

# Assessing the Status and Trends of Spring Chinook Habitat in the Upper Grande Ronde River and Catherine Creek: Annual Report 2017

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## Technical Report

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Columbia River Inter-Tribal Fish Commission

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# Assessing the Status and Trends of Spring Chinook Habitat in the Upper Grande Ronde River and Catherine Creek

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# **Executive Summary**

## **Background and Objectives**

The Columbia River Inter-Tribal Fish Commission (CRITFC) is conducting a fish habitat monitoring program in the Upper Grande Ronde River and Catherine Creek basins designed to evaluate the effectiveness of aggregate restoration actions in improving freshwater habitat conditions and viability of salmonids listed under the Endangered Species Act. Critical uncertainties for fisheries managers in the Columbia Basin are whether habitat restoration actions will yield a net improvement in basin-wide habitat quality, and whether expected improvements in fish production can be brought about by improvements in the quality and quantity of salmon habitat. The primary objectives of this project are to: 1) Assess current status and trends in fish habitat characteristics considered to be key limiting factors (particularly water temperature, pool habitats, streamflow, large wood, and fine sediment) to viability of spring Chinook Salmon populations; 2) Evaluate effectiveness of aggregate stream restoration actions aimed at improving key limiting habitat factors; and 3) Develop a life cycle model to link biotic responses of spring Chinook populations to projected changes in stream habitat conditions.

## **Life Cycle Model as an Organizing Concept**

One of the central components of this project is a spring Chinook life cycle model which provides a means to integrate habitat monitoring efforts with recovery planning. The life cycle model is a tool to simulate fish population trends in relation to projected habitat conditions, and to examine the relative benefits of habitat improvements on fish population recovery potential. The fundamental basis of the model is that intrinsic watershed factors (such as geology, climate, or valley morphology) interact with human actions (such as forest harvest, cattle grazing, or stream restoration) to affect processes that drive known limiting factors (e.g., flow, temperature, pool area, etc.), and therefore fish survival via both density-dependent and density-independent processes. This conceptual model represents the general structure of our research program.

The life history model is comprised of several interacting subcomponents that are built independently, each of which provides critical information about the interaction between landscape characteristics, instream habitat conditions, and fish response. Some of these subcomponents include modeling stream temperature from local riparian and geomorphic conditions, linking key limiting habitat characteristics (e.g., water temperature, fine sediment, pool abundance, and large woody debris) to fish abundance and productivity, estimating food base and growth potential for salmonids from stream macroinvertebrate



data and habitat characteristics, evaluating habitat characteristics and fish use of cold-water refuges, and evaluating the vulnerability of sites to low and high streamflows. Though these components are described here as providing inputs to the life cycle model, each component is a valid research project on its own, likely to yield interesting scientific insights and practical applications for conservation.

## **Progress and Key Findings from Individual Project Components**

### **Stream Habitat Conditions**

- Continued collection and QA/QC of stream habitat condition data using the Columbia Habitat Monitoring Program (CHaMP) methodology. Conducted 25 surveys within the spring Chinook salmon extent in 2017, totaling 175 site surveys performed by CRITFC since 2011.
- Summaries of eight key habitat metrics revealed that, across spatial extents, there exists large variability within the two more heavily disturbed watersheds (Upper Grande Ronde and Catherine Creek) compared with the Minam River watershed, a wilderness area.
- Continued monitoring of stream temperature at 71 sites across three watersheds.

### **Stream Temperature Modeling**

- We used a mechanistic water temperature model to investigate potential thermal benefits of riparian reforestation and channel narrowing to Chinook Salmon populations in the Upper Grande Ronde River and Catherine Creek basins.
- A combination of riparian restoration and channel narrowing was predicted to reduce peak summer water temperatures by 6.5 °C on average in the Upper Grande Ronde River and 3.0 °C in Catherine Creek, translating to predicted increases in Chinook Salmon parr abundance of 590 % and 67 % respectively.
- Although projected climate change impacts on water temperature for the 2080s time period were substantial (i.e., median increase of 2.7 °C in the Upper Grande Ronde and 1.5 °C in Catherine Creek), we predicted that basin-wide restoration of riparian vegetation and channel width could offset these impacts, reducing peak summer water temperatures by about 3.5 °C in the Upper Grande Ronde and 1.8 °C in Catherine Creek.

### **NetMap Data Summary**

- We used the NetMap analysis toolset in ArcGIS to calculate a large suite of landscape and river channel attributes that have important implications for fish survival and distribution including channel hydraulic geometry, floodplain

- complexity, riparian shade, erosion potential, road density, and salmonid intrinsic potential.
- NetMap attributes will be used as covariates in spatial statistical network (SSN) models to extrapolate from point-based measurements of key fish habitat conditions (e.g., large pool frequency, floodplain complexity, water temperature) to spatially continuous population-level predictions. These models will also be used to assess how restoration actions or land use might influence the quantity and quality of fish habitat in the future.

### **Historical Ecology for Setting Restoration Targets**

- General Land Office (GLO) survey field notes from the 19th century helped demonstrate that in two watersheds severely impacted by Euro-American land use, stream channel widths—a metric representing habitat simplification—increased in accordance with a geomorphic stream classification. Conversely, we did not detect significant change in stream widths in an adjacent, wilderness stream with minimal land use impact.
- Using a mechanistic water temperature model (Heat Source), we predicted that the combination of stream width and riparian restoration could decrease average summer water temperatures 6.6 °C in Upper Grande Ronde and 3.0 °C in Catherine Creek.
- The above restoration scenarios translate into substantial changes in the percentage of stream network habitable to salmon and steelhead migration (from 29% to 79%) and to core juvenile rearing (from 13% to 36%).
- Land use legacies leave an important footprint on the present landscape and are critical for understanding historic habitat-forming processes as a necessary first step towards restoration.

### **Fish-Habitat Relationships**

- CRITFC conducted fish snorkeling and electrofishing to determine reach-scale juvenile Chinook Salmon densities per linear stream distance. Based on the combined annual and rotating panel CHaMP survey visits, densities steadily declined from 2011-2016, but showed a small increase in the Upper Grande Ronde River and Catherine Creek in 2017. Juvenile Chinook densities in the Minam River were as high or higher than other watersheds, especially in 2015.
- Structural equation modeling (SEM) of fish-habitat relationships—after accounting for elevation—revealed that large pools, mean August water temperature, and stream power had a direct influence on Chinook parr abundance, whereas large wood had an indirect influence on fish through the formation of pools.

- The SEM was used to generate parr abundance estimates under various management scenarios including restoration of instream pools, riparian restoration and subsequent shading and cooling of water temperature, and climate change projected for the 2080s. Findings indicated that water temperature is likely the primary limiting factor in the upper Grande Ronde whereas pool structure is likely the primary limiting factor in Catherine Creek for the parr life stage. Therefore, the recommended management strategies for increasing habitat capacity in the two basins differ accordingly.

### **Food Web Response to Carcass Additions**

- Steelhead carcasses were added to three stream sections in the Upper Grande Ronde River to evaluate how the reduction of spawning salmon may be influencing stream food webs and in particular, juvenile salmonids.
- After the addition of carcasses in August, juvenile chinook and steelhead (*O. mykiss*) growth rates substantially increased in treatment reaches relative to controls.
- Juvenile *O. mykiss* mean condition factor was significantly greater in the treatment reach of two of the three pairs at the conclusion of the study in October. Juvenile Chinook mean condition factor was significantly greater in October in one of the two pairs in which Chinook were present and in this treatment reach, Chinook were approximately 22% larger than controls.
- Diets of Chinook and *O. mykiss* indicate elevated growth rates were likely attributed to direct consumption of eggs and carcass tissue rather than bottom-up pathways. Diets of non-salmonids indicated that these species were not consuming eggs and carcass tissue, even at times when juvenile salmonids were.
- The addition of carcasses to Grande Ronde tributaries could be an effective way to increase juvenile salmonid production.

### **Temperature Impacts on Prespawn Mortality**

- In September 2017, we submitted a draft manuscript to Transactions of the American Fisheries Society titled “Water temperature and projected climate change impacts on prespawn mortality of Chinook Salmon in Eastern Oregon streams”.
- Our analysis revealed a significant positive effect of average August water temperature on the probability of prespawn mortality (PSM) with average PSM rates increasing from approximately 0.1 at 10 °C to about 0.48 at 18 °C.



- The mean probability of PSM was predicted to increase from approximately 0.06 under current climatic conditions (2000-2013) to approximately 0.12 by the 2040s (102% increase), and to 0.19 by the 2080s (230% increase).
- These findings highlight the need for habitat restoration and other management actions that promote climate change resilience and improve habitat conditions for fish in temperature-impaired watersheds.

### **Climate Change Impacts to Benthic Macroinvertebrates**

- CRITFC has collected benthic and drift macroinvertebrates at CHaMP sites in the upper Grande Ronde River, Catherine Creek, and Minam River from 2011-2017. For analysis of 2011-2016 data, we refer readers to our 2016 annual report. In this year's report, we summarize the conclusions of a three-step climate vulnerability assessment for benthic macroinvertebrates in the Columbia River Basin, in collaboration with the Columbia Basin Partner Forum (CBPF).
- *Habitat suitability* for benthic macroinvertebrate assemblages was proposed as being likely to become marginal, in terms of how climate change will affect exposure, sensitivity, and adaptive capacity; *biotic interactions* within and among benthic macroinvertebrate assemblages including potential shifts in competition, predation, and prey-resource interactions were proposed as being at an elevated risk from climate change; *connectivity* was proposed as being adversely affected by climate change, with continued or increased spatial and temporal isolation within and among benthic macroinvertebrate assemblages.
- Vulnerability to climate change for benthic macroinvertebrate assemblages was rated as "medium-high." We suggest several management strategies with the goal of identifying and protecting existing high-quality habitat while improving the suitability of the entire stream network (including degraded habitats) to support resiliency of macroinvertebrate assemblages.

### **Life Cycle Model**

- We continued development and revisions to a habitat-based spring Chinook Salmon life cycle model in collaboration with Eco Logical Research (ELR). While the model is still under revision, we made several improvements upon the previous version.
- Improvements in this contract period included (a) expanding the model from Catherine Creek to the upper Grande Ronde population, (b) refining differential parr survival rates for fall vs. spring migrants, (c) revising parr capacity estimates to reflect available stock-recruit data, (d) varying spawner capacity estimates based on thermal criteria, and (e) incorporate restoration scenarios from fish-habitat models that were both published and in preparation from our group.

## **Conclusions**

Significant progress has been made over the last nine years in collection of high quality stream habitat and biotic data as well as development of analytical tools needed to quantify status and trends in habitat conditions and fish populations and to evaluate effectiveness of aggregate restoration activities. CHaMP habitat surveys, fish snorkel surveys and benthic macroinvertebrate sampling have been conducted at 160 unique sites (478 site visits) throughout the spring Chinook Salmon distribution area in the upper Grande Ronde, Catherine Creek, and Minam River (in collaboration with Confederated Tribes of the Umatilla Indian Reservation, CTUIR, and Oregon Department of Fish & Wildlife, ODFW), providing precise and spatially referenced data for a large suite of stream habitat and biotic factors. Because fluvial and riparian processes that create fish habitat generally operate over a relatively long time frame (i.e., decades), we do not feel it would be appropriate now to quantify long-term trends in fish habitat conditions. However, the data needed to develop important fish-habitat relationships, habitat-land use relationships, and to parameterize a life cycle model to make projections in fish response to habitat change are now available, and as demonstrated in this report, have been applied successfully to make strides towards meeting project objectives.

## Introduction

The Columbia River Inter-Tribal Fish Commission is conducting a fish habitat monitoring program in the Upper Grande Ronde River and Catherine Creek basins designed to evaluate the effectiveness of aggregate restoration actions in improving freshwater habitat conditions and viability of ESA-listed spring Chinook Salmon populations. A critical uncertainty for fisheries managers in the Columbia Basin involves determining whether freshwater habitat restoration actions will yield a net improvement in basin-wide habitat quality such that remaining human-caused survival impairments elsewhere during the life cycle can be compensated. Through the Columbia Basin Fish Accords, Bonneville Power Administration funds this project and has an interest in determining whether expected improvements in salmon habitat and, thereby, fish production can be brought about by improvements in the quality and quantity of salmon habitat.

Habitat restoration in the Upper Grande Ronde River basin and Catherine Creek basin is being conducted by agencies such as the U.S. Forest Service (e.g., Upper Grande Ronde mine tailings restoration, where channel damage was done by historic dredge mining of the streambed), the Umatilla Tribe (e.g., McCoy Creek Meadows restoration, where natural river meanders are being restored to a channelized stream), the Oregon Department of Fish and Wildlife (ODFW), and the Grande Ronde Model Watershed (e.g., stream channel reconfiguration, riparian fencing and planting, and improvement of irrigation diversions). The U.S. Bureau of Reclamation is also conducting studies of water use and availability in Catherine Creek watershed and may implement projects based on their findings.

There have been many studies conducted in recent years examining the current condition of fish habitat in all the subbasins of the Columbia River. Some of the most common impediments to survival of salmon include high water temperatures, increased concentrations of fine sediment in spawning gravel, loss of riparian vegetation, channelization, loss of large woody debris in the channel, loss of large pools for adult fish holding and juvenile rearing, and summertime depletion of streamflows in the channel. Added to these concerns caused by human influence is climate change, which can lead progressively to changes in the timing of runoff from snowmelt, increased summer air temperatures, and change in the seasonal distribution of precipitation.

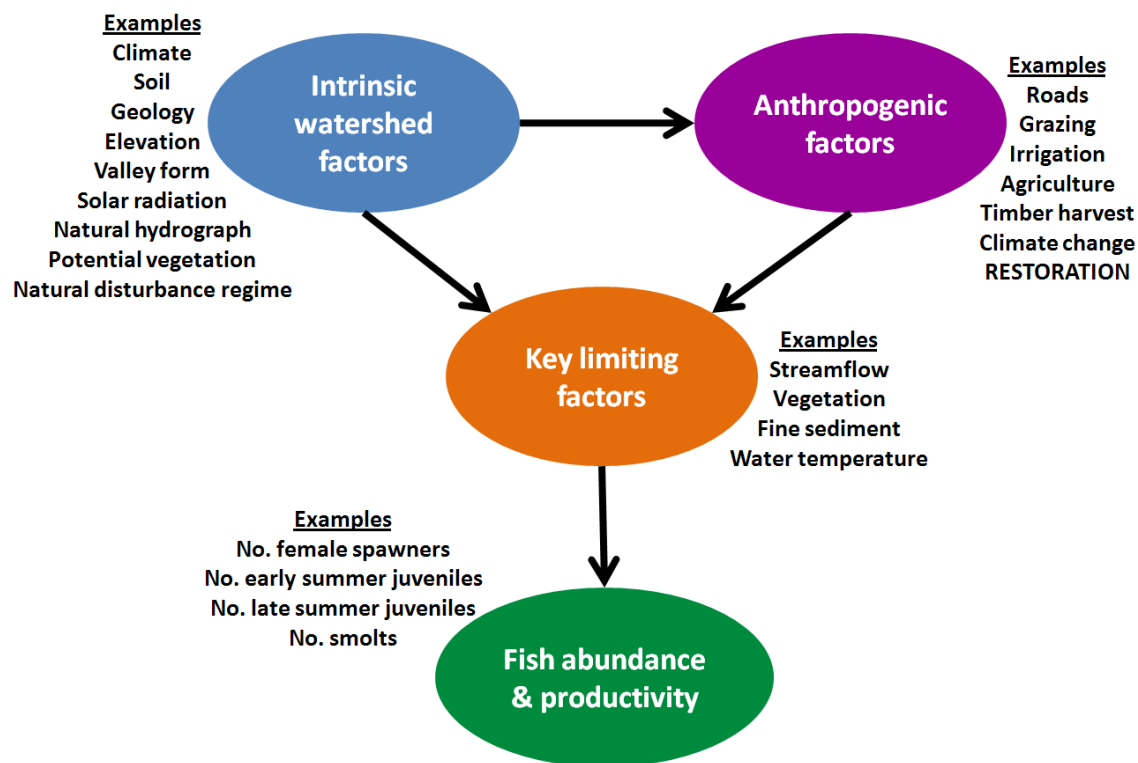
We are attempting to monitor habitat factors that have been identified by previous studies as key limiting factors. Our monitoring plan includes measurement of: water temperature, streambed substrate composition and fine sediment concentrations, streamflow, water chemistry, riparian condition, stream channel morphology (including spawning habitat and large pool distribution), large woody debris, benthic macroinvertebrates (diversity



and density, which indicate long-term water quality), drifting macroinvertebrates (indicating fish food availability), and fish snorkeling (indicating relative abundance of salmonids and qualitative indices of abundance of non-salmonids). Habitat surveys are conducted in 25 sites per year distributed throughout the currently used spring Chinook Salmon spawning and rearing habitat in the Upper Grande Ronde and Catherine Creek basins following methods developed by the Columbia Habitat Monitoring Program (CHaMP; [www.champmonitoring.org](http://www.champmonitoring.org)). In addition, CRITFC funded ODFW to conduct CHaMP surveys in the Minam River from 2013-2015 to provide a set of monitoring sites to act as an unmanaged reference for more heavily disturbed, managed sites in the Upper Grande Ronde and Catherine Creek.

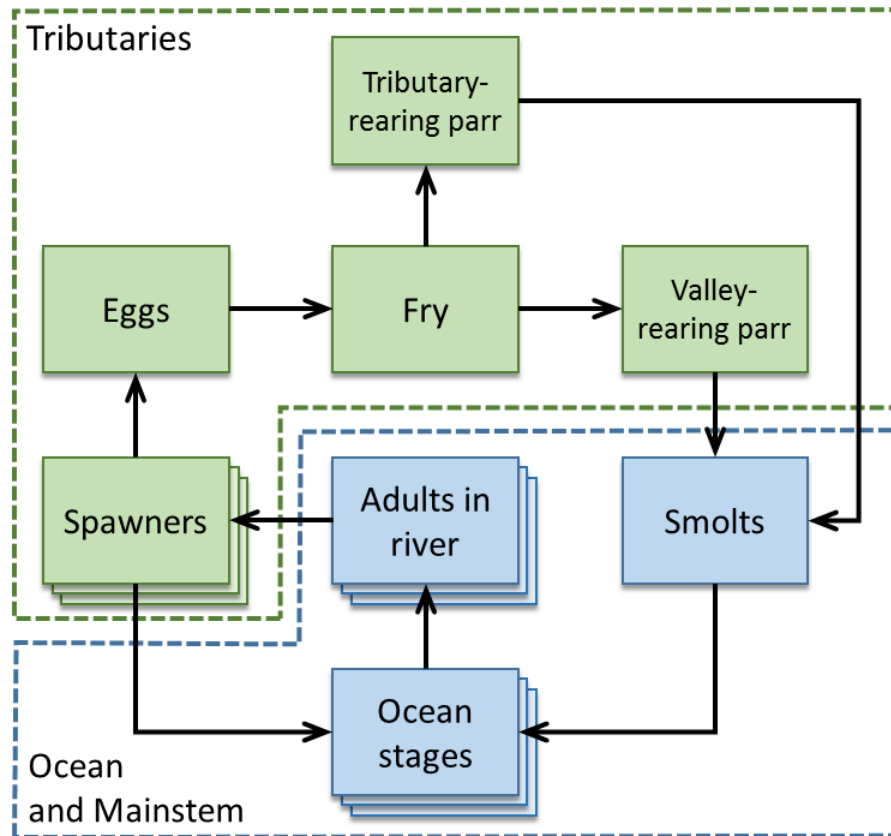
As a means to integrate habitat monitoring efforts with recovery planning, CRITFC is developing a life cycle model. The life cycle model is being designed as a tool to simulate population trends in relation to projected environmental conditions, and to examine the relative benefits of habitat improvements on population recovery potential. Initially, a prototype model was developed which includes all Chinook Salmon populations in the Grande Ronde/Imnaha Major Population Group (MPG) of the Snake River Evolutionarily Significant Unit (ESU), which consists of the Grande Ronde River, Catherine Creek, Lostine/Wallowa, Minam, Wenaha, and Imnaha. That model focused on empirical validation of the survival of outmigrating juveniles and returning adults, with particular focus on the environmental and operational variables that influence survival through the hydro system and ocean. This prototype model provides a strong empirical basis for further refinements of the life cycle model to include effects of freshwater habitat conditions on productivity and abundance of rearing juvenile salmonids.

Future refinement of the life cycle model and the research needed to parameterize the model are guided by a basic conceptual model framework (Figure 1). The fundamental basis of the model is that intrinsic watershed factors (such as geology, climate, or valley morphology) interact with human actions (such as forest harvest, cattle grazing, or stream restoration) to affect processes that drive known limiting factors (e.g., flow, temperature, pool area, etc.), and therefore fish survival via both density-dependent and density-independent processes. Current and future habitat conditions can act as predictors of relative change in survival at different life history stages, and therefore affect recovery potential.



**Figure 1.** Conceptual framework for relating fish abundance and productivity to watershed characteristics and limiting factors. Direction of arrows indicates direction of influence.

The Upper Grande Ronde/Catherine Creek life cycle model for spring Chinook Salmon (Figure 2) is comprised of several interacting subcomponents that are built independently, each of which provides critical information about the interaction between landscape characteristics, instream habitat conditions, and fish response. Individual subcomponents of the life cycle model, and of our research program, include modeling stream temperature from local riparian and geomorphic conditions; a water temperature model that links summer water temperature metrics to summer juvenile survival; a potential natural vegetation map for estimating stream shade; rates of restoration project addition of large woody debris; macroinvertebrate inputs for estimating the food base and growth potential for drift-feeding salmonids; and a low flow model permitting evaluation of site vulnerability to climatic variations in precipitation, snow melt timing, and air temperature variations. Though these components are described here as potentially providing inputs to the life cycle model, each model is a valid research project that is likely to yield interesting scientific insights and practical applications for conservation.



**Figure 2.** Conceptual diagram of the life cycle model (LCM) framework used to assess restoration scenarios in Catherine Creek. Stacked boxes represent life stages for which multiple ages exist and are tracked accordingly. In our LCM, parr can ‘choose’ to migrate downstream and overwinter in the Grande Ronde Valley (‘Valley’) or remain in the upper mainstem and tributaries of Catherine Creek (‘Tributary’). Otherwise, the LCM’s smolts are enumerated at Lower Granite Dam (LGD), where ‘ocean stages’ begin and are modeled via LGD-to-LGD smolt-to-adult return rates. Fish-habitat modeling described in this report provides capacity estimates for spawner and parr freshwater life stages.

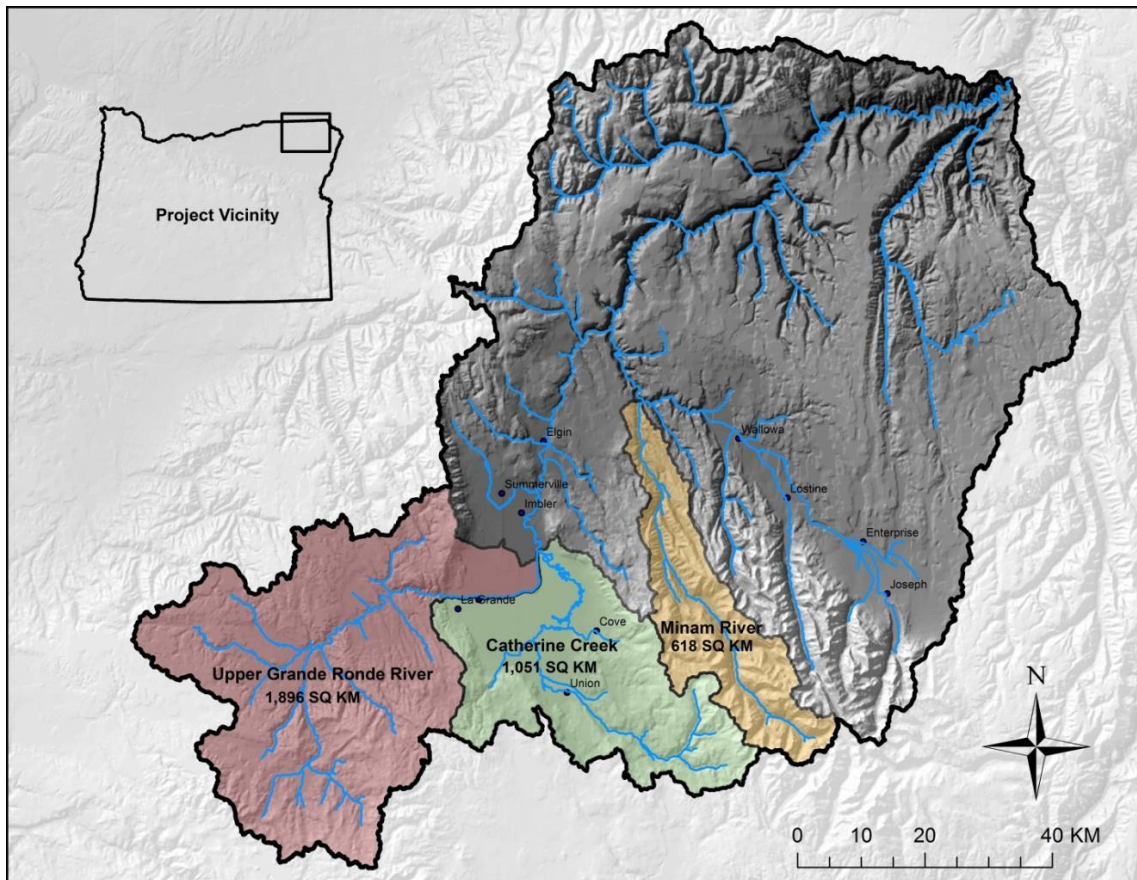
The primary objectives of this project are to: 1) Assess current status and trends in fish habitat characteristics considered to be key limiting factors (particularly water temperature, pool habitats, streamflow, and fine sediment) to viability of spring Chinook Salmon populations; 2) Evaluate effectiveness of aggregate stream restoration actions aimed at improving key limiting habitat factors; and 3) Develop a life cycle model to link biotic responses of spring Chinook populations to projected changes in stream habitat conditions.



## Methods

### Study Area

This study is being conducted in the Grande Ronde River and its tributaries, which originates in the Blue Mountains of NE Oregon and flows 334 km to its confluence with Snake River near the town of Rogersburg, Washington (Figure 3). Focal watersheds include the upper Grande Ronde River above the town of La Grande, Catherine Creek, and to a lesser extent, the Minam River, which drain areas of approximately 1,896, 1,051, and 618 km<sup>2</sup> respectively.



**Figure 3.** Study area in the Grande Ronde River basin, NE Oregon. Focal watersheds include the Upper Grande Ronde River, Catherine Creek, and Minam River. The Upper Grande Ronde and Catherine Creek are the basins with significantly damaged habitat that is undergoing restoration in various locations. The Minam River basin is the local reference basin that has far less current evidence of human impact.

The topography of the upper portion of the subbasin (i.e., upstream of the Wallowa River confluence) is characterized by rugged mountains in the headwater areas and a broad, low gradient valley between the Blue and Wallowa Mountains. Peaks in the Wallowa Mountains reach a maximum elevation of 2,999 m, and provide the source of many of the

Grande Ronde's tributaries including Catherine Creek and the Wallowa River. The Blue Mountains reach elevations of 2,347 m and are the source of the Grande Ronde River, Wenaha River, and other tributaries. Due to the lower elevation of the Blue Mountains, snow melt generally occurs earlier in these tributaries, often resulting in very low flows during summer.

Surface geology of the Grande Ronde subbasin is dominated by rocks of the Columbia River Basalt group, with some older granitic intrusives and volcanics with associated sedimentary deposits present in the headwater areas of the Upper Grande Ronde and Catherine Creek. The climate is characterized by cold, moist winters and warm, dry summers with mean daily air temperatures near La Grande averaging  $-0.42^{\circ}\text{C}$  in January and  $21^{\circ}\text{C}$  in July. Average annual precipitation ranges from 36 cm in the valleys to 152 cm in the mountains, with most of the precipitation in the mountains falling as winter snow.

The vegetation community at lower elevations is dominated by grasslands consisting of Idaho fescue/bluebunch wheatgrass (*Festuca idahoensis*-*Agropyron spicatum*) and bluebunch wheatgrass-Sandberg's bluegrass (*Agropyron spicatum*-*Poa sandbergii*) (Nowak 2004). As elevation increases, the grasslands transition to shrub/scrub plants, and eventually to coniferous forests in the mountains. Forest species consist of low elevation Ponderosa pine (*Pinus ponderosa*) and lodgepole pine (*Pinus contorta*) associations grading into Douglas-fir (*Pseudotsuga menziesii*), grand fir (*Abies grandis*), subalpine fir (*Abies lasiocarpa*), and mountain hemlock (*Tsuga mertensiana*) associations at higher elevations. Riparian vegetation is dominated by black cottonwood (*Populus trichocarpa*) and willow (*Salix* spp.), black hawthorn (*Crataegus douglasii*), mountain alder (*Alnus incana*), and mountain maple (*Acer glabrum*).

Approximately 49% of the land in the Grande Ronde basin is publicly owned, of which about 97% is managed by the US Forest Service. The remaining public land is managed by the Bureau of Land Management and the States of Oregon and Washington. Excepting the Eagle Cap and Wenaha-Tucannon Wilderness Areas, the National Forests are managed for multiple use including timber production, livestock grazing, and recreation. Private property comprises 51% of the land in the basin and is located primarily in lower elevation valleys and along rivers. A large proportion of the private property is used for agriculture including crop production, livestock grazing, and forestry. Only 0.1% of the land in the Grande Ronde Basin is currently owned by the tribes, although the tribes retain fishing and hunting access rights at all usual and accustomed locations as afforded under the treaties of 1855 and 1863.

Spring Chinook populations in these basins were listed as threatened under the Endangered Species Act in 1992. Population declines over the past century were due in

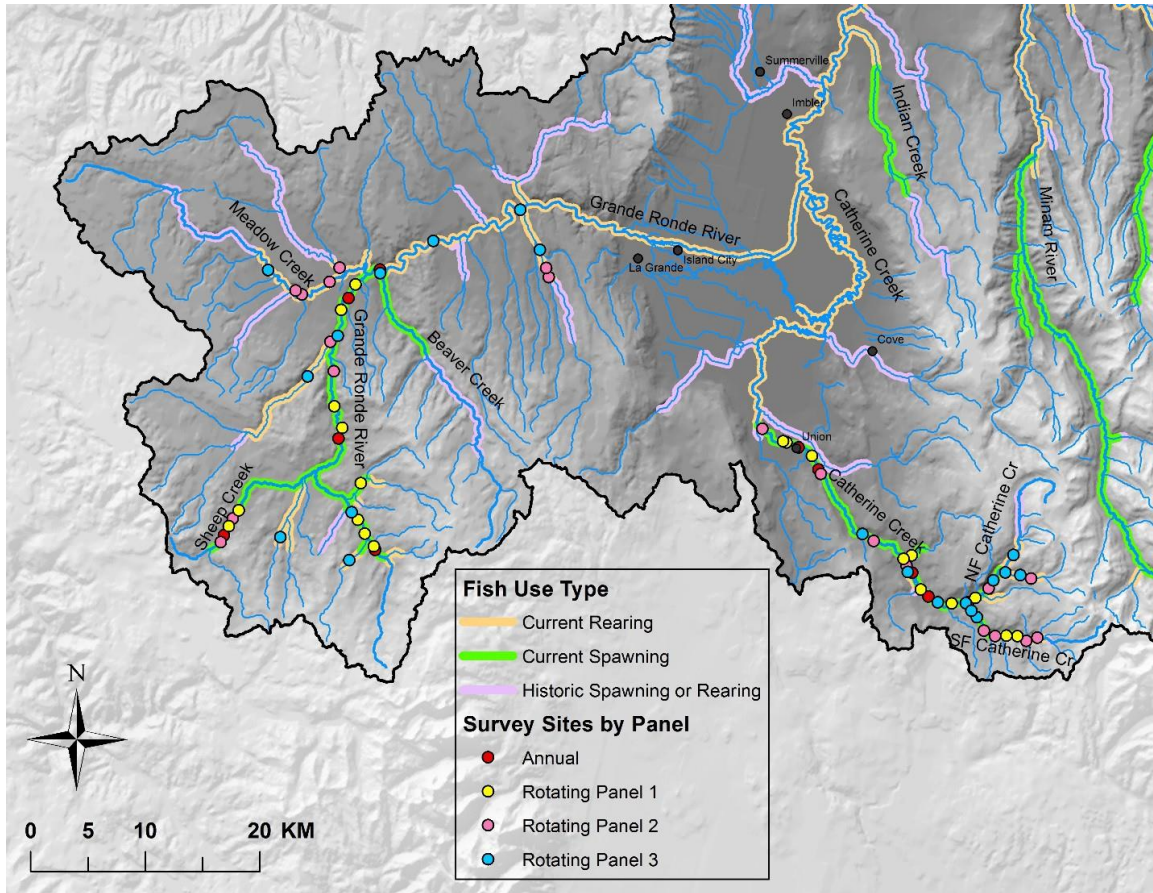
part to severely degraded habitat conditions resulting from intensive anthropogenic disturbances including timber harvest, cattle grazing, levee and road construction, and stream diversions for irrigation. Specifically, stream temperature, streamflow, fine sediment, habitat diversity, and quantity of key habitats such as large pools, have been identified as key limiting factors for recovery of Chinook populations in these basins (Nowak 2004).

## **Stream Habitat**

### **CHaMP Data Collection**

In 2017 we continued to implement the Columbia Habitat Monitoring Program (CHaMP) protocol and sampled 25 sites in the upper Grande Ronde and Catherine Creeks basins. When combined with CHaMP sites surveyed by the Confederated Tribes of the Umatilla Indian Reservation (CTUIR) and ODFW, including the Minam basin, a total of 160 unique sites (478 site visits) have been surveyed in the Grande Ronde basin. CHaMP is designed as a Columbia River basin-wide habitat status and trends monitoring program built around a single protocol with a programmatic approach to data collection and management. CHaMP will result in the collection and analysis of systematic habitat status and trends information that will be used to assess basin-wide habitat conditions and characterize stream responses to watershed restoration and/or management actions. A detailed description of the CHaMP protocol is provided at <http://www.monitoringmethods.org/Protocol/Details/806>.

Survey sites were randomly selected using the Generalized Random Tessellation Stratified (GRTS) survey design (Stevens and Olsen 2004), and were distributed with equal probability across all wadable portions of the stream network that were classified as current or historic spawning and rearing areas for spring Chinook Salmon (Figure 4). The spring Chinook distribution area was modified from maps produced by ODFW, StreamNet (StreamNet 2009), the Grande Ronde Subbasin plan (Nowak 2004), and the NOAA's Interior Columbia Basin Technical Recovery Team (ICTRT), and is described in more detail in McCullough et al. (2012).



**Figure 4.** Sample design showing the distribution of CRITFC habitat survey sites in the upper Grande Ronde River and Catherine Creek basins.

**Table 1.** Rotating panel design for CHaMP sites in the Grande Ronde River and Catherine Creek watersheds. Note that 5 of the 30 sites in our survey design are sampled by ODFW each year.

	Year								
Panel	2011	2012	2013	2014	2015	2016	2017	2018	2019
<b>Grande Ronde Chinook</b>									
Annual Panel	5	5	5	5	5	5	5	5	5
Rotating Panel 1	10			10			10		
Rotating Panel 2		10			10			10	
Rotating Panel 3			10			10			10
<b>Catherine Creek Chinook</b>									
Annual Panel	5	5	5	5	5	5	5	5	5
Rotating Panel 1	10			10			10		
Rotating Panel 2		10			10			10	
Rotating Panel 3			10			10			10
<b>Total Annual Samples</b>	30	30	30	30	30	30	30	30	30
<b>Total Unique Samples</b>	70								

We used a 3-year rotating panel design (Figure 4) with the intention of achieving a good balance between power to describe current status (i.e., accurate description of spatial variation across the entire sampling extent) and power to detect trends over time. This temporal design includes 5 annual sites and 10 rotating panel sites in the upper Grande

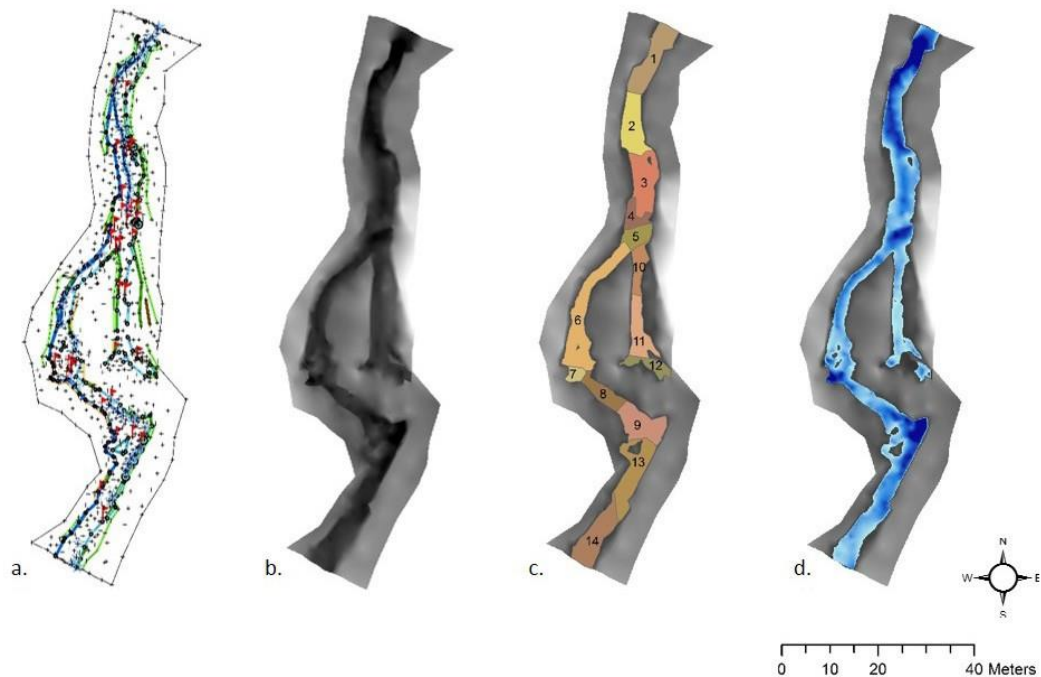
Ronde and Catherine Creek basins. Annual sites are surveyed every year and rotating panel sites are surveyed every 3 years. A total of 30 sites are surveyed each year, with a total sample size of 70 unique sites after 3 years. Note that 5 of the 30 sites within CRITFC's target sample frame are surveyed by ODFW each year. The 2017 field season was the seventh year of implementing the protocol in the upper Grande Ronde watershed. This year's surveys were the seventh visit to annual sites and the third revisit to panel 1 sites since the start of the program in 2011.

Dissimilarly to the previous monitoring seasons (2013 - 2015), CHaMP data were not collected in the Minam River watershed. These data, which were collected in collaboration with the Oregon Department of Fish and Wildlife, were selected as suitable reference site locations that could be used as a comparison to the two more significantly disturbed watersheds (i.e., Catherine Creek and Upper Grande Ronde River watersheds).

A large number of stream habitat variables were measured at each site, generating over 100 metrics describing the condition of the stream. Most of the variables measured were chosen because they are directly related to salmonid fish growth and/or survival or because they provide critical information used to describe ecological processes in the stream or broader landscape that may be indirectly related to fish productivity. The measurements were collected using a combination of traditional habitat data collection methods along with use of technical survey equipment (e.g., Total Stations) that allows for the development of detailed topographic maps of the stream channel. A complete list of metrics generated by CHaMP surveys is available at <https://www.champmonitoring.org/>.

Topographic data are composed of deliberately placed coded points and lines in the stream channel and floodplain that are used to illustrate inflection points in the channel bedform such as pools, the toes and tops of banks, and the thalweg (the location with the highest streamflow) along with important features such as the edge of water, bankfull elevations, and channel unit boundaries (Figure 5, a.). Depending on the channel length and complexity, crews capture between 700 and 1500 points to accurately represent a stream reach. The raw data were imported into ArcGIS and processed to create digital elevation models (DEM), channel unit delineations, and water depth maps (Figure 5, b. - d.). These data were evaluated for quality control during the initial processing that occurs directly after the survey. After the field season is completed, the output products from each survey were processed through the River Bathymetry Toolkit (RBT) to derive metrics that are potentially important to fish, such as residual pool depth. After the RBT metrics were created, a second round of quality control analysis is performed to ensure the accuracy of the topographic data. These data were then made available to the public on the program's website ([www.champmonitoring.org](http://www.champmonitoring.org)).



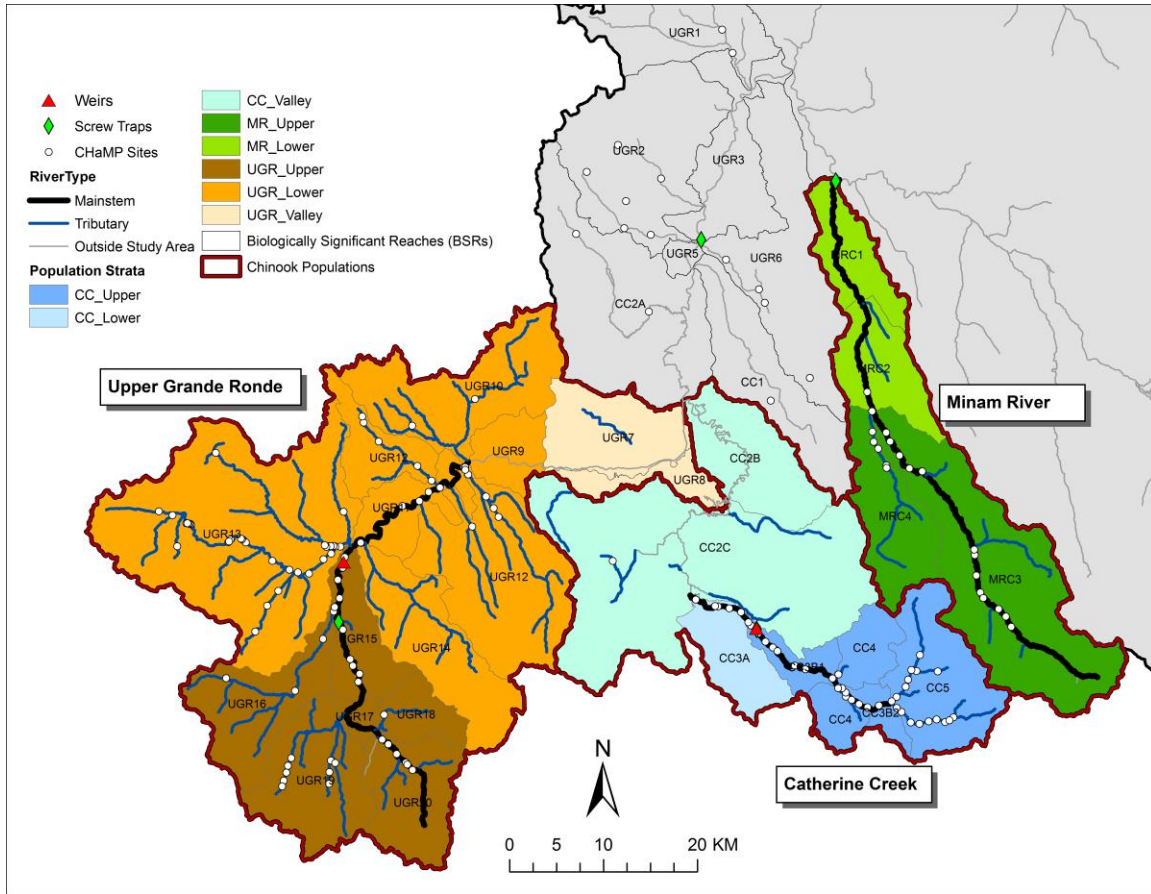


**Figure 5.** Example of data from a topographic survey conducted on Sheep Creek, site CBW05583-490810 using the CHaMP protocol and a Total Station. The panel depict a) the raw point and line data as surveyed in the field, b) the digital elevation model (DEM), c) channel units overlaid on the DEM, and d) the water depth map produced by analysis at each CHaMP site.

These detailed surveys will become increasingly powerful as the CHaMP program progresses because these surveys can be repeated and overlaid to inform topographic change over time. We have just completed the seventh year of the nine year study design and have started the process of revisiting panel sites. At the end of the nine year period we will have a minimum of three visits at each site and will be able to quantify the degree of channel and habitat change over time at individual sites and start to look at patterns of channel erosion and aggradation across the watershed.

Selected habitat metrics important for Chinook Salmon at various life history stages (percentage area pools, large and total pools per km, total fish cover, large woody debris frequency, average august water temperature, and maximum 7-day average maximum water temperature) were summarized for all sites across all years by Chinook population group (upper Grande Ronde, Catherine Creek, and Minam), by biologically significant reaches (BSRs), and population stratification within each of the three population groupings (lower, middle, upper). BSR boundaries correspond approximately to the HUC6 watershed boundaries, but were aggregated or modified to better represent significant breaks in physical channel morphology (e.g., tributary junctions or major changes in valley confinement), land ownership, or fish use. BSRs were developed by

local experts associated with the Federal Columbia River Power System (FCRPS) Biological Opinion expert panel and restoration implementation group (i.e., Atlas Group) for use in planning and implementing stream restoration actions. Chinook population group classifications are based on the historic and current range extents and are further refined by their location (lower, middle, upper) within each grouping (Figure 6).



**Figure 6.** Map showing the spring Chinook population boundaries, CRITFC population strata, and biologically significant reaches (BSR) within the CRITFC study area.

Year-round temperature records were collected at all CHaMP sites in the three focal watersheds using Hobo Tidbit and Pro V2 data loggers following the Water Temperature Probe Installation Protocol (<https://www.monitoringmethods.org/Method/Details/846> ). Sites were visited once in both spring and fall to download the data and to assess the condition of the temperature loggers. In addition to the sites monitored through the CHaMP protocol, CRITFC has temperature loggers deployed at 12 other locations in the Catherine Creek and upper Grande Ronde watersheds. These loggers fill in areas where temperature data are not sufficiently captured by the CHaMP program and often are located up- and downstream of significant tributary junctions. Data validation and storage for non-CHaMP sites was conducted by CRITFC staff and summary statistics were



calculated to match those metrics derived from the CHaMP data. This database will continue to be updated each year as more temperature data are collected in the field.

Temperature metrics were based on a “summer period” as defined by CHaMP, which is between July 15<sup>th</sup> and Aug 31<sup>st</sup>. This timeframe is crucial due to the potentially detrimental influences of high temperatures on salmonids. For every consecutive 7-day period within this 48-day period, rolling averages of maximum temperatures were calculated (7dAM). The number of 7dAM periods where the average maximum temperature exceeds 12°, 13°, 16°, 18°, 20°, and 22°C was evaluated at every site. These water temperature standards were designated based on beneficial uses by various salmonid life history stages as defined by the Oregon Department of Environmental Quality (Sturdevant 2008). Additionally, the average daily temperature, maximum daily temperature, and the maximum 7-day rolling average of maximum temperature (Max7dAM) are reported over the same summer time period. Monthly statistics, average and maximum temperatures, were also calculated. We have reported on the Max7dAM and Average August temperatures as they are used regionally to summarize stream temperatures and would allow for the most direct crossover between monitoring programs.

In future analyses we plan to extrapolate fish density estimates to include un-sampled portions of the watershed by using the combination of the BSR regions and Chinook population group. Using GRTS we can compute population estimates based on population stratification and then calculate BSR-wide estimates based on the proportional stream length of each classification type. We can use the classification to more accurately extrapolate population metrics to BSR reaches where we have few or no CHaMP sites based on stream type.

## **Results and Discussion**

A summary of eight important CHaMP habitat metrics including total fish cover (m<sup>2</sup>), bankfull and wetted large woody debris (LWD) frequency (count/100 m), percentage slow water, large pools and total pools (pools/km), average August water temperature (°C), and the maximum 7-day running average maximum water temperature (Max7dAM, °C) within each Chinook population group, population stratification, and Biologically Significant Reaches (BSR) is provided (Table 2). These data include sites sampled by both CRITFC and ODFW within the historic Chinook habitat extent, which encompassed 124 unique sites (n). A Generalized Random Tessellation Stratified (GRTS) sample approach was used to average the data for each year and across all years (2011-2017). These data were then averaged and standard deviations were summarized across the various spatial units.

The eight metrics for instream habitat supporting the three ICTRT populations (Catherine Creek, Minam River, and Upper Grande Ronde) reveal important variations at scales of the entire population-specific basin, the component BSRs, and population strata. At the coarsest scale (ICTRT populations), within year and between year variability was large for each of the measured physical habitat metrics (Figure 7); most notably for both bankfull and wetted LWD frequency. With the exception of large and total pools/km, the highest values of the reported metrics were observed within the Upper Grande Ronde ICTRT population strata. Interestingly, though the Grande Ronde had the highest percentage of slow water, it also had the least large and total pools compared to Catherine Creek and the Minam River (Figure 8). Though this is unusual, it is likely a remnant of the tiered structure of instream habitat classification whereby fast water non-turbulent (i.e., glide) habitat also fall under the slow water tier of classification. In this instance, the proportion of slow water area classified as fast water non-turbulent would be larger than that of pool habitat for the Grande Ronde.

Of further importance are the distribution, size, and number of pools. Whereas, the Grande Ronde had the lowest large pools and total pools/km, surprisingly the percentage of slow water was higher in the upper strata of the Grande Ronde, particularly for tributaries, than the lower. Compared to both Catherine Creek and the Minam River, which have upper mainstem and tributaries of higher gradient and stream power, the percentage of slow water, and thus pools, were primarily found on mainstem reaches. Additionally, it was surprising that at the ICTRT population spatial scale, the Minam River has the lowest total fish cover (%) and LWD frequency. However, when evaluating these metrics at the population strata scale for these metrics, the Minam was only slightly above Catherine Creek for upper mainstem and upper tributary strata.

At the population strata scale the influence of small typically colder water tributaries, compared to mainstem reaches, can help distinguish some of the observed variation in each metric at the ICTRT scale. At the population strata scale, water temperatures are lower in the upper portions of each strata (i.e., upper mainstem and upper tributary) compared to the lower strata. Though timeseries data are limited for the Minam river (Figure 7) it had the lowest upper mainstem average August and Max7dAM water temperatures, while both Catherine Creek and Upper Grande Ronde strata are nearly equal. In comparison to the upper tributary strata, Catherine Creek water temperatures were lowest while the Upper Grande Ronde were highest. Overall, despite the influence of relatively cooler tributaries to the lower section of the UGR mainstem (e.g., Five Points Creek, Beaver Creek, and upper McCoy Creek) the change in temperature from the upper strata to lower was twice that of Catherine Creek.

The finer spatial resolution of the BSRs highlights particular portions of the two subbasins where extreme temperatures are detrimental to Chinook Salmon. Four of the

five Catherine Creek BSRs exceed the ODEQ beneficial use threshold for juvenile salmonid rearing of 18°C 7dAM for a minimum of one 7-day period, based on the Max7dAM. Additionally, of these four BSRs, two exceeded the 20°C threshold set for migration corridors for adult salmon for a minimum of a 7-day period. In the Grande Ronde subbasin all 7 BSRs exceeded the juvenile rearing and the adult migration 7dAM temperature. The maximum temperature exceedance based on the Max7dAM metric occurred at BSR UGR11, showing average temperatures across the study period exceeding 27°C. By breaking the subbasins down into smaller spatial extents we can identify areas of high risk to Chinook Salmon and focus restoration efforts within particular reaches. Prime examples of which are BSRs UGR15 and UGR16, which show anomalous data that don't follow the expected trend moving downstream in the subbasin. However, from further investigation, the root causes of identified risk in these BSRs are significant and stark shifts in topography. Here the landscape flattens considerably and the river meanders through open meadow habitat with minimal vegetation for cover or for contributing large wood.

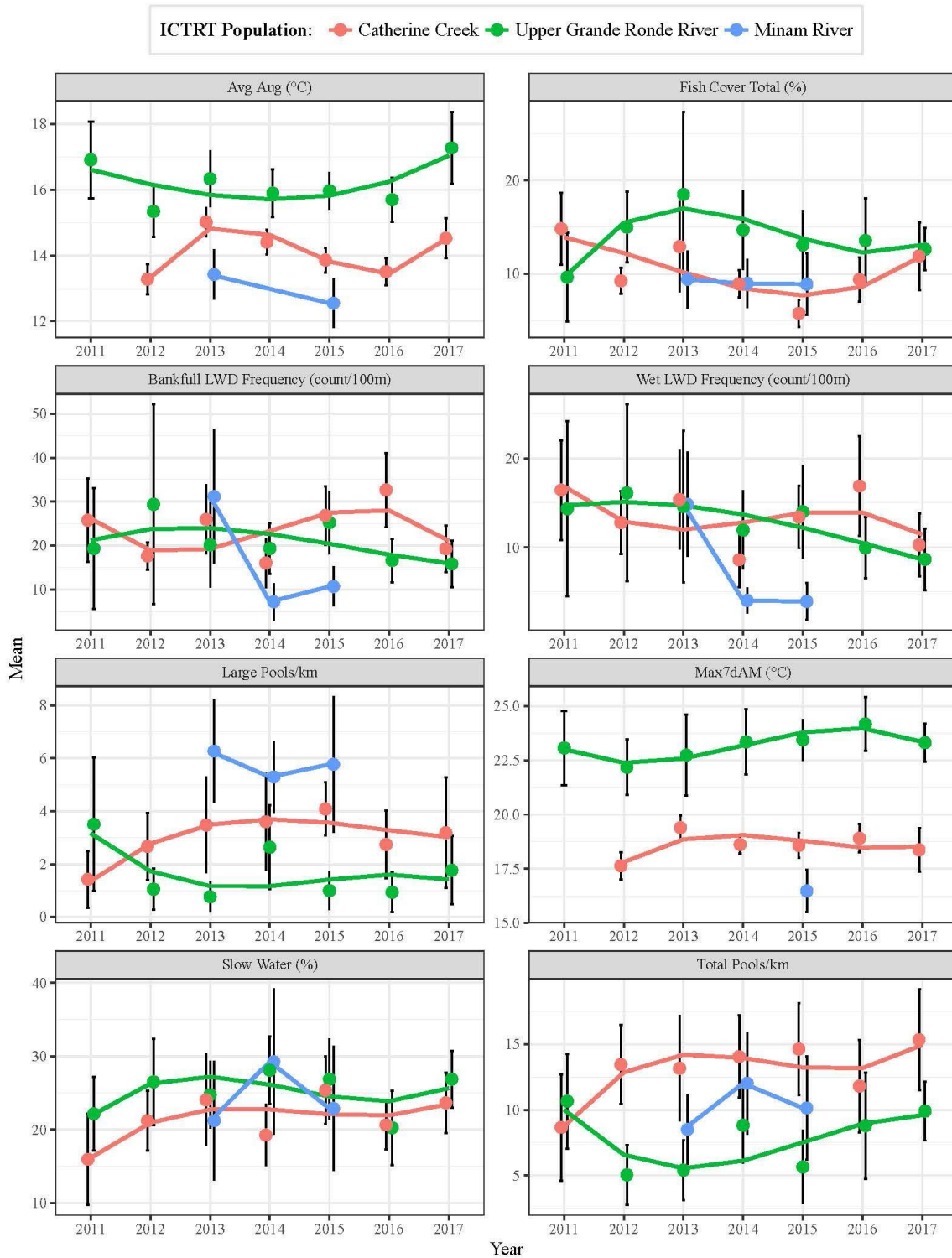
In future modeling efforts we plan to use GRTS to summarize data by population stratification and then use distinct river segments in each BSR to roll-up metrics based on the proportion of each class in a BSR. This will allow us to downscale our GRTS estimates from the whole watershed scale to a finer resolution (e.g., BSR) while leveraging knowledge of habitat data for known historic and current Chinook population ranges. The continued exploration of the trends across spatial extents will further our understanding of how habitat metrics are changing within drainages. We hope to explore how this method may improve our estimates and extrapolation of habitat variables and fish populations.

**Table 2.** Summary statistics for eight habitat variables measured at all CHaMP sites across the Upper Grande Ronde River, Catherine Creek, and the Minam River. GRTS sample values for each site were averaged over all survey years, 2011-2017 and then summarized based on three different spatial extents: Interior Columbia Technical Recovery Team (ICTRT) populations, biologically significant reaches (BSR), and population strata (PopStrata). For each habitat variable, mean, standard deviation (SD), and the number of sites that had data values (n) are recorded.

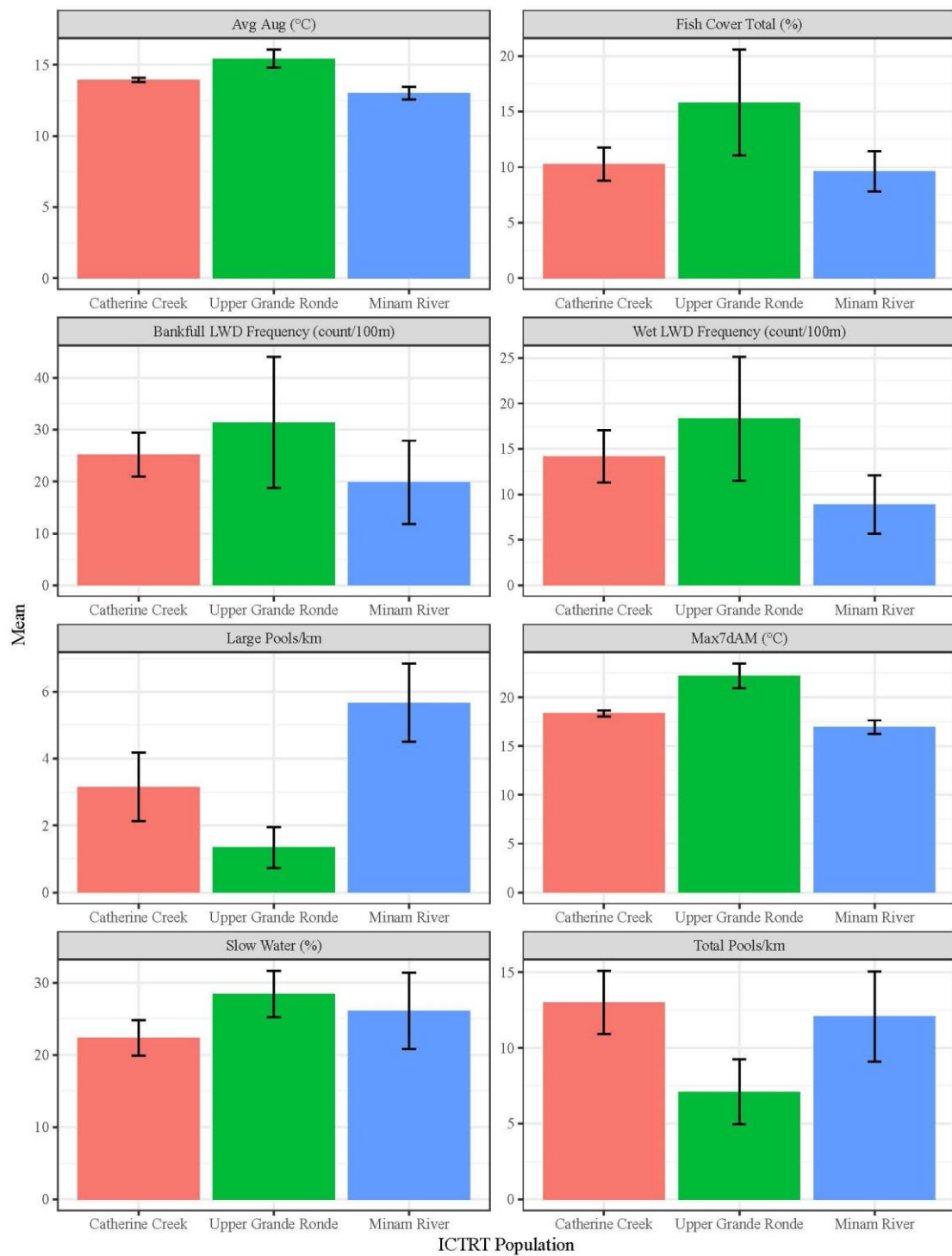
Spatial Extent	Average Aug Temp (°C)			Total Fish Cover (%)			Bankfull LWD Frequency (count/100 m)			Wet LWD Frequency (count/100 m)		
ICTRT Populations	Mean	SD	n	Mean	SD	n	Mean	SD	n	Mean	SD	n
Catherine Creek	13.93	2.51	35	10.28	5.35	40	25.20	15.79	40	14.19	10.53	40
Minam River	13.00	0.59	6	9.62	4.54	20	19.85	20.22	20	8.89	7.93	20
Upper Grande Ronde River	15.42	2.23	47	15.81	11.94	64	31.40	29.87	64	18.33	18.45	64
Biologically Significant Reach	Mean	SD	n	Mean	SD	n	Mean	SD	n	Mean	SD	n
CC3A	17.70	0.35	4	10.53	2.87	4	14.37	10.46	4	8.50	8.42	4
CC3B1	16.96	0.70	6	8.44	3.05	9	22.00	16.36	9	12.43	8.99	9
CC3B2	14.32	0.66	7	7.43	3.94	7	24.72	11.54	7	12.69	6.36	7
CC4	14.13	0.43	3	21.55	7.10	3	32.50	8.35	3	19.44	6.71	3
CC5	11.49	0.89	15	10.37	4.21	17	28.34	17.30	17	16.15	12.57	17
MRC3	12.94	0.05	2	8.59	3.31	13	18.46	21.90	13	8.20	7.80	13
MRC4	13.03	0.72	4	13.44	3.57	6	25.09	16.19	6	11.74	7.58	6
UGR11	19.42	0.75	4	6.64	2.61	5	1.14	0.76	5	0.56	0.58	5
UGR12	15.83	1.44	5	11.41	2.70	7	14.60	11.34	7	7.87	6.61	7
UGR13	15.19	2.04	13	19.21	12.36	23	41.57	32.94	23	23.47	19.64	23
UGR15	17.26	0.56	8	7.73	4.02	9	15.68	10.88	9	8.51	6.37	9
UGR16	16.33	0.79	3	7.08	2.08	4	10.90	1.35	4	5.12	1.88	4
UGR19	13.81	1.02	7	22.59	17.56	8	27.14	11.74	8	17.55	7.95	8
UGR20	12.67	0.68	6	16.48	7.41	6	53.92	34.56	6	36.28	24.48	6
Population Strata	Mean	SD	n	Mean	SD	n	Mean	SD	n	Mean	SD	n
CC_Lower_Mainstem	17.70	0.35	4	10.53	2.87	4	14.37	10.46	4	8.50	8.42	4
CC_Upper_Mainstem	15.54	1.48	13	8.00	3.50	16	23.19	14.51	16	12.54	7.95	16
CC_Upper_Tributary	11.93	1.29	18	12.05	6.21	20	28.97	16.34	20	16.65	11.93	20
MR_Upper_Mainstem	12.94	0.05	2	8.59	3.31	13	18.46	21.90	13	8.20	7.80	13
MR_Upper_Tributary	13.03	0.72	4	13.44	3.57	6	25.09	16.19	6	11.74	7.58	6
UGR_Lower_Mainstem	19.42	0.75	4	6.64	2.61	5	1.14	0.76	5	0.56	0.58	5
UGR_Lower_Tributary	15.29	1.97	18	17.80	11.77	31	36.91	32.10	31	20.74	19.10	31
UGR_Upper_Mainstem	15.29	2.35	14	11.23	7.08	15	30.97	29.99	15	19.62	21.19	15
UGR_Upper_Tributary	14.47	1.66	11	16.94	15.48	13	22.97	12.89	13	14.83	10.18	13

**Table 2.** Continued.

Spatial Extent	Large Pools/km			Max7dAM (°C)			Slow Water (%)			Total Pools/km		
ICTRT Populations	Mean	SD	n	Mean	SD	n	Mean	SD	n	Mean	SD	n
Catherine Creek	3.15	3.83	40	18.36	3.12	35	22.35	10.68	40	13.01	7.72	40
Minam River	5.68	4.64	20	16.95	0.90	5	26.06	15.88	20	12.08	8.36	20
Upper Grande Ronde River	1.34	3.00	64	22.19	3.65	46	28.44	15.38	64	7.10	9.12	64
<b>Biologically Significant Reach</b>	<b>Mean</b>	<b>SD</b>	<b>n</b>	<b>Mean</b>	<b>SD</b>	<b>n</b>	<b>Mean</b>	<b>SD</b>	<b>n</b>	<b>Mean</b>	<b>SD</b>	<b>n</b>
CC3A	8.50	8.42	4	22.20	0.22	4	35.07	10.32	4	9.56	4.98	4
CC3B1	12.43	8.99	9	22.26	0.23	6	21.24	8.65	9	11.95	4.33	9
CC3B2	12.69	6.36	7	19.25	0.96	7	25.93	13.65	7	17.10	8.24	7
CC4	19.44	6.71	3	18.81	1.70	3	28.63	9.98	3	6.16	8.72	3
CC5	16.15	12.57	17	15.28	1.31	15	17.37	5.81	17	13.89	7.99	17
MRC3	8.20	7.80	13	16.53	0.79	3	30.42	14.15	13	12.46	7.90	13
MRC4	11.74	7.58	6	17.59	0.64	2	20.95	15.05	6	13.27	8.48	6
UGR11	0.56	0.58	5	27.40	1.21	4	12.56	9.96	5	4.46	3.34	5
UGR12	7.87	6.61	7	24.50	2.30	5	16.58	4.43	7	4.78	4.02	7
UGR13	23.47	19.64	23	21.07	3.69	11	31.46	12.31	23	4.29	7.10	23
UGR15	8.51	6.37	9	24.87	1.24	8	17.40	7.87	9	8.84	4.25	9
UGR16	5.12	1.88	4	24.86	1.19	3	17.12	1.45	4	5.72	3.44	4
UGR19	17.55	7.95	8	20.39	1.98	7	47.33	18.03	8	9.18	9.12	8
UGR20	36.28	24.48	6	18.02	1.09	6	32.18	16.05	6	25.47	12.64	6
<b>Population Strata</b>	<b>Mean</b>	<b>SD</b>	<b>n</b>	<b>Mean</b>	<b>SD</b>	<b>n</b>	<b>Mean</b>	<b>SD</b>	<b>n</b>	<b>Mean</b>	<b>SD</b>	<b>n</b>
CC_Lower_Mainstem	5.99	5.08	4	22.20	0.22	4	35.07	10.32	4	9.56	4.98	4
CC_Upper_Mainstem	4.60	3.27	16	20.64	1.66	13	23.29	11.36	16	14.21	6.84	16
CC_Upper_Tributary	1.42	3.05	20	15.87	1.91	18	19.06	7.74	20	12.73	8.56	20
MR_Upper_Mainstem	7.55	4.56	13	16.53	0.79	3	30.42	14.15	13	12.46	7.90	13
MR_Upper_Tributary	2.56	2.00	6	17.59	0.64	2	20.95	15.05	6	13.27	8.48	6
UGR_Lower_Mainstem	2.10	2.09	5	27.40	1.21	4	12.56	9.96	5	4.46	3.34	5
UGR_Lower_Tributary	1.00	3.00	31	21.90	3.54	17	29.03	12.60	31	4.34	6.67	31
UGR_Upper_Mainstem	3.34	3.57	15	21.93	3.59	14	23.31	13.88	15	15.49	11.88	15
UGR_Upper_Tributary	0.15	0.53	13	21.48	3.11	11	38.11	20.21	13	9.35	8.96	13



**Figure 7.** GRTS sample mean summary statistics for eight habitat variables measured at all CHaMP sites across the Upper Grande Ronde River, Catherine Creek, and the Minam River for years 2011-2017 based on Interior Columbia Technical Recovery Team (ICTRT) populations.



**Figure 8.** GRTS sample mean summary statistics for eight habitat variables measured at all CHaMP sites across the Upper Grande Ronde River, Catherine Creek, and the Minam River averaged across all years (2011-2017) based on Interior Columbia Technical Recovery Team (ICTRT) populations.



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## **Stream Temperature Modeling**

In March 2017 we published an article in the Journal of Environmental Management titled: “Can stream and riparian restoration offset climate change impacts to salmon populations?”. A link to the journal online is provided here:

<https://www.sciencedirect.com/science/article/pii/S0301479716309793?via%3Dihub>

### **Abstract**

Understanding how stream temperature responds to restoration of riparian vegetation and channel morphology in context of future climate change is critical for prioritizing restoration actions and recovering imperiled salmon populations. We used a deterministic water temperature model to investigate potential thermal benefits of riparian reforestation and channel narrowing to Chinook Salmon populations in the Upper Grande Ronde River and Catherine Creek basins in Northeast Oregon, USA. A legacy of intensive land use practices in these basins has significantly reduced streamside vegetation and increased channel width across most of the stream network, resulting in water temperatures that far exceed the optimal range for salmon growth and survival. By combining restoration scenarios with climate change projections, we were able to evaluate whether future climate impacts could be offset by restoration actions. A combination of riparian restoration and channel narrowing was predicted to reduce peak summer water temperatures by 6.5 °C on average in the Upper Grande Ronde River and 3.0 °C in Catherine Creek in the absence of other perturbations. These results translated to increases in Chinook Salmon parr abundance of 590 % and 67 % respectively. Although projected climate change impacts on water temperature for the 2080s time period were substantial (i.e., median increase of 2.7 °C in the Upper Grande Ronde and 1.5 °C in Catherine Creek), we predicted that basin-wide restoration of riparian vegetation and channel width could offset these impacts, reducing peak summer water temperatures by about 3.5 °C in the Upper Grande Ronde and 1.8 °C in Catherine Creek. These results underscore the potential for riparian and stream channel restoration to mitigate climate change impacts to threatened salmon populations in the Pacific Northwest.

### **Citation**

Justice, C., S. M. White, D. A. McCullough, D. S. Graves, and M. R. Blanchard. 2017. Can stream and riparian restoration offset climate change impacts to salmon populations? *Journal of Environmental Management* 188:212–227.

## **NetMap Data Summary**

### **Introduction**

There is a pressing need among natural resource managers in the Grande Ronde River basin and elsewhere to extrapolate or roll-up empirical measurements of fish habitat conditions or fish abundance to a spatially continuous scale that encompasses the entire spatial extent of interest (e.g., fish population boundaries or assessment units). Stream and biological monitoring data in the Grande Ronde basin is largely limited to a set of randomly-selected sites ranging in size from 120-600 meters as in the Columbia Habitat Monitoring Program (CHaMP). Other monitoring programs such as the Oregon Department of Fish and Wildlife's (ODFW) Aquatic Inventories Program (AIP) or U.S. Forest Service's Level II stream surveys cover more extensive stream segments, but they too have considerable gaps in data coverage in some privately-owned portions of the basin or in smaller tributaries. These data gaps make it difficult for managers to assess the overall condition of a population or other key management units. To address this need, we intend to use spatial statistical network models (SSN; Peterson et al. 2013) coupled with NetMap data to link remotely-sensed landscape/land use characteristics with empirical measurements of fish habitat conditions and fish abundance and use model predictions to fill in the gaps where monitoring data does not exist. In addition, SSN models can be used to evaluate how aquatic habitat conditions might change under future restoration or land use scenarios. To obtain the input data (i.e., covariates) for these models, we contracted with TerrainWorks to provide and run the NetMap analysis toolset for the upper Grande Ronde River and Wallowa River subbasins (Figure 9; <http://www.terrainworks.com/terrainworks>). NetMap is a geographic information system (GIS) application that can be run in ArcMap and provides a suite of analysis tools to characterize fluvial and geomorphic processes within a watershed.

### **Methods**

As a first step in this process, we compiled all available LiDAR imagery (0.5-1 m resolution) within the study area and merged it with the 10 m digital elevation model (DEM) from the National Elevation Dataset (NED). This merged DEM (rescaled to 2 m resolution) was then used to generate NetMap's virtual watersheds and stream network, which form the spatial template for all NetMap analyses. NetMap produces over 250 stream and watershed attributes pertaining to fluvial processes (e.g., streamflow, hydraulic/planform geometry, floodplain mapping), aquatic habitats (e.g., fish intrinsic potential, beaver habitats), riparian condition, erosion, roads, and fire/climate change. NetMap outputs can be summarized at various spatial scales from entire watersheds, to HUC6 sub-watersheds, to individual stream reaches (approximately 100 m in length). Here we provide a summary of some key NetMap attributes and how they apply our

project objectives. The application of this data in SSN models and other landscape analyses will be forthcoming in the 2018 annual report.

## **Results and Discussion**

In portions of the basin with LiDAR coverage (Figure 9)—which encompasses most of the Chinook Salmon distribution area—the NetMap stream layer more accurately follows true channel morphology compared with coarser-scale hydrography layers such as the National Hydrography Dataset High Resolution 1:24K layer (NHDHR) or the NHD 1:100K layer (Figure 10). This is particularly true in broad, low-gradient valleys with highly sinuous stream channels. This improved accuracy translates to more accurate estimates of channel gradient and sinuosity, attributes that often have a strong influence on fish distribution and abundance. The NetMap stream layer also provides a more accurate estimate of true stream distance, which has important implications for spatial statistical network models that rely on distance between observations in calculations of spatial autocorrelation.

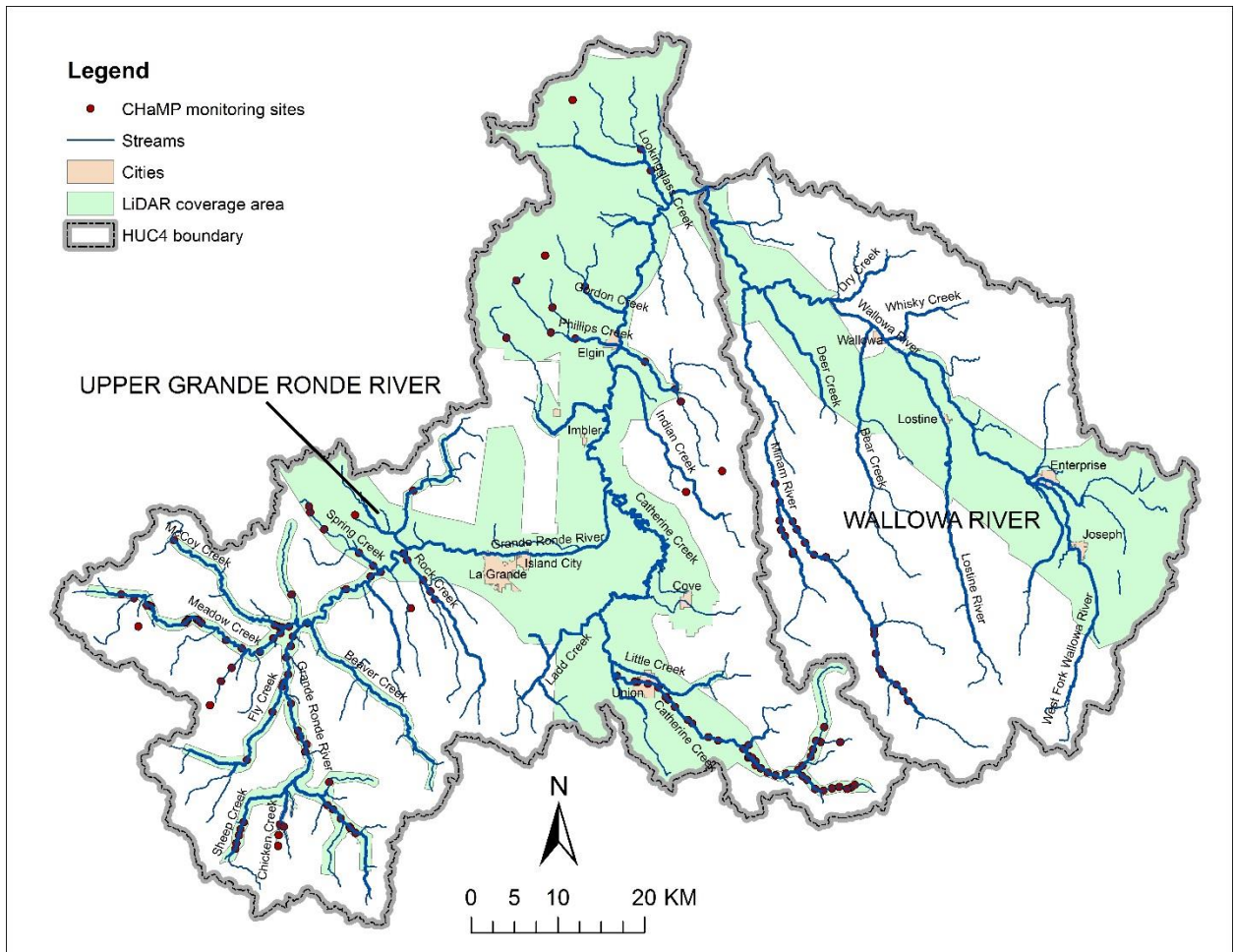
NetMap's floodplain mapping tools can be used to map and quantify valley floor elevations, or to calculate commonly used channel metrics such as floodplain width, or valley width index (valley width at 5X bankfull depth/bankfull width). Figure 11 shows an example of the floodplain mapping in the upper Grande Ronde River at Vey Meadows, with different colored shading indicating valley floor elevations at different multiples of bankfull depths. The cross-section profile tool can be used to quickly examine a cross sectional profile for any location along the stream network and identify floodplain features such as side channels, relic channels and roads. The tool can also easily produce longitudinal profiles (not shown here). We plan to use this tool to calculate an index of floodplain complexity for each NetMap reach (e.g., floodplain complexity index = (area of valley floor under 1 bankfull depth – mainstem area)/total valley floor area). Reconnection of river floodplains and increasing floodplain complexity are key objectives of modern stream restoration actions throughout the Pacific Northwest and in the Grande Ronde basin specifically, and therefore it's important that we have tools to quantify floodplain complexity. We hypothesize that fish abundance and diversity (both in terms of species and size classes) would be positively influenced by the higher floodplain complexity. We intend to test this hypothesis using empirical snorkel survey data in the coming months. If a strong relationship exists between fish abundance and floodplain complexity, then these data could be very useful for modeling fish population response to stream restoration.

NetMap also include methods for assessing riparian shade and thermal energy reaching the stream. The shade/thermal energy tool estimates the amount of solar radiation reaching the stream channel (Watt-hours / m<sup>2</sup>) as a function of remotely-sensed tree

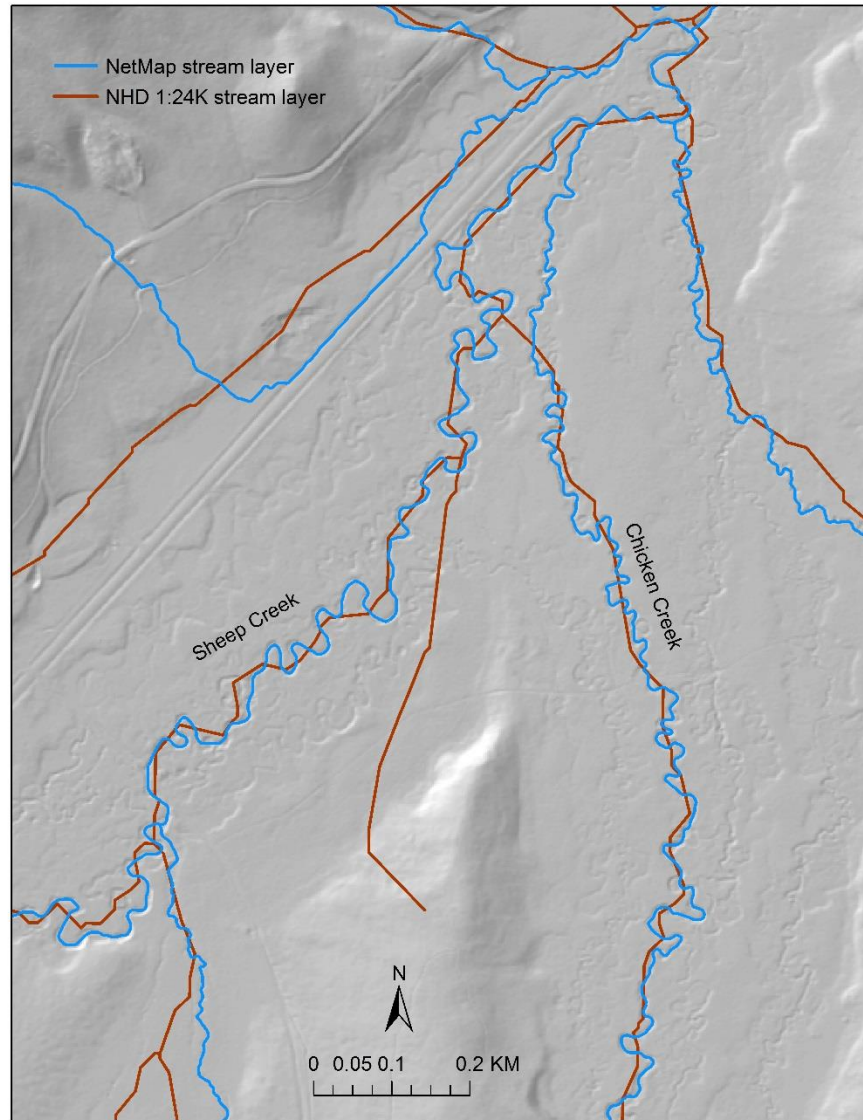
height and basal area (LEMMA; <https://lemma.forestry.oregonstate.edu/>), channel width, channel orientation, topographic shading and solar angle. The tool also estimates the difference in thermal loading between current conditions and maximum estimated shade conditions (Figure 12), thereby allowing users to identify areas where riparian restoration could be applied most effectively to increase shade and reduce thermal loading into streams. While this approach is somewhat simplified compared to our previous work on riparian mapping and Heat Source modeling in the upper Grande Ronde and Catherine Creek (Justice et al. 2017), the NetMap tools will allow us to rapidly assess shade and thermal loading across a much larger spatial extent and potentially use these data to improve on current water temperature models.

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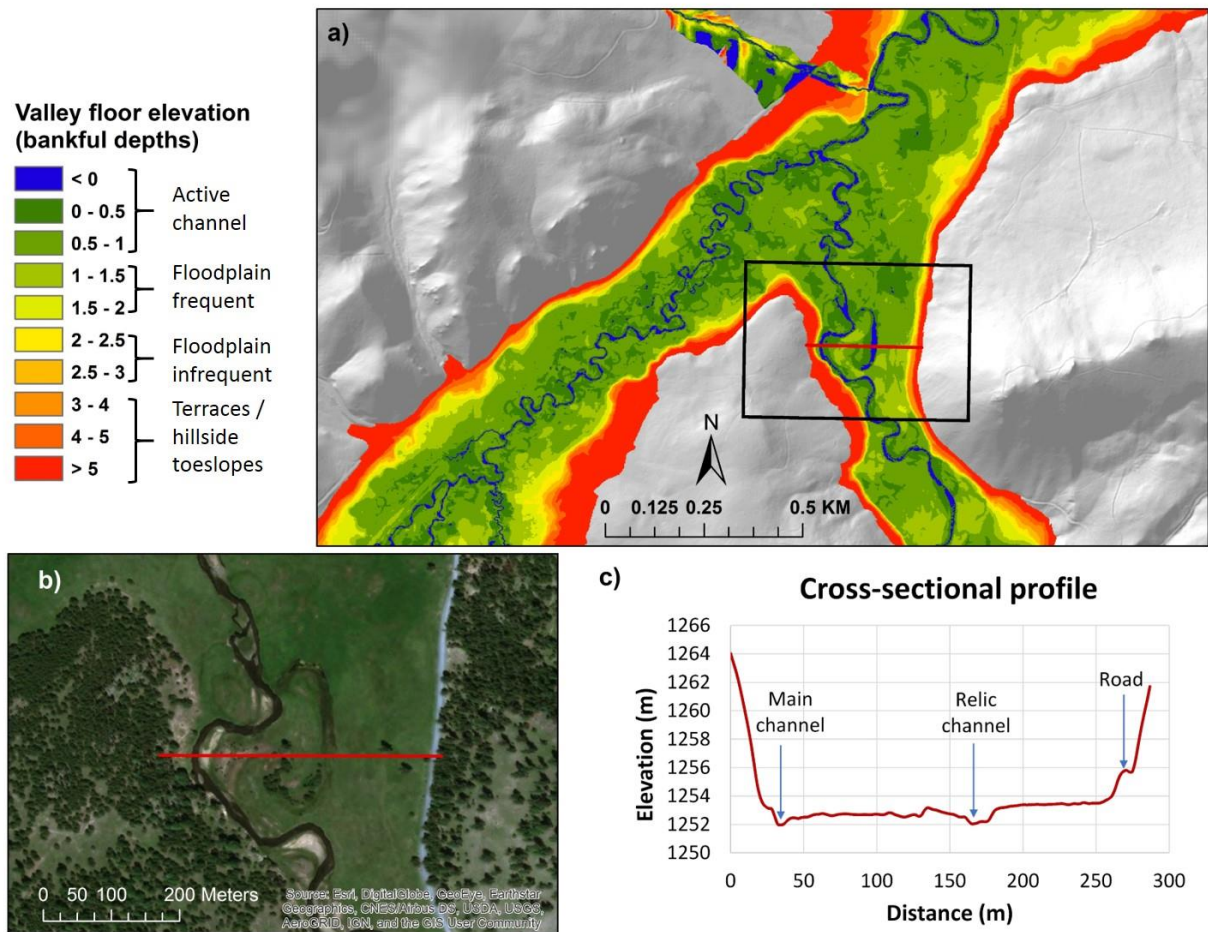


**Figure 9.** Map showing the two HUC4 subbasins where NetMap data was generated (grey dashed line) and the LiDAR coverage area (highlighted in green).

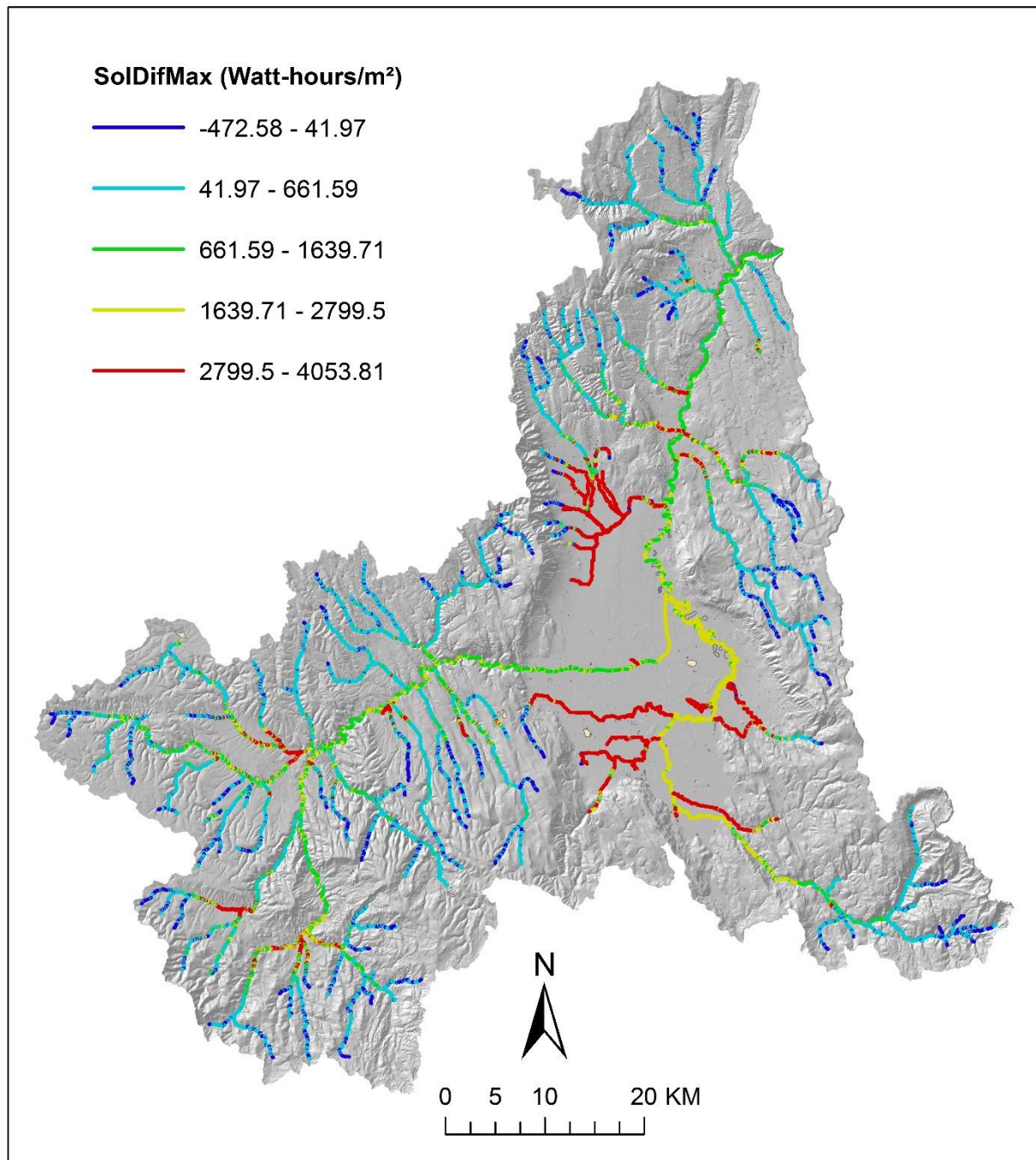


**Figure 10.** Comparison of the NetMap stream layer (blue line) with the National Hydrography Dataset High Resolution (1:24K) stream layer in the upper Grande Ronde River basin near the confluence of Sheep Creek and Chicken Creek. The grey background shows the NetMap hillshade layer generated from a 2-meter resolution digital elevation model.





**Figure 11.** Output from NetMap's floodplain mapping tools and cross section profile tools from the upper Grande Ronde River at Vey Meadows showing a) the valley floor elevation in multiples of bankfull depth, b) a magnified segment of the river showing satellite imagery and the location of a channel cross section (red line), and c) the cross-sectional profile plot corresponding to the location of the red cross section line.



**Figure 12.** Output from NetMap’s “Shade/Thermal Energy” tool showing the estimated difference in solar radiation reaching the stream channel (SolDifMax in Watt-hours / m<sup>2</sup>) from current riparian conditions to maximum shade conditions (i.e., solar radiation current – solar radiation maximum). Higher numbers (i.e. warmer colors) indicate areas that provide much less shade under current conditions compared with maximum shade conditions.

## Historical Ecology for Setting Restoration Targets

### Abstract

The following abstract summarizes the manuscript by White et al. (2017). Please see <https://www.elementascience.org/articles/10.1525/elementa.192/> for further details.

Land use legacies can have a discernible influence in present-day watersheds and should be accounted for when designing conservation strategies for riverine aquatic life. We describe the environmental history of three watersheds within the Grande Ronde subbasin of the Columbia River using General Land Office survey field notes from the 19th century. In the two watersheds severely impacted by Euro-American land use, stream channel widths—a metric representing habitat simplification—increased from an average historical width of 16.8 m to an average present width of 20.8 m in large streams; 4.3 m to 5.5 m in small, confined or partly confined streams; and 3.5 m to 6.5 m in small, laterally unconfined streams. Conversely, we did not detect significant change in stream widths in an adjacent, wilderness stream with minimal human impact. Using a mechanistic water temperature model and restoration scenarios based on the historical condition, we predicted that stream restoration in the impacted watersheds could notably decrease average water temperatures—especially when channel narrowing is coupled with riparian restoration—up to a 6.6°C reduction in the upper Grande Ronde River and 3.0°C in Catherine Creek. These reductions in water temperature translated to substantial changes in the percentage of stream network habitable to salmon and steelhead migration (from 29% in the present condition to 79% in the fully restored scenario) and to core juvenile rearing (from 13% in the present condition to 36% in the fully restored scenario). We conclude that land use legacies leave an important footprint on the present landscape and are critical for understanding historic habitat-forming processes as a necessary first step towards restoration.

### References

White, SM, C Justice, DA Kelsey, DA McCullough, T Smith. 2017. Legacies of Stream Channel Modification Revealed Using General Land Office Surveys, with Implications for Water Temperature and Aquatic Life. *Elem Sci Anth* 5(3):1-18. doi:10.1525/elementa.192.

# **Stream Biota**

## **Fish-Habitat Relationships**

### **Introduction**

A general approach to most fish habitat monitoring programs is to measure a suite of habitat conditions and infer how those conditions change (a) over space and time and (b) in response to alternative management strategies or policies. The importance of habitat condition to fish can either be gleaned from extensive literature on fish-habitat relationships (Jackson et al. 2001) or empirically determined by relating fish response to habitat conditions in a statistical model (Fausch et al. 1988). Most fish habitat monitoring programs occur as observational studies in natural systems rather than as controlled experiments, making it challenging to predict how management decisions directly translate into habitat conditions, or how habitat conditions influence fish response. This exemplifies the “correlation does not imply causation” problem of observational studies, where observed correlations among predictors (e.g., habitat conditions) and responses (e.g., fish performance) cannot be relied upon to infer mechanisms or direct causal effects (Shipley 2002).

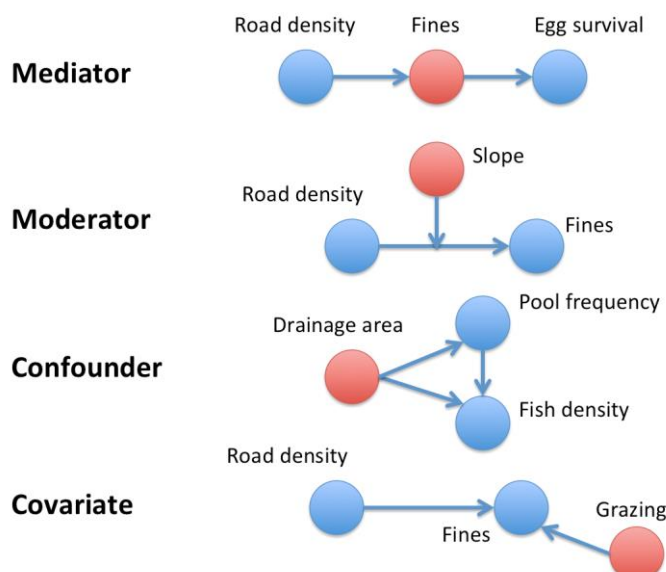
Structural equation modeling (SEM) is a multivariate approach that emerged from various scientific disciplines and builds upon numerous statistical techniques such as regression, path analysis, factor analysis, and latent variables (Grace 2006). The SEM approach can help address the problems mentioned above, and is one potential approach to estimating tributary habitat carrying capacity. To our knowledge, SEMs have not been employed to estimate carrying capacity of individual fish species. However, the approach has been used by aquatic ecologists for understanding patterns in biodiversity (Belovsky et al. 2011; Duffy et al. 2016), water quality and temperature (Zou and Yu 1994; Isaak and Hubert 2001), ecosystem indicators (Arhonditsis et al. 2006; Maloney and Weller 2011; Irvine et al. 2015) and fish performance (i.e., growth) (Budy et al. 2011).

The advantages of SEM for observational studies and differences from conventional univariate and multivariate approaches are reviewed in (Grace 2007). SEMs graphically relay complex hypotheses about how system components interrelate in a manner easily comprehended by stakeholders (Figure 13). Theoretical knowledge is typically used to develop models, which represent alternative hypotheses about processes leading to observed patterns in the data. The approach is based on the analysis of covariance relations, with maximum-likelihood estimation being the most common method for obtaining solutions; however numerous procedures can be used including Bayesian estimation. Several recent advances to SEMs make it an ideal approach for non-normal or nonlinear data, categorical responses, and hierarchical data structure. Overall, the approach is well suited to elucidating how different processes work in concert, how

effects propagate through as system, and evaluating the relative importance of different stimuli (Figure 13) (Wu and Zumbo 2008; Grace et al. 2010).

## Methods

We conducted snorkel surveys to quantify juvenile Chinook Salmon and steelhead abundance and size in their summer rearing habitats. These data were used to inform fish assemblage structure and to assess fish-habitat relationships. CRITFC, ODFW, and CTUIR perform snorkel surveys at CHaMP sites where habitat data



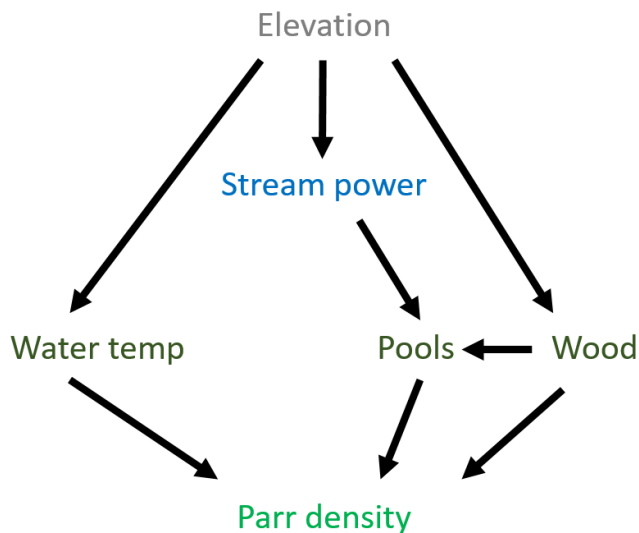
**Figure 13.** Language for causal models. Paths (arrows) between variables (circles) represent the direction of influence. Structural equation modeling (SEM) is well suited to analysis of these complex relationships.

are collected. These three agencies have recognized the need to use a common snorkel survey protocol so that information collected by individual entities can help managers determine whether aggregate habitat restoration actions will yield a net improvement in basin-wide habitat quality and viability of ESA-listed fish species. To this end, CRITFC developed a snorkeling protocol, drawing heavily from the methods of Thurow (1994) and O'Neal (2007) and integrated with the Pacific Northwest Aquatic Monitoring Program (PNAMP) methods, with the intention that this protocol will be implemented by all agencies responsible for data collection in the Upper Grande Ronde, Catherine Creek, Minam River, and potentially other nearby basins. Details about the snorkel survey



methodology can be found in White et al. (2012). Snorkel counts at each CHaMP site are expanded using a correction factor developed from paired mark-recapture to account for fish that were not observed by snorkelers (Jonasson et al. 2015) and screened to include only visits falling within the summer low-flow period of Julian Day 200-260.

We developed a fish-habitat SEM using data from the Columbia Habitat Monitoring Program (CHaMP 2016) coupled with snorkel surveys of salmonid densities (McCullough et al. 2017). The SEM (Figure 14) allowed us to address hypotheses that a) mean average August water temperature ( $^{\circ}\text{C}$ ), large pool frequency ( $\text{pools}\cdot\text{km}^{-1}$ ), and large wood frequency (wetted wood pieces $\cdot\text{km}^{-1}$ ) directly influenced parr density ( $\text{fish}\cdot\text{km}^{-1}$ ), b) that large wood indirectly influenced parr density through the formation of pools, and c) that elevation (m) was a contextual variable driving patterns in water



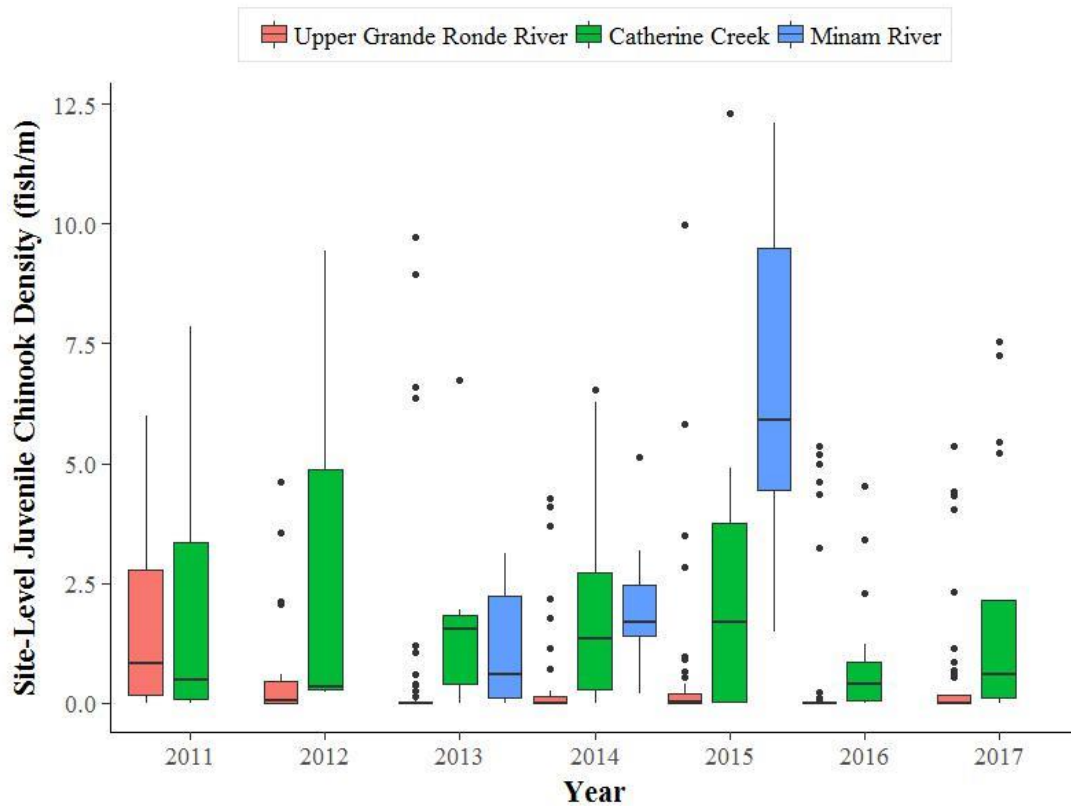
**Figure 14.** Hypothesized structural equation model (SEM) linking Chinook parr density with habitat variables in the upper Grande Ronde River.

temperature, bankfull specific stream power ( $\text{Watts}\cdot\text{m}^{-2}$ ), and large wood frequencies. Piecewise SEM (Lefcsek 2016) allowed for incorporating random effects, with *Site* as a random effect in each sub-model and *Year* as a random effect predicting parr density. Piecewise SEM also allowed for incorporating a nonlinear effect of water temperature on parr density in an equivalent manner as Justice et al. (2017), where negative impacts to fish did not occur until maximum weekly maximum water temperature reached  $18^{\circ}\text{C}$ .

## Results and Discussion

The 2017 field season marks the seventh consecutive visit to annual panels, and the third visit to Rotating Panel 1 (RP1) for CHaMP sampling. Site visits were coupled with associated fish snorkeling and electrofishing to determine late summer rearing capacity. Whereas analyses of fish distribution and its linkages to local and landscape conditions are underway, a cursory look at juvenile Chinook Salmon rearing densities in three ICTRT Chinook Salmon populations (upper Grande Ronde, Catherine Creek, and Minam River) was instructive (Figure 15). Fish densities in the Upper Grand Ronde River are

typically lower than other populations, and have substantially decreased since 2011. Though highly variable, average densities for both Catherine Creek and the upper Grande Ronde increased in 2017. Juvenile Salmon densities in the Minam River wilderness area tended to be as high or higher than Catherine Creek during the three years of sampling there, with the final year of sampling (2015) exhibiting the most significant difference from other populations.

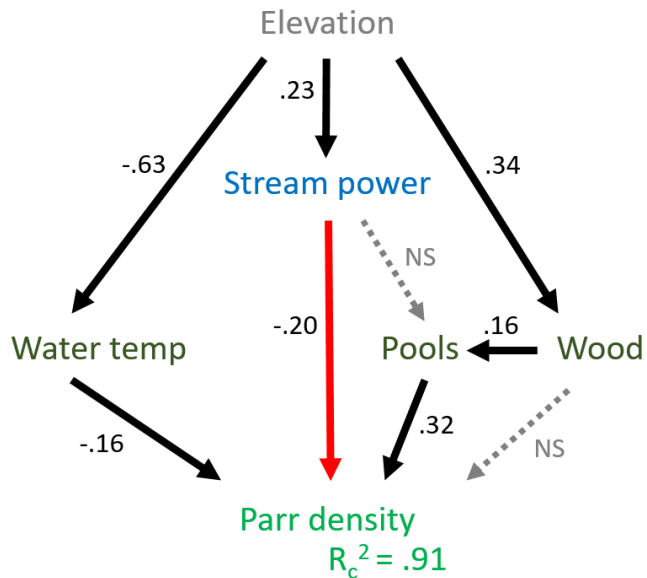


**Figure 15.** Juvenile spring Chinook Salmon density (fish/m) by ICTRT population from 2011-2017 in Upper Grande Ronde River and Catherine Creek, and from 2013-2015 in the Minam River for Julian days 200-260.

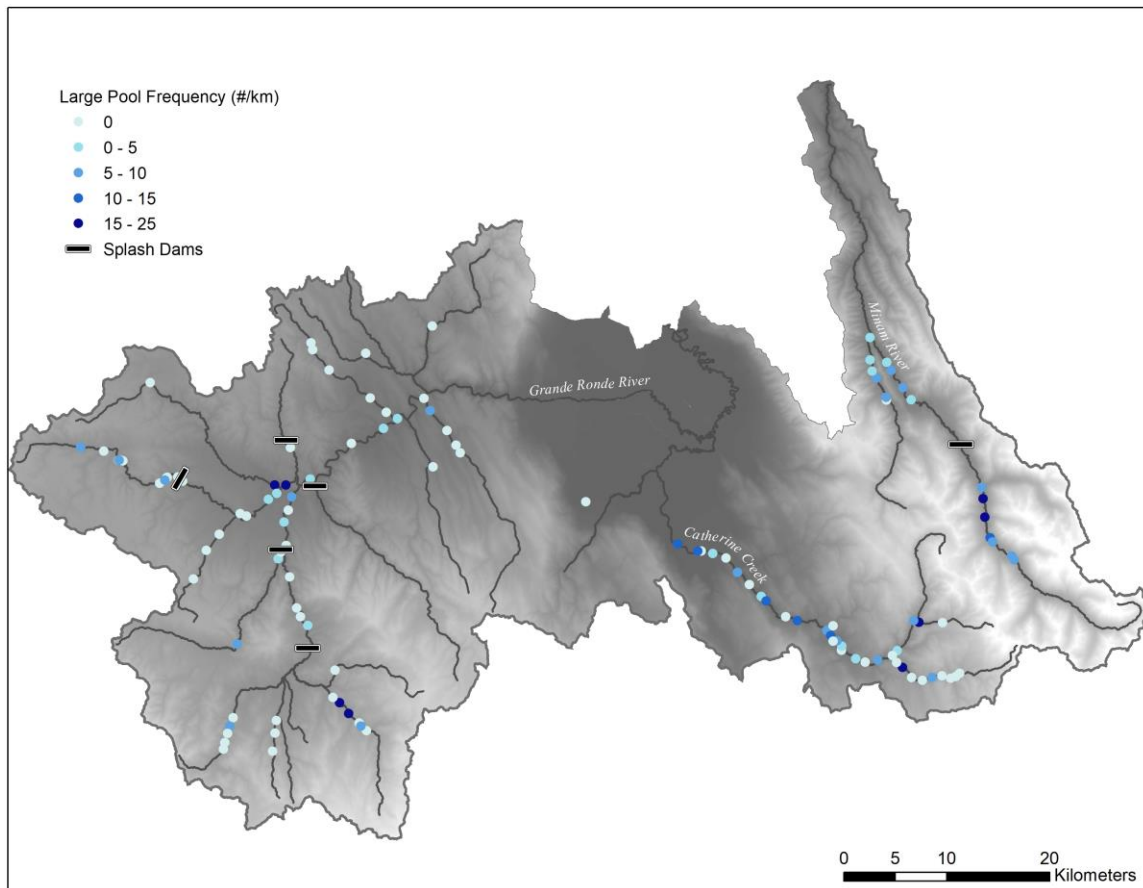


Piecewise SEM revealed that Chinook parr densities were positively influenced by pool frequencies and negatively influenced by water temperature as expected (Figure 16). Contrary to our hypothesis, SEM modification indices indicated that specific stream power was negatively associated with parr densities in a direct manner, and that wood frequencies did not have a direct influence. Rather, large wood appeared to have an indirect effect on fish through its positive association with pools. Landscape context was also an important consideration in this model, with higher elevations associated with cooler water temperatures, greater stream power, and higher frequencies of large wood.

**Figure 16.** Piecewise SEM results (variable defined in text). Direction of arrows indicates the direction of effect; magnitude of standardized path coefficients indicates the direction and strength of the relationship (coefficients closer to |1| are stronger). Dashed arrows are hypothesized interactions found to be non-significant; and  $R_c^2$  is conditional coefficient of determination of the mixed effects model.

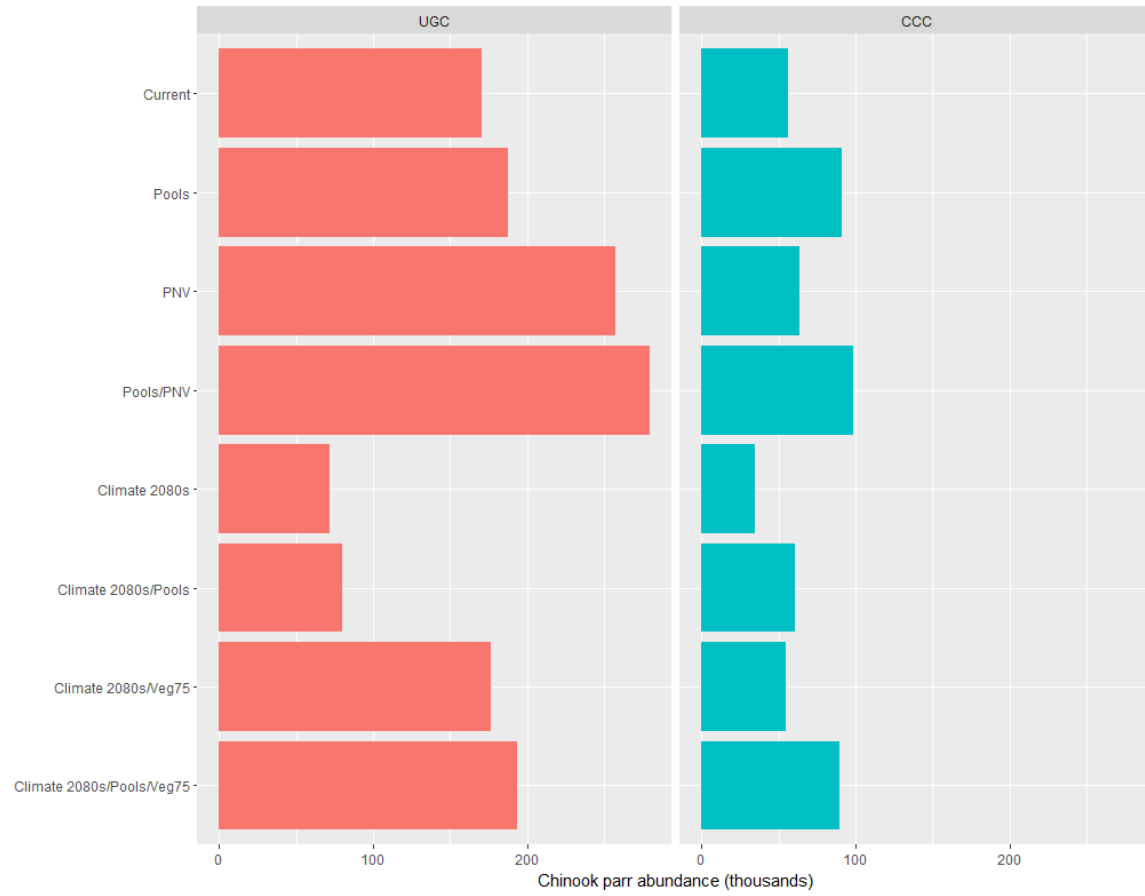


The effects of local-scale habitat conditions on fish density would have been obscured without incorporating a variable accounting for the position of reaches in the stream network. Mossop and Bradford (2006) used a similar conceptual model explaining associations among juvenile Chinook salmon density, wood, pools, and reach gradient in small tributaries of the upper Yukon River, Canada. Their study was based on visualizing pairwise correlation coefficients, however, and did not account for the variance-covariance structure inherent in modern SEM. Unlike their study, our findings revealed that the primary benefit of large wood to fish in this system is not through direct benefit from wood (e.g., velocity refuge, cover from avian predators, etc.), but instead through an indirect path of the pool forming function of large wood in streams. The distribution of pools in upper Grande Ronde and Catherine Creek (Figure 17) is currently far below values documented in the 1930s-1940s (McIntosh et al. 2000).

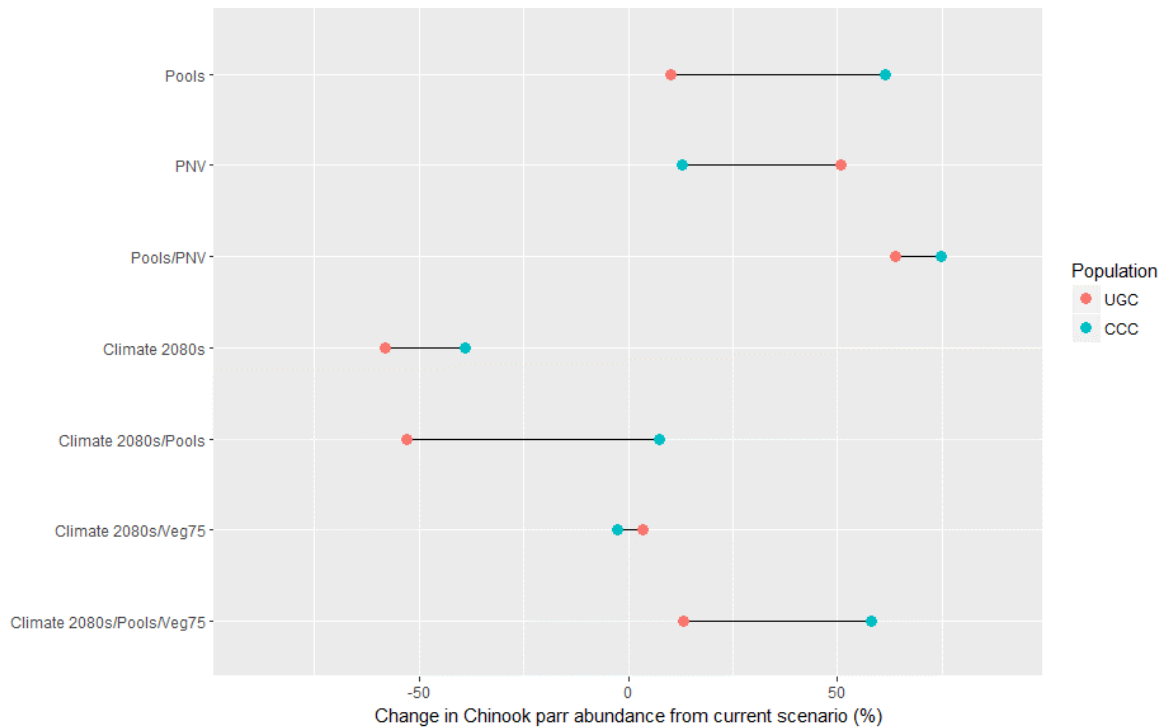


**Figure 17.** Current distribution of “large pools” as measured in CHaMP reaches in the study area, in relation to splash dams. Large pools meet the dual criteria of being  $\geq 0.8$  m in depth and  $\geq 20$  m<sup>2</sup> surface area during low flow conditions.

We used the SEM for evaluating the consequences to Chinook parr abundance of various restoration scenarios (Figure 18, Figure 19), including the current scenario (“Current”), restoration of large pool frequencies to their historical state (“Pools”) (McIntosh 1992; White, S.M., unpublished data), ensemble predictions of climate change in the 2080s coupled with a Heat Source model (“Climate 2080s”) and potential natural vegetation (“PNV”) or riparian vegetation growth to 75 years (“Veg75”) (Justice et al. 2017). In the upper Grande Ronde River, restoration to historical pool frequencies provides only a small benefit in both the current and 2080s time period, whereas restoring vegetation and subsequent shading provides a larger benefit to fish and potentially mitigates future climate change. In Catherine Creek where water temperatures are cooler and therefore less of a limiting factor, restoration of pools provides a greater benefit to fish; however riparian restoration provides significant additional benefit, indicating a holistic strategy of restoration is warranted.



**Figure 18.** Chinook parr abundance for various scenarios (described in text) in the upper Grande Ronde River (UGC, left panel) and Catherine Creek (CCC, right panel).



**Figure 19.** Percentage change in Chinook parr abundance from current status for various scenarios (described in text). Scenario predictions for Chinook parr abundance in Current, Pool restoration, potential natural vegetation (PNV), and Climate 2080s scenarios in the upper Grande Ronde River (UGC) and Catherine Creek (CCC).

This analysis does not consider effects of restoration across the entire life cycle of Chinook salmon (as in a life cycle model), nor do we account for density dependence in projected parr abundance under the scenarios. For example, parr abundance under complete riparian restoration and current water temperature (“PNV” in Figure 19) of c. 250,000 summer parr exceeds Oregon Department of Fish and Wildlife’s estimate of parr capacity of c. 150,000 based on stock-recruitment curves; however, it is not entirely unrealistic given ODFW has estimated parr capacity in recent years as high as c. 225,000 in the upper Grande Ronde River and (ODFW, unpublished data). Rather, the SEM and subsequent scenario predictions should be used to evaluate the relative potential of restoring habitat conditions for the parr life stage of Chinook salmon corresponding to various strategies available to managers in the basin.

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## Food Web Responses to Carcass Additions

### Introduction

Over the last century, human impacts mediated by harvest, dam construction, and habitat degradation have led to a marked decline of naturally spawning salmon within the interior Columbia River Basin (Nowak 2004, NOAA 2016). Returning adult salmon and steelhead transport large amounts of nutrients from the ocean to streams and these nutrient and carbon (eggs and tissue) subsidies can promote enhanced algae, invertebrate and fish productivity (Bilby et al. 1998, Kohler et al. 2012). With fewer spawning adults and reduced marine nutrient subsidies, the growth rates and production of juvenile salmonids may be much lower than historically. But in interior Columbia Basin streams, where Spring Chinook Salmon (*Oncorhynchus tshawytscha*) spawn during the summer/early fall, the effect of marine subsidies on juvenile salmonids is not well understood and likely varies throughout their spawning distribution due to shifts in temperature and fish species assemblages. Further, temperature regimes and species assemblages are expected to change over time (Ficke et al. 2007), which may influence the effects of salmon subsidies on stream ecosystems and biota in the future. To assess how the reduction of spawning salmon may be influencing stream food webs and, in particular juvenile salmonids, we added carcasses (steelhead; *O. mykiss*) to three experimental reaches in the Upper Grande Ronde River along a temperature and fish assemblage gradient. This project was conducted in collaboration with Oregon State University (Department of Fisheries & Wildlife) and Oregon Department of Fish and Wildlife (East Region Fish Research).

Carcass additions have been used as a management tool to promote enhanced production of salmonid species, but a thorough review of studies evaluating carcass additions reveals key knowledge gaps that justify the need for this study. First, two meta-analyses of studies evaluating fish responses to carcass additions (Janetski et al. 2009, Collins et al. 2015) have shown that results are mixed, with some exhibiting fairly drastic increases in juvenile salmonid growth rates and production, while others have observed little to no response. Second, there have been no published studies evaluating juvenile Chinook (including Spring Chinook) responses to carcass addition (see table in Collins et al. 2015). Third, the overwhelming majority of carcass addition studies have been in coastal drainages of the Pacific Northwest (Janetski et al. 2009, Collins et al. 2015) which differ substantially from interior Columbia River Basin streams in species composition, seasonal hydrology, climate, disturbance, and notably the timing of spawning salmon. Lastly, there have been no studies evaluating how the broader community of non-salmonid fish species can influence the responses of target salmonid species. This latter point is especially important as we consider expected shifts in fish assemblages as streams warm with climate change (Daufresne and Boët 2007, Ficke et al. 2007).

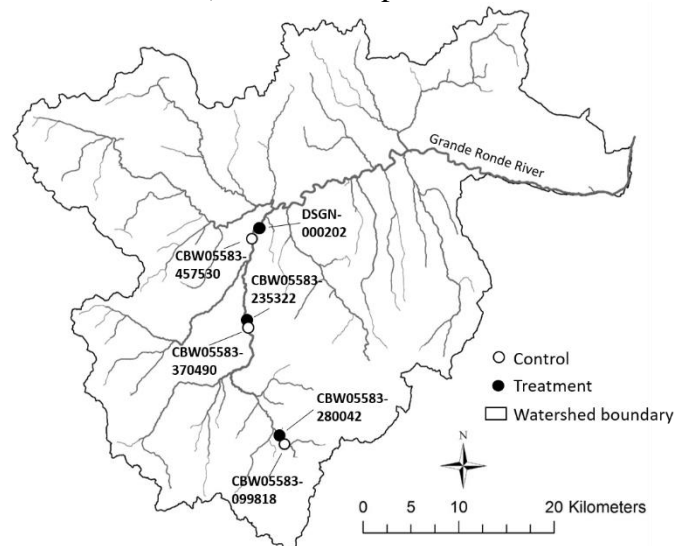
Our approach to evaluate the influence of carcass additions on fish communities involved quantifying fish growth rates, fish diets, stable isotope assimilation (fish, invertebrates, and algae), and algal (biomass) and invertebrate (drift and benthic biomass) responses in three sections of the upper Grande Ronde River. These three sections were positioned along a temperature and fish community assemblage gradient. We established three pairs of sites, with each pair containing an upstream control reach and a downstream treatment reach in which carcasses were added in mid-August. A before-after control-impact (BACI) design was utilized in which sampling was conducted before and after the addition of carcasses in both control (reference) and treatment (carcass addition) reaches.

## Methods

All sites were associated with the Columbia River Habitat Monitoring Program (CHaMP). The upper pair was located upstream of Vey Meadows (Control reach - CBW05583-099818; Addition reach - CBW05583-280042); the middle pair was located between Vey Meadows and Fly Creek (Control - CBW05583-370490; Treatment - CBW05583-235322); and the lower pair was located between Fly Creek and the confluence with Meadow Creek (Control - CBW05583-457530; Treatment - dsgn4-000202)(Figure 20).

Due to temperature and habitat differences, the fish communities in each pair differed. Pair 1 was dominated by juvenile Chinook and *O. mykiss*, but also contained sculpin (*Cottus* spp.), Bull trout (*Salvelinus confluentus*), and brook trout (*Salvelinus fontinalis*) in lower abundances. The community in pair 2 was dominated by Chinook but had increased numbers of *O. mykiss*, sculpin, longnose dace (*Rhinichthys cataractae*) and speckled dace (*Rhinichthys osculus*). In pair 3, Chinook were rare and we were not able to quantify growth rates. The fish community was numerically dominated by northern pikeminnow (*Ptychocheilus oregonensis*), bridgelip sucker (*Catostomus columbianus*), and reidside shiner (*Richardsonius balteatus*) but *O. mykiss* were still common. Reaches were 200-300 meters in length.

Due to differences in the discharge and bankfull width of each pair, different amounts of carcasses were added to each pair. Approximately 90, 121, and 166 steelhead carcasses



**Figure 20:** Map of carcass addition reaches (closed circles) and associated paired controls (open circles) within the upper Grande Ronde Basin.

were added to pair 1, pair 2, and pair 3, respectively. The average weight of male and female carcasses was 2.68 and 2.89 kg, respectively. The vast majority of females contained eggs (eggs from 40 females were removed during hatchery operations).

### ***Biofilm***

Substrate biofilm was sampled 2 times before carcass addition and 3 times after carcass addition. During each sampling event, we selected 3 rocks from each of 6-7 evenly spaced transects. The three rocks from each transect were pooled for a total of 6-7 replicates from each reach. A PVC pipe with a seven cm diameter opening was positioned on a rock and the area around the pipe was scraped and then washed away. The area inside the pipe was then scraped and washed into a container. Three subsamples were then pulled from this container; one for chlorophyll *a* analysis; one for biomass analysis; and one for stable isotope analysis. Chlorophyll *a* was analyzed using methods outlined in EPA method 445.0 (Arar and Collins 1997).

### ***Fish sampling***

Fish were sampled using snorkel-herding methods. Fish were herded by a snorkeler into a seine net equipped with a 1 x 1 x 1 m bag. Each site was sampled between 5-6 times over the summer with 2-3 sampling events before carcass addition and 3 events after the addition of carcasses. All *O. mykiss* during all sampling events except the last event were tagged with 9 or 12 mm PIT-tags (9 mm tags usually used on fish <80 mm and 12 mm tags used on fish >80 mm). *O. mykiss* individual growth rates between sampling events were obtained by recapturing and remeasuring individuals. We collaborated with the Oregon Department of Fish and Wildlife (ODFW) La Grande Fish Research's Early Life History Program (ELH) who tagged juvenile chinook in our study reaches in August around the time carcasses were added. Prior to the tagging of juvenile chinook, we used the change in mean size (both weight and length) of a minimum of 60 chinook to obtain relative growth rates between intervals. At the lower pair, we also tagged northern pikeminnow (n = 359), bridgelip sucker (n = 154) and reidside shiner (n = 165); however recapture rates during subsequent events were extremely low (<1%) which precluded us from obtaining individual growth rates on these species and tagging was abandoned.

**Diets:** The goal of diet sampling was to determine whether fish were directly consuming carcass tissue and eggs from carcasses. The other pathway for fish to obtain nutrients and carbon from carcasses is for carcasses to decompose and be incorporated into algae which is then consumed by invertebrates that are ultimately consumed by fish. We sampled salmonids using gastric lavage methods in which a small flexible plastic tube is inserted into the fish's stomach and water is pumped in, forcing stomach contents to be expelled. Gastric lavage is not an effective method on non-salmonids including pikeminnow, dace, suckers, and shiners. We therefore obtained diet samples of these species through lethal take (in accordance with permits). Our goal during each sampling

event was to obtain diet samples of 25 individuals for each species. We collected diet samples during three sampling events (1 week after carcass addition; 3 weeks after carcass addition; and 7-8 weeks after carcass addition). We quantified the number of consumed eggs per individual fish by species and the percentage of fish of each species that contained carcass tissue in their diets.

### ***Stable Isotopes***

Stable isotope analysis will allow for evaluation of the degree to which salmon nutrients are being incorporated in fish and it will provide information about the pathways in which nutrients are flowing through the food web. Samples have been prepared and sent to the UC-Davis stable isotope lab with an expected return date in late March, 2018. We prepared samples of algae (collected twice before carcass addition and 3 times after), stream conditioned leaves (an important source of food for some invertebrates; e.g. shredders), invertebrates (shredder, scrapper, and predator functional feeding groups and crayfish) and the dominant fish species at each site. Fish and invertebrate stable isotope samples were collected once prior to the addition of carcasses and twice after the addition of carcasses (3 weeks after and 7-8 weeks after). We obtained non-lethal fin clips from salmonids and we obtained fin clips from individuals obtained for lethal take for other fish species.

### ***Benthic and Drift Invertebrates***

Benthic invertebrates and drift invertebrates were collected during 3 sampling events (2-3 weeks before carcass addition; 3 weeks after carcass addition; and 7-8 weeks after carcass addition). In each reach, 8 0.25 x 0.25 m samples were collected using targeted riffle sampling protocols adopted by CRITFC and ODFW in the basin (Hayslip 2007). These samples were composited into a single pooled sample. Following the CHaMP protocol for drift sampling, nets were placed at the bottom of reaches for approximately 3-6 hours during days in which no other sampling occurred within the reach. Benthic and drift samples will be processed by a professional laboratory in spring of 2018.

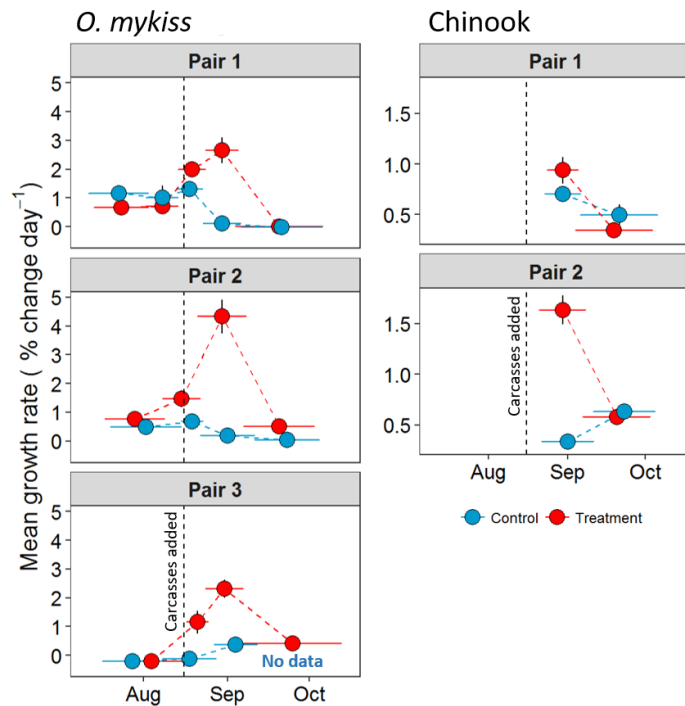
## **Results and Discussion**

In pairs 2 and 3, removal of carcasses by terrestrial scavengers was minimal and the majority of carcasses decomposed within the stream. Due to warm water temperatures, carcasses decomposed rapidly and little carcass material or eggs remained after 8 weeks. Trail camera footage revealed that bald eagles (*Haliaeetus leucocephalus*) were the dominant scavenger but most consumption occurred within the stream and thus carcass material (eggs and tissue) were released to streams during scavenging. Scavenging of carcasses in pair 1 by black bears (*Ursus americanus*) resulted in a rapid loss of carcasses from the reach and after three weeks almost no carcasses remained. Due to cool water temperature, carcasses in this reach decomposed very little before they were scavenged.

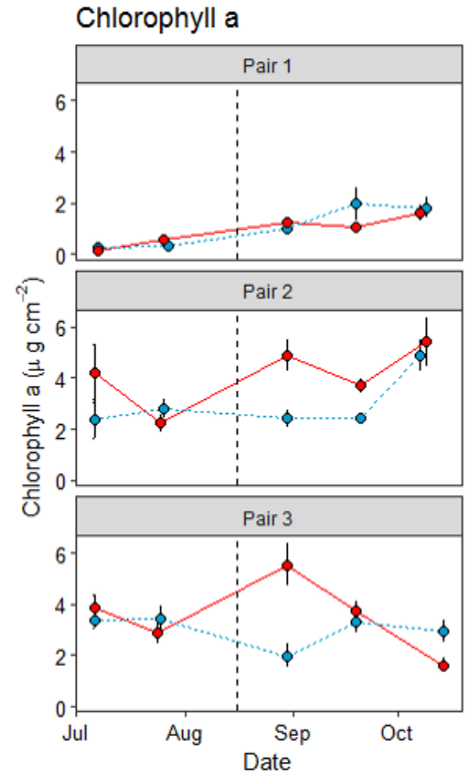
During the three weeks in which carcasses were present, eggs were commonly observed being released from carcasses into the stream. Bears were not observed on trail cameras at pairs 2 or 3.

The response of biofilms to carcass additions was mixed. Prior to carcass addition, chlorophyll *a* concentrations on benthic substrates were similar between control and treatment reaches of all three pairs, as evidenced by overlapping confidence intervals (Figure 21). However, approximately 2 weeks after the addition of carcasses, chlorophyll *a* was significantly greater in treatment reaches compared to controls in pairs 2 and 3 but not pair 1. Approximately 5 weeks after carcasses were added, chlorophyll *a* was only significantly greater in the treatment reach of pair 2 compared to controls. During the last sampling event, 8 weeks after carcass addition, chlorophyll *a* concentrations were similar between control and treatment reaches of pairs 1 and 2 but were significantly lower in the treatment reach of pair 3 compared to the control reach. Overall these results suggest nutrients released from carcasses resulted in a short-term positive biofilm response in pairs 2 and 3. However, there was considerable temporal variability at both control and treatment sites which complicates interpretation of these results. Stable isotope analysis will provide a tool to determine the degree to which carcass nitrogen was incorporated into algae.

The addition of carcasses resulted in increased growth rates of both *O. mykiss* and Chinook (Figure 22). Growth rates of juvenile *O. mykiss* were similar between control and treatment reaches prior to carcass addition and then increased substantially in treatment reaches 0-4 weeks after carcass addition in all pairs. From 4-8 weeks after carcass additions, growth rates in treatment reaches decreased to levels similar to control reaches. Chinook growth rates were greater in treatment reaches compared to controls 0-4 weeks after addition and then were similar to controls 4-8 weeks after addition. The difference in Chinook growth rates between the treatment and control reach was far more pronounced in pair 2 compared to pair 1. Because Chinook growth rates were only quantified after the addition of carcasses, we also evaluated changes in mean weight and condition factor over



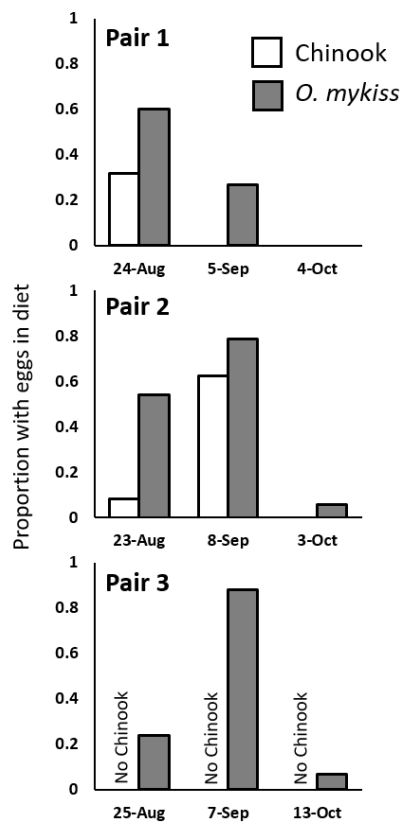
**Figure 22:** Growth rates of juvenile *O. mykiss* (left) and Chinook (right) in treatment (red) and control (blue) reaches. Horizontal error bars indicate the dates for each growth interval. Chinook growth was not quantified in pair 3 due to low abundance. Chinook growth rates were not measured prior to the addition of carcasses.



**Figure 21:** Biofilm chlorophyll a on benthic substrates in treatment reaches (red) and control reaches (blue). Error bars indicate 95% confidence intervals. The dashed line indicates the date carcasses were added to streams.

time as proxies for growth. In pair 1, the change in mean weight was substantially greater in the treatment reach 0-2 weeks after carcass addition and then remained similar to controls after. In pair 2, change in mean weight was greater 0-3 weeks after carcass addition and this resulted in Chinook being 22% larger in October. Mean condition was originally lower in the treatment reach compared to the control in pair 1 prior to carcass addition but was greater

than controls in all three sampling events after carcass addition. In pair 2, mean condition in the control and treatment reach were similar before carcass addition but were substantially greater 3 and 7 weeks after the addition of carcasses.



**Figure 23:** Proportion of juvenile *O. mykiss* (grey bars) and Chinook (open bars) with eggs in the diets during three events corresponding to approximately 1 week, 3 weeks, and 7-8 weeks after carcasses were added.

Diet samples from all three treatment reaches indicate that *O. mykiss* were consuming numerous eggs from carcasses 1 and 3 weeks after carcass addition. Chinook were also consuming eggs and carcass tissue with more eggs observed in diets of Chinook in pair 2 compared to pair 1 (Figure 23), a result consistent with growth rates. Diets of non-salmonids indicated little or no consumption of eggs or carcass material. This occurred during the same sampling events in which *O. mykiss* and Chinook diets contained eggs and carcass tissue. In addition, visual observation indicated that eggs were present and abundant on the stream benthos during these events, suggesting that competition for eggs and tissue between salmonids and non-salmonids was not inhibiting non-salmonids from consuming eggs and tissue. Stable isotopes of non-salmonids will provide additional evidence to determine the relative contribution of carcass material to non-salmonids.

Overall, these results suggest that the addition of carcasses to sections of the Upper Grande Ronde River increases growth rates and condition of juvenile salmonids. Our preliminary data suggests that these increases were attributed primarily to direct consumption of eggs and carcass tissue rather than through bottom-up pathways. Diets of Chinook

and *O. mykiss* contained abundant eggs and carcass tissue and other studies have found that direct consumption of these high energy food resources can result in rapid growth rates (Bilby et al. 1998, Bentley et al. 2012). Given the limited and short-term response of biofilm chlorophyll *a* to carcass additions, it is unlikely that changes in primary production and associated invertebrate production could have resulted in the short-term salmonid growth responses that occurred. However, forthcoming stable isotope analysis will allow for evaluation of the degree to which carcass nitrogen was incorporated into biofilm, invertebrates, salmonids and non-salmonids.



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## Temperature Impacts on Prespawn Mortality

In September 2017, we submitted a draft manuscript to Transactions of the American Fisheries Society titled “Water temperature and projected climate change impacts on prespawn mortality of Chinook Salmon in Eastern Oregon streams”. The reviewers requested some revisions to the manuscript, which we intend to complete and resubmit for publication in 2018. The draft manuscript is provided in Appendix A.

### Abstract

High mortality rates of adult salmon prior to spawning can significantly influence a population’s long-term viability, a problem that is exacerbated by warming stream temperatures associated with climate change. In this study, we used a 14-year time series of salmon carcass recovery data from spring- and summer-run Chinook Salmon *Oncorhynchus tshawytscha* collected in six rivers within the Grande Ronde and Imnaha River basins in Northeast Oregon coupled with water temperature predictions from the NorWeST model to evaluate the relationship between water temperature and prespawn mortality (*PSM*). We then used climate change projections for the 2040s and 2080s to predict how *PSM* rates would change in response to warming streams. Despite a high degree of spatial and temporal variability in estimates of *PSM*, our analysis revealed a significant positive effect of average August water temperature on the probability of *PSM*. Specifically, predicted *PSM* rates averaged across all rivers increased from approximately 0.1 at 10 °C to about 0.48 at 18 °C, with *PSM* increasing more sharply as temperatures exceeded approximately 15 °C. The mean probability of *PSM* was predicted to increase from approximately 0.06 under current climatic conditions (2000-2013) to approximately 0.12 by the 2040s (102% increase), and to 0.19 by the 2080s (230% increase). These results add to a substantial body of evidence linking high water temperature to increased *PSM* in Pacific Northwest salmon populations and provide insights into future climate change impacts to a critical life stage for these threatened species. These findings also highlight the pressing need for habitat restoration and other management actions that promote climate change resilience and improve habitat conditions and for fish in temperature-impaired watersheds.

### Citation

Justice, C., J. Feldhaus, D. Kelsey. (In review). Water temperature and projected climate change impacts on prespawn mortality of Chinook Salmon in Eastern Oregon streams. Submitted to Transactions of the American Fisheries Society in September 2018.

# **Conservation Planning for Climate Change Impacts to Benthic Macroinvertebrate Assemblages in the Columbia River Basin (October 2017 Draft)**

## **Preface**

In collaboration with the ODFW La Grande Field Office, CRITFC has collected benthic and drift macroinvertebrate from 2011 through 2017 at CHaMP sites in the upper Grande Ronde River, Catherine Creek, and Minam River. Our objective in collecting macroinvertebrate samples has been to characterize the capacity of river reaches to generate a food base for the fish communities and to link the macroinvertebrate communities with current conditions and trends in water quality, habitat characteristics, in context of position in the stream network (e.g., River Continuum). In our previous annual report for this project (McCullough et al. 2017), we described patterns in benthic and drift abundance, biomass density, and various derived metrics (Index of Biotic Integrity, % sensitive taxa, etc.) in relation to various environmental metrics for samples collected between 2011-2016. During the 2017 field season, we initiated a more comprehensive field sampling design that included the previous spatial coverage, but added temporal (monthly) sampling at six treatment/control sites in conjunction with the carcass addition study (See section: *Food Web Responses to Carcass Additions*). At the time of writing this report, metrics from the 2017 samples are still being finalized, and so we refer readers to our previous annual report (McCullough et al. 2017) for a detailed analysis of previous data. In this year's annual report, we chose instead to summarize the conclusions from a benthic macroinvertebrate vulnerability assessment (White et al. 2017a) as part of a collaboration with the Columbia Basin Partner Forum.

## **Introduction**

This assessment considers the vulnerability of benthic macroinvertebrate assemblages and their vulnerabilities to climate change impacts in the Columbia River Basin. We also explore potential strategies for resource managers to mitigate for these vulnerabilities. A three-step decision support framework for climate adaptation (Nelson et al. 2016) has been used to present these findings in a consistent format with other ongoing vulnerability assessments within the Columbia Basin Partner Forum (CBPF). The CBPF is an interdisciplinary group of federal, state, local, and tribal resource managers and scientists that meets periodically to address issues relating to climate change and the Columbia River Basin, with the support and guidance of the Great Northern Landscape Conservation Cooperative (2017).

This document is intended to assist resource managers and researchers in the following ways:

- By referencing selected literature, expertise, and data on the likely impacts of climate change on benthic macroinvertebrate assemblages;
- By highlighting research needs to better understand future impacts;
- By generating a set of management actions towards mitigating negative impacts on these assemblages, and aide their long-term integrity;
- By providing this information using a standard approach and format, which is being used to assess the vulnerabilities of other species and habitats in the Columbia River basin.

### **Conservation Target**

This climate vulnerability assessment for benthic macroinvertebrates employs an assemblage-level approach, rather than the population-level approach developed for salmonids by Nelson et al. (2016). In this report, an assemblage or community refers to multiple interacting populations of organisms having substantial overlap in timing and occurrence. In general, biotic community composition is considered one of several essential biodiversity variables, relevant for indicating trends in condition and vulnerability of ecosystems to climatic impacts (Pereira et al. 2003). Climate change may have more profound effects on overall community structure (Burgmer et al. 2007) or water quality metrics calculated from taxonomic composition (Hamilton et al. 2010) than on individual species. We place special focus on taxa within the benthic macroinvertebrate assemblage having life history, ecological, and behavioral traits that make them important food resources for stream-dwelling salmonids (Rader 1997; Sullivan and White 2017), or otherwise critical components of riverine food webs (Figure 1) (ISAB 2011; Naiman et al. 2012). Our decision to take an assemblage-level approach necessarily overlooks vulnerability of individual taxa—for example native freshwater mussels (Nedea et al. 2009) or crayfish (Larson and Olden 2011)—that may benefit from their own vulnerability assessments.



**Figure 24.** Aquatic insect larvae, such as this caddisfly (top), stonefly (middle), and mayfly (bottom), are excellent indicators of stream health and are important components of food webs in aquatic ecosystems. Photo copyright: Guenter A. Schuster <https://www.flickr.com/photos/ksnpc/6289538084>

### **Geographic Scope**

This assessment is focused on the streams and rivers of the Columbia River Basin that provide habitat to salmonids, including anadromous salmon, steelhead, and resident trout. Freshwater lotic ecosystems (flowing rivers and streams) are the principal habitat type for a wide range of benthic macroinvertebrates that are food sources for these fish, including species of mayflies (order Ephemeroptera), stoneflies (order Plecoptera), caddisflies (order Trichoptera), and others. Figure 25 shows the major watersheds of this region, which provide habitat for benthic macroinvertebrate assemblages.



**Figure 25.** Major rivers of the Columbia River basin.



Increasing air temperatures predicted for the 21<sup>st</sup> century are expected to reduce seasonal snowpack as more precipitation falls as rain and less as snow (Mote et al. 2014). This is expected to alter the timing of stream flow, with increases to seasonal and peak flows occurring between December and April, and lower stream flows between June and September (ISAB 2007). The extent of these changes is expected to be greatest in mid-elevation “transient” watersheds with moderate winter air temperatures, where precipitation will change from snow to rain (Mantua et al. 2010; Hamlet et al. 2013). Higher air temperatures and reduced stream flows are expected to increase water temperatures in regional streams (ISAB 2007; Isaak et al. 2012). Summer water temperatures in streams with larger relative groundwater inflows are expected to warm less than streams with lower groundwater contributions (ISAB 2007; Isaak et al. 2012). Climate change is expected to reduce cold-water fish habitat in Columbia River Basin tributaries, with the greatest effects to occur in streams of mid-elevation watersheds, areas east of the Cascade crest, southern portions of the basin, and in areas without permeable geology (ISAB 2007).

### **Step 1: Assess Vulnerability**

The vulnerability assessment was performed under a three-step process recommended by the CBPF of the Great Northern Landscape Conservation Cooperative. Under step 1 of this process, the vulnerability of benthic macroinvertebrate assemblages of the Columbia River Basin to climate change impacts was assessed. This was achieved through a review of key literature and regional professional expertise on the status and ecological characteristics of macroinvertebrates, and the likely climate change impacts to benthic macroinvertebrate assemblages. The findings are delineated into sections on habitat suitability, biotic interactions, and habitat connectivity. These findings are summarized in a vulnerability matrix (Table 3).

#### ***Habitat Suitability***

In general, invertebrates worldwide (terrestrial, marine, and freshwater) are highly susceptible to climate change impacts (Prather et al. 2009). Previous research has demonstrated several characteristics that can make invertebrates vulnerable to climate change, including narrow thermal tolerances, narrow moisture tolerances, synergistic effects to multiple stressors, low fecundity, low abundance, restricted range, long recovery time after disturbances, climate-related breeding requirements, low within-population genetic variation, specialist behavior, mutualist behavior (dependent on other species), and temporal mismatches with other species (Prather et al. 2009). A study of caddisflies (Trichoptera) across 23 European ecoregions revealed several traits making certain species more vulnerable to climate change: endemism, preference for springs, preference for cold water temperatures, short emergence period, and restricted ecological niches in terms of feeding types (Hering et al. 200). Because of their strict physiological

tolerances, caddisflies, stoneflies, and mayflies may be among the most vulnerable to climate-induced changes in water temperature (Li et al. 2013).

Haidekker and Hering (2008) found that benthic insect assemblages in mountainous areas of central Europe become less specialized with warming river temperatures, and that those in small streams are affected more than those in large streams. Durance and Ormerod (2007) examined data in a UK headwaters streams to test relationships between macroinvertebrate abundance and climatic variables. They found that abundance declines with increasing temperatures, and that less common species are most affected. However, an assessment of changes in abundance of individual stream macroinvertebrates in North Carolina and the Mid-Atlantic Highlands of the U.S. revealed that historically common taxa can also be highly vulnerable to biological, physical, and chemical water quality (Hawkins and Yuan 2016). A study in New South Wales demonstrated that families of stream macroinvertebrates that favor cold water and higher streamflow were most vulnerable to conditions expected under climate change (Chessman 2009). Thermal preference of benthic macroinvertebrates showed promise towards climate sensitivity using a long-term dataset from Maine, North Carolina, and Utah. Across the entirety of the U.S., stream temperature is a primary determinant of the macro-spatial distribution of benthic macroinvertebrate assemblages (Hill and Hawkins 2014).

Because of likely similarities across the Holarctic region in how benthic macroinvertebrates respond to environmental gradients, we presume the findings described above hold for taxa that evolved in the Columbia River basin. In the Pacific Northwest, many benthic macroinvertebrates rely on cool, flowing streams and rivers for their habitat. In fact, stream temperature is thought to be the “master variable” affecting the life-history strategies of most aquatic macroinvertebrates in the Columbia Basin; development rates, metabolism, and feeding of aquatic macroinvertebrates are highly affected by water temperature (ISAB 2011).

The Washington State Habitat Action Plan (2015) assessed the vulnerability of different taxa of benthic macroinvertebrates to climate change impacts. In general, mayflies were predicted to have a lower vulnerability, while stoneflies and caddisflies were predicted to have a higher vulnerability. For example, *Goreilla baumanni* (a caddisfly) was placed on a “climate watch list” due to its high vulnerability. These caddisflies use headwater habitats during their larval and pupae stages, and this link to a specialized habitat makes them especially vulnerable because these areas may be more likely to dry out. Stoneflies of the *Lednia* genus were also placed on this watchlist because their habitat preferences are centered around high-elevation coldwater locations, which may warm significantly in the future, especially as glaciers recede.

Another important distinction to make when evaluating macroinvertebrate vulnerability is whether taxa are fully aquatic (e.g., gastropods, crustaceans, etc.) and therefore exposed to water temperature year-round, or semi-aquatic and therefore able to escape extreme water temperatures after emergence, with emergence from the aquatic phase hypothetically arriving earlier with climate change (Plotnikoff pers. com. 2017).

Because many of the aquatic habitats in the Columbia basin are already in a disturbed state, a high proportion of benthic macroinvertebrate assemblages may be weighted towards taxa with wider physiological tolerances to environmental degradation. Ironically, it may therefore be more difficult to detect climate-induced changes to tolerant assemblages in these degraded habitats, since members of the biotic assemblage can withstand exposure to environmental gradients that would be detrimental to more sensitive taxa (Plotnikoff pers. com. 2017). A recent analytical approach developed by Hawkins and Yuan (2016) shows promise for evaluating the sensitivity of individual taxa to human disturbance, based on observed taxa-specific increases or decreases in abundance as compared to an expected reference condition.

### ***Biotic Interactions***

Benthic macroinvertebrates form an important part of the lotic food web, consuming organic matter and nutrients from the riparian forest and in-stream primary production thus representing a critical transformative food-web link for fish and other insect predators (ISAB 2011; Sullivan and White 2017). Benthic macroinvertebrates also contribute many other important services in freshwater ecosystems, including the conversion of organic material into food for other organisms, nutrient cycling, and the aeration of sediments. The loss of species diversity can reduce these ecological benefits (Covich et al. 1999). Climate change affects macroinvertebrates throughout their “ecological hierarchy” including changes to physical emergence, development, migration, reproduction, and the timing of resource consumption (Prather et al. 2009). Higher temperatures can disrupt the interactions between species through changes in phenology, survival, symbioses, and other pathways (Traill et al. 2010).

Climate change is expected to affect riparian plant communities through changes to air temperature and hydrologic-geomorphic regimes (Meyer et al. 1999; Bendix and Hupp, 2000). Resulting changes to benthic macroinvertebrate assemblages are expected following climate-induced changes to plant growth and subsequent shifts in hydrology (Suren and Riis 2010). However, some traits such as high crawling rate and armoring may buffer the response of certain benthic macroinvertebrate taxa to extreme low-flow events induced by climate change (Walters 2011). In Northern California streams, a commonly-applied index of biotic integrity (IBI) was not sensitive to climate-related changes, but using a trait-based approach revealed that macroinvertebrates with a life cycle >1 y and body size >40 mm declined with increasing temperature or decreasing

precipitation. These collective findings indicate that a trait-based approach may provide more sensitive indicators for the effects of climate change on benthic macroinvertebrate assemblages.

In general, there is a lack of understanding about how the loss of global invertebrate species may affect ecosystem services (Prather et al. 2009). Information is lacking about the diversity of most freshwater macroinvertebrate assemblages, but elevated levels of local endemism and species richness are typically found (Balian et al. 2008). Better monitoring and understanding of aquatic food web structure and function is needed to understand the effect of climate change on stream biota and to prevent losses of macroinvertebrate populations (ISAB 2011; Wisseman and Johannes 2015).

The presence of salmon carcasses in natural streams increases macroinvertebrate density and biomass because of the greater availability of nutrients and organic matter (Janetski et al. 2009). Warmer water temperatures predicted with climate change are expected to negatively influence the abundance of salmonids (Justice et al. 2017; White et al. 2017), thereby impacting the abundance and potentially assemblage structure of benthic macroinvertebrates through the loss of spawner carcasses as a marine-derived nutrient subsidy (Wipfli and Baxter 2010).

### ***Habitat Connectivity***

Benthic macroinvertebrates benefit from stream habitat connectivity that exist on a continuum, where migration is possible upstream or downstream. This habitat connectivity is limited, however, by the ability of individual species to migrate, as defined by their ability to drift downstream or—after emerging as adults—fly upstream to establish new habitats. Wisseman and Johannes (2015) examined the potential impacts of climate change on freshwater macroinvertebrates of the Deschutes, Klamath, and John Day basins of eastern Oregon. They found the most vulnerable habitat types to be in the alpine/subalpine streams and wetlands and forested headwater streams and springs. The macroinvertebrates associated with these habitats tend to have a lower adaptive capacity and upward migration as a response to warmer temperatures is limited because of their headwater locations. Streams that become ephemeral or are severely disrupted also limit the connectivity of macroinvertebrate assemblages, which may become trapped there and placed at risk of extirpation. Macroinvertebrate assemblages highly adapted to springs in Switzerland were likewise predicted to suffer from climate change (von Fumetti et al. 2017). In the semi-arid west, connectivity may also be considered as thermal refugia and the availability of water connecting these refugia. Climate change may impact this connectivity through increases to groundwater temperatures, and changes to summer stream flows as it relates to upwelling (Plotnikoff pers. com. 2017).

Hogg and Williams (1996) performed an experimental water temperature increase in a headwaters channel near Toronto, Ontario to study the effects on stream macroinvertebrates. They found that temperature increases stimulated the growth rates of macroinvertebrates but suppressed their overall abundance. They also found that macroinvertebrate life history parameters were even more sensitive to gradual changes in water temperature than their abundance and density, and advocated for maintaining diverse, connected habitats to facilitate gene flow among macroinvertebrate populations and reduce their vulnerability to climate change. Domisch, et al. (2011) modelled the susceptibility of stream macroinvertebrates in a submontane region of Central Europe and found that a changing climate will likely alter the range of these species along the river continuum, with possible effects including a loss in population abundance and diversity in headwaters areas, and the establishment of non-native macroinvertebrates in lower reaches.

Human activities that disrupt the continuity of river systems may also have a profound effect on stream macroinvertebrates (Vannote et al. 1980), especially when climate change is expected to increase demand for electricity derived from hydropower. The establishment of benthic macroinvertebrates is hindered in river reaches where hydroelectric dams alter natural flow patterns and cause daily and seasonally fluctuating water levels, and reduced flow velocities can favor non-native macroinvertebrates (ISAB 2011). Kennedy et al. (2016) found that hydropеaking regimes —with discharge often varying by a factor of 10 or more per day to meet energy demands—can lead to extirpation of aquatic insects that lay eggs near the river edge, a common life history strategy for many taxa including ecologically-important mayflies. White et al. (2017) examined the relationships between stream flow, temperature and macroinvertebrate responses in impounded river in the UK. They found macroinvertebrate assemblages to be particularly sensitive to changes in extreme flows in these regulated systems. Other human activities including land use and agriculture creates thermal barriers, diminished water recharge, and changes in the timing of flow in Columbia River Basin streams and rivers.

### ***Adaptive Capacity***

The adaptive capacity of macroinvertebrate assemblages (and individual taxa) will be a crucial factor in how resilient they are to climate change impacts. Knowledge of baseline macroinvertebrate assemblages in the Pacific Northwest pre-European settlement period is lacking (ISAB 2011). Anthropogenic effects of the development of the Columbia River and its tributaries since the 1800's include increased sedimentation, slower flows, greater water depths, and higher summer water temperatures, which have all contributed to drastic changes to benthic macroinvertebrate assemblages. Many native benthic macroinvertebrates have been extirpated or decreased in abundance (ISAB 2011).

Wisseman and Johannes (2015) identified the different classes of macroinvertebrates with the highest and lowest adaptive capacity to climate change in Eastern Oregon basins, with eurythermal (able to tolerate a wide range of temperatures) species likely having the highest adaptive capacity, and those species associated with snow-melt and glacier-fed habitats to have the lowest adaptive capacity.

Resource managers should embrace practices that help to understand and assist the adaptive capacity of benthic macroinvertebrates. In general, ecosystem function is more resilient to disturbances when species diversity or key functional groups of species are maintained (Traill et al. 2010). Prather et al. (2009) made several recommendations for researchers and managers regarding macroinvertebrate protection and climate change: (i) Research macroinvertebrate ecosystem services; (ii) perform interdisciplinary research; (iii) Conduct biomonitoring of macroinvertebrate assemblages to monitor changes ; (iv) Initiate informed decisions with stakeholders and policy makers; and (v) Increase public awareness of the role macroinvertebrates play in ecosystem service managers (i.e. through local schools, parks visitors and media).

### ***Other Interacting Stressors***

Several other stressors acting individually or synergistically on benthic macroinvertebrates assemblages may compound the impacts of climate change. These include:

- Use of pesticides/persistent toxins in landscape;
- Land use practices in the riparian zone and hillslopes (e.g., logging, grazing, ranching, road building);
- Degradation of physical in-stream habitat and floodplains;
- Increasing exposure to wildfires; and
- Increasing population density, intensified by climate-driven human migrations.

**Table 3.** Climate vulnerability assessment for benthic macroinvertebrates assemblages (BMAs) in riverine ecosystems of the Columbia River Basin.

Key Factor of Vulnerability	HABITAT SUITABILITY: To what extent will climate change alter habitat suitability affecting exposure, sensitivity, and adaptive capacity?	BIOTIC INTERACTIONS: To what extent will climate change influence competitor, predator, and prey/resource interactions?	CONNECTIVITY: To what extent will climate change alter the degree of connectivity to larger networks of populations or suitable habitats?
Climate-related questions to consider	<ul style="list-style-type: none"> <li>• Are stream temperatures expected to remain or become unsuitable?</li> <li>• Are streamflow conditions (peak, low flow, seasonality) likely to become unsuitable?</li> <li>• Could changes in human water use act synergistically with climate change (e.g., de-watering, diversions, hydropeaking)?</li> <li>• What is the adaptive capacity of BMAs to climate change impacts on physical habitat?</li> <li>• Are there stressor thresholds to habitat, which when surpassed, will cause BMAs to shift or decline?</li> <li>• Will increased storm intensity increase pollution, turbidity stressors?</li> <li>• Will food quality decrease in carbon enriched atmosphere?</li> </ul>	<ul style="list-style-type: none"> <li>• Will climate change adversely affect predation on BMAs?</li> <li>• Will climate change adversely affect prey or resource availability to BMAs?</li> <li>• Will climate change adversely affect competition or other adverse biotic interactions within BMAs?</li> <li>• How will climate-induced range shifts of invasive species impact BMAs?</li> <li>• Will climate-induced reductions in salmon populations affect nutrient availability/food webs in tributaries?</li> <li>• Will spatial/temporal phenologic mismatches harm predator/prey relationships?</li> </ul>	<ul style="list-style-type: none"> <li>• Are BMAs currently isolated, or connected to larger networks of populations and habitat?</li> <li>• Will this connectivity remain as a result of climate change?</li> <li>• Are human-caused barriers (culverts, diversions, dams, etc.) present that could become barriers to BMAs under changing stream flows?</li> <li>• Will extreme climatic events (e.g., drought or flooding) increase the degree of isolation of BMAs?</li> <li>• Will BMAs have sufficient thermal refuges, which occur at finer scales than fish?</li> </ul>
Assess Vulnerabilities	<p>A-Habitat likely to remain or become suitable</p> <p>B-Habitat likely to become marginal (i.e., at or near thresholds)</p> <p>C-Habitat likely to become unsuitable</p>	<p>D-Threats from biotic interactions likely to be low</p> <p>E-Threats from biotic interactions likely to be high</p>	<p>F-Assemblages likely to be connected to a larger network</p> <p>G-Assemblages likely to remain or become isolated</p>
	Answer: <b>B</b>	Answer: <b>E</b>	Answer: <b>G</b>



The Table 3 matrix was developed based on selected literature and expert knowledge of current conditions and expected impacts of climate change on benthic macroinvertebrate assemblages. It is important to note that while the process required deterministic answers to “Assess Vulnerabilities,” the reality is likely to be more nuanced. While habitat suitability will likely become marginal for many assemblages, some habitats will also likely become unsuitable. Regarding connectivity, while many assemblages will likely become isolated, many others will likely remain connected to a larger network. In producing the matrix of strategies in Step 2, we considered these nuances.

## Step 2: Climate Adaptation Strategies

In Step 2, the CBPF’s Decision Support Framework was utilized to calculate the overall vulnerability of Columbia River Basin benthic macroinvertebrate assemblages to climate change impacts, and to develop summary goals and strategies for their survival and continued resilience.

**Table 4.** Climate adaptation strategies derived from vulnerability matrix (Step 1).

Relative vulnerability to climate change:	Medium-High
Relative value for stream macroinvertebrate conservation:	High value in the short and long term, but may require investment to remove invasive predators or competitors and adjustments to land management practices
Potential goal:	Identify and protect existing high-quality habitat while improving the suitability of the entire stream network (including degraded habitats) to support resiliency of stream macroinvertebrate populations
Strategies:	<ul style="list-style-type: none"> <li>• Protect existing climate refugia and networks;</li> <li>• Moderate stream temperature increases;</li> <li>• Moderate base flow decreases;</li> <li>• Moderate changes to peak flow timing and magnitude;</li> <li>• Remove/suppress invasive predators and competitors;</li> <li>• Prevent invasion of non-native predators and competitors;</li> <li>• Adjust land use practices that work synergistically with climate change;</li> <li>• Reduce uncertainty through research and monitoring;</li> <li>• Determine additional strategies after clarifying management goal(s)</li> </ul>

### Step 3: Climate Actions

In Step 3, we employed the approach of the CBPF Template for Decision Frameworks to select a set of objectives and actions that respond to the Climate Adaptation Strategies developed in Step 2. This list of actions was further refined to ensure that they are appropriate to the challenges and opportunities to climate mitigation of benthic macroinvertebrate communities of the Columbia Basin. Nevertheless, these actions, summarized in Table 4Table 5, do not apply to specific geographies or communities, and should only be considered as a general example for resource managers.

**Table 5.** Climate adaptation objectives and actions derived from decision support framework (Step 2).

Strategies	Objectives	Actions
Protect existing climate refugia and networks	<ul style="list-style-type: none"> <li>Identify and protect areas likely to remain climatically suitable over the long-term</li> <li>Protect and restore critical or unique habitats that buffer survival during vulnerable periods (i.e. seasonally or at particular life history stages)</li> </ul>	<ul style="list-style-type: none"> <li>Understand and map where tributary confluences and groundwater inputs may buffer projected stream temperature increases</li> <li>Protect/restore off-channel habitats, spring brooks, and seeps</li> <li>Protect/restore flood or thermal refugia and stream segments that are important as connectors</li> </ul>
Moderate stream temperature increases	<ul style="list-style-type: none"> <li>Reconnect floodplains</li> <li>Restore incised (widened) channels</li> <li>Restore stream flows</li> <li>Maintain/enhance riparian vegetation to shade streams</li> </ul>	<ul style="list-style-type: none"> <li>Resolve thermal barriers</li> <li>Reconnect floodplain features (e.g. side channels, ponds)</li> <li>Designate and restore natural floodplain boundaries</li> <li>Remove infrastructure (e.g., roads, levees, rip rap, etc.) from floodplains</li> <li>Reintroduce beaver or build beaver dam analogs to increase sediment storage</li> <li>Work to restore natural flow regimes</li> <li>Reduce water withdrawals, restore summer baseflow</li> <li>On regulated streams, pulse flows during critical times, sourcing from lower in the thermocline</li> <li>Reduce grazing pressure (e.g. reduce stocking rates, use rest-rotation systems, fence riparian areas, provide off-stream water sources, retire vacant allotments in priority areas, increase monitoring in priority areas to ensure good practices)</li> <li>Restore riparian vegetation in degraded areas</li> <li>Adjust riparian vegetation to favor species that are better suited for future climate conditions</li> </ul>
Moderate base flow decreases	<ul style="list-style-type: none"> <li>Restore or replicate natural stream flows</li> <li>Reduce water withdrawals and/or water diversions</li> <li>Restore riparian and hillslope vegetation</li> <li>Increase natural water storage in groundwater aquifers</li> </ul>	<ul style="list-style-type: none"> <li>Remove or breach dams</li> <li>Increase water storage of water in floodplains by encouraging natural flooding and groundwater infiltration</li> <li>On regulated streams, pulse flows during critical times, sourcing from lower in the thermocline</li> <li>Increase efficiency of irrigation techniques</li> <li>Explore potential to combine sprinkler and flood irrigation to capture increasing spring floods (and recharge groundwater supplies) and then switch to more efficient sprinkler irrigation when stream flows are lower</li> <li>Consider alternative water supplies for public land operations to retain in-stream flows</li> <li>Legally secure water rights/agreements for in-stream flows</li> <li>Reform water laws to enable increased acquisition of in-stream water rights</li> <li>Explore the use of water trusts/funds to increase investments in the protection of watershed health and function</li> <li>Use water pricing to encourage water conservation</li> <li>Establish native riparian vegetation</li> </ul>

		<ul style="list-style-type: none"> <li>• Remove non-native riparian vegetation</li> <li>• Reintroduce beaver and/or install artificial beaver-mimic dams where compatible with conservation goals</li> <li>• Increase off-channel habitat and protect refugia in side channels</li> <li>• Protect wetland-fed streams which maintain higher summer flows</li> <li>• Maintain/restore forest and wetland vegetation cover</li> <li>• Reduce road density</li> </ul>
Moderate changes to peak flow timing and magnitude	<ul style="list-style-type: none"> <li>• Restore floodplain connections</li> <li>• Restore incised (scoured) channels</li> <li>• Restore riparian vegetation</li> <li>• Restore stream flow regimes</li> <li>• Reduce rain-on-snow flooding</li> </ul>	<ul style="list-style-type: none"> <li>• Remove infrastructure (e.g., roads, levees, rip rap, etc.) from floodplains</li> <li>• Reconnect floodplain features (e.g. channels, ponds)</li> <li>• Create new or restore degraded floodplain habitats</li> <li>• Reintroduce beaver to encourage dam-building that increases sediment storage and deposition</li> <li>• Establish riparian vegetation; remove non-native vegetation</li> <li>• Remove stressors that cause riparian damage (illegal or degraded trails, cattle, etc.)</li> <li>• Disconnect road drainage from streams</li> <li>• Restore natural drainage systems, create retention ponds</li> <li>• Maintain/restore forest, wetland and riparian vegetation cover</li> </ul>
Remove/suppress invasive predators and competitors	<ul style="list-style-type: none"> <li>• Remove/suppress invasive predators and competitors</li> </ul>	<ul style="list-style-type: none"> <li>• Remove or control invasive predators or competitors (via electrofishing, chemical removal, genetic swamping)</li> <li>• Encourage increased harvest of non-natives</li> </ul>
Prevent invasion of non-native predators and competitors	<ul style="list-style-type: none"> <li>• Prevent non-native predator invasions</li> <li>• Restore habitats that convey an advantage for native species over non-native species</li> </ul>	<ul style="list-style-type: none"> <li>• Strategically use physical or electrical barriers to prevent further spread of non-native species</li> <li>• Model future changes in stream flow and habitat to anticipate future invasion hotspots</li> <li>• Promote expansion native populations in areas where trying to prevent invasion of non-native species</li> </ul>
Reduce uncertainty through research and monitoring	<ul style="list-style-type: none"> <li>• Monitor changes in aquatic food web dynamics</li> <li>• Enhance research and monitoring to better understand the current status of macroinvertebrate communities and their vulnerabilities to climate change</li> <li>• Communicate science about macroinvertebrate communities to the public</li> </ul>	<ul style="list-style-type: none"> <li>• Assess food webs for baseline data; monitor food web dynamics in space and time</li> <li>• Research key questions related to climate change impacts on macroinvertebrate communities (<i>For examples, see Table 1, climate-related questions</i>)</li> <li>• Expand research of macroinvertebrate ecosystem services</li> <li>• Perform interdisciplinary biomonitoring of macroinvertebrate assemblages to monitor changes</li> <li>• Increase public awareness of the role macroinvertebrates play in ecosystem service managers (i.e. through local schools, parks visitors and media).</li> </ul>
Determine additional strategies after clarifying management goal(s)	<ul style="list-style-type: none"> <li>• To be determined</li> </ul>	<ul style="list-style-type: none"> <li>• To be determined</li> </ul>

The objectives and actions listed in Table 5 focus primarily on protecting and restoring the coldwater stream habitat that is important to macroinvertebrate communities. Habitat actions can be designed to assist macroinvertebrate taxa that cannot escape a changing aquatic environment. Any effort at habitat preservation should include unique habitats such as groundwater upwelling zones or tributary confluences, which will offer a refuge under future climate conditions (Plotnikoff pers. com. 2017). The current condition of riparian vegetation also needs to be considered when considering the effects of climate change. Riparian vegetation provides stream shading and organic litter inputs, improving the species composition and production of aquatic macroinvertebrates (Vannote et al., 1980; ISAB 2011). Floodplain management and connection to thermal refugia are also important opportunities for stream habitat protection and restoration. In addition to stream habitat actions, changes to water management, basin wide land use, suppression of invasive species, and research and monitoring are highlighted in the range of helpful objectives and actions.

## Data Sets

Many regional fish habitat programs collect benthic macroinvertebrates as part of ongoing habitat monitoring (Table 3), and maintain regional datasets of their distribution, abundance, and diversity. As programs increasingly adopt the Northwest Standard Taxonomic Effort (NWSTE) guidelines (<https://www.pnamp.org/project/4210>), data can be shared widely across the region.

**Table 6.** Type of macroinvertebrate samples collected in common fish-habitat programs of the Pacific Northwest (from Sullivan and White 2017).

Program	Abbreviation	Targeted riffle	Reach-wide	Multi-habitat	Drift
USFS-BLM Aquatic and Riparian Effectiveness Monitoring Program	AREMP <sup>1</sup>	X			
California Department of Fish and Game	CDFG <sup>1</sup>	X			
EPA Environmental Monitoring Assessment Program	EMAP <sup>1</sup>	X	X		
National Aquatic Resource Surveys	NARS <sup>2</sup>		X		
USFS-BLM Biological Opinion Effectiveness Monitoring Program	PIBO <sup>1</sup>	X			
Upper Columbia Monitoring Strategy	UC <sup>1</sup>	X			
Columbia Habitat Monitoring Program	CHaMP <sup>3</sup>				X
BPA Action Effectiveness Monitoring	AEM <sup>2</sup>	X		X	
BLM AIM-National Aquatic Monitoring Framework	AIM-NAMF <sup>2</sup>	X	X		
USGS National Water-Quality Assessment	NAWQA <sup>2</sup>	X			
Status and Trends Monitoring for Watershed Health and Salmon	WA <sup>2</sup>		X		
Oregon Department of Environmental Quality	ODEQ <sup>2</sup>	X			

<sup>1</sup> Reviewed in (Roper et al. 2010)

<sup>2</sup> Pers. comm. (various sources). NARS is an outgrowth of EMAP.

<sup>3</sup> CHaMP 2016

Additionally, the following data sets will be helpful in assessing the likely future impacts of climate change on water temperature and flow in Columbia River Basin streams and rivers:

NorWeST Statistically Modeled Historic and Future Stream Temperature Data

Isaak, et al. (2015) developed spatial statistical network models of summer water temperatures using historic measurements collected in the NorWeST database and a series of other climatic and geomorphic data. They used these models to produce simulated August mean water temperatures throughout the Pacific Northwest for a range of historic years (1993-2011), and for the future climate change periods based on the A1B emissions scenario of the CMIP3 global assessment.

Regional Joint Monitoring Operating Committee (RMJOC) II Streamflow Projections for Future Climate Change Scenarios (expected to be complete in June 2017)

This multiyear project will provide an updated set of streamflow projections for the streams and rivers of the Columbia River Basin for future climate change periods based on the most recent (CMIP5) global climate scenarios. The projections are a result of a modeling effort by the University of Washington with the VIC Hydro Model, which is supported by BPA and a regional consortium.

The Ambient Biological Monitoring Program dataset (1993-2004) from the Washington State Department of Ecology is a long-term monitoring project to explore spatial and temporal trends in benthic macroinvertebrate communities. It is available as a diagnostic tool for identifying stressors to macroinvertebrates and the settings in which they occur.

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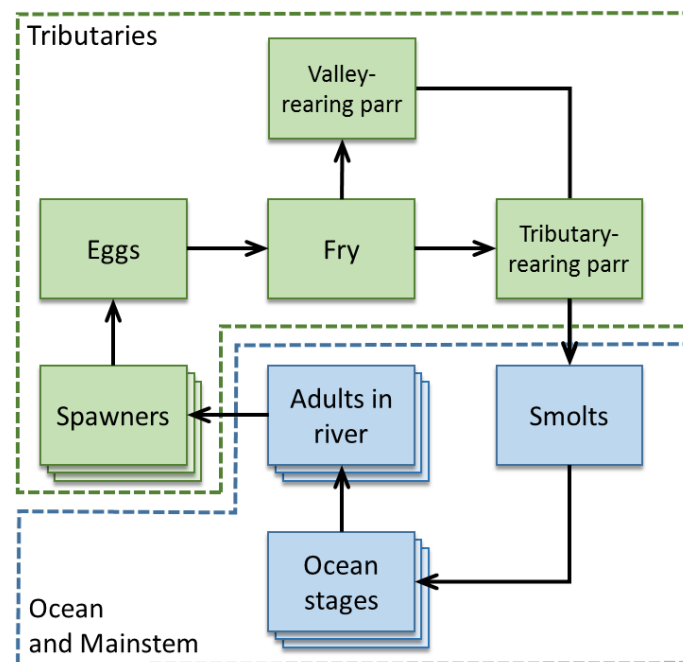
## Life Cycle Model

### Preface

We continued development and revisions to a habitat-based spring Chinook Salmon life cycle model in collaboration with Eco Logical Research (ELR) as described in our previous annual report (McCullough et al. 2017). While the model is still under revision, we made several improvements during this contracting period. Here, we reiterate the overall objectives of the life cycle model, document updates to the model from the previous version, and summarize key preliminary findings. A draft report documenting the background, methods, results, and discussion of life cycle model findings in greater detail is provided in Appendix B.

### Overview of Updates

This summary describes continued progress toward assessment of Chinook salmon populations and habitat restoration opportunities within Catherine Creek and the Upper Grande River watersheds using a life-cycle modelling (LCM) framework (Figure 26).



**Figure 26.** Conceptual diagram of the life cycle model (LCM) framework used to assess restoration scenarios in Catherine Creek and the Upper Grande Ronde. Stacked boxes represent life stages for which multiple ages exist and are tracked accordingly. The model also tracks two dominate out-migration strategies which include migrating downstream and overwinter in the Grande Ronde valley (‘Valley’) or remaining in the upper mainstem and tributaries of each system (‘Tributary’). Smolts are enumerated at Lower Granite Dam (LGD), where ‘ocean stages’ begin and are modeled via LGD-to-LGD smolt-to-adult return rates.

The work presented here extends life-cycle development efforts previously evaluated by the Columbia River Inter-Tribal Fish Commission (CRITFC). Ultimately, these efforts are intended to provide an analytical tool that can be used to guide the development of long-term restoration and recovery strategies for two threatened Chinook populations under changing climatic conditions. However, much of the work presented here focuses on the continued refinement of the LCM framework and addresses several revisions and extension designed to increase the effectiveness of the LCM investigation. Accordingly, the current LCM development efforts sought to address the following specific objectives:

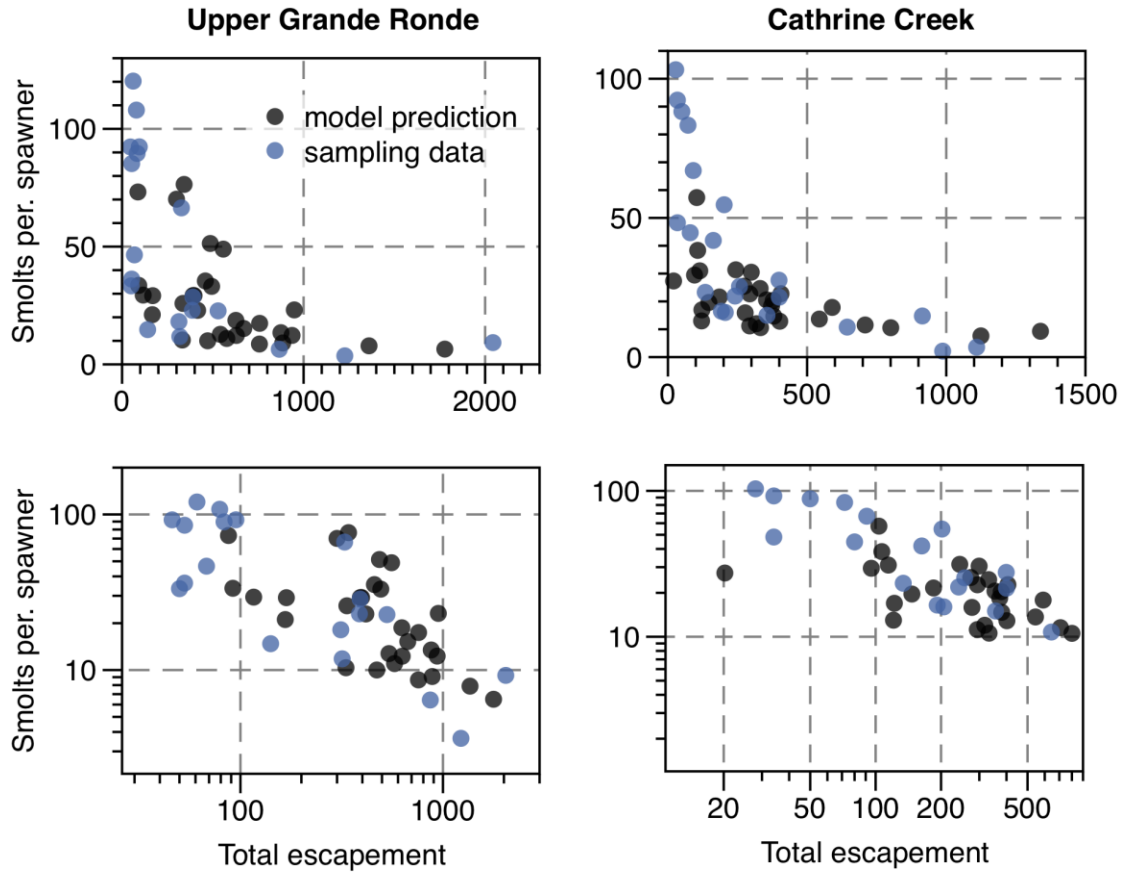
1. Complete the development and validation of LCM productivity and capacity terms for the Upper Grande Ronde Chinook population and incorporate into model simulations of potential restoration effectiveness.
2. Use available data to refine the differential parr survival terms for the dominate life-histories (i.e. fall vs. spring migrants) exhibited within in each population.
3. Incorporate revised parr capacity estimates based on parameters obtained via a Beverton-holt stock-recruit function fit to available abundance data.
4. Develop a means to vary spawner capacity according to the availability of habitat that is thermally suitable for pre-spawn survival under the restoration scenarios described in Justice et al. (2017).
5. Provide a more comprehensive description of the analytical approaches used to develop capacity and productivity terms across life-stages.
6. Simulate population performance under restoration scenarios described by a linear mixed-effects fish habitat model Justice et al. (2017) and a structural equation model (White et al., unpublished data) that consider the effect of future temperature, habitat, and climate conditions on early life-stage and spawner capacity.

### **Preliminary Findings**

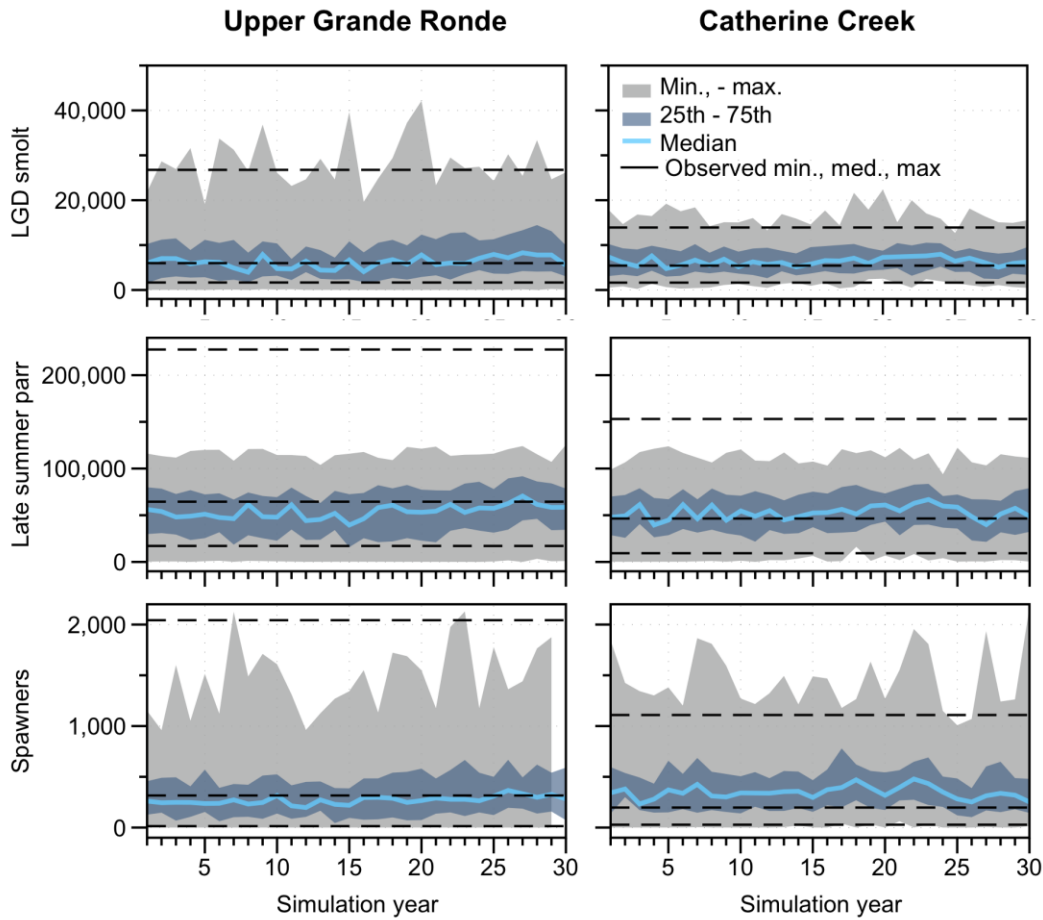
Several significant changes to the LCM framework continue to increase the realism and utility of the model to assess recovery potential for Upper Grande Ronde and Catherine Creek Chinook within a LCM context. Specifically, providing a means to address the influence that extreme temperatures have on spawning ground survival likely improves the model's ability to represent each scenario its influence on freshwater productivity. In addition, allowing the model to account for life-history dependent smolt survival probabilities, and inclusion of mortality for hatchery smolts prior to passing LGD are also more accurately reflect the dynamics of these systems.

Overall, our base parameterization (i.e., 'Curr') appeared to accurately capture both the stage specific (i.e., freshwater vs. marine, Figure 28) and total life cycle productivity (Figure 29) of spring Chinook salmon in Catherine Creek and the Upper Grande Ronde,

despite being developed from disparate and somewhat independent datasets in a piecewise fashion (e.g., freshwater survivals from long-term monitoring datasets vs. a habitat-based parr capacity model that predicts how many fish a basin can produce).



**Figure 27.** Base model validation of life-cycle productivity showing relationship between smolts (at LGD) per spawner and total spawners (hatchery + natural) for the Upper Grande Ronde and Catherine Creek Chinook populations. Sampling data includes years 1992 – 2014, and model predictions depict a single 30 – year model iteration.



**Figure 28.** Base model validation of stage specific abundance showing total range, 25<sup>th</sup> and 75<sup>th</sup> percentile, and median abundance as predicted by the LCM for smolts (at LGD), parr, and spawners. Horizontal lines show the observed range and median values for observed data from 1992 – 2014.

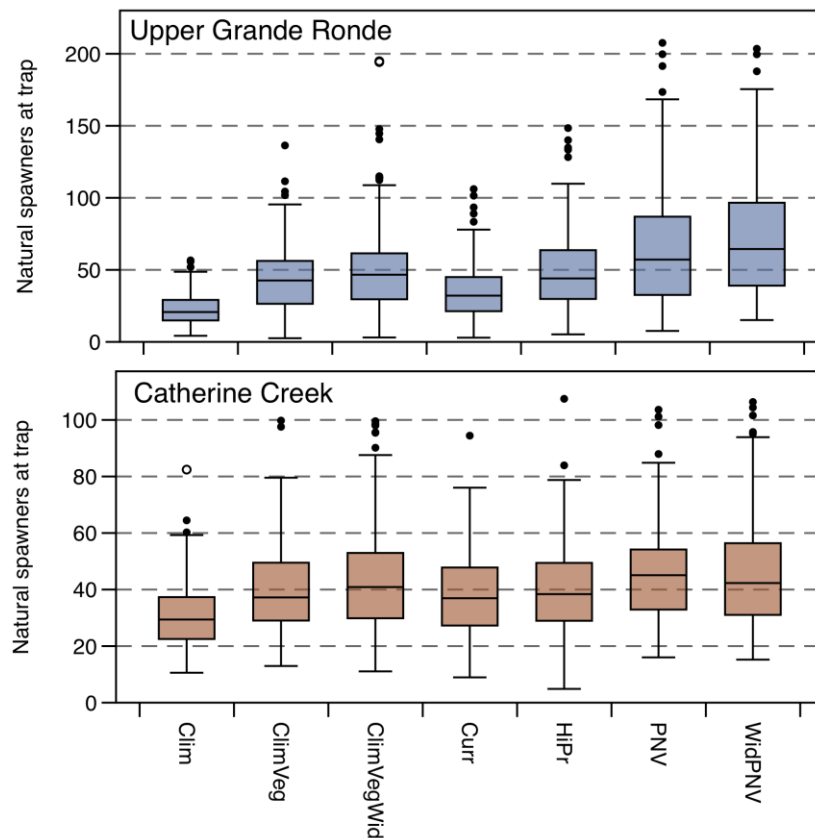
Given these patterns, the current LCM parameterization appears to reasonably capture the demographics of the intended natural populations. Accordingly, the model, with the capacity modifications was used to assess the effects of the Justice et al. (2017) scenarios on total spawner abundance. While no attempt is made to infer meaning here, LCM simulation results illustrate the following:

- The proportional changes in parr and spawner carrying capacity reflecting the Justice et al. (2017) habitat scenarios translated into an increase in the abundance of natural origin adult spawners that was similar on a rank-order basis. However, the magnitude of the population response did not increase/decrease on a one-to-one basis to the capacity changes under each scenario.
- The absolute response to population productivity under most management scenarios was marginal (i.e., increase of only 10 to 50 natural spawners).



However, relative increases/decreases in spawners were more pronounced, and represented a potential doubling of returning natural spawners under the most aggressive restoration scenarios (i.e. PNV and WidPNV).

- Marginal response to restoration scenarios may be explained by two factors: reliance of these populations on a heavy supplementation scheme, and an SAR that is independent of spawning and rearing productivity.
- Productivity responses (i.e. increase/decrease) were more pronounced within the UGR where current temperature regimes in much of the rearing network reside close to upper temperature threshold for this species.
- Overall, the ‘PNV’ and ‘WidPNV’ scenarios, which assume maximum cooling of stream temperatures in the absence of climate change effects, showed the greatest response, with an approximate 80-100% increase in spawner abundance within the UGR. On the other hand, ‘doing nothing’ vegetation/restoration wise in the face of the anticipated effects of climate change on conditions in the UGR (scenario ‘Clim’) translated into a 36% reduction in spawner abundance.



**Figure 29.** Distribution of the geometric mean of naturally reared spawners returning to the adult trap/weir for UGR and CC under each restoration scenario (“Curr” = current scenario, “Clim” = climate 2080s, “Veg” = vegetation restoration, “Wid” = channel width restoration, “HiPr” = vegetation restoration in high priority areas, “PNV” = restoration to full potential natural vegetation).

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- Justice, C., White, S.M., McCullough, D.A., Graves, D.S., Blanchard, M.R., 2017. Can stream and riparian restoration offset climate change impacts to salmon populations? *Journal of Environmental Management* 188, 212-227.
- McCullough, D.A., S.M White, C. Justice, L. Burns, D. Kelsey, and D. Graves. 2017. Assessing the Status and Trends of Spring Chinook Habitat in the Upper Grande Ronde River and Catherine Creek. Annual Report to Bonneville Power Administration. Portland, OR: Columbia River Inter-Tribal Fish Commission.

## Dissemination of Project Findings

### Publications

- Justice, C., S. M. White, D. A. McCullough, D. S. Graves, and M. R. Blanchard. 2017. Can stream and riparian restoration offset climate change impacts to salmon populations? *Journal of Environmental Management* 188:212–227.
- Justice, C., D.A. McCullough, and S. White. Habitat Characteristics and Fish Use of Cold-water Refuges in the Upper Grande Ronde River. 2017. Columbia River Inter-Tribal Fish Commission Technical Report 17-09, Portland, OR. 59p.
- Sullivan S.P, White S.M. 2017. Methods supporting the development of food web metrics from benthic macroinvertebrate data. Report prepared for the Bureau of Indian Affairs Rights Implementation Climate Change Contract AO9AV00480 by Rhithron Associates, Inc., Missoula, MT, and Columbia River Inter-Tribal Fish Commission, Portland, OR.
- White, S.M., C. Justice, D.A. Kelsey, D.A. McCullough, T. Smith. 2017. Legacies of Stream Channel Modification Revealed Using General Land Office Surveys, with Implications for Water Temperature and Aquatic Life. *Elementa: Science of the Anthropocene* 5(3):1-18. doi:10.1525/elementa.192.
- White, S.M. 2017. Using history to help shape the river's future (public outreach article). *Ripples in the Grande Ronde*, Grande Ronde Model Watershed Education Outreach Program.

### Draft publications

- Justice, C., J. Feldhaus, D. Kelsey. Influence of water temperature on spawning success of spring Chinook Salmon in the Grande Ronde and Imnaha River basins, northeast Oregon. Intended for *North American Journal of Fisheries Management*.
- White, S.M., Graves, D., Barton, D., Gephart, L. Conservation planning for climate change impacts to benthic macroinvertebrate assemblages in the Columbia River basin. Prepared for the Columbia Basin Forum.
- White, S.M., Justice, C., McCullough, D., Blanchard, M., and Sedell, T. Climate change meets the ghosts of land use past: Revisiting historical changes in salmon habitat in the Columbia River Basin. Intended for *Ecological Applications*.
- White, S.M., McHugh, P., Naman, S., Baxter, C., Bellmore, R., Naiman. R., and Danehy, R. What's in the toolbox? A primer of food web perspectives and methods for riverine fish conservation. Intended for *Fisheries*.

### Presentations

- Justice, C. 2017. Influence of River Restoration and Climate Change on Stream Temperature and Fish Abundance in the Upper Grande Ronde River. Presentation to Grande Ronde Model Watershed, ISRP, NPPC, La Grande, OR.
- Justice, C. 2017. Can Stream and Riparian Restoration Offset Climate Change

Impacts to Salmon Populations? Presentation to the US Geological Survey, Fall Seminar Series, Portland, OR.

- Justice, C., S. M. White, D. A. McCullough, D. S. Graves, and M. R. Blanchard. 2017. Can stream and riparian restoration offset climate change impacts to salmon populations? Sandy River Basin Watershed Council, Climate Adaptation Workshop, Troutdale, Oregon.
- White, S.M., Justice, C., McCullough, D., Kelsey, D., Sedell, T. 2017. Using historical ecology to guide stream habitat restoration. River Restoration in Eastern Oregon and Eastern Washington Session, Oregon Chapter of the American Fisheries Society, Bend, Oregon.
- White, S.M. 2017. Historical pool analysis and restoration mapping in the upper Grande Ronde River. Presentation to Grande Ronde Model Watershed, ISRP, NPPC, La Grande, OR.
- White, S.M. 2017. Storytelling as common ground between traditional and scientific ecological knowledge. Indigenous Science and the Environment seminar, Portland State University.

### **Media coverage**

- Television: “Ash and Sediment from Eagle Creek Fire expected to wash into streams, clog spawning ground,” KATU News, Portland, OR. [katu.com/news/local/ash-and-sediment-from-eagle-creek-fire-expected-to-wash-into-streams-clog-spawning-ground](http://katu.com/news/local/ash-and-sediment-from-eagle-creek-fire-expected-to-wash-into-streams-clog-spawning-ground)
- Television: “Forest Service plans fast action in gorge,” KGW News, Portland, OR. [www.kgw.com/news/local/forest-service-plans-fast-action-in-gorge/474258867](http://www.kgw.com/news/local/forest-service-plans-fast-action-in-gorge/474258867)
- Radio: “Tribes worry over lower Columbia Gorge Coho runs with forest fires,” Oregon Public Broadcasting, Portland, OR. <http://www.opb.org/news/article/oregon-salmon-columbia-gorge-eagle-creek-fire-tribes/>
- Podcast: “Would you trust the feds to make decisions about your land?” Terrestrial, National Public Radio (NPR), Seattle, WA. [kuow.org/post/would-you-trust-feds-make-decisions-about-your-land-terrestrial](http://kuow.org/post/would-you-trust-feds-make-decisions-about-your-land-terrestrial)

## **Appendix A - Temperature Impacts on Prespawn Mortality**

# Water temperature and projected climate change impacts on prespawn mortality of Chinook Salmon in Eastern Oregon streams

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September 2017

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## Abstract

High mortality rates of adult salmon prior to spawning can significantly influence a population's long-term viability, a problem that is exacerbated by warming stream temperatures associated with climate change. In this study, we used a 14-year time series of salmon carcass recovery data from spring- and summer-run Chinook Salmon *Oncorhynchus tshawytscha* collected in six rivers within the Grande Ronde and Imnaha River basins in Northeast Oregon coupled with water temperature predictions from the NorWeST model to evaluate the relationship between water temperature and prespawn mortality (*PSM*). We then used climate change projections for the 2040s and 2080s to predict how *PSM* rates would change in response to warming streams. Despite a high degree of spatial and temporal variability in estimates of *PSM*, our analysis revealed a significant positive effect of average August water temperature on the probability of *PSM*. Specifically, predicted *PSM* rates averaged across all rivers increased from approximately 0.1 at 10 °C to about 0.48 at 18 °C, with *PSM* increasing more sharply as temperatures exceeded approximately 15 °C. The mean probability of *PSM* was predicted to increase from approximately 0.06 under current climatic conditions (2000-2013) to approximately 0.12 by the 2040s (102% increase), and to 0.19 by the 2080s (230% increase). These results add to a substantial body of evidence linking high water temperature to increased *PSM* in Pacific Northwest salmon populations and provide insights into future climate change impacts to a critical life stage for these threatened species. These findings also highlight the pressing need for habitat restoration and other management actions that promote climate change resilience and improve habitat conditions and for fish in temperature-impaired watersheds.

## Introduction

Pacific salmon that have successfully migrated to their spawning grounds often die prior to spawning, a process commonly referred to as prespawn mortality (*PSM*). High *PSM* rates observed in salmon populations throughout the Pacific Northwest (up to 90%; Bowerman et al. 2016) have raised awareness among natural resource managers of the potential threat that *PSM* poses to population persistence and recovery potential (Keefer et al. 2010, Bowerman et al. 2016). While the threat of high *PSM* to salmon persistence is well understood in many systems, there is a need to better identify and quantify the environmental and biological factors influencing *PSM* (Roumasset 2012, King et al. 2013) so that correction actions can be designed and implemented. Some factors that have been associated with high *PSM* include elevated water temperature (Contor and Schwartz 2007, Quinn et al. 2007, Keefer et al. 2010, Roumasset 2012, Bowerman et al. 2017), low streamflow (Quinn et al. 2007, Tillotson and Quinn 2017), pollutants (Feist et al. 2011, Scholz et al. 2011), infectious disease (CDFG 2004, Benda et al. 2015), higher percentage of hatchery origin spawners (Bowerman et al. 2017), and high fish density (Quinn et al. 2007, Tillotson and Quinn 2017). Of these factors, high water temperature is most consistently associated with elevated rates of *PSM*.

Recent research has shown a high degree of variability in *PSM* rates both within and across salmon populations (Bowerman et al. 2016, Roumasset 2012), suggesting that the environmental or biological factors influencing *PSM* are likely to be somewhat specific to a given area or fish population. For example, spring and summer-run Chinook Salmon *Oncorhynchus tshawytscha* are likely more susceptible to high summer water temperatures than fall Chinook because they typically spawn during August and September when temperatures are at their peak, while fall Chinook enter freshwater and spawn in fall

when temperatures are cooler. Additionally, salmon population response to climate change is likely to vary substantially among populations because of geographic differences in hydrology, topography, vegetation, and climate patterns (Crozier and Zabel 2006, Battin et al. 2007) as well as genetic differences among populations (Steel et al. 2012). Thus, the development of predictive models relating environmental factors to *PSM* should be tailored to the area and population of interest or otherwise account for population variability.

Although spawning ground surveys are commonly conducted throughout the Pacific Northwest to estimate salmon spawner abundance, distribution and *PSM*, data on reach-specific environmental conditions, particularly water temperature, is often lacking. Researchers and managers often struggle to draw linkages between commonly collected spawning survey data and environmental conditions because stream and reach-specific data is difficult and expensive to acquire, or because the data was simply not collected historically. In this study, we demonstrate the use of a broad-scale regional water temperature model (NorWeST; Isaak et al. 2016) to evaluate potential relationships between modeled temperature data and empirical reach-scale estimates of *PSM*. Additionally, we were able capitalize on the climate change projections available from NorWeST to predict how climate-related stream warming would impact *PSM*.

Human-caused climate change has already resulted in significant alterations to lotic habitats in North America including shifts in the magnitude and timing of seasonal flow patterns, increased frequency of extreme flow events such as flooding and drought, and long-term increases in mean annual stream temperatures, and these trends are likely to continue (Lynch et al. 2016). Isaak et al. (2012) estimated that summer water temperatures in Pacific Northwest streams increased by approximately 0.22 °C/decade between 1980 and 2009, and August stream temperatures were projected to increase on average, +2.83°C by the 2080s (Isaak et al. 2016). Although the response of Pacific salmon to climate change is likely to be diverse, a growing body of evidence suggests that recent climate change has already led to spatial and temporal shifts in salmon growth, phenology, and population dynamics (Lynch et al. 2016). Climate change has also been implicated in the 2015 run failure of Sockeye Salmon in the Columbia River, poor survival of Winter-run Chinook Salmon in California and Oregon coastal Coho Salmon, ocean acidification and associated reductions in growth of Atlantic Salmon, and spread of exotic predators (Crozier 2016).

Given the increasing threat of climate change and associated warming of streams on fish populations across the Pacific Northwest and North America (Lynch et al. 2016), it is important to evaluate the extent to which water temperature and future climate change influence prespawn mortality in threatened salmon populations. In this study, we used salmon carcass recovery data from spawning ground surveys to estimate prespawn mortality rates of spring- and summer-run Chinook Salmon in six rivers within the Grande Ronde and Imnaha River basins of Northeast Oregon. The objectives of this research were 1) to evaluate the relationship between prespawn mortality and summer water temperature under current climate conditions (2000-2013), and 2) to predict how prespawn mortality rates would change in response to future climate change projections for the 2040s (2030-2059) and 2080s (2070-2099).



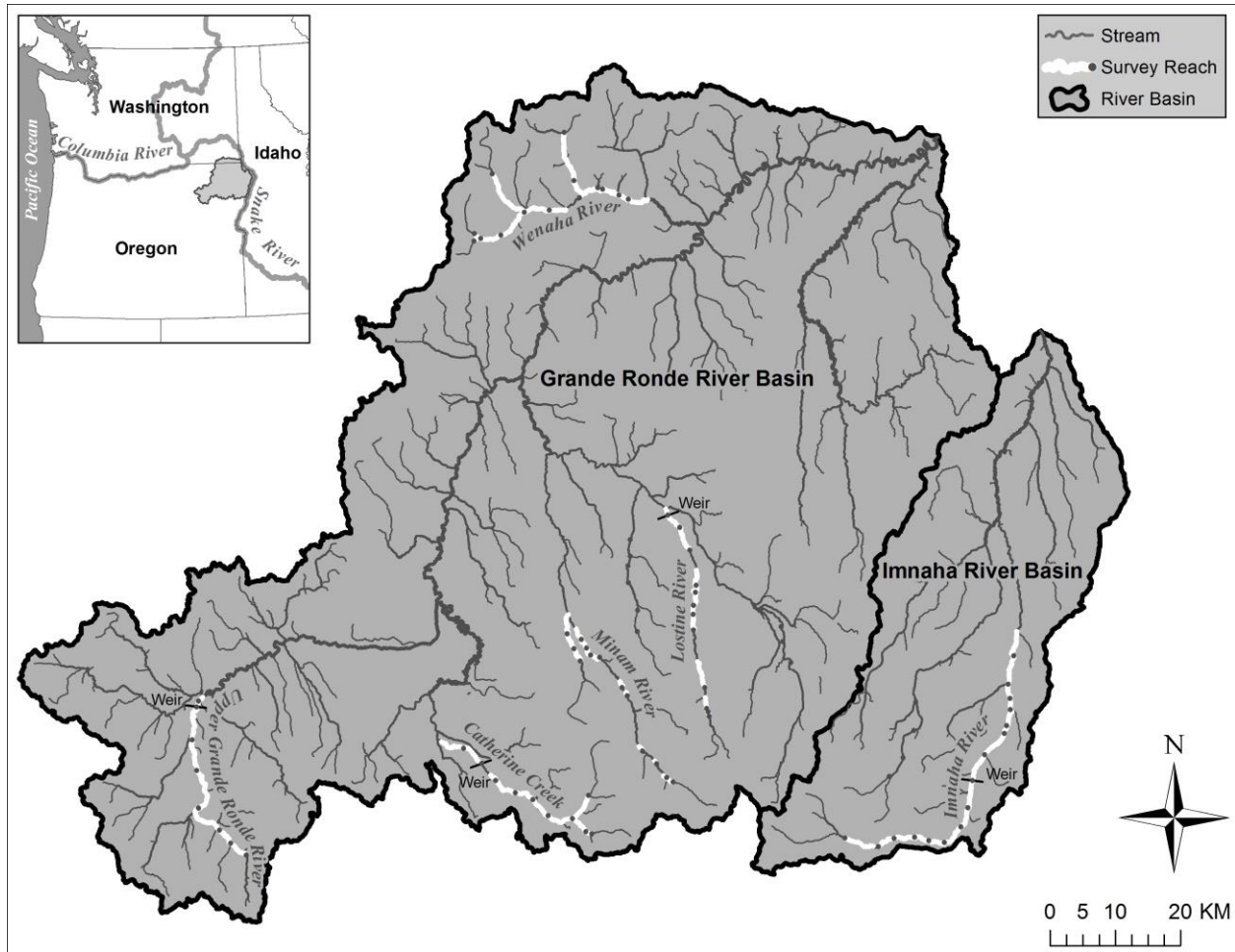
## Methods

### Study site

The study area included six streams within the Grande Ronde and Imnaha River basins of Northeast Oregon, USA (Figure 1). The Grande Ronde River is a major tributary of the Snake River, originating in the Blue Mountains of NE Oregon and flowing approximately 340 km north/northwest before joining the Snake River in Southeast Washington. From there, the Snake River flows another 272 km to its confluence with the Columbia River near river kilometer 523. The adjoining Imnaha River originates on the eastern side of the Wallowa Mountains of NE Oregon and flows north approximately 117 km to its confluence with the Snake River near river kilometer 309. Salmon must cross eight major dams during their migration from the Pacific Ocean to their spawning grounds in the Grande Ronde and Imnaha River basins.

This area is typified by cold winters with ample snow in its headwaters, and hot, dry summers. Basin tributaries are primarily fed by snowmelt, with peak flows occurring during the spring, and base flows occurring during late summer. The six rivers included in this study spanned a wide range of sizes and habitat conditions. Watershed area above each survey reach ranged from 35 to 589 km<sup>2</sup> (mean = 247 km<sup>2</sup>). Average catchment elevation ranged from 825 to 2,035 m (mean = 1,259 m) while 30-year average annual precipitation ranged from 37 to 134 cm (mean = 72 cm; data from Hill et al. 2016). Four of these rivers (Catherine Creek, Imnaha River, Upper Grande Ronde River, and Lostine River) have weirs installed near the lower extent of the spawning area to enumerate spawning escapement and to capture fish for hatchery brood stock. Two of the rivers (Minam River and Wenaha River) are managed as wilderness streams with no weirs, no hatchery releases, and relatively intact instream and riparian conditions.

Habitat for fish and other aquatic life in the Grande Ronde and Imnaha River basins has been steadily degraded since the mid-1800s due to land use, with water temperature being arguably one of the most impaired and influential factors for Chinook Salmon, steelhead, and bull trout (Ecovista 2004, Nowak 2004). Tributary and estuarine habitat degradation, combined with other factors such as hydroelectric operations in the mainstem Snake and Columbia rivers, predation, and commercial and sport fishing contributed to the decline and subsequent listing of the Grande Ronde/Imnaha Chinook Salmon populations as threatened under the Endangered Species Act (ESA) (NOAA 2008). The focal populations in this study were chosen because high prespawn mortality rates have been observed in some of these populations and managers need to better understand the factors influencing prespawn mortality. In addition, a long time series of carcass recovery data was available to conduct this analysis.



**Figure 1.** Study area showing locations of spawning survey reaches (highlighted in white) in the Grande Ronde and Imnaha River basins in NE Oregon.

### Prespawn mortality estimates

We used Chinook Salmon spawning survey data from the Oregon Department of Fish and Wildlife (ODFW) spanning 14 years (2000-2013), six rivers, and 66 unique survey reaches within the Grande Ronde and Imnaha River basins (Figure 1). Spawning surveys were conducted within selected river reaches ranging in size from 0.3 to 8.5 km (mean = 3.0 km). Surveys were intended to cover the entire spawning extent within each river, with gaps occurring because access to the survey was denied by landowners or for other safety or logistical reasons. Reaches were surveyed two to three times per year corresponding to the start, peak, and conclusion of spawning. For each female salmon carcass encountered during a survey, field crews recorded the “percent spawned” based on a visual estimate of egg retention (i.e., the percentage of eggs remaining in each carcass). Female salmon carcasses were designated as prespawn mortalities if greater than 50% of the eggs were remaining in the carcass (Feldhaus et al. 2017). Percent spawned was not estimated for male Chinook Salmon.

For each unique combination of reach, river, and year, we calculated prespawn mortality (*PSM*), defined as the proportion of mature (age 4-5) female Chinook Salmon that died prior to successfully spawning (*Y*) out of the total number of female carcasses that were sampled and for which spawning success could be

estimated ( $n$ ). This metric did not account for fish that died during upstream migration (en route mortality), those removed by fisheries, or fish collected for hatchery bloodstock. Data from reaches with fewer than five carcasses sampled per year were excluded from the analysis to avoid bias associated with low sample size. While greater than 10 carcasses per reach would be preferable (Bowerman et al. 2017), we were concerned that a higher sample size threshold would bias our analysis by disproportionately eliminating data from reaches with warmer water temperature and consequently, lower fish abundance. Additionally, data from rivers with fewer than 20 reach- and year-specific estimates of *PSM* were excluded from the analysis to ensure sufficient sample size to estimate a temperature effect on *PSM*. The total number of carcasses sampled per reach ranged from 5 to 369 (mean = 22.6). The final dataset consisted of 7,499 female fish carcasses, totaling 332 unique estimates of *PSM*.

### Water temperature data

We used water temperature predictions from the NorWeST model (Isaak et al. 2016) to estimate average August water temperature (*AvgAugTemp*) within each survey reach and year. The NorWeST model provided year-specific temperature predictions from 2000 through 2013 at points spaced approximately every kilometer throughout the stream network. This spatial statistical network model has been shown to produce accurate and unbiased predictions of water temperature (Mid-Columbia region model:  $R^2 = 0.94$ ; Root Mean Square Prediction Error =  $0.91\text{ }^{\circ}\text{C}$ ; Isaak et al. 2016). NorWeST prediction points that overlapped spawning reaches were joined to each reach using a spatial join tool in ArcGIS (ESRI 2012) and all prediction points were then averaged by reach and year.

NorWeST predictions of *AvgAugTemp* for the 2040s (2030-2059) and 2080s (2070-2099) future time periods were based on projected changes in August air temperature and stream discharge from an average ensemble of 10 global climate models with the lowest bias in simulating observed climate across the Northwest United States (Hamlet et al. 2013). Climate model simulations were downscaled using a spatially explicit delta method (Hamlet et al. 2013) to represent the A1B greenhouse gas emissions trajectory (IPCC 2007).

### Modeling analysis

We used a generalized linear mixed-effects model implemented with the lme4 package in Program R (Bates et al. 2015) to examine the relationship between August water temperature and prespawn mortality. Because the carcass recovery data was collected in the same locations year after year (i.e., repeated measures design), it wasn't appropriate to assume that annual replicates were statistically independent—a key assumption in general linear modeling. Additionally, river- and year-specific differences in environmental conditions (Rand et al. 2006, Roumasset 2012) or other population-specific parameters such as run timing (Hinch et al. 2012) could also create a lack of independence among observations of prespawn mortality. We accounted for this lack of statistical independence by using a mixed-effects model which included *Year*, *River* and *Reach* as random grouping effects. *Year* represented the random variation between years, and was intended to capture additional variation in prespawn mortality that was not explained by water temperature. The factor *River* represented the variation between rivers, and *Reach* represented the variation between reaches. Treating these grouping factors as random effects rather than fixed effect categorical variables is a more efficient analysis method because it uses fewer degrees of freedom (Zuur et al. 2009). In addition, treating *River* as a random effect allows us to make general predictions about the influence of water temperature on prespawn mortality in the typical river, including rivers outside of those included in our dataset.

We assumed the number of Chinook Salmon that died prior to spawning ( $Y$ ) was binomially distributed ( $Y \sim B(n, p)$ ), where  $p$  is the probability of prespawn mortality ( $PSM$ ). The final model formula was given by:

$$\text{logit}(p) = \ln\left(\frac{p}{1-p}\right) = \text{AvgAugTemp} + \text{Year} + \text{River} + \text{Reach},$$

where *AvgAugTemp* is the fixed effect explanatory variable, and *Year*, *River* and *Reach* are random effects as described above. This model structure assumed a common slope for the effect of temperature on *logit(p)* across all rivers and years. Given the sparse and unbalanced nature of the carcass recovery dataset, we assumed that inclusion of an interaction term between temperature and river (i.e., variable slopes by river) would likely lead to a spurious model fit. Model predictions were transformed from the linear logit scale to the non-linear “real” scale using the formula:

$$p = \frac{e^{\text{logit}(p)}}{1 + e^{\text{logit}(p)}}.$$

Model assumptions were evaluated using standard diagnostic plots generated with the DHARMA package in R (Hartig 2016) and by computing the dispersion parameter  $\hat{\phi}$ , which provides an indication of potential overdispersion in the data (Harrison 2014). The proportion of deviance explained by the model (pseudo- $R^2$ ) was computed using methods described in Nakagawa and Schielzeth (2013) and implemented in package MuMIn in R. We used the effects package (Fox 2003, Fox and Hong 2009) and sjPlot packages (Lüdtke 2017) in program R to compute and plot model predictions and associated confidence intervals.

## Results

Modeled August water temperatures varied considerably across years and rivers, as well as longitudinally within rivers (Figure 2). Mean August water temperatures were highest in the Upper Grande Ronde River (mean = 15.6 °) and lowest in the Wenaha (mean = 11.5 °C) and Minam rivers (mean = 12.9 °C). Longitudinal variation in water temperature within a river, as denoted by the spread of the data shown in each of the boxes in Figure 2, was lowest in the Minam River followed by the Lostine River, and was considerably higher among all other rivers. Inter-annual variation in water temperature was generally similar across rivers, with the highest water temperatures occurring in 2012, 2001, and 2005, and the lowest in 2002, 2010, and 2006.

Reach-specific prespawn mortality ( $PSM$ ) estimates between 2000 and 2013 were highly variable across rivers and years, ranging from 0 to 1 (mean = 0.15). Estimated  $PSM$  was lowest in the Wenaha River (mean = 0) and Minam River (mean = 0.04) and highest in the Upper Grande Ronde River (mean = 0.36; Figure 3).  $PSM$  rates were most variable in the Upper Grande Ronde, followed by the Imnaha and Lostine rivers. In contrast,  $PSM$  variability was moderately low in Catherine Creek, and was very low or negligible in the Minam and Wenaha rivers, the two wilderness streams. Average  $PSM$  by year ranged from 0.06 in 2000 to 0.29 in 2009 (Figure 4), with the highest degree of variability occurring in 2009.

The generalized linear mixed-effects model explained approximately 50% of the deviance in  $PSM$  (conditional quasi- $R^2 = 0.50$ ), while water temperature alone explained approximately 11% of the deviance (marginal quasi- $R^2 = 0.11$ ). Random- and fixed-effect model coefficients are shown in Tables 1

and 2, and random effect intercepts for each river, year, and reach are provided in supplemental materials (Tables 4-6). Average August water temperature had a statistically significant positive effect on the probability of *PSM* (coefficient for *AvgAugTemp* = 0.62,  $p < 0.0001$ ).

Model diagnostic plots indicated that assumptions of uniformly distributed residuals and homogeneity of variance were satisfied. The estimated dispersion parameter  $\hat{\phi}$  was 1.72, indicating a small to moderate amount of overdispersion in the data. Because the effect of water temperature on *PSM* was so highly significant, a small correction to the standard errors to account for overdispersion would not alter our biological conclusions (Zuur et al. 2009) and therefore we chose not to take additional steps to correct for overdispersion.

The model implies one average curve for the effect of water temperature on the predicted probability of *PSM* (i.e., thick black line shown in Figure 5) that can be shifted up or down for each river, reach and year. This average curve, also called the marginal effect, represents the effect of water temperature on the predicted probability of *PSM* when all other model terms (i.e., random effects *River*, *Year*, and *Reach*) are held fixed at their mean value. The average predicted probability of *PSM* increased from approximately 0.01 at 10 °C to about 0.48 at 18 °C, with predicted probability of *PSM* increasing sharply as temperatures exceeded approximately 15 °C (Figure 5). Confidence intervals around this curve were quite wide, indicating a high amount of river-, year-, and reach-specific variation as well as random noise in the data. Model uncertainty increased substantially with increasing water temperature, a likely result of increased variability in *PSM* estimates and smaller sample size at higher temperatures.

Inclusion of a random intercept for *River* produced very different predictions of *PSM* across rivers for a given value of water temperature (Figure 6). For example, the Wenaha River had the lowest predicted *PSM* across the range of temperatures sampled, Catherine Creek and the Minam River showed a moderate increase in predicted *PSM* as a function of water temperature, and all other rivers showed a relatively rapid increase in predicted *PSM* with increased water temperature.

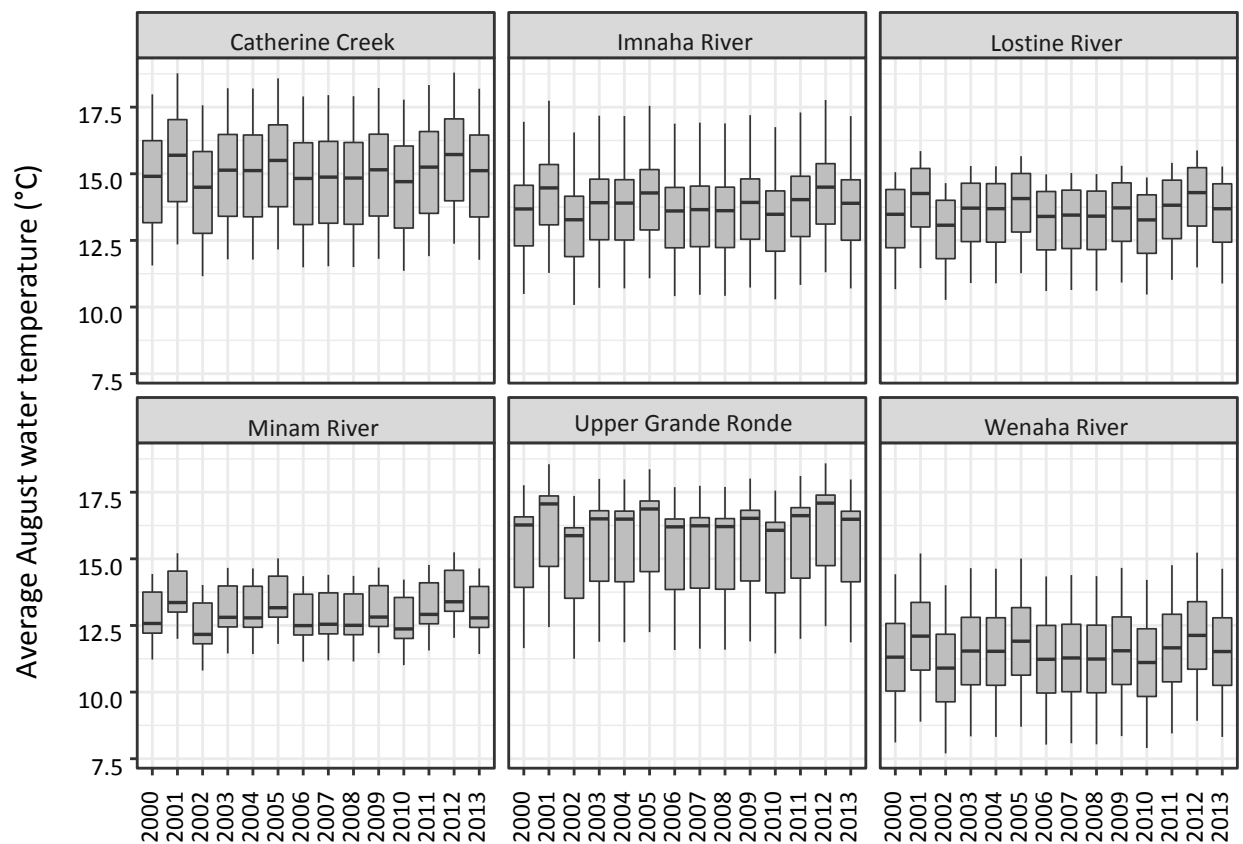
## Climate change projections

August water temperatures averaged across the entire study area were projected to increase from 13.6 °C under current conditions (2000-2013) to 14.8 °C by the 2040s (1.2 °C difference) and to 15.8 °C by the 2080s (2.2 °C difference; Table 3). Projected increases in water temperature did not differ substantially across rivers (range for 2040s = 1.2-1.3 °C; range for 2080s = 2.1-2.2 °C).

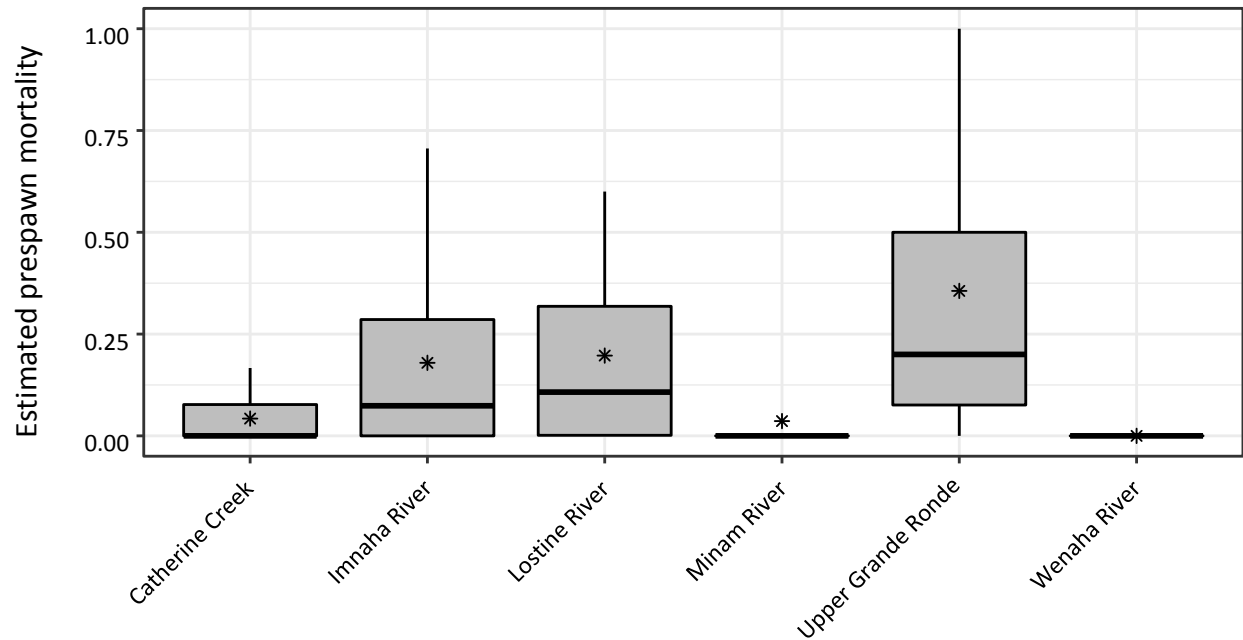
Prespawn mortality predictions for future climate change scenarios increased substantially in response to warming stream temperatures (Figure 7). Mean predicted probability of *PSM* increased from approximately 0.06 under current climatic conditions to approximately 0.12 by the 2040s (102% increase), and to 0.19 by the 2080s (230% increase). Uncertainty around these predictions was relatively large and increased substantially for future climate scenarios, coincident with higher predicted water temperatures.

The predicted increase in probability of *PSM* associated with climate change varied considerably across rivers (Table 3). For example, mean probability of *PSM* in the Upper Grande Ronde River was projected to increase from approximately 0.38 under current conditions to 0.57 by the 2040s (52% increase) and to 0.71 by the 2080s (88% increase). In contrast, probability of *PSM* in the Wenaha River, a much cooler wilderness stream, was predicted to increase from only 0.005 under current temperatures to 0.01 by the

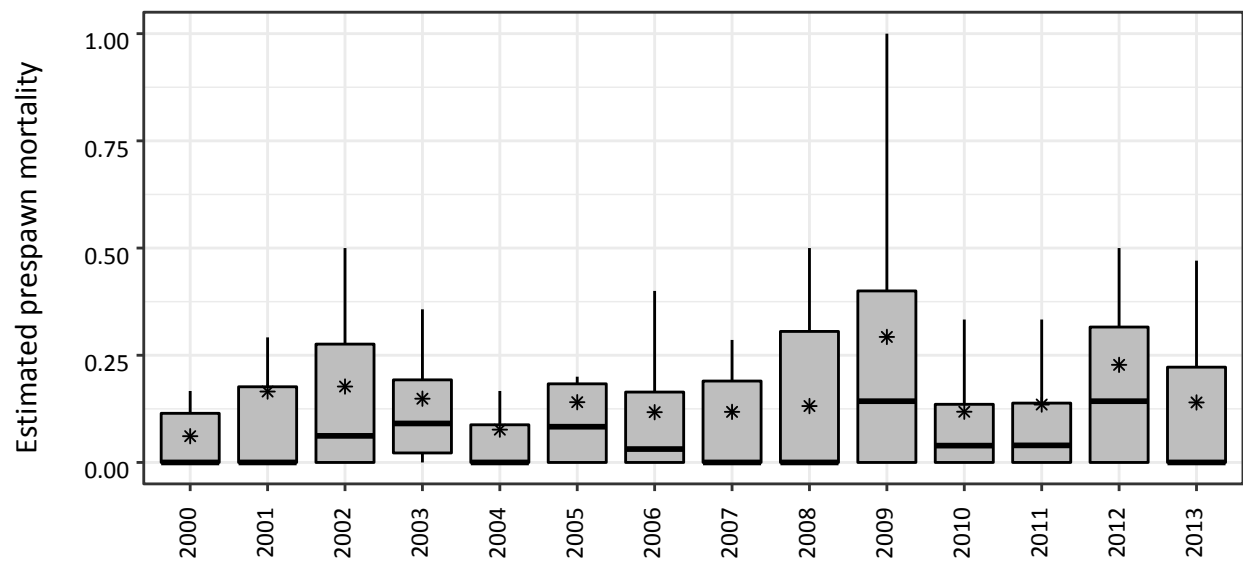
2040s (110% increase) and to 0.02 by the 2080s (265% increase). Although the relative (i.e., percentage) increase was much higher for the Wenaha River, the mean predicted probability of *PSM* in this stream remained very low for all climate scenarios. Similarly, predicted *PSM* remained relatively low in the Minam River for all climate scenarios, increasing from a mean of 0.02 under current conditions to 0.05 by the 2040s and to 0.09 by the 2080s.



**Figure 2.** Boxplots of average August water temperature (°C) by year and river. Bold horizontal lines represent medians, boxes represent the 25<sup>th</sup> and 75<sup>th</sup> percentiles, and vertical lines represent roughly two standard deviations.



**Figure 3.** Boxplot of reach-specific prespawn mortality estimates grouped by river for all years combined. Bold horizontal lines represent medians, asterisks represent means, boxes represent 25<sup>th</sup> and 75<sup>th</sup> percentiles, and vertical lines represent roughly two standard deviations.



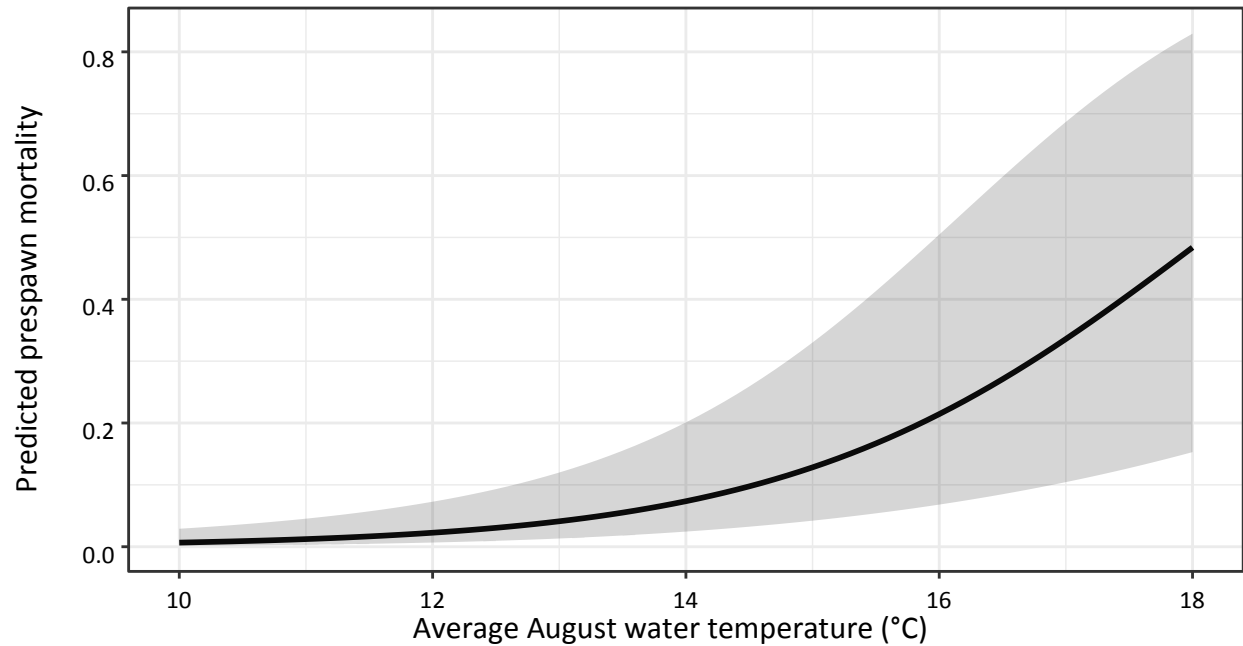
**Figure 4.** Boxplot of reach-specific prespawn mortality estimates grouped by year for all rivers combined. Bold horizontal lines represent medians, asterisks represent means, boxes represent 25<sup>th</sup> and 75<sup>th</sup> percentiles, and vertical lines represent roughly two standard deviations.

**Table 1.** Summary of random-effect model coefficients.

Random effects	Variance	Std. Dev.
<i>Reach</i>	2.035	1.427
<i>Year</i>	0.347	0.589
<i>River</i>	1.467	1.211

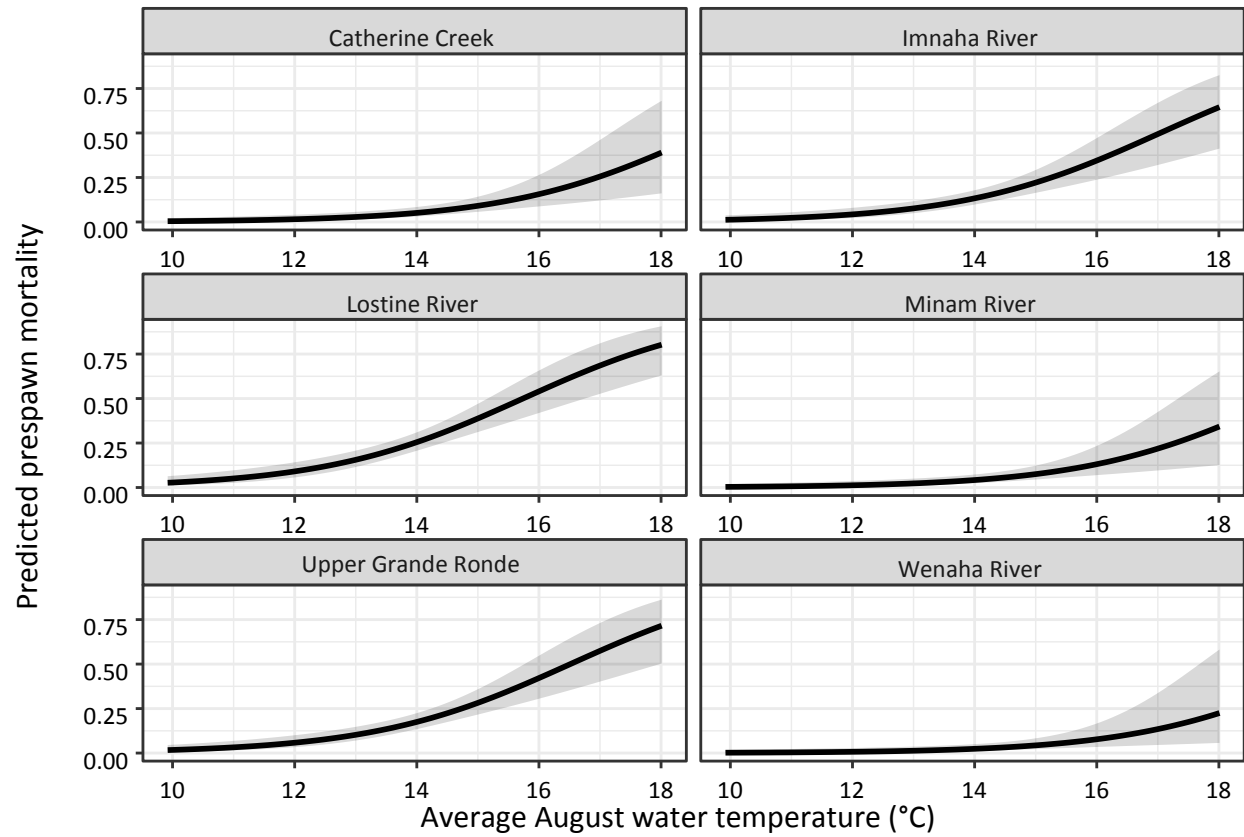
**Table 2.** Summary of fixed-effect model coefficients

Fixed effects	Estimate	Std. Error	z	P-value
(Intercept)	-11.170	1.939	-5.760	8.41E-09
<i>AvgAugTemp</i>	0.617	0.136	4.529	5.91E-06



**Figure 5.** Predicted probability of prespawn mortality as a function of mean August water temperature (°C) averaged across all reaches, rivers and years (i.e., marginal effect of *AvgAugTemp*). Grey shading represents 95% confidence intervals.

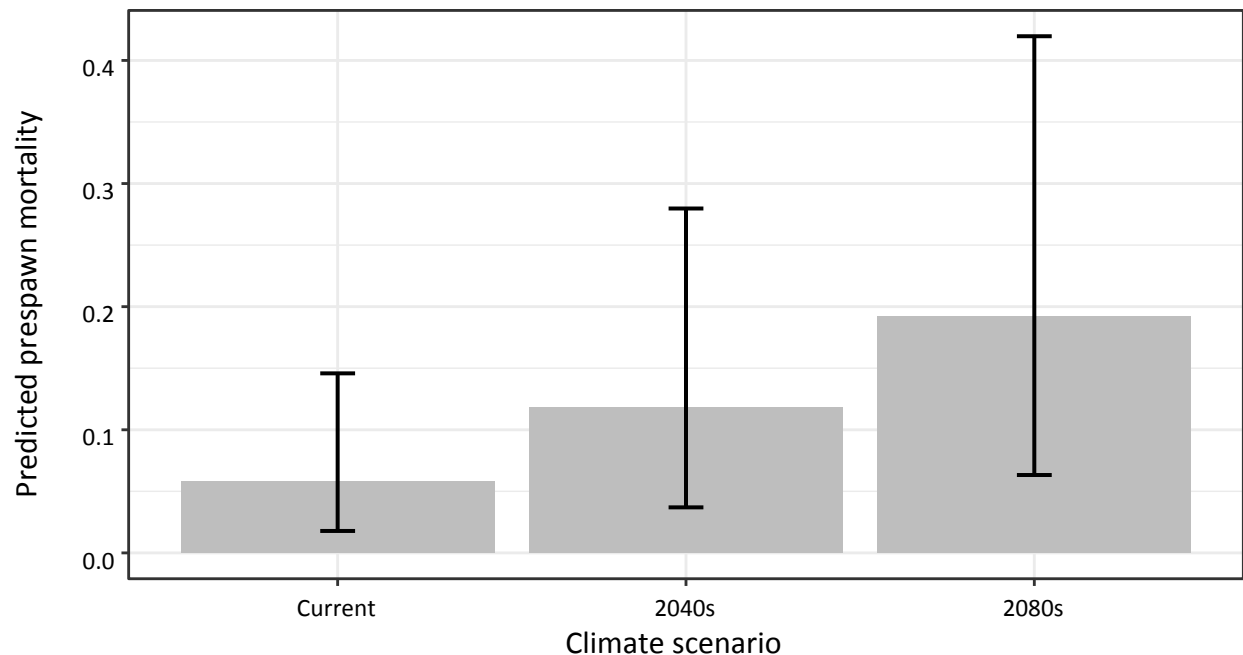




**Figure 6.** Predicted probability of prespawn mortality as a function of mean August water temperature (°C) grouped by river. Grey shading represents 95% confidence intervals.

**Table 3.** Predicted probability of prespawn mortality by climate scenario and river.

River	Avg. August Temperature (°C)	Predicted probability of <i>PSM</i>	Std. Error	Lower 95% Conf. Int.	Upper 95% Conf. Int.
<b>Current climate scenario (2000-2013)</b>					
Catherine Creek	14.9	0.084	0.140	0.017	0.565
Imnaha River	13.9	0.124	0.103	0.009	0.414
Lostine River	13.2	0.176	0.090	0.007	0.358
Minam River	13.0	0.024	0.075	0.006	0.299
Upper Grande Ronde	15.7	0.376	0.164	0.034	0.679
Wenaha River	11.5	0.005	0.038	0.003	0.153
Total	13.6	0.058	0.033	0.018	0.146
<b>2040s climate scenario (2030-2059)</b>					
Catherine Creek	16.1	0.167	0.186	0.034	0.763
Imnaha River	15.1	0.234	0.156	0.020	0.630
Lostine River	14.5	0.315	0.135	0.014	0.541
Minam River	14.3	0.050	0.116	0.014	0.467
Upper Grande Ronde	17.0	0.571	0.196	0.064	0.832
Wenaha River	12.7	0.011	0.066	0.007	0.267
Total	14.8	0.118	0.062	0.037	0.280
<b>2080s climate scenario (2070-2099)</b>					
Catherine Creek	17.1	0.265	0.206	0.055	0.864
Imnaha River	16.1	0.353	0.190	0.031	0.776
Lostine River	15.4	0.450	0.163	0.025	0.662
Minam River	15.2	0.085	0.146	0.021	0.594
Upper Grande Ronde	17.9	0.706	0.208	0.098	0.913
Wenaha River	13.7	0.019	0.096	0.012	0.386
Total	15.8	0.192	0.091	0.063	0.420



**Figure 7.** Predicted probability of prespawn mortality by climate scenario averaged across all reaches and rivers. Error bars represent 95% confidence intervals.

## Discussion

The observed positive effect of summer water temperature on prespawn mortality of spring- and summer-run Chinook Salmon populations in the Grande Ronde and Imnaha River basins is largely consistent with findings from similar studies in the Columbia and Willamette River basins (Contor and Schwartz 2007, Keefer et al. 2010, Roumasset 2012, Bowerman et al. 2017) and elsewhere in the Pacific Northwest (Heard 1991, Hinch et al. 2012). Perhaps the most relevant comparative study of *PSM* in Chinook Salmon populations was conducted in the Umatilla River, a nearby Eastern Oregon river with similar hydrologic and climatic characteristics (Contor and Schwartz 2007). These researchers found that *PSM* rates estimated from carcass recoveries increased exponentially as a function of seasonal maximum stream temperature, with the sharpest increases in mortality occurring at temperatures exceeding about 22 °C. For comparative purposes, we translated our predictor variable from *AvgAugTemp* to maximum annual water temperature (*MaxTemp*) to match the predictor variable used in Contor and Schwartz (2007) using a linear regression model developed from stream temperature measurements collected at 574 sites throughout the Columbia River Basin (CHaMP 2017;  $MaxTemp = 2.188 + 1.246 * AvgAugTemp$ ;  $R^2 = 0.72$ ). The resulting relationship between maximum annual water temperature and prespawn mortality was remarkably similar to the prediction curve from the Umatilla River, with predicted *PSM* rates at a given water temperature differing by only 3 percentage points on average and similar inflection points (i.e., steep increase in *PSM*) occurring at approximately 21-22 °C. One caveat to consider with this comparison is that the shape of the relationship between *MaxTemp* and *PSM* would likely change somewhat if the Grande Ronde/Imnaha model had actually been fit to maximum water temperature rather than simply transforming the x-axis as we did here. Similarly, Bowerman et al. (2017) found that *PSM*

rates of spring Chinook Salmon in Willamette River tributaries were consistently high (> 80%) in reaches with maximum water temperatures exceeding 20 °C. The similarity in predicted temperature effects on *PSM* rates across a range of different salmon populations in the Pacific Northwest is compelling and lends credibility to the broader applicability of our model results.

The positive association between water temperature and *PSM* likely reflects a combination of both direct thermal stress and indirect or sub-lethal effects. Laboratory studies on the acute lethality of water temperature on adult salmonids indicate that the upper incipient lethal temperature (i.e., the temperature at which 50% of the fish die after an exposure period of 1 to 7 days at constant temperature) is approximately 21-22 °C (McCullough et al. 2001), though this threshold can vary somewhat depending on the initial acclimation temperature. A lethality threshold of 21-22 °C corresponds fairly well with the temperature at which average *PSM* rates in the Grande Ronde/Imnaha basins began to rise steeply (i.e., average August temperature of 15-16 °C, or maximum annual temperature of 20.9-22.1 °C as derived from the regression equation described above). Though direct lethal impacts from high water temperature likely contributed to elevated *PSM*, particularly in very warm reaches such as the lower portions of the Upper Grande Ronde River, the cumulative thermal stress from indirect factors such as increased metabolic demand (Fenkes et al. 2016), migration delay (Sauter et al. 2001), and elevated disease risk (Materna 2001, Karvonen et al. 2010, Ray et al. 2012) also likely contributed to an overall reduction in migration fitness and associated increased prespawn mortality.

Our results indicated that, in the absence of rapid population adaptation, future climate change and associated stream warming could increase the average probability of prespawn mortality in the Grande Ronde/Imnaha River basins from approximately 0.06 under current conditions to 0.12 by the 2040s and to 0.19 by the 2080s. This translates to a decline in spawner abundance of 6% by the 2040s and 14% by the 2080s relative to current conditions (2000-2013). These projected impacts are likely conservative because they do not account for other life stages that could potentially be impacted by climate warming such as reduced viability of gametes in holding adults (McCullough et al. 2001), reduced juvenile rearing survival and abundance (Crozier and Zabel 2006, Justice et al. 2017), increased energetic demands during upstream migration (Rand et al. 2006, Fenkes et al. 2016), or changes in ocean productivity (ISAB 2007).

While numerous studies have identified climate change and associated changes to freshwater and marine habitats as a serious potential threat to Pacific salmon survival and reproductive success (Crozier and Zabel 2006, Ficke et al. 2007, Mantua et al. 2010, Beechie et al. 2013, Lynch et al. 2016), relatively few have attempted to quantify future climate change impacts to salmon abundance or survival. Among the studies that have, Justice et al. (2017) predicted that water temperature increases associated with climate change could reduce juvenile Chinook Salmon parr abundance in the Grande Ronde River basin by 36-53% by the 2080s in the absence of other habitat perturbations. Similarly, Battin et al. (2007) predicted a 20-40% decline in Chinook Salmon spawner abundance by 2050 resulting from climate change in the Snohomish River basin. In a modeling analysis of spring Chinook Salmon in the Salmon River drainage of Idaho, Crozier et al. (2008a) predicted that increased air temperature and reduced streamflow associated with climate change could reduce mean salmon abundance by 20-50% and increase the probability of quasi-extinction from 0.1-0.4 to 0.3-0.9 by the 2040s. In addition, Muñoz et al. (2014) predicted a 5-55% chance of catastrophic loss of Quinsam River Chinook Salmon populations by 2075 resulting from climate warming. The potential climate change impacts to salmon populations we described for the Grande Ronde/Imnaha River basins as well as impacts identified by researches in other

Pacific Northwest watersheds underscore the serious potential threat that climate change poses to future viability of these threatened salmon populations and emphasizes the urgent need for mitigation measures such as stream restoration and reduction of greenhouse gas emissions. We acknowledge that models of climate change and salmon population dynamics are inherently uncertain and that responses of salmon populations to climate change are likely to vary considerably because of geographic differences in hydrology and habitat conditions (Crozier and Zabel 2006), genetic differences how individual fish and populations respond to future changes in water temperature regimes (Steel et al. 2012), and because of salmonids' ability to adapt to environmental changes with plastic and evolutionary changes in behavior (Crozier et al. 2008b, Lynch et al 2016, Mantua et al. 2015) and physiology (Muñoz et al. 2014). However, this uncertainty should not preclude managers from identifying populations that are at high risk from climate change and taking prudent measures to mitigate for future climate impacts.

Although we did not evaluate the potential for habitat restoration to offset projected climate impacts to salmon survival, similar studies have shown that restoration actions such as improving riparian vegetation cover, narrowing and deepening of stream channels, reduction of impervious surface cover, and improving floodplain and side channel connectivity can substantially reduce water temperature and thereby mitigate climate impacts to salmon (Battin et al. 2007, Justice et al. 2017). For example, Battin et al. 2007 found that habitat restoration in the Snohomish River basin had the potential to partially or completely mitigate climate change impacts, particularly in lower elevation streams. Similarly, Justice et al. (2017), found that complete restoration of riparian vegetation cover and channel narrowing in the Grande Ronde River basin could more than offset projected climate impacts to juvenile salmon abundance. Given that adult salmon in the Grande Ronde basin hold and spawn in many of the same habitats that juveniles use for rearing, it's likely that the projected reductions in water temperature from stream and riparian restoration described in Justice et al. (2017) would have similar benefits for survival of adults prior to spawning. Based on a decision support framework and literature review of the ability for habitat restoration actions to ameliorate climate change effects, Beechie et al. (2013) concluded that restoration actions that restore floodplain connectivity, restore stream flow regimes, and re-aggrade incised channels are most likely to improve population resilience to climate change. Additionally, actions that preserve or enhance heterogeneity in habitats and genetics of salmon populations will likely play a very important role in promoting salmon resilience to future climate change (Beechie et al. 2013, Mantua et al. 2015).

Although we detected a statistically significant relationship between probability of *PSM* and water temperature, there was a substantial amount of uncertainty in model predictions. The combination of fixed and random effects explained only 50% of the variability in the data, indicating that other environmental or biological factors not included in the model likely influenced the probability of *PSM*. Some factors that may influence *PSM* but were not included in our model include the percentage of hatchery-origin spawners in the population (*PHOS*; Bowerman et al. 2017), spawner density (Quinn et al. 2007, Tillotson and Quinn 2017), and the presence of weirs (Newton et al. 2017). Bowerman et al. (2017) identified a significant positive effect of *PHOS* on the probability of *PSM* of spring Chinook Salmon in Willamette River tributaries. However, *PHOS* in the Grande Ronde and Imnaha River basins is largely confounded with water temperature—rivers with high water temperatures generally have larger proportions of hatchery fish and visa-versa. For this reason, we chose not include *PHOS* as a variable in our model, and instead focused on water temperature, which has clear physiological implications for salmon survival and substantial support in the literature as an important factor affecting prespaw mortality (Heard 1991,

Contor and Schwartz 2007, Keefer et al. 2010, Roumasset 2012, Hinch et al. 2012, Bowerman et al. 2017). High spawner density has also been linked to elevated rates of prespawn mortality in salmon populations. In a study of Sockeye Salmon in small Alaska stream, Tillotson and Quinn (2017) found that a very large spawning escapement (~ 12 times the 54-year median) led to *PSM* rates of approximately 55%. The authors concluded that this high rate of mortality was ultimately a density-dependent process that was exacerbated by low-flow conditions and reduced dissolved oxygen. In the Grande Ronde/Imnaha basins, salmon populations are well below target abundance levels for recovery (NOAA 2008) and therefore, we assumed that *PSM* was not significantly influenced by fish density. It is also possible that weirs could impact survival of salmon through handling stress or by impeding upstream passage, thereby forcing fish to spend longer amounts of time in lower river reaches where water temperatures are higher and spawning habitat may be less suitable (Newton et al. 2017). While it would be tempting to include a weir factor in the model to test for a weir effect on *PSM*, we determined that this would not be appropriate for several reasons. First, a weir effect would be potentially confounded with water temperature since weirs are located near the downstream extent of the spawning range where water temperature is higher. Thus, it would be unclear whether prespawn mortalities found below a weir perished because of migration blockage or water temperature or a combination of both. Second, fish spawning above a weir may have experienced similar handling stress or delayed migration as compared to fish spawning below a weir, but these impacts are not evident from the final carcass recovery location. Third, the Minam and Wenaha rivers, which do not have weirs, are wilderness streams with generally cooler water temperatures and better habitat conditions, so the expected probability of *PSM* would likely be lower regardless of the lack of weirs in these streams. A more robust method for evaluating a weir effect would be a before and after control impact (BACI) design or similar control/treatment design in which *PSM* is estimated before and after the installation of a weir, and is compared with a control stream or set of streams in which no weirs were installed.

High uncertainty in our model of *PSM* rates was also likely related to inherent bias and poor accuracy associated with carcass recovery methods. Estimates of prespawn mortality based on carcass recoveries can be influenced by variation in detection probability of fish carcasses (DeWeber et al. 2017) with factors such as stream size, discharge and water clarity, observer effort and experience, fish behavior, fish sex, fish size, and the timing and frequency of spawning surveys all having a potential influence on detection probability (Bowerman et al. 2016, Zhou 2002). Among these factors, the timing of spawning surveys relative to when fish arrive at the spawning grounds has been shown to be one of the biggest contributors to bias in estimates of prespawn mortality (Bowerman et al. 2016). Using a detailed dataset of Chinook Salmon carcass recovery data from central Idaho, Bowerman et al. (2016) found that spawning surveys conducted over a limited time window (i.e., during the spawning period only) accounted for only 12.8% on average of the total prespawn mortality that was observed over the entire sampling period (i.e., surveys starting 4-5 weeks prior to spawning and ending 1 week after the last spawner was observed). Given that spawning surveys in the Grande Ronde and Imnaha River basins were generally conducted during the spawning period only (July 15–September 15), we assumed that estimates of total prespawn mortality were negatively biased. That said, it's unclear how this potential bias would influence the relative effect of water temperature on probability of *PSM* as described by the model. Given that surveys were conducted during the middle of the summer when water temperatures are most likely to influence fish survival, we assumed that our model captured the most biologically relevant temperature effects on the probability of *PSM* and that the slope of this relationship would not change significantly as

a result of the truncated timing of the spawning surveys. Future estimates of prespawn mortality could be improved by expanding the time window of spawning surveys (Bowerman et al. 2016), and by using a mark-recovery method to estimate prespawn mortality which accounts for imperfect detection probability and unknown spawning success (DeWeber et al. 2017). However, such methods may not be feasible depending on run size (i.e., very low run size could result in insufficient numbers of marked recoveries to generate a reliable estimate) and funding constraints.

Model precision was also likely related to the use of modeled water temperature data as a predictor variable. We used model predictions of average August water temperature instead of field-based water temperature measurements because field measurements were not consistently available at the reach-scale, a requirement for evaluating how the probability of *PSM* would vary in response to longitudinal differences in stream temperature. One consequence of this is that modeled temperatures are imperfect estimates of true water temperature, and therefore using them as a predictor variable introduces some additional variability into the model predictions. Thankfully, the model from which these predictions were generated was unbiased and very precise ( $R^2 = 0.94$ ; Root Mean Square Prediction Error =  $0.91\text{ }^{\circ}\text{C}$ ; Isaak et al. 2016). Another potential drawback of using the NorWeST data was that temperature predictions were only available as August means. It's possible that alternative temperature metrics such as cumulative degree days, maximum 7-day average daily maximum temperature (Bowerman et al. 2017) or maximum annual water temperature (Contor and Schwartz 2007) would provide more precise or informative relationships with probability of *PSM*. For example, fish that spawn in cooler headwater reaches of the Upper Grande Ronde River may still exhibit higher prespawn mortality due to cumulative exposure to high temperatures during holding or migration in warmer downstream reaches, and therefore, a metric such as cumulative degree days may yield a more precise model fit. Unfortunately, the requisite data on holding locations and migration timing for this type of analysis were not available. In addition, Isaak et al. (2016) found that several commonly used metrics used to describe water temperatures during the low flow summer period were highly correlated, suggesting that alternative temperature metrics would show a similar relationship with *PSM*. Furthermore, the potential drawbacks of the NorWeST data are largely outweighed by its broad availability throughout the Pacific Northwest and Mountain West regions and its inclusion of climate change scenarios, traits which open the possibility for similar analyses investigating temperature impacts to salmon populations across a much larger geographic range.

This study focused on factors influencing the probability of spawning success for mature Chinook Salmon that were staged on the spawning grounds, but did not account for en route mortality. En route mortality can account for a significant portion of the total mortality experienced by a salmon population throughout its life cycle and is often associated with high water temperatures (Rand et al. 2006, Crossin et al. 2008, Macdonald et al. 2010), infectious disease (CDFG 2004, Benda et al. 2015), or depleted energy reserves (Rand et al. 2006, Cooke et al. 2006). While en route mortality is an important metric to consider for overall management of salmon populations, it was not included in this analysis because accurate estimates of en route mortality were not available. Additionally, characterizing the effects of environmental factors on en route mortality would require detailed information about migration timing, holding locations, and temperature exposure along the migration route (Cooke et al. 2006), which was not available. A radio-telemetry study focusing on the causes and overall rates of en route and prespawn mortality of Grande Ronde and Imnaha salmon populations could help identify important survival bottlenecks and lead to improvements in future management and recovery strategies.

## Summary

Describing how environmental conditions such as water temperature influence the probability spawning success of ESA-listed salmon populations will become increasingly important as water temperatures rise in response to climate change (Bowerman et al. 2016). Despite considerable spatial and temporal variability in the effect of water temperature on the probability of prespawn mortality, these results add to a substantial body of evidence linking high water temperature to increased *PSM* rates in Pacific Northwest salmon populations. Projected increases in mortality associated with climate change highlight the need for continued habitat restoration or other management actions to improve temperature conditions for fish in temperature-impaired watersheds. In addition, the functional relationship between water temperature and probability of spawning success described here could be useful in salmon life cycle models to predict how future changes in water temperature could influence salmon population viability parameters. Finally, the concept of applying a freely-available, broad-scale temperature model to explain fine-scale patterns in vital rates of a threatened species may prove useful for researchers lacking empirical temperature data and wishing to explore similar relationships in other basins or for other species or life stages.

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## Supplemental Material

**Table 4.** Random effect intercepts by year.

<i>Year</i>	(Intercept)
2000	-0.380
2001	0.618
2002	0.691
2003	0.298
2004	-0.486
2005	-0.436
2006	0.086
2007	0.203
2008	-0.084
2009	0.713
2010	-0.305
2011	-1.072
2012	-0.340
2013	0.716

**Table 5.** Random effect intercepts by river.

<i>River</i>	(Intercept)
Catherine Creek	-0.388
Imnaha River	0.658
Lostine River	1.460
Minam River	-0.592
Upper Grande Ronde River	0.982
Wenaha River	-1.179

**Table 6.** Random effect intercepts by reach.

<i>Reach</i>	(Intercept)	<i>Reach</i>	(Intercept)
CATH1	1.913	LOS5	0.583
CATH2	-0.154	LOS6	1.094
CATH3	0.495	LOS6.1	0.658
CATH4	0.238	LOS7	-0.220
CATH5	-0.578	LOS8.1	0.763
CATH6	0.078	LOS8.2	-1.780
CATH7.1	-1.904	LOSW	3.171
CATH7.2A	-0.625	MIN1	-0.160
IMN3	-1.248	MIN3	1.546
IMN4.1	-0.458	MIN4	-0.612
IMN4.1A	-0.944	MIN5	0.878
IMN4.1B	-1.444	MIN6	-0.448
IMN4.2	-0.322	MIN7.1	-0.100
IMN4.2A	-1.590	MIN7.2	-0.353
IMN4.2B	-1.139	MIN7.3	-1.204
IMN5	0.782	UGR1	0.726
IMN5.1	0.679	UGR1.1	1.358
IMN5.2	1.102	UGR1.2	1.492
IMN6	-0.311	UGR3.2	-0.972
IMN7.1	0.562	UGR3.2B	-2.487
IMN7.1A	0.662	UGR3.2x	1.598
IMN7.2A	2.018	UGR3.3	0.197
IMN7.3	-0.464	UGR3.3X	-1.589
IMNW	3.028	UGR3.4A	-1.658
LMIN8	-0.173	UGR3.4B	1.338
LMINA	-0.196	UGRA	1.359
LOS1.3	-0.500	WEN3.1	-0.053
LOS2.1	-0.451	WEN3.2	-0.295
LOS2.2	0.440	WEN4.1	-0.204
LOS3	0.783	WEN4.2	-0.357
LOS4	-0.843	WEN5.B	-0.319
LOS4.1	-0.862	WEN6.1	-0.182
LOS4.2	-0.812	WEN6.5	-0.224

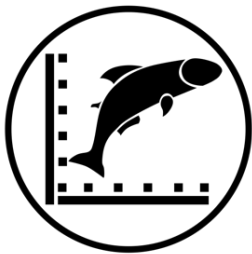
## **Appendix B – Life Cycle Model**



# LIFE CYCLE MODEL FOR THE UPPER GRANDE RONDE AND CATHERINE CREEK

*AN EVALUATION OF SPRING CHINOOK RECOVERY POTENTIAL*

*JANUARY 2018 – MARCH 2018 PROGRESS REPORT*



PREPARED BY: ECO LOGICAL RESEARCH

Nick Weber, Carl Saunders, & Peter McHugh



FOR: THE COLUMBIA INTER-TRIBAL FISH COMMISSION

**March 2018**

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## INTRODUCTION

The following report describes continued progress toward assessment of Chinook salmon populations and habitat restoration opportunities within Catherine Creek and the Upper Grande River watershed using a life-cycle modelling (LCM) framework. The work presented here extends life-cycle development efforts previously initiated by the Columbia River Inter-Tribal Fish Commission (CRITFC) and included as part of the Independent Scientific Advisory Board's (ISAB) 2017 review of LCM efforts in the interior Columbia Basin (chapter 9.f in ISAB 2017). Ultimately, these efforts are intended to provide an analytical tool that can be used to guide the development of long-term restoration and recovery strategies for two threatened Chinook populations under changing climatic conditions. However, much of the work presented here focuses on the continued refinement of the LCM framework. Accordingly, the current LCM development effort for Catherine Creek and the Upper Grande Ronde River sought address the following specific objectives:

1. Complete development and validation of LCM productivity and capacity terms for the Upper Grande Ronde Chinook population and use this updated model to evaluate simulations of potential restoration effectiveness described by Justice et al. (2017).
2. Use up-to-date data from Oregon Department of Fish and Wildlife (ODFW) to refine parr survival terms and incorporate life-history specific survival rates for the dominate life-histories (i.e., fall vs. spring migrants) exhibited within in each population.
3. Incorporate revised parr capacity estimates based on parameters obtained via a Beaverton-holt stock-recruit function fit to annual estimates of parr and spawner abundance.
4. Develop restoration scenarios for spawner capacity according to the availability of habitat that is thermally suitable for pre-spawn survival as described in Justice et al. (2017).
5. Provide a more comprehensive description of the analytical approaches used to develop capacity and productivity terms across life-stages.
6. Simulate population performance under restoration scenarios described by Justice et al. (2017) that consider the effect of future temperature, habitat, and climate conditions on early life-stage and spawner capacity.

The following pages provide a general overview of the underlying LCM framework, a description of population parameter estimation, and results of population performance under varying climatic and restoration scenarios. Current model performance is evaluated to guide future model refinement.

# MODELLING FRAMEWORK

## ISEMP LIFE-CYCLE MODEL

The LCM used here is based on a stage-structured, stochastic projection model, adapted from Sharma et al. (2005) and implemented by the Integrated Status and Effectiveness Monitoring Program (Nahorniak and Armour 2017) in the R statistical computing language (R Core Team 2014). This LCM propagates cohorts across a complete life cycle according to a series of stage-specific Beverton-Holt ‘spawner’ ( $N_i$ ) and ‘recruit’ ( $N_{i+1}$ ) relationships (after Moussalli and Hilborn 1986) governed by stage-specific capacity ( $c_i$ ) and productivity ( $p_i$ , maximum recruits per spawner) parameters:

$$\text{Equation 1.} \quad N_{i+1} = \frac{N_i}{\frac{1}{p_i} + \frac{1}{c_i} N_i}$$

In addition to stage-specific survival and capacity, the modelling framework allows users to specify estimates of parameter spatial and temporal variability in  $p_i$  and  $c_i$  giving rise to a stochastic model that is more reflective of natural processes.

Although the functional form of the model implies that density dependence occurs at all stage transitions, density-independent transitions are modeled by setting capacity to an infinitely large value and using empirical estimates of survival ( $S_i$ ) as the productivity input ( $p_i$ ). Otherwise, realized survival ( $S_i = N_i / N_{i+1}$ ) is a function of both the capacity and productivity for each modelled life-stage of the population. These parameters, modeled within the Beverton-holt stock-recruit relationship, form the basis for modelling population responses to restoration and climatic scenarios evaluated for the Upper Grande Ronde and Catherine Creek chinook populations (see Restoration Scenarios below).

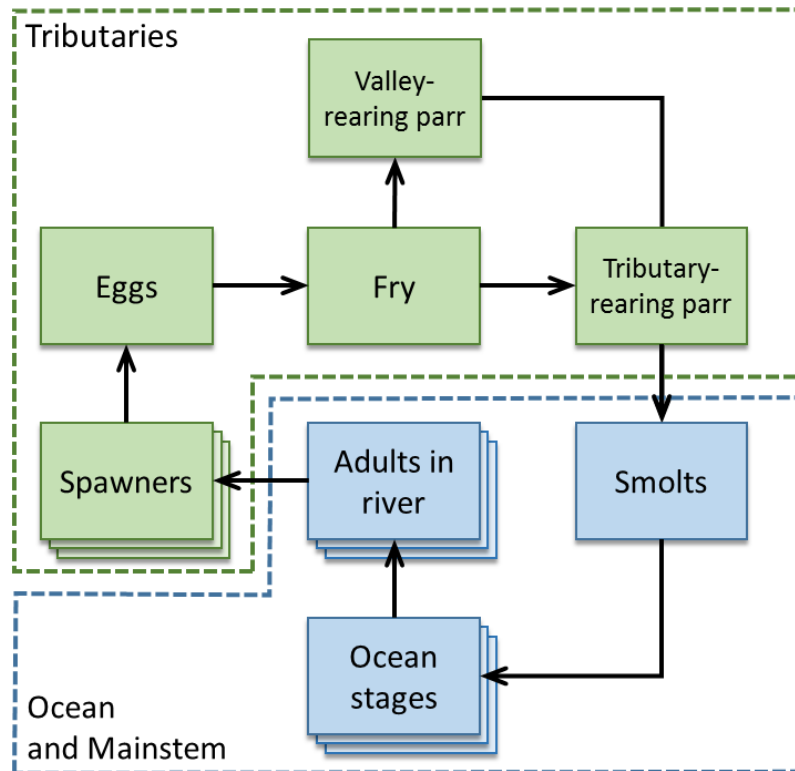
## LCM FRAMEWORK

In most LCM applications, capacity and productivity parameters appropriate for the population of interest are difficult to obtain, as these parameters are rarely estimated as part of routine monitoring programs. Here, however, a combination of entities monitors the Upper Grande Ronde (UGR) and Catherine Creek (CC) in such a way that estimating these parameters using population-specific data is possible. the Oregon Department of Fish and Wildlife (ODFW), Confederated Tribes of Umatilla Reservation (CTUIR), Columbia River Inter-Tribal Fish Commission (CRITFC), and other parties collect data needed to estimate spawner, parr, and smolt abundance and/or survival each year. Additionally, as part of the Columbia Habitat Monitoring Program (CHaMP) and Integrated Status and Effectiveness Monitoring Program (ISEMP), population-specific habitat data are collected and analyzed in such a way that permits the estimation of adult salmonid carrying capacities as a function of current habitat conditions.

Currently, our LCM framework uses egg-to-parr and parr-to-smolt productivity inputs derived from estimates generated by ODFW’s ‘Early Life Histories’ monitoring program. ‘Ocean’ (i.e., from LGD to LGD) survival and adult maturation schedules are estimated using age-specific smolt-to-adult return rate estimates (SARs) generated by the Fish Passage Center, similar to those used in B. Lessard’s LCM

(McCann et al. 2016)<sup>1</sup>. Our model uses capacity terms estimated and modelled at two freshwater tributary life stages, summer parr and adult spawners (expressed in terms of eggs). Parr capacity was estimated by fitting a simple spawner-recruit curve to available abundance estimates obtained from ODFW. Estimates of spawner capacities were based on a habitat suitability model (see McHugh et al. 2017 for an example) developed by ISEMP that relies on habitat metrics generated using the CHaMP protocol.

Where necessary, other LCM parameters were informed by literature values, and complete details describing the base model parameterization (scenario ‘Curr’ in Table ) for the UGR and CC, including assumptions about stochasticity are presented in Appendix B-i.



**Figure 1.** Conceptual diagram of the life cycle model (LCM) framework used to assess restoration scenarios in the Upper Grande Ronde (UGR) and Catherine Creek (CC). Stacked boxes represent life stages for which multiple ages exist and are tracked accordingly. Our model also tracks two dominate out-migration strategies within the UGR and CC which include migrating downstream and overwinter in the Grande Ronde valley (‘Valley’) or remaining in the upper mainstem and tributaries of each system (‘Tributary’). Smolts are enumerated at Lower Granite Dam (LGD), where ‘ocean stages’ begin and are modeled via LGD-to-LGD smolt-to-adult return rates.

<sup>1</sup> Note that the SARs assumes a single ‘average’ outmigrant experience and make no attempt to parse transported vs. in-river individuals; thus the assumption is that conditions during the 2001-2014 smolt migration years are a reasonable average of future mainstem/ocean conditions.

## HATCHERY SUPPLEMENTATION

The UGR and CC spring Chinook populations are presently supplemented by the annual release of *ca.* 250K (UGR) and 150K (CC) hatchery-reared smolts ( $N_{sm-sup}$ ), that are released into acclimation ponds near the spawning grounds in both systems. Adults returning from supplementation releases (HOR) are meant to home to the spawning grounds and spawn naturally with natural-origin (NOR) fish. To support an integrated hatchery program a fraction of natural-origin fish returning to spawn in the wild are retained for spawning in the hatchery. Weirs and adult traps situated in both streams are used to manage three aspects of escapement under the supplementation goals: (i) the proportion of the total natural-origin return retained for the hatchery ( $P_{NOS-R}$ ), (ii) The proportion of natural spawners of hatchery origin ( $P_{HOS}$ ), and (iii) the proportion of hatchery broodstock/egg-take of natural origin ( $P_{NOB}$ ). These supplementation efforts have a strong influence on abundance dynamics in both the CC and UGR populations. Thus, measures have been taken to alter the original ISEMP LCM code to allow a more accurate representation of this management strategy in LCM simulations. More specific descriptions and representations of how the supplementation management strategy was incorporated into the CC and UGR LCM frameworks are described in Appendix B-ii.

## BASE MODEL PARAMETERIZATION

Although an overview of the parameters used in our LCM framework can be found in Appendix B-i, in some cases a more thorough description of the analytical methods used in model development may be warranted with the intent that increased detail will assist in future efforts to refine and apply the LCM to novel management scenarios.

### EGG TO PARR SURVIVAL

Model egg-to-parr survival was based abundance data supplied by ODFW's early life-history monitoring program. This monitoring includes annual estimates of redds within CC and the UGR LCM extent as well as an estimate of parr abundance during late summer prior to the downstream fall migration exhibited by a portion of the population. Egg-to-parr survival estimates were constructed by first estimating egg equivalents based on the number of redds in year  $t$  as a function of ocean-age specific fecundity values (see below). Annual survival was then estimated as the ratio of eggs in year  $t$ , to the estimate of late summer parr in year  $t+2$ . We used the median and standard deviation of annual survival estimates as inputs to the LCM. It is also worth noting, that years prior to 2002 when hatchery spawners were excluded from spawning naturally above the trap/weir locations were excluded from the final estimate. This modification considerably reduced the egg-to-parr survival estimate for the Catherine Creek population.

### PARR CAPACITY

The previous version of the CC LCM (see (ISAB 2017)) used a statistical model that estimated parr capacity based on CHaMP habitat survey metrics. In the revised LCM framework, parr capacity is estimated using spawner and parr abundance data for the CC and UGR populations, obtained from ODFW. Carrying capacity was estimated by fitting a Beverton-Holt relationship between spawners in year  $t$ , and the late-summer parr estimate in year  $t+2$ . Although these approaches are considerably different, parr capacity as estimated from the Beverton-Holt fit were similar to those previously estimated using the statistical approach.

## PARR TO SMOLT AT LGD

In both the CC and UGR watersheds, spawning primarily occurs in headwater reaches, above the location of the smolt trap. However, a substantial proportion of the parr produced in these areas migrate downstream during fall and overwinter below the smolt trap, in lower valley of the Grande Ronde. Further, data collected during 1994-2016 by ODFW suggest that the survival rates between these two life history strategies differ. Based on tributary tagging efforts, and detections at LGD and elsewhere in the mainstem Columbia, survival rates for the two life history strategies can be estimated thereby increasing the realism of the LCM framework. This approach integrates spatially explicit survival rates for parr rearing in either the headwaters or valley reaches of these watersheds.

One challenge to integrating different overwinter survival rates for headwaters and valley occupants is that the life stages modeled in the LCM for both CC and UGR do not coincide with the timing of juvenile chinook tagging efforts conducted by ODFW. That is, the LCM for both populations incorporates a productivity term to model the transition from parr to smolts at LGD that approximately coincides with the time period between late summer parr (for which survival and abundance are estimated, see above) and smolts at LGD. In contrast, empirical estimates of overwinter survival for tributary and valley life history variants are derived from sampling efforts conducted during mid-autumn (ca. Oct-Dec). Therefore, raw survival estimates of fall migrants and winter residents (i.e., spring migrants) represent too short a time period for integration into the LCM. As a result, rather than directly incorporating empirical estimates of survival into the LCM, a correction factor was calculated based on available Cormack-Jolly-Seber estimates of survival for both life-history strategies, with the following assumptions:

- 1) Estimates of survival from the late summer parr until fish reach LGD accurately reflect survival rates of all fish in the population. That is, without accounting for life history strategy, all fish experience an average survival rate, and the average of survival rates for each life history strategy is equal to the mean pooled survival for the population as late summer parr to smolts at LGD, and more importantly:
- 2) That fish that overwintering in tributary habitat experience the same survival rate as fall migrants during the short period of time between when fall migrants are tagged and winter sampling occurs in the tributaries.

The previous version of the LCM for CC (ISAB 2017) used mean CJS survival estimates (mean of migration years) for late summer parr to smolts at LGD as the stage-specific productivity. Acknowledging the previously identified assumptions, it is reasonable to calculate a correction factor that accounts for different survival rates for headwaters and valley migration life-histories that can be used to adjust the pooled survival rate as:

$$C_{life\ history} = \frac{Mean(S_l)}{Mean(S_p)}$$

Where  $C$  is the correction factor applied to the pooled late summer parr - LGD smolt survival rate,  $S_l$  is the survival rate of each specific life history (i.e. fall migrant or winter resident), and  $S_p$  is the survival of parr tagged as both fall migrants and winter residents (i.e., the entire population), which is directly comparable to the late summer parr tagging effort applied to the entire population).

**Table 1.** Estimates of total parr survival and life-history specific survival for parr exhibiting fall migrant and winter resident life-histories within the UGR and CC. All survival estimates are based on CJS estimates of survival from ODFW PIT-tag surveys and were used to develop survival corrections for parr to LGD survival within the base model.

	pop. Survival ( $S_p$ )	life-history survival $S_L$		Correction factor	
		fall migrant	winter resident	fall migrant	winter resident
UGR	0.19	0.22	0.14	1.20	0.77
CC	0.17	0.18	0.15	1.10	0.92

## HATCHERY SMOLT SURVIVAL TO LGD

While smolt survival to LGD is modelled from the late summer parr stage for naturally rearing Chinook, hatchery reared smolts experience a shortened survival period from the time of their introduction until they reach LGD. Thus, survival from the time of supplementation to LGD was modelled separately for hatchery and naturally reared smolts. Survival parameters for hatchery reared smolts use annual estimates from Justice (2017) spanning migration years from 2000 – 2012, the median of which were used as inputs within to the LCM. Within the model, hatchery smolt survival affects abundance of smolts introduced to the system by the models modified supplementation scheme. It is worth noting that after LGD hatchery and naturally reared smolts experience the same SAR to LGD (discussed below).

## OCEAN SURVIVAL AND MATURATION

The LCM framework uses smolt to adult return rate (SAR) data to parameterize the marine survival and maturation schedules for each population. The SAR is taken from information supplied in the Comparative Survival Study (FPC 2016) documenting survival and age structure of smolts released as part of Catherine Creek supplementation from Lower Granite Dam (LGD) and returning to LGD as adults. Thus, within our modelling framework, ocean life-stages begin and end at LGD.

Smolt-to-adult return rates and age structure were further decomposed into ocean age-specific survival ( $Soa1$ ,  $Soa2$ ,  $Soa3$ ) and maturation probabilities ( $m1$ ,  $m2$ , and  $m3$ ) that could yield an overall SAR and returning adult age structure on par with the observed data. While the observed return-at-age pattern is a function of non-identifiable parameters (i.e.,  $N*Soa1*m1 = Noa1$ ), the approach used here places constraints on survival and maturation probabilities by age that allow parameter estimation (see McHugh et al. 2017 for an example and rationale).

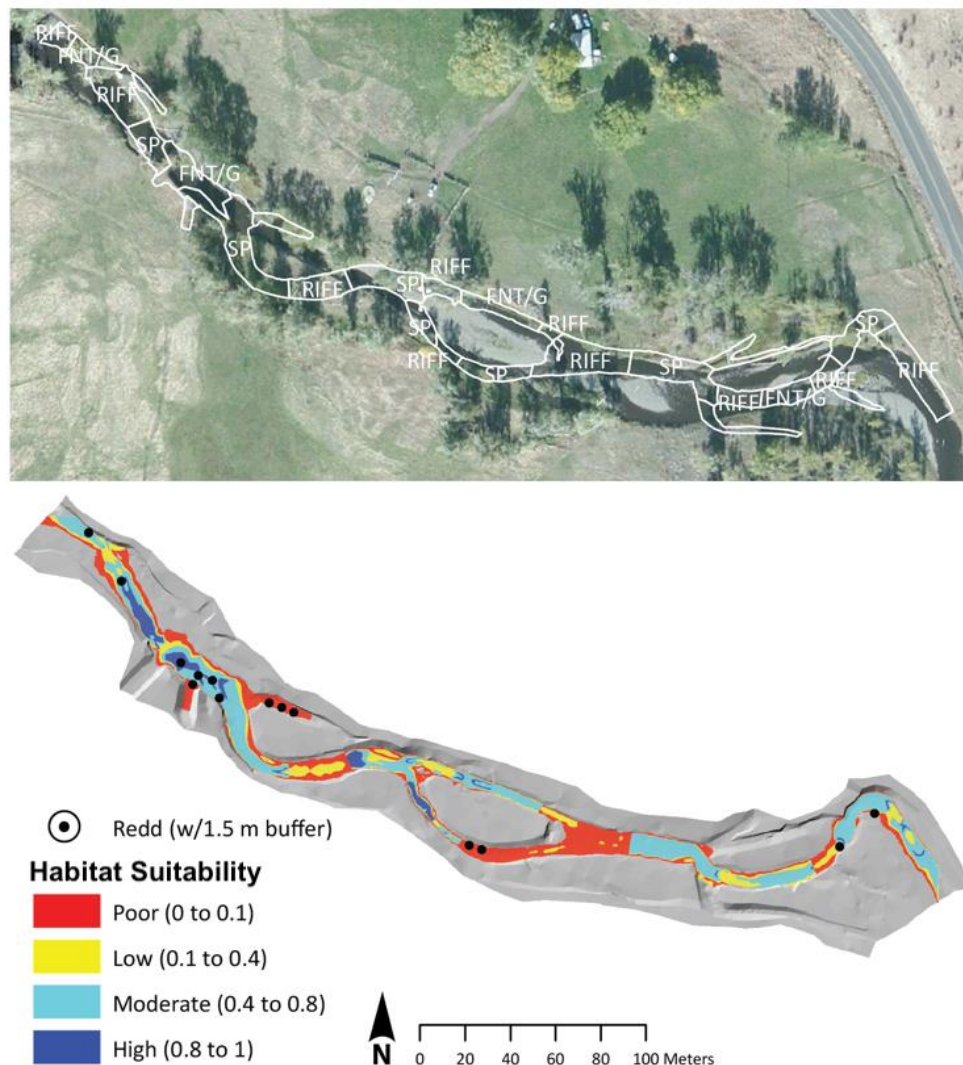
## SPAWNER CAPACITY

Spawner capacity plays a significant role within the current UGR and CC LCM frameworks as it is used to account for pre-spawn survival on the spawning grounds given restoration scenarios that affect temperature (see restoration scenarios below). Additionally, the spawner capacity estimates also capitalize on one of the strengths of the current LCM frameworks, in that they are based on a habitat suitability index (HSI) model that provide a spatially explicit depiction of spawning habitat quality within modelled reaches.



## Spawner Habitat Suitability Index Model

(Catherine Creek, CBW05583-147626, Visit 4286, 2016)



**Figure 2.** Example of spawner habitat suitability index (HSI) model output for a CHaMP visit on Catherine Creek. The HSI model was ultimately used to produce network estimates of spawner capacity used within the LCM framework based off of 186 CHaMP visits occurring between 2011 – 2015.

The HSI model predictions utilized habitat information collected by CRITFC using the CHaMP habitat protocol throughout the UGR and CC historic Chinook spawning network. The HSI model uses depth and velocity results from Delft3D hydraulic model runs, as well as geo-referenced field observations of substrate size (i.e., gravel, cobble, etc.) to compute a spawning HSI score for every 10-cm raster cell within each surveyed reach. HSI scores are then translated into a reach-scale estimate of available spawning habitat, weighted by habitat suitability (i.e. weighted usable area, WUA). The total spawning capacity for a reach is then estimated by dividing the WUA by the average territory size for a spawning Chinook (estimated here as 3.2m<sup>2</sup> for CC and the UGR).

Habitat Suitability Index estimates of reach-scale spawner capacity (estimated as redds) were available for a total of 186 CHaMP survey visits (CC = 81, UGR = 105) that occurred between 2011 and 2015. To

obtain an estimate of total spawner capacity for each population domain, reach scale estimates were first divided by survey reach length as a measure of redd density per linear stream distance. Total redd capacity was then estimated by linear extrapolation of mean redd density within strata that roughly correspond to position in the stream network and channel size. Finally, capacity estimates for each strata were summed as an estimate of base model spawner capacity for each population.

## SPAWNER FECUNDITY

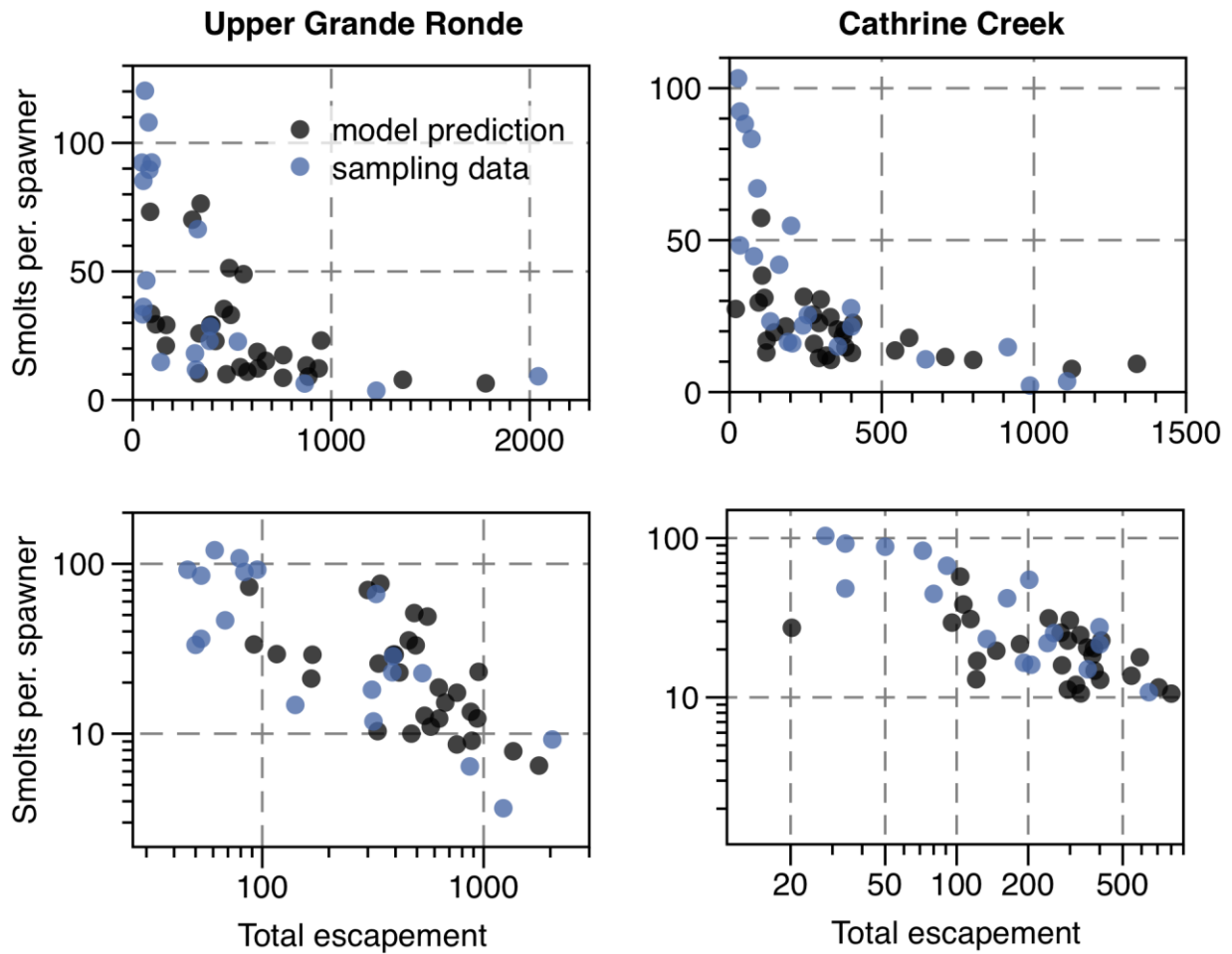
As the current modelling framework tracks the ocean age-specific survival and maturation schedules, efforts were made to model the reproductive contribution of spawning Chinook based on age at return. Age specific fecundity values currently used in the LCM are those reported by Kareiva (2000, Table 2). It is worth noting, that taking an average of the age-specific fecundity values weighted by age composition (see SAR section) closely matched the annual mean of fecundity reported for Catherine Creek by ODFW (i.e., 3871). Also relevant to the modelling framework is that sex ratio of returning spawner are modelled here as being equivalent.

**Table 2.** Female fecundity by ocean age, and ocean age of returning spawners.

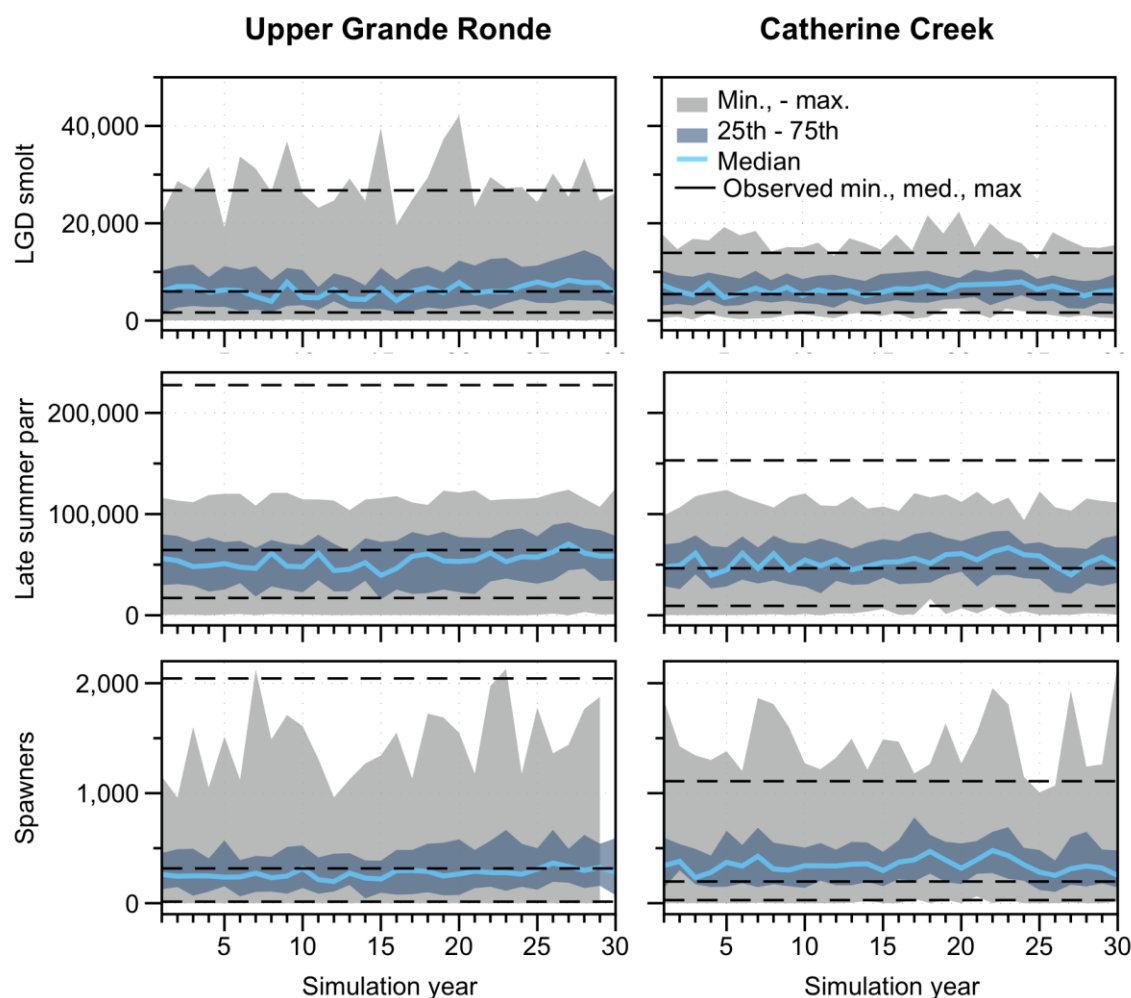
	Spawner Ocean Age		
	Age 1	Age 2	Age 3
Eggs per female	3257	4095	5149
Age composition	23%	75%	2%

## BASE MODEL VALIDATION

Several measures were taken to describe the accuracy of our LCM framework in reproducing UGR and CC population dynamics. This effort relied on comparing LCM predictions of freshwater productivity and abundance to available data. To describe productivity, model predictions of natural smolts at LGD per spawner derived by a single 30-year base model run were compared to available smolt per spawner data for the years 1993 to 2014. To further validate model dynamics, the range and median of late summer parr, smolts at LGD, and adult spawners from 50 base model iterations was compared to corresponding abundance data for each population.



**Figure 3.** Base model validation of life-cycle productivity showing relationship between smolts (at LGD) per spawner and total spawners (hatchery + natural) for the Upper Grande Ronde and Catherine Creek Chinook populations. Sampling data includes years 1992 – 2014, and model predictions depict a single 30 – year model iteration.



**Figure 4.** Base model validation of stage specific abundance showing total range, 25<sup>th</sup> and 75<sup>th</sup> percentile, and median abundance as predicted by the LCM for smolts (at LGD), parr, and spawners. Horizontal lines show the observed range and median values for observed data from 1992 – 2014.

## RESTORATION AND CLIMATE SCENARIOS

Following validation of the base ('Curr') LCM, the modelling framework was used to investigate Chinook population performance in the UGR and CC according to an array of restoration and climatic scenarios (Table 3). First, the population was modelled under 'current conditions using the base model. This is intended to provide insight on population performance under a 'do nothing' case, and provides a standard for comparison to other restoration and climate scenarios. In addition to the base case ('Curr'), six different scenarios were modelled that were based on the recent temperature modeling work of Justice et al. (2017). Specifically, the population-level effects of alternative temperature/habitat futures ranging from a scenario that considered the effects of global warming, manifested as increased stream flow and hydrological change, in the absence and presence of habitat restoration (i.e., scenarios 'Clim', 'ClimVeg',

‘Clim...’, etc.) to scenarios that considered the effects of riparian restoration under current thermal conditions (e.g., scenario ‘PNV’).

**Table 3.** Scenarios modeled using the CC LCM, after Justice et al. (2017). For LCM inputs, each scenario is represented as a proportion increase or decrease in summer parr rearing and spawner capacity.

Scenario Abbreviation	Description
Curr	Baseline model calibrated using 2010 temperature, climate, vegetation, and hydrologic conditions
PNV	Vegetation across the entire model extent set to potential natural vegetation (PNV) cover and height.
HiPr	Vegetation in high priority areas set to PNV and other areas set to current conditions.
WidPNV	Channel width set to historic conditions and vegetation set to PNV.
Clim	Air temperature and streamflow set to 2080s climate projections.
ClimVeg	2080s climate projections and vegetation set to potential cover and height at 75 years.
ClimVegWid	2080s climate projections, vegetation set to potential cover and height at 75 years, and channel width set to historic conditions.

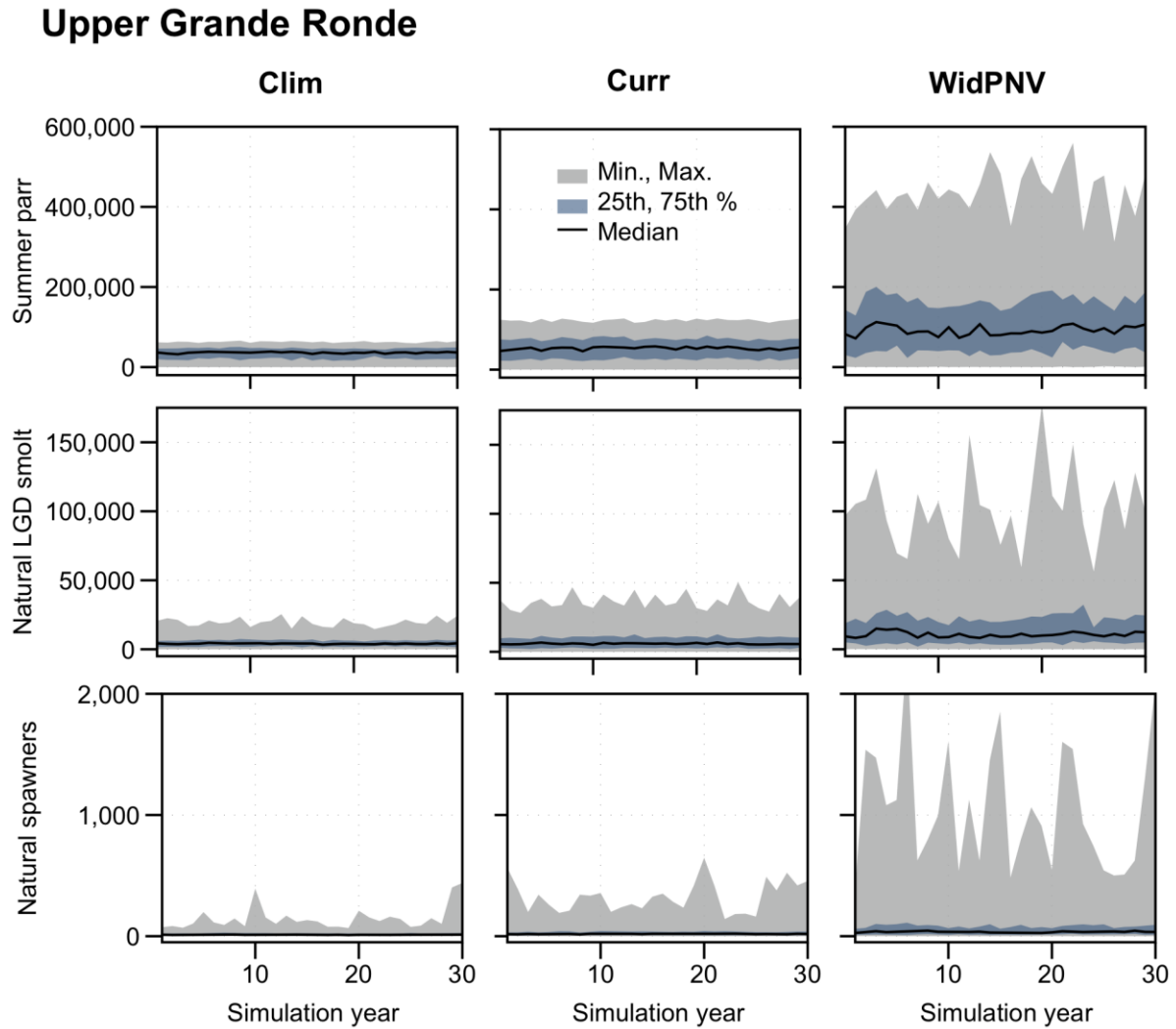
Benefits/changes were expressed within the modelling framework at two stage transitions: 1) via a simple proportional change in summer parr rearing capacity based on the abundance changes that Justice et al. (2017) predicted under each scenario, and 2) by proportionally adjusting spawner capacity (as egg equivalents) based on temperatures that exceed a threshold for pre-spawn survival under each temperature and restoration scenario.

**Table 4.** Proportional increase/decrease and absolute capacity estimate used under each restoration and climate scenario for the Upper Grande Ronde (UGR) and Catherine Creek (CC) Chinook populations.

Population	Scenario Abbreviation	Parr capacity		Spawner capacity (redds)	
		% change	input	% change	input
UGR	Curr	0%	149543	0%	14198
	PNV	377%	713318	83%	25930
	HiPr	100%	299085	29%	18379
	WidPNV	590%	1031844	83%	25930
	Clim	-53%	70285	-60%	5745
	ClimVeg	63%	243754	0%	14242
	ClimVegWid	114%	320021	10%	15632
CC	Curr	0%	165687	0%	12827
	PNV	61%	266757	4%	13362
	HiPr	30%	215394	0%	12848
	WidPNV	67%	276698	4%	13362
	Clim	-36%	106040	-36%	8200
	ClimVeg	20%	198825	-1%	12762
	ClimVegWid	37%	226992	1%	12977

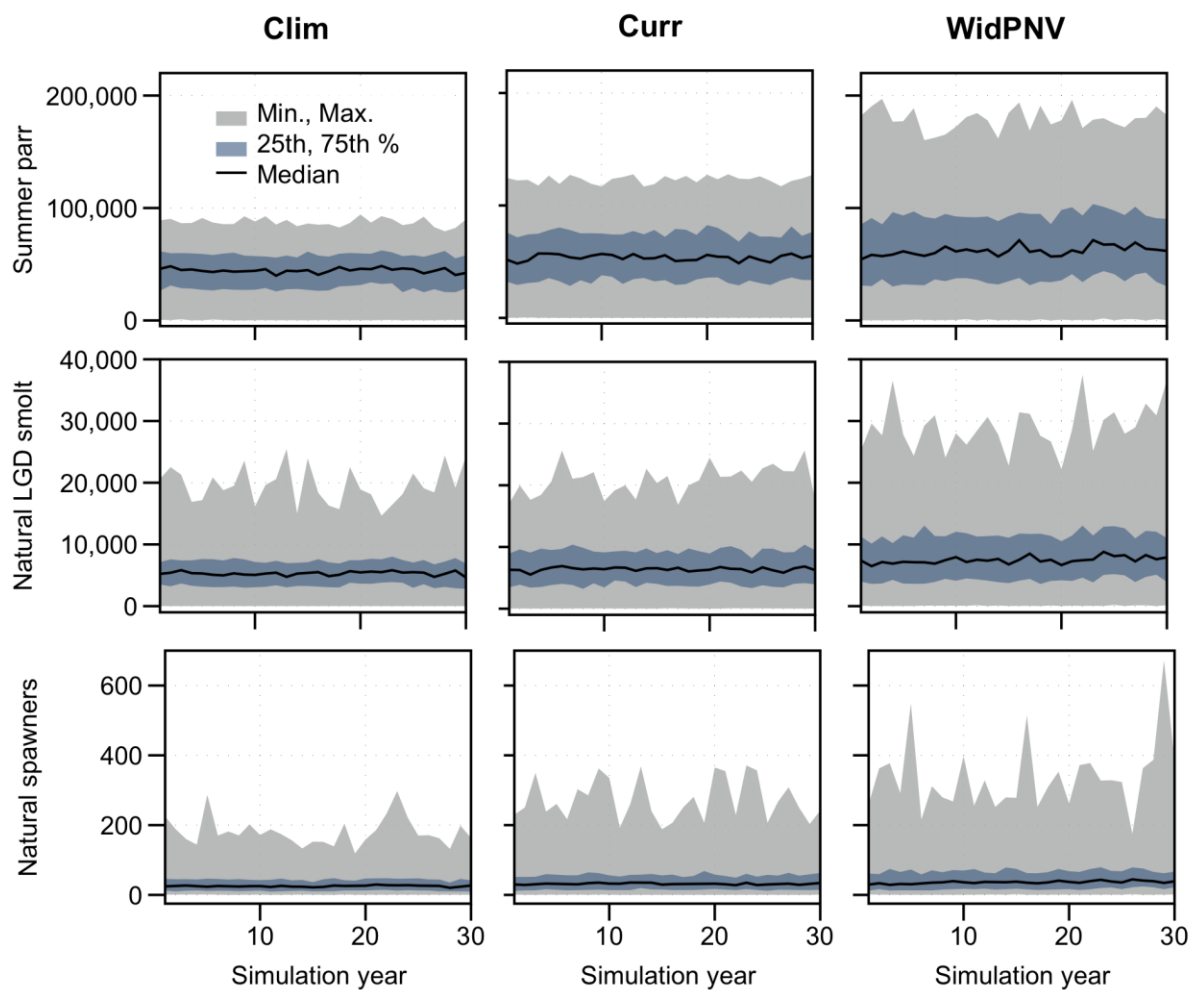
For each scenario (Table 3) the model was used to simulate population dynamics for a 30 - year period (i.e., 50 years, less a 20 year burn in period) for  $N = 200$  separate Monte Carlo iterations. All models were initialized with 1,000,000 natural-origin fry and 150,000 hatchery-origin smolts. Supplementation releases were tracked as hatchery (H1) fish until their death; all progeny spawned in the wild were tracked and treated as being ‘natural’, regardless of parentage (e.g.,  $H1 \times H1 = \text{natural}$ ,  $\text{natural} \times H1 = \text{natural}$ ).

Results of the LCM simulation are presented in several ways to foster a greater understanding of the system dynamics under each scenario. Of primary focus to the viability of the population is the number of spawners returning to the spawning grounds, which is presented for naturally reared and total spawners as the geometric mean of each 30-year model simulation (Figure 7, Table 5). Figures comparing the range of model abundance estimates across each of the 200 MCMC simulations for the 30 - year time series for parr, smolt, and total spawners (i.e. hatchery + wild) are also presented for the base model and the two future management scenarios that feature the largest increase/decrease in parr and spawner capacity (i.e., Clim and WidPNV).



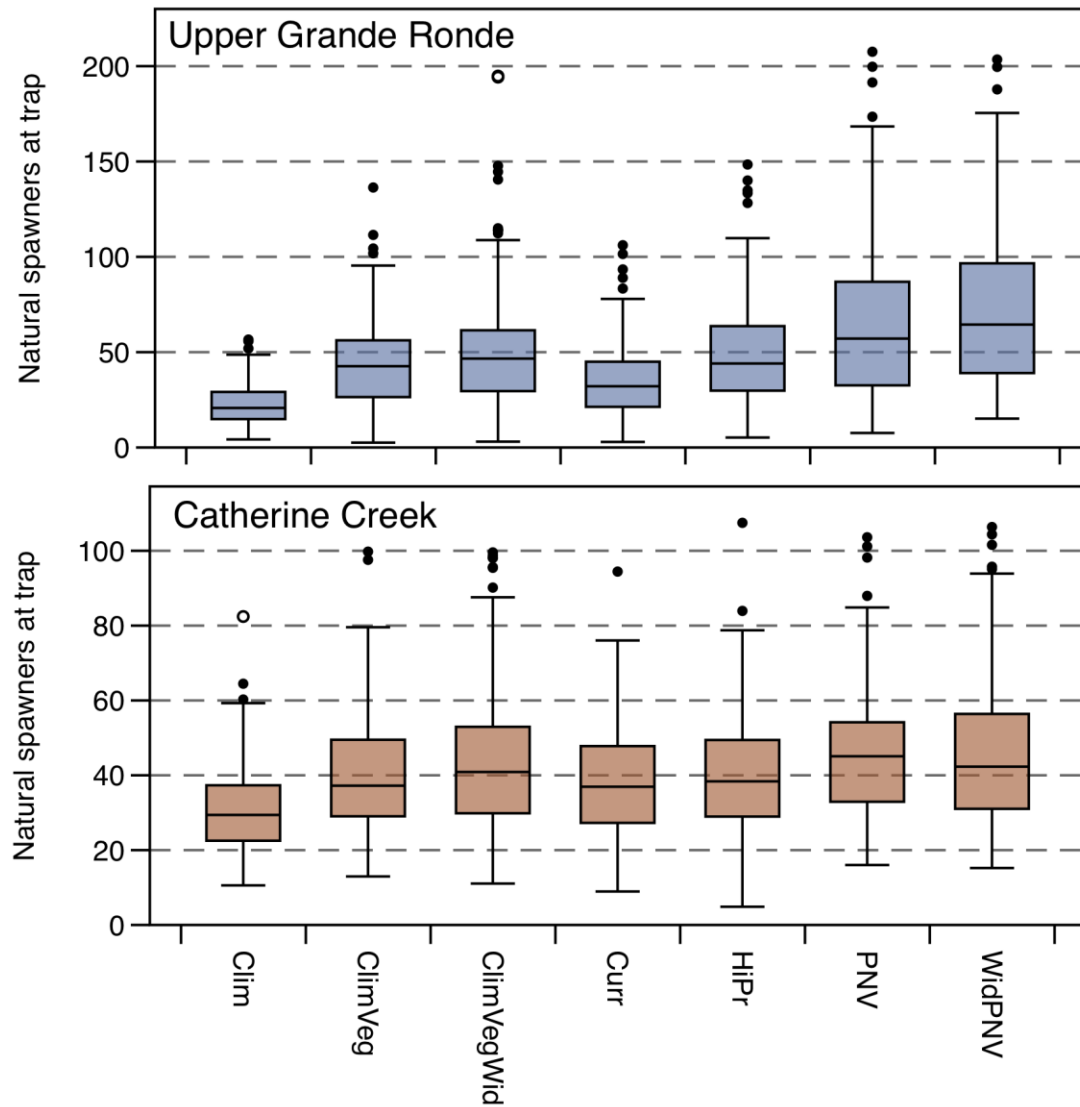
**Figure 5.** Stage specific range, 25<sup>th</sup> and 75<sup>th</sup> percentile, and median of abundance predicted over 200 MCMC model simulations for Upper Grande Ronde chinook. Predictions show the management scenarios for the base model (Curr), and largest decrease (Clim) and increase (WidPNV) in parr and spawner capacity.

## Catherine Creek



**Figure 6.** Stage specific range, 25<sup>th</sup> and 75<sup>th</sup> percentile, and median of abundance predicted over 200 MCMC model simulations for Catherine Creek chinook. Predictions show the management scenarios for the base model (Curr), and largest decrease (Clim) and increase (WidPNV) for parr and spawner capacity.





**Figure 7.** Distribution of the geometric mean of naturally reared spawners returning to the adult trap/weir for the UGR and CC under each restoration scenario.

**Table 5.** Increase/decrease in natural spawners returning to the weir/trap on UGR and CC under each management scenario based on 200 MCMC simulations. Increase/decrease presented relative to the change in the geometric mean for each 30-year simulation.

<b>Population</b>	<b>Scenario</b>	<b>Mean</b>	<b>SD</b>	<b>Range</b>	<b>Relative to Curr (%)</b>
UGR	Curr	35	19	3 - 106	-
	Clim	23	10	4 - 57	-36%
	ClimVeg	44	23	3 - 136	26%
	ClimVegWid	50	30	3 - 195	44%
	HiPr	49	26	5 - 148	41%
	PNV	66	40	8 - 208	88%
	WidPNV	73	43	15 - 203	108%
CC	Curr	39	16	9 - 94	-
	Clim	31	11	11 - 82	-23%
	ClimVeg	40	15	13 - 100	4%
	ClimVegWid	43	18	11 - 100	12%
	HiPr	40	16	5 - 107	3%
	PNV	46	19	16 - 160	22%
	WidPNV	45	19	15 - 106	19%

## RESULTS AND DISCUSSION

Several significant changes to the LCM framework introduced here increase the realism and utility of the model to assess recovery potential for Upper Grande Ronde and Catherine Creek Chinook within a LCM context. Specifically, accounting for capacity differences for spawners addition to parr improves the model's ability represent the impact of each restoration and climatic scenario on life-cycle productivity. In addition, allowing the model to account for life-history dependent smolt survival probabilities, and inclusion of mortality for hatchery smolts prior to passing LGD also more accurately reflects the population dynamics of these systems, and substantially improves the agreement between model predictions and observed population dynamics.

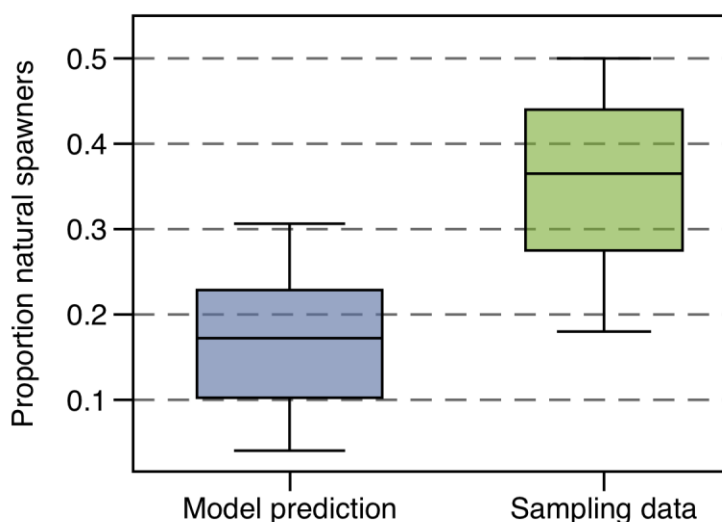
Overall, our base parameterization (i.e., 'Curr') appeared to accurately capture both the stage specific (i.e., freshwater vs. marine, Figure 4) and total life cycle productivity (Figure 3) of spring Chinook salmon in Catherine Creek and the Upper Grande Ronde, despite being developed from disparate and somewhat independent datasets in a piecewise fashion (e.g., freshwater survivals from long-term monitoring datasets vs. a habitat-based parr capacity model that predicts how many fish a basin can produce). For example, estimates of LGD smolts per spawner in our modeled dataset closely mirror that depicted in ODFW's 20-year sampling dataset (Figure 4). Similarly, model estimates of late summer parr, total smolts, and total spawners returning to CC and the UGR featured high degree of correspondence with the minimum, median, and maximum observed for these populations (i.e., in Carmichael et al. 2011). Given these patterns, the current LCM parameterization ('Curr') appears to reasonably capture the demographics of the intended natural populations. Accordingly, the model, with the capacity modifications described in Table 3 was used to assess the effects of the Justice et al. (2017) scenarios on total spawner abundance. While no attempt is made to infer meaning here, LCM simulation results (summarized in Table 5 and Figures 5-7) illustrate the following:

- The proportional changes in parr and spawner carrying capacity reflecting the Justice et al. (2017) habitat scenarios translated into an increase in the abundance of natural origin adult spawners that was similar on a rank-order basis. However, the magnitude of the population response did not increase/decrease on a one-to-one basis to the capacity changes under each scenario.
- The absolute response increasing capacity under most management scenarios was marginal (i.e., increase of only 10 to 50 natural spawners). However, relative increases/decreases in spawners were more pronounced, and represented a potential doubling of returning natural spawners under the most aggressive restoration scenarios (i.e. PNV and WidPNV).
- Marginal response to restoration scenarios may be explained by two factor; reliance of these populations on a heavy supplementation scheme, and an SAR that is independent of spawning and rearing productivity.
- Productivity responses (i.e. increase/decrease) were more pronounced within the UGR where current temperature regimes in much of the rearing network reside close to upper temperature threshold for this species.
- Overall, the 'PNV' and 'WidPNV' scenarios, which assume maximum cooling of stream temperatures in the absence of climate change effects, showed the greatest response, with an approximate 80-100% increase in spawner abundance within the UGR. On the other hand, 'doing nothing' vegetation/restoration wise in the face of the anticipated effects of climate change on conditions in the UGR (scenario 'Clim') translated into a 36% reduction in spawner abundance.

## FUTURE WORK AND LCM FRAMEWORK IMPROVEMENT

Perhaps the most glaring result of the current LCM evaluation is that increases/decreases in parr and spawner capacity did not result in life-cycle productivity responses on a similar magnitude. For example, the most optimistic restoration scenario (WidPNV) predicted a 590% increase in parr and 83% increase in spawner capacity within the UGR (Table 5). While these changes did result ca. 100% increase in the abundance of naturally reared adult spawning chinook, this increase is still likely below that which would be needed to establish a viable population independent from hatchery supplementation despite a predicted 5 fold increase in early life-stage capacity. This verity makes apparent several considerations that are relevant to restoration and recovery planning for the UGR and CC chinook populations. These results may suggest that increasing habitat quantity alone may not be sufficient increase the recovery potential of salmonid populations that feature low early life-stage productivity. In addition, predicting increases/decreases to habitat availability and corresponding capacity as a function of habitat restoration or climate change may be more attainable than survival. Incorporating the potential influence that restoration and climate change may have on survival and productivity into the LCM framework could provide greater insight about the recovery trajectory of the UGR and CC populations as a function of future management scenarios.

In addition, the changes made to the LCM framework served to increase the correspondence between base model predictions of stage-specific abundance and those observed in available population monitoring data. However, one area where model predictions continue to deviate from available data is in the ratio of returning natural to hatchery reared spawners (Figure 8).



**Figure 8.** Distribution of the ratio of between naturally and hatchery reared spawning Chinook returning to the adult trap/weir on Catherine Creek as predicted by the base LCM and observed data from ODFW

This discrepancy may be due to several characteristics of the modelling framework that could be addressed in future LCM efforts or as new data become available. Within the model, the SAR from LGD to LGD is based on the CSS information for hatchery reared smolts. This SAR may feature mortality parameters that are too high for naturally reared smolts and adult that often exhibit greater performance

than hatchery fish. Additionally, little data exists to compare stray rates for hatchery and naturally reared chinook between LGD and the adult trap/weir at the spawning grounds on the UGR and CC. However, an increased rate of straying for hatchery reared adults (as has often been shown) might also allow the hatchery:natural spawner ratio within the LCM to more accurately capture system dynamics.

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## APPENDIX B-i. BASE PARAMETERIZATION AND REFERENCES

Input category	Life stage(s)	Parameter	Value(s)	Stochasticity	Source(s)	Comments
Survival/ productivity	Egg-to-parr	Egg-to-summer parr productivity	UGR = 0.20 CC = 0.14	Target CV: 40%	B. Jonasson, ODFW, monitoring spreadsheet	Computed from ODFW as the mean of late summer parr, relative to ODFW's redd counts (above trap), assuming 1 female/redd and age specific fecundity. Includes only years > 2001 to include natural spawning of hatchery reared fish passed above traps.
	Winter parr-to- smolt	Winter parr-to-smolt survival	UGR Trib. = 0.18 UGR Val. = 0.12 CC Trib. = 0.10 CC Val. = 0.13	Target CV: 70%	B. Jonasson, ODFW, monitoring spreadsheet	Survival from fall to LGD as a smolt; computed for entire cohort of summer parr for separate Tributary vs. Valley overwintering component
	Ocean rearing stages [by ocean age (oa)]	Survival by ocean age, Soa1, Soa2, Soa3	Soa1 = 0.02, Soa2 = 0.32, Soa3= 0.56	Target CV: 50, 30, 30%, respectively	Based on CSS age-specific smolt-to-adult return rates (McCann et al. 2016)	Survival inputs & maturation probabilities (below) yield adult return age composition of Catherine Creek Acclimation Pond releases oa1 = 23%, oa2 = 75%, oa3 = 3%. Values also used for UGR population.
	Spawners	Prespawn mortality (on spawning grounds)	UGR = 0.80 CC = 0.87	Target CV < 10%	J. Feldhaus, ODFW, and T. Bowerman prespawn mort summaries.	Median value from available data, also incorporates additional 10% stray/mortality rate from LGD to spawning grounds.
Capacity	Summer parr	Total capacity	UGR = 149,543 CC = 165,687	n/a (static)	B. Jonasson, ODFW, monitoring spreadsheet	Beaverton-holt fit of total spawner estimate to late-summer parr estimate.
	Adult (spawners)	Total capacity	UGR = 14198 CC = 12,827	n/a (static)	HSI-based female spawner (redd) capacity, CHaMP	Modeled in units of eggs assuming average spawner age structure and age-specific fecundity.
Other life history or mgmt parameters	Fall/early winter parr	Probability of moving downstream to overwinter	UGR = 0.27 CC = 0.76	n/a (static)	B. Jonasson, ODFW, monitoring spreadsheet	Allows for spatial redistribution of fish in fall, in manner reflective of obs'd fall vs. spring migrant abundance
	Ocean rearing stages [by ocean age (oa)]	Maturation probability, Moa1, Moa2, Moa3, ...	Moa1 = 0.11, Moa2 = 0.95, Moa3= 1.00	Target CV: 80, 15, 0, respectively	Estimated from SARs data (see note for marine survival above)	see McHugh et al. (2017) appendix for detailed estimation approach.
	Fecundity, by total age	f3, f4, f5	f3 = 3257, f4 = 4,095, f5= 5,149	n/a (static)	Kareiva et al. (2000)	Used in CC and UGR populations
	Smolts	Supplementation releases	UGR = 250,000 CC = 150,000	n/a (static)	Carmichael (2011)	Current management target; varies in LCM as f(run size) if hatchery broodstock req's aren't met.
	Hatchery smolt	Survival - release to LGD	UGR = 0.45 CC = 0.68	n/a (static)	Justice (2013) Migrant survival analysis	Median value survival estimate from release to LGD

## APPENDIX B-ii. MODELLING HATCHERY SUPPLEMENTATION

Supplementation efforts strongly influence abundance dynamics in both CC and UGR populations, and the following measures were taken to accurately represent this management strategy within the LCM framework. In order to implement the supplementation scheme, several simplifying assumptions were made so that a mixed hatchery-natural broodstock population could be modelled for each the CC and UGR case:

1. All hatchery-reared fish are produced from supplementation fish captured in the wild and reared according to conventional hatchery practices; thus, a captive broodstock component is not included, due to its discontinued status and previously documented life history differences.
2. To model a feedback between total (HOR+NOR) adult returns ( $N_{AD-tot} = N_{AD-h} + N_{AD-N}$ , adults eligible for retention as broodstock ( $N_{BS}$ ), and  $N_{sm-sup}$  in year  $y+2$ , we assumed an egg take requirement of 85 females to produce 250K smolts (M. McLean, CTUIR, pers. comm.); or an equivalency of 1,470 smolts per adult, assuming a 50:50 sex ratio. Thus,  $N_{sm-sup}$  in year  $y+2$  is determined as, where  $N_{BS}$  is determined according to stream-specific rules below. Hatchery broodstock goals for UGR and CC are 170 and 103 adults (fem+male), respectively.

If  $N_{BS}$  in year  $y \geq$  hatch. Goal {  $N_{sm-sup}$  in  $y+2$  = smolt goal (i.e., 150K CC, 250K UGR) }  
Else {  $N_{sm-sup} = N_{BS}$  in year  $y+2 = N_{BS} * 1,470$  }

3. All supplementation fish, like their NOR counterparts, have intact adipose fins and are thus minimally affected by mark-selective fisheries targeting Chinook in the mainstem Columbia and Snake rivers; thus, modeled LGD-to-LGD SARs already accounts for harvest. Harvest above LGD is not presently accounted for in the LCM, however it can be incorporated into revised model runs as needed and tailored to affect hatchery fish differently than natural fish as data/information prescribe.

**Catherine Creek.** In CC (rule type = 4 in input files), where adult trapping operations span the entire run, the management of hatchery and natural fish on the spawning grounds and in hatchery broodstock follows a ‘sliding scale’ framework under which target levels for  $P_{NOS-R}$ ,  $P_{HOS}$ , and  $P_{NOB}$  vary across three levels of run size ( $N_{AD-tot}$ ), <250, 250-500, and >500 (Carmichael et al. 2011). At low abundance, is  $P_{NOS-R}$  is capped at 40% there are no constraints on  $P_{HOS}$ , and  $P_{NOB}$ , and hatchery fish are otherwise retained to meet egg-take requirements. At high abundance,  $P_{HOS}$  is minimized (max allowed = 50%),  $P_{NOB}$  is maximized (target  $\geq$  30%), and is  $P_{NOS-R}$  more tightly constrained (max allowed = 20%). At moderate abundance,  $P_{NOS-R}$  is similarly constrained (20% cap) and there is tolerance for higher  $P_{HOS}$  (max allowed = 70%) and lower  $P_{NOB}$  (target  $\geq$  20%). While this rule set is straightforward, there are cases for which it is impossible to meet all constraints simultaneously, which made implementing it in code somewhat complicated. Consider for example a case in which returns fall between 250 and 500, but are almost exclusively (say 80+%) of hatchery origin. In this case, it’s quite likely that egg take needs will not be met if weir management strictly follows the sliding scale’s  $P_{NOB}$  and  $P_{NOS-R}$  constraints; nor is it clear what passage goals (to spawning grounds) should be in cases for which returns exceed the scale’s upper abundance threshold but based predominantly on hatchery origin returns. Thus, to operationalize the CC sliding scale, several additional ‘rules’ were imposed, via the following pseudo-code:



Case 1 (in the low abundance tier)

```
If  $N_{AD-tot} < \text{lower threshold (i.e., } <250 \text{ adults, } H+N \text{)}$  then ... {  
  If  $N_{AD-N} > 0$  ... {  
    Retain up to 40% of the natural run for hatchery broodstock  
    Make up the balance of broodstock needs with hatchery fish  
  } Else (i.e., case for which  $N_{AD-N}$  is 0, but  $N_{AD-h}$  is  $> 0$ ) ... {  
    Collect hatchery fish up to the broodstock goal  
  }  
  Rescale H and N run sizes to account for fish taken to hatchery  
  Pass everything that wasn't collected to the spawning grounds  
  If adult collections  $<$  broodstock goals {  
    Recompute smolt releases for year  $y+2$  as function of fish retained (i.e., per #3 above)  
  }  
}
```

Case 2 (in the mid abundance tier)

```
If  $\text{lower threshold} < N_{AD-tot} < \text{upper threshold (i.e., } 250\text{-}500 \text{ adults)}$  then ... {  
  If  $N_{AD-N} > 0$  ... {  
    Retain up to 20% of the natural run for hatchery broodstock  
    Make up the balance of broodstock needs with hatchery fish  
  } Else (i.e., case for which  $N_{AD-N}$  is 0, but  $N_{AD-h}$  is  $> 0$ ) ... {  
    Collect hatchery fish up to the broodstock goal  
  }  
  Rescale H and N run sizes to account for fish taken to hatchery  
  Determine  $P_{HOS}$  would be if remaining fish were allowed to spawn  
  If  $P_{HOS} > 70\%$  {  
    If you can meet  $P_{HOS}$  goal without dropping into lower abundance tier {  
      Remove hatchery fish until you meet  $P_{HOS}$  goal  
    } Else { Don't worry about it }  
  }  
  Rescale H run size to account for removals  
  Pass remaining fish to spawning grounds  
  If adult collections  $<$  broodstock goals {  
    Recompute smolt releases for year  $y+2$  as function of fish retained (i.e., per #3 above)  
  }  
}
```

Case 3 (in the high abundance tier)

```
If  $N_{AD-tot} > \text{upper threshold (i.e., } 500 \text{ adults)}$  then ... {  
  If  $N_{AD-N} > 0$  ... {  
    Retain up to 20% of the natural run for hatchery broodstock  
    Make up the balance of broodstock needs with hatchery fish  
  } Else (i.e., case for which  $N_{AD-N}$  is 0, but  $N_{AD-h}$  is  $> 0$ ) ... {  
    Collect hatchery fish up to the broodstock goal  
  }  
  Rescale  $N_{AD-h}$  and  $N_{AD-N}$  run sizes to account for fish taken to hatchery  
  Determine  $P_{HOS}$  would be if remaining fish were allowed to spawn  
  If  $P_{HOS} > 50\%$  {  
    If you can meet the system total escapement goal with natural fish alone {
```

```

    Pass only natural fish to the spawning grounds ( $P_{HOS} = 0$ )
  } Else If  $N_{AD-N}$  / upper threshold  $\geq 50\%$  {
    Remove hatchery fish until you're at or under  $P_{HOS} = 50\%$ 
  }
}
Rescale H run size to account for removals
Pass remaining fish to spawning grounds
If adult collections < broodstock goals {
  Recompute smolt releases for year  $y+2$  as function of fish retained (i.e., per #3 above)
}
}

```

As the above pseudo-code illustrates, no attempt was made to constrain the modeled supplementation program based on the stated hatchery  $P_{NOB}$  goals, because (a) as doing so introduced additional coding complications (i.e., due to exceptions and circular dependencies), and (b) was virtually impossible to meet under the natural population's current total life cycle productivity/capacity assumptions.

**Upper Grande Ronde River.** In the Upper Grande Ronde (rule type = 5 in input files), the management of HOS/NOS on the spawning grounds and HOB/NOB at the hatchery is less formalized than in Catherine Creek, due to the fact that the weir is typically pulled before the majority of the run makes its way through to the spawning grounds and because the its supplementation program is generally less restrictive regarding target  $P_{NOS-R}$ ,  $P_{HOS}$ , and  $P_{NOB}$  values. In brief, the main constraint imposed on weir/program management is that no more than 50% of the natural run can be retained for hatchery broodstock. This is executed in code according to the following pseudo-code:

```

If  $N_{AD-N} > 0$  ... {
  Retain up to 50% of the natural run for hatchery broodstock
  Make up the balance of broodstock needs with hatchery fish
} Else (i.e., case for which  $N_{AD-N}$  is 0, but  $N_{AD-h}$  is  $> 0$ ) ... {
  Collect hatchery fish up to the broodstock goal
}
Rescale H and N run sizes to account for fish taken to hatchery
Pass everything that wasn't collected to the spawning grounds
If adult collections < broodstock goals {
  Recompute smolt releases for year  $y+2$  as function of fish retained (i.e., per #3 above)
}

```