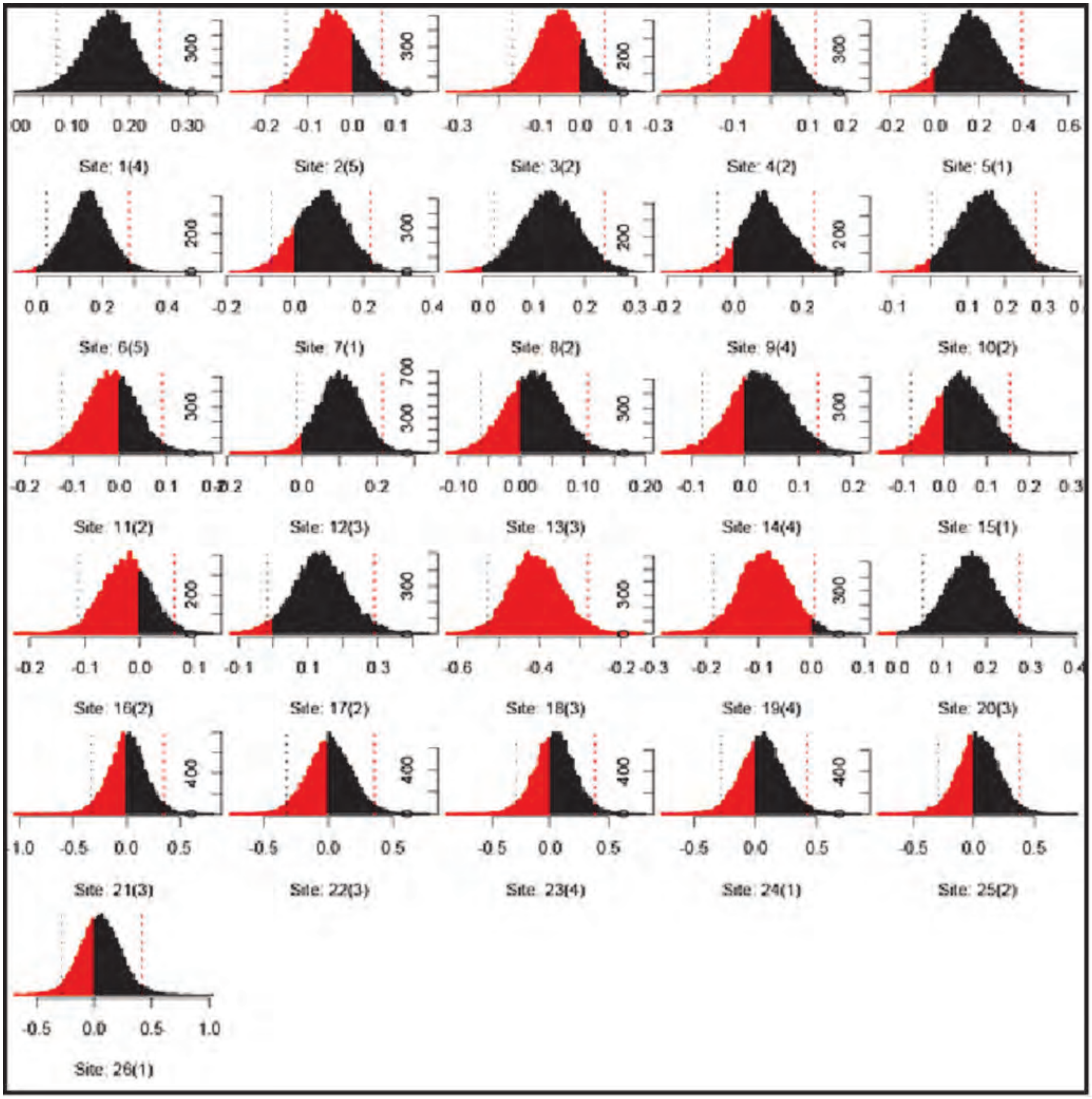


Figure1.2. (This page and opposite page). A similar Bayesian trend analysis run on each of the sites comprising the subwatersheds in Figure 1.2 reveals the variation in estimated site specific trends. For example, there is strong evidence for a negative trend in only a couple of sites (distribution is 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 1.3 and 1.4) 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.

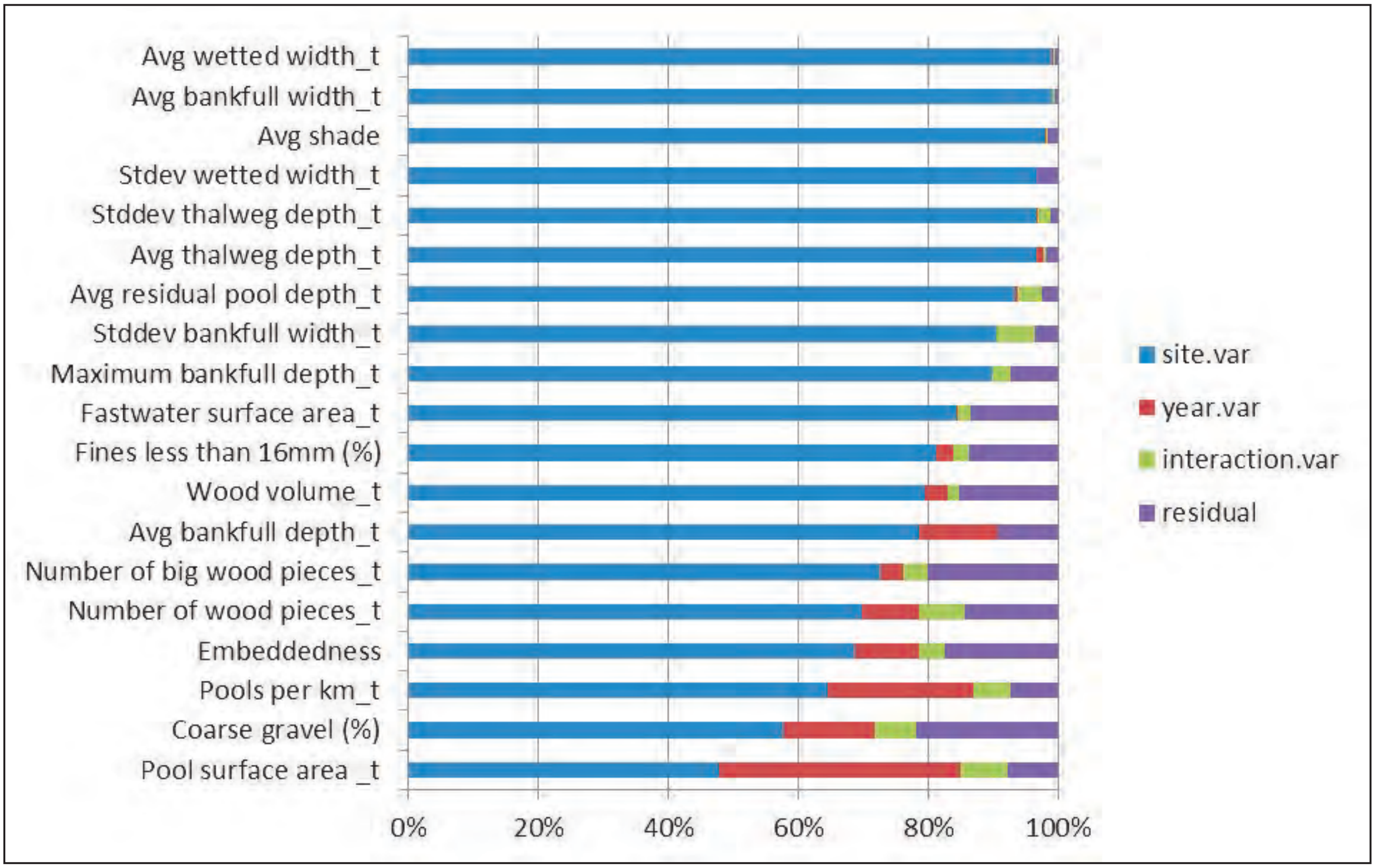


Figure 1.3. This graph illustrates the relative proportion of total variation that is attributable to site, concordant (year), interaction, and residual variation, as described in the text. Data come from the ISEMP habitat surveys in the Wenatchee watershed collected during the sampling seasons from 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.

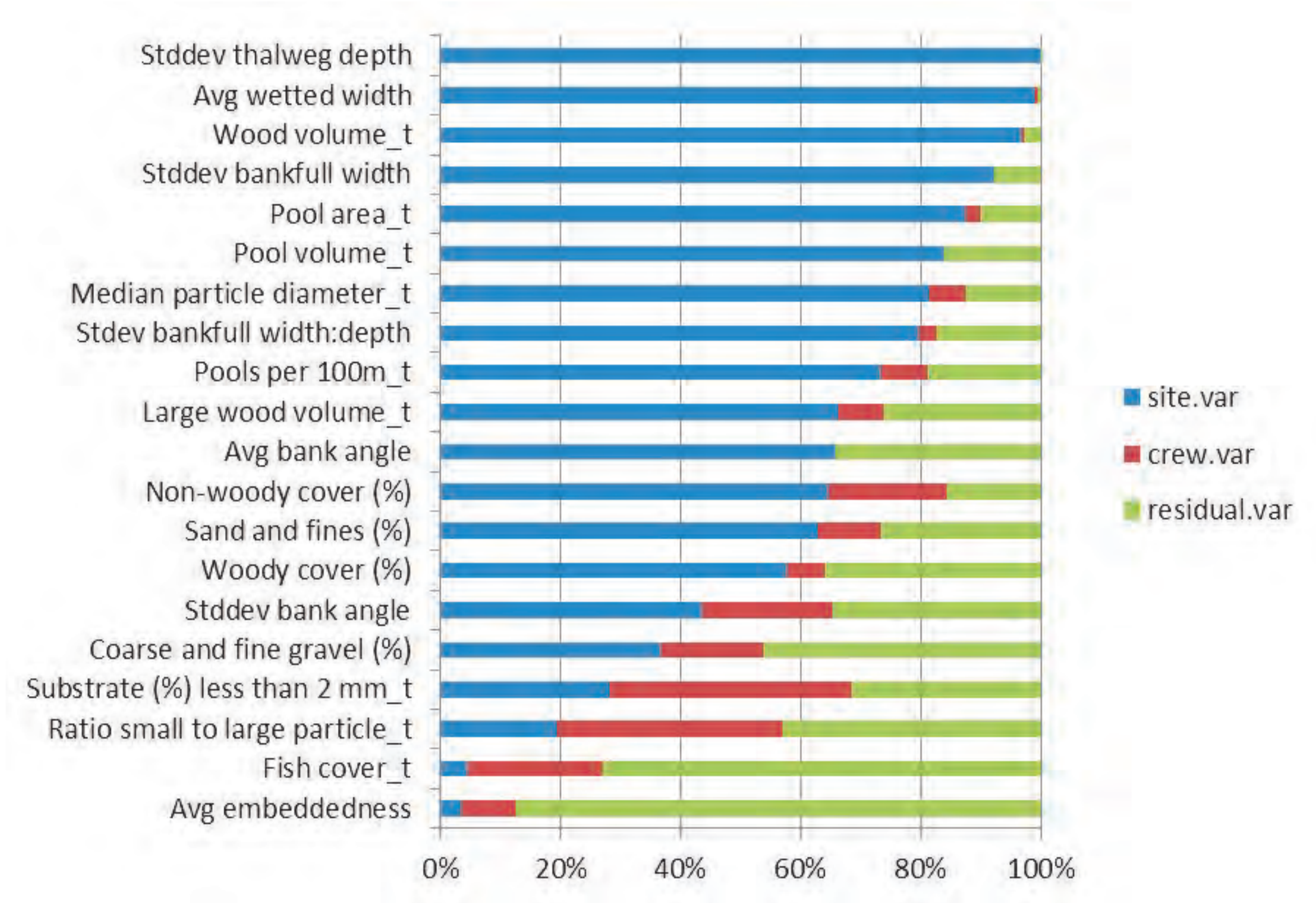


Figure 1.4. As part of the CHaMP surveys during 2011, a design to evaluate the performance of different crews at the same sites was developed in which several crews sampled the same six sites in the Upper Grande Ronde during a short time interval.

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.

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

Authors: Jody S. White and Brice X. Semmens

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, RPA 50.5), tributary/population escapement estimates (RPA 50) as well as age and sex composition required to meet the information needs of the ISEMP watershed model and the BiOp (NMFS 2008). Based on the preliminary success of this application in the Snake River Basin, a similar run decomposition program has been adopted in the upper-Columbia, relying on adult PIT tagging at Priest Rapids Dam.

Assuming a known run size past LGD and a known tagging rate, we can estimate the total number of fish in any location PIT tags are detected with known efficiency (e.g., at instream PIT tag arrays, weirs, and dams). Unfortunately, estimating the run-at-large at LGD is complicated by "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 typical-

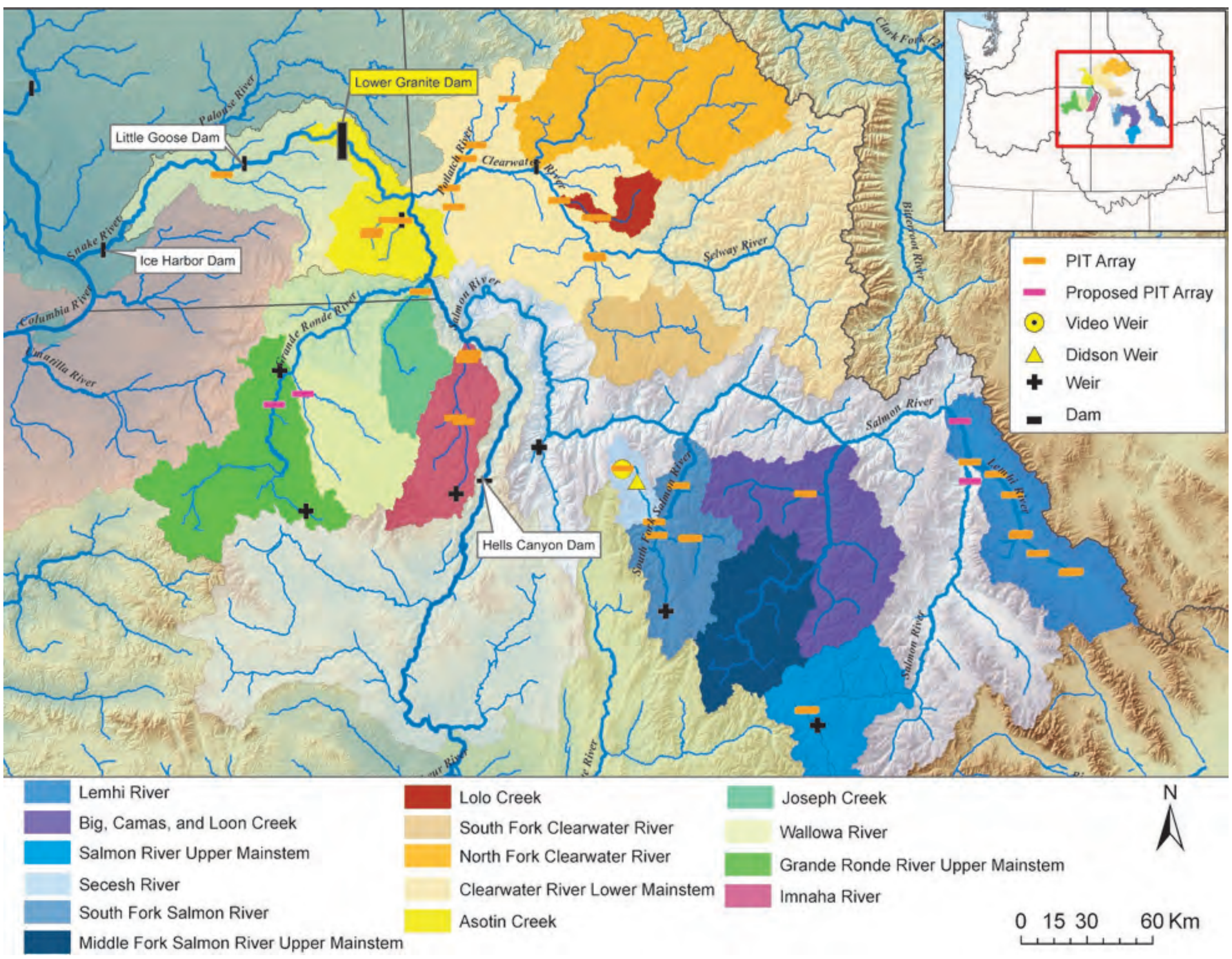

Figure 2.1. ISEMP and other state agency Snake River Basin instream PIT detection sites and PIT collection locations (weirs) relative to the Interior Columbia Basin Technical Recovery Team's population designations for spring/summer-run Chinook salmon.

ly 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 statistical approaches to estimate tributary/population escapement. During periods of consistent tagging rates, lower temperatures, and consistent ladder count schedules (e.g., the spring/summer Chinook salmon migration) we can use maximum likelihood based mark/recapture models. During periods of inconsistent tagging rates, count periods, and trap operations (e.g., the 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 that explicitly account for sampling inconsistencies from fish ladder counts and trap operation.

Notably, ISEMP PIT tagging at Lower Granite Dam is coordinated with BPA Project 2010-026-00, which uses genetic techniques to assign natural origin adult spring/summer Chinook salmon and steelhead to a population or Major Population Group of origin. Approximately half of the roughly 4,000 natural origin spring/summer Chinook salmon and 4,000 natural origin steelhead targeted for ISEMP PIT tagging are genotyped by this project. The coordination of these two sampling efforts both reduces total fish handling and enables a side-by-side comparison of the efficacy of the two methods for generating population, major population group (MPG), and distinct population segment (DPS) adult escapement estimates. Genetic analysis of the samples enables estimates of gender, allowing the resulting estimates of escapement to be partitioned into male and female components. Additionally, the two projects share the cost

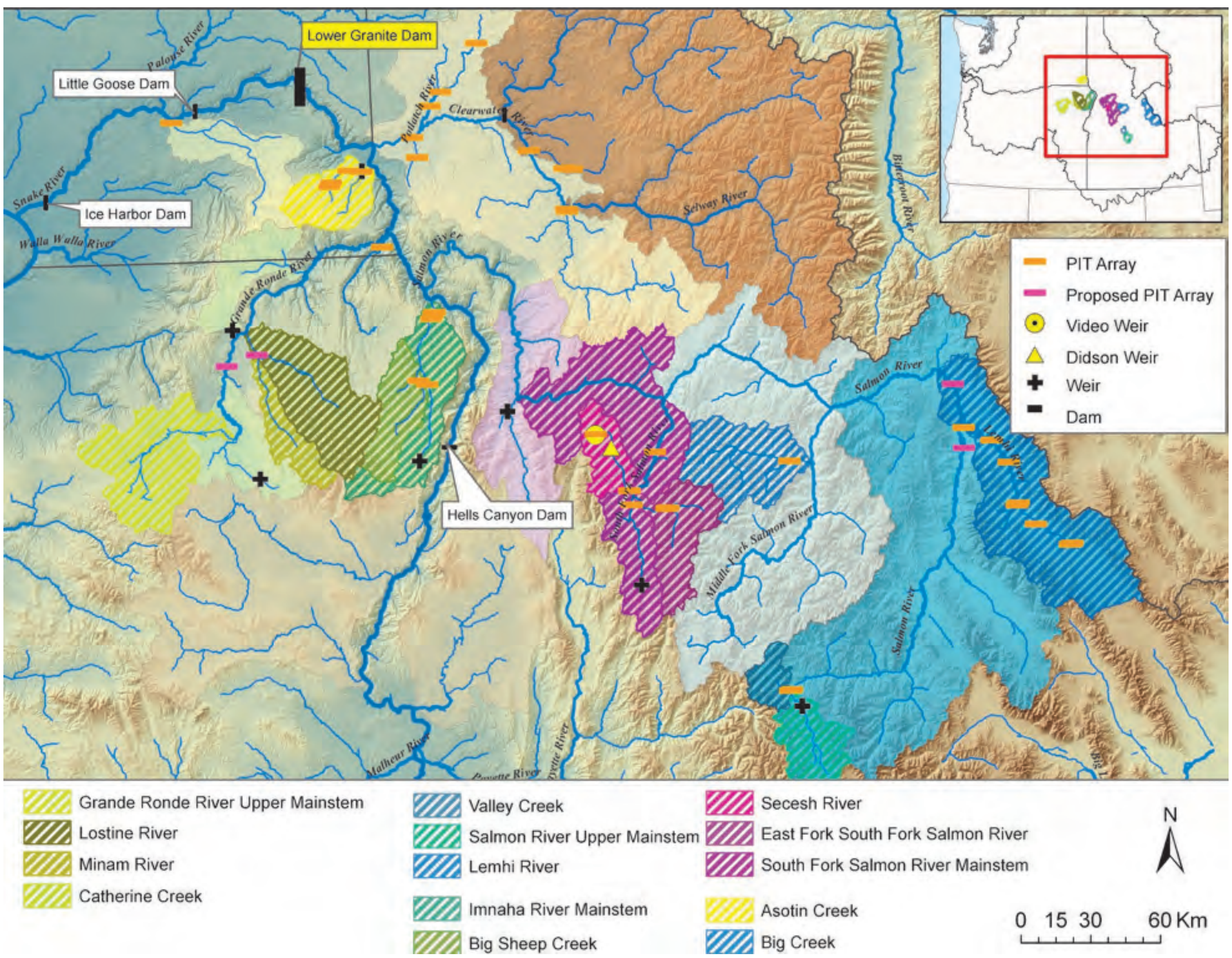

Figure 2-2. ISMEP and other state agency Snake River Basin instream PIT detection sites and PIT collection locations (weirs) relative to the Interior Columbia Basin Technical Recovery Team's population designations for steelhead.

of aging scales, allowing estimation of escapement by age, which is necessary to calculate returns-per-spawner as described in the BiOp.

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 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. During these periods, estimates of the number of returning adults are generated from window counts. In order to assign these counted (but not tagged) fish to 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 populations 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 at the window. 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):

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

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 that are 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):

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

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

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):

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

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

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

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 Table X.

Table 2.1. Spring/summer Chinook salmon and 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 2010 and 2011 run-year Chinook salmon tributary escapement estimates are found in Table X. 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. Run-year 2011 was tagged at a 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).

Table 2.2. Spring/summer Chinook salmon run year, Major Population Group (MPG), population, fraction of population sampled, escapement estimate, coefficient of variation (CV), and independent estimate (if available) monitored by ISEMP PIT tag arrays in the Snake River Basin.

Run-Year	MPG	Population	PIT Tag Decomposition				Independent Estimate		
			Fraction Sampled[1]	Escapement	CV	95% CI	Escapement	CV	95% CI
2010	South Fork Salmon	Mainstem	100%	4,671	3.7%	± 340			
2010		Secesh	100%	1,308	5.6%	± 143			
2010		East Fork	100%	1,026	0.5%	± 11	1,032	N/A	N/A
2010	Middle Fork	Big Creek	100%	285	24.2%	± 135			
2010	Upper Salmon	Valley Creek	100%	235	9.6%	± 44			
2010		Lemhi	95%	262	3.7%	± 19			
2011	Grande Ronde/Imnaha	Imnaha	100%	2,421	6.3%	± 297			
2011	South Fork Salmon	Mainstem	100%	3,318	6.5%	± 423			
2011		Secesh	100%	779	2.2%	± 34			
2011		East Fork	100%	652	0.2%	± 3			
2011	Middle Fork	Big Creek	100%	449	18.1%	± 159			
2011	Upper Salmon	Valley Creek	100%	460	8.9%	± 80			
2011		Lemhi	95%	337	16.2%	± 107			

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 natural origin 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). 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) natural origin steelhead over LGD. Table X summarizes tributary run estimates above instream PIT tag arrays and independent estimates where available.

Table 2.3. Steelhead run year, Major Population Group (MPG), population, subpopulation fraction of population sampled, escapement estimate, coefficient of variation (CV), and independent estimate (if available) monitored by ISEMP PIT tag arrays in the Snake River Basin. Shaded rows identify opportunistic independent estimates of escapement, primarily comprising locations where PIT tag wands are utilized to interrogate PIT tags.

Run-Year	MPG	Population	Subpopulation	PIT Tag Decomposition				Independent Estimate		
				Fraction Sampled[1]	Escapement	CV	95% CI	Escapement	CV	95% CI
2009-2010	Lower Salmon	Asotin Creek[2]		95%	1,687	8.5%	± 280	1,500	N/A	N/A
2009-2010	Salmon River	South Fork		90%	1,497	9.1%	± 268			
2009-2010		Secesh		100%	298	22.1%	± 129			
2009-2010		Middle Fork	Big Creek	100%	753	21.8%	± 322			
2009-2010		Upper Salmon	Valley Creek	100%	237	17.7%	± 82			
2009-2010		Upper Salmon	Pahsimeroi River[2]	100%	138	22.9%	± 62	115	N/A	N/A
2009-2010		Lemhi River		95%	630	14.2%	± 175			
2009-2010		Little Salmon	Rapid River[2]	95%	136	24.0%	± 64	150	Census	Census
2009-2010	Clearwater River	Lochsa River	Fish Creek[2]	100%	246	58.1%	± 117	205		164-255
2010-2011	Lower Salmon	Asotin Creek[2]		95%	890	10.0%	± 175	1,128	2.0%	± 44
2010-2011	Grande Ronde	Joseph Creek[3]		100%	1,627	1.4%	± 45	1,698	22.4%	± 744
2010-2011	Imnaha River	Imnaha River		100%	3,298	1.5%	± 97			
2010-2011		Imnaha River	Cow Creek	100%	147	1.4%	± 4			
2010-2011		Imnaha River	Big Sheep Creek	100%	765	2.2%	± 33			
2010-2011	Salmon River	South Fork		90%	2,540	1.9%	± 93			
2010-2011		Secesh		100%	397	3.1%	± 24			
2010-2011		Middle Fork	Big Creek	100%	687	1.6%	± 22			
2010-2011		Upper Salmon	Valley Creek	100%	232	1.5%	± 7			
2010-2011		Lemhi River		95%	428	1.7%	± 14			

[1]Fraction sampled refers to the fraction of spawning believed to occur above PIT tag arrays.
[2]Weirs that capture and enumerate steelhead and use handheld wands to identify PIT tags, but do not have PIT tag arrays.
[3]Locations with weirs that capture and enumerate steelhead and use handheld wands to identify PIT tags and also have neighboring PIT tag arrays.
[4]Independent estimate generated from a video weir paired with a single PIT tag array.

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 PIT tag demonstrates the potential for instream PIT tag arrays to provide efficient, cost-effective, and accurate estimates of tributary escapement.

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CHAPTER 3: Watershed Production Model

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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.

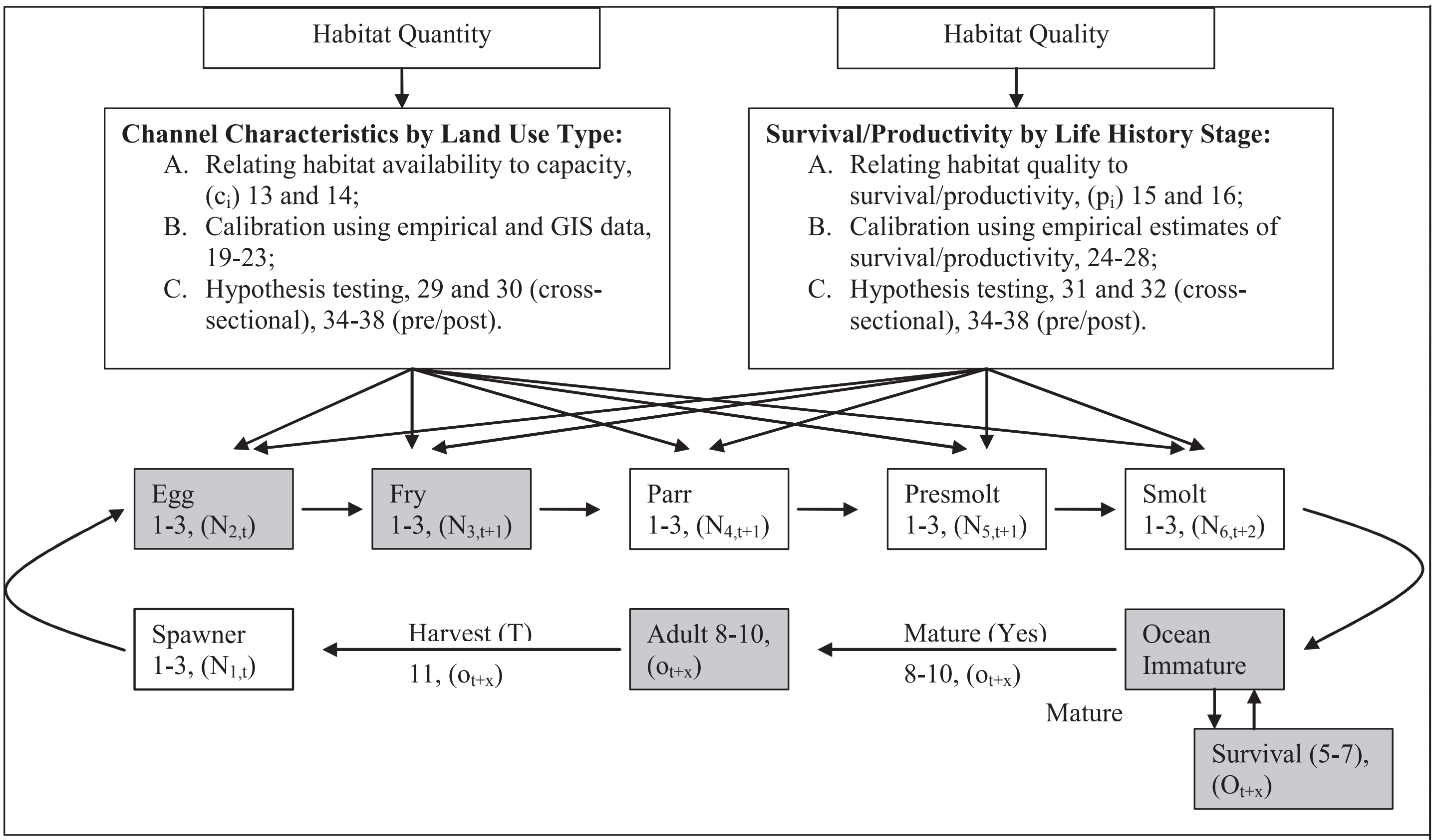

Figure 3-1. Schematic illustrating how the model develops relationships between habitat quantity (capacity) and quality (survival/productivity) and stage-based abundance, productivity, and survival. Grey boxes indicate those life stages for which metrics will be inferred, notation in parentheses refers 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):

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

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.

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

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.

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

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:

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

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. CH— spring/summer Chinook, SH—steelhead

	Instream (Quantity/Quality) Wading Surveys (Yearly Panel)				Fish Surveys (carrying capacity and productivity estimates) Measurement = Density and Survival								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 CH Age 3	SH Age 3 CH Age 2	SH Age 2 CH Age 1	SH Age 1 CH Age 0	SH Age 0										
2010			37		SH Age 5 CH Age 4	SH Age 4 CH Age 3	SH Age 3 CH Age 2	SH Age 2 CH Age 1	SH Age 1 CH Age 0	**SH/CH Spaw** **SH Age 0**									
2011		55			SH Age 6 CH Age 5	SH Age 5 CH Age 4	SH Age 4 CH Age 3	SH Age 3 CH Age 2	SH Age 2 CH Age 1	**SH Age 1** **CH Age 0**	SH/CH Spaw SH Age 0								
2012	X		55		SH Age 7	SH Age 6 CH Age 5	SH Age 5 CH Age 4	SH Age 4 CH Age 3	SH Age 3 CH Age 2	**SH Age 2** **CH Age 1**	SH Age 1 CH Age 0	SH/CH Spaw SH Age 0							
2013				**55**	**SH Age 8**	**SH Age 7**	**SH Age 6** **CH Age 5**	**SH Age 5** **CH Age 4**	**SH Age 4** **CH Age 3**	**SH Age 3** **CH Age 2**	**SH Age 2** **CH Age 1**	**SH Age 1** **CH Age 0**	**SH/CH Spaw** **SH Age 0**						
2014		55				SH Age 8	SH Age 7	SH Age 6 CH Age 5	SH Age 5 CH Age 4	SH Age 4 CH Age 3	SH Age 3 CH Age 2	SH Age 2 CH Age 1	SH Age 1 CH Age 0	SH/CH Spaw SH Age 0					
2015			55				SH Age 8	SH Age 7	SH Age 6 CH Age 5	SH Age 5 CH Age 4	SH Age 4 CH Age 3	SH Age 3 CH Age 2	SH Age 2 CH Age 1	SH Age 1 CH Age 0	SH/CH Spaw SH Age 0				
2016	X			55				SH Age 8	SH Age 7	SH Age 6 CH Age 5	SH Age 5 CH Age 4	SH Age 4 CH Age 3	SH Age 3 CH Age 2	SH Age 2 CH Age 1	SH Age 1 CH Age 0	SH/CH Spaw SH Age 0			
2017		55							SH Age 8	SH Age 7	SH Age 6 CH Age 5	SH Age 5 CH Age 4	SH Age 4 CH Age 3	SH Age 3 CH Age 2	SH Age 2 CH Age 1	SH Age 1 CH Age 0	SH/CH Spaw SH Age 0		
2018			55							SH Age 8	SH Age 7	SH Age 6 CH Age 5	SH Age 5 CH Age 4	SH Age 4 CH Age 3	SH Age 3 CH Age 2	SH Age 2 CH Age 1	SH Age 1 CH Age 0	SH/CH Spaw SH Age 0	
2019				55							SH Age 8	SH Age 7	SH Age 6 CH Age 5	SH Age 5 CH Age 4	SH Age 4 CH Age 3	SH Age 3 CH Age 2	SH Age 2 CH Age 1	SH Age 1 CH Age 0	SH/CH Spaw 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	1991-073-00	INPMEP	IDFG	Adult Biosampling	t, c, o	t, c, o		
Lower Granite Dam	1990-055-00	ISMES	IDFG	Adult Biosampling		t, c, o		
Lower Granite Dam	2003-017-00	ISEMP	QCInc	Adult PIT Tagging/Biosampling	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
Lower Granite Dam	1991-073-00	INPMEP	IDFG	Juvenile Biosampling			s, t, c, o	
Lower Granite Dam	1990-055-00	ISMES	IDFG	Juvenile Biosampling				s, 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			

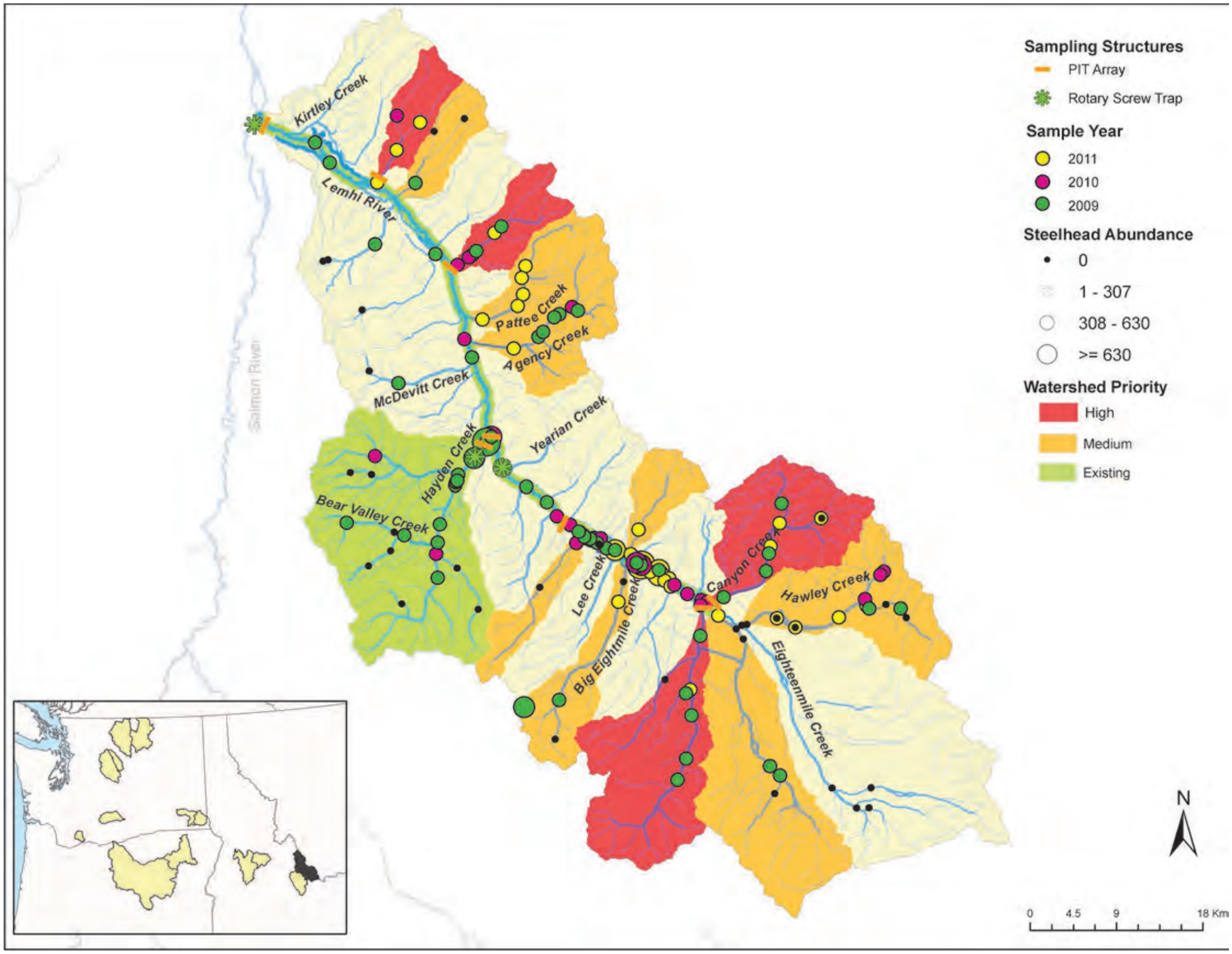

Figure 3-2a. Abundance of steelhead at Lemhi River remote juvenile capture and tagging locations sampled in 2009, 2010, and 2011. Also shown is the distribution of sampling infrastructure. Subwatershed coloration identifies habitat available at the inception of ISEMP (existing) and high and moderate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

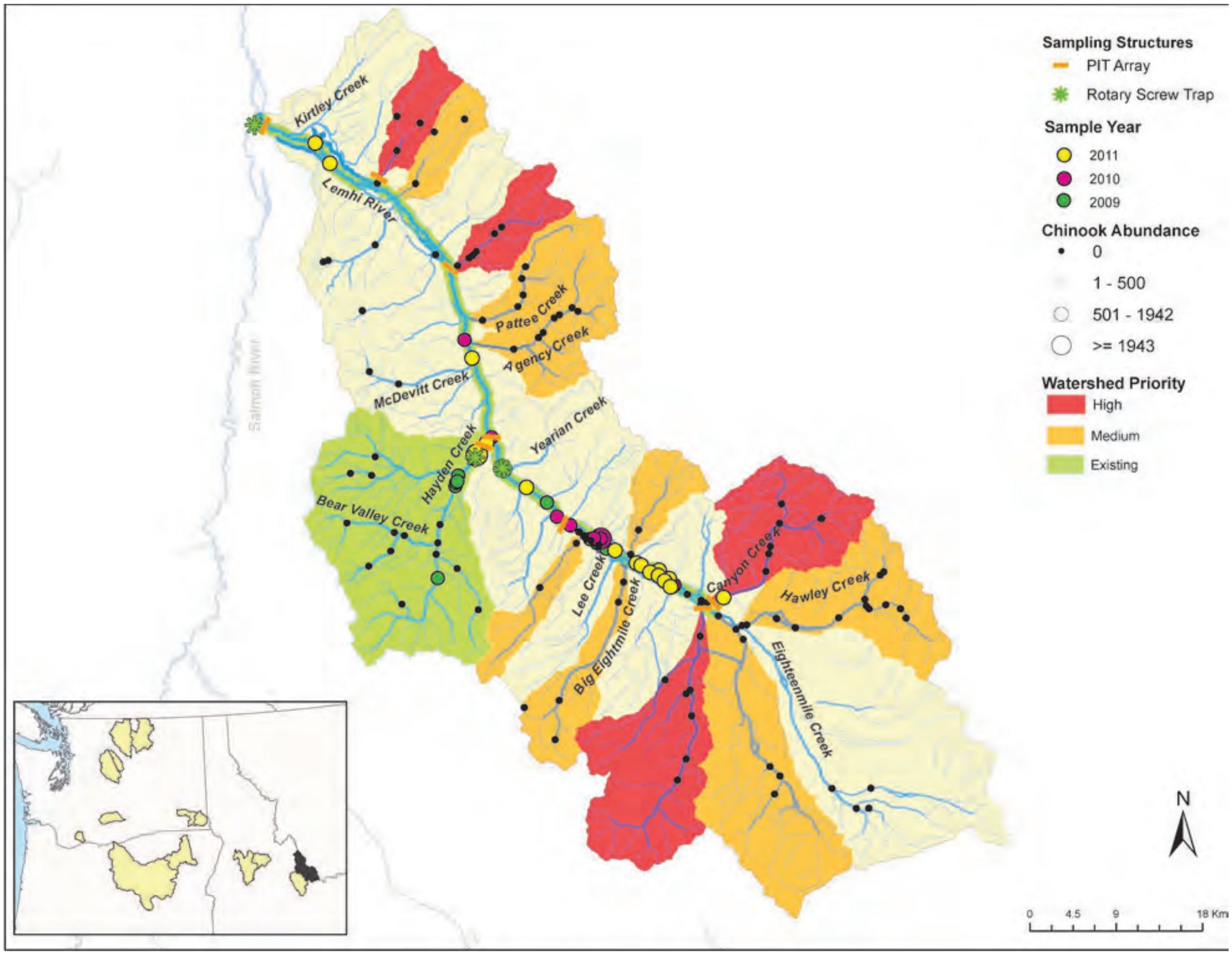

Figure 3-2b. Abundance of spring/summer Chinook salmon at Lemhi River remote juvenile capture and tagging locations sampled in 2009, 2010, and 2011. Also shown is the distribution of sampling infrastructure. Subwatershed coloration identifies habitat available at the inception of ISEMP (existing) and 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).

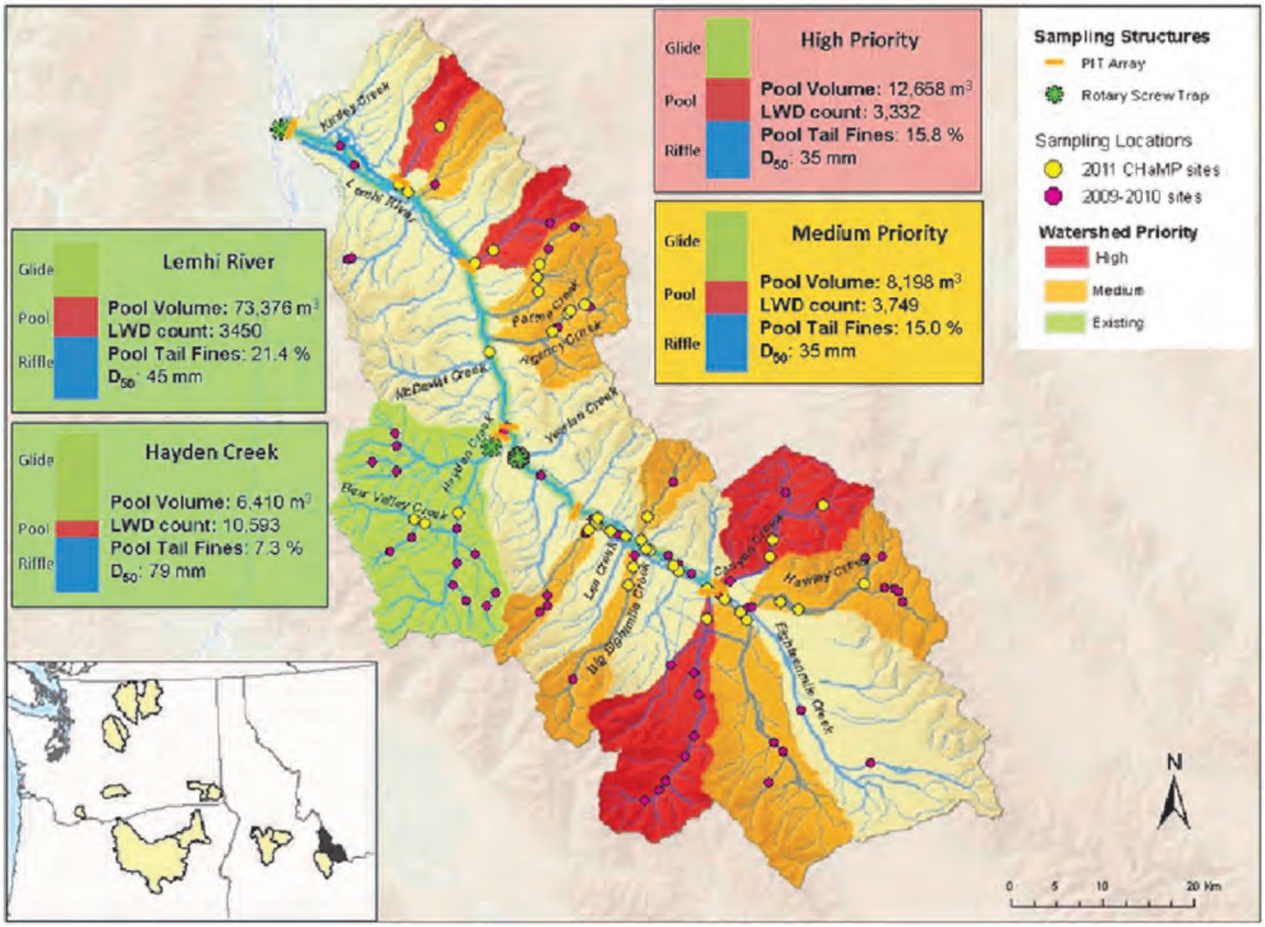

Figure 3-3. Location of 2009, 2010, and 2011 habitat surveys in the Lemhi River. Habitat indicators are summarized by habitat available at the inception of ISEMP (Lemhi mainstem and Hayden Creek; existing) and high and moderate priority subwatersheds identified for reconnection.

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	Survival
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 (egg to smolt survival) 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 3-10) 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 3-4 and 3-5.

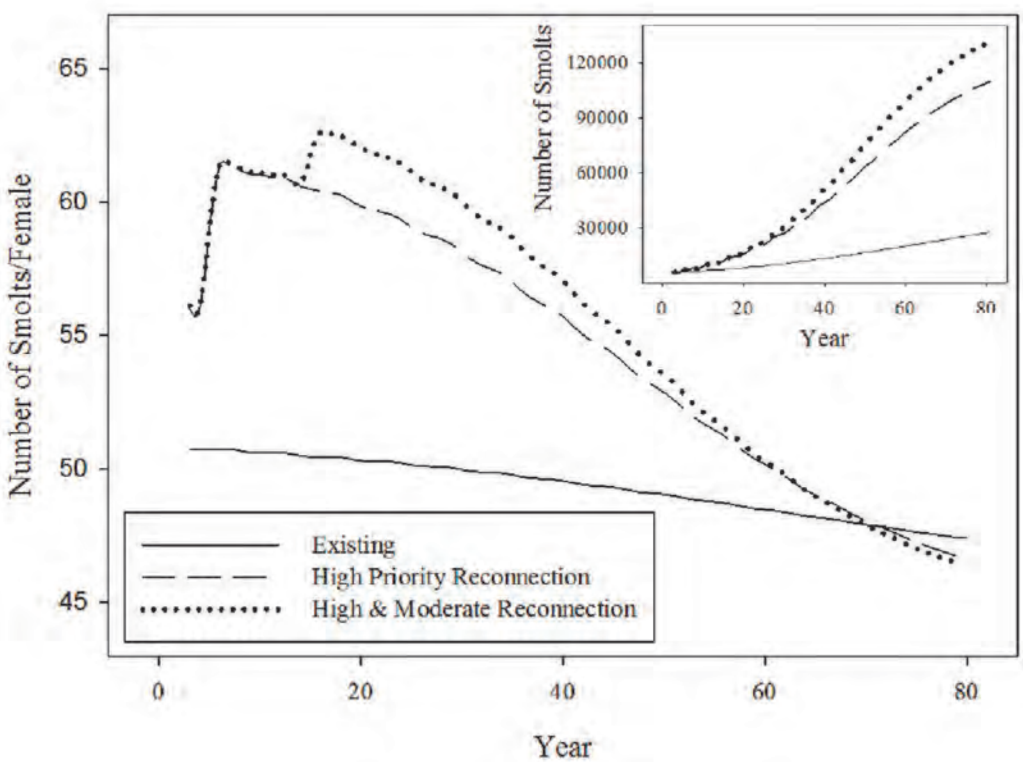

Figure 3-4. Number of spring/summer Chinook smolts per female and total estimated smolt production (inset) given existing habitat, reconnection of high priority tributaries, and addition of high and moderate priority tributaries.

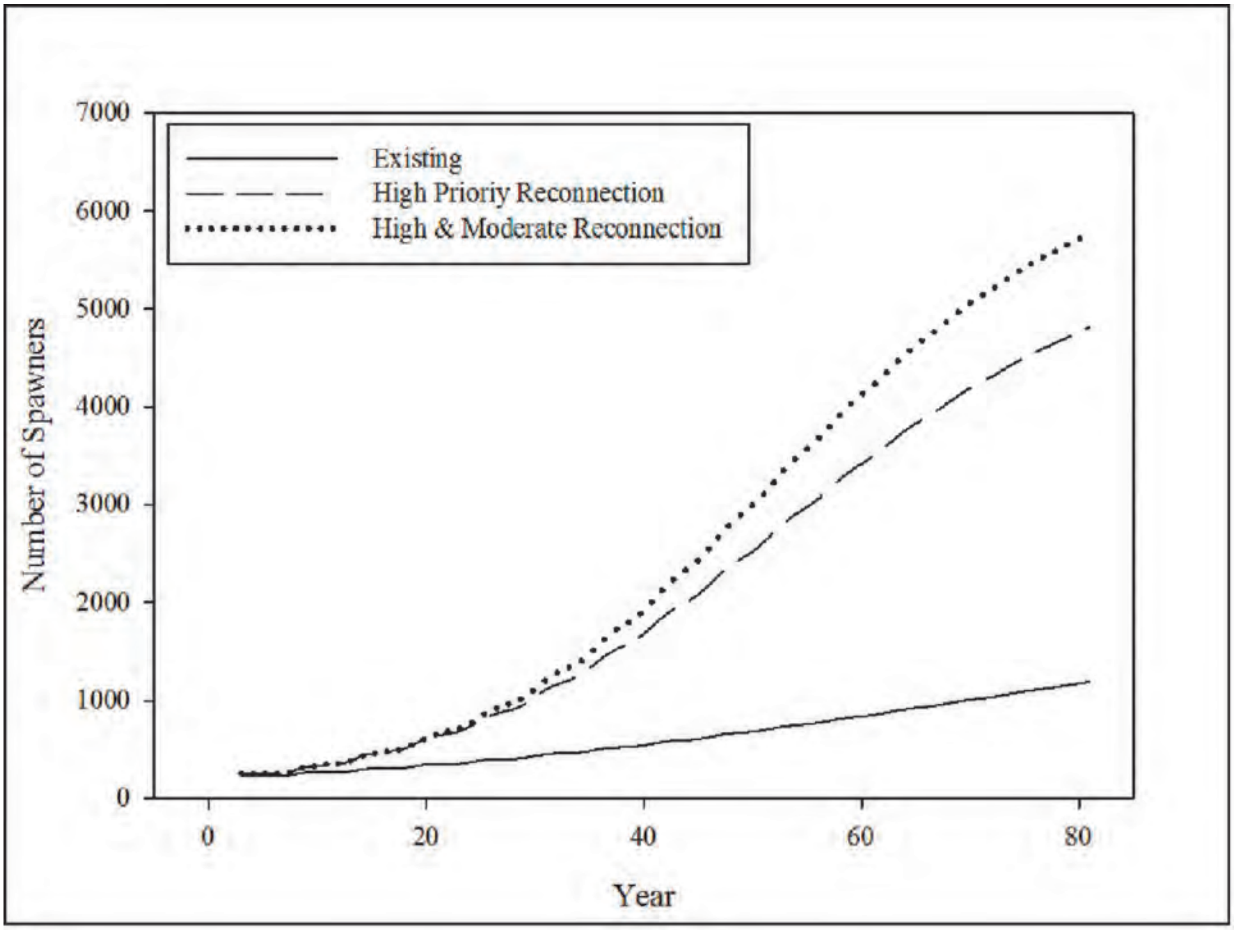

Figure 3-5. Number of spring/summer Chinook salmon adults returning to the Lemhi River given existing habitat, reconnection of high priority watersheds, and reconnection of high and moderate priority watersheds.

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 Subbasin 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, Figure 4.1) (Schapire 2002).

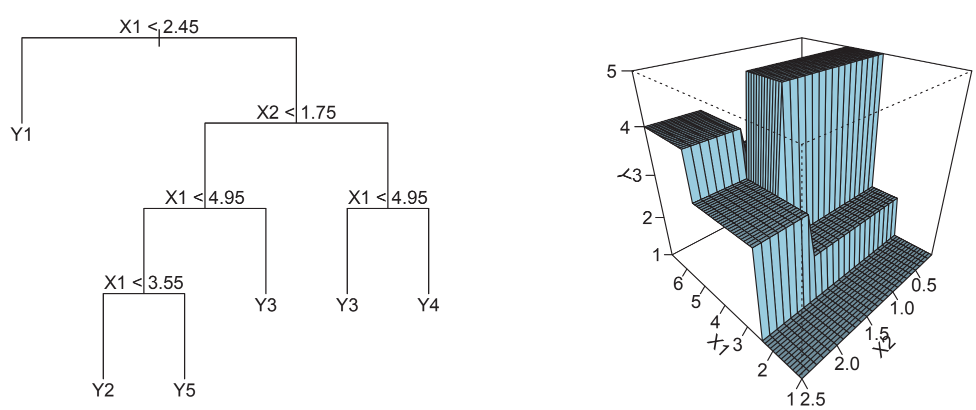

Figure 4.1. A single decision tree (left panel) based on a response variable, Y, and two predictor variables, X1 and X2. The panel on the right depicts the prediction surface.

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.

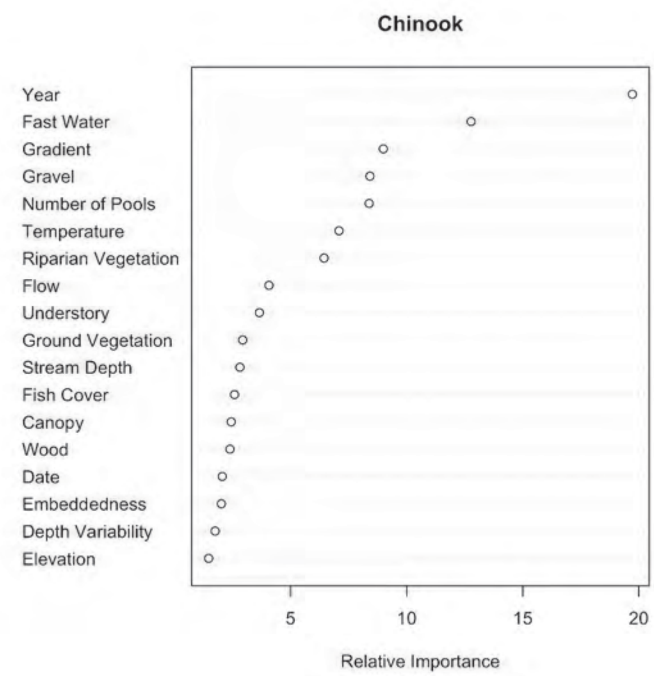

Figure 4.2. The relative importance of various habitat metrics in predicting the density of juvenile Chinook using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010 analyzed using a boosted regression tree approach.

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.

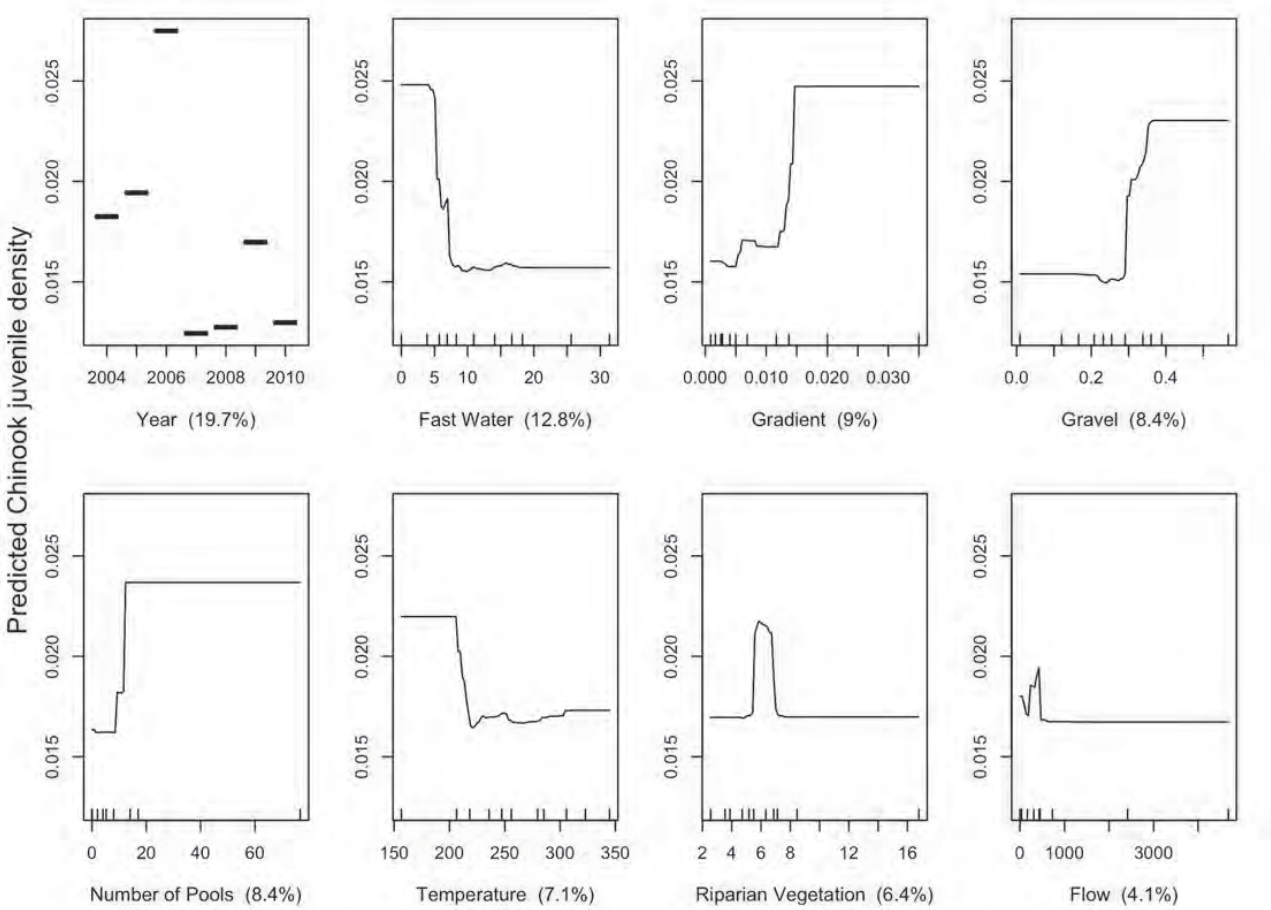

Figure 4.3. Partial dependence plots showing the marginal effect of the eight most important habitat metrics identified from a BRT on juvenile Chinook densities using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010. The y-axis is the predicted value of juvenile density. Along the bottom 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.

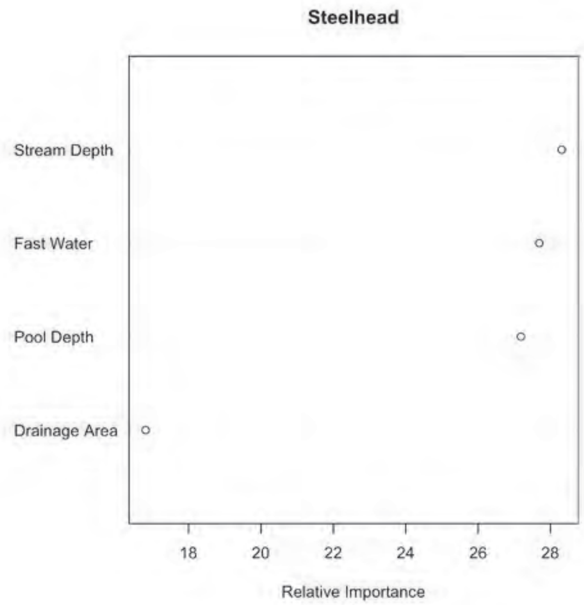

Figure 4.4. The relative importance of the four most important habitat metrics in predicting the density of juvenile steelhead using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010.

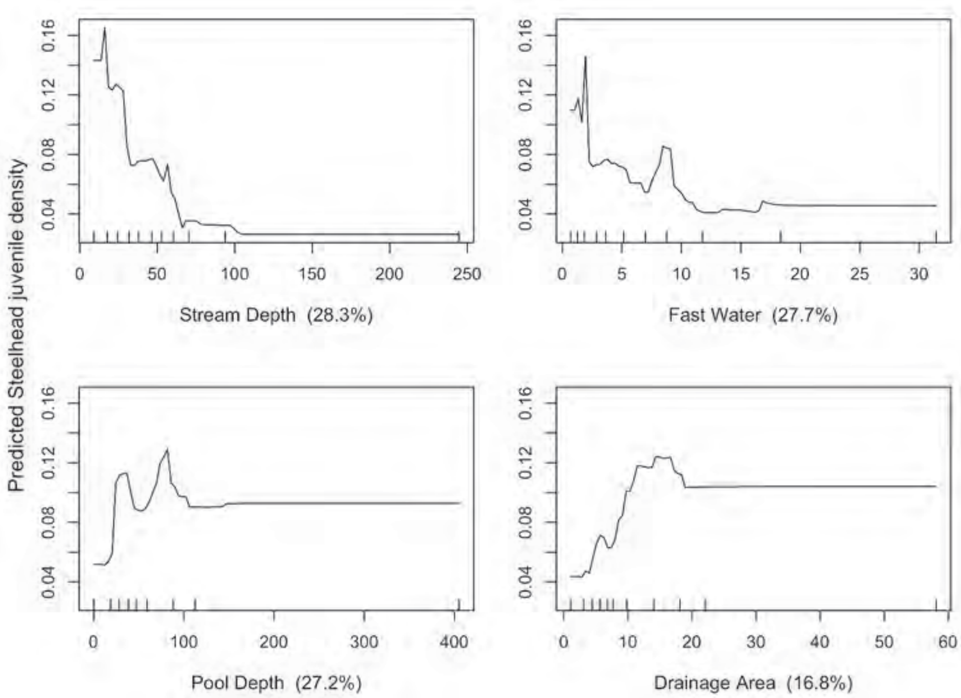

Figure 4.5. Partial dependence plots showing the marginal effect of the four most important habitat metrics identified from a BRT on juvenile steelhead densities using fish density data and habitat data collected by ISEMP in the Wenatchee River subbasin 2004-2010. The y-axis is the predicted value of juvenile density. Along the bottom of each plot, the tick marks show the deciles of the data for that habitat metric.

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.

<u>Habitat monitoring</u>

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 electrofishing equipment to herd fish a seine or multiple dip nets. O. mykiss that were greater than or equal to 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 5.1). A statistically significant response was also found in stream shading at all of the exclosed sites that were surveyed (Figure 5.2).

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 5.3).

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 5.4).

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 5.5). 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.

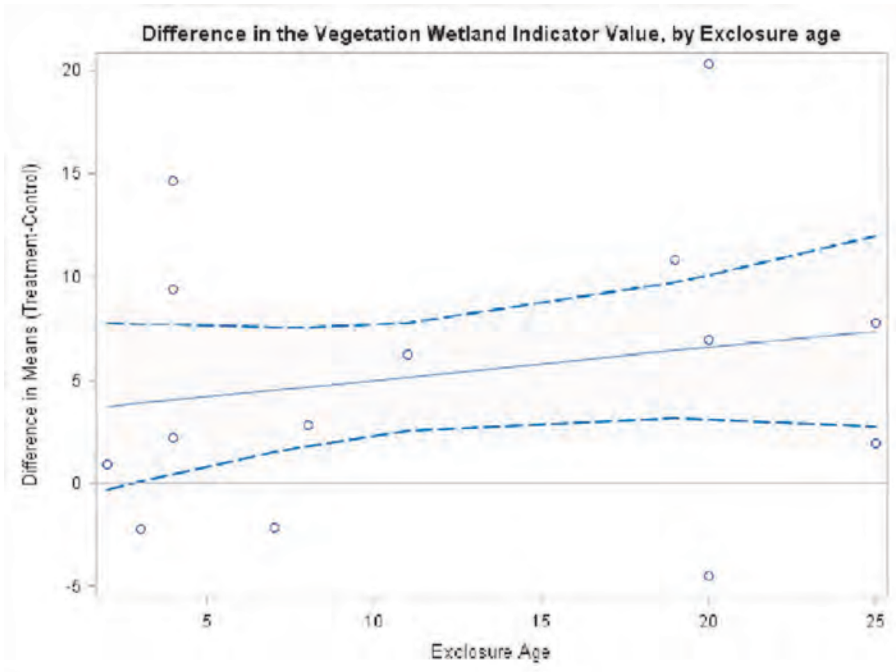

Figure 5.1. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, in wetland indicator values for the greenline plant communities. Statistically different values observed at exclosure sites > 6 years old.

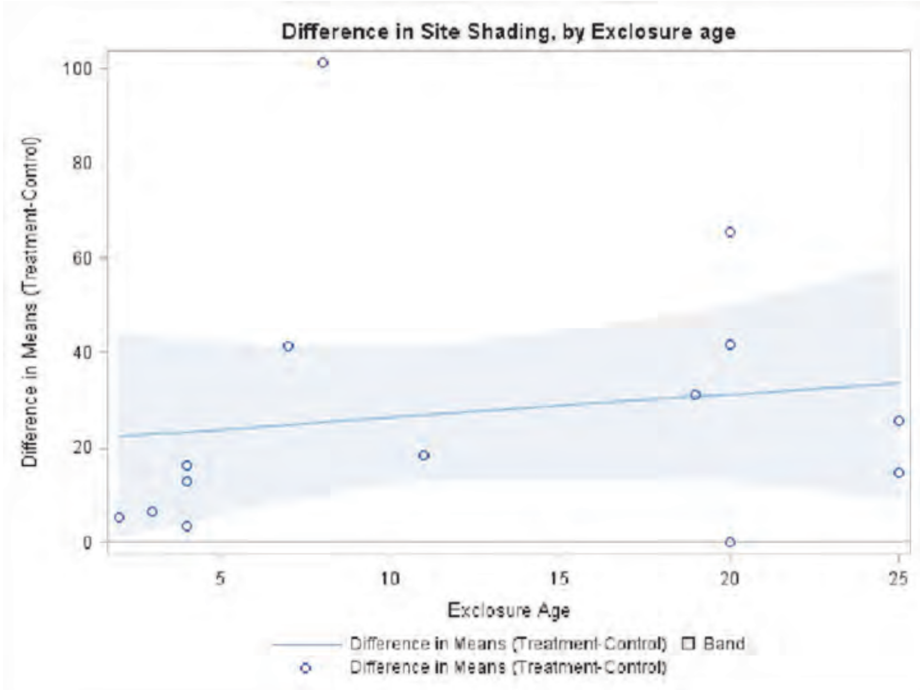

Figure 5.2. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, in shading, as measured using the Solar Pathfinder.

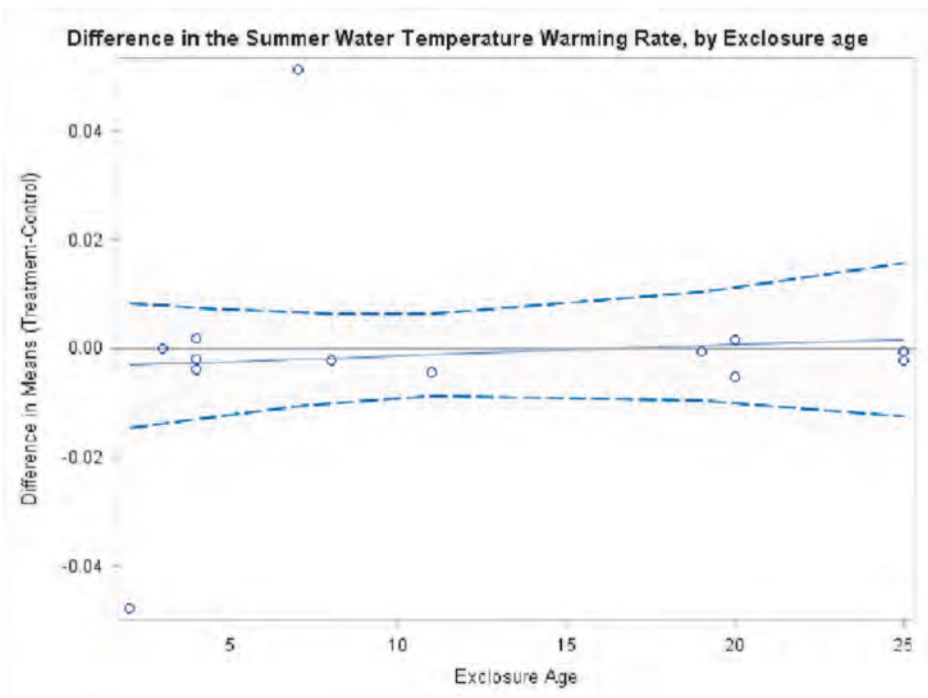

Figure 5.3. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, in rate of temperature change.

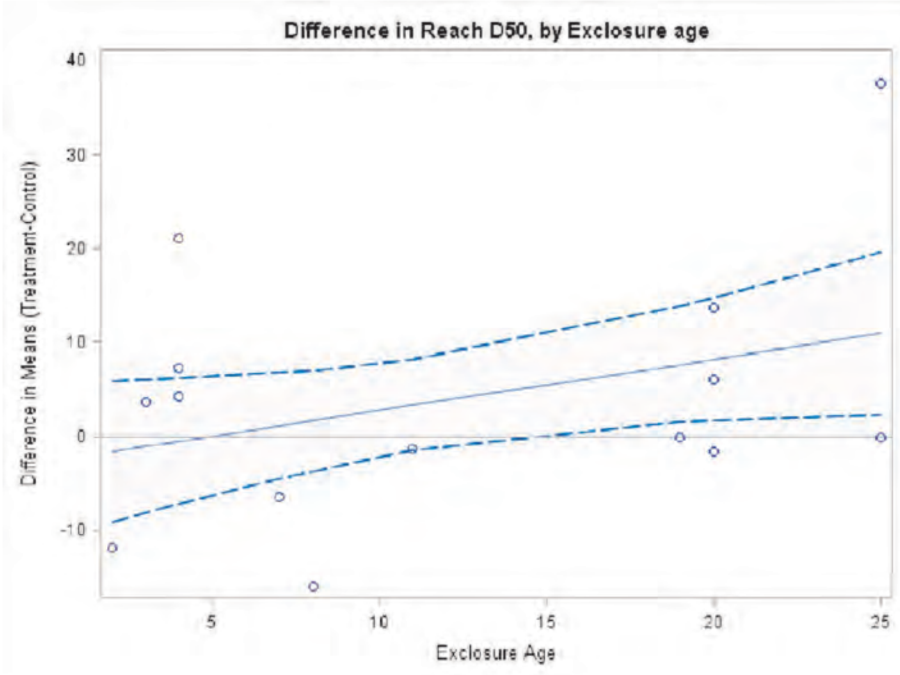

Figure 5.4. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, of the median grain size (D50).

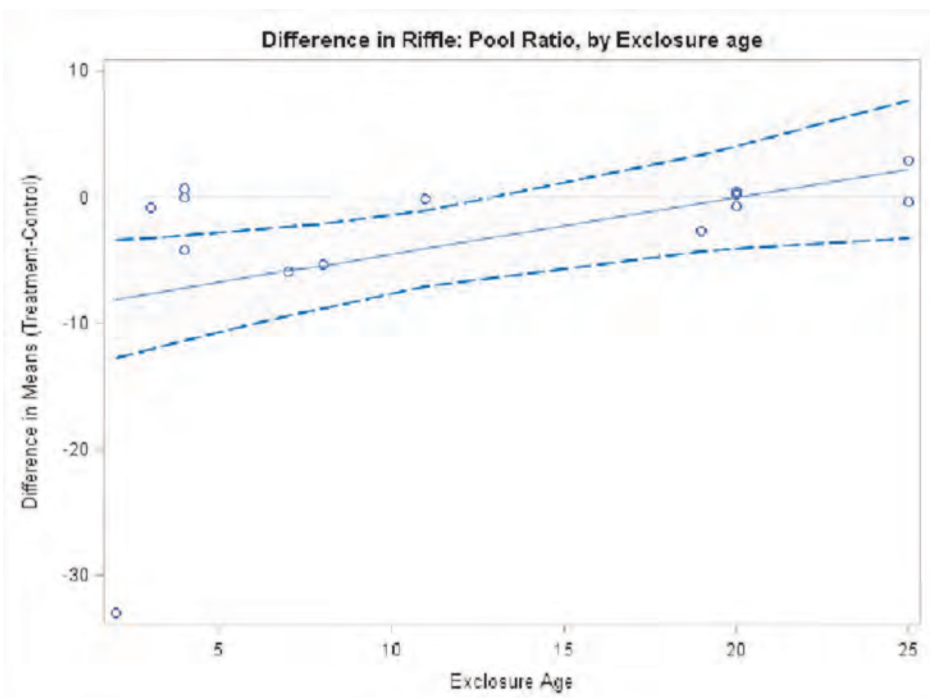

Figure 5.5. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, in the riffle: pool ratio to describe the difference in habitat arrangement.

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 5.6).

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 5.7).

We were unable to detect a treatment effect on steelhead summer growth rates at any sites (Figure 5.8). Since summer growth rates were not statistically significant it is no surprise to find no statistically significant difference in summer fish production (Figure 5.9).

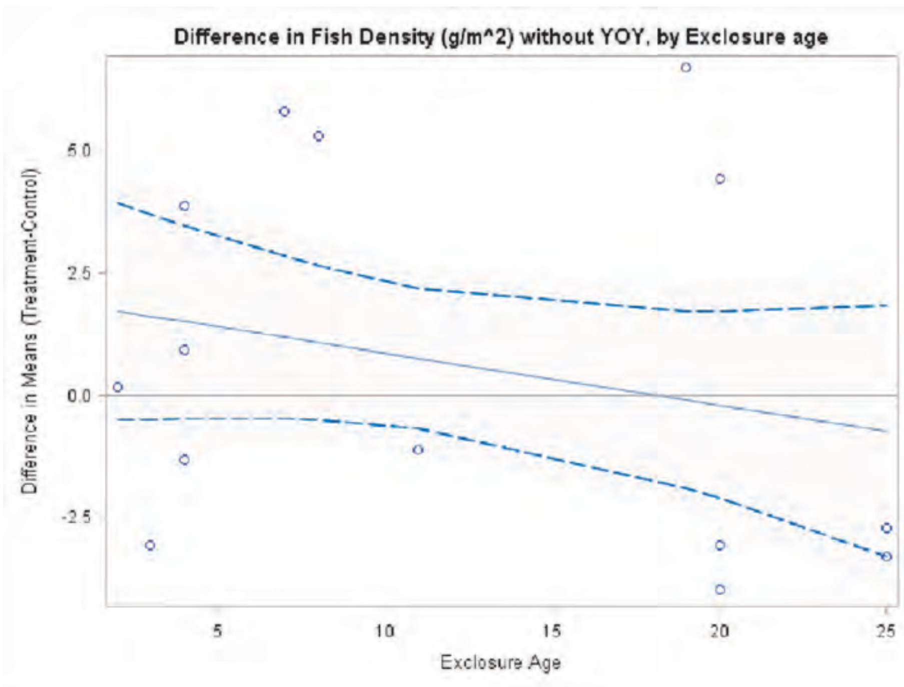

Figure 5.6. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, in fish biomass (g/m2) excluding age 0 steelhead.

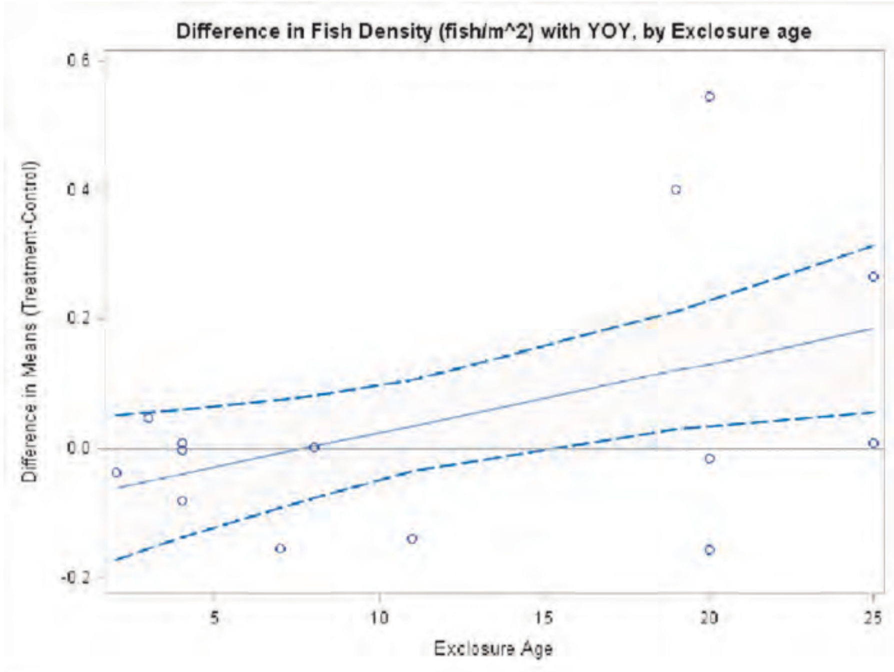

Figure 5.7. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, of fish density (fish/m2), including age 0 steelhead.

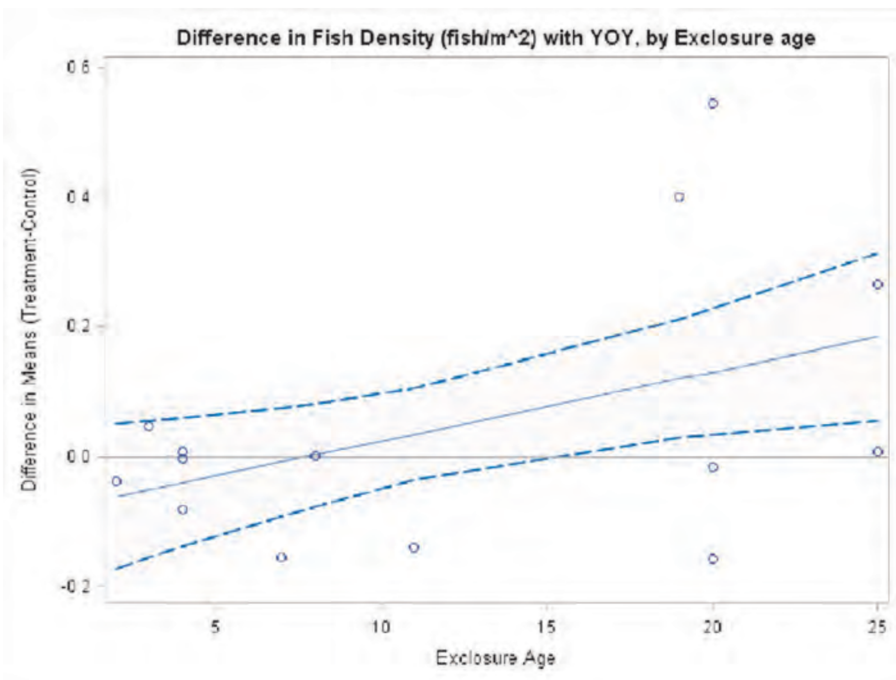

Figure 5.8. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, in steelhead summer growth rates, excluding age 0 steelhead.

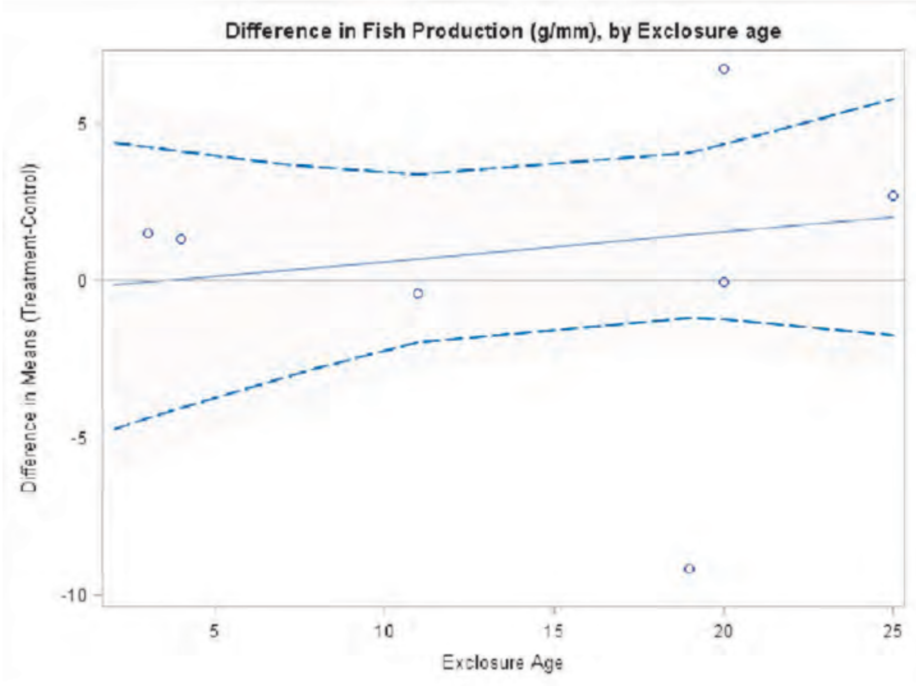

Figure 5.9. Difference between exclosure and control reaches (treatment mean - control mean; with 95% CIs), across different ages of exclosures, in fish production, excluding age 0 steelhead.

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). 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 subwatersheds in Asotin Creek. Treatments will be implemented in a staircase design after 4 years of pre-treatment monitoring (Figure 6.1). 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 6.1). 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.

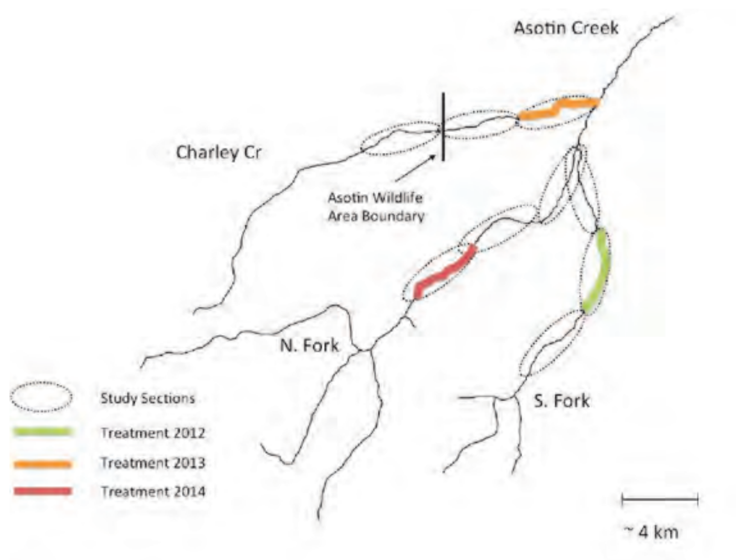


Figure 6.1. Experimental design and restoration schedule for the Asotin Creek IMW.

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 6.2). 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 6.2). 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.

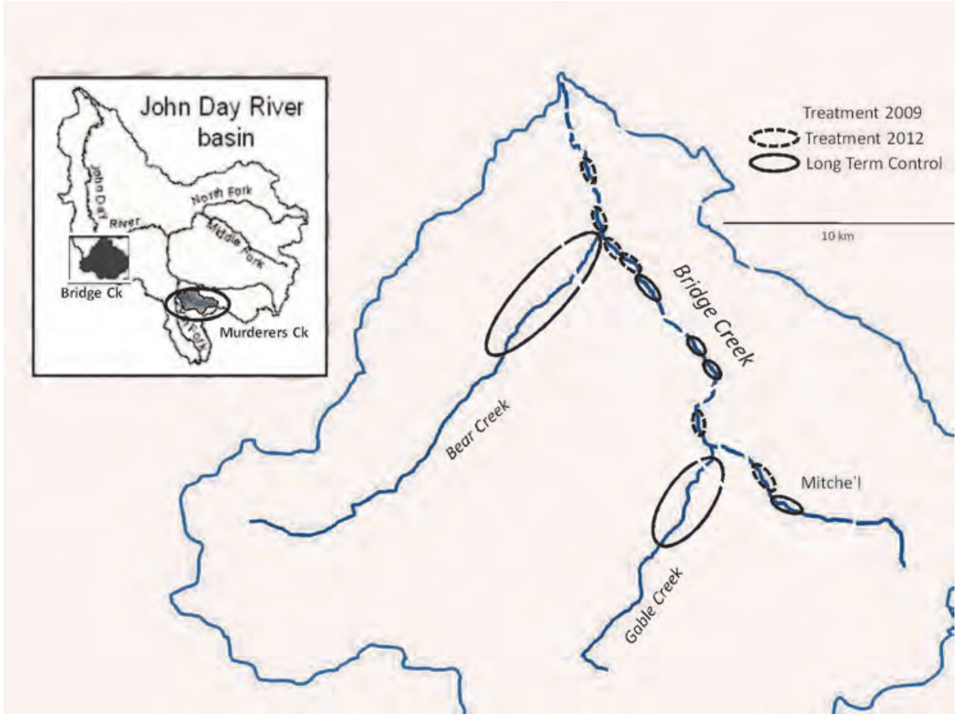


Figure 6.2. The Bridge Creek IMW experimental design. White, black-dashed, black-solid oval represent restoration units, sub-watersheds, watersheds that will be treated in 2009, 2013, or act as long term controls, respectively. Gable Creek and Bear Creek will act as long term sub-watershed controls and 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 6.3) 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 6.4 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.

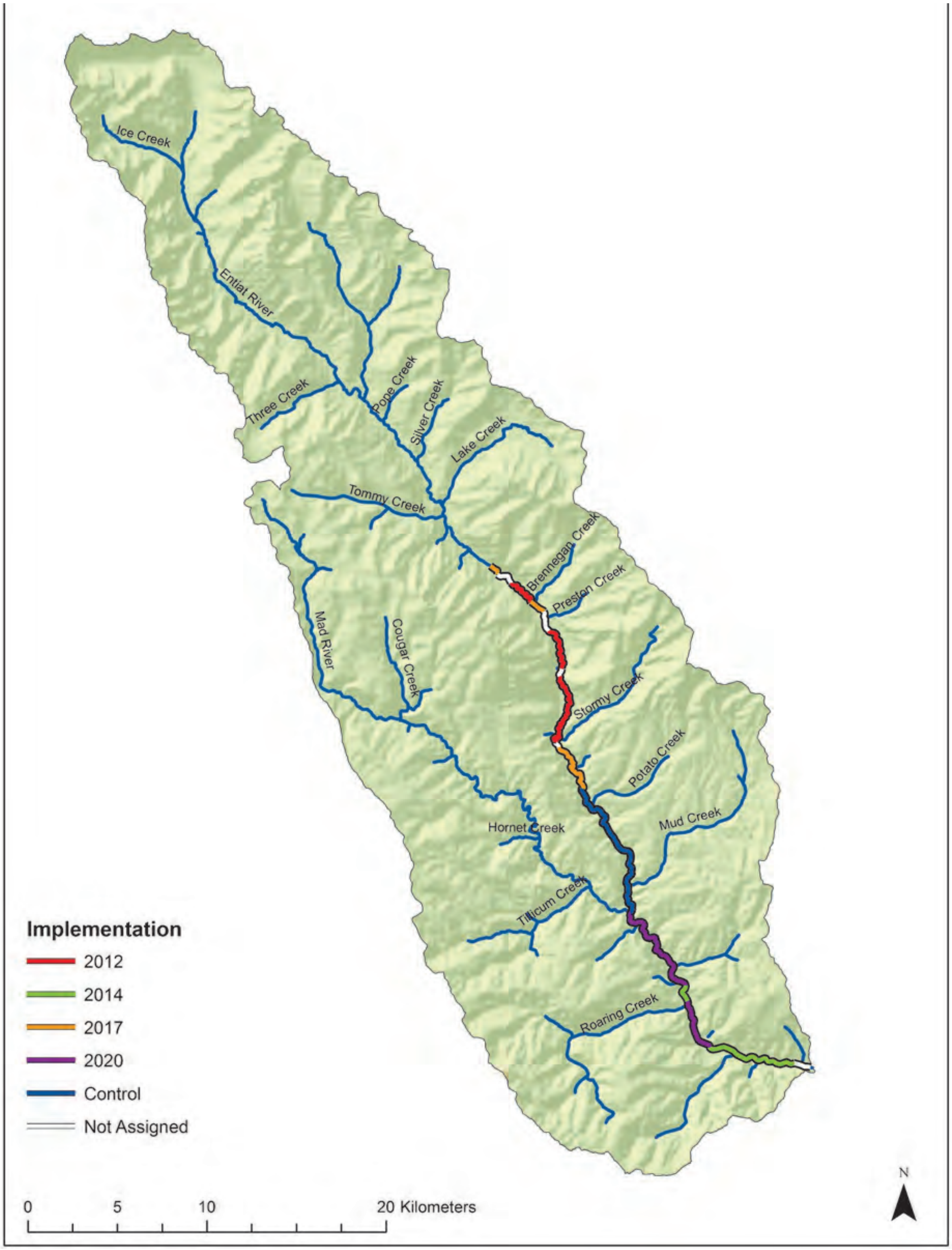
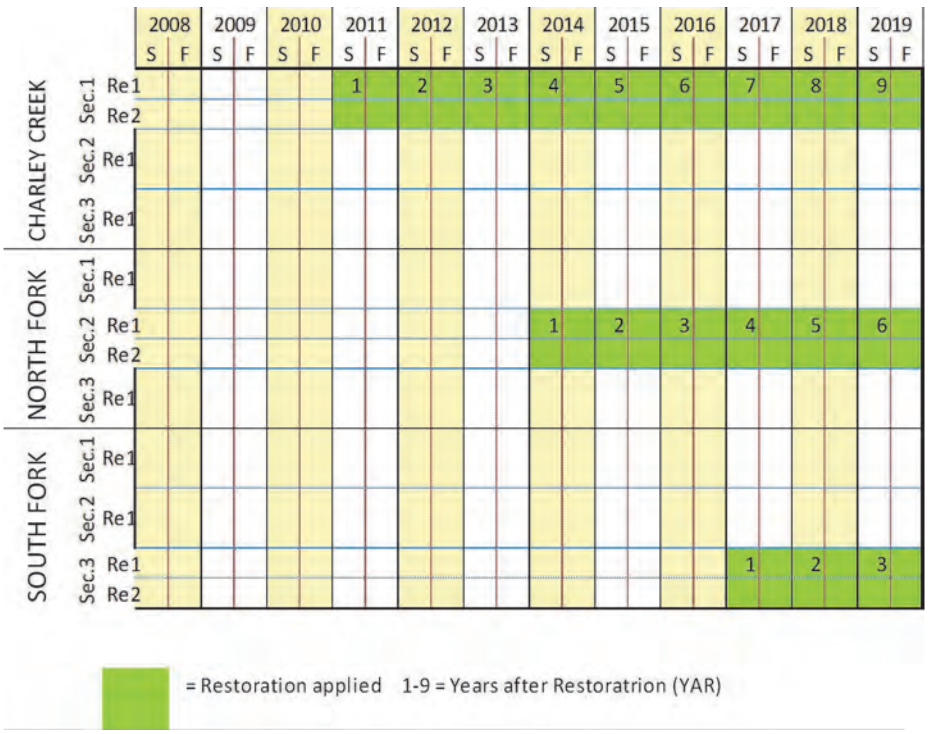


Figure 6.3. The Entiat River IMW experimental design. Treatments are stratified by valley types. Numbered letters represent reaches. Red reaches will be treated in 2012, green reaches in 2014, orange reach in 2017, and purple reach in 2020. The Mad River (large tributary coming in at the upstream section of 1F), will act as sub-watershed control and will not be treated.

a)



b)

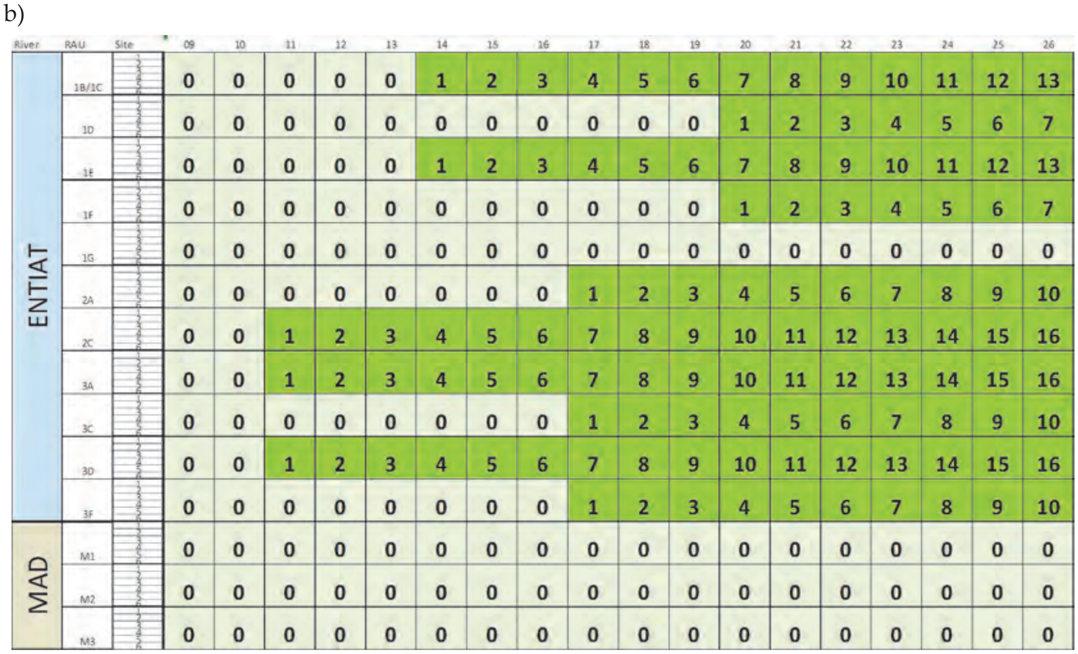


Figure 6.4. Schematic of the treatment and control reaches and location and timing of treatments used to develop statistical models for a) the Asotin Creek IMW and b) the Entiat River IMW.

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 6.1 a and b).

Table 6.1. 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 6.1. 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:

- "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.

- "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.

- "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.

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

- "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 (Figure 6.5). 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.

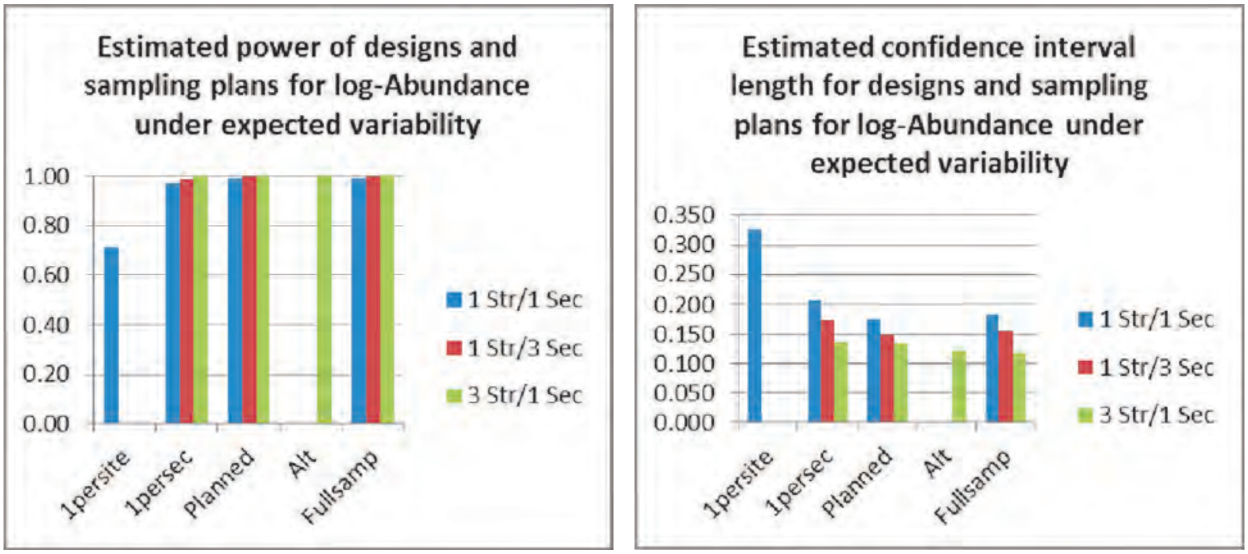


Figure 6.5. The estimated power of designs and sampling plans (left panel) and their associated estimated confidence intervals (right panel) for detecting a 25% change in juvenile steelhead abundance using Asotin Creek IMW historic data and ***best case*** estimates of variance.

Under the worst-case variability, greater differences among the methods begin to emerge (Figure 6.6). 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.

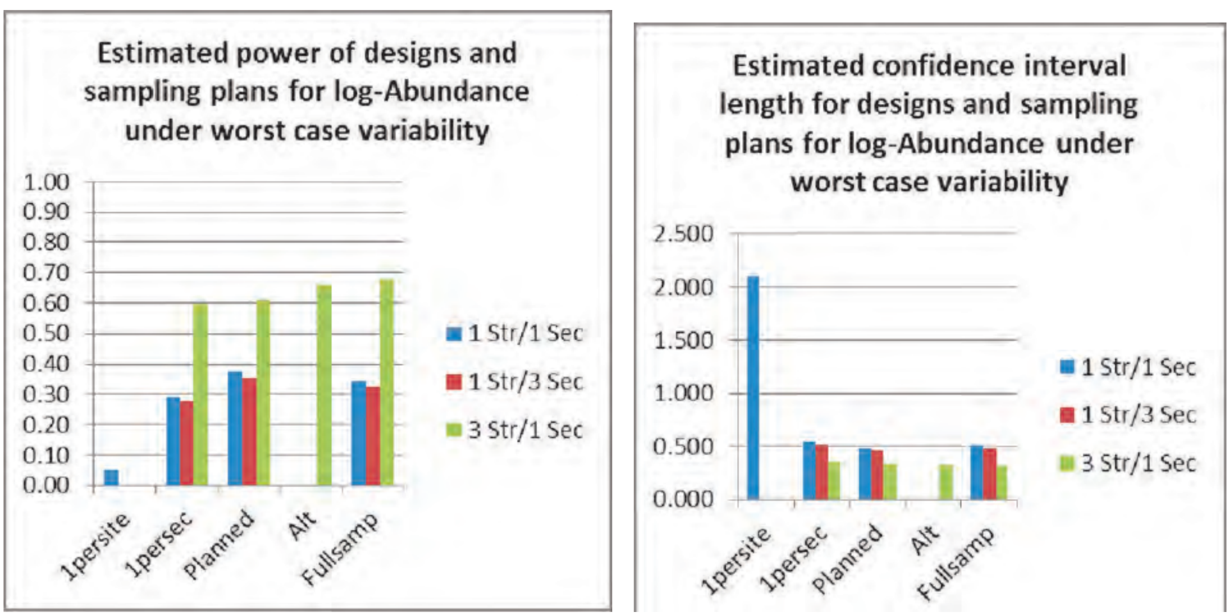


Figure 6.6. The estimated power of designs and sampling plans (left panel) and their associated estimated confidence intervals (right panel) for detecting a 25% change in juvenile steelhead abundance using Asotin Creek IMW historic data and ***worst case*** estimates of variance.

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 6.7).

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.

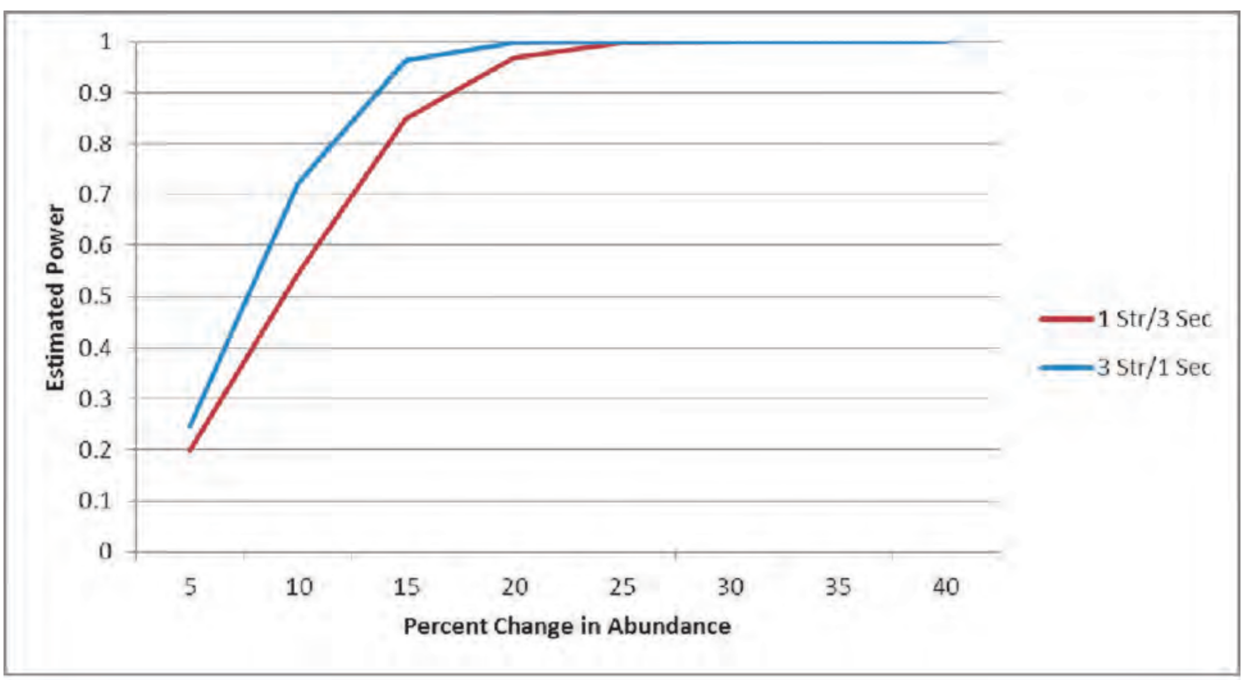


Figure 6.7. Power for varying alternatives for log-abundance under ***best case*** scenario variability.

In the worst case variability, the horizontal difference can be as much as 15% (Figure 6.8). 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.

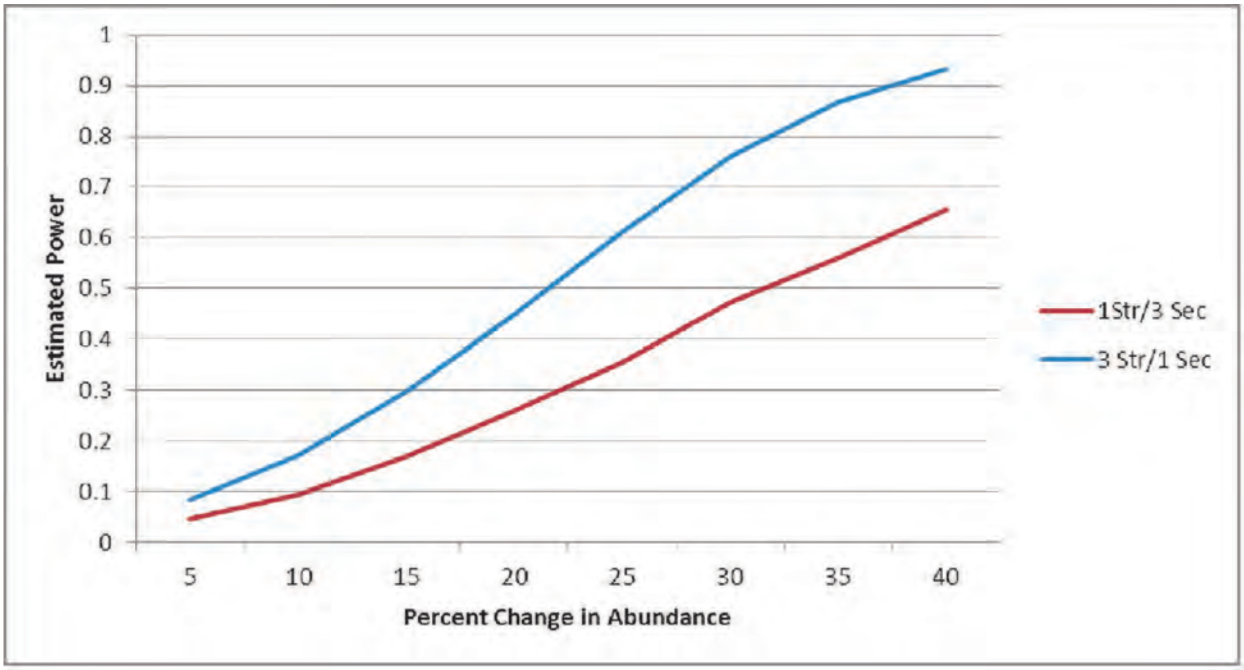


Figure 6.8. Power for varying alternatives for log-abundance under ***worst-case*** variability.

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 6.2), 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, 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. These survey methods have been implemented throughout 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 (Table 7.1). 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.

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.

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 7.1, Figure 7.1). 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 7.1. 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

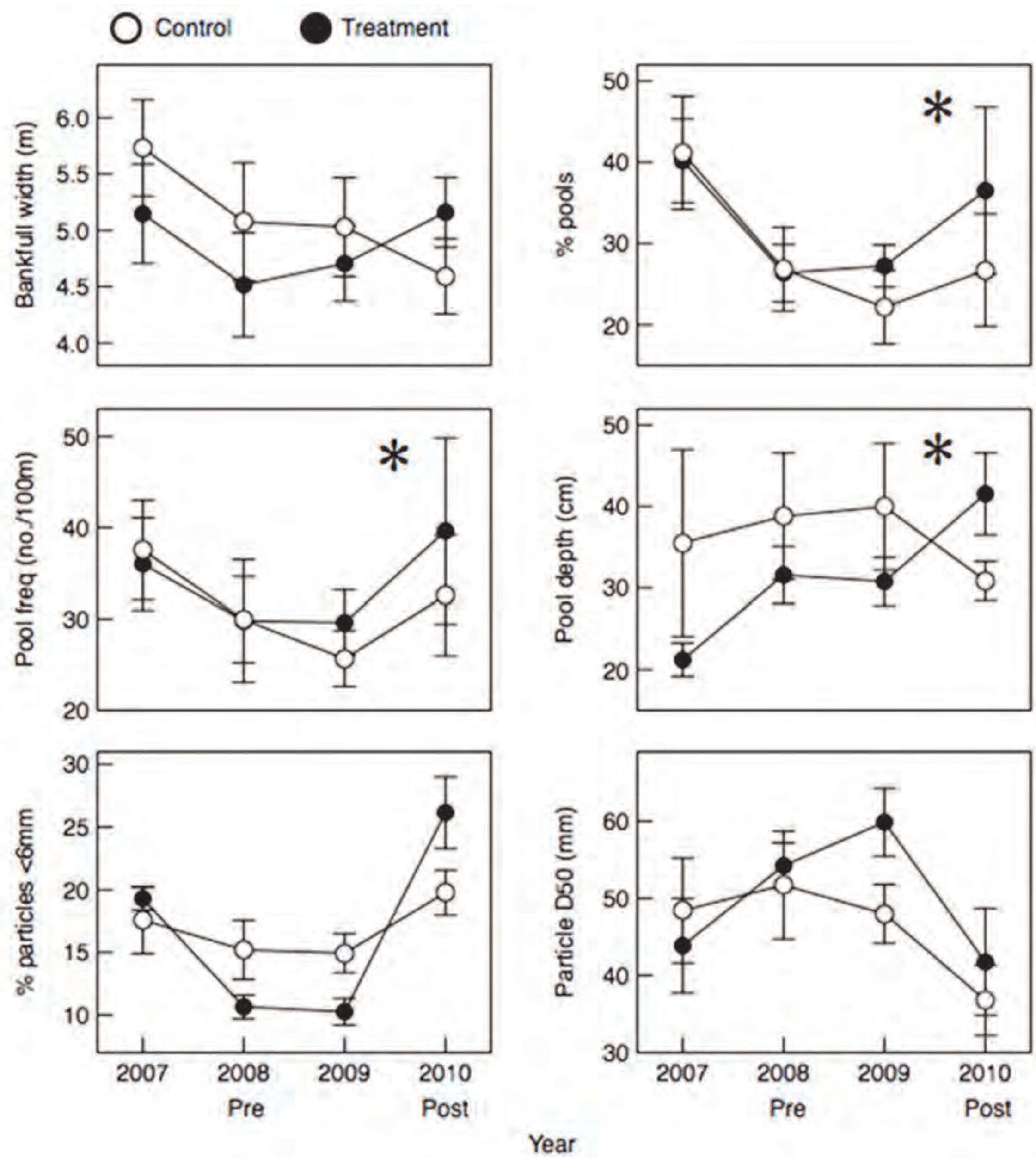


Figure 7.1.Average of channel attributes for treatment and control reaches across years, * indicates significant differences for pre and post-restoration.

Aerial photography was provided by AggieAir Flying Circus of the Utah Water Research Laboratory using unmanned aerial vehicles (UAV; Figure 7.2). 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 7.3).

A variety of geospatial outputs are created from the raw topographic data points acquired via RTK GPS, TS, and TLS (Figure 7.4). 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 7.4— 7.10, we display products from the one of treatment reaches.

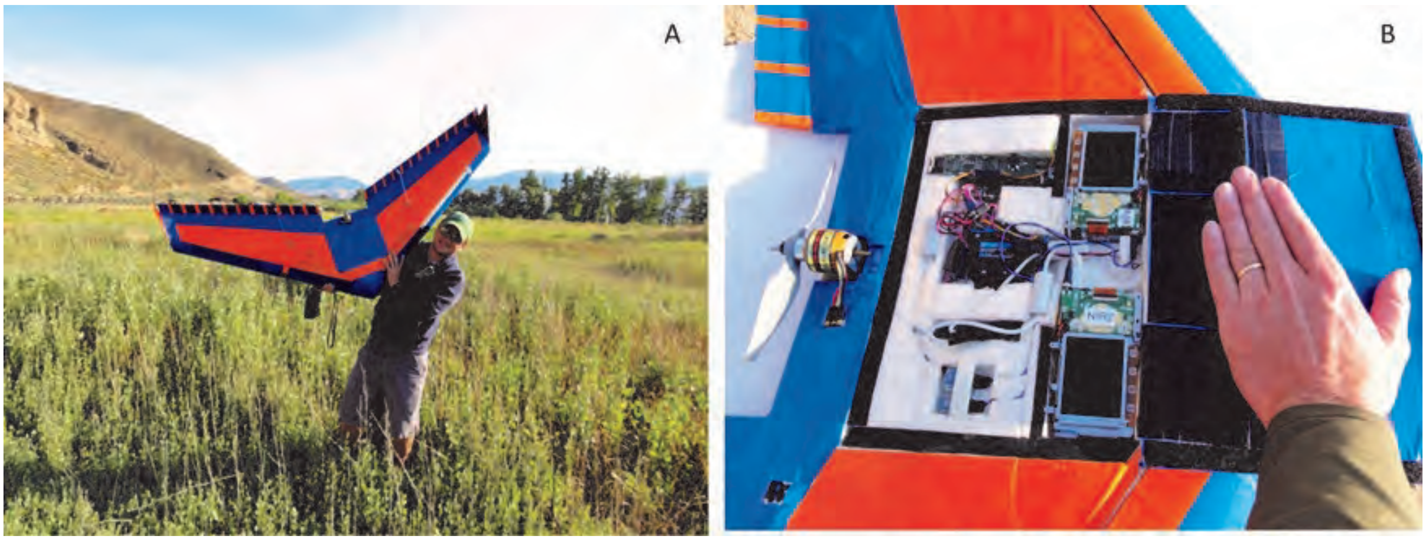


Figure 7.2. UAV Drone (A) equipped with RGB and NIR digital cameras (B).

Figure 7.3. Example of high resolution of UAV imagery at the Painted Hills National Monument visitor center.

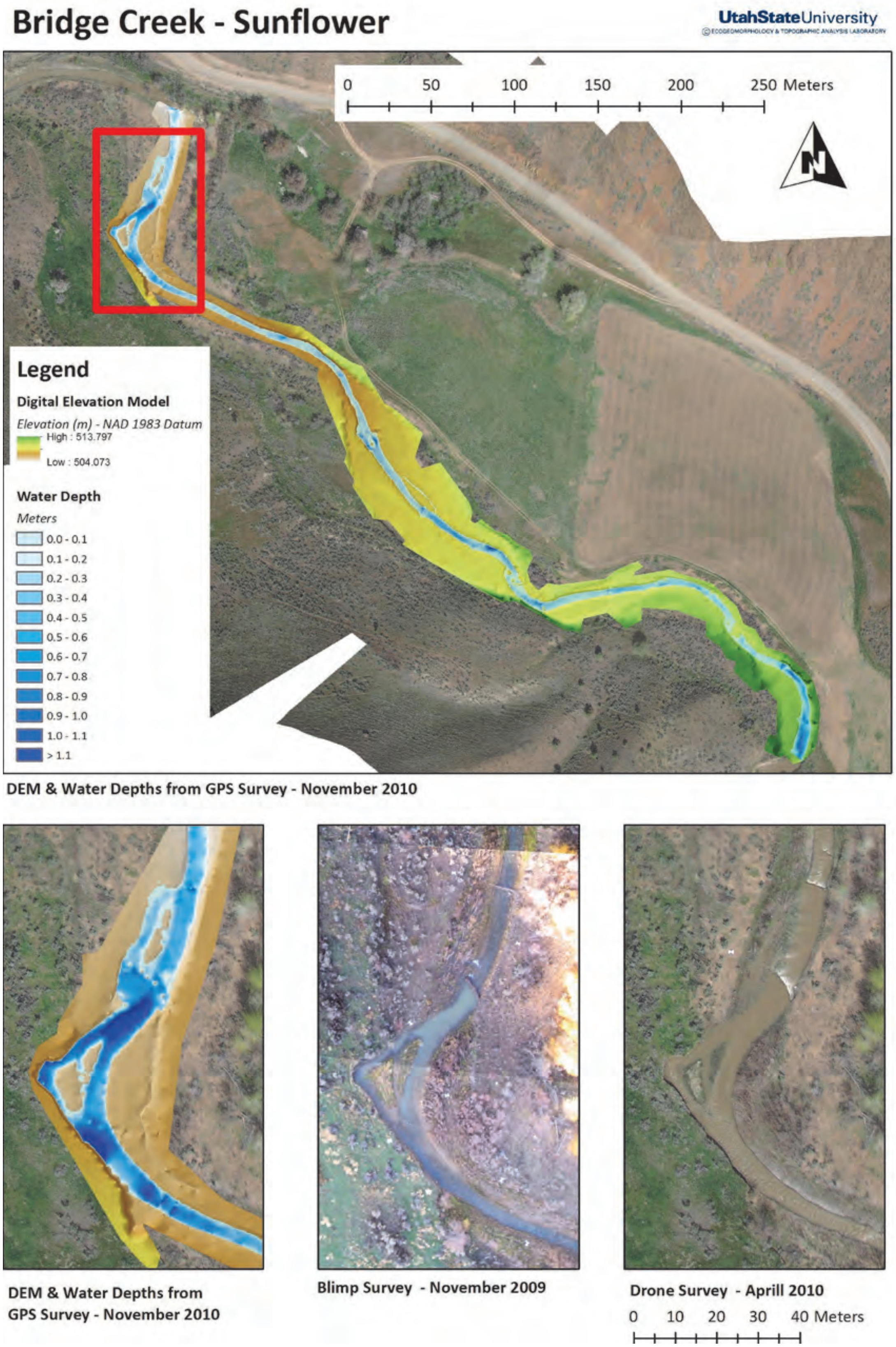


Figure 7.4. Examples of derived products from topographic and aerial surveys.

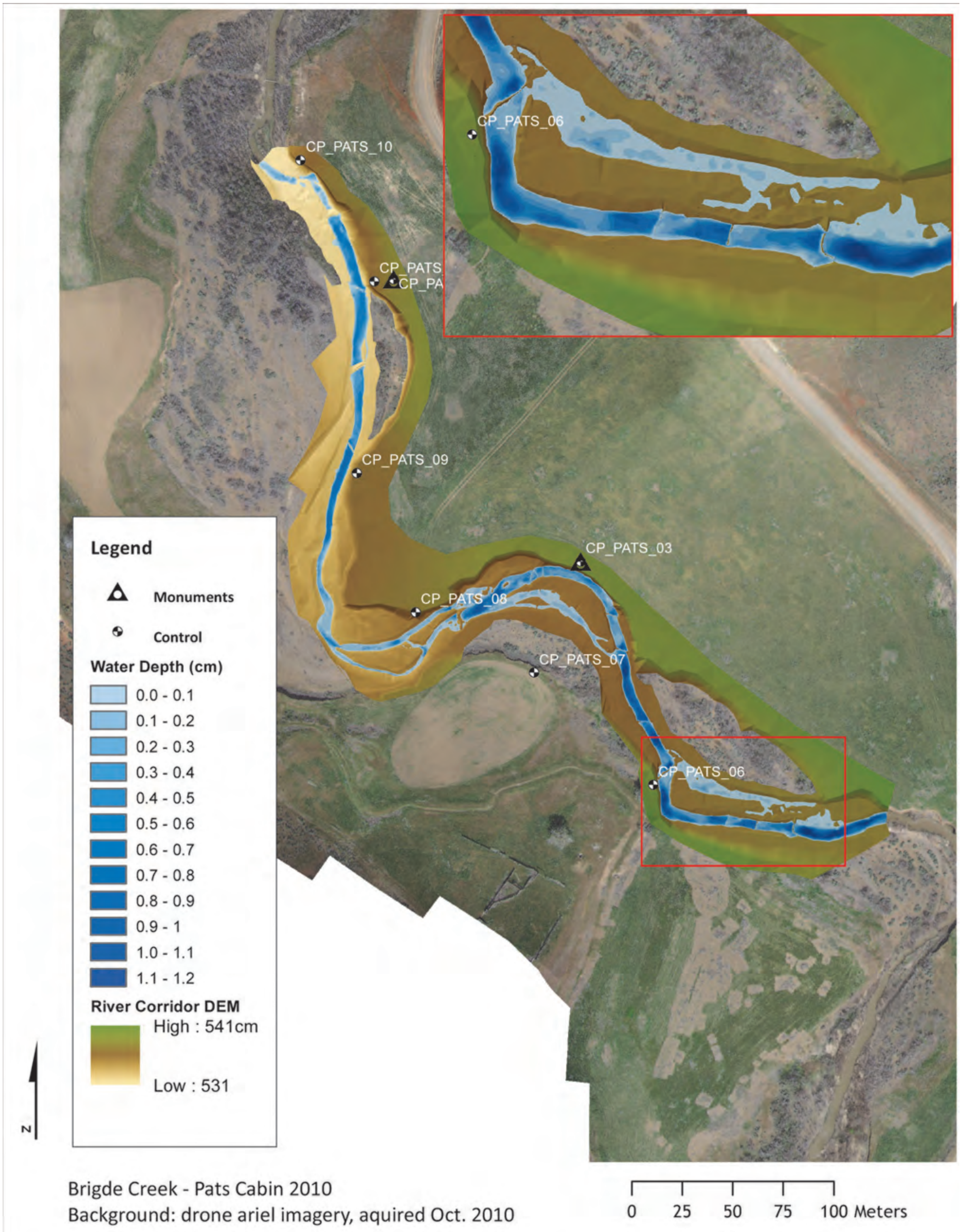


Figure 7.5. Pats Cabin treatment reach showing the digital elevation model, water depth maps, derived from topographic data. Also shown is the location of BDSS.

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 7.6). From a geomorphic perspective, these represent the 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).

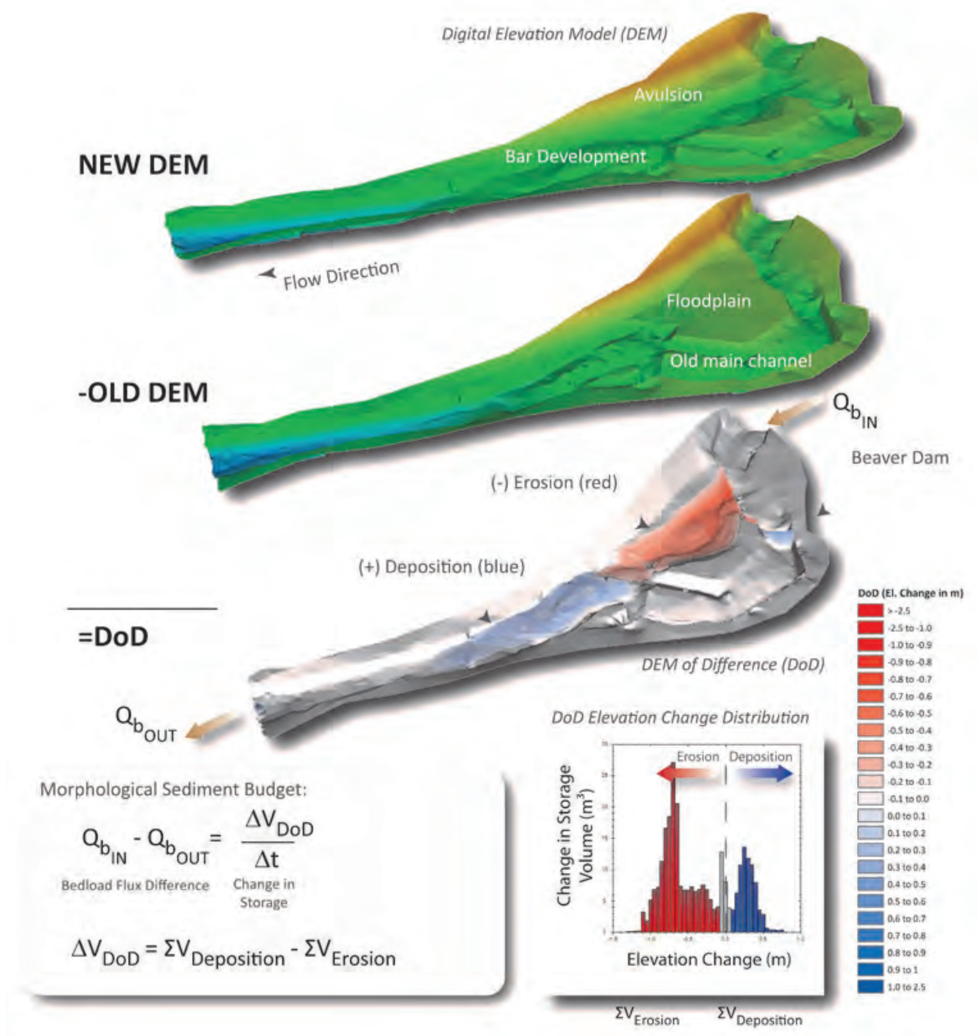
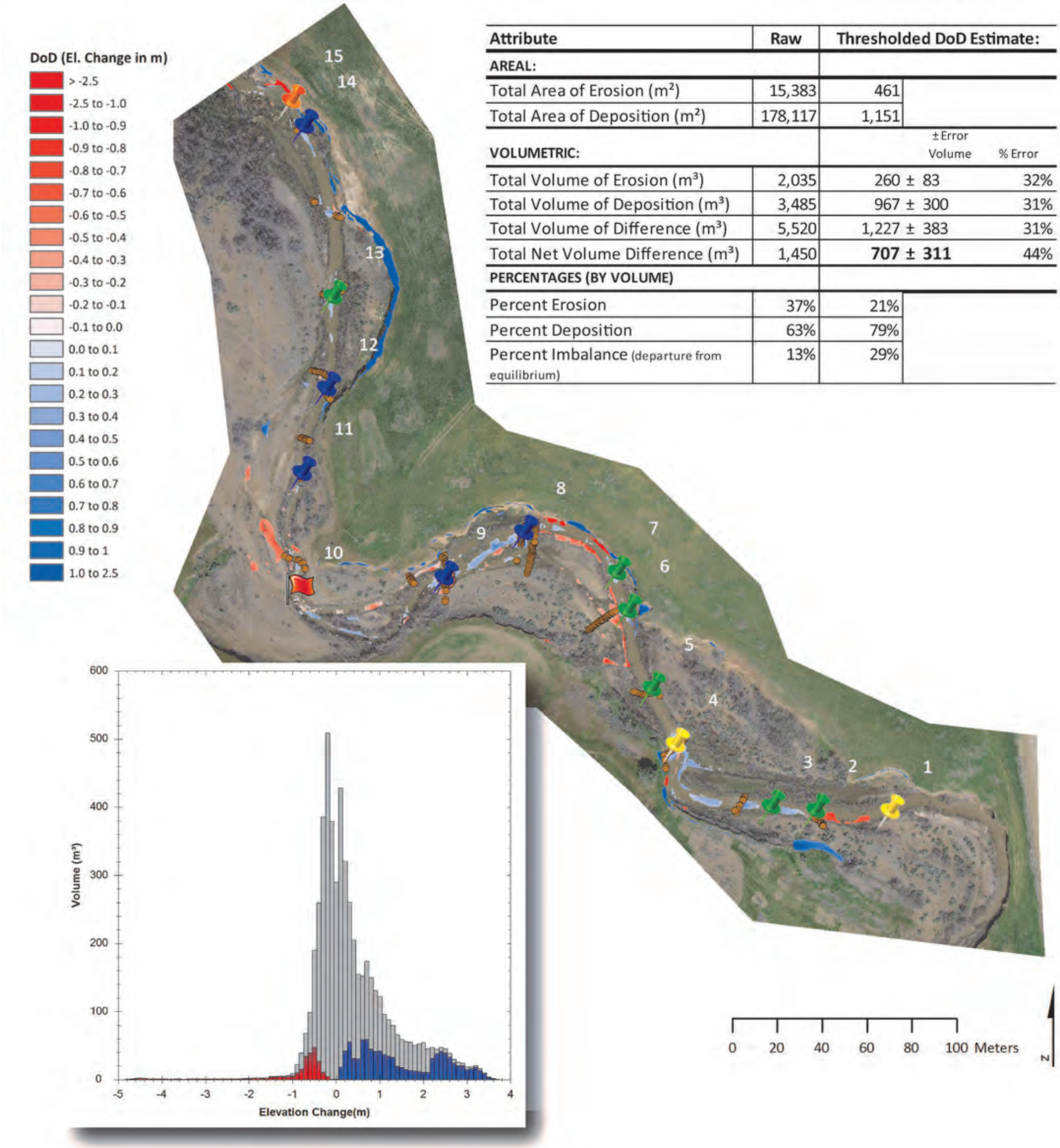


Figure 7.6. Concept of DEM differencing. For an X, Y pixel the old elevation (Z) is subtracted from the new elevation. A negative value (represented in red) indicates erosion, where a positive value (new elevation is higher than old; represented in blue) indicates deposition, and neutral change (represented 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 7.7). 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 7.7 insert.). DEMs of difference clearly capture the general pattern of deposition, scour, deposition seen at most BDSS (Figure 7.8).



Attribute	Raw	Thresholded DoD Estimate:	
AREAL:			
Total Area of Erosion (m²)	15,383	461	
Total Area of Deposition (m²)	178,117	1,151	
VOLUMETRIC:		±Error Volume	% Error
Total Volume of Erosion (m³)	2,035	260 ± 83	32%
Total Volume of Deposition (m³)	3,485	967 ± 300	31%
Total Volume of Difference (m³)	5,520	1,227 ± 383	31%
Total Net Volume Difference (m³)	1,450	**707 ± 311**	44%
PERCENTAGES (BY VOLUME)			
Percent Erosion	37%	21%	
Percent Deposition	63%	79%	
Percent Imbalance (departure from equilibrium)	13%	29%	

Figure 7.7. 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.

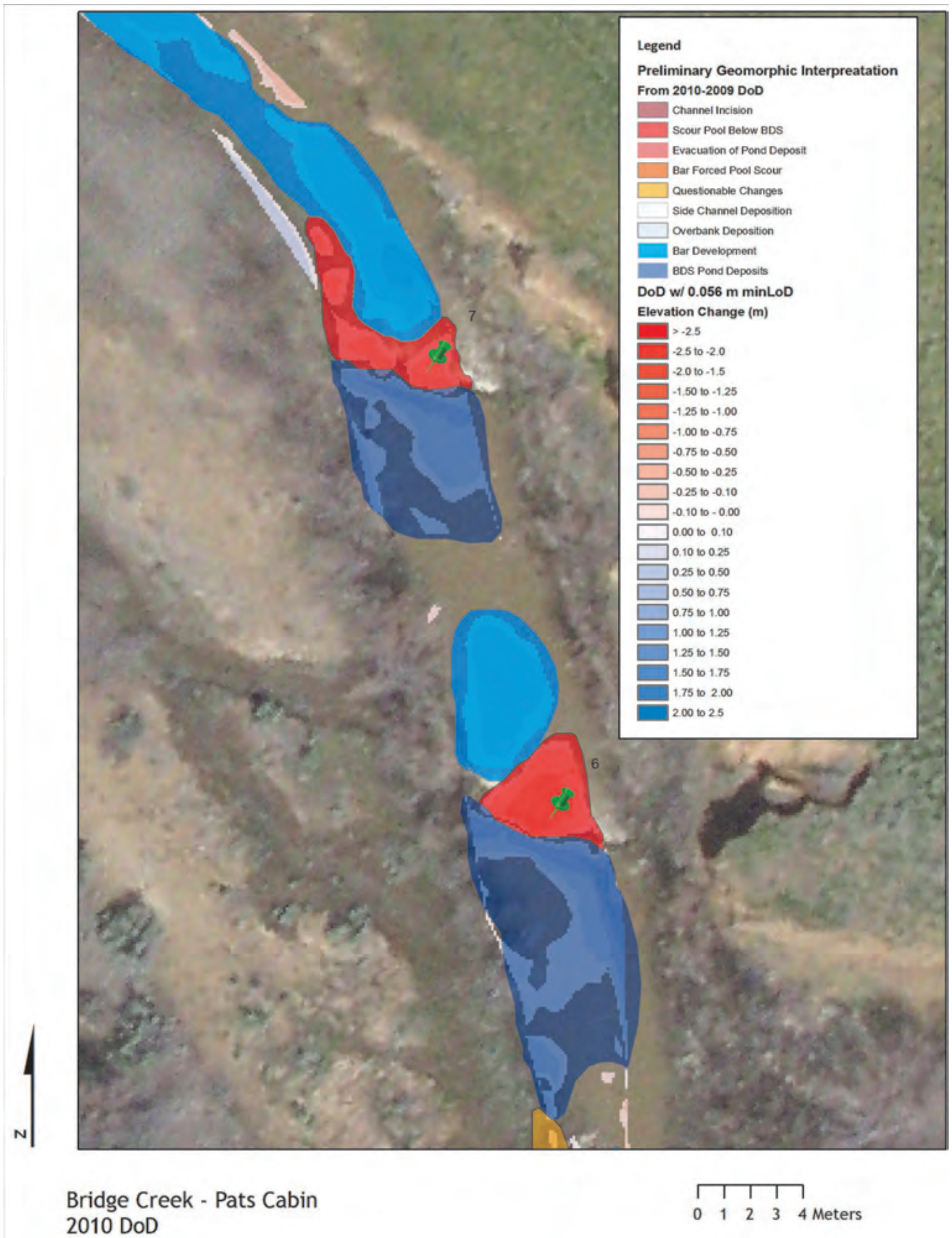


Figure 7.8. DEM of difference (post-restoration minus pre-restoration) from topographic surveys for a portion of treatment reach in Bridge Creek. Pushpins represent structure location. Blue color represents aggradation (deposition of sediments), and red represents erosion. General pattern was to have 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 7.8. 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 the changes. For the example shown in Figure 7.9, 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).

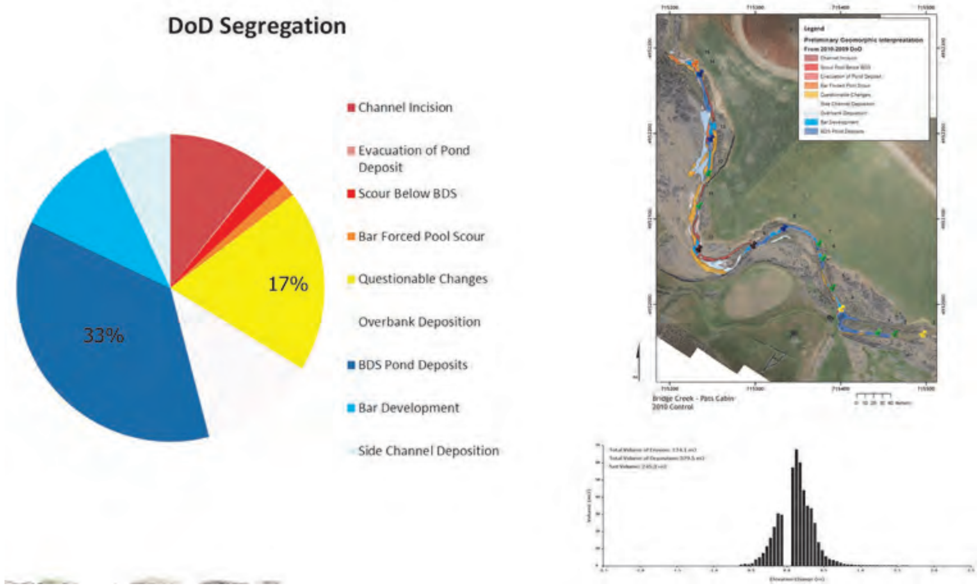


Figure 7.9. DEM of difference (post-restoration minus pre-restoration) from topographic surveys for a treatment reach in Bridge Creek. Pushpins represent structure location. Changes to the stream channels were identified and quantified.

Table 7.2. 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 7.2 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 7.10 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. 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.

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. 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.

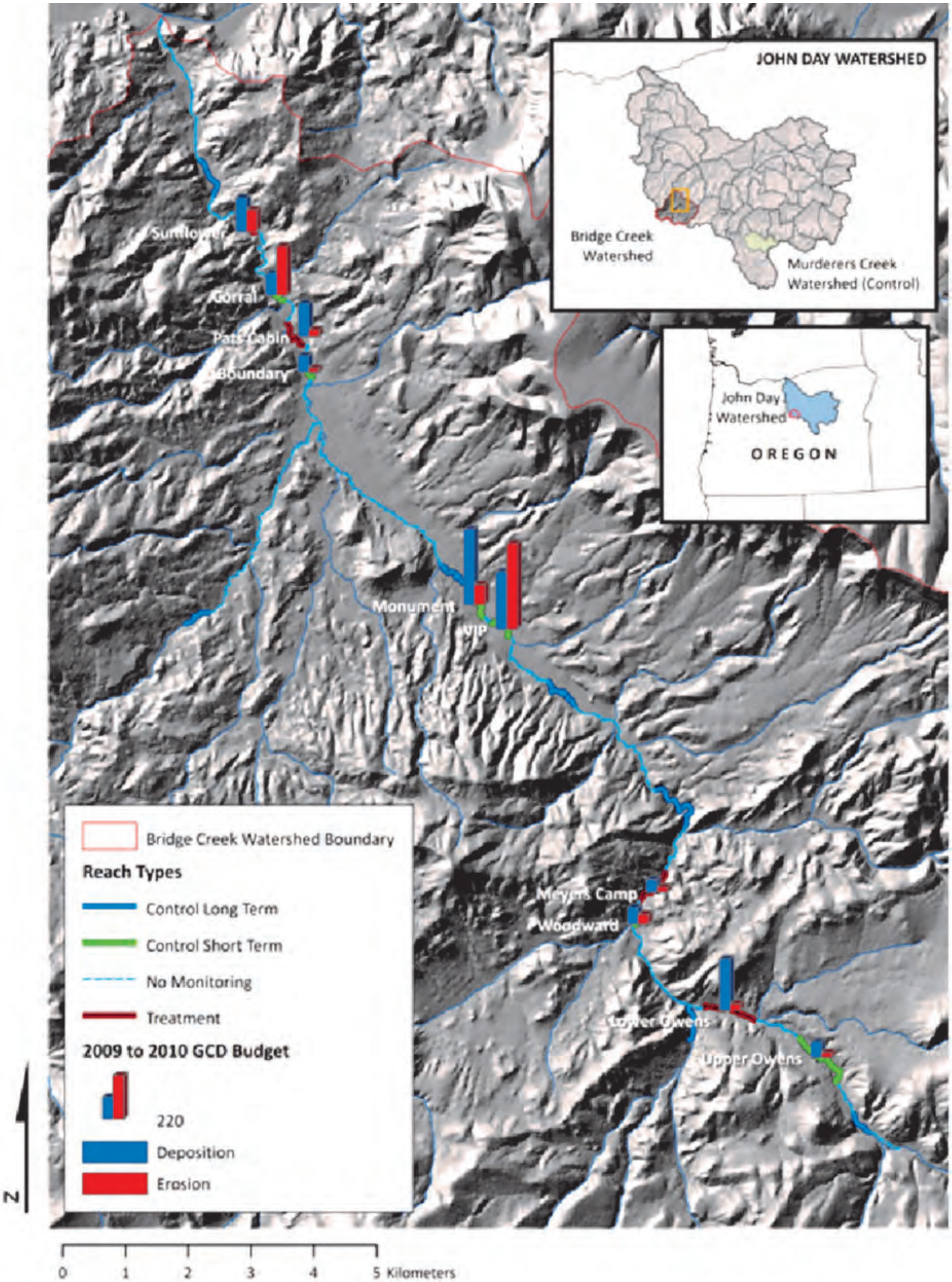


Figure 7.10. Graphical representation of relative magnitudes of erosion and deposition experienced at each of the 10 monitoring reaches between November 2009 and November 2010. Values based on geomorphic change detection using a fuzzy inference system and Threshold to show only changes with a 90% or greater probability of being real

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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 8.1). In fact, this model was used to estimate the impacts of the Middle Fork IMW on temperature (Crown 2010; Figure 8.2).

Under the Clean Water Act, biologically based critical thresholds have been established (Figure 8.1). 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 8.2. 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 8.1). 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

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 8.1; 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 8.1; Figure 8.1).

Table 8.1: 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. Model results were produced every 0.5 min and 200m. The extent was 112.95.
Restored Vegetation	System Potential Vegetation and increased hyporheic exchange in meadow reaches.
Restored Flow	No points of diversion or ditch inputs and tributary flows adjusted to OWRD's estimates of natural flow.
Restored Morphology	Bankfull widths reduced by 10-50% while cross-sectional area preserved.
NTP	Natural Thermal Potential: combining the inputs of system potential vegetation , natural stream flow, and range of reduced bankfull width estimates. Hyporheic flow restored in meadow reaches . No other temperature adjustments were made to tributary inputs.
Pre-restoration	Scenarios estimating instream temperatures before major current restoration projects started.
Post-restoration	Scenario estimating instream temperatures when major current restoration projects near natural thermal potential.

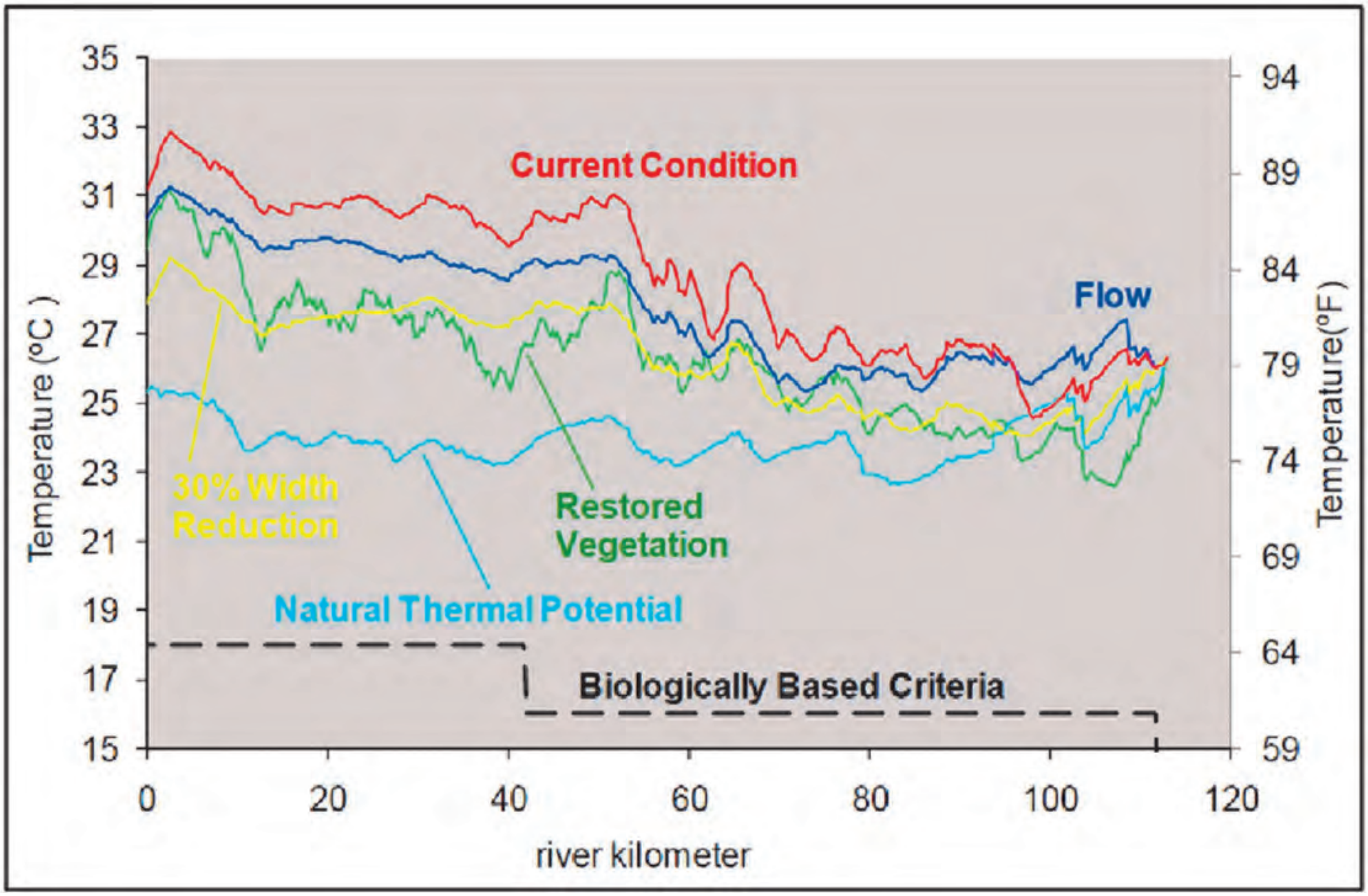

Figure 8.1. Longitudinal profile of 7 day daily average maximum stream temperatures for the Middle Fork John Day in summer 2002 as modeled by ODEQ (2010) using HeatSource, across different scenarios. Scenarios include current conditions (2002; red line), restored flows (no withdraws and reconnected tributaries; 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:

consumption = growth + (respiration + 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.

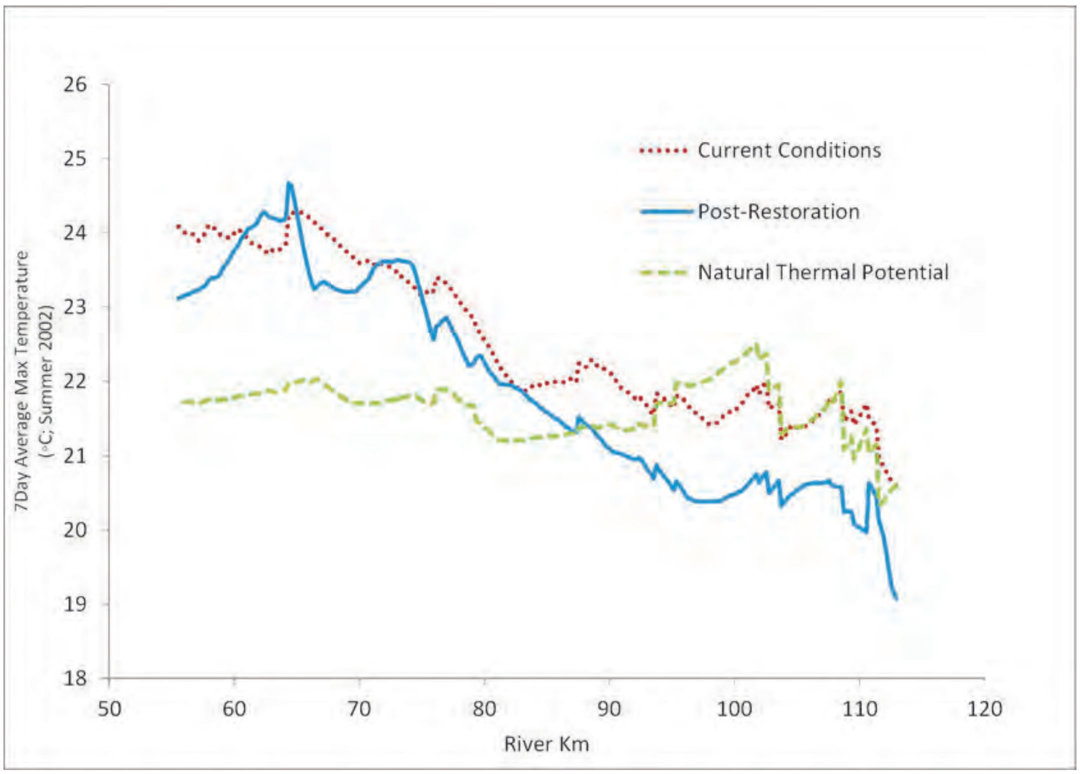

Figure 8.2. Longitudinal profile of average maximum weekly stream temperatures for the Middle Fork John Day in summer 2002 as modeled by ODEQ (2010) using Heat Source, across different scenarios. Scenarios include current conditions (2002; red solid line), and post-restoration implemented in the Middle 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:

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 8.3). 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.

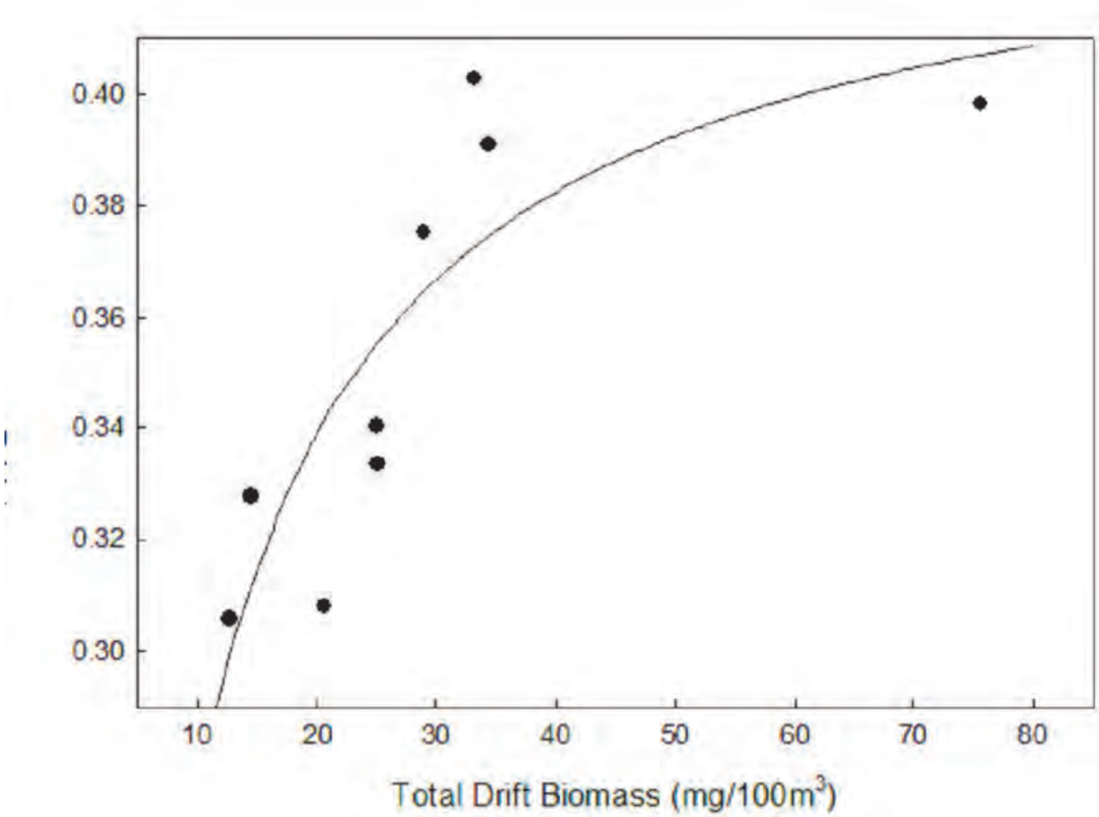

Figure 8.3: Non-linear regression of *O. mykiss* consumption (proportion of maximum consumption for given temperature regime and fish size or the P-value) and total drift biomass (mg/100m^3)

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 8.4).

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.

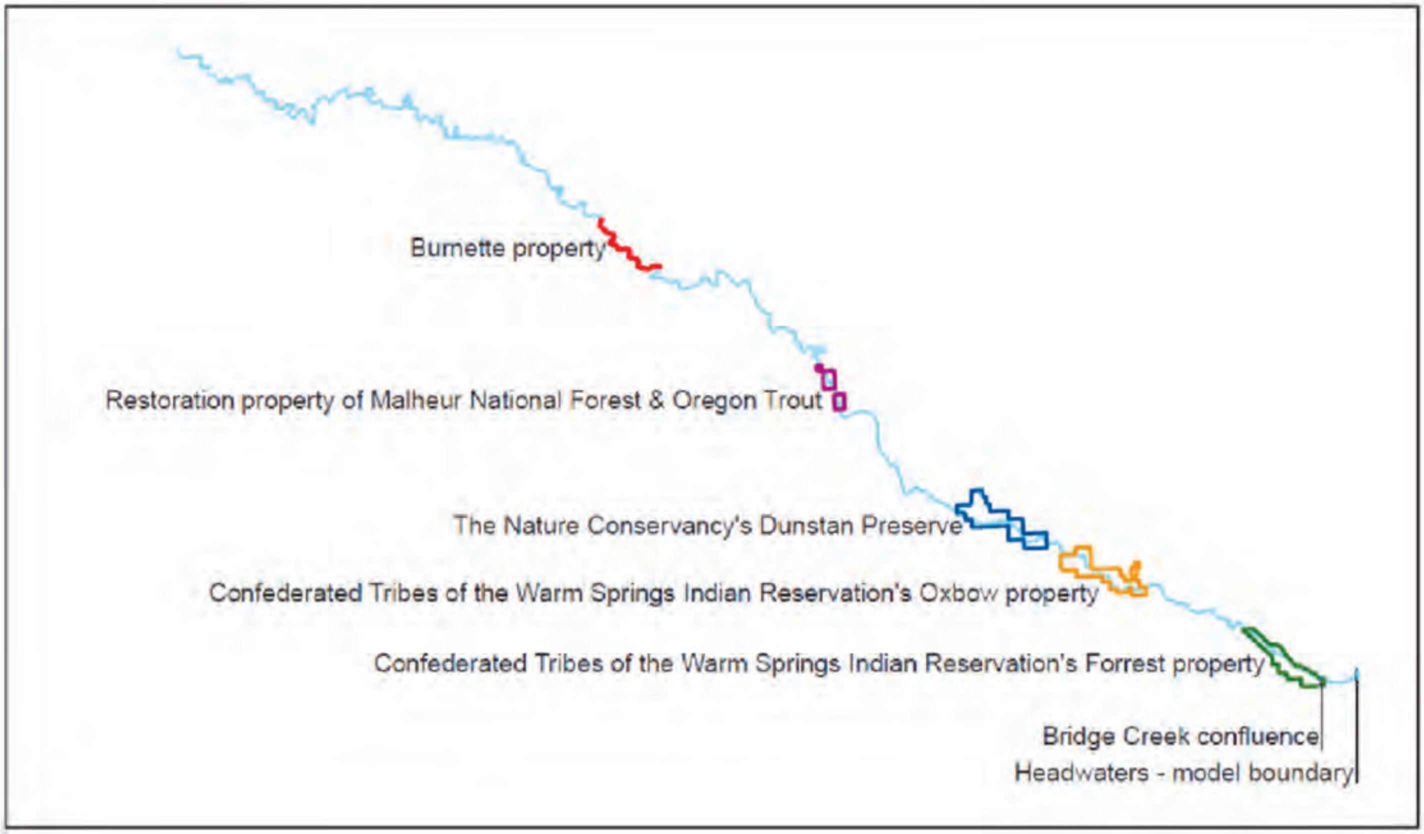

Figure 8.4: Location of restoration actions of the Middle Fork John Day Intensively Monitored Watershed study.

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 8.5. 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 8.3 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 8.5). 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 8.6). The results of the Heat Source modeling are discussed in more detail in Crown (2010).

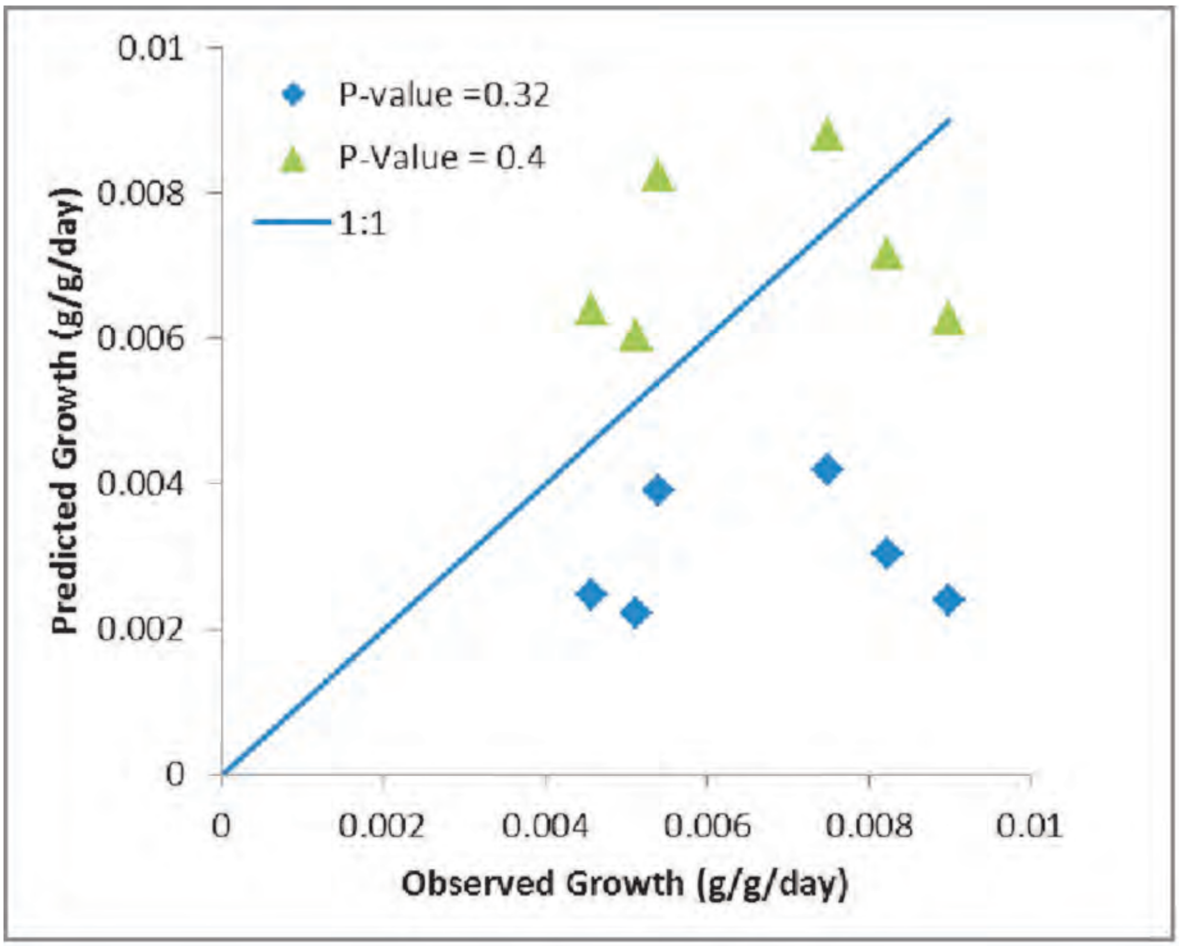

Figure 8.5. Observed versus predicted growth (normalized by grams of growth per gram of fish per day) at six sites in the upper Middle Fork John Day River. If we were able to predict actual growth, the points would fall on the 1:1 line. Blue diamonds are original P-value of 0.32, and green triangles are growth modeled with a P-value of 0.4.

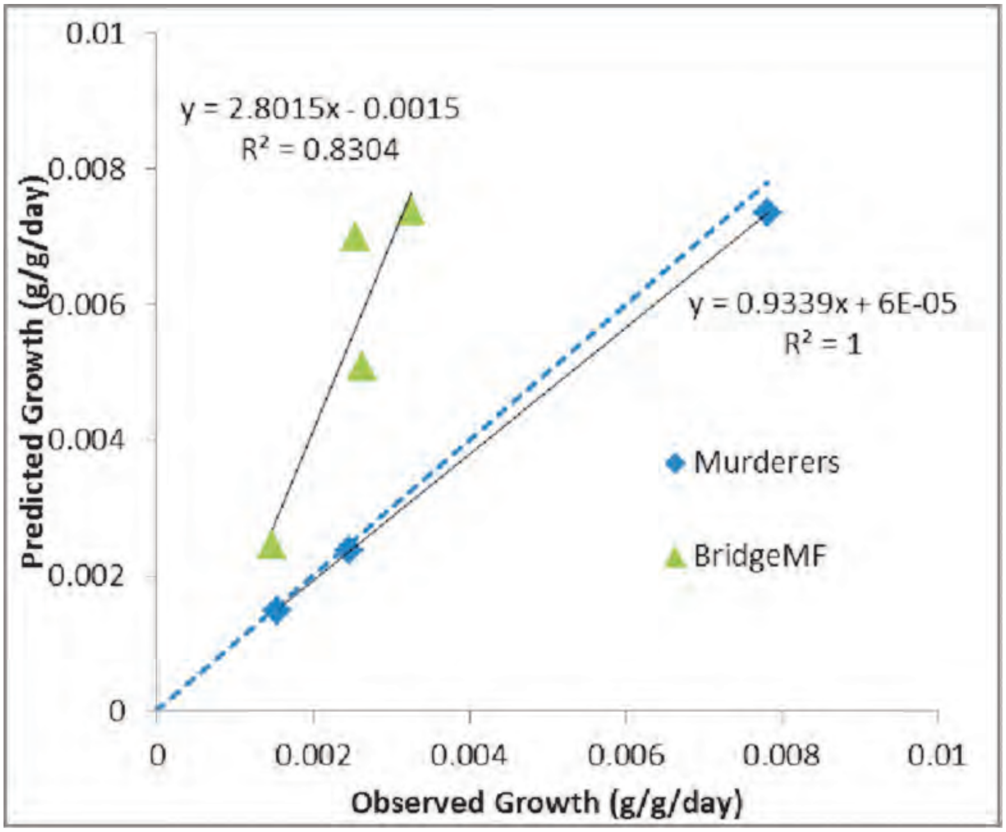

Figure 8.6. Observed versus predicted growth (normalized by grams of growth per gram of fish per day) at four sites on Bridge of the Middle Fork (green triangles), and 3 sites on Murderers Creek (blue diamonds). If we were able to predict actual growth, the points would fall on the 1:1 line (dotted blue 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 8.7). 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 8.8). 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.

Discussion

The growth model accurately predicted growth on the stream that it was partially developed in (Murderers Creek); the relationship (Figure 8.3) was developed 5 years 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.

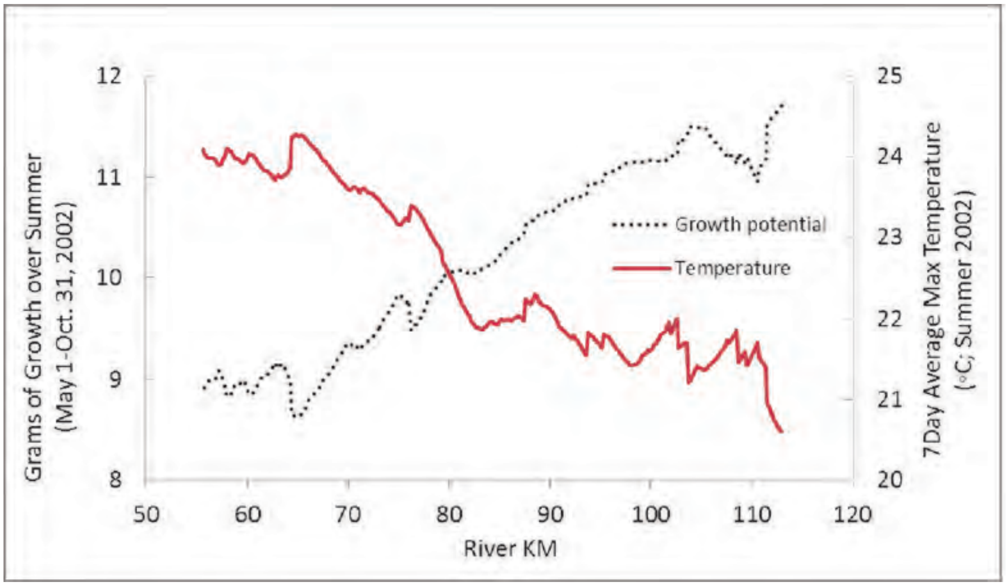

Figure 8.7. Relationship between temperature profile as modeled from Heat Source in 2002 (red solid line) and the modeled growth potential of 20 g juvenile steelhead during May 1-Oct. 31, 2002 (black dotted line) along the upper Middle Fork John Day River.

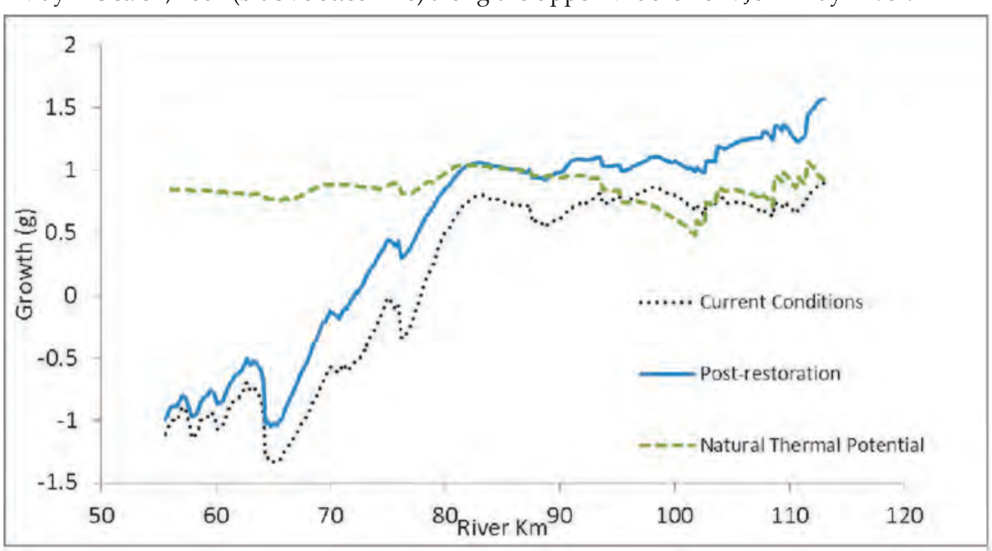

Figure 8.8. Growth (g) potential of a 20 g *O. mykiss* between July 1-Aug 15 (2002), for current thermal conditions (black dotted line), post-restoration as planned by the Middle Fork IMW study (blue solid line), and under natural thermal potential (green dashed line) for the upper Middle Fork John Day River.

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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 9.1). 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 9.1). 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 9.1).

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 9.1 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 forag-

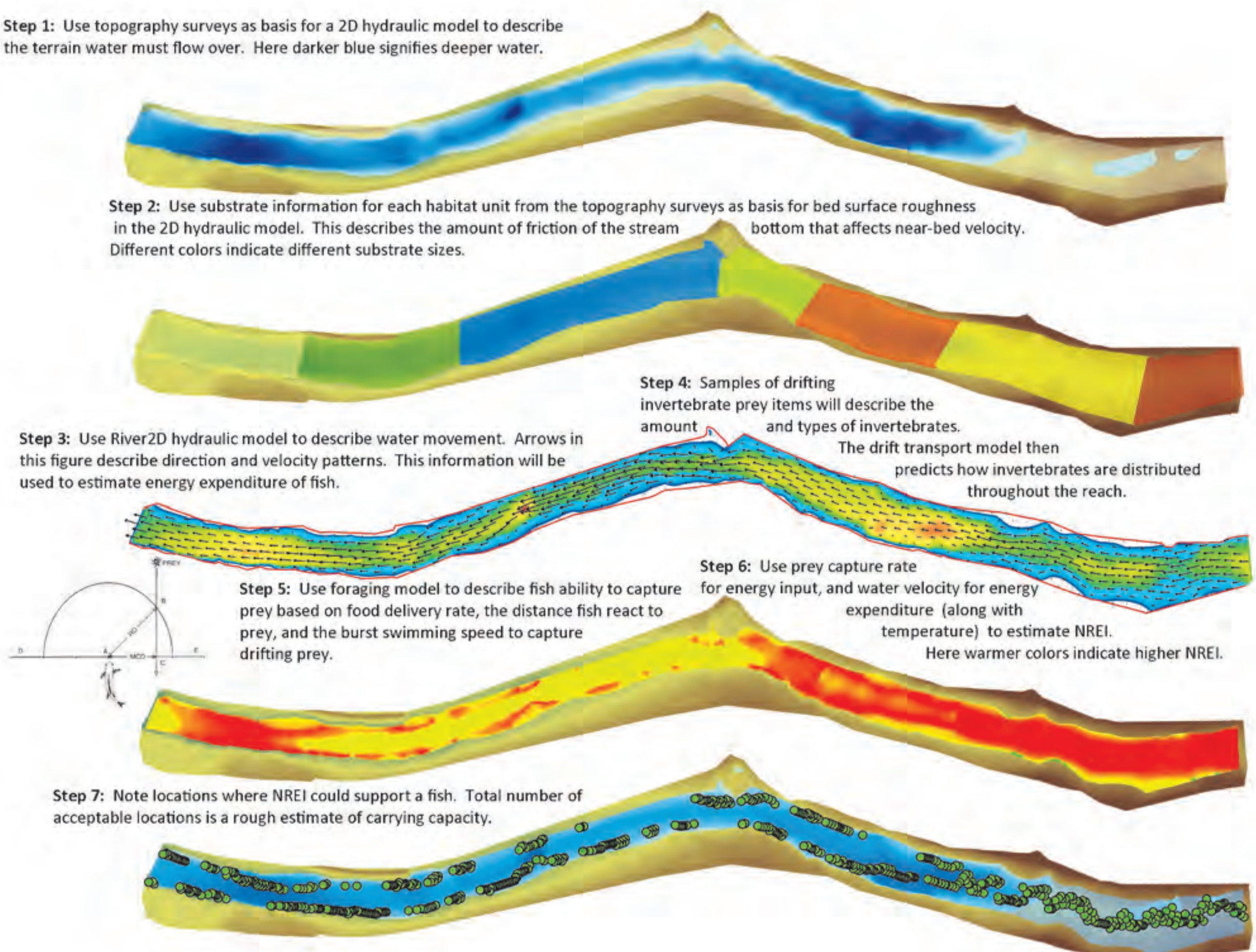

Figure 9.1. Estimating energy available (net rate of energy intake or NREI) and carrying capacity of juvenile steelhead in a stream reach.

ing spacing for 150mm steelhead), creating a three-dimensional array of cells, each of which contain an equal portion of the total discharge (Figure 9.1 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 9.1 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 9.1 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 9.1 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 9.1 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 9.1 Steps 7).

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 9.2).

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 9.3).

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.

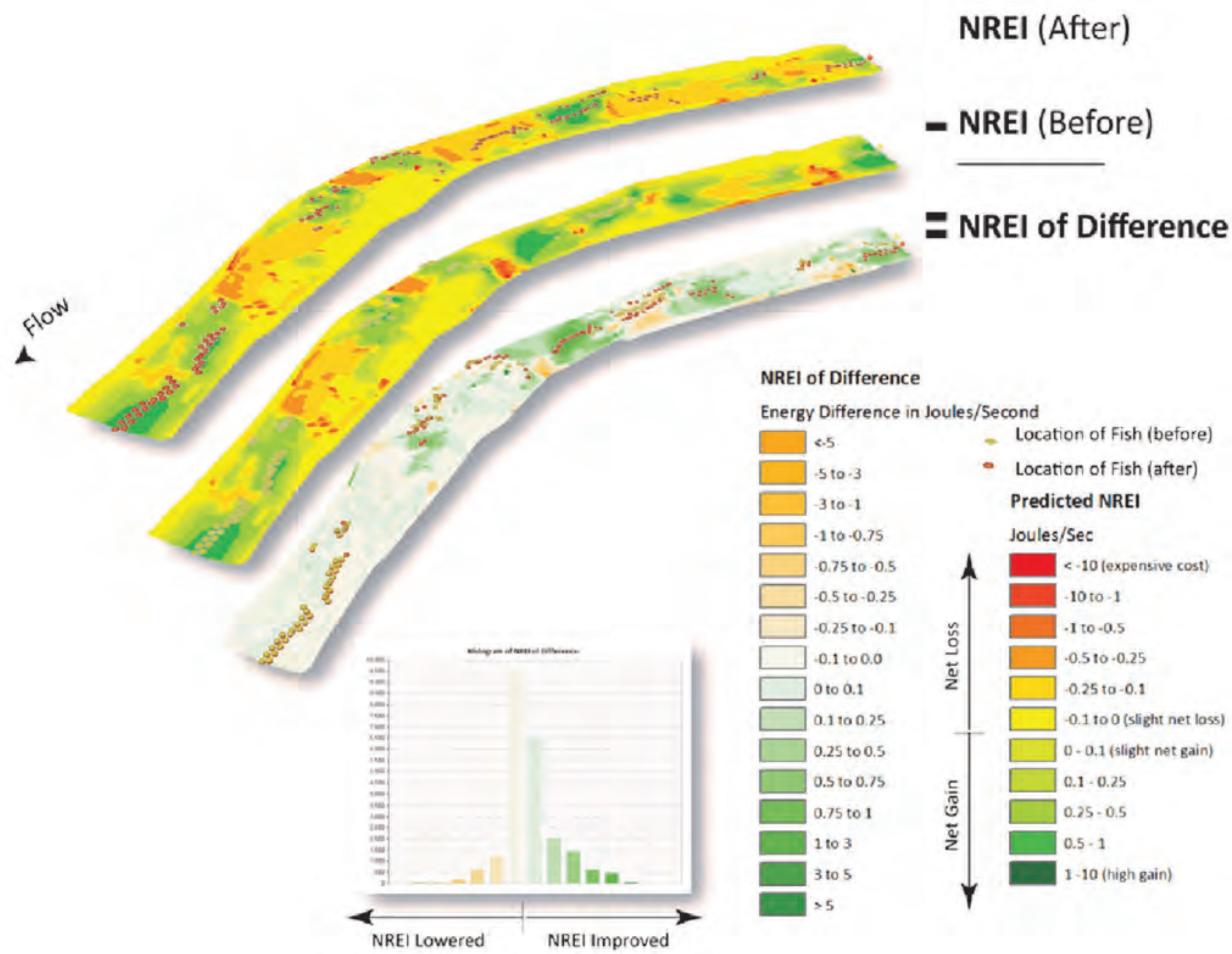

Figure 9.2. The energy available (net rate of energy intake or NREI; the colored surface) and abundance (dots represent placement of fish) pre-treatment (Before) and hypothetical post-treatment (creation of pools via wood additions) in a reach of the South Fork of the Asotin. If NREI (Before) is subtracted 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.

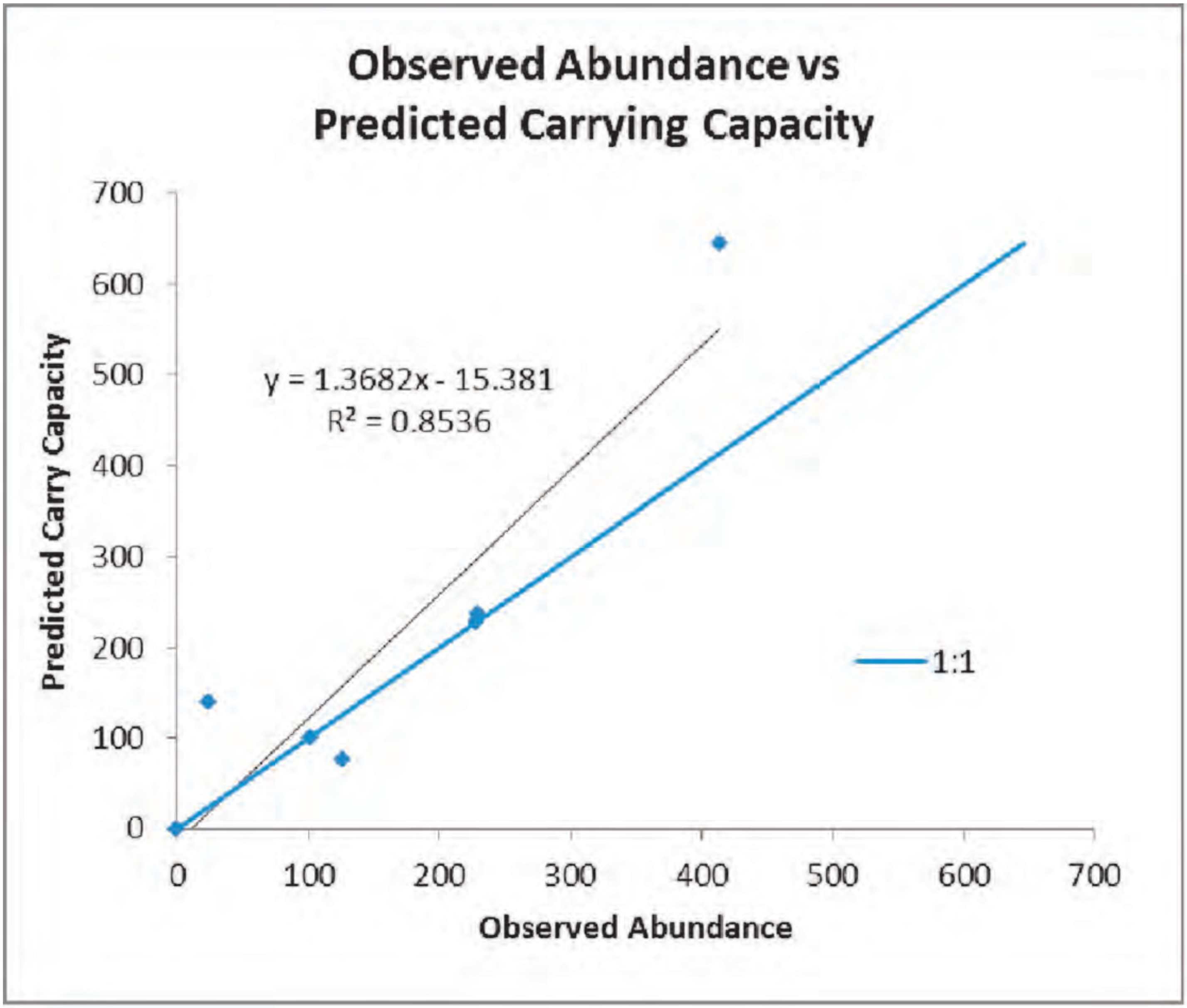

Figure 9.3. Observed versus predicted abundance of fish across reaches in 8 different streams. If we were able to predict actual abundance, the points would fall on the 1:1 line.

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CHAPTER 10: Analyzing Juvenile Salmonid Growth Data from the Salmon Basin

Author: Kevin E. See

Introduction

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. Differences in growth before and after restoration could be one measure of the effectiveness of a restoration action. As a test of the analysis framework to detect differences in growth, growth data from several watersheds in the Secesh and the Lemhi basins were analyzed.

Data & Methods

ISEMP has collected juvenile growth data in the Secesh and Lemhi from 2005-2011. However, most of the fish were captured in 2009-2011. The data were filtered to focus on fish recaptured in the same spatial area as they had been tagged, to allow for the assumption that all growth took place within that spatial area. Fish recaptured less than twenty days after being tagged were excluded from this analysis, to eliminate effects on growth due to the tagging experience (Bateman & Gresswell, 2006). This led to a total of 529 steelhead and 1484 Chinook available for this analysis.

Because fish were captured at multiple lengths and ages, the Schnute growth model was used to estimate differences in growth regimes (Tables 10.1 and 10.2) (Quinn & Deriso, 1999). This model takes the length at tagging (Y1), the length at recapture (Y2) and the time in between (delta t) and models it with two parameters. From these two parameters, a growth curve can be calculated, which graphically shows the difference in growth between different watersheds.

$$Y_2 = (Y_1^{\gamma} + \eta^{\gamma} * \Delta t)^{\frac{1}{\gamma}}$$

Results & Discussion

The results show distinct differences in the growth regimes between and within watersheds. 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 10.1 and 10.2, 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 10.2).

Table 10.1. Parameter estimates of the Schnute growth model for steelhead from the Salmon River basin 2009-2011.

	n	Eta	std dev.	Gamma	std dev.
Secesh Tribs	71	21.97	1.62	0.88	0.00
Secesh Mainstem	101	127.00	4.43	2.40	0.01
Hayden	79	161.65	18.23	2.45	0.05
Lower Lemhi Mainstem	65	195.58	26.56	4.29	1.10
Lower Lemhi Tribs	77	120.00	9.14	2.60	0.03
Upper Lemhi Mainstem	76	129.48	28.31	1.13	0.00
Upper Lemhi Tribs	60	128.71	7.91	2.21	0.01

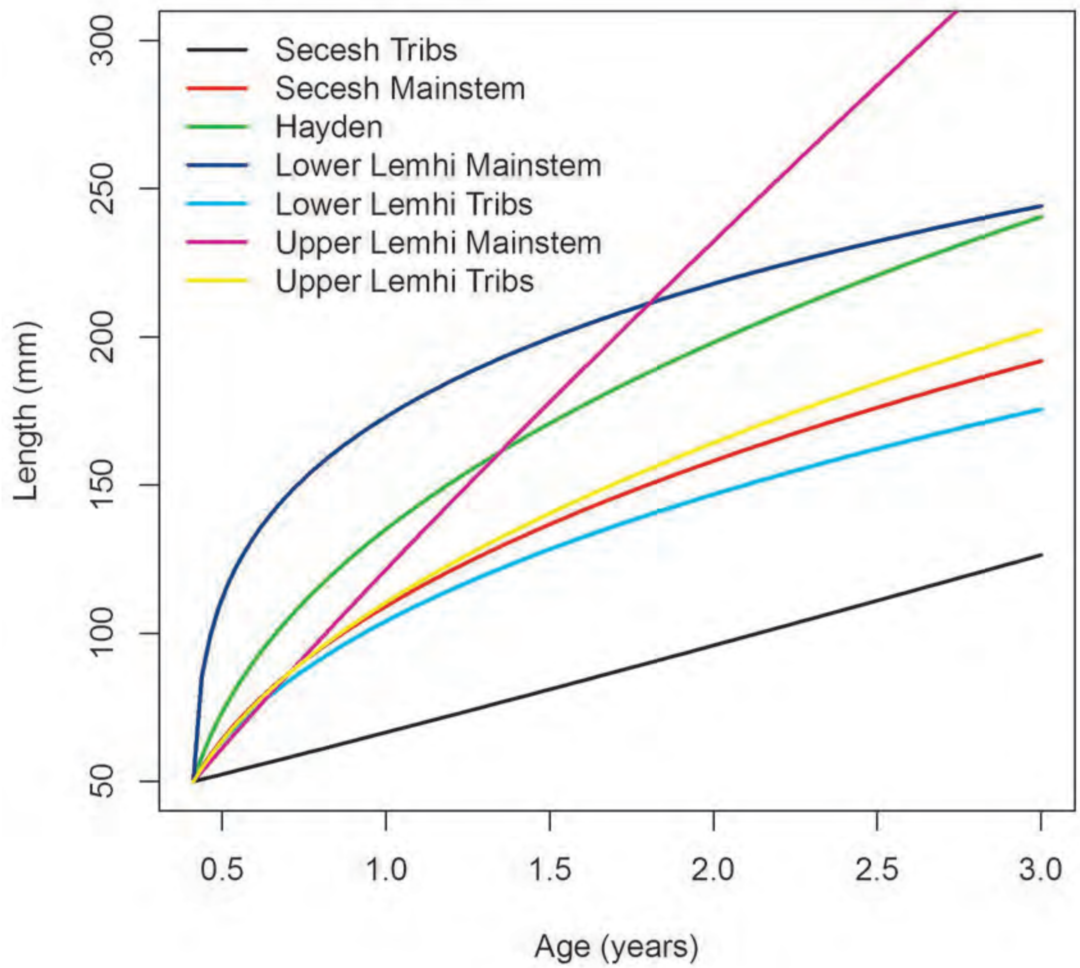

Figure 10.1. Fitted growth curves using mark-recapture data of steelhead from the Salmon River basin 2009-2011.

Table 10.2. Parameter estimates of the Schnute growth model for Chinook from the Salmon River basin 2009-2011.

	n	Eta	std dev.	Gamma	std dev.
Secesh Tribs	259	91.01	0.84	3.63	0.06
Secesh Mainstem	925	91.85	0.24	4.11	0.02
Hayden	37	130.86	11.20	3.19	0.10
Lower Lemhi Mainstem	59	121.37	2.05	6.29	2.92
Upper Lemhi Mainstem	199	118.11	2.65	3.57	0.11

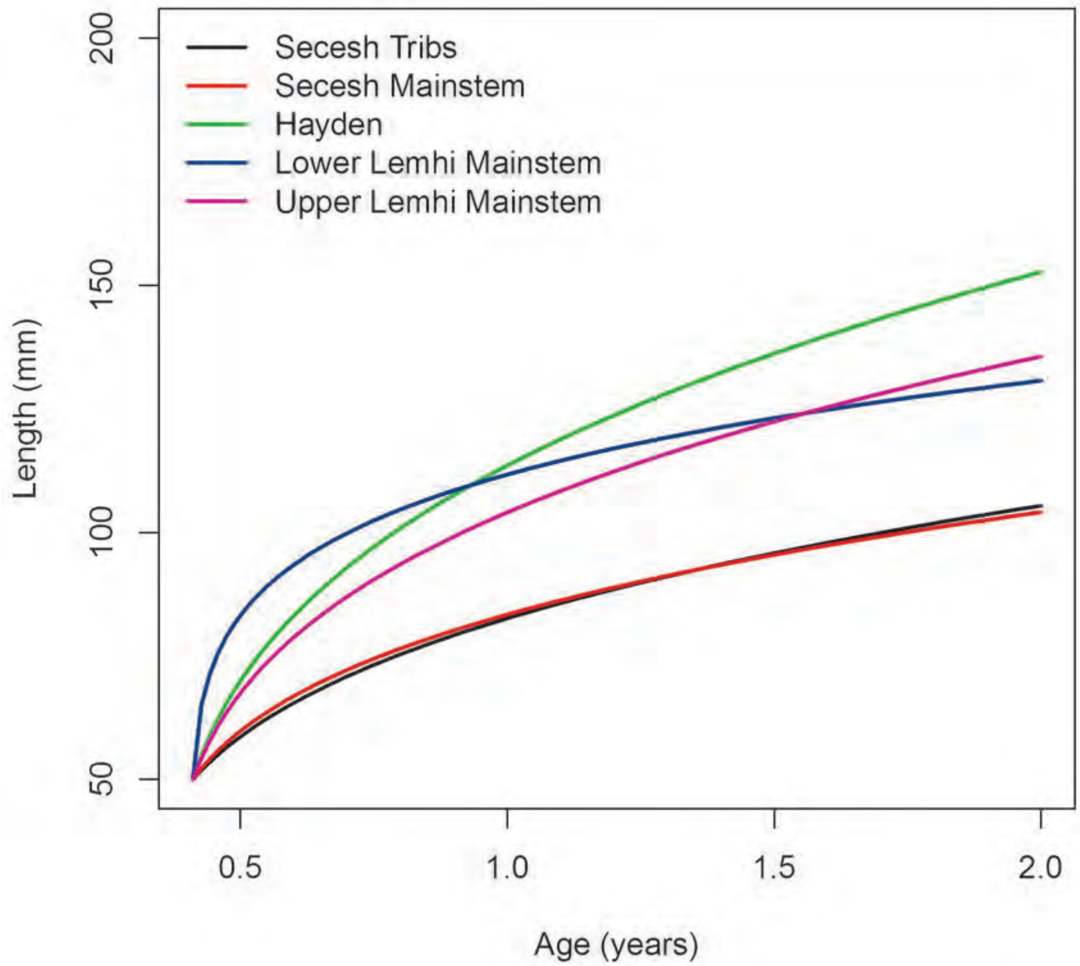

Figure 10.2. Fitted growth curves using mark-recapture data of Chinook from the Salmon River basin 2009-2011.

Growth has been linked to survival, although explicit quantitative relationships have yet to be established in the ISEMP basins. This work demonstrates that growth models can be fit with this type of monitoring data, and differences between watersheds can be established. As a piece of the larger question relating freshwater habitat to survival, growth may offer one way to differentiate between good and poor habitat quality, or to assess the effectiveness of habitat restoration actions. 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.

Literature Cited

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CHAPTER 11: Products from ISEMP: A Closer Look

Large-scale experimental manipulations of stream habitat condition to test mechanistic linkage between stream habitat and fish population processes

Effectiveness Monitoring of Riparian Fencing (June 2013)

ISEMP has also conducted fish and habitat surveys within riparian exclosures installed by ODFW (funded by BPA) and compared them to paired control reaches (areas still subject to grazing). The effectiveness monitoring strategy employed in this study is to use the 20 year time series of structures to assess the expect time and benefits of this restoration approach. The work demonstrates the ability of alternative post-hoc opportunistic designs to detect changes due to restoration. If possible, benefits of this major restoration strategy may be documented now rather than 20 plus years it will take for riparian vegetation to establish and influence fish habitat. This work will evaluate the changes due to removal of grazers on fish habitat including geomorphic changes, fish habitat, and riparian vegetation and evaluate the changes due to removal of grazers on fish abundance, growth and survival.

Lemhi Intensively Monitored Watershed study. (October 2017)

The 2018 BiOp check-in requires quantitative evaluations of the effectiveness of habitat restoration at achieving targeted increases in freshwater productivity. The ISEMP project in the Lemhi River will report changes in freshwater productivity that accompanied tributary reconnections and site-specific habitat restoration actions (e.g., channel realignment). These estimates will be accompanied by uncertainty, and where changes are not detected an accompanying *post-hoc* power analysis will identify the minimum change that could have been detected given realized sampling effort. If supported by the data, an analysis of remaining limiting factors will be conducted.

Bridge Creek Intensively Monitored Watershed study (December 2017)

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. ISEMP will have implemented 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. the long term, beaver dams will facilitate stream geomorphic changes that includes 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, raise water tables in the alluvial aquifer and thus help to greatly expand the amount of riparian forest and reduce stream temperatures.

The goals and outcomes of this work will be to:

- Demonstrate whether beaver dam support structures, to encourage stable beaver dam construction that eventually reconnects incised streams to the floodplain, is a viable and efficient stream restoration tool. (December 2012)
- Document how the restoration action results in changes to habitat and fish, and the expected benefits to this approach to improving fish production, so expected and estimated benefits to similar actions can be extrapolated (Demonstrate impacts of restoration to: temperature response, fish responses, riparian vegetation responses, ground water responses, beaver population response; December 2014).

Entiat River Intensively Monitored Watershed study - Analysis of the effectiveness of instream watershed restoration on the populations of spring Chinook salmon and steelhead. (December 2020)

A hybrid staircase hierarchical design is being used to guide instream habitat restoration actions in the Entiat River subbasin. The first round of habitat action implementation begins in 2012, with actions implemented in 2014, 2017 and 2020. Habitat (CHAMP) and fish (ISEMP) effectiveness monitoring has been ongoing since 2011 and will measure any change in the habitat and resulting change in the fish population's abundance, growth and survival as a result of restoration actions. Results from this monitoring will help guide restoration actions in other subbasins.

Broad-scale juvenile and adult fish population monitoring aligned with existing, ongoing habitat monitoring, to form the basis for extrapolating mechanistic fish-habitat relationships beyond experimental watersheds

ISEMP is developing a hierarchical survey strategy that will not only address status and trends of salmonids and their habitat but also provide potential information for the development of fish-habitat relationships, limiting factor analyses, prioritization and planning of restoration and management, and information for Intensively Monitored Watersheds. In addition, the strategy also provides data to help determine the appropriate sampling scales and effort for monitoring, protocol precision, accuracy and efficiency, protocol comparisons, and development of protocol crosswalks.

Estimating adult abundance from redds

Redd surveys are an important measure of how many spawners are present within a watershed, but they are fraught with potential errors. In addition, most common methods of estimating the total number of spawners from redd counts have no measure of uncertainty. This product will establish methods to incorporate observation error so as to calibrate observed redds to true number of redds, with uncertainty. Building on these results, this product will take the adjusted redd counts, including the uncertainty in those adjusted counts, and provide an estimate of the true escapement, with uncertainty. These results will be compared to alternative escapement estimates when available. The product of this work will be recommendations for future monitoring based on the cost effectiveness, bias, precision, and information value of alternative escapement estimation methodologies.

- *Recommendations for steelhead redd surveys in the Wenatchee (December 2012)*
    - ◊ This white paper will provide management focused recommendations for designing redd surveys and analyzing the data from such surveys to generate escapement estimates in the Wenatchee subbasin.
- *Recommendations for Chinook redd surveys in the Snake (December 2013)*
    - ◊ This white paper will provide management focused recommendations for designing redd surveys and analyzing the data from such surveys to generate escapement estimates in the Snake basin.
- *Use of redd surveys manuscript (March 2014)*
    - ◊ This manuscript, to be published in a peer-reviewed journal, will explain potential biases in redd counts, explore ways to correct those biases, and detail how to best convert corrected redd counts to adult abundance estimates, including the appropriate uncertainty.
- *Guidance on the use and validity of steelhead index spawning ground counts. (June, 2013)*

The total number of steelhead redds the Wenatchee River subbasin is estimated annually by the WDFW using index spawning ground counts in all known reaches/tributaries with significant steelhead spawning populations. From 2005-2009 ISEMP conducted redd surveys in at least 25 probabilistically selected streams reaches that represent the entire subbasin to determine if index spawning ground counts adequately characterize the abundance of steelhead spawning for an entire population. Product includes:

- ◊ Estimate bias in abundance estimates from index spawning ground counts by comparing results from redd surveys in probabilistically selected reaches.
- ◊ Compare the number and distribution of steelhead redds from index and probabilistically selected stream reaches.
- ◊ Describe relationship between abundance estimates generated from probabilistic and index reach-based surveys.
- ◊ Maps of geo-referenced steelhead redds in index and probabilistically selected stream reaches to understand the natural variability in the distribution of steelhead spawning.

Calibrating Snorkel Counts to Fish Abundance Estimates (November 2013)

Fish were surveyed in the Wenatchee subbasin from 2004-2010 and in the Entiat subbasin from 2006-2010 using snorkeling methods. In 2011, and moving forward, abundance estimates were calculated using three pass depletion and mark-recapture methods. By conducting snorkel surveys simultaneously with these other methods, we are able to model the bias in snorkel counts, and adjust the older counts to maintain and extend this valuable time-series.

- Calibration of historical snorkel surveys using mark-recapture studies so that historical snorkel data can be converted into a population estimate.
- Analysis of temporal variation in fish counts at multiple time scales (at diurnal, daily, weekly, monthly time scales) to guide sampling design.
- Stratified correlations between key habitat metrics and fish habitat utilization metrics (i.e., abundance, distribution, and size of juvenile anadromous salmonids) to guide habitat restoration actions.
- Analysis of the precision of snorkel surveys.
- Quantification of the number, species, and size of fish present within each site for use by ISEMP collaborators.

Approaches to the uncertainty associated with downstream migrant trapping methods. (December 2012).

Abundance estimates from rotary screw trap data are often fraught with high levels of estimation error, this error is often not well reported, and managers may not realize the level of imprecision in these estimates. Additionally, methods to reduce the error in estimation can be expensive and ineffective. ISEMP has been conducting a series of investigations to highlight the importance of these generally overlooked weaknesses and to suggest Improvements that will reduce sampling costs while improving the value of these estimates of fish emigration. Products of these analyses include:

Three analyses:

- Comparing The Relative Performance Of Downstream Migrant Abundance Estimation Methods Through Simulation

- Expanding the previous catch expansion analysis to incorporate the intentional use of trap downtimes in an effort to reduce trapping effort/costs.
- Using multi-year trap efficiency data to generate modeled trap efficiency relationships within a given year to reduce trap efficiency trial effort.
- Comparison of life stage-specific outmigration patterns between watersheds and the subbasin.
- Comparison of estimates for watersheds versus subbasin.

Precision, Bias, Reliability, and Cost-Comparison of RST Versus Instream PIT tag detection (IPTD) Based Juvenile Abundance and Survival (December 2014)
This product will evaluate the tradeoffs between rotary screw traps and IPTDS for the purpose of estimating juvenile abundance and survival. The product of this work will be recommendations for future monitoring based on the cost effectiveness, bias, precision, and information value of each approach.

Lower Granite Dam and IPTDS escapement estimates – (Beta - October 2010; Final – March 2013)
Final Bayesian run decomposition for Snake River natural origin spring/summer Chinook salmon and steelhead coded in R, with parameter inputs based on "canned" DART queries.

Juvenile salmonid summer rearing (standing crop) population estimation (December 2013)
This product will determine whether the current estimation methods are statistically valid, whether the current effort in terms of the number and distribution of remote juvenile surveys is sufficient, and evaluate alternative sampling designs.

Juvenile Survival Models (December 2013)
This product will formalize statistical approaches to estimate survival, movement and residency rates for juvenile anadromous salmonids PIT tagged in remote juvenile surveys. This product will identify sample size requirements, the optimal distribution of remote juvenile surveys and the appropriate modeling approach.

Basin-wide estimates of juvenile abundance and juveniles per spawners (December 2013).
ISEMP and local co-managers will develop basin-wide estimates of juvenile abundance and juveniles per spawners. In the more intensively surveyed watersheds, estimates of survival, growth, and production will also be estimated (first annual estimates 12/2012).

Description of steelhead life history patterns in the Wenatchee and Entiat River subbasins. (June 2013)
Description of the size, life stage, timing, and spatial distribution of steelhead outmigrants trapped throughout the subbasin at rotary screw traps, and at instream PIT tag detection arrays and mainstem dams.

Landscape context data collection and analysis to support extrapolating mechanistic fish-habitat relationships beyond experimental watersheds

Introduce and produce a geomorphic framework (known as River Styles) to provide the watershed context to reach scale habitat information for Bridge Creek, Middle Fork, South Fork of the John Day; the Lemhi and Secesh of the Salmon; and the Wenatchee and Entiat. This framework will describe the current status of different habitat types, their expected trends, opportunities for restoration and land management, and a means prioritize and develop processed based subbasin management strategies. (expect draft product 12/2014 on this approach and to host several workshops for managers). The long-term goal is to combine the River Styles framework with the Watershed Production model to provide the estimated influence of mitigation strategies on population performance metrics (have working model by 6/2015)

Survey and sampling designs to support population-scale inference of fish-habitat relationships

Determine strata for organizing standard stream habitat survey designs (April 2010)
Using habitat data collected from 2007-2008 in the Wenatchee subbasin, best practices were established for grouping or combing various habitat metrics. These groupings were then used to design the valley class strata, based on the Beechie classes, which was adopted by CHaMP.

Design of observation error study (June 2011)
Designed a crew variability study to investigate the levels of observation error associated with various CHaMP habitat metrics.

Standardize fish sampling methods across ISEMP (June 2011)
Determined that removal and mark-recapture methods provided more data at multiple spatial scales compared to snorkel surveys, because they generated not only site-level abundance estimates (versus indices of abundance), but also involved tagging fish which allows for an investigation into movement and survival. Therefore, all ISEMP fish surveys in 2011 involved either three-pass depletion or mark recapture methods.

Power analysis of habitat data (April 2011)
Investigated the trade-off between sample size to standard error of mean habitat metrics across a watershed, based on habitat data from the Wenatchee subbasin. One result was the validation of the sample size of sites within each subbasin for CHaMP.

PIT tag array detection analysis with DIDSON imaging sonar (June 2011)
The DIDSON is an imaging sonar often used for counting fish in riverine environments. This project developed an analytical method to estimate escapement, taking into account human observer error, errors in the automated "Convolved Samples over Threshold" (CSOT) algorithm that DIDSON uses to count passage, and downtime when the DIDSON sonar is not operating.

Evaluate channel unit fish sampling (June 2012)
Analyzed one year of data where fish were sampled at the Tier 2 channel unit level to determine whether information at that spatial scale was worth the additional effort. Recommended surveying at the site level, and allocate effort savings into sampling more sites.

Set necessary sample size for calibrating single pass fish surveys (June 2012)
Articulated the tradeoff between the number of sites sampled using three-pass depletion or mark/recapture and the accuracy of a ratio estimator that could be used to estimate site abundance from a single pass survey.

Decision tree to determine fish sampling design (December 2012)
There are many considerations to account for when designing fish survey plans, including budget, number of sites, site size, methods, etc. This project will formalize the trade-offs that should be considered and assist in determining an optimal fish sampling design.

Standardized GRTS processing scripts for fish and habitat data (December, 2013)

Data management to support population-scale inference of fish-habitat relationships

IPTDS Data Management (October 2009)
ISEMP has produced a standard suite of IPTDS infrastructure with supporting data management tools that enable automated remote data downloading, parsing, and upload to PTAGIS. Additionally, these tools return real-time site diagnostics; distribute automated alerts via email informing site stewards of potential malfunctions and indicating successful data uploads to PTAGIS, and allow a number of optimizations to be completed remotely without site visits.

PIT tag Database (December 2013)
ISEMP is currently operating a beta version of a database to store PIT tag interrogation and auxiliary data from remote juvenile surveys, juvenile re-sight efforts (e.g., mobile PIT tag detection), and IPTDS. An exportable version of this database will enable seamless data transfer to PTAGIS and support desktop queries.

Regional PIT Tag Data Queries (January 2012)
ISEMP in collaboration with DART have developed a number of automated PTAGIS data queries that perform automated data reduction in support of common analyses accompanying the operation of IPTDS.

PTAGIS Database Version II (December 2013)
ISEMP is currently leading a PSMFC workgroup tasked with developing IPTDS requirements for the PTAGIS database. This work will develop metadata standards that will be enforced in the next version of the PTAGIS database. Ultimately these data standards will ensure the proper use of IPTDS data, particularly for analyses that utilize multiple IPTDS sites and/or require estimates of IPTDS efficiency for PIT tag based expansions (e.g., LGR IPTDS run decomposition).

STEM Databank (January 2008 – present):
The Status, Trend and Effectiveness Monitoring Databank was created as a highly flexible, metadata rich data repository to hold the

diverse array of data generated and collated by ISEMP. STEM has gone through 3 major releases and is currently developing metric storage and retrieval capacities, as well as metadata linkages to monitoringmethods.org and the ISEMP database that supports tagging operations (located in Boise, ID and maintained/managed by QCI).

- Measurement storage and retrieval (1/1/2008, 10/1/2011 updated release). Raw measurement data collected by biologists is often not shared publically due to limited data serving capacities, time limitations, and inconsistent data formats. ISEMP strives to publish raw data and make it available upon request to the public to increase data sharing.
- Metric storage and retrieval (10/1/2012). Metric calculations and definitions will be stored concurrently with measurement data, which will make it one of the few data repositories that can document the path between raw field data and metric data.
- Metadata linkage to MonitoringMethods.org (12/1/2012). ISEMP plans to support regional metadata documentation efforts by linking the STEM repository to MonitoringMethods.org.
- Flexible online data upload (6/1/2013). ISEMP plans to develop a flexible online data upload tool to allow input of data from spreadsheets directly into STEM. This will greatly increase the flexibility and range of data that can be put in and retrieved from STEM.
- Metadata and metric linkage to ISEMP's fish database (4/1/2013). The fish database maintained by QCInc in Boise currently holds limited metadata, as STEM Databank has been the metadata repository for ISEMP. ISEMP will link these two databases (along with considering linkages to MonitoringMethods.org) to ensure complete metadata documentation for all ISEMP databases. A similar linkage and documentation of ISEMP fish metrics will also take place.
- Metadata linkage to cbfish.org (proposed and under review for 6/1/2013). ISEMP is considering linking project metadata currently stored in cbfish.org to metadata documentation in STEM.
- Data distribution to DART (8/1/2013). Distribution of data to DART via web services would allow public review and limited graphing functionality of data stored in STEM via DART's online applications.

Aquatic Resources Schema (ARS) (September 2009)
The Aquatic Resources Schema was developed by Environmental Data Services for ISEMP as a global schema to facilitate data entry, storage and retrieval of ISEMP data. It is an MS Access database with a pre-defined structure that streamlined and standardized much of the data terminology and structure currently used by ISEMP. Although this product is no longer supported as a user interface for ISEMP, its utility and metadata documentation standards were one of the first joint metadata and data documentation efforts for ISEMP and the PNW.

Automated export of data between ISEMP and PTAGIS repositories (June 2009)
The Aquatic Resources Schema began supporting the export of ISEMP generated data to PTAGIS in 2009 by pulling data from PTAGIS' desktop software, P3. These tools streamlined data flow from data collectors to P3 to ISEMP data storage and finally to PTAGIS, allowing project metadata and supporting auxiliary information to be collected and stored in addition to PTAGIS data requirements. The original export functionality has now been replicated in the ISEMP fish database.

Protocol documentation and requirements (January 2008)
Initial migration of ISEMP data to the STEM Databank repository discovered many inconsistencies in the data dictionary and protocol documentation within ISEMP. This inspired all ISEMP data collection efforts to rigorously document methods and protocols for data collection and update these annually. This effort allowed significant progress in robust tool development to support data collection and storage needs. ISEMP projects and MonitoringMethods.org have been influenced by the data structure and requirements necessary for well-documented protocols.

Data Management lessons learned paper (July 2012)
A manuscript describing several data management lessons learned will be submitted to the peer-reviewed journal Fisheries for review. This manuscript is one of several efforts to relay practical lessons about data management learned through ISEMP to local data managers.

Data flow paper (August 2012)
A manuscript summarizing the difficulties and frustrations experienced by local data managers during the data compilation and use process will be submitted to a peer-reviewed journal targeting local data managers in the summer of 2012. The paper documents the need and desire of fisheries data managers in the northwest to improve data handling.

Data quality guidelines (May 2009)
In 2009 ISEMP generated an outline of data quality guidelines that has been used as the backbone of QA checks customized for each data stream within ISEMP. ISEMP's data quality checks are in continual development, but this outline marked the onset of standard-

ized and more rigorous data quality from ISEMP data streams.

Ongoing water quality compilation for John Day Basin (January 2006 - ongoing updates)
This data compilation effort was one of ISEMP's initial efforts to collate historic data from multiple agencies. ISEMP has continued to compile data from local organizations annually, which has provided source data for 3 water temperature models to date, one of which will likely be expanded for use in CHaMP watersheds in 2013.

PITA inventory (February 2011, update September 2012)
There is widespread use of instream PIT tag detectors in small streams throughout the PNW, although not all of these in-stream structures currently reside within PTAGIS. In 2011 ISEMP compiled a list of all in-stream detectors and this list is available upon request.

IMW locations (June 2011, update September 2012)
In 2011 ISEMP compiled a list of watersheds that are commonly referred to as Intensively Monitored Watersheds (IMWs). Boundaries were reviewed by local agencies and will be updated in 2012.

Uniform GIS layers compiled and clipped to PNW (January 2011, updated September 2012)
ISEMP curates GIS layers for the PNW for ISEMP specific projects. Many of these uniform layers are available nationally (e.g., DEMs, Land Use/Land cover/precipitation, temperature, etc) but several are maintained on a state-by-state basis and in these cases, ISEMP has generated standardized GIS layers across states.

Spatio-temporal analysis of fish-habitat relationships to develop quantitative rule set that links abundance and productivity to habitat quality and quantity.

Fish-habitat relationship modeling (December 2014)
Understanding fish/habitat relationships can help direct restoration efforts, such as determining where effort should be focused, and what types of restoration will have the biggest impact in a given location.

- ***Preliminary analysis on habitat data (April 2010)***
    - ◊ Using habitat data from the Wenatchee, collected in 2007-2008, best practices for summarizing a multitude of habitat information were established.
- *Mechanistic models (June 2013)*
    - ◊ Use mechanistic models to estimate growth and abundance of salmonids. Models will be used to make prediction of BiOp related fish metrics (juvenile production) or metrics related to these metrics (growth) across multiple ESUs based on habitat information collected from CHaMP – NREI and growth potential.
- *Final analysis (December 2014)*
    - ◊ This product will describe the fish/habitat relationships at the site level, based upon juvenile surveys and the complete three-year panel of CHaMP habitat data. Several different methods of characterizing these relationships will be explored. In addition to describing these relationships at the site level, a hierarchical approach will be employed to estimate the fish/habitat connections at the watershed or population scale.
- *Incorporate relationships in the watershed production model (December 2014)*
    - ◊ This product will formalize the fish/habitat relationships utilized in the watershed production model based on empirical relationships developed from the above analysis.

Watershed production models to evaluate the impact of management action scenarios for key populations and habitat action tactics.

The Salmon Subbasin ISEMP is based on a Beverton-Holt multi-stage model that views fish survival, abundance, and distribution as a function of habitat quality and quantity. This approach was selected to address the primary goal of ISEMP, namely to develop approaches to assess the effectiveness of habitat restoration on the freshwater productivity (e.g., egg to smolt survival) of anadromous salmonids. The model requires the following data to develop those functions:

- Adult abundance.
- Juvenile abundance, survival, and distribution within reaches subject to restoration initiatives and reference reaches (more generally, tributaries of the Lemhi River).
- Spatially representative habitat survey data capable of yielding estimates of habitat quantity that can be "rolled-up" to spatial

scales of interest.

The model is being developed initially from the ISEMP Salmon basin data streams, but is also in parallel being constructed for more general application. The model will be customized for the other ISEMP basins, and a final product from this task will be a fully exportable version of the life-cycle modeling framework.

R-Code Watershed Model with Flexible Input and Parameterization (December 2012)

Initially the watershed model was programmed in Excel, later transferred to Visual Basic, and is now being re-programmed in R.

Watershed Production Model populated with data from the Lemhi and Sesech Subbasins. (December 2012)

Watershed Production Model populated with data from the Wenatchee and Entiat Subbasins. (July 2013)

Watershed Production Model populated with data from the John Day Subbasin. (December 2013)

Reduced Watershed Model (June 2015)

In its current form, the watershed model utilizes data streams that are not widely available across most Columbia Basin populations. Utilizing sensitivity analyses and data from cooperating subbasins, we will identify minimum data requirements to reliably populate the watershed model. The product of this work will be a streamlined version of the model, and accompanying sampling design requirements, to enable export of the model to populations/locations with "standard" data streams. This product will include substantial guidance on output interpretation. In essence this product will test the exportability of fish/habitat relationships generated empirically in the Secesh and Lemhi, and place bounds on the interpretation of model output given data of varying quality and time series of varying duration.