Characterizing Watershed-Scale Effects of Habitat Restoration Actions to Inform Life Cycle Models: Case Studies Using Data-Rich vs. Data-Poor Approaches

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Characterizing Watershed-Scale Effects of Habitat Restoration Actions to Inform Life Cycle Models: Case Studies Using Data-Rich vs. Data-Poor Approaches

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U.S. DEPARTMENT OF COMMERCE

National Oceanic and Atmospheric Administration
National Marine Fisheries Service
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Abbreviations

AICc	Akaike Information Criterion, corrected
AP	abundance and productivity
AQI	Aquatic Inventories Project (ODFW)
BDA	Beaver Dam Analog
BH	Beverton–Holt
BiOp	biological opinion
BLM	U.S. Bureau of Land Management (DOI)
BOR	U.S. Bureau of Reclamation
BPA	Bonneville Power Administration
BSR	biologically significant reach
cfs	cubic feet per second
CHaMP	Columbia Habitat Monitoring Program
CRB	Columbia River Basin
CRBFH	Columbia River Basin Floodplain Habitat
CRITFC	Columbia River Inter-Tribal Fish Commission
DOC	U.S. Department of Commerce
DOI	U.S. Department of the Interior
EP	expert panel
ESA	Endangered Species Act
ESU	evolutionarily significant unit
FCRPS	Federal Columbia River Power System
FLIR	forward-looking infrared
GAA	globally available attribute
GC	geomorphic condition
GDU	growing degree unit
GEI	gross energy intake
GLO	U.S. General Land Office (now BLM)
GR	Grande Ronde
GRTS	generalized random tessellation stratified
HEM	habitat expansion model
HQQ	habitat quality and quantity

HSI	habitat suitability index
ICRT	Interior Columbia Recovery Team
ICTRT	Interior Columbia Technical Recovery Team
IMW	intensively monitored watershed
ISAB	Independent Science Advisory Board
ISEMP	Integrated Status and Effectiveness Monitoring Program
LCM	life cycle model
LiDAR	light detection and ranging
LS	large streams
LWD	large wood debris
MAR	multivariate autoregressive
MCMC	Markov chain Monte Carlo
MIC	modified information criterion
MPG	major population group
MWMT	maximum weekly maximum temperature
NEI	net energy intake
NMFS	National Marine Fisheries Service (NOAA)
NOAA	National Oceanic and Atmospheric Administration (DOC)
NorWeST	northwest stream temperature
NREI	net rate of energy intake
NWFSC	Northwest Fisheries Science Center (NMFS)
ODFW	Oregon Department of Fish and Wildlife
PAG	plant association group
PCA	principal component Analysis
PDO	Pacific Decadal Oscillation
PHABSIM	physical habitat simulation
PIT	passive integrated transponder
pNI	proportionate natural influence
PNSHP	Pacific Northwest Salmon Habitat Project
PNV	potential natural vegetation
PNW	Pacific Northwest

Abbreviations (continued)

PUD	public utlity district
QET	quasi-extinction threshold
QRF	quantile regression forest
RFM	random forest model
RT	reach type
RVD	riparian vegetation departure
SAR	smolt-to-adult return
SC	small confined streams
SC	swimming cost
SEM	structural equation model
SPCH	spring Chinook
SPS	salmonid population summary
SR	stock–recruit
SREFS	East Fork Salmon River
SRLEM	Lemhi River, Salmon River
SRLMA	Lower Mainstem Salmon River
SRNFS	North Fork Salmon River
SRPAH	Pahsimeroi River, Salmon River
SRPAN	Panther Creek, Salmon River
SRUMA	Snake River Upper Mainstem
SRVAL	Valley Creek, Salmon River
SRYFS	Yankee Fork, Salmon River
SSD	spatial structure and diversity
SSN	spatial stream network
SU	small unconfined streams
UCSRB	Upper Columbia Salmon Recovery Board
UGR	Upper Grande Ronde
USACE	U.S. Army Corps of Engineers
USAL	Upper Salmon River
USFWS	U.S. Fish and Wildlife Service
WDFW	Washington Department of Fish and Wildlife

1. Executive Summary

1.1. Life cycle models (LCMs)—a solid ecological foundation

Life cycle models are an essential tool for developing effective management strategies to conserve and recover salmonid populations. LCMs link the salmonid life cycle to physical and biological processes through a series of population biology parameters. Since salmon encompass large geographic ranges, as well as multiple habitat types, understanding the quantitative connection between the life stages and the stream and ocean environment is paramount to understanding what may limit their productivity and abundance at the population scale (McElhany et al. 2000).

LCMs and their use in understanding salmonid population dynamics is not a "new" concept (Larkin and Hourston 1964). LCMs have been in use, coupled with restoration scenario planning and development, for over 25 years (Nickelson et al. 1992). Others have also utilized the LCMs with restoration scenarios in the Pacific Northwest at both the watershed and regional scale (Beechie et al. 1994, Lichatowich et al. 1995). As the extent of use for this particular type of application grew, it became apparent that the necessary fish and associated habitat data for developing a standardized suite of population-scale biological parameters (e.g., stage specific capacity, productivity) to use as model inputs was not available for every population. This limitation led to several approaches to generate such information at the spatial scales needed for the broad application of LCMs (Mobrand et al. 1997). Still other studies attempted to estimate the effects of restoration on biological parameters used in LCMs, in turn characterizing the demographic benefit of restoration to the population (e.g., Bartz et al. 2006). These and other examples suggest that LCMs are a useful platform to help quantify how potential watershed restoration actions affect the protection and restoration of salmonid populations (Beechie et al. 1994, Mobrand et al. 1997, Nickelson and Lawson 1998, Pollock et al. 2004, McHugh et al. 2017).

1.2. How did we use LCMs to evaluate watershed restoration scenarios and salmon population response?

The use of LCMs and watershed restoration scenarios to explore the potential population-scale outcome of reach-scale habitat management actions that we describe in this document is different in several ways from previous LCM applications. Our process includes a combination of literature review, watershed-scale examples that utilize local watershed information, extrapolation methods to expand data gaps, and the use of real restoration scenarios developed by local watershed managers. Lastly, we include population-specific downstream factors (e.g., mainstem dam passage survival, marine survival), that impact Columbia River salmon populations. The combination of these elements makes our effort more comprehensive than previous efforts because we include a wide range of modeling approaches that are based on the scale and intensity of fish and habitat data that are available for each population and restoration scenario.

We chose three example watersheds that have varying levels of local fish and habitat information to gain a better understanding of how the extent and resolution of data affects the ability to see a restoration "signal" relative to the natural variation in fish, habitat, and restoration effectiveness data. The role that LCMs can play in up-scaling restoration actions is important because with many individual restorative actions the population level "signal" is small and their inclusion in LCMs may be limited to small impacts on only one or two specific metrics such as habitat capacity or fish density. At the population level, these relatively common reach level restoration actions may not translate into a detectable increase in outmigrating juveniles or returning adults. However, population signals become more detectable when either the aggregate restoration impact is larger, or the uncertainty in underlying biological mechanism of effect is lower. In this sense, higher quality fine-scale fish and habitat data can allow for evaluation of smaller, less detectable, reach level actions and their benefit to salmon at the population level.

In our process, we assume that the actions we model can achieve a salmon population response. We recognize that there are actions that are beneficial for fish, but for which we do not currently have enough information to incorporate into LCMs. We also recognize that we did not model all necessary actions that could lead to a positive salmon population response. For example, none of the case studies examined increased summer flows, which would likely have a positive effect on several parameters in our models. Not having these quantitative fish–habitat relationships limits our ability to estimate appropriate parameters for each restoration action type; however, the LCM framework is sufficiently general that as the community develops specific relationships between habitat quality and quantity and demographic parameters, they can be integrated into the population assessments.

Third, we focus on evaluating metrics of salmonid population dynamics that are important for management - change in average adult abundance or risk of falling below a given threshold, probability of quasi extinction). We then describe how these approaches can differ and provide examples of this framework to illustrate the potential approaches to this process of linking LCMs to restoration scenarios.

Finally, we focus on freshwater tributary habitat rehabilitation because there is a need to understand the benefits of a diverse set of restoration options, where the majority of restoration aiding salmon populations occurs. Ultimately, there is a need to understand how the relative population benefits of various tributary actions compare, while simultaneously accounting for harvest, hatchery, and dam actions. LCMs can make these comparisons while identifying population specific limiting factors to develop cost effective population recovery strategies.

1.3.   What did we find out?

Our efforts using LCMs for watershed and salmon population restoration identify several general patterns important to understanding how potential tributary habitat actions affect salmon species in the Columbia River Basin. First, regardless of the inputs to LCMs, the output in terms of common metrics is similar among the case studies - a comparable change in adult abundance and quasi-extinction risk with each restoration scenario. This standardization is valuable because it allows us to generate comparison among and between any group of populations.

Second, in each of the case studies, a certain magnitude of change in habitat capacity or condition needs to occur in order for there to be any type of change to capacity, survival, or overall population size. This critical effect size could be due to a limiting factor threshold being passed or the population response finally becoming sufficiently large so as to be detectable relative to background variation. This was evident in the comparison of results between the Grande Ronde River populations, where large-scale changes to juvenile parr capacity at the scale of kilometers can translate to moderate changes in adult abundance. Conversely, smaller scale projects in the Wenatchee River did not lead to detectable predicted changes in adult abundance despite a relatively rich fish abundance data. The Upper Salmon River suggests a similar pattern; the magnitude of the response scales directly with the magnitude of the change in habitat quality or quantity, with the smaller perturbations having no predicted effect on the population status. In each of the watersheds we evaluated, over the range of scenarios we included, predicted adult abundance scaled with the magnitude of restoration. This is a strong indicator that tributary restoration will benefit these populations, and density-dependence at later life stages (e.g., marine survival) will not dilute its benefits.

We again assume that a change in the modeled response is a change in the biological response. We understand that there can be biological benefits to specific restorative actions that LCMs do not capture due to the scale of the action (local) relative to the scale of the response variables (population level). We also understand that there likely are benefits or synergistic effects of actions (e.g., changes in fish growth), which we cannot currently include in our LCMs.

Third, including scenarios that go beyond the tributary habitat actions is important to give the relative context of how restoration to the salmon population(s) may occur. For example, our scenarios for the Grande Ronde River include both recent (i.e., high) and baseline (i.e., low) predation by pinnipeds on returning adults which is strongly correlated with adult return timing that varies among populations in the Grande Ronde River. Thus, a very wide range of management situations can be incorporated into LCMs and the LCMs can be used to examine how limiting factors change through time.

Fourth, there are positive synergistic effects, which can occur over the long-term. The best example of this is in the Grande Ronde where the results of riparian restoration, in combination with in-channel restoration, resulted in a larger than additive effect at the population scale over the longer time projections. This is important because the value of specific restoration actions may require timescale longer than our typical management evaluation timeframes. LCMs can thus help us better understand what suite of actions may lend themselves to the greatest overall benefit.

1.4.   What does this mean for watershed restoration planning?

The use of LCMs can allow for developing restoration prioritization schemes across a range of options through the use of common performance metrics. The exact nature of these performance metrics is flexible and depending on the need, the focus can range from stakeholder engagement, extinction risk, population viability, and narrow or broad sense recover, to dollars per fish. The other value of LCMs, as shown with the Grande Ronde

example, is how much time it may take to see benefits. Explicitly considering the time course of biological response to management actions helps manage expectations in terms of restoration investment. An "All-H" approach can easily be incorporated into LCMs, and the LCMs can be used over the course of time to examine how limiting factors change. As with any population, limiting factors will vary over time, changing with actions undertaken, varying environmental conditions, and different management actions. Thus, LCMs allow for an extremely broad range of scenario development and can help redirect both management and restoration efforts to identify actions that reduce these constraints.

1.5. What does this mean for research, monitoring, and evaluation?

LCMs currently incorporate much of the fish abundance, survival, and habitat information collected throughout the CRB. However, using the LCM approach for evaluating watershed and population level responses to habitat restoration scenarios identifies important gaps in our understanding of linkages between habitat quality and quantity and fish population processes. For example, we cannot readily assess actions that modify watershed-scale flow due to the lack of quantitative relationships between in-stream flow and rearing capacity or stage specific survival. These knowledge gaps can aid in structuring data collection and research activities as these uncertainties limit the projection of population-scale benefits of particular management strategies. Using LCMs in a decision support fashion is a form of model-based inference, and as such, addressing any limitations on parameterizing LCMs needs to be included in the overall management strategy, in our opinion, with equal importance to more tangible actions, such as direct habitat manipulation. In particular, the ability of LCMs to evaluate many different restoration actions and track their population level synergistic or competing effects through time provides a powerful predictive utility. As such, research or monitoring that can reduce model uncertainty or include habitats or life stages not currently modeled, will make LCMs a more powerful component of the watershed restoration planning process. Ultimately, LCMs can provide cost effective predictions about the efficacy of competing or complementary large-scale restoration actions to support the planning process prior to implementation.

2.   Introduction

2.1.   What have we learned from applying salmon life cycle models to measure salmon population response to tributary habitat actions?

Identifying and prioritizing salmon habitat restoration actions is guided by answering three key questions (Beechie et al. 2010): 1) How have habitats changed and altered salmon populations? 2) What are the root causes of observed habitat changes? 3) What are the constraints on habitat restoration? Answering the first two questions provides the information needed to diagnose which habitat problems have the most significant effects on a salmon population, and to identify restoration actions that are needed to address those problems. Answering the third question informs the feasibility of restoration in specific locations, as a function of socio-economic factors such as land use, water use, and land owner cooperation. Answering these questions may involve several assessments, including watershed process analyses, habitat change analyses, and salmon population analyses (e.g., Beechie et al. 2013a, Wheaton et al. 2018).

Once the necessary restoration actions and potential constraints have been identified, alternative restoration scenarios at the watershed-scale can be developed and evaluated to determine which suite of actions will likely provide the largest benefit to salmon populations (e.g., Bartz et al. 2006, Scheuerell et al. 2006, Battin et al. 2007, Roni et al. 2010, Justice et al. 2017). Ultimately, the combined effect of all restorative actions will determine the potential magnitude of change in salmon populations. The purpose of these analyses is to help focus restoration efforts on the types, location, and level of actions that lead to predictable and understandable improvement to salmon populations. Specific methods for these analyses depend on local habitat and fish data availability, and may range from simple analyses based on coarse spatial and/or temporal resolution data to more detailed evaluations with higher resolution data. Therefore, the richness of the data will determine the appropriate analysis to estimate the population response to a suite of potential restoration actions.

In this report, we describe a general approach to developing and evaluating alternative restoration scenarios and present examples of alternative methods for each step in the evaluation. Scenarios can be developed based on the habitat change analyses (Beechie et al. 1994), expected habitat configurations based on geomorphic potential (Beechie et al. 2015), or suites of actions of known effectiveness (Roni et al. 2010). Habitat change analysis identifies how current habitat types and conditions important to salmonids compares to either historic and/or potential conditions (Nickelson et al. 1992, Beechie et al. 1994, Beechie et al. 2015). Once the scenarios are developed, biological data can estimate how each restoration scenario alters estimates of life-stage habitat capacity or survival. These biological models help determine which restoration alternatives are likely to be the most important to salmon population response. The bulk of this report is devoted to examples of evaluations associated with data rich and data poor environments. In each, we show the methods and rationale for habitat and restoration scenario evaluation and incorporation into life cycle models to assess the benefit for each population.

3.   Background

Habitat isolation and degradation, the harvest of salmonids, hatchery practices, and the introduction of non-native species over the last 175 years have affected salmon populations throughout Pacific Northwest (PNW) streams (Ruckelshaus et al. 2002, Waples et al. 2008). Habitat isolation and degradation in the PNW has resulted in the virtual eradication of certain habitat types, such as large freshwater wetland and forest floodplain habitats in the lower portion of river basins (Collins et al. 2003). Slower water habitats in both the freshwater and estuarine environment have been reduced to less than 20% of their historic occurrence in various parts of the PNW region (Collins et al. 2003). Similarly, in the Columbia River basin, agriculture and urbanization have reduced the area of side channel habitats by 26% (Bond et al. 2019). In addition, mainstem and tributary habitats have been significantly lost due to hydropower or fish passage blockages such as culverts, or degraded due to forest practices, land conversion, and stream-cleaning practices that have led to the loss of in-stream wood and stream-channel types beneficial to salmon (Beechie et al. 1994, Montgomery et al. 1995, Pess et al. 2003, Sheer and Steel 2006).

Coincident with habitat degradation and isolation in Pacific Northwest streams has been a dramatic decline in the abundance of numerous salmonid populations. Gresh et al. (2000) estimated that salmon abundance (defined as the number of salmon returning to spawn) in the Pacific Northwest (from Alaska to northern California) has declined 20% to 40% since European settlement. The distribution of salmon during this time period has also changed. Historically, 84% of wild salmon returned to rivers in Alaska and British Columbia, and 16% returned to rivers in Washington, Oregon, Idaho, and California. Currently, 99% of wild salmon return to Alaska and British Columbia, while 1% return to Washington, Oregon, Idaho, and California (Gresh et al. 2000).

To combat this reduction in salmon abundance and aquatic habitat quantity and quality, stream and watershed restoration actions have been occurring throughout the Pacific Northwest for the last several decades (Reeves et al. 1995, Roni et al. 2002). Many of these restoration efforts have been at the stream reach (i.e., hundreds of meters to kilometers) or site scale (meters to hundreds of meters; Bernhardt et al. 2005, Roni et al. 2014). The subsequent physical effects of these restorative actions at the site and reach-scale, and to a lesser extent biological aspects of aquatic riverine habitats, have been well documented (Roni et al. 2002, Wohl et al. 2015). Manipulation of stream channel form and the addition of stream channel obstructions, including rock and wood, has received the most attention with regards to stream restoration effectiveness monitoring (Roni et al. 2014, Louhi et al. 2016). These efforts have documented an increase in the occupancy and density of juvenile Coho Salmon (*Oncorhynchus kisutch*), Chinook Salmon (*O. tshawytscha*), and steelhead (*O. mykiss*) in areas where structures were placed (Roni and Quinn 2001, Pess et al. 2012, Polivka et al. 2015). The increase in abundance in improved habitats was typically related to an increase in habitat capacity and not due to a redistribution of fish from other habitats of the same stream reach (Polivka et al. 2015). Another common practice is barrier removal for the longitudinal, and in some cases the lateral, connectivity of stream networks (Pess et al. 2005, 2014). Efforts related to barrier removals typically result in positive population responses to fish populations within years to decades (Pess et al. 2012, Allen et al. 2016,

Jolly et al. 2018). Riparian restoration, floodplain reconnection and enhancement, and habitat protection have also been restorative actions that have been well documented as effective in most cases at the site or reach scale (Bouwes et al. 2016, O'Neal et al. 2016), and for floodplain reconnection and enhancement at a larger scale, including population level effects (Sommer et al. 2001, Ogston et al. 2014).

Documentation of the biophysical effectiveness of such restoration actions has been important, but does have limitations. Action effectiveness monitoring, while useful for simple modeling purposes to examine the potential effects of restoration actions on physical habitat and aquatic biota at the site or reach scale (Roni et al. 2010), does not allow us to examine restoration benefits at the population scale and within the context of the entire salmonid life cycle. Survival or capacity benefits resulting from restoration actions in one life stage could potentially be rendered insignificant by survival bottlenecks occurring in other life stages. Moreover, restoration projects may be locally beneficial, but if they do not target the limiting factors for a population their benefit will be limited. Therefore, accounting for multiple life stages through life cycle modeling is a necessary and important step to linking food web and salmon population level response to restorative actions (Beechie et al. 2013a, Ogston et al. 2014, Bellmore et al. 2017), and has been done for several populations throughout the Columbia River Basin (Honea et al. 2009, McHugh et al. 2017).

Intensively monitored watersheds (IMW) are another study design used to examine watershed scale effectiveness of restoration actions on entire salmonid populations, including each life stage. Starting in the late 1950s with the Alsea Watershed Study in the Oregon Coast Range, these watershed-scale restoration experiments have been demonstrated to be the most direct techniques to understand salmonid population responses to stream and watershed-scale actions (Bennett et al. 2016). In some cases, salmonid response to these actions has resulted in increases to the survival, productivity, and overall abundance of juvenile salmonids (Bennett et al. 2016). For example, watershed-scale restoration efforts focused on floodplain connection in the upper Chilliwack watershed identified that between one-half to one-third of the overall Coho Salmon smolt outmigration was attributed to those created and/or restored habitat types (Ogston et al. 2014). Increases in juvenile Coho and steelhead densities and smolt productivity have been documented at the watershed and population scale in response to multiple restorative actions including in-stream wood and nutrient additions, despite reductions in adult escapement in the Keogh River, British Columbia, Canada (Ward et al. 2003). Large-scale restoration actions such as dam removal have also shown population level responses. Dam removal on the Elwha River has resulted in Coho Salmon population level responses of relocated hatchery-dominated Coho Salmon adults into newly available habitat (Liermann et al. 2017). Specifically, Liermann et al. (2017) documented immediate freshwater production that was comparable to other systems throughout the Pacific Northwest.

Efforts in the Columbia River basin have also shown salmon population level responses to various metrics important to their recovery. Significant increases in the density, survival, and production of *O. mykiss* were documented in the John Day River, OR, using beaver dam analogs (BDAs), a restoration technique that increases in-stream habitat quantity and quality (Bouwes et al. 2016). The Entiat River IMW uses a hierarchical-staircase design to examine

habitat actions that have been implemented to test the hypothesis that increasing in-stream complexity will result in increases in density, growth rates, survival, and productivity of juvenile salmonids (Hillman et al. 2016). To date, habitat monitoring has shown a significant increase in the volume of wood in the Entiat River, but treatment or monitoring has not occurred long enough yet to determine population level responses (Hillman et al. 2016). However, there are higher densities of juvenile Chinook and steelhead using off-channel habitats compared to main channel locations within the Entiat River (Hillman et al. 2016). In addition, off-channel habitats located in the upper watershed produced more yearling spring Chinook smolts than those in the lower watershed (Hillman et al. 2016).

While these and other studies support the hypothesis that stream- and watershed-scale restoration efforts can affect salmonid populations in a positive fashion, differences in the natural characteristics, anthropogenic history of a watershed, and restoration actions and magnitude make extrapolation to the population level difficult. Extrapolation of such information from site, reach, and watershed-scale response studies is a necessary and important restoration planning and prioritization step because watershed-scale restoration and associated salmon population level response is typically the primary goal of such actions. This issue becomes more acute with limited financial resources to implement stream and watershed restoration for the purposes of salmon recovery. The pressure to implement effective actions and efficiently use restoration funds necessitates the evaluation of restoration actions in the context of the entire life cycle of each population, and ultimately the benefits to adult salmon abundance.

For decades, life cycle models (LCMs) have been used to estimate demographic response to changes in capacity and/or survival in salmon populations (e.g., Nickelson and Lawson 1998, Kareiva et al. 2000, McHugh et al. 2004, Scheuerell et al. 2006, Zabel et al. 2006, Honea et al. 2009). The simplest LCMs evaluate only a single stage, typically adult spawner abundance and the resulting adult offspring, with relationships between them that can represent the dependence of offspring numbers on the abundance of their parents (e.g., Ricker 1954, Beverton and Holt 1957, Barrowman and Myers 2000, Buhle et al. 2018). By incorporating density-dependence, these models predict population size as a function of the parent spawner abundance. However, similar density-dependent processes (e.g., growth, survival) may decouple the observed benefits of restoration at juvenile life stages from meaningful increases in adult abundance. To account for this decoupling, LCMs for salmon have become more life history specific, often including several juvenile stages (e.g., egg, fry, parr, pre-smolt, smolt) and density-dependent processes therein (Crozier et al. 2008, Honea et al. 2016, McHugh et al. 2017). Thus, the multi-stage aspect of these models has become a powerful tool for simultaneously evaluating the population response from multiple habitat or management actions, or single actions that affect more than one life stage. In addition, the LCM approach can determine what the magnitude of a restoration action must be to achieve a desired population increase. These outputs can provide a more effective cost-benefit analysis to choose among a suite of potential habitat actions. In addition, in an LCM framework we can effectively compare the relative merits of restoration actions and other management scenarios (e.g., harvest, dam operations, predator removal).

Some studies have evaluated the proximate effects of stream restoration in an attempt to optimize the size or type of restoration activity (Roni 2019). The benefits of restoration may be difficult or impossible to observe for several reasons. First, restoration actions need to be implemented with sufficient intensity—large spatial extent over a short temporal duration—to result in a detectable population benefit. That is to say, a change in habitat quality or quantity (HQQ) will result from stream habitat restoration actions, and fish biological processes at the individual scale (growth, survival, movement, life history expression) will be affected in a manner that positively benefits the population (fresh water productivity, population growth rate, extinction probability) such that the signal can be detected above the measurement and process error that plague the estimation of population-scale fish abundance through time. Second, restoration project selection rarely requires an analysis to determine whether projects are addressing the factors that are currently limiting population growth. Two primary practical considerations, intensity and constraint, prevent this approach from being the default strategy for linking habitat restoration actions with fish population benefits (Bennett et al. 2016). With respect to intensity, it is extremely difficult to coordinate a watershed-scale set of actions that result in a significant change in HQQ, with the possible exception of dam removal (e.g., the Elwha River project). In terms of issues of population process constraint, often multiple ecological impairments are present and, as such, a restoration strategy must address all before capacity or productivity improvements occur. Similarly, multiple conflicting management strategies might also be present in a population, thereby confound our ability to detect the impact of restoration actions (e.g., supplementation programs overwhelming natural production, or concurrent in-stream or upland resource extraction practices). Thus, only in rare cases is the tributary environment of a salmon or steelhead population amenable to direct manipulation to demonstrate the positive benefit of restoration actions at the population scale (e.g., IMWs; cf. Bennet et al. 2016). However, an LCM approach can address issues of project scale and limiting factors within the confines of the available abundance data to determine whether a project or suite of projects are likely to produce a useful population benefit.

A parsimonious approach to generating the necessary estimates of population benefits to suite of watershed restoration actions is to combine the best available science on individual fish response to changes in habitat quality and quantity, with measured changes in habitat features from restoration actions. LCMs represent an important management support tool because a population forecast model will generate alternative futures based on a standardized suite of input population biological parameters (e.g., stage specific capacity, productivity) and action scenarios (e.g., habitat action type or extent, climate change, migration corridor conditions). LCMs estimate the aggregated (over time and space) effects of changes in habitat on entire populations of salmon or steelhead. LCMs are thus a standard approach to understanding the physical and biological processes underlying population dynamics (Nickelson and Lawson 1998), and LCMs have a long history of application in conservation and population management situations (Morris and Doak 1998, McClure et al. 2003, Heinrichs et al 2017).

The management of ESA-listed salmonid populations presents an ideal setting for applying LCMs as a key component of a decision support system. Salmonid life cycles encompass vast geographic ranges, and given this necessary degree of habitat diversity, the opportunities for impacts due to human activity is manifold. Developing effective management strategies

to conserve and recover ESA-listed salmonid populations involves balancing a portfolio of potential actions that are applied across life-stage, habitat type, jurisdiction, and anthropogenic impact type (Bartz et al. 2006). The most commonly utilized tool for salmonid conservation and recovery is freshwater tributary habitat rehabilitation. However in order for population recovery to occur we need to incorporate additional factors that affect population dynamics including harvest, hatchery, and dam actions in order to develop cost-effective population recovery strategies.

In this report, we aim to address two main questions: What methods and data are needed to estimate salmonid population response to various stream and watershed restoration actions? And, how can current and potential stream and watershed restorative actions increase the potential salmon population response in a suite of watersheds in the Columbia River basin in the short (i.e., years) and long-term (i.e., decades)? First, we document methods for translating habitat actions into changes in salmon LCM inputs and demonstrate how those actions influence LCM based projections of abundance, productivity, spatial structure and diversity. We then use the combination of habitat and restoration data with LCMs in a "data rich" and "data poor" environment to estimate changes in three general locations - the Upper Grande Ronde, the Wenatchee River, and Upper Salmon River basins. In all three cases, we focus on Columbia River Basin (CRB) spring/summer Chinook salmon populations to contrast results from more detailed habitat and fish information with results generated using the types of information more generally available across CRB spring/summer Chinook populations. In each watershed, we constructed LCMs that account for changes in habitat among restoration scenarios, and used these models to estimate a demographic response of the population. We focus on metrics of salmonid population dynamics that are important for management of these salmon populations (e.g., average abundance or risk of falling below a given threshold). We then describe how these approaches can differ and provide examples of this framework to illustrate the potential approaches to this process.

4.   Methods

4.1.   Overview

Evaluating the likely outcomes of alternative restoration scenarios includes four steps (Table 4.1). Each step can be accomplished through a variety of methods, depending on the types and resolution of habitat and fish data available. In the first step, we develop and/or use existing LCMs to develop habitat capacity estimates for each life stage. This first step requires a compilation of all necessary available data. These data typically include habitat specific capacity, survival, and abundance. This allows for the development of an LCM at the appropriate level of detail given the data that we have and our understanding of limiting factors (i.e., which key habitat or management variables will be included in the model). If data are lacking to parameterize the LCM, a choice between collecting the necessary data or utilizing the parameters and functional relationships from nearby basins or the general literature to inform the model is necessary.

Table 4.1. Approach and steps for using LCMs to estimate and compare the benefits of restoration actions at population scales.

Analysis step	Description	Example citations
1. Develop salmon LCM and habitat capacity, growth, and survival models.	Capacity, growth, and survival estimates are derived for each salmon life stage using a variety of models. These models are then calibrated and compared to existing salmon population estimates.	—
2. Develop restoration scenarios.	Alternative scenarios quantify habitat changes based on: 1) habitat change analyses, 2) expected habitat configurations based on geomorphic potential, or 3) suites of actions of known effectiveness.	Beechie et al. (1994) Beechie et al. (2015) Roni et al. (2010) Wheaton et al. (2017) McHugh et al. (2017)
3. Estimate restoration effects on habitat capacity or survival.	Each restoration action or action type is translated into effects on habitat capacity and/or survival at specific life stages for each species for input to life cycle models.	Beechie et al. (1994) Bartz et al. (2006) Jorgensen et al. (2009) McHugh et al. (2017)
4. Estimate population-level outcomes of each restoration alternative.	Life cycle models (or other life cycle analyses) incorporate changes in life stage capacities and survivals to project changes in population performance.	Scheuerell et al. (2006) Battin et al. (2007) Honea et al. (2009) McHugh et al. (2017) Justice et al. (2016)

In the second step, there is a need to develop alternative restoration scenarios to represent a range of real or hypothetical options for restoration. The purpose of the scenarios is to provide a reasonable estimate of the magnitude of restoration benefit for each scenario in order to help managers select the best options for improving a salmon population. Scenarios should be diverse enough to allow comparison across a range of realistic options, and can focus on the effect of individual action types in order to help diagnose which habitat actions have the largest impacts on a population.

In the third step of the analysis, there is a translation of the effects of each restoration scenario into changes in either habitat capacity or survival at specific life stages. For example, addition of wood structures to a channel may increase both summer and winter rearing capacity, as well as changes in life-stage survivals (Solazzi et al. 2000, Gregory et al. 2003). By contrast, a change in spawning gravel quality by decreasing percent fine sediment would not alter spawning capacity, but would increase egg-to-fry survival (e.g., Jensen et al. 2009).

Finally, in the fourth step the changes in capacity and survival from the restoration scenarios are inputs to a LCM to assess the overall change in salmon abundance and productivity, and potentially to estimate change in spatial structure and diversity as well. LCMs can vary considerably in complexity, particularly in the number and specificity of life-stages included in the model. In general, more complex models allow for a greater range of restoration scenario development, however require more data. Conversely, less complicated models have a more limited range of restoration scenario development, but do not require the same amount of input data.

4.2.   Developing salmon life cycle (LCM) and habitat capacity models

4.2.1.   Capacity estimation

As LCMs can evaluate the benefits of various restoration or management strategies, thus there is a need for estimates of the potential effects of management actions at multiple life stages. The advantage of multi-stage LCMs is realized through a limiting factors analysis, where the demographic benefit of habitat actions can be evaluated on each life stage simultaneously. LCMs typically predict the abundance at one life stage as a function of the previous life stage through a recruitment relationship that assumes density-dependence (Crozier et al. 2008). These transition functions typically include productivity and capacity terms (e.g., Beverton–Holt, Ricker). This process becomes a powerful evaluation of restoration because increases at capacity at one stage may result in little or no gain in adult abundance if one or more successive stages are at capacity. Therefore, accurate representation of capacity terms becomes vital as stream restoration projects seek to maximize abundance, growth, or survival at one or more stages. This requirement has led to the development of a suite of approaches to estimate juvenile salmon rearing capacity that are sensitive to specific restoration actions at multiple spatial scales. Similar to capacity, stage transition survival estimates are additionally important components of many LCMs.

An important distinction is that "capacity" can refer to either the habitat capacity or population capacity. Here, we define habitat capacity as the maximum number of individuals that can be supported by the habitat and life stage of interest, often referred to in ecology as the carrying capacity (del Monte-Luna et al. 2004). In contrast, population capacity is the asymptotic capacity, or maximum equilibrium population size that is often derived from fitting models to observational data (Milner et al. 2003). Both of these terms differ from production or productive capacity, which often refers to the sum of all individuals or biomass produced in a habitat per unit time (Ricker 1975, Wurtsbaugh et al. 2014). This distinction between habitat and population capacity is important because most life cycle modeling efforts require or directly estimate population capacity, but evaluations of stream restoration actions typically estimate habitat capacity. The discrepancies between these estimates can, in part, be reconciled by scaling habitat capacity estimates (e.g., % change in habitat capacity in place of number of individuals) or only using a portion of the available data (e.g., 90th percentile, see quantile regression below) to more closely approximate population capacity.

Capacity estimation methods can be partitioned into two broad classes; empirical models that are fitted to data, and mechanistic models that use a set of mathematical equations to describe the underlying processes driving fish distribution, growth, survival, or abundance. In the latter case, model parameters are typically derived from independent analyses of fish–habitat relationships or from related literature. To capture the breadth of current knowledge on physical and biological determinants of salmonid population processes, "fish–habitat relationships" is very broadly defined to include bioenergetics mechanisms and ecological interactions, as well as the more classical physical and physiological limitations imposed by a gradient of habitat quality and quantity. These estimation methods, and models on which they are based, vary widely because of the grain and extent of the estimation needs and available data, life stage of interest, and the specificity of the attributes included in the evaluation. Thus, when applied to evaluate restoration objectives, each approach may be suited to evaluate the effects of different restoration actions at different spatial scales. Here, we provide a brief review of models employed to estimate contemporary capacity in the river basins.

4.2.2. Empirical models

There is a suite of approaches to estimate juvenile rearing capacity currently being employed in the Columbia Basin. Although each approach can estimate contemporary capacity, each approach utilizes a different suite of attributes. Thus, each approach may be more or less suited to evaluate the effects of different restoration actions depending on the spatial scale of interest. Empirical models are models developed using observational or experimental data. This is a common approach in fisheries management, and a primary component of most modeling efforts.

4.2.2.1. Stock–recruitment models

When a series of stage-specific abundance data are available, stock–recruitment (S–R) models can explore the transition dynamics between successive life stages. With salmonids, S–R models are typically fit to estimates of redds or spawners, and the resulting juvenile production at a life stage of interest (Liermann et al. 2010), or from one juvenile stage to another where data are available. Typically, juvenile production estimates are produced from emigrating juveniles using downstream migrant traps, or for resident stages with snorkeling or multi-pass electrofishing. Although these models can take many forms (e.g., Ricker, Beverton–Holt, hockey stick, etc,), all estimate the density-independent productivity of the population at low abundances and the asymptotic capacity, or long-term average capacity, of the population (Ricker 1954, Beverton and Holt 1957).

For data-rich populations, S–R models are a robust method for estimating contemporary capacity, and are the "gold standard" for estimating population parameters. However, the weaknesses of S–R model become apparent when populations lack numerous years of monitoring data, are contrast poor, lacking the range in abundance needed to estimate population capacity, or when the population dynamics are overwhelmed by abiotic drivers with strong temporal patterns (e.g., PDO, climate change). Additionally, S–R models are famously noisy, owing to both the observation error in abundance estimates at each life stage, and the range of environmental conditions captured by decades of monitoring. New techniques, including hierarchical models and integrated population models have sought to alleviate some of the traditional limitations of S–R models by sharing information among populations (shrinkage) and modeling observation and process error separately (Buhle et al. 2018). Although these new approaches offer the ability to incorporate coarse environmental covariates, few populations have enough data to explicitly model how ecological conditions have changed over the monitoring period (Neuswanger et al. 2015), or restoration actions (Scheuerell et al. 2015).

4.2.2.2. Quantile regression forest models (QRF)

Quantile regression forests (QRF) is another empirical model fitting approach that attempts to deal with some of the shortcomings of other empirical approaches (Meinshausen 2006). QRF models evaluate the relationship between environmental co-variates and a quantile of observed fish densities (commonly 90th percentile; Cade and Noon 2003). Here, the assumption is that the upper percentiles of observed fish densities are at or near capacity at reach or habitat unit scales, even if the overarching population is below capacity. QRF models can also describe the entire distribution of predicted fish densities for a given set of habitat conditions, not just the mean expected density. QRF models area used in a variety of ecological systems to estimate the effect of limiting factors (Cade and Noon 2003, Prasad et al. 2006).

Quantile regression forests are an extension of a machine learning approach called random forests, which are an ensemble of many regression trees. Each tree is a subset of predictors and data, producing the most parsimonious relationship between the predictor and response. Random forests can effectively deal with non-linear responses, and are typically resistant to overfitting and collinearity. Random forest models (RFM), in

some cases, outperform more standardized parametric models in predicting fish–habitat relationships (Knudby et al. 2010). The same benefits are shared between QFRs and RFMs including the ability to capture non-linear relationships between the independent and dependent variables, naturally incorporate interactions between covariates, and work with untransformed data while being robust to outliers (Breiman 2001, Prasad et al. 2006).

The habitat data used to develop recent QRF models used in the Columbia River Basin are part of the Columbia Basin Habitat Monitoring Program (ISEMP/CHaMP 2015).[1] CHaMP sites are 120- to 600-m reaches within wadeable streams across select basins within the interior Columbia River Basin and were selected based on a spatially balanced Generalized Random Tessellation Stratified (GRTS) design (Table 4.2; Stevens and Olsen 2004). CHaMP habitat data include, but are not limited to, measurements describing: channel units, channel complexity, fish cover, disturbance, riparian cover, size (depth, width, discharge), substrate, water quality, large woody debris, and temperature.

4.2.2.3. Structural equation modeling (SEM)

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. 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. Traditional SEMs, like most other regression-based approaches, estimate the influence of predictor variables (e.g., habitat condition) on the average value of a response variable (e.g., fish abundance). SEMs provide a flexible structure that allows for more data types and structures than habitat expansion or QRF methods. They are currently being employed in the upper Grande Ronde River to evaluate the relative effects of landscape position, large woody debris, pool availability, and water temperature on abundance of Chinook parr in the Grande Ronde River basin (White et al. 2018). However, many watersheds currently lack the habitat and fish data needed to utilize the SEM approach.

4.2.2.4. Habitat expansion models

In some watersheds, lacking habitat and fish data means extrapolation is necessary in order to gain estimates of fish utilization at the appropriate scale for the development of watershed-wide restoration scenarios. Habitat expansion models directly extrapolate habitat capacity at any spatial scale by multiplying the amount of available habitat by the maximum density at which fish occur in each habitat, and summing all habitats of interest (Bartz et al. 2006, Beechie et al. 2015, Bond et al. 2019). This calculation assumes that maximum densities are constant across different units of the same habitat type. Depending upon the availability of habitat estimates for a given watershed, fish densities at any spatial

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[1] https://www.champmonitoring.org

Table 4.2. CHaMP-derived habitat metrics included in the summer parr rearing capacity QRF model.

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

grain could be used, from fine (habitat unit) to coarse (channel form). These models do require that habitat estimates be spatially extensive, but require no further extrapolation. Application of habitat expansion models, which are typically built with coarse scale geomorphic controls are best suited to estimate the effects of large scale planform channel restoration (Beechie et al. 2015, Bond et al. 2019). Mechanistic models are likely to inform smaller spatial scale restoration actions (e.g., wood addition, riparian cover) that evaluate the bioenergetics of fish in response to changes in stream conditions.

4.2.3. Mechanistic models

Empirical models typically lack the experimental manipulations needed to identify and validate causal mechanisms, and thus understanding why these complex assemblages of variables interact to describe fish habitat requirements is difficult. The basis of mechanistic models such as drift-foraging bioenergetics models are typically experimental or comparative studies confirming patterns described by mathematical models based on ecological theory (Fausch 1984, Hughes and Dill 1990). Therefore, these models can be more robust for predictions, but they still allow for the evaluation of alternative management scenarios (Nislow et al. 1999, Hayes et al. 2016, Wall et al. 2016). However, due to their complexity, they can be data intensive and difficult to calibrate and validate (Piccolo et al. 2014, Rosenfeld et al. 2014).

Microhabitat models, such as habitat suitability models, straddle empirical and drift-foraging bioenergetics approaches (Rosenfeld et al. 2016). Proximate cues that fish are responding to such as depth, velocity, and substrate, describe habitat quantity and quality and can in part be driven by mechanistically based hydraulic models. However, the rule sets used in building these relationships are based on fish preferences of these variables which are often site-specific, making extrapolation difficult or uncertain (Rosenfeld 2003).

The basis for drift foraging models is typically optimal foraging theory to describe the feeding behavior and location position of drift feeding salmonids, positing that fish choose locations with optimal energetic value (Hughes and Dill 1990, Hill and Grossman 1993). A drift-feeding fish's net energy intake (NEI) or net rate of energy intake (NREI) is energy gains through capture and consumption of drifting invertebrates minus energy cost through swimming to maintain a foraging position. These models were initially validated with intensive observations of feeding locations (Fausch 1984, Hughes and Dill 1990, Addley 1993, Hill and Grossman 1993, Guensch et al. 2001), but have also been used to successfully predict growth and abundance (Nislow et al. 2000, Hayes et al. 2007, Urabe et al. 2010, Wall et al. 2015). The NEI model incorporates data on depth, focal velocity, prey abundance (drifting invertebrates) to predict prey encounter rates, capture success, and consumption rates at locations throughout the modeled environment. These variables can simply be measured throughout a reach and converted to NEI (Guensch et al. 2001, Urabe et al. 2010). Alternatively, depth and velocity results from hydraulic models can also be used to estimate these inputs (e.g., Wall et al. 2015), and additionally drift transport rates (e.g., Hayes et al. 2007). Bioenergetics models estimate gross energy input (GEI) from prey consumed and swimming costs (SC) at the focal velocity under a given temperature, with GEI − SC = NEI. In watersheds where hydraulic models have been developed, LCMs can include a change in HSI or NREI as a change in the capacity.

4.2.4. What is the scale of the fish–habitat relationships being estimated?

The scale at which fish–habitat relationships are estimated is important. There are typically three: stream reach, watershed, and a hybrid of the two, each with advantages and disadvantages.

The reach scale typically has the advantage of a large amount of data and more detailed habitat metrics. This also means the need for more assumptions to aggregate the data, such as the movement patterns associated with the habitats and the juxtaposition of the habitats in relation to such movement patterns. In addition, there is typically more variability at the finer grain due to factors such as the location of spawning adults and the dispersal of their offspring.

We do not need to aggregate predictions at the watershed scale. In addition, the average incorporates a large amount of variability and naturally accounts for the juxtaposition of different habitat types. Typically, there are fewer data at this scale, as well as less detailed habitat data. Similarly, most restoration occurs at the reach scale and a mismatch between watershed scale fish and habitat estimates and reach scale restoration leads to compromises in evaluating restoration actions.

Hybrid models typically use stream reach-scale relationships to create indices and then establish relationships between these indices and watershed-scale fish data using a fitting or calibration process. This is a form of dimension reduction (like PCA) driven by reach scale data and expert knowledge. More than one metric typically is used. Using both types of data side steps some of the disadvantages described in the preceding methods. This does assume that watershed-scale habitat can be effectively described using the habitat index derived from the reach-scale data.

It is important to note that we take advantage of all of the different types of models. They each have their strengths and weaknesses. For example, the sub-reach scale models can provide detailed predictions about how changes in bathymetry would affect LCM parameters. Ultimately, however, fish data are the gold standard. These bottom-up models are useful for developing our understanding of the mechanisms but if they cannot predict patterns we see in the fish data, we need to use them with care.

4.2.5. Calibrating or "fitting" the models

We fit model parameters to data. For example, in the situation where stream reach-scale relationships are used, the fit could be as simple as calculating the average density of fish observed in each habitat type, or as complex as quantile regression forests or structural equations models. There are two stages where hybrid models are fit to data. The first step is the development of initial parameter estimates or habitat capacity metrics using fitted fish–habitat relationships at the stream–reach scale. In the second step, these values are adjusted to agree with watershed basin scale fish data. These data could be adult spawners for the entire basin or could include fish abundance at different stages along with survivals between stages.

There are two approaches used in the LCMs described in the methods and used as examples in this document—calibration and statistical model fitting. In the calibration procedure, the LCM parameters are first developed independently based on the literature and reach scale data, and then a subset of the LCM parameters are adjusted to produce fish population predictions that are in closer agreement with the basin scale fish data. Statistical model fitting is similar in spirit but all of the model parameters are fit in a single process (e.g., the state space model fit using Markov chain Monte Carlo (MCMC). Instead of adjusting/ calibrating LCM parameters based on stream–reach scale analyses, the analysis is used to derive metrics that are then related to parameters in the LCM. For example, parr capacity might be modeled as $parrCap = a \times X$, where X is a metric of parr habitat derived from the reach scale data and a is a slope parameter that is estimated in the modeling process, which includes fish data. This allows for a description of uncertainty in this relationship, which in turn percolates through the statistical analysis into uncertainty in model projections.

4.3.   Developing restoration scenarios

The development of stream and watershed restoration scenarios requires a habitat change analysis prior to examining how a suite of actions can change salmon habitat capacity and potential population size. The basis for the habitat change analyses is a comparison between historical habitat conditions, current habitat conditions, and potential habitat conditions developed from the geomorphic settings in a watershed. Stream and watershed restoration scenarios can then examine the benefits of potential restoration effectiveness including climate change.

4.3.1.   Habitat change analyses

A habitat change analysis, in its simplest form, identifies how current habitat conditions important to salmonids and watershed health compares to either historic and/or potential conditions. Habitat conditions typically include but are not limited several categories including habitat quantity (i.e., stream channel area, pool frequency, floodplain condition), habitat quality (i.e., pool frequency, floodplain condition, wood loading, fine sediment levels, riparian condition), environmental conditions (i.e., stream temperature, streamflow), indicators of habitat quality (i.e., adjacent land use), and causes of habitat degradation (i.e., water diversions and barriers). Each of these variables mentioned can have an impact on salmon habitat capacity and their survival, and can affect or impact at a single or multiple life stages (Montgomery et al. 1999, Pess et al. 2011, Walters et al. 2013, Ogston et al 2014, Bouwes et al. 2016).

Analyses of change in habitat quantity or quality from a reference or historical condition can be a powerful approach to informing restoration scenarios. Here, we define habitat quantity as the amount of suitable habitat on a linear (m), aerial (m^2), or volumetric basis (m^3). Typical examples of habitat quantity include river km accessible to migrating fish below passage barriers (Sheer and Steel 2006) or surface area of a certain habitat type (Bartz et al. 2006). Habitat quantity is often linked to carrying capacity in life stage and life cycle models (Scheuerell et al. 2006). Habitat quality refers to biophysical conditions ranging from poor to excellent within an organism's range, as compared against some

reference or historical baseline. Organisms respond to habitat quality in several ways including changes in abundance on a linear (fish abundance/m), aerial (fish/m^2) or volumetric basis (fish/m^3); or through differential survival or productivity (Holtby and Scrivener 1989). In many cases, the concepts of habitat quantity and quality are interrelated. For example, the amount of river habitat above a tolerable water temperature threshold could be considered a loss of habitat quantity (Beechie et al. 2008), but even if water temperature is reduced to below threshold values the benefit to fish would only gradually increase as temperatures reached optimal conditions (habitat quality).

4.3.1.1. Channel area

Habitat quantity is one of the most important categories to quantify in developing restoration scenarios (Beechie et al. 2015). A change in habitat quantity, in its simplest form can be comparing the current v. historic conditions of stream channel width (White et al. 2017). One example of how this can be accomplished is through the current stream habitat surveys and the development of estimated stream channel widths using historic records such as General Land Office (GLO) notes (White et al. 2017). However, care must be taken when considering stream habitat quantity as indicated simply by relative planform channel area since the biologically useable habitat is given by the wetted portion of the channel planform, and with intensive seasonal water withdrawals can dramatically reduce usable area. Similarly, estimates of wetted habitat area from remote sensed imagery or site visits will be sensitive to the flow on the date of imagery or survey.

4.3.1.2. Pool frequency

Pool frequency is another common metric used to gain a better understanding of both habitat quantity and quality (Montgomery et al. 1999, Bouwes et al. 2016, Favrot et al. 2018). Pool frequency can affect juvenile salmon rearing capacity (Beechie et al. 1994), adult spawning preferences (Montgomery et al. 1999), and the overall quality of juvenile rearing habitats which also allow for increases in densities and survivorship (Bouwes et al. 2016). Utilizing historic information, coupled with geomorphic classification and historic riparian condition can allow for a comparison similar to stream channel width. White et al. (2018) compared present-day estimates of large pool (maximum depth >0.8 m, surface area >20 m^2) frequency (pools/km) to historical information (McIntosh et al. 2000) in the Grande Ronde Basin (Figure 4.1). Such information can spatially identify areas most impacted by historic land use activities including splash dams (White et al. 2018). Historic and current pool frequency data can be examined in relation to other types of habitat change analyses, such as current vs. historic riparian condition and give a better spatial understanding of how overall in-stream and riparian conditions have changed together (Bilby and Ward 1991, McFarlane et al. 2017, Goss and Roper 2018).
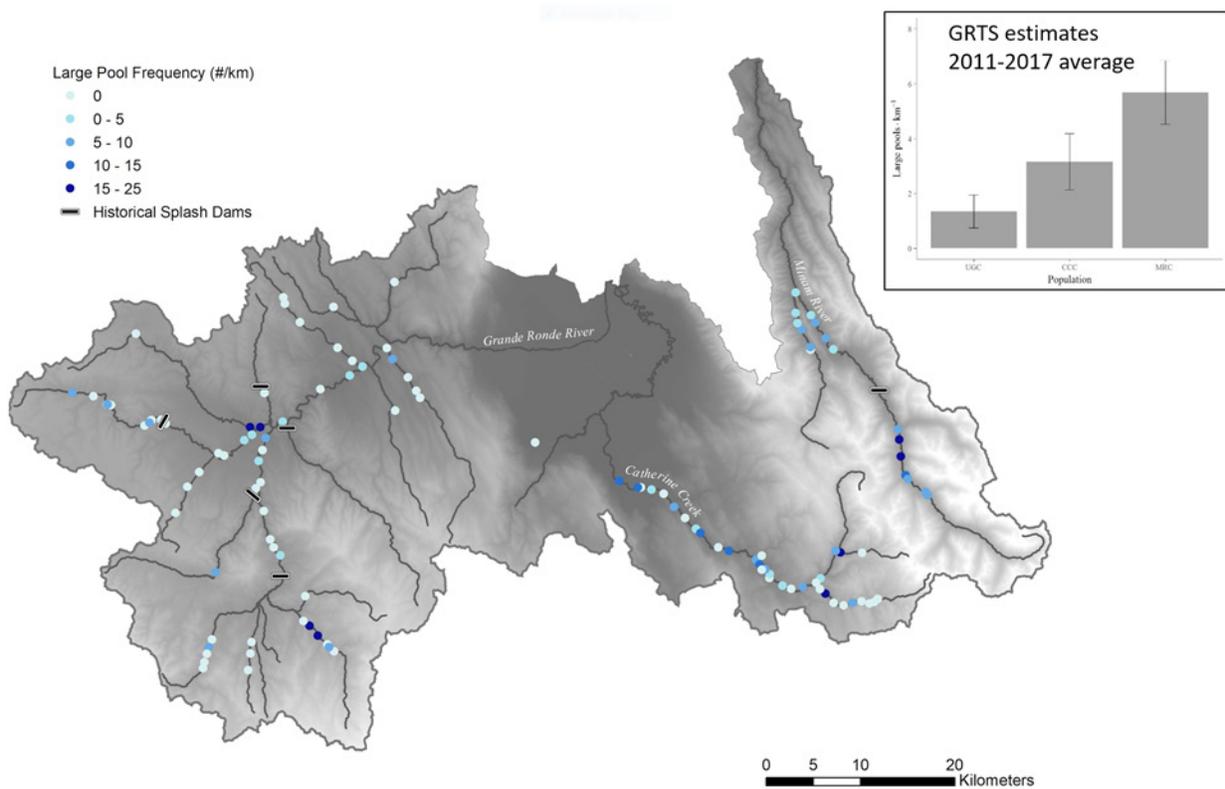
Figure 4.1. Present-day pool frequencies demonstrated important to rearing juvenile Chinook salmon in the upper Grande Ronde (UGC), Catherine Creek (CCC), and Minam River (MRC) watersheds (White et al. 2018).

4.3.1.3. Floodplain condition

Floodplain habitat extent and condition is another habitat quantity metric important to salmonids (Quinn and Peterson 1996, Whited et al. 2013). One concept that captures the importance of floodplains to salmonids is habitat complexity. Habitat complexity, in this context, is the expansion of the distribution of velocities, depths, habitat types, stream temperature, and food resources due to the existence and connection of floodplains to main stems and tributaries (Hicks et al. 1991, Bunn and Arthington 2002). Floodplains allow for the development and maintenance of multi-threaded channels, which allows for a greater diversity of the preceding physical and biological attributes. This, even under altered conditions, give salmonids multiple options for each life stage and can result in greater growth and survival opportunities (Sommer et al. 2001, Jeffres et al. 2008). In most cases, floodplains offer salmonids additional rearing capacity, especially during the wet months when juvenile where fish seek rearing opportunities (Nickelson et al. 1992, Sommer et al. 2001). These areas also facilitate increased growth and survival by offering abundant prey, optimal rearing temperatures, and refuge from predators (Jeffres et al. 2008, Bouwes et al. 2016, Bellmore et al. 2017). Given widespread losses and the potential benefits of floodplain habitats, scientists are increasingly recognizing the importance of restoring habitat complexity and floodplain connectivity to recovery of threatened species (Takata et al. 2017).

Differences in the historic vs. contemporary floodplain connectivity can have large-scale changes to the amount and utilization of habitats by salmonids. Historic loss of floodplain habitats in North Puget Sound, Washington State resulted in reductions of juvenile coho salmon capacity of 34% to 86% (Beechie et al. 1994, Pollock et al. 2004). Even when salmonid populations are well below habitat capacity, salmonids show preferential utilization of floodplain habitats (Pess et al. 2008). For example, salmonid carrying capacity in the floodplain habitats of a Columbia River tributary relative to the main stem were 251% higher, on average, for anadromous salmonids based on preferred food resources (Bellmore et al. 2013). Thus whether there is loss or utilization the restoration floodplain connectivity, heterogeneity and complexity is important to their recovery and persistence of salmonids (Bellmore et al. 2013). Comparing historic or potential to current condition, regardless of the method utilized is an important component to changes in potential salmonid capacity and productivity and should be part of any analysis of freshwater habitat capacity.

4.3.1.4. Anthropogenic barriers

Anthropogenic barriers to migration such as culverts, dams, levees and dikes associated with floodplains and estuarine areas, and water diversions are a common cause of a reduction in the total area available to salmonids . Much has been published on the topic of salmon habitat loss due to barriers (cf. McClure et al. 2008). Barriers, at a watershed scale, can reduce the amount of salmonid habitat anywhere from less than 1% to over 90% of historic capacity (references). Sheer and Steel (2006) found that almost 15,000 km (approximately 40% of total stream fish habitat) was blocked to salmon access in the Willamette and Lower Columbia River basins. Furthermore, they found that population-based abundance scores for spring Chinook salmon were strongly correlated with the magnitude of habitat lost and the number of lowland fish passage barriers (Sheer and Steel 2006). Even when items such as culverts are permitted to pass salmonids at all life stages, they can in fact be a barrier to migration (Price et al. 2010). One important habitat assessment procedure that is used is the evaluation of fish migration blockages (Beechie et al. 2013b). Portions of tributaries and other habitats that are blocked from fish access can be mapped and estimates or inventories of habitat upstream of migration barriers can be quantified to estimate the amount of habitat which is disconnected (Beechie et al. 1994, Pess et al. 2003, Sheer and Steel 2006).

4.3.1.5. Riparian conditions

Understanding the historic changes or potential to riparian condition is an important component to the development of restoration scenarios for streams (Wissmar and Beschta 1998). Riparian conditions effect a vast assortment of stream channel conditions and functions including but not limited to stream channel width, stream channel type, stream temperature, and wood loadings. In turn these factors affect both habitat quantity and quality for salmonids at each life stage in the freshwater environment.

Habitat change analyses of riparian vegetation can be determined either through historic reconstruction from historic datasets (Beechie et al. 2010) or through understanding the potential for growth based upon soil conditions, landscape attributes (i.e., elevation, ecological zonation, slope, aspect) or a combination of both historic and potential (Pollock et al. 2012, Macfarlane et al. 2017). Riparian condition has been robustly estimated at a watershed-scale, integrating the natural patchiness across time and space (Reeves and Bisson 2009). An excellent example within the Columbia River basin that was recently conducted which gives potential riparian vegetation is in the Grande Ronde River Basin (see Grande Ronde River case study below and Justice et al. 2017).

Another method that focuses on remote sensing data rather than more localized field data is the riparian vegetation departure index (RVD; Macfarlane et al. 2018). The RVD index calculates riparian vegetation's departure from its historic condition as the ratio of current vegetation cover to estimated historic riparian vegetation cover (Macfarlane et al. 2017). The objective with this method is to quantify the proportion of native riparian cover within each polygon (Macfarlane et al. 2017). The scale goes from "0" (a complete departure from native riparian vegetation) to "1" (the same as historic native riparian vegetation), with numbers near one meaning small differences between historic and current (Macfarlane et al. 2017). Macfarlane et al. (2018) uses this method in several major watersheds in the Columbia Basin (Entiat, Wenatchee, John Day, Upper Grande Ronde, Tucannon, Asotin, Lower Clearwater, South Fork Clearwater, Lochsa, Lemhi, Yankee Fork, and Upper Salmon) to identify at the stream network-scale riparian, floodplain, in-stream, and adjacent land conditions for restoration and conservation purposes.

4.3.1.6. Stream temperature

Riparian vegetative cover has a primary and direct influence on one of the most important environmental factors associated with the salmonids – stream temperature. Stream water temperature is widely recognized as one of the most important environmental factors naturally influencing the distribution, growth, and survival of salmonids and other aquatic organisms (McCullough 1999, Justice et al. 2017). Stream temperatures directly affect the physiological processes of salmonids (Whitney et al. 2016, Martin et al. 2017, Bowerman et al. 2018), as well as their migration and spawn timing windows (Berman and Quinn 1991, Lisi et al. 2013). Salmonids will seek out cold-water refuges (Torgersen et al. 1999, Brewitt et al. 2017) as well as mobilize into relatively warmer water areas for metabolic and assimilative capacity purposes (Armstrong et al. 2013, Brewitt et al. 2017). Stream water temperature regimes can be modified by land use activities including but not limited to water diversions, reduced or non-existent riparian vegetation, excessive livestock grazing, and the simplification of streams due to channelization, levees, mining, and road construction (Justice et al. 2017). These activities can lead to increases in stream temperatures due to decreased streamflow, loss of sub-surface streamflow and hyporheic exchange, and increased solar radiation reaching a stream (Poole and Berman 2001).

Understanding how stream temperature responds to the degradation and potential restoration of riparian vegetation and channel morphology is an important component to prioritizing freshwater salmon restoration actions (Justice et al. 2017). Where and how stream temperatures can be restored is thus directly linked to the preceding list of cause and effects that have occurred throughout a watershed. The question then becomes where and how much has stream temperature been altered and is there anything that could be implemented to restore stream temperatures? Once again a comparison between historic and/or potential vs. current conditions becomes an analysis that can be used to determine these potential restoration locations as well as actions.

To evaluate the current, and to a certain extent, historic condition of stream temperatures throughout a watershed requires several types of data and information. Stream water temperature data measured using stationary thermographs (i.e., stream temperature loggers) in representative areas throughout a watershed is the foundation of any type of stream temperature analysis. Temperature loggers that collect data over time, coupled with a spatially robust sampling scheme can utilize spatial stream network (SSN) models to interpolate and expand measured stream temperatures throughout the entire stream network (Isaak et al. 2014). The coupling of stationary thermograph data and SSN models have allowed for a variety of analyses to be implemented across the Pacific Northwest evaluating spatial patterns and differences in river systems (Fullerton et al. 2018), fish utilization and density patterns (Isaak et al. 2016), and climate change scenarios (Isaak et al. 2017). Remotely sensed data such as forward-looking infrared imagery (FLIR) can capture a snapshot of peak daily water temperatures throughout the stream network and used to understand spatial differences at the watershed-scale and how that may impact utilization, movement, and growth of salmonids (Fullerton et al. 2017). In short, the recent onset of stream temperature data and associated tools can identify the thermal landscape that describe ecological and management concerns and that are linked to human actions in a manner that establishes a baseline of information at the watershed scale (Steel et al. 2017).

Riparian vegetative cover, stream channel morphology, hydrology, and climatic conditions are some of the most important parameters to quantitatively measure or estimate in order to determine the potential change from historic to current stream temperature conditions. LiDAR and/or historic aerial photographs are key tools that can be used to assess historic/potential vs. current stream temperature conditions.

4.3.1.7. In-stream channel complexity

A second component linked to riparian zones and floodplains is in-stream channel complexity. In-stream channel complexity can be defined as obstructions associated with a stream channel such as individual pieces of wood, sediment substrate larger than the average diameter (i.e., boulders associated with a gravel streambed), and naturally accumulated (i.e., log jams) or naturally built (i.e., beaver dams) obstructions that alter the topography of the stream channel and adjacent landforms. In doing so these structures create habitat complexity which is the expansion of the distribution of velocities, depths, habitat types, stream temperature, and food resources (Hicks et al. 1991, Bunn and Arthington 2002).

Much has been published with regards to the natural functions of wood and other obstructions and the numerous ecosystem benefits associated with wood in particular (Montgomery et al. 2003, Roni et al. 2014). These functions include but are not limited to; sediment storage and conversion of streambed to alluvial channels, an increase in the amount and quality of spawnable salmonid gravels, the creation of slower water environments such as deep pools, cover for feeding and refuge, the creation and maintenance of floodplain habitats, and increased water residence time and nutrient retention (Roni et al. 2014). This has been known for quite some time and as a result, there have been many, many efforts over the years to incorporate additions of wood, structures, and other obstructions into stream and watershed restoration (Roni et al. 2002, Roni et al. 2014). These efforts have been well documented and have consistently led to increases in slow-water habitat area, pool frequency, and pool depth, habitat complexity, and in several cases increases in the density and abundance of juvenile salmonids (Pess et al. 2012, Polivka et al. 2014, Roni et al. 2014).

As with the other aspects of stream ecosystems, understanding where and how in-stream complexity restoration can occur correlates to the preceding list of cause and effects that have occurred throughout a watershed. The question then becomes where and how much have obstruction been altered and is there anything that could be implemented to restore stream channel complexity? Again, a comparison between historic and/or potential vs. current conditions becomes an analysis that can be used to determine these potential restoration locations as well as actions (Sedell and Luchessa 1982, Sedell and Froggatt 1984).

The digital "reconstruction" of historic riverine landscapes, and associated floodplains, riparian, and in-channel structures associated is a tool that helps us understand watershed restoration opportunities and constraints at multiple spatio-temporal scales (Collins et al. 2003, Collins et al. 2012). In addition, these historic reconstructions can and have been put into a geomorphic context, which allow for the identification of restoration opportunities that include land use history, physical dynamics, and geologic settings to be considered (Collins et al. 2003, Collins and Montgomery 2011, Beechie and Imaki 2014).

The basic approach used to reconstruct what occurred in terms of in-stream channel complexity has been a combination of archival studies, field investigations, and remote sensing or digital analyses; Collins et al. (2002, 2003, 2012) and Collins and Montgomery (2011) have outlined an approach that combines archival and field studies methods to develop quantitative historic information on the change in wood abundance and characteristics of stream channels and valley bottoms across most of the major river systems entering Puget Sound. They found that wood jams were an integral component in creating and maintaining a dynamic, anastomosing river patterns with numerous floodplain channels and abundant edge habitat and routed floodwaters and sediment onto floodplains (Collins et al. 2002).

Such an approach is useful to develop multiple hypotheses associated with restoration efforts in the Columbia River Basin. For example, Woelfle-Erskine et al. (2012) used a similar approach for the Clark Fork River in Montana to help identify predictive metrics for channel patterns. Their results suggested a geomorphic setting that was transitional between braided and meandering, and thus the creation of a single-thread meandering channel, which incorporates structures to limit erosion and channel movement, was inconsistent

with the historical range of variability associated with a particular restoration location (Woelfle-Erskine et al. 2012). They suggested multiple working hypotheses to provide a means to incorporate uncertainty in order to maximize the potential for site-specific restoration success (Woelfle-Erskine et al. 2012).

Another approach to gain a better understanding of restoration opportunities and the identification of restoration targets is the use of reference conditions to determine how much and potentially where restoration can occur with regards to in-stream channel complexity. Fox and Bolton (2007) developed regional and geomorphic reference quantities for in-stream wood in unmanaged forested basins throughout Washington State for a large range of stream channels (gradients ranging between 0.1% and 74% and bankfull widths between 1 and 100 meters) and physiographic regions. Fox and Bolton (2007) argue that these levels are useful as restoration targets for basin-scale wood loads to reestablish the central tendencies and functions associated with such obstructions.

4.3.1.8. Fine sediment levels

Increased levels of sedimentation, particularly fine sediment levels less than 1-2mm in diameter, which deposit in stream channels due to a variety of natural and anthropogenic disturbances can have deleterious effects on one of the most important life stages to salmonids – the egg-to-fry life stage (Everest et al. 1987). Fine sediment infiltration into salmon redds and their associated egg pockets can reduce intergravel flow, dissolved oxygen, and lead to the suffocation of eggs in the gravels (Chapman 1988; Greig et al. 2005). Fine sediment can also fill the interstices of gravels and prevent fry from emerging from the gravel (Chapman 1988). Increased fine sediment levels in the streambed can increase in frequency and magnitude of scour through the overall reduction in streambed size from a large amount of fine sediment, thereby reducing survival (DeVries 1997, 2008). This is in addition to biological factors such as parental fitness or fitness of their gametes, which in some years can have a strong effect on survival of eggs and embryos (Johnson et al. 2012).

The egg-to-fry or parr life stage and the survival associated with this life stage, particularly for Chinook salmon, is critical to understand with respect to salmonid limiting factors and their change in population abundance over time (Honea et al. 2009, Roni et al. 2015). Life cycle models report egg-to-fry or early life stage survival as one of the main factors limiting population recovery (Kareiva et al. 2000, Honea et al. 2009). Roni et al. (2015) has reported that "little data exist on Chinook (salmon) egg-to-fry survival in the natural environment (Healey 1991, Bradford 1995; but see Merz and Setka 2004), and population models rely on combining survival from egg deposition to smolt outmigration or using relationships developed in the laboratory between fine sediment and egg-to-fry survival (Kareiva et al. 2000, Scheuerell et al. 2006, Honea et al. 2009, Jensen et al. 2009)."

There have been recent efforts to develop meta-analyses to bring together information on egg-to-fry survival associated with fine sediment as well as numerous other variables (Jensen et al. 2009). There has also been larger scale efforts, particularly in the Columbia River basin, to provide more field studies on egg-to-fry survival for Chinook salmon and other salmonids that are thoughtful in their methods (Johnson et al. 2012), spatially

extensive across a particular watershed and multiple watersheds (Roni et al. 2015), and include multiple years of data (Roni et al. 2015). Roni et al. (2015) directly measured egg-to-fry survival throughout the Yakima River Basin over a four-year period and found mean egg-to-fry survival ranged from 49% to 69%. The primary variables to explain differences in survival by reach and year across the Yakima included parentage and between reach variation (Roni et al. 2015). Parentage was most significant in years when high flow events were minimal (Johnson et al. 2012, Roni et al. 2015).

4.3.1.9. Water diversions

Water diversions and their impacts on salmonid habitat capacity and survival are important factors to consider with regards to stream and watershed restoration. Water diversions can reduce habitat capacity, survival, and have a deleterious effect on returning adult salmonids (Arthaud et al. 2010, Walters et al. 2013). Specifically, increases in the magnitude and duration of summer low flows can result in reductions in over summer survival of salmonids (Grantham et al. 2012). Walters et al (2013) projected juvenile Chinook salmon survival reductions of over 50% due to water diversions in the Lemhi River basin. Arthaud et al. (2010) found that juvenile rearing flows have a subsequent effect on returning adults in watersheds were water diversions have pronounced effects. In addition, water diversions, coupled with projected climate change scenarios can have negative additive effects with respect both juvenile salmonid capacity and survival (Walters et al. 2013).

One of the primary potential mechanisms for the reduction in oversummer survival of juvenile salmonids is the disconnection of portions of a stream and isolated pools (Obedzinski et al. 2018). Increases in the number of days where disconnection occurs can result in a decrease in the overall survival for a season (Obedzinski et al. 2018). Changes in survivorship at one life stage/season can have even more dramatic impacts during drought years. For example, Notch (2017) found that total outmigration survival from tagging as juveniles to outmigration to the ocean during drought years in the Sacramento River where 0.3%, the lowest ever recorded for outmigration Chinook salmon to the Pacific Ocean. The majority of the mortality was associated with water diversion in a tributary to the Sacramento River, Mill Creek, which was exacerbated by the drought conditions (Notch 2017). Other life stages can also be impacted from water diversions. Simpson (2018) found that a disproportionately large number of adult steelhead were entrained, relative to adult Spring Chinook salmon in the Umatilla River, particular outmigrating from the system as post-spawning adults, or kelts. According to Simpson (2018) this was due to their propensity for entrainment and their difficulties escaping through the water intakes of irrigation canals. Thus, keeping connection within a stream or between tributaries and a main stem has been implemented in the Columbia River Basin and has resulted in positive trends for salmonids (McCoy et al. 2018).

One method that has been used is a physical search to locate water diversion via fieldwork (Herren and Kawasaki 2001). A survey of waterways, combined with a GPS receiver to identify locations of water diversion, combined with a physical description, and photographs is an effective method to accurately identify water diversions (Herren and Kawasaki 2001). Attributes to either be measured or noted can include the type of diversion, intake

size, type of discharge, bank location, screen type (if and when present), river system or waterway, and likely primary use of the diverted water (Herren and Kawasaki 2001). Determination of ownership can also be attempted through personal communication with the owners themselves (Herren and Kawasaki 2001). Herren and Kawasaki (2001) found over 3,300 diversions in the Sacramento–San Joaquin Delta (Delta) and the Suisun Marsh of the San Francisco Bay–Delta area, over 98% of which were unscreened to prevent fish entrainment.

Staff at NWFSC have been examining the potential of impacts of water withdrawal in the Interior Columbia River Basins. Holzer (D. Holzer, unpublished data) identified potential water rights relative to the estimated

Figure 4.2. Potential water rights relative to the estimated 1 Aug streamflow conditions. The light-green areas are near-zero difference between the natural flow regime and water withdrawal, while dark-green and blue areas are trending more streamflow than water rights.

1 Aug streamflow conditions. For example, in Figure 4.2, the light green area indicates where the natural flow rates are of equal magnitude to the existing in-stream water rights, while darker green to blue areas is a gradient of increasing streamflow relative to in-stream water rights. This map illustrates an effective way to document areas where restorative actions with regards to water diversions would be important at a larger spatial extent.

4.4.   Estimating restoration effects on salmon habitat capacity or survival

Once restoration scenarios are identified, each habitat change must be translated into a change in a life-stage capacity or survival in the life cycle model. In general, habitat quantity or area changes tend to affect habitat capacity, whereas habitat quality change tend to affect life-stage survival. The functional relationships between a habitat change and the change in capacity or survival are typically developed from literature values or from local empirical relationships. For example, numerous studies of fine sediment effects on egg-to-fry survival show that egg-to-fry survival decreases with increasing fine sediment, and both general and species-specific equations can be developed to translate changes in fine sediment in a change in survival (Jensen et al. 2009). On the other hand, local data may indicate that smolt production of a species is related to a measured stream parameter such as summer stream flow, and the statistical relationship between stream flow and survival may be used to quantify rearing survival in a life cycle model.

4.4.1. Translating habitat quantity into habitat capacity estimates

Spawning capacity estimates occur two main ways. First, spawning gravel area can be translated into redd capacity by dividing spawning gravel area by the average redd area of spawners for a species. Egg capacity is then the number of redds multiplied by fecundity for the species, and by number of females per redd if it is a number other than 1. Second, spawning capacity estimates can be a function of changes in wood abundance in smaller streams. Redds per km have been quantified by channel type and land-cover class, and egg capacity is number of redds multiplied by fecundity for the species, and by number of females per redd if it is a number other than 1.

Rearing capacity for any life stage is typically estimated by summing all habitat areas of each habitat type and then multiplying the total area of each habitat type by type-specific fish density. Thus, the production potential of a habitat for each life stage (e.g., summer rearing, winter rearing) can be expressed mathematically as

$$c = \sum_{i=1}^{n} \sum_{j=1}^{n} A_{ij} \cdot d_i$$

where A_{ij} is the sum of areas of all habitat units (j =1 through n) of type i, and d_i is the density of fish in habitat type i. See Table 4.3 for examples of habitat types and their densities.

Table 4.3. Examples of rearing densities of rearing juvenile salmonids used to estimate rearing capacities. Chinook salmon densities are from Beamer and Henderson (1998) and Jamie Thompson (WDFW, unpublished data). Coho salmon densities are from Reeves et al. (1989) for pool, riffle, and pond values, Beechie et al. (1994) for lake values, Beamer and Henderson (1998) for large river values, and Henning et al. (2004) for marsh values. Steelhead densities are from Johnson (1993) for pool, riffle, and pond values, and Beamer and Henderson (1998) and Thompson (unpublished) for large river values.

	Fish density (fish/m²)				
Habitat type	Chinook subyearling	Coho (summer)	Coho (winter)	Steelhead (summer)	Steelhead (winter)
Small stream (bfw <20 m)					
Pool	0.09	1.70	3.50	0.63	0.14
Riffle	0.0057	0.68	0	0.46	0.10
Pond	0.09	1.30	3.75	0	0.03
Large river (bfw >20 m)					
Backwater Pool	0.97	0.79	0.06	0.20	0
Natural Bank	0.68	0.41	4.53	0.30	0.10
Hydromodified Bank	0.27	0.20	0.48	0.10	0.10
Bar–Boulder	0.05	0	0	0.26	0.20
Bar–Gravel	0.15	0.13	0.0014	0.21	0.07
Bar–Sand	0.04	0.07	0	0.03	0
Floodplain					
Slough	0.12	1.28	2.50	0	0
Pond	0	1.50	3.75	0.10	0
Marsh	0	0	0.32	0	0
Side Channel	0.04	1.28	1.28	0	0

4.4.2. Translating habitat quality into survival estimates

In general, habitat quality attributes tend to affect growth potential, and thus survival more than capacity, although to some extent it can be argued that they affect both (Chapman 1966). Examples of habitat attributes that affect survival are fine sediment in spawning gravels, stream temperature, and water quality parameters (e.g., dissolved oxygen). One challenge with estimating survival parameters is that the life cycle model requires a single value for a population, but habitat attributes vary by reach and cannot be summed to the basin scale. There are at least three ways to handle this problem. First, reach-level survivals can be averaged across all reaches in a population, and weighted by habitat type if necessary. For example, in coho winter rearing habitats, survival values can be calculated as the weighted average survival across all reaches or habitats in a subbasin based on the proportion of rearing capacity in each habitat area:

$$s_3 = (f_{ss} \times s_{ss}) + (f_{lr} \times s_{lr}) + (f_p \times s_p)$$

where:
- s_3 is the weighted average winter rearing survival for that spatial unit,
- f_{ss} is the proportion of rearing capacity in small streams, and s_{ss} is the base survival value applied to that proportion of the spatial unit,
- f_{lr} is the proportion of rearing capacity in large rivers, and s_{lr} is the base survival value applied to that proportion of the spatial unit, and
- f_p is the proportion of rearing capacity in ponds and marshes, and s_p is the base survival value applied to that proportion of the spatial unit.

The weighted average winter rearing survival can change among scenarios in two primary ways: a change in the proportions of fish rearing in each habitat type (a function of changes in habitat area or type-specific density), or a change in the survival of fish within a habitat type. Second, survival can change as a function of a habitat quality change, such as a change in fine sediment in spawning gravels. Modeled reach level estimates of fine sediment proportion in spawning gravels can be translated to reach level estimates of egg to fry survival (e.g., using equations from Jensen et al. 2009), and then averaged across all reaches in a sub-basin. Finally, survivals estimates can be scaled with a change in sub-basin or population-scale capacity (e.g., increased wood in large river habitats, Bouwes et al. 2016).

4.5. Estimating population-level outcomes of restoration alternatives

Restoration alternatives can be evaluated by developing a set of restoration scenarios, with each scenario representing either single or multiple restoration action types, and specified locations for each action type. For example, a restoration scenario may represent removal of passage barriers in specific locations, which reconnects spawning and rearing habitats above each barrier and increases spawning and rearing capacity for that scenario. For a more complex restoration scenario, the combined effects of multiple actions such as barrier removal, riparian planting, and floodplain habitat connection can be evaluated simultaneously. Each restoration action type influences specific life-stage capacities or

survivals, and in most cases capacities and survivals of multiple life stages are increased. Locations may be reach-specific where there are data to do so, or they may be generalized to sub-basins or sub-populations. The life cycle model then estimates the combined effects of the multiple life stage improvements on a salmon population.

Beechie et al. (2015) used geomorphic and biological analyses to estimate restoration potential under three alternative scenarios for a 64-km section of the Trinity River, California, between the North Fork Trinity River and Lewiston Dam, which is the focus of habitat rehabilitation efforts under the Trinity River Restoration Program. They found that the potential increase in *Oncorhynchus tshawytscha* (Chinook salmon) and *O. mykiss* (steelhead) fry rearing capacity ranged between a low of 62% and a high of 112% depending upon the species and scenario (Beechie et al. 2015).

A more direct life cycle model example from the Columbia River Basin focused on developing restoration scenarios for steelhead in the John Day River Basin (McHugh et al. 2017). McHugh et al. (2017) used a modelling framework that linked reach-scale stream habitat models with a basin-scale LCM, bridged by statistical extrapolation models, to evaluate recovery opportunities for steelhead in the Middle Fork John Day River, USA. They used a LCM to quantify population performance under current conditions and under two classes of restoration that aim to increase survival for juvenile steelhead: riparian revegetation, which reduces (otherwise limiting) stream temperatures during the warm summer months; and woody structure addition, which increases in-stream hydraulic complexity and thus juvenile rearing capacity (McHugh et al. 2017).

Their evaluation of the restoration scenarios revealed that while both strategies have the potential to improve the conservation status of steelhead, the benefits of woody structure addition were relatively minor compared to those resulting from stream temperature reductions (McHugh et al. 2017). They suggested that in thermally stressed systems the benefits of wood addition will be optimized if structures are added at a considerably higher rate than is often done, focusing on reaches that are not thermally limited (McHugh et al. 2017). In addition, if these efforts would address thermal limitations and offer long-term sources of future wood recruitment if coupled with extensive riparian planting in stream reaches that have the highest potential for effective shading (McHugh et al. 2017). Both examples area data-rich restoration scenario examples even though they each utilized extrapolation techniques.

5. Estimating population-level outcomes of restoration alternatives in data-rich watersheds: An example from the Grande Ronde basin focusing on spring Chinook salmon populations

The Grande Ronde River Basin included six historical populations of Spring Chinook Salmon (Figure 5.1). Since the early 1990s, the Oregon Department of Fish and Wildlife (ODFW) has conducted annual studies of juvenile Chinook salmon production in four of these populations (Upper Grande Ronde, Catherine Creek, the Minam River, and the Lostine River). These four Spring Chinook salmon populations represent a range of habitat conditions. The Minam River is relatively pristine basin, although there were historical mining impacts in some parts of the drainage. The upper sections of the Lostine River are also relatively intact; however, the lower sections are impacted by water withdrawals and other land use activities. Both Catherine Creek and the Upper Grande Ronde watersheds have been extensively modified by land use including timber harvest, overgrazing, beaver trapping, and mining. In addition, low gradient reaches in the Grande Ronde Valley that likely supported a diversity of juvenile Chinook salmon habitats and associated juvenile rearing patterns were extensively converted to agricultural use beginning in the mid- to late 19th century.

Figure 5.1. From Anderson et al. (2011). Location of fish traps in the Grande Ronde River subbasin during the study period. Shaded areas delineate spring Chinook salmon spawning and upper rearing areas in each study stream. Dashed lines indicate Grande Valley and Wallowa Valley, major agricultural areas in these watersheds.

The Grande Ronde is a basin with a rich set of demographic data for Chinook salmon. Redd counts have been made throughout much of the available spawning habitat for over 60 years (Tranquili et al. 2004). Similarly, there are 23 years of fall and spring juvenile emigrant estimates from screw traps on major tributaries. In addition, several years of mid-summer in-stream tagging with passive integrated transponders have led to size and survival estimates of multiple life stages from the Grande Ronde River tributaries to Lower Granite dam on the Snake River. These data have been used in a state–space model to estimate juvenile rearing capacity.

5.1.   Overview/Summary

The four Grande Ronde Spring Chinook salmon population LCMs are framed in the matrix life cycle modeling format originally described in Zabel et al. (2006). We used information generated from the spawner to smolt life-stage monitoring as the basis for incorporating detailed juvenile life stage survival and density-dependent relationships into the freshwater juvenile stages of full life cycle models for each of the populations. Life cycle models were developed based on long-term data series including three main components: estimation of annual spawning escapements (mid-1950s to present); presmolt emigration (1992–2016 migration years) estimates of late summer parr densities at sample sites within each population; and PIT tag-based survival rates to Lower Granite Dam for summer parr, fall downstream migrants, winter parr, and spring downstream migrants (e.g., Jonasson et al. 2017).

For each population, we estimated the total amount of rearing habitat in reaches designated as current use by ODFW above and below the location of the juvenile outmigrant traps. We used the results from a systematic survey of pools, fast water and run habitat units in Grande Ronde basin tributaries in combination with parr density estimates for each habitat category to generate standardized habitat estimates of the total amount of habitat above and below the juvenile sampling weirs for each population.

The basic approach for incorporating habitat change effects starts with current life stage capacities and survival estimates derived from the >20-year juvenile series for each population. Using the results of ODFW Aquatic Inventory surveys in each population, we calculate the total amount of pool equivalent habitat currently supporting spawning and/ or rearing. Other than scaling the expression of juvenile life stage parameters to the total amount of pool equivalent habitat within a population, our Grande Ronde MLCMs do not directly include habitat parameters. We use multipliers on life stage specific survival and capacity terms as inputs to model the impact of habitat actions or environmental changes.

We analyzed a range of habitat restoration scenarios starting with maintaining baseline conditions and adding: the 2009–16 actions; minimum 2018–21 actions; current projections for proposed 2019–24 actions: implementation of 20-year habitat restoration scenarios including Recovery Plan actions plus riparian restoration in high and moderate priority reaches identified in Justice et al. (2017). At this point, the last three habitat scenarios have been run only for Catherine Creek and the Upper Grande Ronde populations. For Catherine Creek and the Upper Grande Ronde populations we added another scenario to simulate the potential of additional habitat restoration downstream of current use. For that scenario we assume that the current area production has been extended downstream sufficiently after 25 years.

The habitat actions were analyzed in combination with two variations on future hydropower operations: continuation of current operations under the FCRPS 2014 (NMFS 2014) guidelines, and implementation of the proposed 2018 spill program assuming a 120% gas cap. We ran the gas cap spill scenario under four different assumptions bracketing a range of potential impacts on subsequent ocean-stage mortality (no effect, or a 10%, 25%, or 50% improvement in ocean-stage survival for in-river migrants subject to increased spill). The scenario analyses also incorporated the current sliding-scale harvest schedule for Snake River spring/summer Chinook and projected impacts of increased marine mammal predation.

Modeling the addition of the 2009–16 habitat actions reduced extinction risks for the Catherine Creek and Upper Grande Ronde populations relative to updated baseline habitat projections. Incorporating supplementation into the model runs resulted in reduction in the risks of gong below the 24-year quasi-extinction thresholds for both the Catherine Creek and Upper Grande Ronde River populations. The largest decreases in projected risks were for habitat actions in combination with hydrosystem spill operations incorporating reductions in ocean latent mortality. The projected 24-year QET risk across model runs dropped to 0.4–2.4% (QET30) and 3.2–24.4% (QET-50) for Catherine Creek. 24-year QET risks remained high in this scenario for Upper Grande Ronde, while the risk of going below QET50 remained relatively unchanged, the risks of going below QET30 dropped further to range from 12.4–71.3%. across 500 runs. The largest increase in short-term abundance (+16%) from the 2014 Biological Opinion (BioOp) tributary habitat actions was projected for the Catherine Creek population, where the actions were directed at expanding summer rearing habitat, identified as a key limiting life stage.

Expressed as proportional changes from baseline conditions, the Catherine Creek recovery plan short and intermediate response actions would result in an 84% gain in parr habitat capacity by year 24. This increase includes the projected benefits of the 2019–24 in-stream actions described above. The initial responses to riparian restoration would increase that gain to a projected 125% improvement in parr rearing capacity by year 24. Benefits from increasing shading and restoration of natural stream channel characteristics would continue to accrue over time, reaching 165% over baseline conditions 48 years out. The benefits projected for the shading corresponding to fully mature riparian tree heights at approximately 100 years out would increase to approximately 206% of baseline. The Upper Grande Ronde River has a greater amount of current production habitat subject to high summer stream temperatures. As a result, riparian restoration actions have a higher proportional impact than for Catherine Creek. The projected increases in parr production potential from implementing the tributary habitat improvements from the Upper Grande Ronde 20-year restoration scenario at 24 and 48 years would be +99% and +140% respectively. Adding in the potential increase in survival gained by successfully addressing the high Grande Ronde Valley outmigration mortality would project to increase the cumulative improvements at 24 and 48 years to 199% and 262%.

We generated additional long-term scenarios to illustrate the potential for further expansion of natural production into reaches below current spawning and rearing that are currently precluded by loss of historical rearing habitat and extremely high summer temperatures (Upper Grande Ronde) along with reduced summer flows (Catherine Creek).

In both cases restoring production to these lower reaches would almost certainly require successful restoration of the upstream reaches targeted in the 20-year scenario in order to extend spawning downstream enough to generate juveniles to use newly restored habitat below current spawning/rearing range.

Under the long-term restoration scenarios, both populations showed large proportional increases in projected natural-origin spawner abundance. For the Upper Grande Ronde population, the cumulative impact of the long-term habitat scenario combining expansion into reaches downstream of Fly Creek, reduced Grande Ronde Valley migration mortality and returning Lower Columbia marine mammal mortalities to pre-2013 averages resulted in a 525% projected increase. The corresponding scenario for Catherine Creek resulted in a median proportional improvement of 527%. However, in absolute terms, the projected abundance for Catherine Creek showed the highest increase relative to Interior Columbia Technical Recovery Team minimum abundance thresholds. More than 50% of 500 simulation runs for the long-term habitat plus Grande Ronde Valley survival improvements scenario for that population exceeded the minimum abundance threshold under the 25% and 50% latent mortality reduction assumptions. Adding reductions to current lower Columbia River predation mortalities, presumably by decreased marine mammal predation, resulted in greater than 70% of simulation runs exceeding the abundance threshold under all spill latent mortality assumptions modeled.

5.2.   Grande Ronde LCM structure

Our four Grande Ronde Spring Chinook salmon population LCMs are framed in the matrix life cycle modeling format originally described in Zabel et al. (2006). Detailed LCMs for several Salmon River basin populations (Crozier et al. 2016) and the Wenatchee River (Jorgensen et al. 2017) use the same basic framework, although each set is adapted to use the different levels of information available to populate freshwater life stages. We expanded the tributary habitat life stage components using the detailed information on juvenile life stages for each of the Grande Ronde populations (Figure 5.2). We also replaced the fixed harvest rate feature of the 2007 model with an abundance-driven functional relationship mimicking current harvest management practices. The matrix has the form:

$$
A(t) = \begin{bmatrix}
0 & 0 & 0 & b_4 \cdot s_A \cdot F_4(t) & s_A \cdot F_5(t) \\
s_2(t) & 0 & 0 & 0 & 0 \\
0 & s_3(t) & 0 & 0 & 0 \\
0 & 0 & (1-b_3) \cdot s_o(t) & 0 & 0 \\
0 & 0 & 0 & (1-b_4) \cdot s_o(t) & 0
\end{bmatrix}
$$

The s terms represent the survivals between life stages, the b_t and $F(t)$ terms represent the rates of maturity at age(t) and relative female fecundity by age. In our Grande Ronde models, the term $s_2(t)$ is a composite representing the production of smolts as a function of parent spawners and the downstream survival of those smolts to entry in the estuary. It includes both density-dependent components (summer parr per spawner, spring outmigrants per parr) and density-independent elements (spring outmigrant to Lower
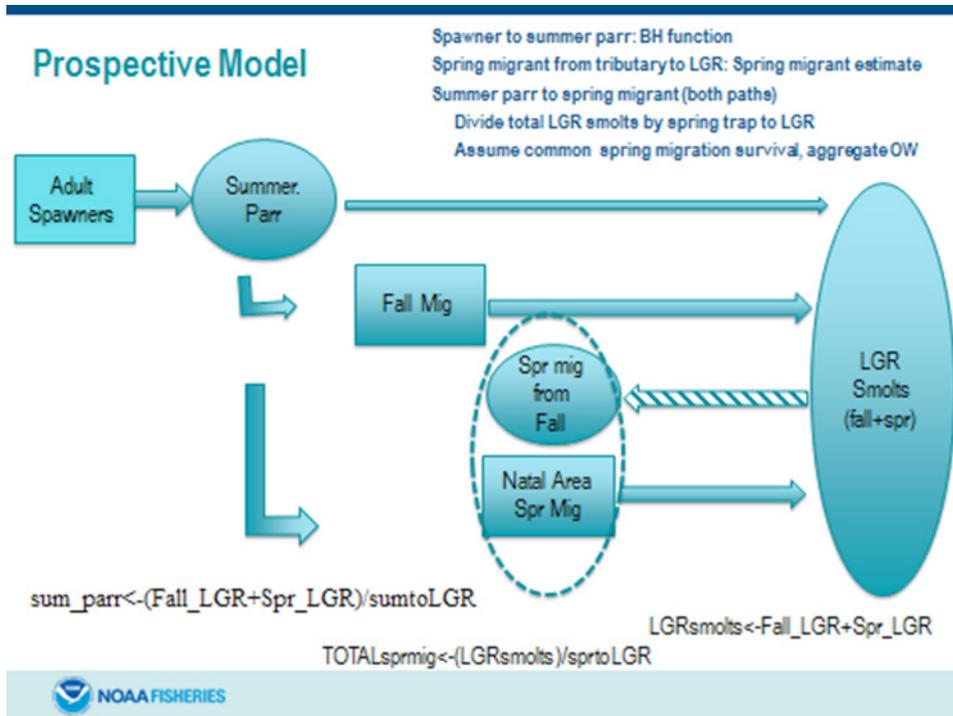
Figure 5.2. Tributary life history stage survivals and abundance estimates used to estimate current baseline model parameters.

Granite Dam smolt, smolt to below Bonneville Dam). The spawner to Lower Granite smolt elements within this stage are directly linked to tributary habitat conditions as described in detail below. Survival through the mainstem Snake and Columbia Rivers are estimated based on PIT tag data representative of the aggregate natural-origin Snake River spring-summer Chinook run (Crozier 2019). The $s_3(t)$ term represents estuarine/early ocean survival through age-3. s_A represents adult migration mortalities from arrival at the Columbia River mouth to the spawning grounds. It includes estimated marine mammal predation in the Lower Columbia River, mainstem Columbia River harvest, upstream passage mortalities and prespawn mortality above Lower Granite Dam.

A detailed description of the freshwater tributary life stage elements of the models follows (Table 5.1). Descriptions of the remaining components are available in Cooney et al. 2017. Briefly, the models incorporate estimated survivals derived from data on annual aggregate Snake River spring Chinook salmon production in subsequent life history stages—downstream migration to the estuary, estuary/ocean, Columbia River entry and upstream migration (Crozier et al. 2017, ISAB 2017). Snake River spring/summer Chinook are subject to in-river harvest that is managed according to a sliding scale (WDFW 2017). We incorporated the sliding scale with estimates of management uncertainty derived from 1995–2014 post-season run reconstructions. Three of the four Grande Ronde populations have active local broodstock supplementation programs. Broodstocking for each of those programs is managed with population specific schedules. We include modules in the Grande Ronde population models that mimic the schedules and recent performances of the supplementation programs (including survivals to release and smolt-to-adult return rates).

Table 5.1. Grande Ronde River basin LCM input parameters: Summary.

Life stage	Function	Derivation	Parameter uncertainty	Variance
Spawner to parr	Beverton–Holt	R `nls` package	Bootstrap	Lognormal
Fall parr to spring migrant	Logistic on density	R `nls` package	Maximum likelihood	Lognormal
Spring migrant to Lower Granite (LG) Dam	Logistic on density	R `nls` package	Maximum likelihood	Lognormal
Juvenile Columbia River migration	Random draw, most recent 10 years	Annual system survival estimates		
Ocean: Year 1	Random start to fixed series with random error component	Multiple regression	Poor ocean conditions, recent ocean conditions, long-term ocean conditions	Lognormal
Ocean: Years 2–5	Constant	0.8	No	
Harvest	U.S. *v* Oregon sliding scale		Management error	Lognormal
Broodstocking	Catherine Creek Schedule	Hatchery Genetic Management Plan	Management error	Lognormal

The Grande Ronde models are calibrated to the 1993–2016 adult data series prior to being used in prospective simulations. We compare estimated adult brood-year returns for the 1993–2011 brood years with model generated estimates using the inputs described above. We include the year-specific estimates of upstream and downstream passage survivals and estimated brood-year ocean smolt to adult return rates (SARs). Observed brood-year returns have consistently been higher than modeled estimates for each population. We calculate a brood-year adjustment factor (the slope of a zero intercept regression between logit transformed estimated and observed SARs) and apply it in prospective analyses.

5.2.1. Estimating life stage capacities using population-specific fish and habitat data

The combination of longer-term estimates of fish data (adult and juvenile life stages) and habitat survey information at the population level allows us to address steps 1 and 2 in the generalized process simultaneously. Those data sets allowed for extrapolating annual estimates of summer parr abundance for each population. Parr production relationships were then generated for each population using the corresponding parent spawner abundance estimates. We also developed survival relationships for two additional juvenile life stages: summer parr to spring outmigrant and spring outmigrant to Lower Granite Dam.

We use the Northwest Stream Temperature (NorWeST) estimate database as a starting point for temperature indices for each population. NorWeST modeled annual temperatures are expressed as Aug averages for 1 km segments of the stream network. We compared NorWeST modeled temperature estimates to empirical data sets available for a subset of reaches in the populations (Isaak et al. 2016). Average NorWeST temperatures for those

locations were also highly correlated with empirically based estimates of maximum weekly maximum stream temperatures, and index that has been used in studies relating adult and juvenile Chinook densities and survival rates (e.g., Justice et al. 2017).

Stream flow data for the four populations were downloaded from the Oregon Water Resources Department.[2] Stations were Catherine Creek (13320300), Minam River (1332000), Upper Grande Ronde River (13317850) and Lostine River (1333000). Stream flow estimates were available for all years of the juvenile study for the Lostine River. There were gaps (one to three years duration) in the annual flow records for the other three populations. Annual stream flows in Grande Ronde tributaries generally peak in May or Jun and decrease to relatively low levels by early Aug. We calculated two indices of summer flow conditions for use in the statistical analyses of the population-specific stage survival relationships: Sep flows during the spawning and initial incubation stage and the average Aug and Sep flows one year after spawning, corresponding to the conditions encountered during the initial year of freshwater rearing. In each case, we compared annual fluctuations in the population-specific data series, dividing the individual-year estimates by the average flow for the series.

Juvenile spring/summer Chinook salmon prefer low gradient reaches with deep pools for summer rearing (e.g., Bjornn and Reiser 1992). In addition, adult spring/summer Chinook salmon redds are generally concentrated in gravels associated with pool habitats. For each population, we estimated the total amount of rearing habitat in reaches designated as current use by ODFW above and below the location of the juvenile outmigrant traps. We used the results from a systematic survey of pools, fast water and run habitat units in Grande Ronde basin tributaries in combination with parr density estimates for each habitat category to generate standardized habitat estimates of the total amount of habitat above and below the juvenile sampling weirs for each population. The estimates were calculated by summing the habitat above and below weirs by stream reach category (pool, riffle, and fastwater) and multiplying the sums by the average relative density for each of those habitat categories. Two of the four populations had potential AQI rearing habitat with summer MWMT stream temperatures above 18°C. We used a relationship between relative parr density and MWMT temperature reported in Justice et al. 2017 to discount the estimated AQI habitat in those reaches where temperatures exceeded 18°C. We also standardized juvenile abundance data for each population to a common unit of habitat (10,000 m^2 of AQI pool equivalent habitat) to explore general relationships between habitat conditions and juvenile production that might be common across one or more populations.

Parent spawner estimates were generated by ODFW for stream reaches upstream of the rotary screw trap sites in each population. Based on the ODFW survey results, we assumed negligible spawning below the juvenile screw trap. We developed production relationships for the reaches above the weir site standardized to a common unit of habitat (10,000 m^2 of equivalent pool area) using the habitat data sets described above. We compared summer parr per spawner ratios (per 10,000 m^2 AQI habitat) to flow and temperature indices representative of averages across spawning and summer rearing locations as well as against parent spawning densities. There were no significant trend relationships in the annual parr per spawner estimates for the environmental indices. However, the parr per spawner estimates did group at relatively distinct temperature levels for each population. There were significant relationships between spawner densities and parr densities for each population.

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[2] http://apps.wrd.state.or.us/apps/sw/hydro_report/

5.2.1.1. Spawner to summer parr stage

We fit linear and Beverton–Holt (BH) relationships to AQI standardized annual estimates of spawner escapement and summer parr production using the `nls` package in R. We assumed a lognormal error structure and weighted age-5 parent spawners by 1.26 (ICRT 2007) to account for higher fecundity of the age-5 females. The Beverton–Holt model, with its density-dependent term was a better fit to the data series for each population (AICc criteria):

$$\ln\left(\frac{Parr_{p,y+1}}{Spawners_{p,y}}\right) = a - \ln\left(abs\left(1 + \frac{e^a}{b}Spawners_{p,y}\right)\right)$$

where the spawner estimates are age-weighted using the following formula:

$$AWSpawners_{p,y} = \left(\left(1 - age5\,prop_{p,y}\right) + 1.26 \cdot age5\,prop_{p,y}\right) \cdot Spawners_{p,y}$$

We addressed parameter uncertainty in the fitted model parameters by generating a set of 1,000 replicate paired estimates of the Beverton–Holt a (natural log parr per spawner) and b (asymptotic parr capacity) using the `nlsboot` bootstrap estimation routine in R. The approach we used to estimate a production relationship for this stage assumed that the spawner estimates were measured without error. Future iterations of this model are under development; they will use a hierarchical framework that includes accounting for potential measurement error. Initial results indicate that the stage-specific relationships derived from that approach are similar.

5.2.1.2. Summer parr to spring tributary outmigrant stage

The combination of life-stage PIT-tag groups available for the four Grande Ronde populations represent a unique opportunity to evaluate survivals within the two predominant parr to oceanward migration pathways (natal area and downstream overwintering). We made a simplifying assumption, that annual early spring to Lower Granite Dam survival for the downstream overwintering components of each population was the same as the estimated survival to Lower Granite Dam for the natal overwintering group passing the smolt trap in the spring. This allowed us to estimate the total number of smolts leaving the tributary from both pathways. We considered framing juvenile life stages in more detail, using the estimates of fall migrant and winter natal area parr survival. Incorporating that level of detail requires making some assumptions about monthly mortality rates that are not directly informed by the available data for these systems. Summer parr estimates are generated based on sampling in Aug, fall downstream migrants passing the smolt traps generally peak in mid-Oct. Parr remaining above the smolt traps to overwinter pass downstream the following spring. The proportion of juveniles overwintering downstream of the trap varies across the four populations is not significantly related to annual variations in density or environmental indices. Survival from summer parr to either of these stages is not directly estimated. We calculate an aggregate overwintering mortality from summer parr to spring tributary outmigration by assuming that the estimated spring outmigrant to Lower Granite Dam survival applies to

the fish surviving overwintering below the weir site (the fall downstream migrants). That assumption is generally supported by patterns in survivals across tag groups in the Grande Ronde including survival estimates derived from winter tagging above the smolt traps after fall emigration. We are exploring alternative approaches to estimating pathway specific overwintering mortalities for future iterations of the Grande Ronde detailed LCMs.

We compared annual estimates of survival from summer parr to spring outmigrant against summer parr density, summer temperatures and relative flow levels after transforming the annual survival series for each population as logits. There was a significant negative relationship of the summer parr to spring presmolt survivals and summer parr abundance for each population. Summer maximum stream temperatures and flow levels were not significant in the analyses and were not included in generating the fitted estimates.

$$Logit\left(Sow_{p,yr}\right) = \frac{Sow_{p,yr}}{\left(1 - Sow_{p,yr}\right)}$$

$$Est\left[Logit\left(Sow_{p,yr}\right)\right] = A_{Sow,p} \cdot Parr_{p,yr} + B_{Sow,p} + \varepsilon_{0,sd}$$

$$Sow_{p,yr} = \frac{Logit\left(\widehat{Sow_{p,yr}}\right)}{\left(1 + Logit\left(\widehat{Sow_{p,yr}}\right)\right)}$$

5.2.1.3. Spring outmigrant to Lower Granite Dam stage

Population-specific estimates of survival for the spring outmigrant to Lower Granite Dam were also evaluated as logistic regressions on parr density. The density-dependent terms were not significant, the relationships incorporated into the life cycle were expressed as a constant multiplier with a randomly drawn error term reflecting the variability in each population series.

$$Logit\left(\widehat{S \lg r_{p,yr}}\right) = B_{S\lg r,p} + \varepsilon_{0,sd}$$

Survivals during the spring migration from the smolt traps to Lower Granite Dam are consistently lower for Catherine Creek and Upper Grande Ronde smolts in comparison to Lostine and Minam River spring migrants. In some years ODFW has also tagged spring outmigrating smolts at Elgin on the mainstem Grande Ronde River below the upper two populations and above Minam and Lostine Rivers. Survival rates to Lower Granite Dam from Elgin are comparable or higher than those estimated for smolts entering downstream from the Minam and Lostine Rivers, indicating that considerable mortality is being incurred in the upper Grande Ronde Valley during the spring outmigration.

5.2.1.4. Catherine Creek summer rearing downstream of trap

In recent years, parr sampling at Catherine Creek CHaMP sites below the weir and smolt trap determined that parr were rearing in the reach extending downstream to the Davis Dam irrigation diversions (e.g., Jonasson et al. 2017). As a result, we incorporated a second tributary habitat summer rearing area into the Catherine Creek model. Given the relatively

low rates of observed downstream passage from initial trap operations in the early spring to the fall, it is likely that these juveniles were produced from spawning upstream of the weir, likely migrating downstream as fry or after a short period of initial rearing. That early redistribution would be prior to the initiation of large-scale irrigation withdrawals that drastically reduce summer/fall flows in the reaches below the trap site. For the Catherine Creek model, we estimated the number of summer parr rearing below the weir site using the same combination of ODFW Aquatic Inventory data (reflecting the impacts of irrigation withdrawals) and CHaMP parr densities by reach type. We assume that the average proportion of parr production observed in the recent years (~30%) applied to the earlier study years before systematic sampling was initiated in the downstream reaches. ODFW has expanded their ongoing summer parr tagging program to include groups in the downstream area. Initial results indicate substantially lower survivals from late summer to detection at Lower Granite Dam the following spring.

5.3.   Developing restoration scenarios: Habitat change analyses

White et al. (2017) used contemporary estimates of channel width based on Oregon Department of Fish and Wildlife's Aquatic Inventories Project (AIP; Moore et al. 2008) to evaluate the impact of channel widening on the distribution of Chinook summer parr. The AIP survey is a rapid assessment of common fish habitat characteristics collected in a spatially continuous fashion across the stream network. AIP data from the 1990s were used to examine channel width as a proxy of stream channel width:depth ratio—a metric strongly tied to integrity of stream channels (e.g., Beschta and Platts 1986; Myers and Swanson 1996) and commonly used in fish–habitat models (Fausch et al. 1988)—because historical estimates of water depths were not available. Data for this analysis were limited to the low flow period to provide consistency in discharge over the years that would allow change in width to be a valid surrogate for change in width:depth ratio.

Historic channel width was estimated using information from GLO notes and then compared to current conditions to get an estimate of percentage change in channel width. A direct comparison for each location typically using this method cannot always be made, thus quantifying the magnitude of change in relation to the geomorphic valley setting is important. This is where it is important to understand the geomorphic setting utilizing various classification schemes. Streams were classified into small and large using an 8-m bankfull width threshold (Beechie and Imaki 2014), and then further divided based on valley confinement (laterally unconfined, partly confined, and confined) following the methodology described in the River Styles Framework (Brierley and Fryirs 2005). This resulted in three classes: large streams (LS), small/partly confined and confined streams (SC), and small/laterally unconfined streams (SU). One-way ANOVA was used to test the effect of valley setting on magnitude of channel change in impacted watersheds.

One key finding of White et al. (2017) was that these streams have yet to recover from severe anthropogenic disturbance such as cattle grazing, logging, and mining (Figure 5.3). This channel widening analysis was then coupled with other factors such as examination of stream temperature to examine how changes in one variable affects changes in another variable that could lead to alterations in fish utilization—both positive and negative. A mechanistic water temperature model demonstrated that channel widening resulted in
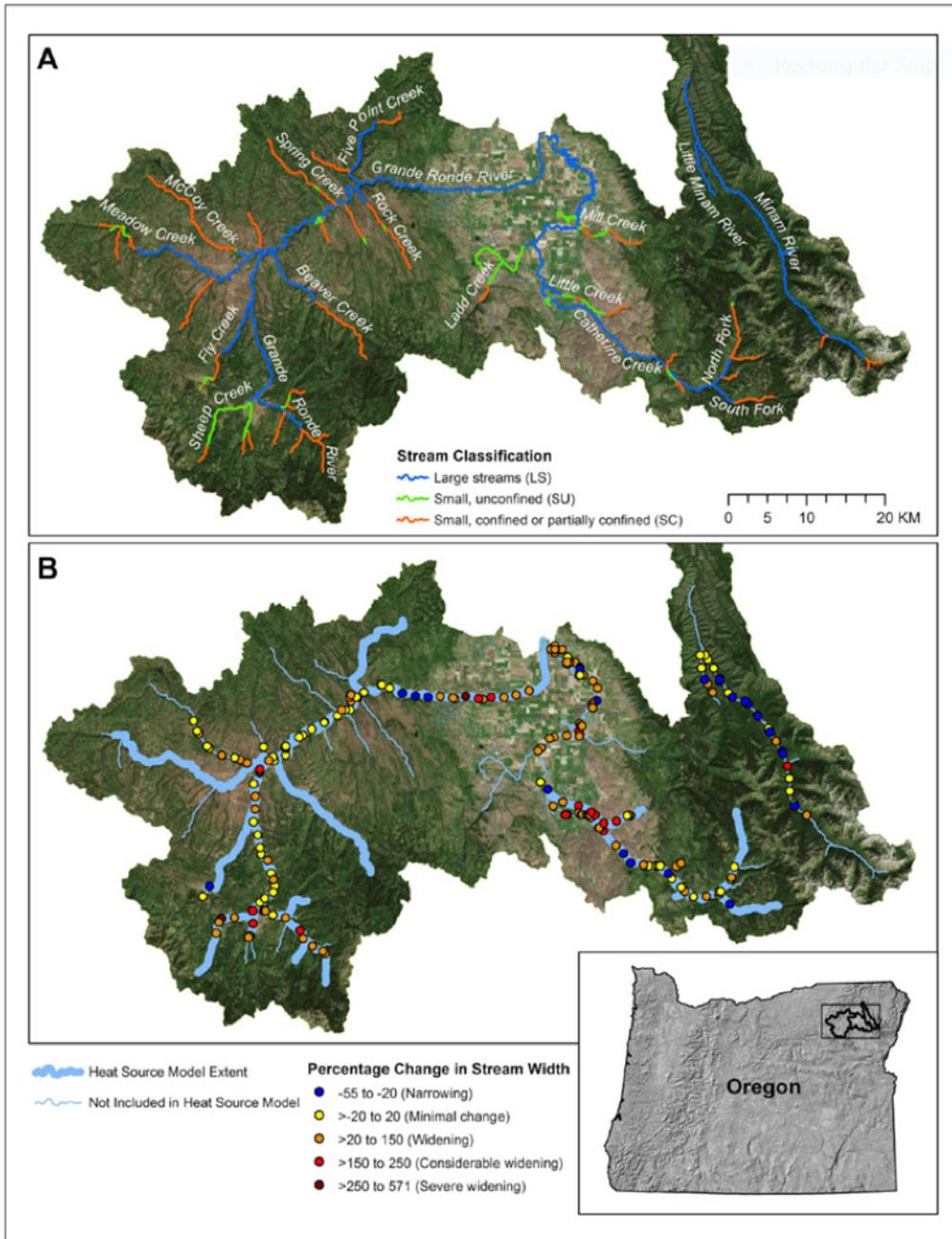
Figure 5.3. Study area, stream classification, and historical changes to channel widths in three focal watersheds. Location of study watershed in northeast Oregon including: A) major salmon-bearing tributaries and the stream classification described in the methods, and B) values of channel change estimates where historical General Land Office surveys intersected with contemporary Aquatic Inventory Program surveys. Focal watersheds include the upper Grande Ronde River, Catherine Creek, and Minam River. The upper Grande Ronde River and Catherine Creek have significantly modified stream conditions from over a century of intensive land use. The Minam River is in the Eagle Cap Wilderness area and most approximates historical reference conditions. From White et al. (2017).

warmer water temperatures through increased surface area exposed to solar radiation. This resulted in a drastic loss of suitable habitat meeting minimum thresholds for salmonids. Based on projections, 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 would translate 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; Figure 5.4).

Justice et al. (2017) then used a deterministic water temperature model called Heat Source (Boyd and Kasper 2003) to investigate potential thermal benefits of riparian reforestation and the channel narrowing analysis from White et al. (2017) to Chinook Salmon populations in the Upper Grande Ronde River and Catherine Creek basins in Northeast Oregon, USA. Inputs to the model included LiDAR data such as channel topography, local climate data, streamflow information from gaging station and manual flow measurements, and water temperature data from thermographs. In addition, extensive field measurements associated with each plant association group (PAG) and potential tree height estimates were used to determine historic/potential and current riparian vegetation conditions. A combination of local knowledge from experienced riparian ecologists was used, as well as detailed maps of current vegetation and potential natural vegetation (PNV) for a 100-m wide stream buffer throughout the Chinook-bearing portions of the Upper Grande Ronde and Catherine Creek watersheds that incorporated physiography, geomorphology, soils, vegetation, and disturbance (Wells et al. 2015). Potential tree height was estimated from species-specific dominant tree height growth curves from regional forestry literature. Weighted-average growth curves within each PAG were then used to estimate the average tree height under fully restored PNV conditions, which was assumed to occur at 300 years. Potential shrub heights were obtained from local sources and from species descriptions in the Fire Effects Information System.
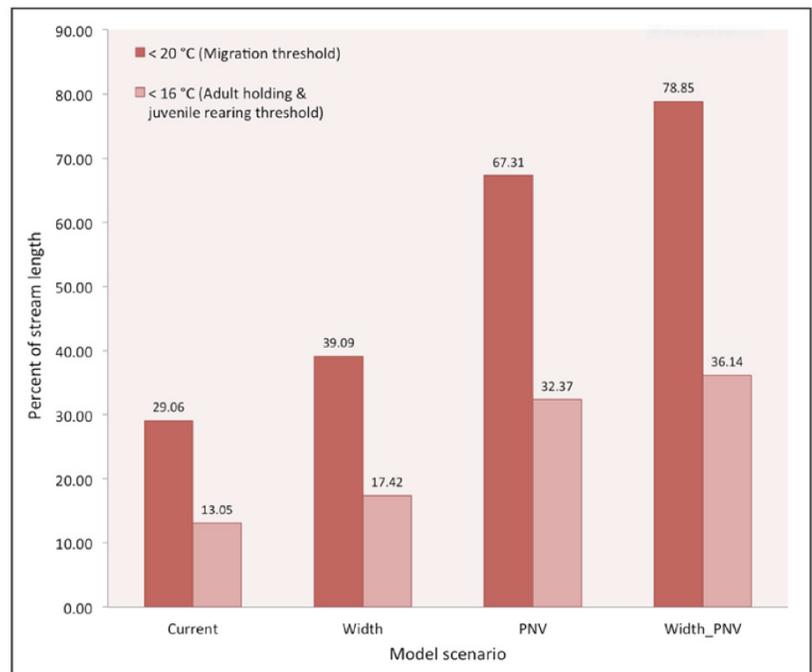
Figure 5.4 Percentage stream length below biological water temperature thresholds for model scenarios. Estimated percentage of stream length below critical salmon and steelhead thresholds for maximum weekly maximum water temperatures (MWMT; EPA 2003) in the upper Grande Ronde River and Catherine Creek watersheds combined. Model scenarios represent current conditions (Current), restored channel width (Width), restored potential natural vegetation (PNV), and the combination of vegetation and channel-width restoration (Width_PNV). From White et al. (2017).
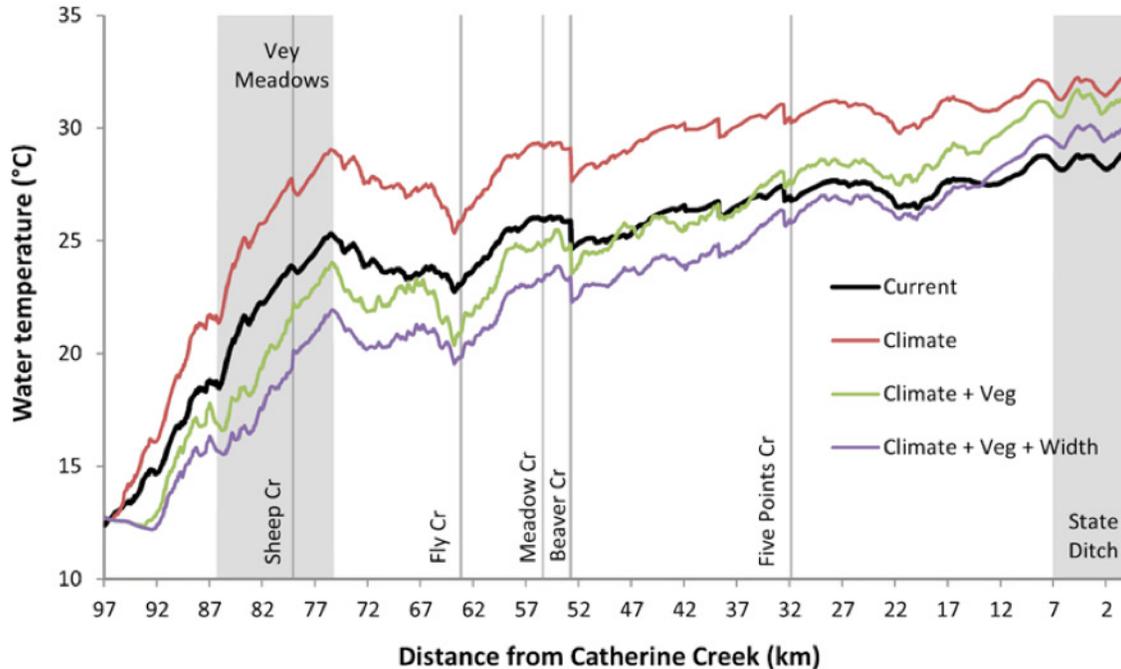
Figure 5.5. Simulated maximum weekly maximum water temperature (MWMT) in the mainstem Grande Ronde River from the headwaters to the Catherine Creek confluence for four model scenarios including current conditions, 2080s climate conditions, 2080s climate conditions plus riparian vegetation restoration, and 2080s climate conditions plus riparian vegetation and channel-width restoration from Justice et al. (2017).

By combining restoration scenarios with climate change projections, Justice et al. (2017) evaluated 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 (Figure 5.5). These results translated to long-term, stable increases in Chinook Salmon parr abundance of 590% and 67% respectively once the modeled actions impacted the population dynamics (Figure 5.6). 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), the model 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. This translated to potential increases in Chinook Salmon parr abundance of 67% to 590 %, respectively. These results underscore the potential for riparian and stream channel restoration to mitigate climate change impacts to threatened salmon populations in the Pacific Northwest.
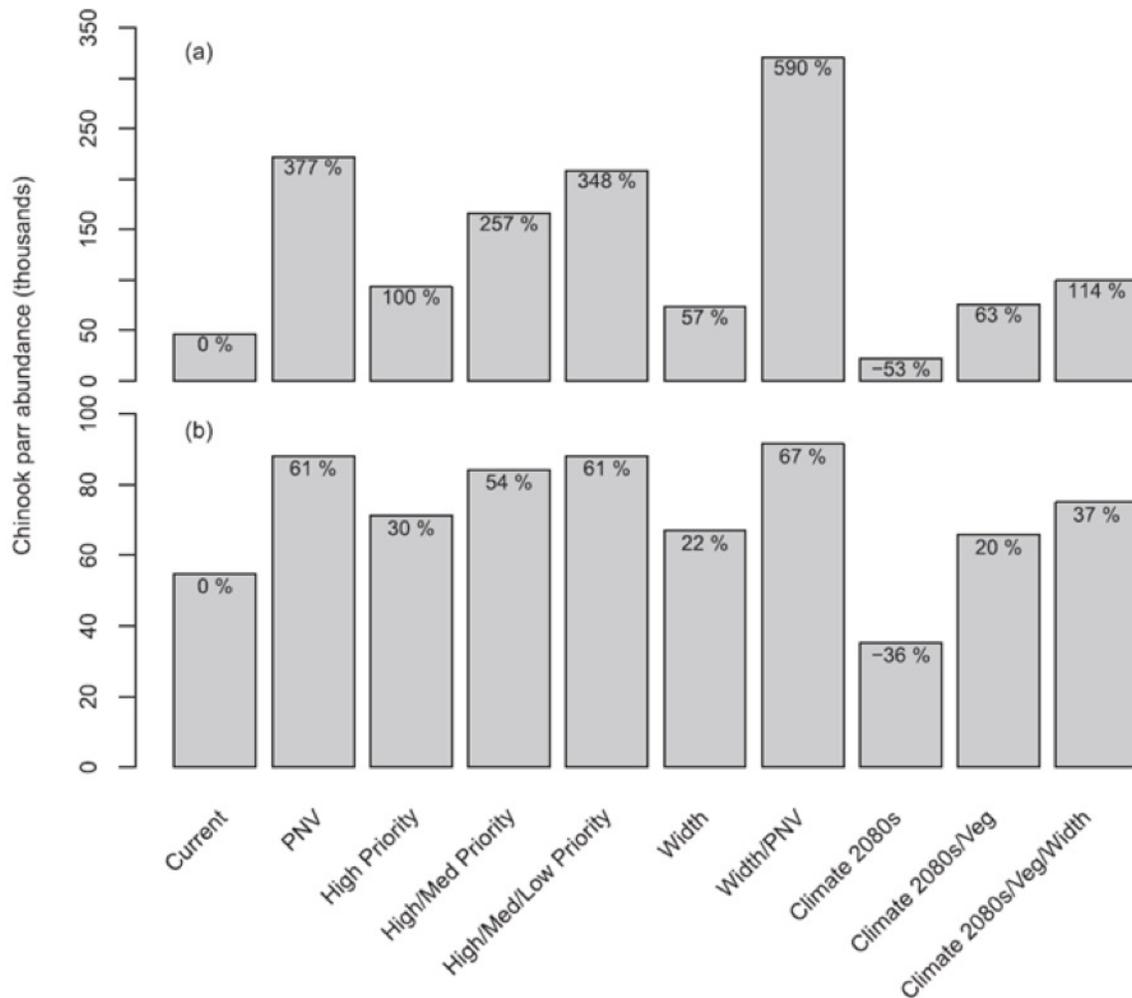
Figure 5.6. Predicted abundance of Chinook Salmon summer parr for each model scenario in a) the Upper Grande Ronde River, and b) Catherine Creek basins. Numbers at the top of each bar indicate the percentage change in abundance from the current condition (Justice et al. 2017).

The basic approach for incorporating habitat change effects starts with current life stage capacities and survival estimates derived from the 20+year juvenile series for each population. Using Catherine Creek summer parr stage as an example, we calculate the total amount of pool equivalent habitat currently supporting spawning and/or rearing. Other than scaling the expression of juvenile life stage parameters to the total amount of pool equivalent habitat within a population, our Grande Ronde MLCMs do not directly include habitat parameters. We use multipliers on life stage-specific survival and capacity terms as inputs to model the impact of habitat actions or environmental changes. The basic approach for incorporating habitat change effects starts with current life stage capacities and survival estimates derived from the >20-year juvenile series for each population as described above. We translate proposed actions into changes in the amount of pool equivalent habitat in the treatment reaches and express the results as a ratio of the new total to the current estimate. That ratio is than used as a multiplier to increase the summer rearing capacity in the model. Life stage survivals can be increased by habitat actions in three ways; in cases where a direct survival impact is alleviated (e.g., irrigation diversion screening-related mortality), a multiplier on survival weighted for the proportion of current rearing area benefiting from the action is used. Restoring riparian cover, reconnecting stream channels to associated groundwater sources, or creating localized water storage (Wondzell et al. 2007) can directly reduce stream temperatures.

Although the MLCMs can be used to model the effects of individual reach scale habitat actions, assessment of larger scale restoration strategies is a more effective use of their capabilities. In practice, larger scale restoration strategies will take time to implement. In addition, actions such as restoring riparian habitat will take additional time to result in changes to conditions affecting juvenile or adult life stages in the reach. For example, developing canopy cover providing effective shade to adjacent stream reaches can take decades to reach full maturity. Our procedures for translating proposed actions into life stage model inputs use a simple set of assumptions to address these factors. We use results from a long-term habitat study in the upper sections of the Grande Ronde basin (Justice et al. 2017, White et al. 2017) as a starting point for translating potential restoration actions into temperature effects on juvenile Chinook production.

We estimated the potential changes in juvenile rearing capacity for restoring high and medium priority reaches in Catherine Creek by applying the mixed effects model described in Justice et al. (2017) that relates late summer juvenile densities to stream temperatures. We applied the model to each 200-m segment of stream in two priority sections of Catherine Creek (the current core spawning and rearing habitat above the town of Union, and the contiguous downstream section from Union to Pyles Creek). We combined the incremental implementation schedule with the generalized riparian response time described in Justice et. al. 2017 using a polynomial equation corresponding to their estimated response times (40% of benefits after 25 years, 85% after 75 years).

5.3.1. Estimating restoration effects on habitat capacity or survival: Developing historical, current, and strategy-specific restoration scenarios

We modeled three incremental habitat action sets; 1) specific actions called for in the current draft NE Oregon Recovery Plan, 2) expanded actions targeting priority reaches identified through the Catherine Creek ATLAS project and 3) implementation of stream/ riparian restoration in high and moderate priority reaches identified in Justice et al. (2017). The Grande Ronde Model Watershed project is currently compiling a six-year strategic work plan identifying projects to be developed and implemented over the next six years. We are prepared to analyze the potential effects of those actions when the descriptions of the component actions become available for that action plan.

Although the MLCMs can be used to model the effects of individual reach scale habitat actions, assessment of larger scale restoration strategies is a more effective use of their capabilities. In practice, larger scale restoration strategies will take time to implement. In addition, actions such as restoring riparian habitat will take some time to fully realize potential changes to conditions affecting directly juvenile or adult life stages in the reach. For example, developing canopy cover providing effective shade to adjacent stream reaches can take decades to reach full maturity. Our procedures for translating proposed actions into life stage model inputs use a simple set of assumptions to address these factors.

The impacts of restoring 10 cfs in flows were estimated using data from CHaMP sampling in the Union to Davis Dam reach analyzed using the U.S. Forest Service River Bathymetry Toolkit (McKean et al. 2009). The effect of the action was expressed as a proportional increase in suitable pool habitat. The draft Recovery Plan also calls for restoring 3 miles of side channel or meander habitat. We assumed that reconnected or reconstructed channel habitats would be in the same low gradient reach (Union to Davis Dam), and that the resulting additional channel habitat would average 80% pool frequency. We assumed these actions would increase the juvenile Chinook summer rearing capacity for the population, but that temperatures would not be changed from current ranges.

For evaluating the impacts of habitat projects implemented in 2009–16, we used summaries of the expected change in key habitat parameters estimated by the Upper Grande Ronde/ Catherine Creek Expert Panel (EP). The U.S. Bureau of Reclamation (BOR) compiled tables capturing the results of the EP process including their identification of the specific reach locations (length treated) and their estimates of the potential change in key factors (e.g., side channel added or activated, floodplain accessed, increase in LWD, increase in sinuosity, riparian plantings, etc.). Where appropriate, the EP included estimates of the relative effectiveness of the methods used to implement the action. We used the standard action categories and the conclusions of the EP in our modeling application.

The third increment of change was based on the high and moderate priority reach restoration scenario described in Justice et al. 2017 and White et al. 2017. This scenario focuses restoring stream structure and reducing temperatures through the combined effects of riparian shade and achieving natural channel structure and width/depth ratios (White et al. 2017). Most of the reaches identified as high priority for riparian restoration along Catherine Creek course through private lands. Implementing these large-scale restoration actions will require extensive landowner cooperation and coordination. In some circumstances, restoring natural channel structure may require direct intervention given the degree of degradation (e.g., extreme channel widening due to historical splash dam activities). Given the time requirements to get agreements in place and limitations on the resources required to actually implement large-scale riparian restoration, we assumed a 20-year implementation schedule.

We have emphasized habitat opportunities within and immediately (8–10 km) downstream of current production areas in these analyses. With the possible exception of the Minam River population, extending sustained natural production into those reaches would provide a basis for further restoration in the historically productive wide valley habitats immediately below.

5.3.1.1. Grande Ronde Valley outmigrant survivals

As described above, outmigrating smolts from Catherine Creek (and to a lesser extent the Upper Grande Ronde River) are subject to relatively high mortalities either during active migration or just prior to beginning that phase (e.g., Favrot et al. 2018). The factors contributing to this increased mortality are not well understood. Two possible contributing mechanisms have been suggested, both at least partially driven by the unique spring flow condition at the lower end of the Grande Ronde Valley. Flows from the Upper Grande Ronde bypass the old Grande Ronde channel via the State Ditch, which begins near La Grande, Oregon well upstream of the former Catherine Creek confluence and rejoins the old main stem channel approximately 22 km below that confluence. Spring flows from the Upper Grande Ronde are backed up when they encounter the relatively confined geology at the lower end of the valley. As a result, migrants from Catherine Creek encounter slack water or even an upstream flow as they pass downstream. Reasons for the documented high levels of mortality during the transition through this reach are unclear. It is possible that migrating smolts delayed in this reach are highly vulnerable to avian or piscine predation. It also is possible that the interruption in normal migration timing is a contributing factor. An ODFW study is underway to gain an understanding of the causes and to identify strategies to reduce this documented mortality (Favrot et al. 2018). To illustrate the potential benefits of reducing mortality levels during this life stage, we have run scenarios including an assumption that managers will identify and implement an approach that will reduce the mortality associated with this reach to average levels observed for migrants from the Lostine and Minam Rivers, which enter a relatively short distance downstream (~50% stage survival increase).

5.3.2. Estimating population-level outcomes of each restoration alternative: Using LCMs to evaluate differences in fish production among restoration scenarios

We estimated the potential changes in juvenile rearing capacity for restoring high and medium priority reaches in Catherine Creek by applying the mixed effects model described in Justice et al. (2017) that relates late summer juvenile densities to stream temperatures. We applied the model to each 200 m segment of stream in two priority sections of Catherine Creek (the current core spawning and rearing habitat above the town of Union, and the contiguous downstream section from Union to Pyles Creek). We combined the incremental implementation schedule with the generalized riparian response time described in Justice et al. 2017 using a polynomial equation corresponding to their estimated response times (40% of benefits after 25 years, 85% after 75 years).

We run 500 simulations of 105 years each for a particular scenario, drawing randomly from parameter distributions (a single 100-year simulation) and random variability elements (annually). The results are saved in arrays, the standard set includes annual spawners (total, natural-origin, and hatchery-origin), brood-year returns (natural-origin) and annual adult harvest rate. For runs invoking local supplementation, annual estimates of natural-origin broodstock removals, spawning area hatchery proportions and accumulated fitness effects are also stored. These arrays can be used to generate different summary statistics and graphics, both within and across scenarios.

Outputs can be summarized in ways that directly correspond to risk and recovery metrics used in status reviews, Biological Opinion evaluations and recovery planning. For example, summarizing frequency distributions of 10-year geometric mean natural-origin spawners at selected years (e.g., 25, 50, or 100 years) or reporting the proportion of runs that fall below a selected quasi-extinction threshold. The ICTRT recommended using a QET of 50 fish averaged over four years as a long-term recovery benchmark. Risk assessments used in prior FCRPS hydrosystem biological opinions also included a QET of 30.

5.3.3. Estimating population-level outcomes of each restoration alternative: Using LCMs to evaluate a six-year strategy for the Upper Grande Ronde River and Catherine Creek

Proposed actions and locations have been developed for Spring Chinook salmon populations in the Upper Grande Ronde River and Catherine Creek (Figure 5.7, Table 5.2). This is based upon current habitat conditions and an overall understanding of the limiting factors associated with Spring Chinook salmon in these basins (Table 5.3).
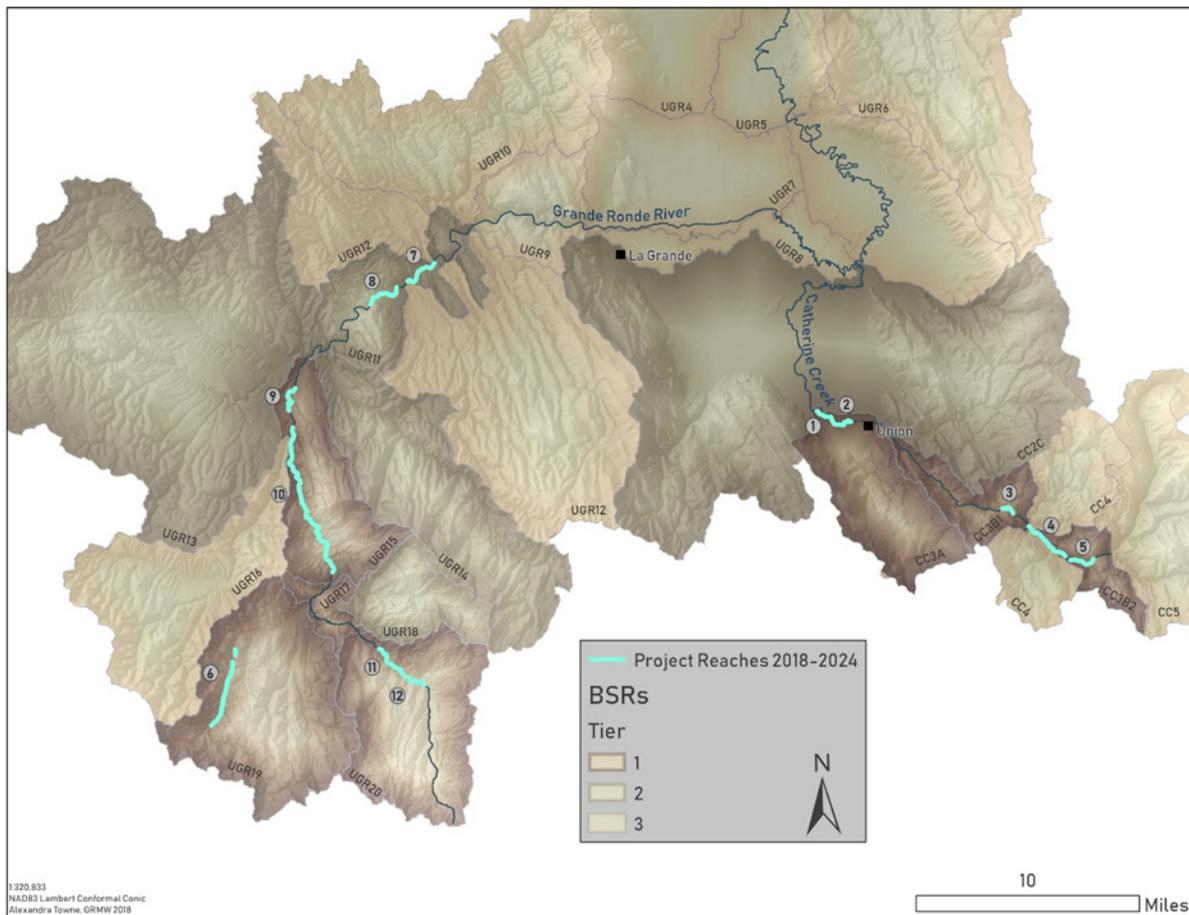
Figure 5.7. Stream restoration project areas in the upper Grande Ronde River. Projects are slated for construction in 2018–24. Numbers correspond to the segment numbers in Tables 5.2 and 5.3.

Table 5.2. Proposed restoration actions for 2018–24 within the current spring Chinook spawning and rearing domain in the Upper Grande Ronde River (UGR) and Catherine Creek (CC) populations. Segment number corresponds to segment number on map in Figure 5.7.

Segment number	River	Reach name	Flood-plain acres	Stream miles	CFS dedicated in-stream	Sinuosity	LWD pieces/ 100 m	Total pools/km	Large pools/km	Side channel (m)
1	CC	CC 37 LWD	21	0.75	0	1.38	15	7	2	119
2	CC	CC Red Mill Reach	13	1.44	0.24	1.4	35	20	8	1,136
3	CC	CC State Parks	8	0.62	0	1.1	18	17	4	625
4	CC	CC Hall Ranch	123	2.25	0	1.3	22	15	5	5,000
5	CC	CC LDS Camp	8	1.2	0	1.1	40	10	5	0
6	Sheep Creek	Sheep Creek	85	4.5	0	—	20.7	—	—	TBD
7	UGR	UGR Longley Meadows/ Gun Club	75	1.6	0	TBD	TBD	TBD	TBD	TBD
8	UGR	Bird Track Springs	114	1.8	0	1.3	84	31	31	1,770
9	UGR	UGR Bowman Property	27	1.5	0	1.1	50	18	5	804
10	UGR	UGR River Canyon	60	8.1	0	—	36	—	—	TBD
11	UGR	Woodley Campground	30	2	0	>1.2	27	8	4	690
12	UGR	UGR Mine Tailings	50	3	0	>1.2	35	8	4	776

Table 5.3. Current habitat conditions for large woody debris (LWD) and pool frequency at proposed restoration reaches. Habitat data come from Aquatic Inventories Project reports, U.S. Forest Service Level 2, and the Columbia Habitat Monitoring Program (CHaMP).

Segment number	River	Reach name	LWD pieces/ 100 m	Pools/km	Large pools/km
1	CC	CC 37 LWD	3.8	10	5.9
2	CC	CC Red Mill Reach	—	—	—
3	CC	CC State Parks	7.6	16.1	3.3
4	CC	CC Hall Ranch	12.3	10.7	1.5
5	CC	CC LDS Camp	8.7	4.7	1.4
6	Sheep Creek	Sheep Creek	24.7	18.9	0.4
7	UGR	UGR Longley Meadows/Gun Club	1	8..8	0.4
8	UGR	Bird Track Springs	3.1	18.7	0.8
9	UGR	UGR Bowman Property	3.7	16.3	1.0
10	UGR	UGR River Canyon	15	29.3	1.0
11	UGR	Woodley Campground	27	52.6	1.1
12	UGR	UGR Mine Tailings	15.4	46.8	2.0

5.4.   Results

Current spawning and juvenile rearing habitats for each of the four populations extend from higher elevation moderate gradient forested valleys downstream through lower gradient alluvial fan and Grande Ronde Valley habitats The Upper Grande Ronde, Catherine Creek populations along with the Wallowa and Lower Lostine River reaches in the Lostine Wallowa population have been substantially altered by human impacts – including channel straightening, diking, LWD removal, degraded riparian habitats and summer baseflow reductions (e.g., White et al. 2017). In recent years, the Oregon Aquatic Inventory surveys (AQI) have generated direct estimates of the relative physical conditions across reaches in each population. We used relative parr densities from snorkel surveys across the three Oregon AQI stream channel classifications (pools, runs, and fastwater) as a basis for expressing the total available habitat in each population in pool-density equivalents (Table 5.4). Although absolute abundance varied across surveys by year and population, average levels in run and fastwater habitats were relatively consistent in proportion to the corresponding pool densities.

The recent CHaMP/ISEMP project compiled reach level stream temperature series for sample sites across the Grande Ronde populations (Figure 5.8). Summer peak temperatures varied from site to site, but the annual patterns across months were similar. All sites had very low winter temperatures extending into early spring, followed by a gradual increase to peak temperatures in Aug. Stream temperatures declined through the fall to winter lows.

Projected summer (Aug average) stream temperatures from the NorWeST regional model were highly correlated with average Aug temperatures at the sample reaches. In addition, direct estimates of maximum weekly maximum temperatures (MWMT) at sample sites were highly correlated with the corresponding empirical Aug average stream temperatures. We used the regression of MWMT on Aug average temperature to project reach-scale estimates of MWMT from the NorWeST Aug average temperatures.

$$MWMT = 1.46 \cdot NorWestAug - 3.65 \quad R^2 = 0.9872$$

Table 5.4. Amounts of tributary spawning and rearing habitat in reaches used for spawning and juvenile rearing above juvenile weirs. Based on estimated areas of pool, run, and fastwater habitat multiplied by relative parr densities observed during CHaMP/ISEMP snorkel surveys. Unit = 10,000 m^2.

	Catherine Creek	Upper Grande Ronde River	Lostine River	Minam River	Relative density factor
Pools	7.61	5.00	3.48	15.54	*1.00*
Runs	1.20	1.91	4.60	5.37	*0.35*
Fastwater	18.45	27.08	29.76	29.76	*0.24*
Total	*27.27*	*33.99*	*37.85*	*50.67*	
Weighted total	**12.44**	**12.13**	**12.21**	**24.53**	
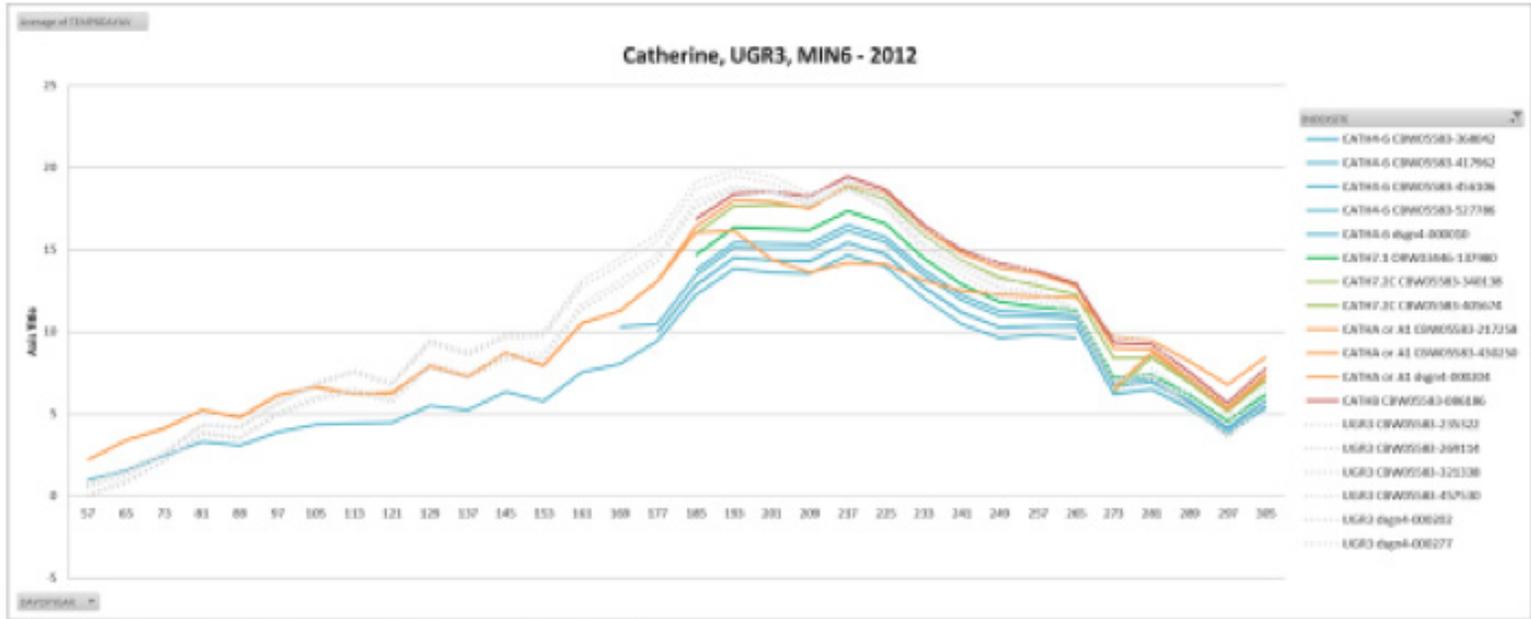
Figure 5.8. Within-year in-stream temperature estimates from CHaMP/ISEMP sampling sites in Grande Ronde spring Chinook populations.

Stream temperature is an important constraint on spring Chinook spawning and juvenile rearing in the Grande Ronde basin. Current summer temperatures in the lower sections of the current use reaches in the Upper Grande Ronde, Catherine Creek and the Wallowa/Lostine populations coincide with substantial declines or absences of spawner and juvenile densities (Figure 5.9). The vast majority (95%) of current spawning in the Upper Grande Ronde population is above where average summer stream temperatures exceed 17.5°C, which extrapolates to 20.5 degrees MWMT (Figure 5.10).

The Oregon AQI surveys identified the amount of accessible side channel habitat associated with mainstem reaches in each population as well as the proportions of that habitat classified as pools, runs or fastwater. Using the Beechie and Imaki (2014) natural potential channel pattern classification system current use reaches in each population are dominated by the meandering pattern, with sections of confined and straight channel patterns (Figure 5.11). The amount of current side channel habitat is well below historical levels based on the relative frequencies of Beechie and Imaki

Figure 5.9. (top) NorWeST Aug mean temperature estimates for Catherine Creek stream reaches. (bottom) Oregon AQI reach-level pools and 2009–16 GPS redd locations within Catherine Creek.

(2014) channel pattern classes and a recent land use-based study of floodplain status in the Interior Columbia Basin tributaries (Bond et al. 2017).

The Oregon AQI surveys indicate that, with the exception of the Minam River, LWD levels are below levels for naturally functioning habitats across reaches in all populations (e.g., White et al. 2017). At the reach level summarized in the Oregon AQI results, fine sediments are not a significant limiting factor on current spawning/rearing—with one major exception, the mainstem Wallowa River.

Figure 5.10. (left) NorWeST 1993–2011 average Aug stream temperatures for the upper Grande Ronde River current spawning/rearing use reaches. (right) Oregon AQI reach level pools and 2009–16 GPS redd locations within upper Grande Ronde population current use reaches.
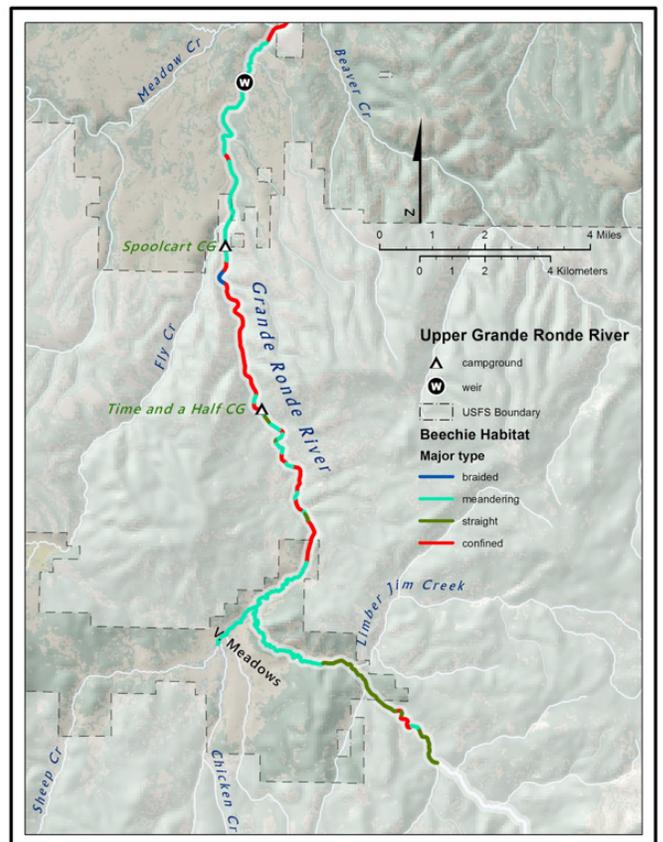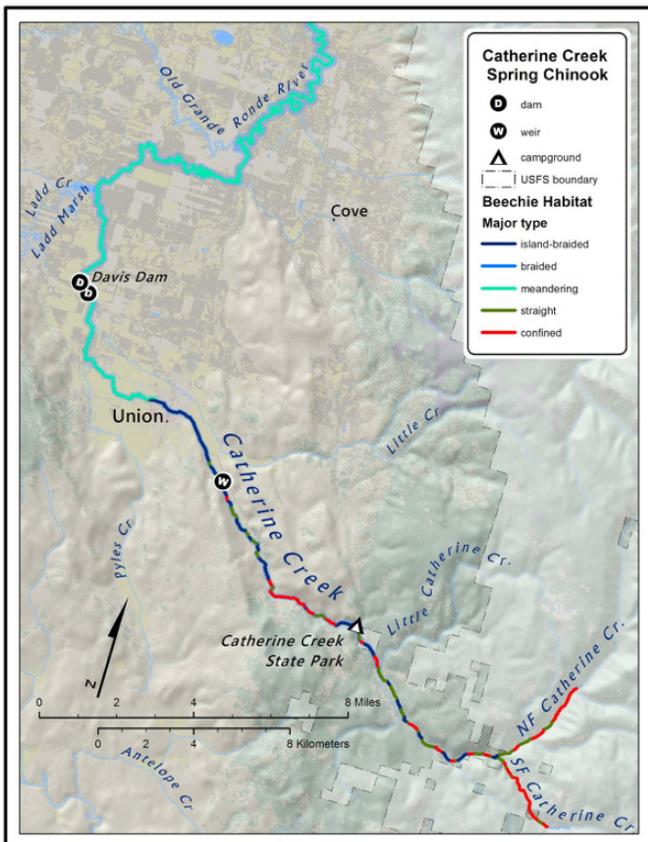
Figure 5.11. (left) Catherine Creek Spring Chinook population 200-m reach level Beechie class ratings. (right) Upper Grande Ronde Spring Chinook population 200-m reach Beechie stream classes.

5.4.1. Steps 1 and 2: Estimating life stage capacities using population-specific fish and habitat data

The amount of habitat associated with current levels of spawning and summer rearing differed considerably across the four Grande Ronde Chinook population tributaries. We standardized each of the four data series to spawner and summer parr per 10,000 m^2 of pool habitat using estimates from the ODFW Aquatic Inventory (AQI) surveys. There were consistent patterns in relative densities (pools, runs and fastwater) across surveys, populations and years. For each population, we expressed the results as an AQI index of pool equivalent habitat by weighing the category habitat subtotals by the relative density index for each category (Table 5.4). We used the resulting population totals to standardize spawner and parr densities to a common unit of habitat. For Catherine Creek, we estimated an additional expansion factor to account for the use of habitat below the weir site for spawning and early rearing.

There were consistent differences in patterns of flow and temperature conditions across the four populations. The Catherine Creek and Upper Grande Ronde study reaches have lower summer flow and higher summer maximum temperature index values. The Lostine and Minam reaches are subject to higher flow levels and lower average maximum summer air temperatures than either the Upper Grande Ronde or Catherine Creek current natural production reaches.

In three of the four study populations, the juvenile screw traps were below almost of the spawning and natal rearing habitat currently in use. The Lostine/Wallowa population is an exception, with spawning and associated rearing occurring in the mainstem Wallowa River and two tributaries, in addition to the Lostine River (Bear and Hurricane Creeks). Direct estimates of juvenile production are not available for the production areas outside of the Lostine River. Since 1995, an average of 65% of the redds counted in the Lostine/Wallowa population have been above the weir and juvenile screw trap. We assumed that the parent escapement estimates and the juvenile production relationship per unit of pool habitat (ODFW AQI) derived from the Lostine smolt trap applied to the other three current production areas.

5.4.1.1. Spawner to summer parr stage

We compared summer parr per spawner ratios (standardized to 10,000 m^2 AQI habitat) against the flow and temperature indices and against parent spawning densities. We used the `nlsboot` routine in R to generate a data set of 1,000 iterations of the fitted *a* and *b* parameters for each curve. We stored the resulting combinations of *a* and *b* parameters for use in the matrix model. The estimates of productivity, asymptotic parr capacity (per hectare of pool-equivalent habitat), and the residual standard errors are summarized in Table 5.5 and depicted in Figure 5.12. There were no significant trends in parr-per-spawner for the environmental indices tested. However, the estimates grouped by population did fall out at relatively distinct temperature levels. For each population, the relationship between parent spawner density and parr density was statistically significant. The standard errors for these estimates are relatively large. The per-hectare estimates of summer parr capacity can be expanded to current population totals by multiplying by the AQI estimates from Table 5.4 . The resulting maximum likelihood estimates of current total parr capacity range

from a low of 88,300 for the Upper Grande Ronde to 481,800 for the Lostine section of the Wallowa/Lostine population. The estimates for the remaining populations were Catherine Creek (118,500) and Minam (351,300).

Low-to-moderate parent escapement levels relative to the range of escapements observed since the early 1950s have a large effect on the population data sets, with very few data pairs within the higher escapements in the range. The resulting fitted curve is representative of the production relationship with the range of recent escapements. It is uncertain how the weighting to lower escapement levels affects the projected shape of the fitted relationship at higher escapement levels.

Table 5.5. Beverton–Holt parameters fitted to ODFW 1992–2016 annual adult spawning and parr abundance estimates. Spawner and parr estimates were standardized to 10,000-m² pool-equivalent habitat. Parameters were generated using the R statistical package `nlsboot` routine. *Key:* CC = Catherine Creek, UGR = Upper Grande Ronde River, LR = Lostine River, MR = Minam River.

Model	B–H *a* (SE)	exp(*a*)	B–H *b* (SE)	sigma
CC	6.326 (0.258)	558.9165	9,528 (5,162)	0.452
UGR	6.287 (0.351)	537.5383	7279 (5,269)	0.439
LR	5.918	371.6676	28,770	0.440
MR	6.181	483	19,640	0.542
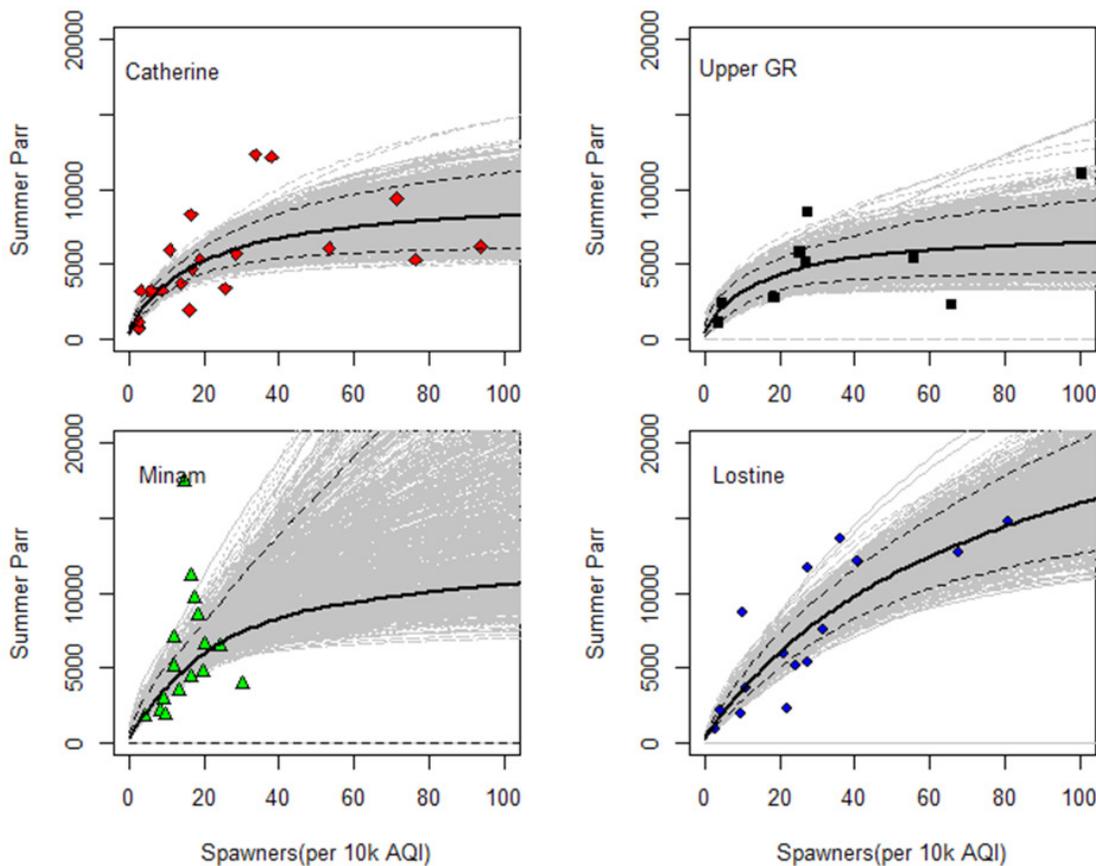
Figure 5.12. Spawner to summer parr relationships fitted to population-specific estimates (points). Gray-shaded zones reflect bootstrap joint parameter evaluation. Solid lines: median across 4,000 iterations; dashed lines contain the central 90% of results. Population estimates standardized to 1-hectare pool-equivalent habitat.

5.4.1.2. Summer parr to spring tributary outmigrant stage

A portion of the juvenile Chinook rearing in each of the four Grande Ronde study populations emigrates downstream in the fall to overwinter before initiating seaward outmigration the following spring. The remainder stay upstream to overwinter, with the survivors emigrating in the spring. The proportion of the estimated population migrating downstream to overwinter below the migrant traps in each population area varied annually, but did not appear to be a function of summer parr density, juvenile length, summer temperature or flow. The average annual ratio of fall migrants to summer parr did vary across populations. The Upper Grande Ronde and the Minam had the lowest average ratios (0.12 and 0.19, respectively). Catherine Creek had the highest (0.37), followed by the Lostine (0.29). These ratios are influenced by several factors, including placement of the migrant traps relative to habitat types utilized.

Survival between summer parr stage and the fall migration (peak in Oct) and winter parr in natal reaches is not directly estimated for either group. The summer parr to spring survival estimates represent the aggregate fall and spring run components (Table 5.6, Figure 5.13). We made a simplifying assumption, that survival from spring migration from downstream overwintering areas to Lower Granite Dam was the same as the estimated survival to Lower Granite Dam for the natal overwintering group passing the smolt trap in the spring. This allowed us to estimate the total number of smolts leaving the tributary (survivors from the fall downstream redistribution and the spring outmigration from the natal rearing areas). Both fall and spring length frequencies are strongly related to summer parr density (Figure 5.14), indicating the potential for density-dependent effects at recent spawning levels.

Table 5.6. Logistic regression results for summer parr to spring migrant stage survivals vs. summer parr density. *Key:* CC = Catherine Creek, UGR = Upper Grande Ronde River, LR = Lostine River, MR = Minam River.

Population	Stage	Intercept	Parr density term	Significance level	sigma
CC	summer to spring	-0.575	-9.61E-05	0.0058	0.420
UGR	summer to spring	0.100	-1.30E-04	0.0422	0.470
LR	summer to spring	-0.856	-2.89E-05	0.0004	0.182
MR	summer to spring	-0.865	-5.31E-05	0.0502	0.388
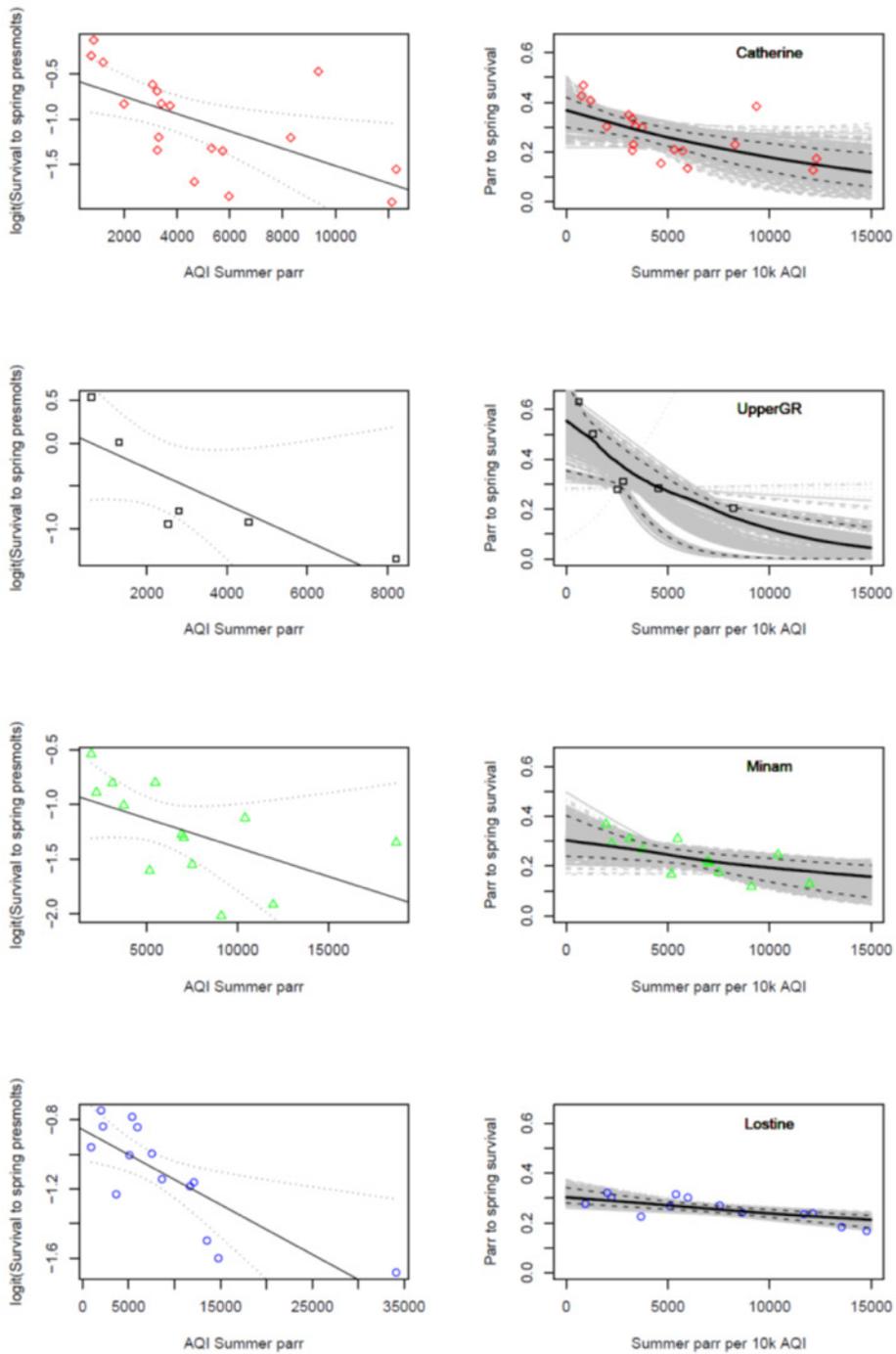
Figure 5.13. Summer parr to spring migrant survivals. 1992–2016 migration-year estimates. Gray zone represents 90% central interval for 4,000 bootstrap samples. (left) Logistic scale; (right) transformed estimates.

ANCOVA Results
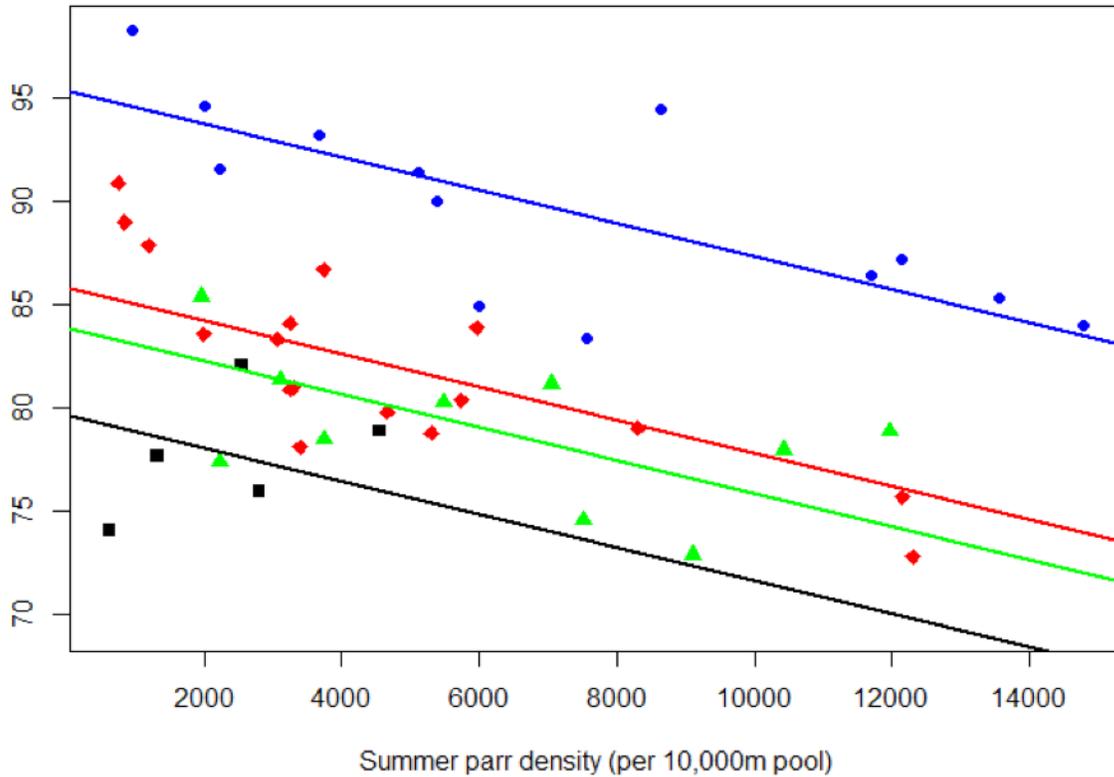
Figure 5.14. Analysis of covariance results. Points are individual-year estimates by population, lines are statistically significant common rate of decline in length vs. summer parr density across populations. Intercepts differ by population.

5.4.1.3. Spring outmigrant to Lower Granite Dam stage

Population-specific estimates of survival for the spring outmigrant to Lower Granite Dam were also evaluated as logistic regressions on parr density. The density-dependent terms were not significant, the relationships incorporated into the life cycle were expressed as a constant multiplier with a randomly drawn error term reflecting the variability in each population series (Figure 5.15). The average estimated spring outmigration survivals averaged 0.40 and 0.42 for the Catherine Creek and Upper Grande Ronde populations respectively. The survivals for this stage were consistently higher for the two populations whose natal tributaries enter below the Grande Ronde Valley (Minam 0.58 and Lostine 0.62). For several years in the study, ODFW operated a smolt trap and conducted pit tagging on outmigranting smolts below the two upper populations but above the Minam and Lostine. Migrating smolts intercepted and tagged at that trap survived at relatively high rates to Lower Granite Dam, indicating that the difference in survivals between the upper and lower populations resulted from factors within the Grande Ronde valley above Rhinehart Gap.
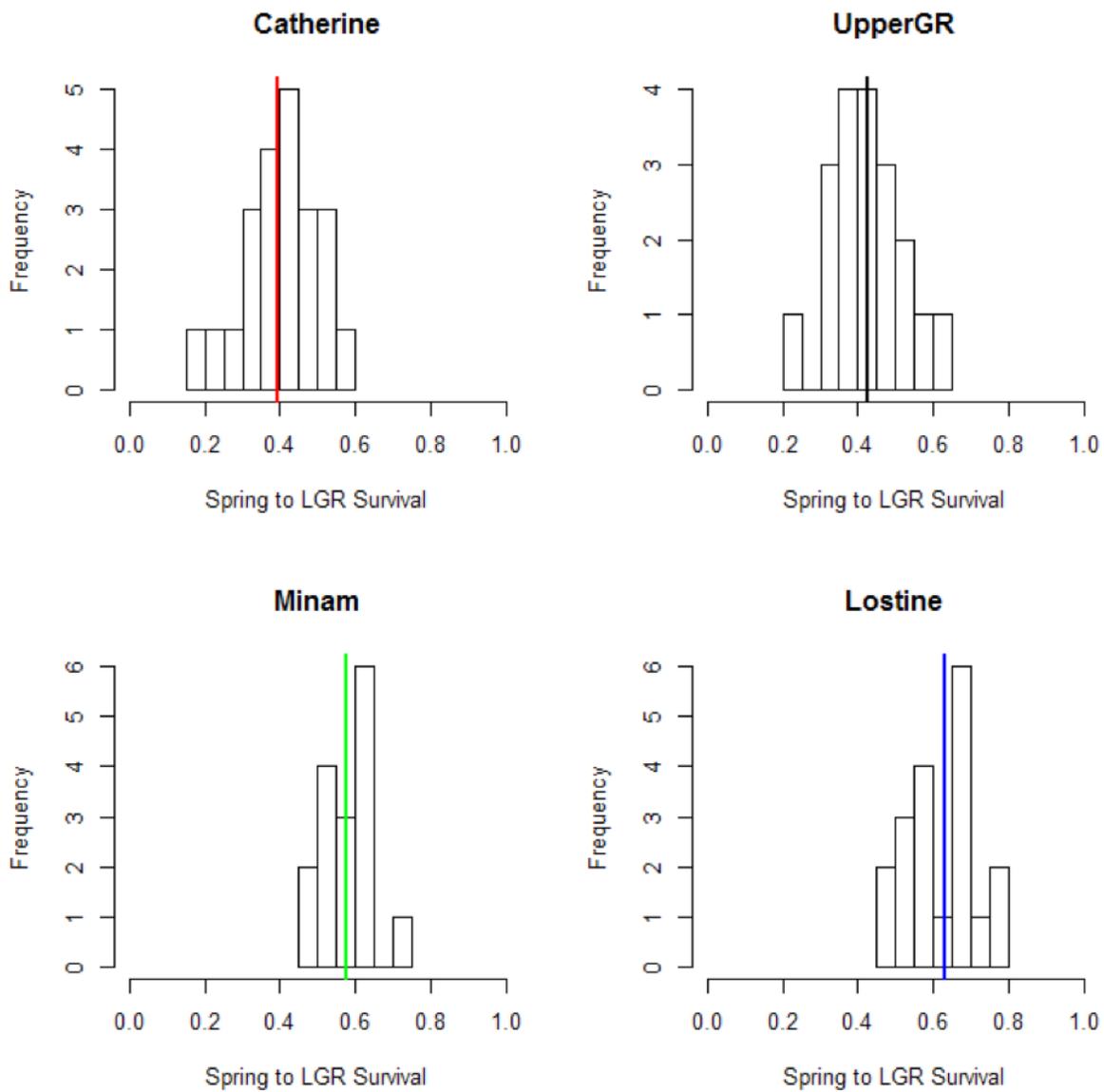
Figure 5.15. Estimated tributary spring migrant to detection at Lower Granite Dam by population. Vertical lines represent medians.

5.4.2. Step 3: Estimating habitat change inputs for the LCMs

The Grande Ronde LCMs were designed to accept estimated changes in specific life stage survivals and capacities. The primary input parameters used to model the scenarios described below are multipliers reflecting the expected changes in parr rearing capacity and outmigrant survivals. In the model, overwintering survival is linked to summer parr density reflecting the strong patterns in the empirical data sets for each population. A key working assumption of the approach is that the tributary stage production and survival relationships we derived from the >20-year adult spawner and juvenile data sets are related to the estimates of available habitat generated using the Oregon AQI data sets. We assume that habitat actions that would increase or decrease those levels over time would proportionally translate into changes from the derived parr capacities for each population.

5.4.2.1. Current habitat conditions

The current distribution of redds in Catherine Creek is largely restricted to reaches upstream of the ODFW weir site (Figure 5.16). Less than 5% of redds counted in annual surveys between 2009 and 2016 were below the weir site. While redd counts prior to 2009 were not georeferenced, ODFW did compile the counts by index reach. A larger proportion of redds were located in the reach extending downstream of the weir site to Union in the 1950–70 period. Potential contributing factors include the impacts of major storm events on stream structure, increased human constraints on channel movement and side channel availability, and increasing summer temperatures.

The majority of redds in the Upper Grande Ronde population are in the upper sections above Sheep Creek (Figure 5.17). Current redd surveys do not cover the mainstem reach passing through Vey Meadows. The Vey Meadows reaches were included in surveys prior to the early 1990s. We extrapolated current estimates for the Vey Meadows reach using average proportions from ODFW surveys and Oregon AQI pool data obtained in the early 1990s. ODFW AQI surveys in Sheep Creek only covered a portion of the reach habitat designated as current spawning and rearing. We used results from historical gravel surveys in the drainage to extrapolate from the AQI survey totals within Sheep Creek to cover the remaining reaches. Both survey methods gave similar estimates of average proportion pools over the common survey reaches. The gravel survey
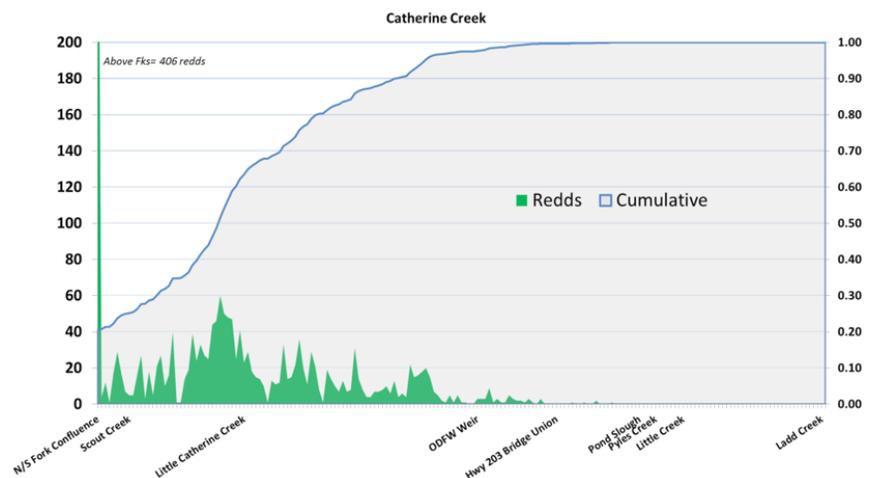
Figure 5.16. Catherine Creek distribution of redds (ODFW 2009–16 GPS) vs. reach location from North/South Fork confluence downstream. Redds in North and South Forks assigned to the first segment at the forks confluence. Green bars: redd counts. Gray shaded area: cumulative proportion moving downstream (secondary axis).

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average pool proportions above the AQI survey reach was roughly 50% of the gravel survey estimates for the AQI reaches. We assumed that the ratio of run to fastwater habitat for the remaining proportion total habitat was the same as in the AQI surveyed reach. We used the resulting estimated proportions to calculate a surrogate AQI estimate for the unsurveyed reaches. The lower reaches of Sheep Creek were also not sampled in either the 2010 or 2015 Oregon AQI survey. The NorWest temperature estimates for these reaches were relatively high, and there is evidence of local influence by hot springs flowing into the reach. We assumed that temperature conditions result in negligible use of lower Sheep Creek for Chinook spawning or summer rearing. The reach may support overwintering—although this has not been confirmed.
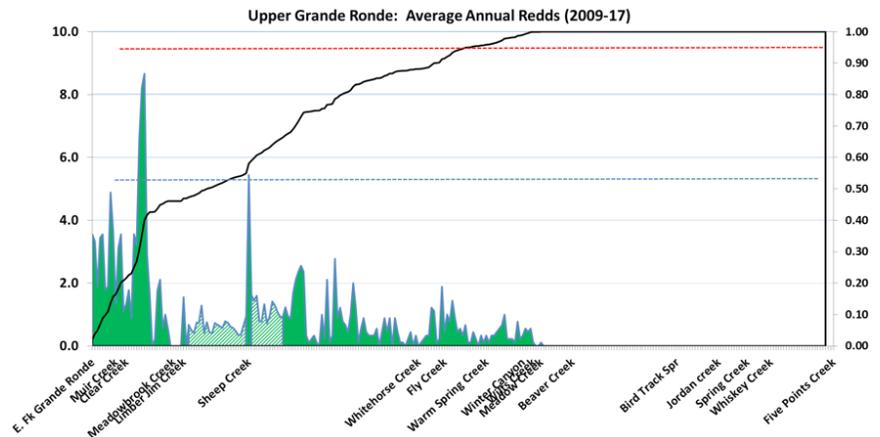
Figure 5.17. Upper Grande Ronde River distribution of redds (ODFW 2009–16 GPS) vs. reach location from upper extent of spawning to Meadow Creek confluence. Redds in Sheep Creek assigned to confluence. Green bars: redd counts. Gray shaded area: cumulative proportion moving downstream (secondary axis). Vey Meadows reach estimated by extrapolation from adjacent reaches using 1991 Oregon AQI survey data. Red dashed line: cumulative 95% of redds above this temperature.

In recent years, ODFW has included geo-referencing of individual redd count (2009–16+) in their annual Spring Chinook redd surveys in the Grande Ronde basin. ODFW complemented their CHaMP/ISEMP summer parr snorkel surveys in 2015 by sampling contiguous reaches from near La Grande upstream to the upper reaches of the East Fork Upper Grande Ronde River. We contrasted the resulting adult spawning and parr density patterns with reach specific NorWeST derived Aug stream temperature and selected Oregon AQI variables (pool area, sediment constituents). In spite of the availability of pool habitat, the presence of redds dropped off rapidly with increasing stream temperature. For the Upper Grande Ronde, 95% of the geo-referenced redds were upstream of the reach where average NorWeST stream temperatures exceed 17.5°C (Figure 5.18).
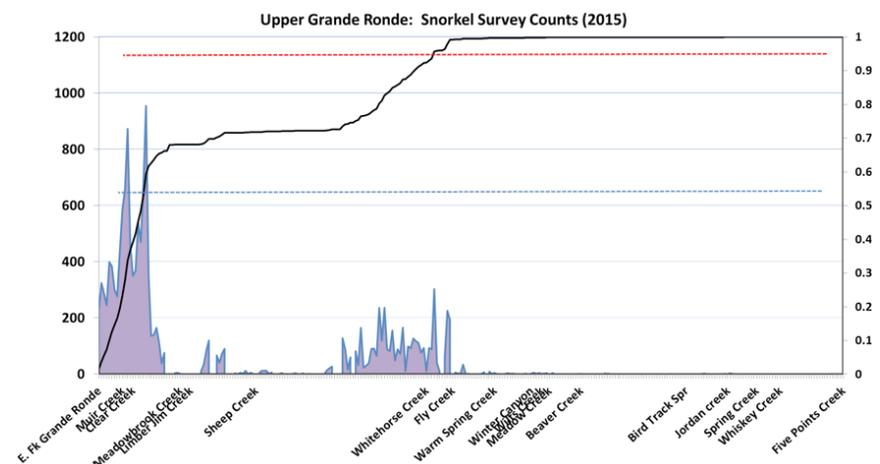
Figure 5.18. Upper Grande Ronde River. 2015 ODFW contiguous juvenile Chinook snorkel surveys (Five Points Creek upstream to upper extent of spawning). Purple: individual reach survey estimates (note: no surveys in Vey Meadows reaches). Black line: cumulative abundance from upstream extent (right-hand axis).

In 2015, ODFW conducted extended longitudinal juvenile snorkel surveys the length of the mainstem Grande Ronde River from the town of La Grande upstream to the upper extent of use in the East Fork Upper Grande Ronde (Figure 5.19). Summer rearing and spawner distributions showed similar relationships to current stream temperatures. Summer juvenile rearing was negligible below Warm Springs Creek. Two of the four study populations (Upper Grande Ronde and Catherine Creek) exhibited relatively high temperatures at the downstream end of current use as defined by ODFW. Other variables quantified by ODFW in the Grande Ronde basin include reach level longitudinal surveys summarized by habitat type (Figure 5.20), sediment characteristics (Figure 5.21), and estimates of LWD.

Justice et al. (2017) developed a temporal model of the temperature influence of riparian canopy development and paired it with results from Heat Source model runs for the Upper Grande Ronde and Catherine Creek mainstems to generate projected temperature impacts of riparian restoration scenarios (Figure 5.22). Full benefit of restoring riparian shading on adjacent stream reach temperatures took up to 300 years of tree growth, but "the most rapid reductions in temperature
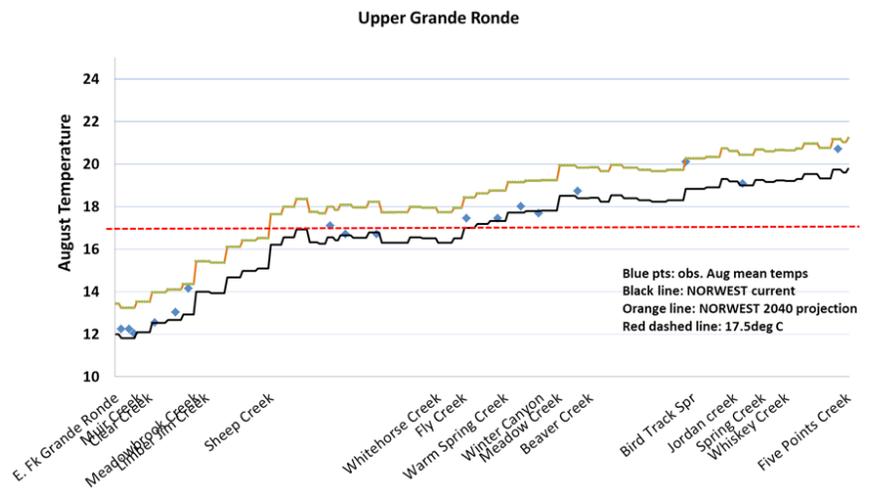
Figure 5.19. Upper Grande Ronde River population. NorWeST Aug average stream temperatures vs. reach. Black line: current temperature (1993–2016 average). Gold line: NorWeST projected 2040 stream temperature. Red dashed line at 17.5°C: estimated temperature threshold for spawning in this population.
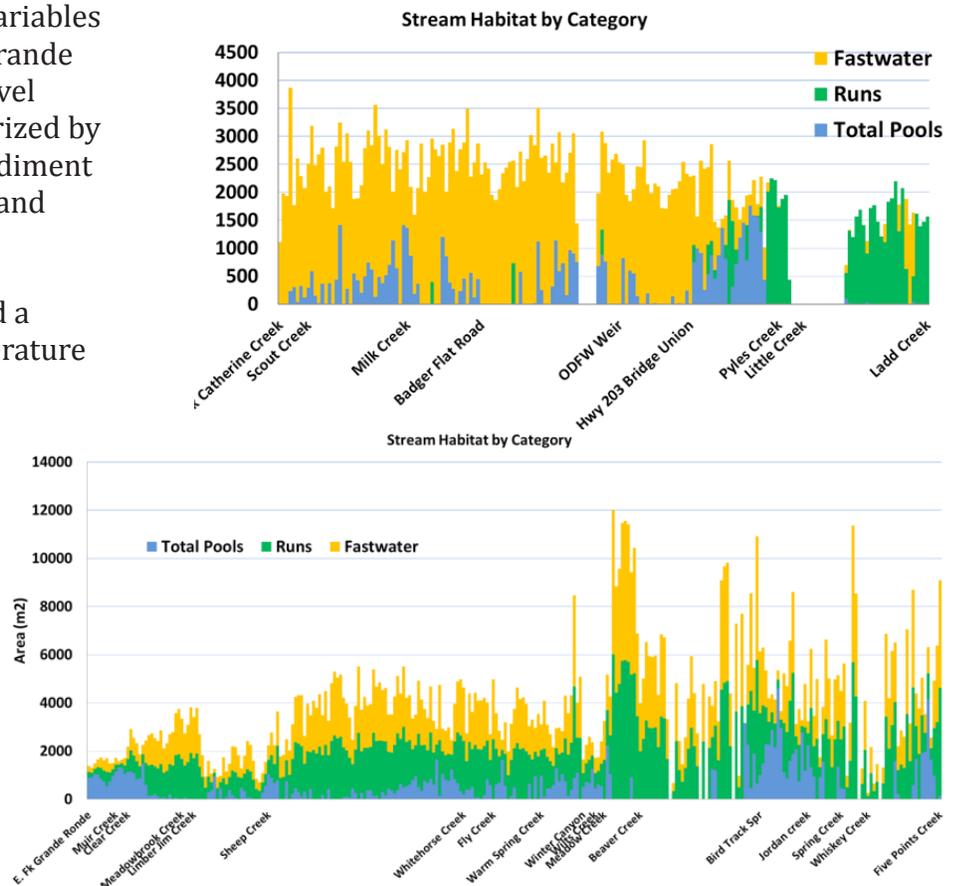
Figure 5.20. ODFW AQI survey results. (top) Catherine Creek. Stream categories by reach, Forks confluence downstream to Ladd Creek confluence. (bottom) Upper Grande Ronde River. Upper extent of spawning downstream to Five Points Creek.

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occurred within the first 25 years, with incremental reductions leveling off over time." Using an example provided in Justice et al. 2017, fully implementing the riparian restoration scenario in the upper Grande Ronde River would result in a potential reduction of 3.4°C at full canopy development (~300 years). A 2.2-degree reduction is projected for the first 25 years (65% of full canopy). Temperatures would be reduced by an additional 0.7°C between years 25 and 75 (reaching 85% of full potential reduction).

In addition to the effects of increased shading, restoring riparian conditions can also reduce stream temperatures through reductions in stream width towards estimated natural conditions (White et al. 2017). Reduced surface area translates into reduced solar heat flux into the stream over a given reach. We used the estimated potential for reduced stream widths projected in White et al. (2017) for large sections of the Grande Ronde and Catherine Creek as the basis evaluating restoration scenarios. The time period required for riparian restoration to result in

Figure 5.21. ODFW AQI survey results, stream sediment comopostion by reach. (top) Catherine Creek. (bottom) Upper Grande Ronde.

changes in stream width is a function of both the level of departure of current riparian from natural levels and the relative degradation of the stream structure. In some cases, restoring historical widths through natural processes may not be possible or would require many decades, for example in situations where low gradient channels have been widened through a combination of historical in-channel scouring (e.g., splash dam effects) and extensive loss of natural riparian restoration. In those cases, restoring potential natural stream widths in a reasonable time period would require direct channel reconstruction. In these analyses we assume that restoration of riparian habitats in designated high/moderate-priority reaches would result in stream widths returning to natural potential over a 15-year period through natural processes (or through direct intervention, where necessary).

Figure 5.22. Riparian restoration scenarios for Upper Grande Ronde and Catherine Creek populations. Justice et al. 2017, Figure 4.

5.4.3. Step 4: Developing historical, current, and strategy-specific restoration scenarios

The starting point for our analysis of tributary restoration scenarios were projections of population performance assuming that base period conditions within the tributary habitats of each population continue into the future. For Catherine Creek and the Upper Grande Ronde populations we also simulated the projected impacts of sequentially accounting for three additional levels of tributary habitat actions. This includes: 1) inclusion of 2009–12 habitat actions, 2) adding minimum target 2018–21 actions, 3) including current five-year planned actions (2019–24, Table 5.7), 4) a combination of actions to restore riparian habitats in the high/moderate priority reaches identified in Justice et. al (2017), and 5) flow and channel restoration actions called for in the 2017 NE Oregon Snake River Recovery Plan (Table 5.2). Longer-term restoration strategies for the Lostine/Wallowa population are under development through the ATLAS process and included in future LCM analysis.

We added another scenario to simulate the potential of additional habitat restoration downstream of current use to the Catherine Creek and the Upper Grande Ronde populations. The potential Chinook salmon increases from restoration in these downstream areas is currently limited due to distances from current spawning reaches and high temperatures. If the restoration scenarios described above result in a downstream expansion of current spawning and rearing, it is possible there would be a source of juveniles to utilize the relatively wide valley habitats below the area of current use. For the last scenario in the sequence, we assume that the current area production has been extended downstream sufficiently after 25 years. We assume that future restoration efforts would prioritize the areas downstream of current production.

5.4.3.1. 2009–16 tributary habitat actions

Catherine Creek habitat restoration actions implemented from 2009 to 2016 were designed to increase flows in a key rearing reaches, increase the amount of functional pool habitat through stream structure improvements, and restoration of floodplain side channel reconnections. Actions also included some riparian restoration in reaches high summer stream temperatures that currently impair or inhibit summer rearing. We reviewed and

Table 5.7. Catherine Creek Recovery Plan habitat actions.

Action	Upstream of Union	Downstream of Union	Implementation timeframe	Response timeframe
Flow restoration	2 cfs	10 cfs addition through reach	5 yr	Immediate increase in rearing pool habitat
Channel structure	Km44 project plus two more equivalent reaches	Restore 3 mi side channel and floodplain	Proportional over 15 yr	0–5 yr
Riparian restoration	High/moderate reaches	High/moderate reaches	Proportional over 20 yr	% of max. shading benefits: 40% at Year 25, 85% at Year 85

adopted the Grande Ronde Expert Panel assessments of the potential change in baseline conditions within Biologically Significant Reaches (BSRs) for incorporation into our LCM habitat effects analysis. The Expert Panel had characterized baseline conditions in each BSR using ODFW Aquatic Inventory survey data augmented by results from CHaMP studies in the basin. We used the same information to characterize current habitat conditions.

The focus of actions implemented from 2009 to 2016 was summer parr rearing capacity, which was identified as the most limiting life stage parameter. It is possible that after substantial habitat restoration efforts another factor (e.g., spawning capacity or overwintering capacity) could become limiting. Actions that improve conditions for summer parr rearing would also increase the capacity for spawning and overwintering capacity so it is not likely that benefits from improving summer parr rearing habitat would override other limitations. Baseline estimates of summer parr rearing were derived from analyzing the 20-year series of adult spawner and juvenile data sets available for Catherine Creek. We translated the impacts of actions to multipliers reflecting the proportional change from baseline habitat conditions. We assumed parr habitat capacities are a simple function of available pool habitat and prevalent stream temperatures. The actions implemented in Catherine Creek addressed five limiting factors directly related to parr rearing capacity: in-stream habitat complexity, bed channel and form, floodplain and side channel access and functionality and stream temperature. The actions are projected to reduce fine sediment levels in the targeted stream reaches. The BOR maintained summaries of the results of the Grande Ronde Expert Panel review of the projected changes in those habitat factors for the collective actions in each Catherine Creek BSR. We accepted those proportional changes and accumulated them into three categories: habitat changes that would be relatively immediate (1–5 years to take full effect), intermediate (10–15 years), or long-term (50–100 years). The Catherine Creek actions implemented between 2009 and 2016 primarily fell into the short-term category and included stream structure (LWD additions, pool construction), bed form enhancement (increased sinuosity), side channel/floodplain restorations, and flow additions (increased pool capacity).

We expect longer-term benefits to accrue from riparian restoration that would increase shading in moderate to high temperature reaches, as well as, restore natural channel widths and depths. The benefits of restoring flows by 10 cubic feet/second (cfs) were estimated using data from CHaMP sampling in the Union to Davis Dam reach and analyzed using the CHaMP Workbench HIS model (Figure 5.23, from C. Horn, Oregon Department of Fish and Wildlife, 2015 memorandum to ODFW). We express the effect of the action as a proportional increase in suitable pool habitat. The draft Recovery Plan also calls for restoring three miles of side channel or meander habitat. We assumed that reconnected or reconstructed channel habitats would occur in the same low gradient reach (Union to Davis Dam), and that the resulting additional channel habitat would average 80% pool frequency. We assumed these actions would increase the juvenile Chinook summer rearing capacity for the population, but that temperatures would not change from current ranges.

We express the proportional changes in population level parr capacity as a weighted percentage to illustrate the relative change from baseline. The actions producing relatively immediate habitat change result in an estimated 21% improvement in functional parr capacity. While the temperature reductions associated with shading would not fully

occur for several decades, we expect shading levels to start contributing to temperatures reductions after 5 to 10 years. By Year 25, the projected benefits of temperature reductions would further increase functional parr capacity by an additional 3–24% over baseline. Additional shading resulting from maturing riparian plantings are projected to further reduce temperatures at 48 years. The cumulative change in functional parr capacity would increase by 27% relative to baseline, an additional 3% increase from Year 25 to Year 48.
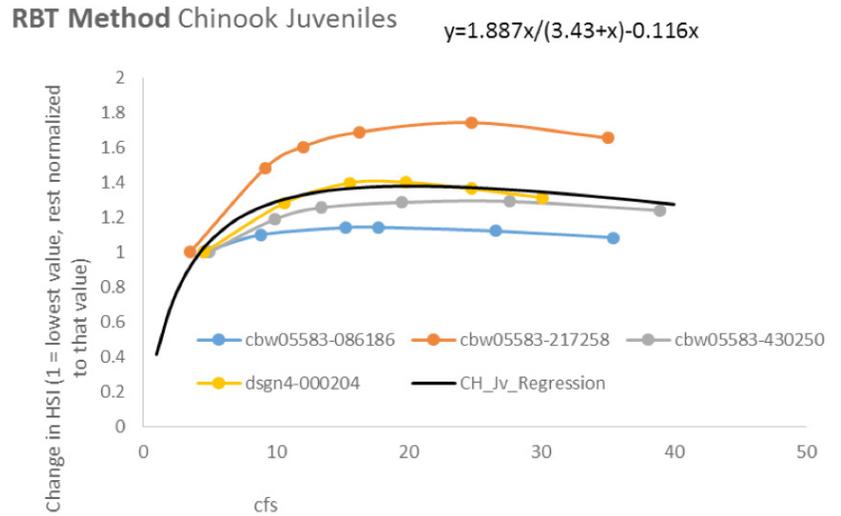
Figure 5.23. Catherine Creek Union to Pyles Creek reach. Estimated change in summer rearing capacity as a function of flow level generated from CHaMP sampling data using the CHaMP Workbench HIS modeling tool.

5.4.3.2. 2018–21 minimum action scenario

The action agencies have committed to pursue additional actions within the Grande Ronde MPG, targeting the same strategic priorities as in the prior Biological Opinion. While the action agencies are targeting higher levels of implementation, past experience indicates that several factors can result in unanticipated delays or require shifting actions among alternatives that are beyond their control. The action agencies have identified improvement targets for key habitat indicators for each major population group but have not provided specific proposed actions. For the purposes of this analysis, we assume that the targets would be achieved in the same populations that were prioritized in the 2000 Hydrosystem Biological Opinion tributary habitat strategy. Assuming that they accomplish the minimum levels of habitat improvement they identify over the three years, the estimated short-term benefits would increase by approximately 2%. Adding in the initial benefits of longer-term actions would increase functional parr capacity at 24 and 48 years to 26 and 37% relative to the original baseline.

5.4.3.3. 2019–24 ATLAS five-year action plan

Participants in the Grande Ronde ATLAS project have identified a series of projects in Catherine Creek and Upper Grande Ronde for implementation in the next five years (Table 5.2). The estimated changes in LWD, total pools and large pool habitats within each project area correspond to current (30–70%) project designs provided by project implementers. Those estimates were generated by summarizing available GIS data layers, and digitizing features (both historical and active channels) from current LIDAR and aerial imagery (Figure 5.24). We assumed that the estimated increase in main channel pool habitat
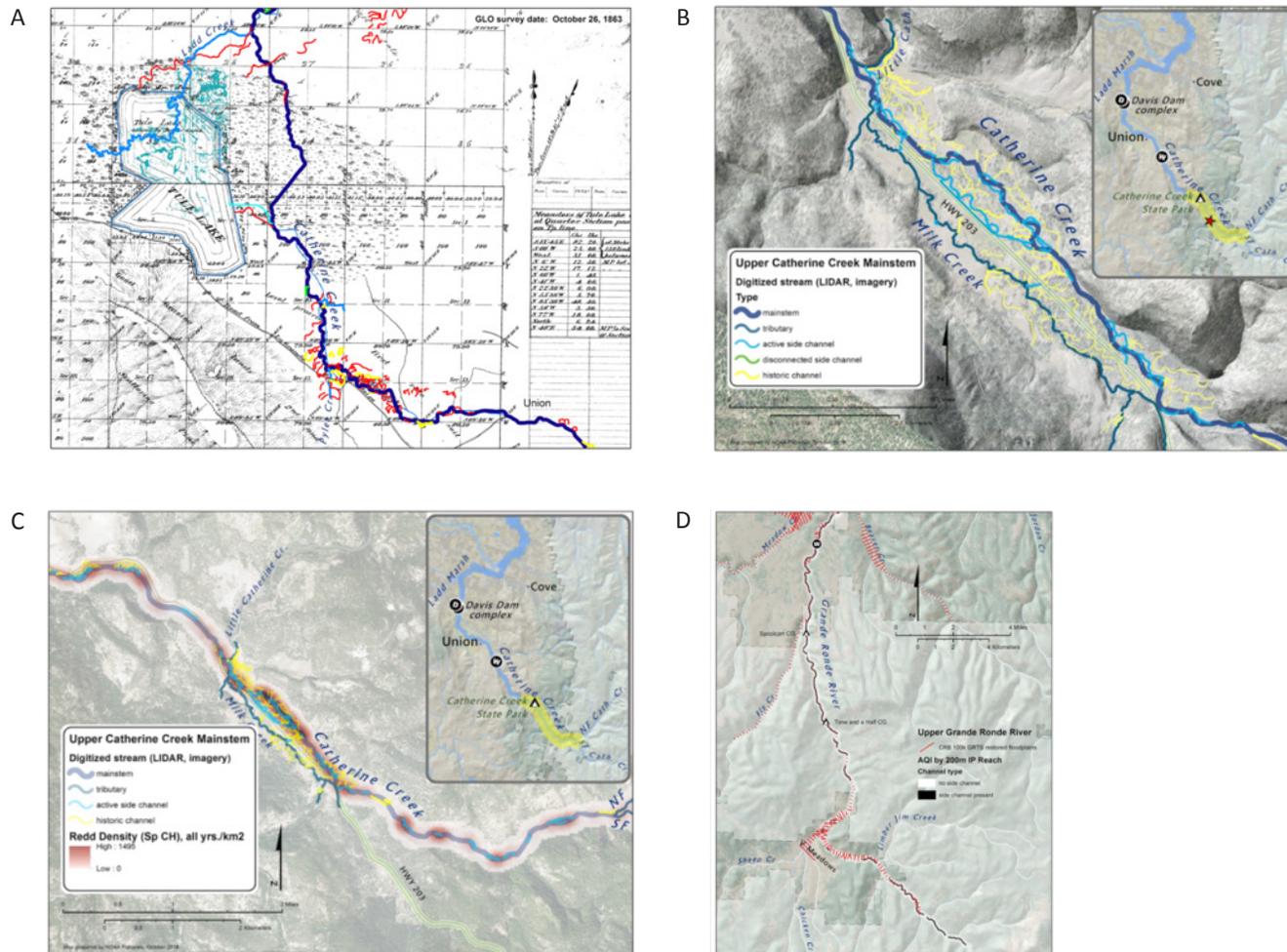
Figure 5.24 Catherine Creek. A) Ladd Marsh reconnection scenario: Current open-water habitat (blue) and potential reconnection pathway (red). Current mainstem overlaid on historical GLO map. Current and historical side channel traces derived from lidar and aerial photos depicted in yellow and red. B) Catherine Creek Hall Ranch project reach. Lidar-based estimates of current and historic side channel habitats. C) Catherine Creek Hall Ranch project reach. Current (2009–16) GPS redd locations vs. flooplain and side channel habitats. D) Upper Grande Ronde River. Upper Grande Ronde current side channel (black lines) and potential based on Bond et al. (2017) floodplain extent (red cross lines).

relative to the corresponding current Oregon AQI reach estimates represented proportional increases in the parr rearing capacity of the target reaches. For each Catherine Creek project, we assume that the estimates of increased pool habitat would be for the main channel and would represent a shift from current run and fastwater area for the target reach.

We made two simple assumptions to convert the linear meters of added side channel habitats projected for each project into increased juvenile rearing habitat. First, we multiplied the estimated additional side channel length by the average wetted width of mainstem habitat in the treatment reach. Second, we assumed that restored side channel habitat would contain 48% pool equivalent juvenile rearing habitat based on average side channel to mainstem information from other studies (Beechie et al. 2005, Goodman et al. 2010). We then applied the run, pool and fastwater proportions estimated from the 2010 and 2015 Oregon AQI surveys of side channel habitats (runs <0.01, total pools = 0.46, fastwater = 0.53). We summed the post-action estimates of reach level parr densities after applying a temperature weighting factor based on the NorWeST current (1993–2011) stream temperature extrapolations as described above, assuming that side channels would have the same stream temperatures as the adjacent mainstem reaches.

Several of the proposed actions include restored floodplain linkages. Previous studies, including several within the Grande Ronde basin, suggest that restoring natural floodplain function can have important benefits to rearing and spawning habitat conditions in associated stream reaches (e.g., Ebersole et al. 2003, Torgersen et al. 2012). It is likely that the combination of restoring floodplain connectivity, natural stream channel depths, and riparian habitats envisioned by several of the actions modeled in this assessment will lead to positive improvements in localized temperature conditions. Quantified estimates are not included in this analysis because there are no adequate methods for quantifying potential improvements resulting from floodplain reconnection based on projected conditions.

The proposed 2019–24 Catherine Creek projects primarily target restoring or enhancing stream structure and expanding side channel habitats to support summer rearing and spawning. All of the projects are in priority restoration reaches identified through the Atlas process (Tier I, either within current core spawning/rearing habitats or immediately downstream). Three of the projects are in the current core spawning and rearing reach above the current adult weir and juvenile screw trap sites upstream of the town of Union. The most extensive of these, the Hall Ranch project, would treat approximately 3.6 km of current mainstem habitat along with associated floodplain habitats (Figure 5.24 A, B) and would notably involve shifting the highway currently limiting mainstem sinuosity and side channel formation. Based on the projected changes in pool habitat for those projects, parr rearing capacity would increase by approximately 26% over baseline conditions associated with the adult and juvenile data series used to estimate life stage parameters in the Catherine Creek LCM. The majority of the increase was projected to result from restoring 2.7 km of side channels.

The remaining two are located downstream between Union and the Pyles Creek confluence, a reach that currently supports juvenile rearing but negligible spawning. Under these actions effective pool habitat in the Union to Pyles Creek reach would project to increase by 16% due to the main channel structure and side channel restoration. Habitat in the Union to Pyles

Creek reaches of Catherine Creek is currently degraded by current stream temperatures as well as by water withdrawals from May into Sep. The potential improvements in physical stream structure projected for this project would increase with proposed flow additions and with riparian restoration included in the 20-year habitat restoration scenario.

For the Upper Grande Ronde population, the proposed 2019–24 actions (Table 5.2) included mainstem channel and side channel restoration projects in three BSRs. Two projects in the East Fork reach (BSR 7) are intended to increase AQI pool equivalents by 17%, largely (95%) as a result of adding side channel habitat. Stream temperatures within this BSR are below the threshold of 18°C MWMT, resulting in no adjustment for temperature effects on parr rearing densities. Two additional BSRs support current spawning and rearing in the Upper Grande Ronde population. Sheep Creek (BSR 9) is a large tributary joining the mainstem Grande Ronde below BSR 7 in Vey Meadows. Actions to improve riparian habitats and to increase in-stream structure were implemented in 2009–16 and are accounted for in the past action inputs described above. At this stage of its development, the new project proposal for additional work in Sheep Creek does not have enough information to quantify potential effects on habitat for input into the life cycle model.

The 2019–24 proposals include two projects in the mainstem below the Sheep Creek confluence (BSR 5). This is also a designated Tier I reach. One of the projects, the middle Grande Ronde canyon reach proposal, would treat approximately 13 km of relatively confined mainstem habitat to increase pool habitat area and restore local floodplain function. The primary objective of the project would be to use placement of LWD to promote localized accumulation of gravels which would lead to increased pool habitat and floodplain function. At this stage in its development, there is insufficient information to translate this action into projected habitat changes for model input. The other proposed project in this BSR would treat a 2.4-km reach downstream of the canyon, increasing pool habitat through mainstem structural enhancement and side channel additions. Based on the estimated improvements in pool area, the project would increase potential parr density in the BSR by 11%, most of the increase resulting from projected side channel access.

Current estimates of stream temperatures in this BSR are relatively high, reducing the potential parr capacities by 40–60% relative to the 18-degree MWMT threshold we incorporate into the modeling analysis. Neither of the proposed projects in this BSR explicitly include riparian restoration during the 2019–24 implementation phase. Reducing stream temperatures by shading and channel effects associated with riparian restoration could substantially increase the potential parr density in this BSR over the projected increases modeled for the 2019–24 actions. The habitat in this BSR falls into the high/moderate priority reach category. As a result, the model projections under the 20-year high/moderate priority restoration scenario would include the combined effects of the proposed changes in pool availability and the potential for decreased temperatures through directed riparian restoration for this BSR.

The five-year action proposal includes two projects in the mainstem Grande Ronde downstream of current spawning and rearing. Current stream temperatures in the reaches targeted by these actions approach 25°C, estimated as a lethal threshold for Chinook juveniles. In addition, the two projects are well below the downstream extent of current spawning. The current project description for the Longley Meadows project is insufficient

to generate an estimated impact on habitat conditions. Projected impacts on pool and side channel availability are available for the Bird Track Springs project. While this project projects to increase available AQI pool equivalent habitat by 41% for the BSR, current temperatures result in negligible potential rearing in the reach or the BSR in general. The increased physical pool habitat would translate into increased juvenile capacity if stream temperatures can be reduced if additional riparian restoration in and above the target reach. Those reductions would also need to be sufficient to support a downstream extension of current spawning to serve as a source of juveniles. While it is unlikely that these projects would contribute to increased spawning/rearing capacities in the near future, there may be benefits to overwintering or outmigrating juveniles in the spring. At this point we do not have a sufficient understanding of the relationship of survival to local habitat conditions during those stages to quantify action effects.

5.4.3.4. 20-year habitat restoration scenario

It is important to put results of the habitat actions to be implemented in the relatively short time-frame of this biological opinion into the context of the effects of longer-term implementation of habitat actions. For instance, life cycle modeling for the Grande Ronde and Catherine Creek populations shows that long-term habitat restoration can have marked effects. To illustrate the potential benefits of continued implementation of potential strategic habitat actions, we modeled a 20-year implementation strategy designed to address the structural changes called for in the Snake River Recovery Plan combined with restoring riparian conditions to those reaches identified as moderate or high priority by Justice et al. (2017). We assumed the implementation would be accomplished at a consistent pace over the 20-year period.

For this scenario we assume that the longer-term Catherine Creek actions explicitly called for in the NOAA Recovery Plan would be implemented over a fifteen year period. In addition, we assume that the high and medium priority riparian restoration reaches identified in Justice et al. (2017) will be replanted at a constant annual rate over the next 20 years. Translating the projected impacts into proportional changes from baseline conditions, the recovery plan short and intermediate response actions would result in an 84% gain in parr habitat capacity by year 24. This increase includes the projected benefits of the 2019–24 in-stream actions described above. The initial responses to riparian restoration would increase that gain to a projected 125% improvement in parr rearing capacity by year 24. Benefits from increasing shading and restoration of natural stream channel characteristics would continue to accrue over time, reaching 165% over baseline conditions 48 years out. The benefits projected for the shading corresponding to fully mature riparian tree heights at approximately 100 years out would increase to approximately 206% of baseline.

5.4.3.5. Upper Grande Ronde population

Summer rearing habitat capacity is likely the most limiting life stage for Upper Grande Ronde population. The same habitat conditions that limit summer parr capacity (availability of large deep pool habitats, high summer temperatures) also impact adult holding/spawning usage. The primary actions implemented during the 2009–16 period were

aimed at restoring riparian habitat conditions. Based on the GR Expert Panel evaluations (Bureau of Reclamation[3]), in-stream complexity across the reaches currently supporting natural production would likely increase by approximately 1% over baseline conditions due to improvements in channel structure (LWD placement). The main focus of restoration efforts during this period was bank stabilization and riparian restoration. Benefits from the actions implemented 2009–16 were projected to contribute to increasing capacity through temperature reduction as shading levels increase. Based on the simple shade model outlined in Justice et al. (2017), functional parr capacity in the Upper Grande Ronde population would project to increase by 12% at Year 24, and approximately 20% by Year 48.

The 20-year continued habitat implementation scenario for the Upper Grande Ronde included a combination of active channel restoration, LWD placement and riparian restoration in reaches above Starkey identified as high or moderate priority by Justice et al. (2017). We summarized the potential changes in spawning/rearing effective capacity within BSRs. We assumed that LWD placement would address reach specific current vs. potential levels over the 20 year implementation period, expressing the results as a proportional increase in effective pool habitat. We assumed that the riparian restoration effort would be implemented at a constant rate over the 20-year implementation period. The modeled response was expressed as a change in effective pool habitat resulting from decreased temperatures and improved channel structure. Direct responses from temperature changes varied across the BSRs as a function of their respective current temperatures. The uppermost BSR (UGR 7: East Fork down to Meadowbrook confluence) exhibited current reach temperatures averaging below 18°C, the level above which relative chinook density begins to decline. The next downstream reach (Meadowbrook Creek confluence to Sheep Creek confluence) averaged 18°C. We assumed that the riparian improvement benefits projected in these two reaches (Justice et al. 2017) would be the result of improved channel/pool structure associated with restored natural riparian conditions. Current BSR average stream temperatures Sheep Creek and in the Sheep Creek to Warm Springs Creek confluence section of the mainstem Grande Ronde are at 20° and 21°C, respectively. We assumed that riparian restoration in these two reaches would increase parr production capacity through a combination of increased shading leading to reduced stream temperatures and corresponding natural channel restoration.

The long-term restoration scenario analyzed for the Upper Grande Ronde population included two components; targeted restoration of pool and side channel habitat in sections of the Grande Ronde Mainstem downstream to Warm Springs Creek (current spawning and rearing) and riparian restoration. The stream channel restoration component of the long-term scenario targeted reaches in wider valley settings classified as meandering using the Beechie index (Beechie and Imaki 2014). We used the Oregon Aquatic Inventory survey data in a gis format to quantify the current levels of pool, run, and fastwater area in 200m reach segments from the upper extent of spawning and rearing from the upper East Fork downstream to Warm Springs Creek. We estimated the median pool and riffle/run areas across the reaches classified as meandering and calculated the change in weighted AQI parr potential of doubling the proportion pools. We accounted for the reduction in fastwater habitat AQI parr potential in estimating the new total AQI parr potential (based on CHaMP sampling, fastwater habitats support approximately 20% of the potential for pool habitats).

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[3] https://www.usbr.gov/pn/fcrps/habitat/panels/2016results.html

We assumed that increased pool habitat would be accomplished by combinations of LWD placement and channel manipulation appropriate for each reach. In addition to the increased parr habitat, we also assumed that restored floodplain connections would result in adding side channel habitat equivalent in area to the associated mainstem reaches for the same meander class reaches. We applied the average side channel pool proportions from the Oregon AQI survey data (Catherine Creek surveys, average proportion of 0.48).

The riparian restoration component targeted reaches classified as high/moderate priority (257% increase, Justice et al. 2017). We discounted that total by 20% assuming that the Vey Meadows reach would not be available for restoration during the 20 year implementation period. The discount level was derived from earlier Heat Source model-based sensitivity analysis that evaluated the impact on temperatures of leaving individual large contiguous sections of the Upper Grande Ronde unrestored (C. Justice, Columbia River Intertribal Fisheries Commission, 2014 memorandum to NMFS). We made some simplifying assumptions to model implementing sufficient riparian restoration to achieve the full increase as a result of actions implemented over a 20 year period. Key assumptions included: a constant rate of implementation (5% of high/moderate priority reach habitat addressed per year); riparian function for a given treated reach would increase over time consistent with the rate of shade development calculated in Justice et al. 2017; riparian habitats lost to grazing, flood scouring, etc., would be replaced. The 20-year restoration strategy also included an assumption that LWD placement would continue to occur targeting the remaining high and moderate priority reaches. We used Expert Panel estimates of current vs. optimum LWD densities (they used comparable reaches in the Minam River as a reference for optimum). Average deficits across Upper Grande Ronde BSRs varied from 35–47%. We assumed LWD placement would reduce LWD deficits in target reaches by 50% in each BSR (excluding the Vey Meadows reach) and that would translate into a proportional shift from fastwater habitat to pool habitat over a five-year period. The projected increases in parr production potential from implementing the tributary habitat improvements from the Upper Grande Ronde 20-year restoration scenario at 24 and 48 years would be +99% and +140%, respectively. We generated results for a variation on the 20-year tributary habitat scenario by also including an improvement in Grande Ronde Valley migration survival of 50% under the assumption actions would be identified and implemented to reduce mortalities to the same levels as experienced by the two downstream populations (Lostine and Minam Rivers). We assumed those improvements would happen over a five-year period beginning in Year 15. Adding in the potential increase in survival gained by successfully addressing the high Grande Ronde Valley outmigration mortality would project to increase the cumulative improvements at 24 and 48 years to 199% and 262%.

5.4.3.6. Downstream of current use scenario

We generated an additional scenario for both populations to illustrate the potential for further expansion of natural production into reaches below current spawning and rearing that are currently precluded by loss of historical rearing habitat and extremely high summer temperatures (Upper Grande Ronde) along with reduced summer flows (Catherine Creek). In both cases restoring production to these lower reaches would almost certainly require successful restoration of the upstream reaches targeted in the 20-year scenario in order to extend spawning downstream enough to generate juveniles to use newly restored habitat below current spawning/rearing range.

For Catherine Creek, the downstream scenario we modelled assumed that access to available deeper water habitats in Ladd Marsh that are currently isolated from the artificially redirected Catherine Creek channel could be reconnected (Figure 5.24 C). In addition, sufficient flow would need to be restored to the reach to ensure that access and egress for juvenile Chinook would be maintained.

Based on GIS analysis, the surface area of open water areas in Ladd Marsh that could potentially support juvenile rearing is approximately 49 hectares (Holzer, unpublished). Based on reported values in the literature (Bartz et al. 2006), expected juvenile Chinook densities in moderately deep marsh habitat would be approximately 37% of mainstem pool habitats. Applying that proportion, adding Ladd Marsh would ultimately increase available juvenile rearing habitat by an additional 75% over the levels projected for the long-term restoration scenario.

For the Upper Grande Ronde, the Phase II long-term scenario targets riparian restoration along with channel and floodplain restoration work in the Starkey to Spring Creek reach. Current temperatures in this reach are high but potentially responsive to riparian restoration (Justice et al. 2017). The floodplain widens considerably in this reach (Figure 5.24 D). There are existing pools and side channels, but the habitat has been substantially degraded due to historical splash dam impacts and riparian habitat loss (White et al. 2017).

5.4.3.7. Lostine/Wallowa population

Development of intermediate (e.g., five-year) and long-term priority habitat restoration scenarios is underway, but not sufficiently complete to incorporate into the current LCM analysis. Previous habitat assessments have highlighted substantial opportunities for restoration benefits in this population, especially in the mainstem Wallowa River. We are continuing to work with ATLAS project participants to develop restoration scenarios for future LCM assessment. We were able to model the incorporation of an approximate 3% improvement in parr rearing potential for the actions implemented as a result of the 2014 Biological Opinion mitigation actions.

5.4.3.8. Grande Ronde Valley outmigrant survivals

Although there is strong evidence for high mortalities associated with spring movements of smolts (both natural-origin and hatchery releases) through the Grande Ronde Valley upstream of Rinehart Gap, the proximate causes are currently not understood (Favrot et al. 2018). Recent studies have suggested that one possible mechanism, floodplain and oxbow stranding, is not a significant source of mortality. Four other hypotheses were identified:

1. Excessive energetic costs result from high spring velocities in the bermed channels throughout the reach.
2. Disrupted migration cues result from the state ditch rerouting of the mainstem Grande Ronde.
3. Reverse flows upstream of Rinehart Gap result from the rerouting of the mainstem, impacting flow timing and accumulation.
4. Some combination of delays in migration timing are due to the flow changes and increased presence of northern pikeminnow.

To illustrate the potential impact of reducing mortalities in this reach, we included scenarios for Catherine Creek and the Upper Grande Ronde populations that assumed that downstream survivals would be improved to levels that would be the same as the average for migrants entering from the Minam and Lostine systems. Those two systems enter the Grande Ronde below Rinehart Gap.

5.4.3.9. Habitat capacity projections

The projected increases in juvenile rearing capacity for the range of scenarios run for the Catherine Creek and Upper Grande Ronde populations are depicted in Figure 5.25. The projections clearly illustrate some of the key assumptions behind the model inputs for habitat restoration actions. We assumed that each proposed action would be implemented proportionally over a 1–5-year time frame depending on the elements (LWD placement, moving a highway, etc.). Habitat responses to actions were also modeled using proportionate time frames (e.g., canopy development resulting from riparian replanting). The intent of this analysis was to generally contrast the potential magnitude of changes in habitat and associated changes in survival and production across a large range of habitat treatments. We recognize that this analysis does not capture the potential impacts of reach level variability in action implementation or habitat response.
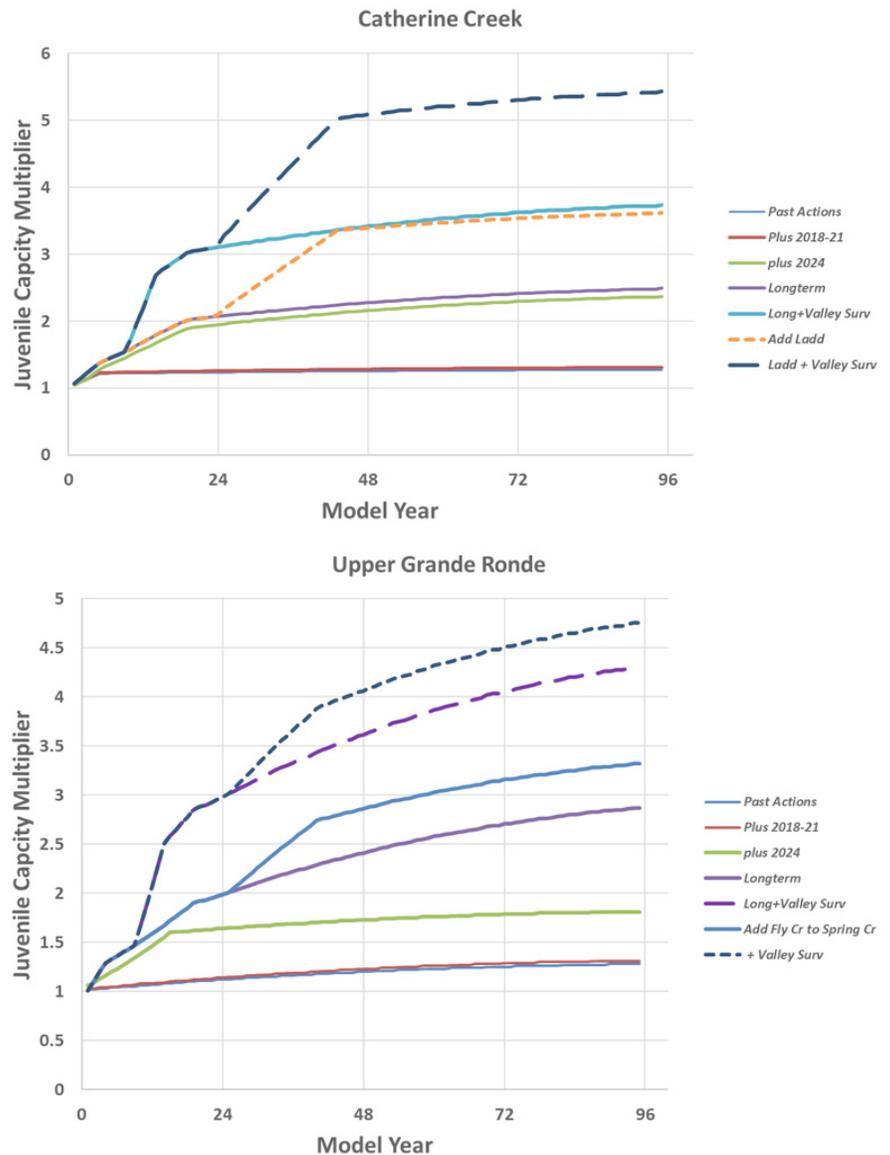
Figure 5.25. Projected proportional changes in juvenile rearing habitat under alternative habitat restoration scenarios. (top) Catherine Creek population. (bottom) Upper Grande Ronde population. Restoration strategies include past actions (2009–16) combined with sequential additions of minimum 2018–21, current 2019–24 proposed, 20-year tributary habitat restoration scenario, adding improved Grande Ronde Valley outmigration survivals, and implementing lower tributary actions after year 24 (i.e, Ladd Marsh reconnection in Catherine Creek, below Fly Creek restoration in Upper Grande Ronde).

For both populations the projected habitat response of implementing the proposed 2019–24 projects results in larger proportional increases than those associated with the past actions plus the minimum 2018–21 actions. The 2019–24 increases for Catherine Creek are proportionally larger, resulting in habitat capacity projections approaching the projections for full implementation of recovery plan stream structure and flow actions. The projected gain in juvenile habitat capacity for the Upper Grande Ronde for the long-term scenario (includes substantial additional riparian restoration) is large, reflecting the importance of reducing temperatures for this population (e.g., Justice et al. 2017). The trend lines for the long-term scenarios also reflect the assumed development rate of canopy cover and the resultant stream surface shading. Under the implementation assumptions modeled, both the 2019–24 and the long-term tributary habitat scenarios result in increasing capacity over the initial 24-year period, potentially increasing abundance and reducing short-term quasi-extinction risks. The effect of reducing outmigrant smolt mortalities to equivalent levels estimated for the Lostine and Minam populations is also substantial.

5.4.4.   Step 5: Using LCMs to evaluate differences in fish production among scenarios

To evaluate short-term effects, we focused on projected natural-origin abundance and the risks of going below quasi-extinction thresholds over the first 24 years. We also evaluated the projected 10-year median natural-origin abundance centered on simulation Year 75 as a measure of response to habitat actions with longer-term benefits (e.g., stream temperature benefits from riparian restoration). We summarized results over 500 iterations for each scenario to capture the impact of uncertainties in life stage parameters and annual environmental effects. The habitat scenarios were run under alternative assumptions regarding the potential impact of the increased spill hydropower regimes on latent mortality. For this summary, we focused on the proportional changes in quasi-extinction risks and natural-origin abundance across those latent mortality assumptions. The effects of the alternative latent mortality reduction assumptions are provided in the figures and tables.

Projected 24-year abundance and quasi-extinction risks differed across the five modeled Grande Ronde River basin spring Chinook populations (Tables 5.8 and 5.9, Figures 5.26 and 5.27). The box outline in each graphic illustrates the middle 50% of modeled outcomes across the 500 runs for each scenario, and the whiskers capture 95% of the outcomes.

The 2014 model scenario reflects average habitat conditions prior to the effects of actions initiated after 2009 and 2014 Biological Opinion hydrosystem operations. The 2018 environmental baseline scenario incorporates three updates: 1) changes to juvenile capacity and survival projected for tributary habitat actions implemented between 2009 and 2016, 2) increases in adult mortality in the Lower Columbia River coincident with a large increase in the abundance of marine mammals, and 3) changes to hydropower operations resulting from implementation of the 2014 federal hydropower system biological opinion. Projecting the impacts of the tributary habitat improvements forward results in a 14% improvement in natural-origin spawner abundance for Catherine Creek and a negligible change for the Upper Grande Ronde population. Adding continued natural stock supplementation resulted in a small reduction in median natural-origin spawners for each of the three populations. It is important to note that for the supplemented populations, adult returns from the natural-origin broodstock hatchery releases also contribute to spawning. For example,

Table 5.8. Projected 24-year natural abundance and quasi-extinction risks for alternative habitat restoration scenarios (5, 25, 50, 75, and 95 percentiles over 500 simulations) for the Catherine Creek population. 2018 Baseline scenario includes increased Lower Columbia predation rates, ongoing hatchery supplementation, and current mainstem harvest schedule. Habitat action scenarios are modeled under current (2018) proposed hydrosystem spill operations constrained by 120% gas cap. Habitat scenarios: 2020: 2018–20 actions at minimum annual rate; 2024: current Grande Ronde Model Watershed proposed 2019–24 actions; LT (long-term) Habitat Actions: 20-year implementation of high/moderate-priority reaches plus recovery plan actions; LT + DS (downstream survival) Habitat Actions: LT Habitat Actions plus improved valley outmigration survivals.

	Median abundance, Years 1–24					24-year risk, QET = 30					24-year risk, QET = 50				
	5%	25%	50%	75%	95%	5%	25%	50%	75%	95%	5%	25%	50%	75%	95%
2018 Baseline	91	118	140	165	206	0	0.002	0.004	0.011	0.044	0.013	0.058	0.139	0.269	0.609
Spill, 120% gas cap	91	117	138	165	204	0	0.002	0.004	0.011	0.037	0.013	0.052	0.117	0.252	0.574
2020 Habitat Actions	91	119	142	168	211	0	0.001	0.004	0.01	0.034	0.013	0.051	0.122	0.256	0.54
10% latent mortality	99	130	151	179	224	0	0.001	0.003	0.008	0.034	0.009	0.036	0.089	0.191	0.444
25% latent mortality	106	140	166	199	246	0	0.001	0.002	0.006	0.024	0.005	0.022	0.06	0.13	0.406
50% latent mortality	126	162	192	231	292	0	0	0.001	0.004	0.018	0.002	0.01	0.028	0.064	0.185
2024 Habitat Actions	108	140	165	199	251	0	0.001	0.002	0.004	0.016	0.005	0.019	0.051	0.112	0.328
10% latent mortality	115	148	179	211	267	0	0	0.001	0.003	0.015	0.003	0.014	0.036	0.082	0.247
25% latent mortality	125	165	196	235	293	0	0	0.001	0.003	0.012	0.002	0.008	0.02	0.053	0.183
50% latent mortality	149	190	226	276	346	0	0	0.001	0.002	0.008	0.001	0.004	0.011	0.026	0.087
LT Habitat Actions	111	146	175	210	268	0	0	0.001	0.004	0.015	0.003	0.015	0.038	0.098	0.254
10% latent mortality	121	155	188	227	285	0	0	0.001	0.003	0.017	0.002	0.011	0.029	0.066	0.234
25% latent mortality	136	175	206	246	319	0	0	0.001	0.002	0.012	0.002	0.007	0.018	0.04	0.153
50% latent mortality	154	201	241	292	370	0	0	0	0.002	0.008	0.001	0.003	0.008	0.021	0.076
LT + DS Habitat Actions	133	172	204	245	321	0	0	0.001	0.003	0.008	0.002	0.009	0.022	0.05	0.152
10% latent mortality	146	188	221	266	342	0	0	0.001	0.003	0.012	0.001	0.006	0.015	0.036	0.106
25% latent mortality	158	206	245	297	376	0	0	0.001	0.002	0.013	0.001	0.004	0.01	0.026	0.086
50% latent mortality	184	240	288	340	440	0	0	0.001	0.002	0.007	0	0.002	0.005	0.012	0.038

Table 5.9. Projected 24-year natural abundance and quasi-extinction risks for alternative habitat restoration scenarios (5, 25, 50, 75, and 95 percentiles over 500 simulations) for the Upper Grande Ronde River population. 2018 Baseline scenario includes increased Lower Columbia predation rates, ongoing hatchery supplementation, and current mainstem harvest schedule. Habitat action scenarios are modeled under current (2018) proposed hydrosystem spill operations constrained by 120% gas cap. Habitat scenarios: 2020: 2018–20 actions at minimum annual rate; 2024: current Grande Ronde Model Watershed proposed 2019–24 actions; LT (long-term) Habitat Actions: 20-year implementation of high/moderate-priority reaches plus recovery plan actions; LT + DS (downstream survival) Habitat Actions: LT Habitat Actions plus improved valley outmigration survivals.

	Median abundance, Years 1–24					24-year risk, QET = 30					24-year risk, QET = 50				
	5%	25%	50%	75%	95%	5%	25%	50%	75%	95%	5%	25%	50%	75%	95%
2018 Baseline	35	47	57	67	86	0.085	0.39	0.653	0.891	0.982	0.763	0.975	0.994	0.999	1
Spill, 120% gas cap	35	45	57	67	85	0.097	0.412	0.684	0.902	0.985	0.774	0.976	0.995	0.999	1
2020 Habitat Actions	35	47	57	67	87	0.089	0.409	0.676	0.883	0.983	0.789	0.976	0.995	0.999	1
10% latent mortality	38	51	62	73	93	0.062	0.261	0.542	0.811	0.965	0.638	0.948	0.988	0.998	1
25% latent mortality	43	55	67	80	102	0.032	0.154	0.382	0.689	0.93	0.422	0.881	0.971	0.995	1
50% latent mortality	51	65	78	92	117	0.01	0.067	0.183	0.41	0.797	0.17	0.701	0.899	0.977	0.998
2024 Habitat Actions	37	50	60	73	94	0.045	0.266	0.54	0.811	0.976	0.593	0.949	0.989	0.998	1
10% latent mortality	40	54	65	77	99	0.031	0.188	0.413	0.705	0.943	0.435	0.908	0.979	0.996	1
25% latent mortality	47	62	72	86	112	0.015	0.096	0.247	0.516	0.857	0.226	0.785	0.942	0.986	0.999
50% latent mortality	53	71	84	99	129	0.006	0.035	0.1	0.281	0.743	0.068	0.457	0.794	0.949	0.997
LT Habitat Actions	44	60	75	88	117	0.009	0.075	0.192	0.502	0.913	0.125	0.747	0.92	0.988	0.999
10% latent mortality	49	67	81	96	127	0.007	0.046	0.126	0.369	0.85	0.081	0.544	0.846	0.97	0.999
25% latent mortality	57	75	91	105	141	0.003	0.022	0.066	0.215	0.622	0.036	0.339	0.672	0.918	0.994
50% latent mortality	66	87	104	123	162	0.001	0.009	0.025	0.08	0.367	0.01	0.12	0.353	0.728	0.973
LT + DS Habitat Actions	50	72	87	102	139	0.004	0.029	0.08	0.241	0.786	0.04	0.406	0.735	0.942	0.998
10% latent mortality	57	77	93	112	154	0.002	0.019	0.057	0.183	0.597	0.017	0.242	0.614	0.904	0.992
25% latent mortality	65	88	106	125	163	0.001	0.009	0.024	0.081	0.42	0.01	0.111	0.329	0.747	0.981
50% latent mortality	75	102	122	146	190	0.001	0.003	0.011	0.033	0.173	0.003	0.03	0.139	0.426	0.907
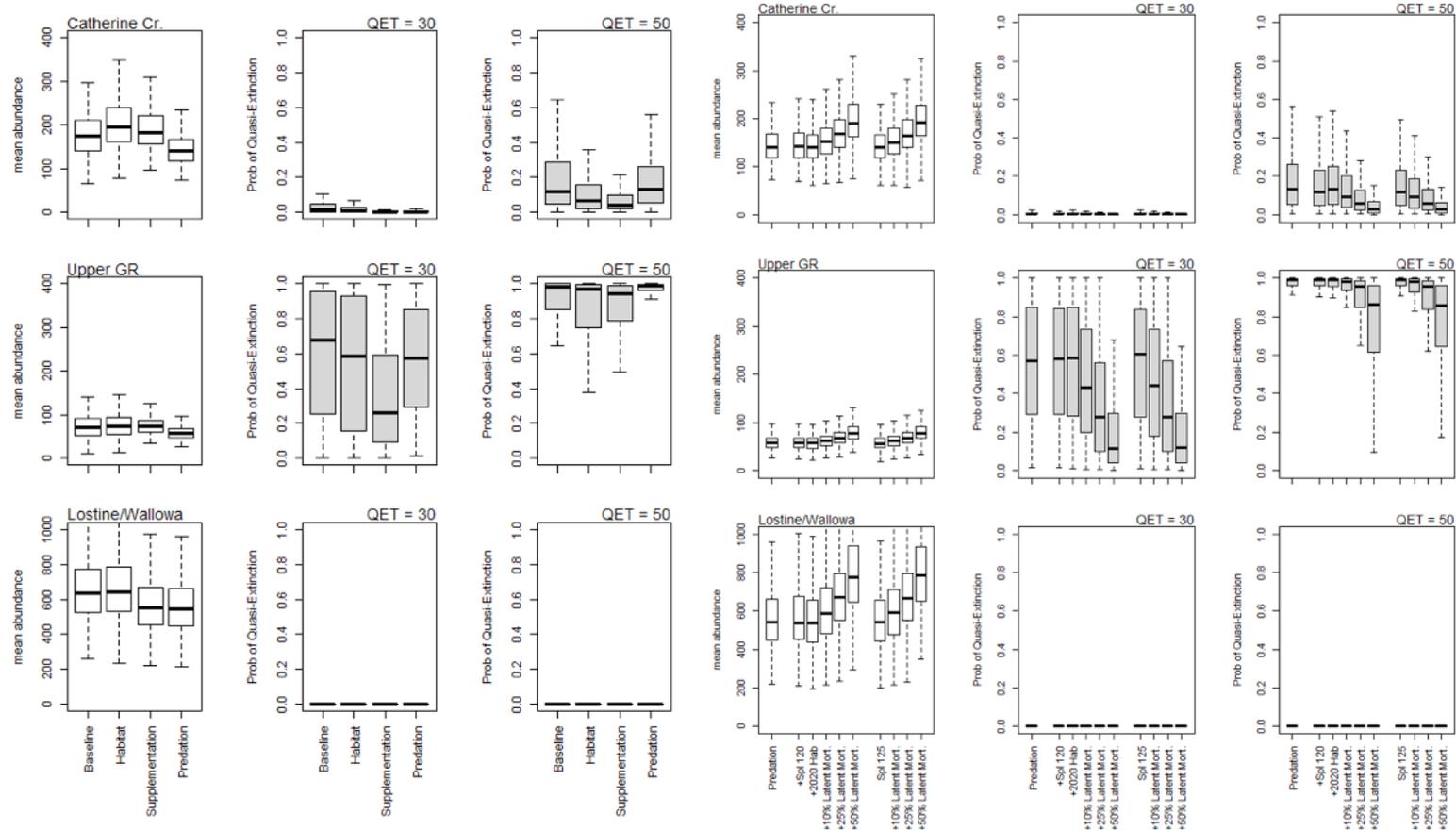
Figure 5.26. Twenty-four-year projected abundance and quasi-extinction risks for Catherine Creek, Upper Grande Ronde, and Lostine/ Wallowa River populations. (left) Baseline, past-action, and 2018–21 minimum actions scenarios. (right) Continued impacts of recent increases in marine mammal predation and hydrosystem gas cap spill operations under alternative latent mortality impact assumptions.
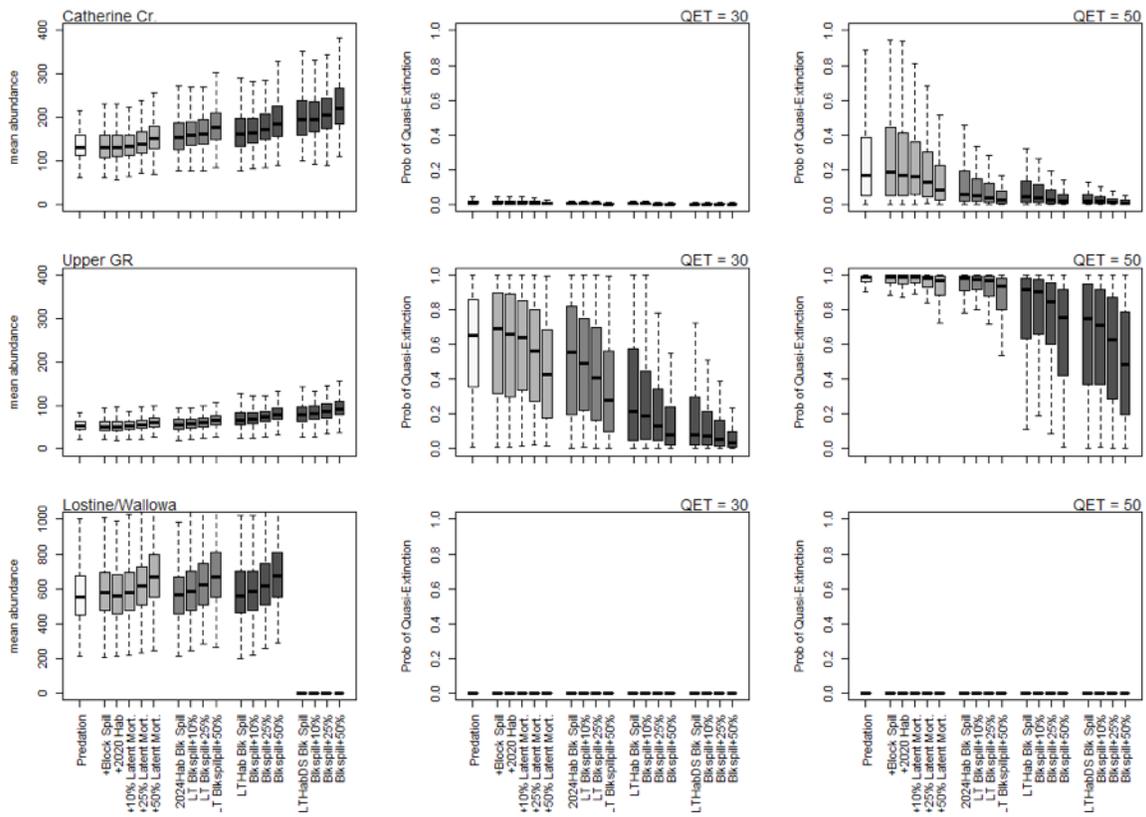
Figure 5.27. Twenty-four-year projected abundance and quasi-extinction risks for Catherine Creek, Upper Grande Ronde, and Lostine/Wallowa River populations for immediate (3–5-year) and long-term (20-year) restoration scenarios. White boxes: current baseline with effects of 2009–16 actions, continuing increased marine mammal predation, and ongoing natural stock supplementation. Light gray boxes: add minimum 2018–21 actions. Medium gray boxes: add proposed 2019–24 actions. Medium-dark boxes: add 20-year implementation of high- and medium-priority reaches. Dark gray boxes: add improved Grande Ronde Valley outmigration survivals. Each habitat scenario is run under four different assumptions regarding spill impacts on ocean survival. Note: 5- and 20-year high-priority strategies for the Lostine/Wallowa population are under development and are not included here.

the median projections for total spawners (natural-origin plus hatchery supplementation returns) increased to 306, 182 and 792 for Catherine Creek, the Upper Grande Ronde and the Lostine/Wallowa River populations (Figure 5.28). From a wild stock return perspective, incorporating supplementation into the model runs resulted in reduction in the risks of going below the 24-year quasi-extinction thresholds for both the Catherine Creek and Upper Grande Ronde River populations. Modeling the addition of the 2009–16 habitat actions and the continuation of current natural stock supplementation programs further reduced extinction risks for the Catherine Creek and Upper Grande Ronde populations. The net impact of all three factors is projected to decrease average abundance by approximately 20% for each population. 24-year risks of going below QET dropped to 0.0–4.0% and 0.1–61% for QETs of 30 and 50, respectively. For the Upper Grande Ronde population, accounting for the effects of 2009–16 habitat actions resulted in a modest reduction to a QET 30 risk of 8.5–98.1%. The risk of going below a QET of 50 over the next 24 years remained very high (76–100%).
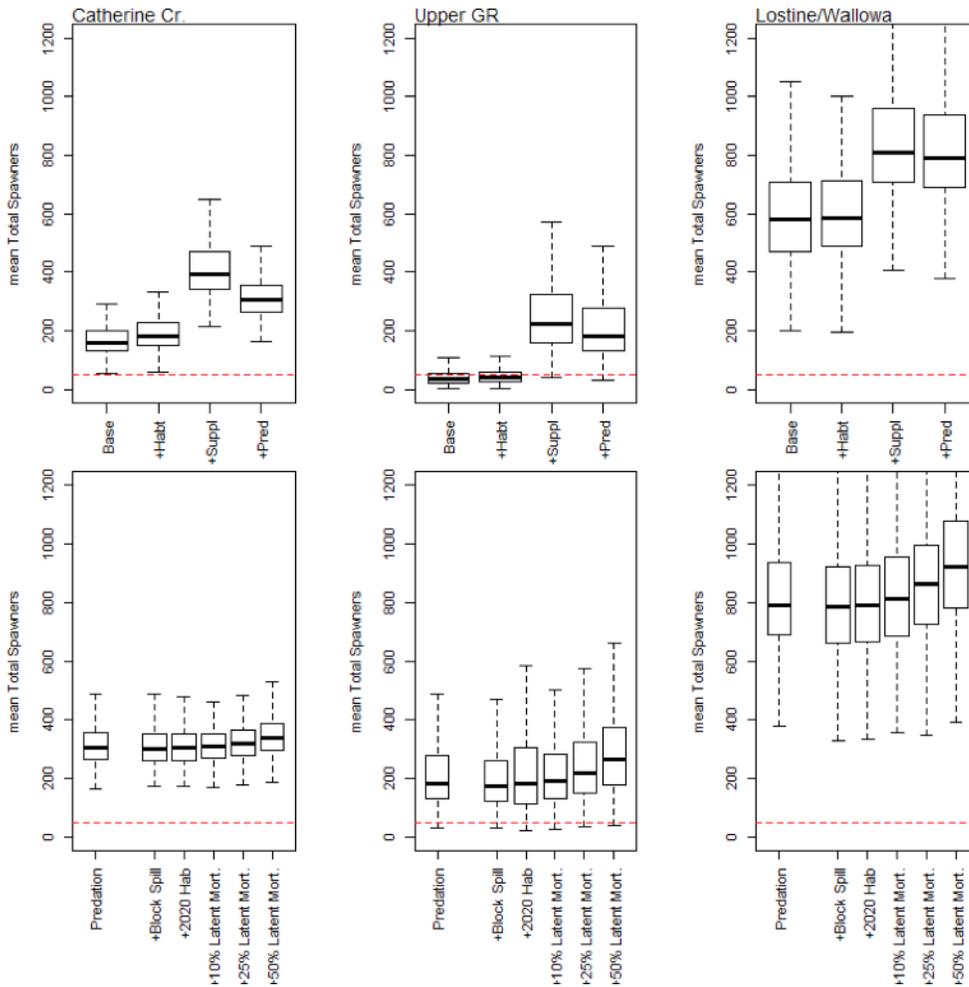
Figure 5.28. Twenty-four-year projected total spawner abundance (natural-origin plus hatchery supplementation returns) for Catherine Creek, Upper Grande Ronde, and Lostine/Wallowa River populations. (top) 24-year average returns for current baseline scenarios. (bottom) Projected returns for the hypothesized "block spill" hydrosystem operation, plus minimum habitat scenarios.

The 20-year habitat restoration strategies modeled for Catherine Creek and the Upper Grande Ronde River populations incorporate both an implementation and a habitat response time frame. Reducing stream temperatures is an important priority identified for habitat restoration actions in the Grande Ronde populations. Restoring riparian canopies associated with high priority reaches is a major mechanism for reducing temperatures. The benefits of increased shading will accrue over several decades as replanted riparian vegetation matures. We evaluated the longer term habitat restoration strategies over the initial 24-year period to estimate potential impacts on short-term abundance and risks of dropping below QETs. To capture the longer term benefits, we summarized the results across 500 runs for each long-term scenario at Year 75 to capture the cumulative effects over time (Figure 5.29). We realize that there is considerable uncertainty about the applicability of the environmental variation assumptions when extended out 75 years, but the projects provide a means of indexing the relative effects of the alternative habitat under

common sets of environmental assumptions. Figure 5.29 summarizes the projected natural-origin spawners at Year 75 for three of the longer-term habitat scenarios: 1) 2019–24 actions, 2) 20-year habitat and riparian high/moderate, and 3) Scenario 2 plus restoration below current spawning/ rearing. Both Catherine Creek and Upper Grande Ronde populations are subject to recent increases in adult survival losses in the lower Columbia River attributed to increased marine mammal predation. The first three scenarios were run assuming the recent year increases are maintained into the future. The fourth scenario depicted in Figure 5.29 assumes that the survivals in the lower Columbia return to base period levels as a result of reduced marine mammal predation. Each of the four scenarios were run under the same set of hydrosystem operations assumptions as the 24-year runs to illustrate the combined impacts of habitat and hydrosystem actions, and all included continuation of the current natural-stock supplementation program and the sliding-scale management schedule.
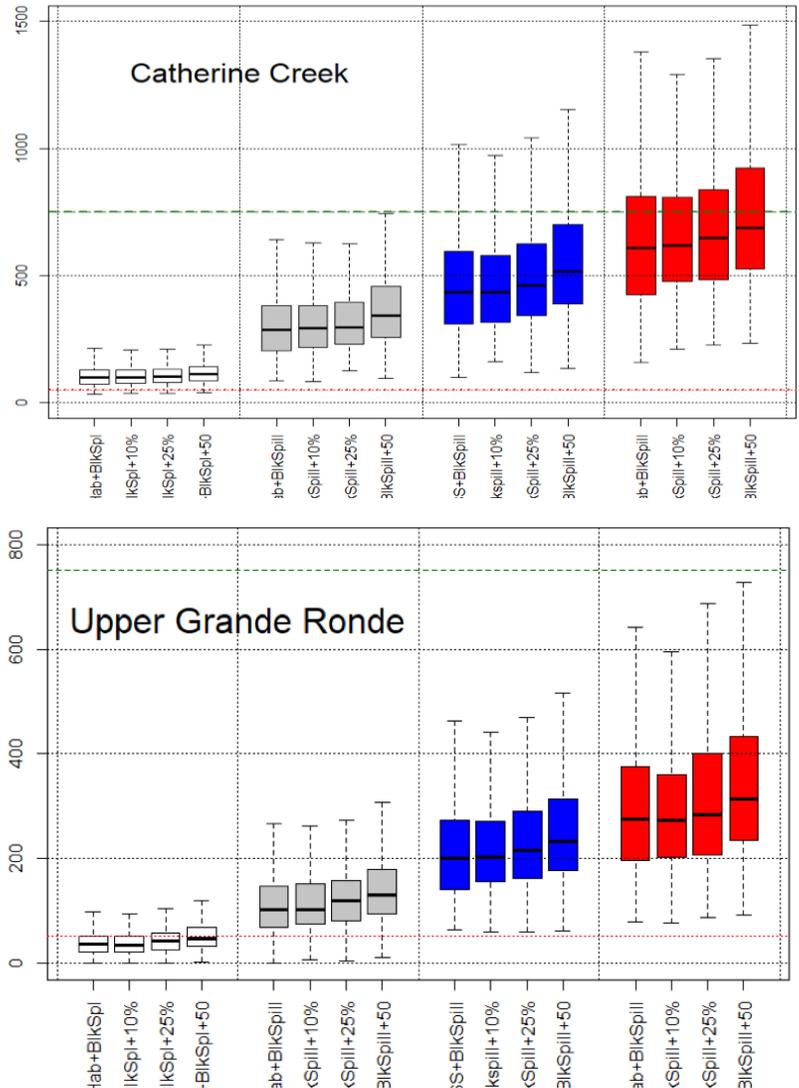
Figure 5.29. Twenty-year restoration scenarios, including restoring riparian habitats in high- and moderate-priority reaches identified in Justice et al. 2017. All runs are based on the hypothesized "block spill" hydro operation. Four possible latent mortality impact assumptions are illustrated for each habitat scenario. (top) Catherine Creek. (bottom) Upper Grande Ronde.

The general pattern of projected increases in abundance with increasing levels of habitat implementation were similar for the populations, as was the response to reduced lower Columbia River mortality. Although the full benefits of implementing the riparian area restoration strategies do not accrue for decades, the initial gains in shading associated with canopy growth did translate into increasing abundance and decreased QET risks projected for the initial 24 years. As would be expected, in each case the most substantial proportional

increase was associated with going from the recent five-year implementation to the 20-year continued habitat action scenarios (Tables 5.9 and 5.10). The range of assumptions regarding potential latent morality reductions resulting from decreased exposure to powerhouse effects in the hydrosystem varied across the scenarios, but generally ranged from 0–19%.

Projected natural-origin abundance under the 20-year habitat restoration scenarios continued to increase past the initial 24 years in response to improving temperature and stream structure. Model projections of 10-year geometric mean abundance centered on model Year 75 increased incrementally across the long-term habitat scenarios (Table 5.10, Figures 5.27 and 5.29). Under the 2024 habitat action plan scenario, the model runs for Catherine Creek projected a large proportional response (+63% relative to the 2018 baseline projections). Projections for the 2024 Upper Grande Ronde scenario were less than 10%. The difference can be explained by the larger emphasis in this strategy on channel restoration in reaches that currently have temperatures conducive to juvenile rearing in the Catherine Creek population. The 20-year high-priority tributary habitat scenarios for both populations projected to double natural-origin abundance for both populations. Addressing outmigration mortality in the Grande Ronde Valley doubled projected natural abundance again for both populations, resulting in 204% and 209% increases for Catherine Creek and Upper Grande Ronde, respectively. Improved spawning and rearing conditions in the downstream sections of current use resulting from the long-term actions opens up opportunities to further extend production downstream. For the Upper Grande Ronde population, the cumulative impact of the long-term habitat scenario combining expansion into reaches downstream of Fly Creek, reduced Grande Ronde Valley migration mortality and returning Lower Columbia marine mammal mortalities to pre-2013 averages resulted in a 525% projected increase. The corresponding scenario for Catherine Creek resulted in a median proportional improvement of 527%.

Based on the distributions of projected abundance across the 500 replicates, the 75-year projected natural abundance estimates for scenarios including 20-year high-priority habitat implementation improved Grande Ronde Valley outmigration survivals and high latent mortality responses to spill resulted in exceeding a threshold of 750 spawners in 11–32% of the model projections. Adding Ladd Creek habitat restoration increased the proportions exceeding 750 to 22–83% under alternative latent mortality reduction assumptions. Combining that habitat restoration scenario with a return to pre-2013 Lower Columbia River predation levels increased the proportions of runs exceeding 750 to 70–99%. While the increases in projected natural-origin returns were substantial for the Upper Grande Ronde population, only the combination of all habitat actions with reduced predation and high latent mortality response resulted in any projected 75-year abundance estimates above 750 (7% of that scenario replicates). Twenty-year restoration strategies for the Lostine Wallowa population have not been fully developed at this point. Previous studies have highlighted this population as having the highest restoration potential among spring Chinook production areas in the Grande Ronde River basin (Mobrand and Lestelle 1997). Extending the LCM analyses to cover specific five- and 20-year habitat restoration strategies for the Lostine/Wallowa population would be a high near-term priority.

Table 5.10. Projected ten-year geometric mean natural-origin abundance at model Year 75 for long-term habitat restoration scenarios (5, 25, 50, 75, and 95 percentiles over 500 simulations). All scenarios include 120% gas cap spill and ongoing natural stock supplementation. Habitat action scenarios: 2024 Hab + Spill: current Grande Ronde Model Watershed proposed 2019–24 actions; LT Hab + Spill: 20-year implementation of high/moderate-priority reaches plus recovery plan actions; DSS + Spill: LT Hab + Spill plus improved valley outmigration survivals; Ladd Hab + Spill, Red Pred + Spill: include actions below current use areas initiated in model Year 25.

	Catherine Creek						**Upper Grande Ronde River**				
	5%	**25%**	**50%**	**75%**	**95%**		**5%**	**25%**	**50%**	**75%**	**95%**
2024 Hab + Spill	158	195	221	253	309	2024 Hab + Spill	29	45	55	64	80
Spill + 10%	175	210	238	274	338	Spill + 10%	36	51	61	70	87
Spill + 25%	198	236	268	309	383	Spill + 25%	44	60	70	81	100
Spill + 50%	226	274	318	374	470	Spill + 50%	59	74	84	96	118
LT Hab + Spill[a]	199	247	282	330	411	LT Hab + Spill[a]	65	90	108	125	155
Spill + 10%	224	269	308	356	450	Spill + 10%	74	102	121	140	176
Spill + 25%	249	302	352	414	510	Spill + 25%	90	120	138	160	195
Spill + 50%	291	368	427	488	604	Spill + 50%	114	144	167	192	235
DSS + Spill[b]	319	398	462	534	668	DSS + Spill[b]	106	143	167	194	237
Spill + 10%	347	443	505	583	719	Spill + 10%	120	159	185	213	271
Spill + 25%	412	491	562	660	811	Spill + 25%	144	183	209	246	299
Spill + 50%	468	586	668	790	965	Spill + 50%	174	219	252	294	361
Ladd Hab + Spill[c]	435	540	622	733	892	Ladd Hab + Spill[c]	168	219	254	296	355
Spill + 10%	476	588	678	800	985	Spill + 10%	190	240	277	324	397
Spill + 25%	539	672	771	907	1084	Spill + 25%	215	272	316	360	456
Spill + 50%	652	798	920	1058	1317	Spill + 50%	260	330	376	444	581
Red Pred + Spill[d]	579	732	838	984	1228	Red Pred + Spill[d]	226	289	341	392	510
Spill + 10%	633	802	912	1074	1362	Spill + 10%	250	314	361	445	554
Spill + 25%	732	909	1038	1215	1487	Spill + 25%	287	351	424	502	650
Spill + 50%	857	1058	1226	1416	1766	Spill + 50%	343	436	521	630	776

[a] Long-term.
[b] Downstream survival.
[c] Ladd Marsh reconnection project.
[d] Reduced marine mammal predation scenario.

The proportional increase in projected natural-origin spawner abundance over all scenarios was the greatest for the Upper Grande Ronde population. For the Upper Grande Ronde population, the cumulative impact of the long-term habitat scenario combining expansion into reaches downstream of Fly Creek, reduced Grande Ronde Valley migration mortality and returning Lower Columbia marine mammal mortalities to pre-2013 averages resulted in a 607% projected increase. The corresponding scenario for Catherine Creek resulted in a median proportional improvement of 529%. However, in absolute terms, the projected abundance for Catherine Creek showed the highest response. While none of the scenarios for either population resulted in more than a 50% chance of exceeding the core area minimum adult spawner threshold of 750, approximately 40% of the runs under the most optimistic scenario for Catherine Creek were above the target level.

Several simplifying assumptions were made in characterizing the potential effects of habitat actions within each of the restoration scenarios we analyzed. We assumed that actions within each Biologically Significant Reach (BSR) would target specific reaches where key factors (e.g., pool structure, riparian cover) were below optimal levels and that follow-up efforts would be taken to restore action effects that might be negated by future events (e.g., major storm events, riparian grazing). We also assumed that riparian restoration would be implemented on a scale that would result in a change in local equilibrium stream temperatures. That requires implementing actions that would affect at least two contiguous kilometers of stream.

The life cycle models assume that the current life history characteristics of each population, including the proportions of juveniles moving into downstream rearing areas in the early spring and in the late fall, would remain constant (i.e., would be drawn from the distributions derived from the >20-year juvenile monitoring studies in each population area). It is possible that each population could adapt to future changes in temperature conditions by changing some or all of these basic life history features. At this time, we do not have a basis for projecting any such changes.

The results described above were all run under the assumption that future variations in climate conditions in the tributaries, the mainstem Columbia River and the ocean would have the same characteristics as the baseline timeframe. The Upper Grande Ronde population is particularly vulnerable to projected increases in summer stream temperatures given that a relatively high proportion of current rearing (Sheep Creek confluence to Warm Springs Creek confluence) is subject to summer temperatures of 17°C or higher. Restoring riparian shading and natural channel form in this degraded reach would be an important hedge against potential climate change. Future climate change scenarios including alternative assumptions for ocean survivals are being developed. Running the Grande Ronde LCM models with those alternative climate scenarios incorporated will be a priority in the near future.

6. Estimating population-level outcomes of restoration alternatives in data-poor watersheds: An example from the Wenatchee River Basin focusing on a spring Chinook salmon population

We detail the process used to quantify biological benefits to Wenatchee River Basin ESA-listed spring-run Chinook salmon from freshwater habitat restoration actions completed from 2009 to 2015 in the basin (Figure 6.1). The process consisted of linking stream and watershed restoration projects completed during this time period to estimated changes in physical habitat. Specifically, we quantified changes in salmon habitat capacity which was used as inputs into a spring Chinook salmon life cycle population dynamics model. The life cycle model allowed us to understand the potential salmon population response as a consequence of the habitat restoration actions. We identified relevant projects, determined the extent of their benefits in terms of changes to fish capacity, and described how we reflected changes to juvenile fish capacity in a life cycle model.

6.1. Overview/Summary

For this analysis, we take advantage of a) a life cycle model developed for a spring-run Chinook salmon population in the Wenatchee basin, and b) a juvenile habitat capacity model. The life cycle model is a product of a collaboration between federal, state, and regional stakeholders, and it uses population-specific information to drive its parameterization and calibration. It was developed to evaluate alternative management actions, including actions centered on freshwater habitat changes as a consequence of restoration actions. We identified Wenatchee basin projects between 2009–15 that were located in areas that contributed to the production of spring-run Chinook salmon. Furthermore, our focus was on projects containing components that altered the landscape through physical geomorphic habitat changes. We chose the approach of translating habitat projects into potential biological benefits using an existing juvenile habitat capacity estimation method (Bond et al. 2019). We focused on physical habitat changes as a result of projects completed during this period that could be quantified into capacity changes in the juvenile summer
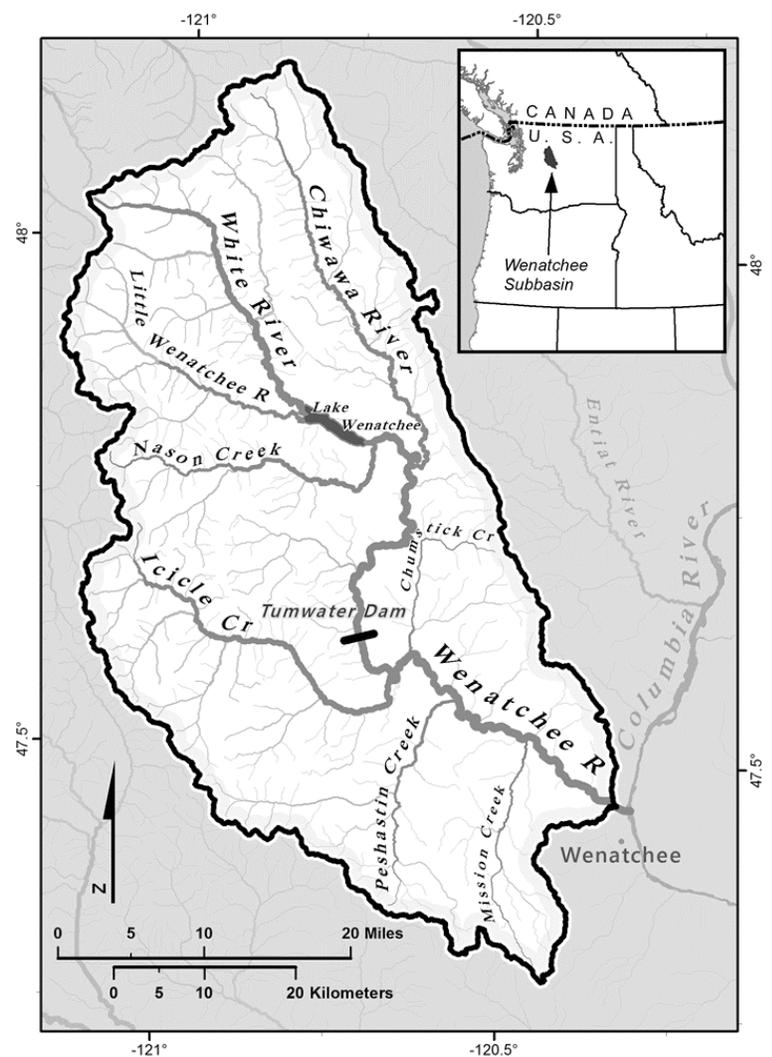
Figure 6.1. Map of the Wenatchee River Basin. Natural production of spring-run Chinook salmon occurs primarily in the main tributaries above Tumwater Dam: Chiwawa, White, and Little Wenatchee Rivers, and Nason Creek. Map by D. Holzer, NMFS/NWFSC.

parr rearing stage: an in-river wood enhancement project in the lower White River, and an oxbow reconnection in Nason Creek. A juvenile habitat capacity model (Bond et al. 2019) was used to estimate changes in capacity as a result of these habitat restoration actions. Analyses suggested that capacity increased by 1.1% and 1.4% in these tributaries, respectively, as a result of the habitat actions. When we introduced these capacity changes into the LCM there was no detectable difference between the scenario with no habitat change compared to the scenario with elevated capacity estimated from the habitat enhancement projects.

6.2. Wenatchee River spring Chinook salmon LCM structure

The Wenatchee River spring-run Chinook salmon life cycle model is an age-structured, stage-based, matrix-type population viability model with stochastic elements (ICTRT and Zabel 2006, Zabel et al. 2006; Figure 6.2). The following overview of the LCM comes from previous reports that contain more detailed information (Jorgensen et al. 2013, 2017).
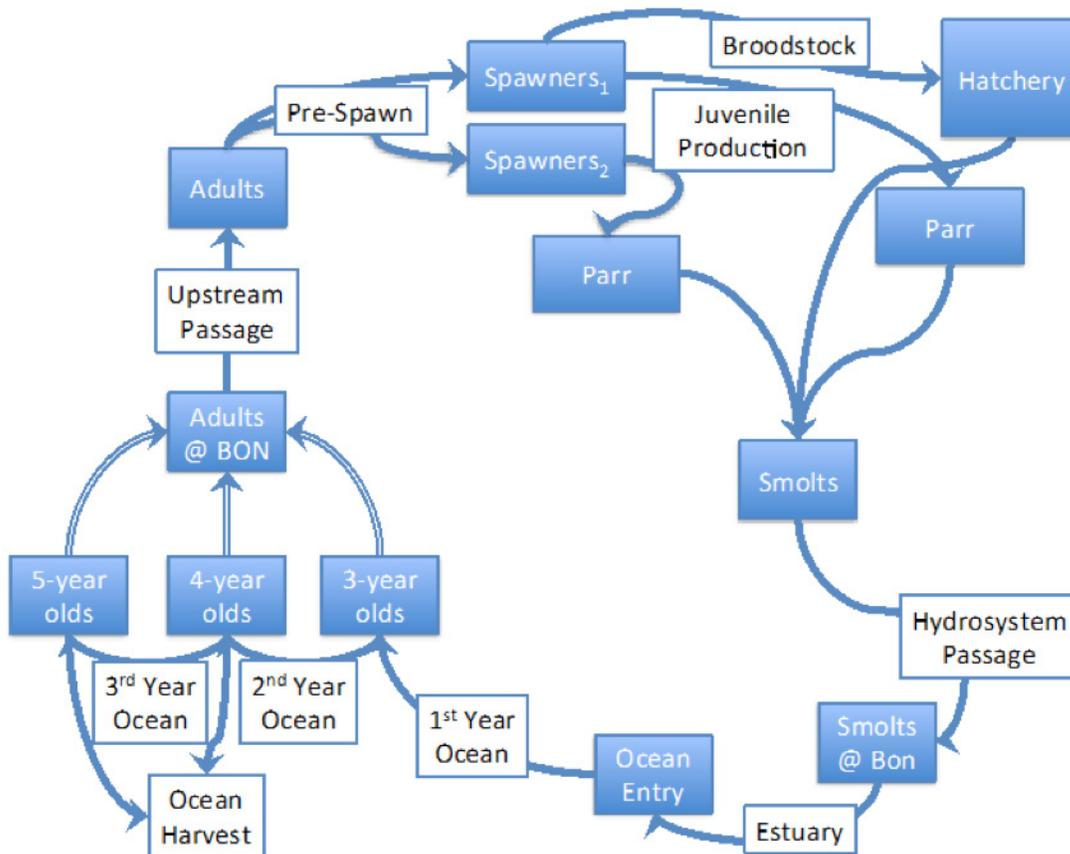
Figure 6.2. Diagram of the life stages included in the Wenatchee River spring-run Chinook salmon life cycle model. The "Spawners" boxes with numbered subscripts denote separate representation in the model of tributary fish production.

In principle, the LCM functions similarly to the traditional Leslie-style matrix structure (Leslie 1945). In this traditional formulation,

$$N(t) = \begin{bmatrix} n_1 \\ n_2 \\ n_3 \\ n_4 \\ n_5 \end{bmatrix}$$

This 5×1 abundance matrix tracks population numbers for five life stage classes across five ages: parr (n_1), smolts (n_2), ocean residence (from one to three years, n_3–n_5), and tributary spawners (four- and five-year-old fish that spent two and three years, respectively, in the ocean, n_4–n_5). The number of individuals at time ($t + 1$) is calculated by multiplying $N(t)$ by a 5×5 transition matrix, $A(t)$:

$$N(t + 1) = A(t) \cdot N(t)$$

The dimensions of the transition matrix, $\mathbf{A}(t)$, reflect the five age classes incorporated into the model, and its entries can change with t. The transition matrix, $\mathbf{A}(t)$, in this more simplified form looks like the following:

$$A(t) = \begin{bmatrix} 0 & 0 & 0 & b_4 \cdot s_A \cdot F_4(t) & s_A \cdot F_5(t) \\ s_2(t) & 0 & 0 & 0 & 0 \\ 0 & s_3(t) & 0 & 0 & 0 \\ 0 & 0 & (1-b_3) \cdot s_o(t) & 0 & 0 \\ 0 & 0 & 0 & (1-b_4) \cdot s_o(t) & 0 \end{bmatrix}$$

The model contains demographic parameters that govern transitions from one life stage to the next. The proportion of three- and four-year-olds leaving the ocean and returning to spawn (their breeding propensities) are noted by b_3 and b_4. Survival of adults from Bonneville Dam to the spawning grounds, s_A, is a product of upstream survival through the entire Columbia River mainstem dam system. Fertility is denoted by the F_t terms. s_2 is the survival probability of parr to the smolt stage (moving from one-year-old fish to two-year-olds), which includes rearing to the smolt stage and downstream migration through the dams to the estuary. $s_3(t)$ is the survival probability of the transition of fish from two- to three-year-olds, the period in which fish enter the estuary and ocean, corresponding to their first year of ocean residency. The s_3 term accommodates stochasticity and varies in time and according to scenarios of climatic and ocean conditions. The proportion of three- and four-year-old fish remaining in the ocean is given by ($1 – b_3$) and ($1 – b_4$). The s_o term represents the annual probability of ocean survival.

Our life cycle model comes from the simplified LCM form that was the basis for the ICTRT and Zabel (2006) model. However, we added a spatial dimension and the ability to track the effects of hatchery supplementation. To represent major fish production areas (Figure 6.2;

Jorgensen et al. 2013) as distinct entities with their own unique characteristics and to account for hatchery production, the abundance array, $\mathbf{N}(t)$, has a modified form to include fish production as discrete spatial units:

$$N(t) = \begin{bmatrix} n_{1,1} & n_{1,2} & \cdots & n_{1,j} & n_{1,h1} & \cdots & n_{1,hk} \\ n_{2,1} & n_{2,2} & \cdots & n_{2,j} & n_{2,h1} & \cdots & n_{2,hk} \\ \vdots & \vdots\ddots & & \vdots & \vdots\ddots & & \vdots \\ n_{5,1} & n_{5,2} & \cdots & n_{5,j} & n_{5,h1} & \cdots & n_{5,hk} \end{bmatrix}$$

where each $n_{x,y}$ element reflects ages (row) of fish originating from a specific subbasin production area (column). Hatchery programs (subscript h) are included and tracked by program type and objective for up to k hatchery program types. Adults of natural and hatchery origin are collected for broodstock at Tumwater Dam to meet the hatchery programs' targets and objectives.

Because of the modification of the $\mathbf{N}(t)$ abundance array to account for tributaries contributing to fish production and to include production from the hatchery programs, the transition matrix is expanded with additional parameters. The additional parameters are applied to each subbasin, j, or hatchery, h, and which are, in some cases, the same and shared among the subbasins (e.g., maturation schedule, upstream survival, fertility, and hydrosystem and ocean survivals) and, in other cases, are different to capture the unique characteristics of a subbasin (e.g., unique characteristics of fish production areas) or hatchery objective. The LCM's spatial resolution is such that it includes parameters specific to the major contributing tributaries to the population, including the two tributaries with the habitat actions described above, White River and Nason Creek.

6.2.1. Inputs to the LCM

Here we provide some additional detail about some of the other parameters used in the life cycle model (Table 6.1).

6.2.1.1. Parr capacity

The NWFSC Watershed Program has initiated efforts to characterize summer parr capacity as a function of geomorphic habitat classes. We include a scenario where capacities are changed as a consequence of implementation of two habitat restoration projects.

6.2.1.2. Parr–smolt

The parr-to-smolt transition, s_2, includes three elements: parr–smolt overwinter survival (s_{ps}), migration survival through the PUD and federal dams to past Bonneville, and the potential for avian predation. Parr–smolt survival is drawn yearly from a distribution determined through a model parameter calibration routine. Hydrosystem survival is determined from the COMPASS model (Zabel et al. 2008). The model has the ability to accommodate potential avian predation management actions (see Jorgensen et al. 2013).

Table 6.1. Parameters used for the Wenatchee River spring Chinook salmon life cycle model for major production areas, which include the Chiwawa River, Nason Creek, and the White River.

Parameter	Chiwawa River	Nason Creek	White River
Spawner(t)-to-parr($t + 1$) Beverton–Holt a	353	328	154
Spawner(t)-to-parr($t + 1$) Beverton–Holt b	0.000298	0.005	0.005
σ^2_1	0.412	0.600	1.04
ϕ_1 (variance term)[a]	0.1	—	—
Parr–smolt survival[b]	Draw from a distribution	Draw from a distribution	Draw from a distribution
Hydrosystem survival	Scenario-dependent, driven by water year and linked to ocean conditions	Scenario-dependent, driven by water year and linked to ocean conditions	Scenario-dependent, driven by water year and linked to ocean conditions
s_3 (first ocean year)	Stochastic, scenario-dependent	Stochastic, scenario-dependent	Stochastic, scenario-dependent
s_0 (ocean survival for years after s_3)	Draw from a distribution	Draw from a distribution	Draw from a distribution
b_3 (propensity of 3-year-olds to breed)	0.046	0.046	0.046
b_4 (propensity of 4-year-olds to breed)	0.514	0.514	0.514
h_r (harvest rate)	0.09	0.09	0.09
s_{pin} (predation from pinnipeds)	Scenario-dependent	Scenario-dependent	Scenario-dependent
s_u (Bonneville-to-basin survival rate)	Draw from a distribution	Draw from a distribution	Draw from a distribution
s_{sb} (prespawning survival rate)	Draw from a distribution	Draw from a distribution	Draw from a distribution
Initial abundance of 4- and 5-year-old tributary spawners used to initialize the model (apportioned geomean of 2008–12)	406	148	38

[a] Chiwawa River production estimates included a Box–Cox transformation as a way to deal with the heteroscedasticity in the data (ICTRT and Zabel 2006, Zabel et al. 2006).
[b] Parr–smolt survival accounts for the period from the summer parr stage to the smolt stage upon exiting the Wenatchee River basin.

6.2.1.3. Ocean and pinnipeds

The ocean phase of salmon in the life cycle model encompasses estuary entry and life at sea. Once smolts pass Bonneville Dam, they reach the estuary and can spend a variable number of years at sea. Survival during the first year in the ocean (s_3) is estimated from a model fit to Wenatchee basin PIT-tagged natural fish detected at Bonneville Dam as juvenile outmigrants and as returning adults (smolt-to-adult returns, SAR), with marine indices and arrival timing of juveniles at the dam. We used a multivariate autoregressive modeling framework (`MAR(1)`, in the `MARSS` package in R; Holmes et al. 2012, 2013) that preserved covariance among the indices and their autocorrelation structure to construct SARs for LCM simulations (M. Sorel, University of Washington, unpublished data; ISAB 2017). First-year ocean survival was calculated by removing mortality estimated for subsequent ocean years from SAR. All subsequent survival in ocean years (s_O) is drawn from a distribution as determined through a parameter calibration process.

The maturation schedule to the adult stage is set by proportions of three- and four-year-old ocean fish returning to spawn (b_3, b_4). The model assumes that all surviving five-year-olds advance to the adult stage and return to spawn. Another important component of survival during this phase for Columbia River-bound adults happens when they pass through the estuary and up through Bonneville Dam. They are vulnerable to predation by pinnipeds (s_{pin}), from which the resulting mortality rates appear to have increased since 2012 (Sorel et al. 2017).

6.2.1.4. Upstream

Survival from Bonneville Dam to the mouth of the Wenatchee River (s_{up}) is drawn yearly from a normal distribution with a mean and variance estimated from recent observations of upper Columbia River PIT-tagged fish (Crozier et al. 2016). The impacts from ocean and Columbia River fisheries (h_r) are also accounted for during the upstream migration, which is set to a constant value of 9% during prospective model runs.

6.2.1.5. Spawners

Several life history events are applied in the life cycle model to adults that migrate upstream before becoming spawners on the spawning grounds. First, a small number of fish migrate upstream in the Columbia River and bypass the Wenatchee River, and some fish stray or disperse to nonnatal tributaries within the Wenatchee basin above and below Tumwater Dam (5% and <3%, respectively; A. Murdoch, WDFW, unpublished data). Those below Tumwater Dam are not considered to contribute to the population and are removed from the life cycle model. The rates of bypass and below-Tumwater dispersal are applied only to hatchery-origin returns (HORs) and can be attributed to several factors: they may be attracted to an earlier rearing location (the "Eastbank effect"), they may not be able to locate or may not have fully acclimated to their release site tributary, or other factors. Second, not all HORs are allowed to spawn in the wild. HORs are held at Tumwater Dam and the yearly number that are passed above and allowed to spawn is determined by annual proportionate natural influence (pNI) targets set by comanagers and by natural-origin return abundance, which govern year-to-year proportion of hatchery-origin spawner (pHOS) rates. Third, all fish that are on the spawning grounds experience some level of prespawn mortality (s_{sb}), which is drawn yearly from a distribution as determined through a model parameter calibration process.

6.2.2. Calibration

Before conducting prospective model runs we calibrated the LCM to recent observations of the population. After a review of an initial LCM calibration routine (ISAB 2017), we modified our procedure that includes simple Approximate Bayes Computation rejection-sampling method (Beaumont 2010, Csilléry et al. 2010, Hartig et al. 2011). In rejection-sampling, approximations of parameters' posterior distributions can be constructed with repeated parameter value-sampling and LCM simulation. Model outputs are compared to observations of recent (2005–14) estimates of spawner abundance (SPS Database[4]) and to smolt estimates (Murdoch, unpublished). Model iterations with parameter sets generating results most similar to observations are retained to form parameter posterior distributions. Parameter values are then drawn from the posterior distributions in the prospective simulation LCM runs.

6.3. Develop restoration scenarios: Habitat change analyses

6.3.1. Habitat projects

The Upper Columbia Salmon Recovery Board (UCSRB) maintains a listing of all upper Columbia River basin restoration projects in a searchable online database (Habitat Work Schedule[5]), and has verified project data and information in the database for projects completed up through 2017 (G. Maier, UCSRB, personal communication). The list of projects includes not only those directed at changing habitats, such as water diversion changes, riparian planting, blockage removals or repairs, and in-stream wood placements, but also other projects that do not directly or immediately manipulate habitat, such as conservation easements and reach assessments, which provide some indirect benefits to spring Chinook salmon and other important species such as ESA-listed steelhead and bull trout.

For the purposes of this report, we focused on projects from the Habitat Work Schedule completed between 2009–15 that were located in areas that contributed to the production of spring-run Chinook salmon. For example, we excluded projects that self-reported that they targeted spring-run Chinook salmon that were located in areas with little or no contemporary occurrence of spring-run Chinook salmon, such as Chumstick and Peshastin creeks. We did not consider effects of projects located in the mainstem Wenatchee River. While the mainstem is important for spring-run Chinook salmon, the focus of this study was to assess benefits of projects with respect to how they might address changes in juvenile rearing capacity in the major fish production tributaries. Currently, there is very limited spawning in the upper Wenatchee mainstem (Name, unpublished), and there is uncertainty about whether mainstem juvenile rearing capacity is limiting. In the absence of quantifiable evidence we assumed for this study that this area is not capacity-limited.

Further, we directed our focus to those projects containing components that altered the landscape through physical geomorphic habitat changes. Our intent was to capture changes to geomorphic features and translate the changes into changes in capacity. Conservation

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[4] https://www.webapps.nwfsc.noaa.gov/sps
[5] http://hws.ekosystem.us/search

easements and land purchases to prevent further development—projects designed to protect intact habitats—are important for the preservation of existing functional habitats; however, these types of projects fell outside of the domain of this study.

Given our approach to ascribe project benefits based on geomorphic changes and given the 2009–15 time window of project completion, we identified two projects for this analysis: 1) CCFEG Large Wood Atonement White River,[6] an in-river wood enhancement project in the lower White River that installed large logs vertically in arrays at multiple sites spread out across 2.8 river km to improve floodplain connection and to provide more habitat complexity by increasing wood retention rates, and 2) an oxbow reconnection in Nason Creek, CCNRD Nason Creek Lower White Pine Reconnection Project,[7] a multiphase project in which the first phase included installation of a bridge in a BNSF railroad track berm. The berm had substantially disconnected an area of off-channel and floodplain habitat from Nason Creek.

We chose the approach of translating habitat projects into potential biological benefits using an existing juvenile habitat capacity estimation method (Bond et al. 2019). We focused on physical habitat changes as a result of project completion during the 2009–15 period that could be quantified into capacity changes in the juvenile summer parr rearing stage. Given the current state of available empirical survival data from this basin, we found it was not possible to translate project benefits into a change in survival, no matter the life stage at which the project could be targeted. This is because overall life stage survivals are composed of incremental survivals across the spatial domain occupied throughout the life stage, and how any one particular location or moment contributes to that survival within the time-frame of the life stage would be difficult to partition and to assess the influence of a project's benefits on survival. However, making a physical change in the landscape can be quantified in terms of physical space available and its quality or suitability—through assignment of fish densities to the habitat types through the framework of Bond et al. (2019) in quantifying capacity as represented by hydrogeomorphological features—can be estimated more directly. From this, some inferences can be made about capacity needs for particular life stages that have been estimated from typical fish data collected in this basin. Therefore, we made the assumption that a project's benefits can be estimated to affect capacity through Bond et al.'s (2018) estimation of capacity from the expansion estimates of fish through a simple summation of fish-density-per-habitat-type multiplication process. We reiterate, however, that if a project had benefits resulting in changes in survival, we do not currently have a methodology to capture survival changes. Thus, we may not be capturing all of the potential benefits attributable to a habitat project or combination of projects. Below, we describe the process of ascribing benefits to capacity by the two projects.

6.3.1.1. CCFEG Large Wood Atonement White River

To estimate potential change in Chinook salmon parr rearing capacity resulting from the White River Large Wood Atonement project, we used an existing model of Columbia River Basin floodplain habitat (CRBFH modeling; Bond et al. 2019) used to estimate juvenile capacity of spring/summer Chinook salmon in the Columbia basin. The model was

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[6] http://waconnect.paladinpanoramic.com/project/290/16940
[7] http://waconnect.paladinpanoramic.com/project/290/14462

constructed from satellite image analysis of 200-m stream segments (2,200 in total) randomly selected throughout the basin. At each selected site, side channel and mainstem wetted habitats were measured. These measurements formed the response in a random forest model with a set of geomorphic and regional predictors. We used existing estimates of parr densities for geomorphic habitats found in the White River. These estimates assign parr capacity densities to bar edge, bank edge, and mid-channel habitat areas separately. Each of these habitat areas was estimated from the modeled wetted width of each 200-m stream segment. We assumed a linear relationship between the widths of edge habitats and stream widths.

The potential effect of large wood additions on the juvenile capacity of the lower White River was calculated from a multi-step process. We estimated the area of the wood installations (consisting of either pile arrays only or pile arrays with engineered wood structures, according to the project documentation) and multiplied these areas by the increase in per-area capacity expected for wooded (0.84 fish/m^2; T. J. Beechie, unpublished data) compared to wood-free banks (0.33 fish/m^2). Based on the project plan's specification for the structures, we estimated that the 32 engineered wood structures or pile arrays could provide an area of 170 m^2 each, for a total wood area of 5,440 m^2 and a net capacity benefit of 2,776 additional parr, or 0.5% above current estimated capacity from the CRBFH modeling. However, as a ground-truth check of the areas of the wood structures that formed as a consequence of the project implementation—rather than relying solely on the project's as-built specifications—we examined the sites using satellite imagery from 2014 (roughly a year after project completion). We were able to clearly view 11 sites that had accumulated wood and to calculate their areas. The mean area calculated from these 11 sites was 295 m^2, which was larger than the generally proposed areal footprint for each site. Assuming that all sites would be in place post-implementation and could be optimistically characterized as having the mean size calculated from these 11 sites, we estimated a total benefit of 4,829 additional parr from the full project post-implementation, representing a 1.1% increase above the current estimated capacity for the White River from the CRBFH modeling.

6.3.1.2. CCNRD Nason Creek Lower White Pine Reconnection Project

To assess the potential change in Chinook salmon parr rearing capacity resulting from the Nason Creek Lower White Pine Reconnection Project, we used the same CRBFH modeling described above (Bond et al. 2019). To estimate the change in wetted floodplain habitat resulting from the reconnection established by this project, we made predictions of the estimated restored floodplain width in place of the current width for each 200-m section of Nason Creek that intersected with this project. Following implementation of the White Pine Reconnection Project, we estimated from the project proposal that an additional 5,058 m^2 of usable side channel floodplain habitat could be created, resulting in an increase in the total Nason Creek capacity of 1.41% over the current rearing capacity.

The habitat modeling does not account for other types of off-channel habitat (e.g., blind channels or seasonally flooded areas) that may be created in the restored floodplain. Further, we did not estimate the eventual quality of habitat, but assumed that restoration would result in a benefit equivalent to typical functional side channel habitat. Finally, we did not model any potential changes to mainstem Nason Creek habitats that may result from this side channel reconnection.

6.4. Estimating population level outcomes of restoration alternatives: Using LCMs to evaluate differences in fish production among restoration scenarios

We estimated habitat benefit effects of the projects by multiplying juvenile (parr) capacity by the estimated percent change in habitat as calculated above using the Bond et al. (2019) habitat expansion method for estimating fish capacity (Table 6.2). These resulted in a 1.1% increase in the White River and 1.41% increase in Nason Creek (Table 6.2) to estimated parr capacity in the two tributaries with habitat actions completed in the 2009–15 period.

As a result of changing capacity in the LCM by these relatively small amounts, there was no detectable difference in either the number of natural-origin spawning adults or the level of extinction risk between the scenario with no habitat change and the scenario with elevated capacity estimated from the 2009–15 projects (Figure 6.3).

Table 6.2. Habitat benefts of Wenatchee River projects. Estimated spring Chinook salmon parr capacity from habitat models, and estimated parr capacity changes from habitat restoration projects (completed 2009–15) in two tributaries of the Wenatchee River basin.

	White River	Capacity change with wood addition/ retention	Nason Creek	Capacity change with side channel reconnection
Capacity without projects	476,407	—	215,938	—
Capacity with projects	481,648	+1.1%	218,973	+1.41%

Wenatchee spring–run Chinook salmon
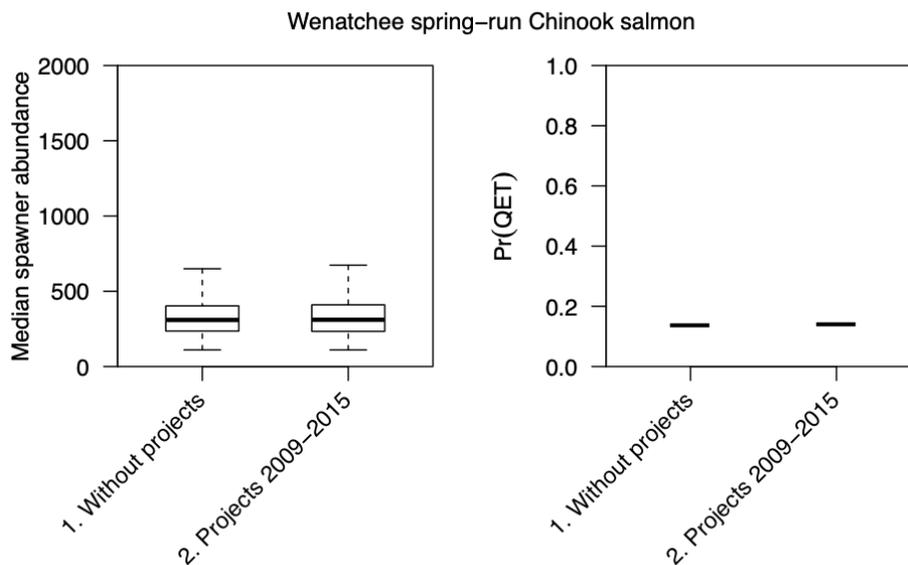
Figure 6.3. Results from the spring-run Wenatchee River life cycle model in response to estimates of habitat conditions with and without completed projects from 2009–15.

6.5.   Discussion regarding the different restoration scenarios

The life cycle model results reflect certain assumptions about project implementation, effects of the habitat changes, and biological conditions that currently exist in the basin. We assumed that the "as-built" conditions after project implementation represented the on-the-ground conditions post-implementation. However, we received reports that, for example, the lower White River pile arrays had mixed results (e.g., in some instances, river flows removed the structures at some sites; in other instances, the amount of retained wood exceeded plan objectives for a site). We were able to view project sites with satellite imagery and adjust the impact of the project shortly after implementation. However, this illustrates that project sites are dynamic and are subject to change after installation in ways that may not necessarily be predictable.

Also, we assumed that the habitat alteration had the anticipated biological response. For example, we assumed that the retained wood in the lower White River enhancement project sites exhibited the higher fish density associated with those habitats, and that the Nason Creek reconnected area resulted in fish densities consistent with those of other off-channel areas. Furthermore, our working assumption was that due to the nature of these projects, the full intended biological benefit was achieved immediately after project implementation, rather than phased in over time.

Our focus was on benefits and, thus, we did not account for the possibility of deleterious effects from the projects. One type of deleterious effect would be the extent to which changes to habitat characteristics could benefit potential Chinook salmon predators, such as bull trout. Another source of uncertainty associated with our modeling exercise is the question of to what extent our actions directed at fish production enhancements above the lake would be realized, amplified, or dampened because of potential lake effects. This is due to a lack of data on the impacts of Lake Wenatchee and fish utilizing tributaries that feed into it.

Our analysis did not account for potential benefits from projects that did not directly manipulate habitat. For example, we did not consider the effects of habitat loss due to not implementing conservation easements and land purchases aimed at protecting intact habitats. This is a useful future exercise in order to quantify effects of habitat loss where existing functional habitat is degraded. Further, information gained from projects focused on reach assessments, monitoring, and research studies helps our understanding of habitat and fish relationships that can add to our knowledge of limiting factors. Even though there were no direct immediate measurable benefits from these types of projects, information gained from past efforts like these areas is useful for the design and location of habitat restoration projects to increase their potential to address limiting factors. Lastly, we did not consider mainstem projects or projects that reduced small amounts of water losses from water diversions. At present, the life cycle model does not consider the lower mainstem to be habitat-limiting. There were some projects in tributaries that currently do not contribute to Chinook salmon production and focus on other species such as steelhead and bull trout (i.e., Chumstick, Peshastin). These projects may provide some benefits for Chinook salmon in the near term and future, but are not accounted for at this time.

7. Estimating population-level outcomes of restoration alternatives in data-poor watersheds: An example from the Upper Salmon River focusing on spring/summer Chinook salmon populations

7.1. Overview/Summary

For the Upper Salmon River Chinook MPG, we have developed life cycle models for eight of the nine populations (the Lower Mainstem population was not modeled at this time due to a lack of habitat and population data). The LCMs are stage specific, with Beverton–Holt-based stage transitions covering spawner to egg, egg to fry, fry to parr, parr to smolt, and smolt to spawner. All of the stage transitions are density-dependent, with the exception of the smolt to spawner component. The redd and juvenile rearing capacity are estimated as functions of stream habitat quality and quantity from Quantile Regression Forest models, a (90^{tile}) regression based on random forest models of parr and redd abundance data relative to a range of reach-scale habitat metrics. Stage-specific survival is based on PIT-tag mark-resight data and internal model calibration to existing data population time series. PIT tagging is extensive in several USAL populations, in particular the Lemhi. From these data, estimates of survival are possible across the life cycle. Assuming that the underlying biology will be similar across the MPG, the survival estimates based on previous work from the Lemhi were applied to the remaining populations. Parr summer survival was used as the free parameter to calibrate overall population projections under baseline conditions. Using the quasi-Bayesian estimation process, population-specific summer parr rearing survival (cast as productivity in the parr–smolt B–H function) estimates were generated for each population with adult or juvenile abundance timeseries.

Habitat restoration scenarios were developed from a baseline of stream habitat quality and quantity built from reach typing and geomorphic condition calibrated to Columbia Habitat Monitoring Program (CHaMP) reach scale habitat monitoring (2011–17). Improvements to habitat quality and quantity were parameterized three ways: 1) from habitat projects listed in the Pacific Northwest Stream Habitat Project (PNSHP) database as having been initiated over the interval 2009–15, 2) a projection of future actions (post-2018) based on random project locations, and 3) a projection of future actions (post-2018) based on applying the level of effort specified for the Upper Salmon Chinook MPG in the Proposed Action across four focal populations within the MPG.

Stream habitat restoration actions were estimated to impact carrying capacity for spawning and rearing, as well as juvenile stage transition rates. As the basis of the freshwater habitat in these models is the reach type and geomorphic condition of the reaches, only in-stream complexity actions were modeled to improve habitat quality and access actions to improve habitat quantity. Since reach geomorphic condition represents habitat quality, the impact of a restoration action within the reach was to improve the geomorphic condition rating by a single step.

The population-level outcomes of restoration alternatives were modeled by running population simulations for 100 years, replicated 500 times. The performance metrics from these simulation sets were the median and quantiles of the size of natural-origin spawner

population at year 50 and the probability that the population met the quasi-extinction criteria by year 24. The quasi-extinction threshold used in these simulations was falling below either 30 or 50 individuals for four consecutive spawning years. All metrics are reported as change relative to a baseline condition, as these standardized metrics are more appropriate than the absolute value of estimated population size and extinction risk.

7.2.   Background

The Upper Salmon River MPG includes nine independent populations (Figure 7.1). Independent populations in the Upper Salmon River MPG include: North Fork Salmon River, Lemhi River, Pahsimeroi River, Upper Salmon River Lower Mainstem (below Redfish Lake Creek), East Fork Salmon River, Yankee Fork Salmon River, Valley Creek, Upper Salmon River Mainstem (above Redfish Lake Creek), and Panther Creek (extirpated). All four population size-classes, based on historic intrinsic production potential, are represented in the MPG. Characteristics of the nine independent populations are listed in Table 7.1.

Hatchery production of spring/ summer Chinook salmon in the Upper Salmon River MPG is primarily related to mitigation or compensation for the impacts of hydroelectric dam development on the Snake River. Pahsimeroi River and Upper Salmon River Mainstem populations are included in integrated hatchery programs based on indigenous

Figure 7.1. Spring/Summer Chinook salmon populations in the Upper Salmon River MPG.

stocks. The East Fork Salmon River, Yankee Fork Salmon River, Lemhi River, and Valley Creek populations have some history of hatchery supplementation with Upper Salmon, local, and Rapid River stocks, but are considered to be persisting because of natural reproduction of the local stocks at present.

Table 7.1. Viability assessments for Snake River spring/summer Chinook salmon populations in the Upper Salmon River MPG. Key: *R/S* = recruits per spawner; *A/P* = abundance:productivity ratio; *SS/D* = spatial structure:diversity ratio.

| Population level: | | Abundance and productivity | | | | | Spatial structure and diversity | | | |
| | | Abundance | | Productivity | | Overall A/P | Goal A | Goal B | Overall SS/D | |
Population	Status	Current natural abundance	Min. threshold	Current estimate (R/S)	Min. R/S @ threshold	Integrated A/P risk	Natural processes risk	Diversity risk	Integrated SS/D risk	Overall viability rating
North Fork Salmon River	Extant	Insufficient data	500	Insufficient data	2.21	High	Low	Low	Low	HIGH RISK
Lemhi River	Extant	79	2,000	1.07	1.34	High	High	High	High	HIGH RISK
Pahsimeroi River	Extant	127	1,000	0.54	1.58	High	Moderate	High	High	HIGH RISK
Upper Salmon River Lower Mainstem	Extant	103	2,000	1.22	1.34	High	Low	Low	Low	HIGH RISK
East Fork Salmon River	Extant	148	1,000	1.07	1.58	High	Low	High	High	HIGH RISK
Yankee Fork Salmon River	Extant	13	500	0.68	2.21	High	Moderate	High	High	HIGH RISK
Valley Creek	Extant	34	500	1.07	2.21	High	Low	Moderate	Moderate	HIGH RISK
Upper Salmon River Mainstem	Extant	246	1,000	1.51	1.58	High	Very Low	Moderate	Moderate	HIGH RISK
Panther Creek	Extinct	Extinct	750	Extinct	1.76	Extinct	Extinct	Extinct	Extinct	EXTINCT

All extant populations in this MPG were at high risk for the integrated viability rating (Table 7.2) at the time of initial population designations by the ICTRT, based on abundance and productivity (A/P) ratings for all extant populations in this MPG being High Risk. At the time, abundance levels for all extant populations were below 25% of the minimum abundance thresholds. The spatial structure ratings varied between populations, from Low Risk to High Risk. Four of the eight extant populations were rated either Low or Moderate for spatial structure and diversity (SS/D) risk and, therefore, could achieve viable status if A/P risk was reduced.

As of the 2015 status review of all ESA listed salmonid stocks, A/P estimates for most populations within this MPG remain at very low levels relative to viability objectives. The Upper Salmon River Mainstem (SRUMA) population has the highest relative abundance and

Table 7.2. Upper Salmon River MPG spring/summer Chinook salmon population risk ratings integrated across the four viable salmonid population (VSP) metrics. *Key*: Very low = <1%, Low = 1–5%, Moderate = 6–25%, High = >25%; HV = highly viable, V = viable, M = maintained, HR = high-risk (does not meet viability criteria).

		Spatial structure/diversity risk			
		Very low	**Low**	**Moderate**	**High**
Abundance/productivity risk	**Very low**	HV	HV	V	M
	Low	V	V	V	M
	Moderate	M	M	M	HR
	High	HR	HRa	HRb	HRc

a North Fork, Lower Mainstem.
b Valley, Upper Mainstem.
c Lemhi, Pahsimeroi, East Fork, Yankee Fork.

productivity combination of populations within the MPG. SS/D ratings vary considerably across the MPG. Four of the eight populations are rated at low or moderate risk for overall SS/D and could achieve viable status with improvements in average A/P. The high SS/D risk rating for the Lemhi population is driven by a substantial loss of access to tributary spawning and rearing habitats and the associated reduction in life history diversity. High SS/D ratings for Pahsimeroi River, East Fork Salmon River, and Yankee Fork Salmon River are driven by a combination of habitat loss and diversity concerns related to low natural abundance combined with chronically high proportions of hatchery spawners in natural areas.

For the entire Snake River Spring/Summer Chinook ESU, long-term trend and population growth rate estimates have been <1 for all natural production data sets, reflecting the large declines since the 1960s. Short-term trends and λ estimates have been generally positive, with relatively large confidence intervals. However, Snake River spring/summer-run

Chinook salmon must migrate past between six and eight mainstem Snake and Columbia River hydroelectric dams to and from the ocean, and all reviews of stock status have concluded that mainstem Columbia and Snake River hydroelectric projects have resulted in major disruption of migration corridors and have affected flow regimes and estuarine habitat, and thus population productivity.

Additionally, tributary habitat conditions vary widely among the various drainages of the Snake River basin. Habitat is degraded in many areas of the basin, reflecting the impacts of forest, grazing, and mining practices. Impacts relative to anadromous fish include lack of pools, higher water temperatures, low water flows, poor overwintering conditions, and high sediment loads. Therefore, to help understand the relative value of management actions, we have constructed a series of population-scale life cycle models that represent the physical and biological settings for eight of the nine Upper Salmon River spring/summer Chinook salmon populations.

The LCM approach is an important tool for exploring the relative value of management actions, such as tributary habitat restoration or mainstem hydropower project operation adjustments. Ideally, large-scale management actions implemented with sufficient intensity—a large spatial extent over a short temporal duration—should result in a measurable population benefit. For example, a change in habitat quality or quantity (HQQ) will result from stream habitat restoration actions and fish biological processes at the individual level (growth, survival, movement, life history expression) will be affected in a manner that positively benefits the population (fresh water productivity, population growth rate, extinction probability). However, there are two practical considerations, intensity and population constraints, that prevent this direct "implement and monitor" approach from being the default strategy for linking management actions with fish population benefits (Bennett et al. 2016).

Why doesn't a direct, "implement and monitor" approach work? It is extremely difficult to coordinate a watershed-scale pulse of actions that results in a significant change in HQQ (with the possible exception of dam removal, e.g., the Condit or Marmot Dam projects); budget constraints, environmental catastrophes, and even scheduling challenges make it essentially impossible to implement watershed-scale restoration that leads to suddenly measurable changes in HQQ. Thus, the effect size necessary to generate a measurable population response rarely results from singular management action implementations. In terms of issues of population process constraint, often multiple ecological impairments are present and, as such, a restoration strategy must address all before capacity or productivity release can be accomplished. Only in rare cases is the tributary environment of a salmon or steelhead population amenable to direct manipulation to demonstrate the positive benefit of restoration actions at the population scale (e.g., IMWs; cf. Bennett et al. 2016). Therefore, a parsimonious approach to generating the necessary estimates of population benefits to management actions is to combine the best available science on individual fish response to changes in HQQ with measured changes in habitat features from restoration actions in a population process model, to estimate the aggregated (over time and space) effects of changes in HQQ on entire populations of salmon or steelhead.

Life cycle models are a standard approach to understanding the physical and biological processes underlying population dynamics (Nickelson and Lawson 1998), and LCMs have a long history of application in conservation and population management situations (Morris

and Doak 2002, Holmes et al. 2004, Schumaker and Brooks 2018). The management of ESA-listed salmonid populations presents an ideal setting for applying LCMs as a key component of a decision support system. Salmonid life cycles encompass vast geographic ranges, and given this necessary degree of habitat diversity, the opportunities for impacts due to human activity are manifold. Developing effective management strategies to conserve and recover ESA-listed salmonid populations involves balancing a portfolio of potential actions that are applied across life stage, habitat type, jurisdiction, and anthropogenic impact type (Bartz et al. 2006). The most commonly advanced tool for salmonid conservation and recovery is freshwater tributary habitat rehabilitation; a restoration industry exists, as anthropogenic disturbance in salmonid spawning and rearing habitat is extensive and has been ongoing for centuries. However, given finite resources available to implement meaningful population recovery actions, developing cost-effective strategies within the freshwater environment, and between these actions and alternatives in other potential management domains (e.g., "Harvest, Hatchery, or Hydro" actions), is necessary. LCMs represent an important management support tool in that a population forecast model will generate alternative futures based on a standardized suite of input population biological parameters (e.g., stage-specific capacity, productivity) and action scenarios (e.g., habitat action type or extent, climate change, migration corridor conditions).

7.3.   Upper Salmon River Mainstem LCM structure

A model for salmon population dynamics, as initially developed and described by Yuen and Sharma (2005), has been coded in the R programming language specifically to facilitate the evaluation of multifaceted management strategies for populations of anadromous salmonids in the interior Columbia River basin.

The model implements the Beverton–Holt spawner–recruit salmon population dynamics model (Beverton and Holt 1957). Inputs describing one or more sites within a watershed, survival estimates by life stage, etc., are user-specified model inputs, as are measures of uncertainty in parameter estimates, and estimates of natural parameter spatial, temporal, and pure variability. Initial salmonid populations, by life stage, are also user-specified. The model calculates fish populations by life stage for each subsequent year up to a user-specified number of years. Hatchery fish introductions into a watershed, and parameters describing the relative robustness and fecundity of hatchery fish and their descendants, can also be user-specified.

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

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

Multiple sites may be modeled simultaneously. "Sites," in this model, refers to a spatial scale over which the user wishes to define the input parameters. A site may be a reach within a tributary, a tributary within a watershed, a watershed within a subbasin, etc. The advantages of concurrent modeling of multiple sites, as opposed to modeling one site at a time, are twofold: First, sites within a watershed are likely not independent. A low-water year for a single site is likely a low-water year for all sites within a watershed, and this model can account for such correlations. Second, modeling multiple sites concurrently allows for inclusion of cross-site migration, where fish at various life stages have some user-specified nonzero probability of migrating to a different site within a watershed. In addition, modeling multiple sites concurrently allows summarization of results at any spatial level after the completion of the simulation (i.e., results may be summarized by site, stream, river, watershed, etc.).

The model has been structured such that it is flexible enough to handle different species of salmonids (though they cannot be modeled simultaneously). This includes steelhead, which are generally more complex than other salmonid species as far as modeling the transitions from one life stage to the next.

7.3.1. Incorporating habitat quality and quantity into a Beverton–Holt spawner–recruit model

The watershed population model follows the Beverton–Holt spawner–recruit model (Beverton and Holt 1957) as has previously been implemented for modeling life stage population dynamics for salmonid populations (Mousalli and Hilborn 1986, Yuen and Sharma 2005). The basic structure of the Beverton–Holt model is as follows:

$$N_{k,i+1,t+1} = \frac{N_{k,i,t}}{\dfrac{1}{p_{k,i,t}} + \dfrac{1}{c_{k,i,t}} \cdot N_{k,i,t}}$$

where:
- $N_{k,i,t}$ is the number of individuals alive at the beginning of life history stage i at time t for site k,
- $p_{k,i,t}$ is the "productivity" at stage i (the maximum survival rate from stage i to $i+1$), and
- $c_{k,i,t}$ is the "capacity" (the maximum number of individuals that could survive from stage i at time t to stage $i+1$ at time $t+1$).

Note that within the salmon model, certain life stages occur within the same year as previous life stages, while others occur the following year. Thus, the value of the subscript t in the above equation will sometimes be the same on the left and right sides of the equation, while in other cases it will be incremented by one.

Productivity at time t for site k is modeled as:

$$p_{k,i,t} = Sr_{k,i,t} \cdot \frac{\sum\limits_{q=1}^{n}\left[E_{i,q}\right]\cdot\left[L_{q,k}\right]_t}{\sum\limits_{q=1}^{n}\left[L_{q,k}\right]_t} \cdot \frac{\sum\limits_{j=1}^{n}\left[F_{i,j}\right]\cdot\left[L_{j,k}\right]_t}{\sum\limits_{j=1}^{n}\left[L_{j,k}\right]_t}$$

where:
- $p_{k,i,t}$ = density-independent productivity for stage i dependent on the relative importance/relationship between productivity and reach type,
- $E_{i,q}$ = a scalar governing the relative value of habitat condition q, and $F_{i,j}$ = a scalar governing the relative value of reach type on overall productivity,
- $L_{j,k}$ and $L_{q,k}$ = proportion of total habitat of type j or in condition q at site k, and
- $Sr_{k,i,t}$ = site- and time-specific average maximum survival rate from one stage to the next in the freshwater life history of the species given average conditions under a baseline in the best possible habitat suited for their survival.

Capacity at time t for site k at life stage i is modeled as:

$$c_{k,i,t} = \sum_{g=1}^{n}\left[H_{k,q,t} \cdot D_{k,q,i,t}\right]$$

where:
- $H_{k,q,t}$ is the amount of reach type specific to habitat q in watershed k at time t, and
- $D_{k,q,i,t}$ is the maximum density, in fish per unit area (m²), of reach type q at site k during life stage i at time t.

$H_{k,q,t}$ is modeled as:

$$H_{k,q,t} = A_{k,t} \cdot \sum_{q=1}^{n}\left[L_{k,q,t} \cdot \sum_{j=1}^{n}\left[M_{k,j,q,t} \cdot G_{k,q,i,t}\right]\right]$$

which can be rewritten as:

$$H_{k,q,t} = \sum_{q=1}^{n}\left[L_{k,q,t} \cdot A_{k,t} \cdot \sum_{j=1}^{n}\left[M_{k,j,q,t} \cdot G_{k,q,i,t}\right]\right]$$

The product $L_{k,q,t} \cdot A_{k,t}$ is entered as a single user input, $M_{k,j,q,t}$, the proportion of each reach type q of geomorphic condition j at time t for site k. $G_{k,q,i,t}$ is a scalar that governs the relative capacity of each reach type by its geomorphic condition for each life stage i.

7.3.2. Reach typing and geomorphic condition assessment using a River Styles framework

The River Styles framework (Brierley and Fryirs 2005) is a methodology for understanding why rivers appear and behave the way they do under current sediment and flow regimes, and how they are likely to appear and behave in the future. At the core of the River Styles framework is the recognition that rivers operate and adjust under the strong influence of a nested hierarchy of landscapes, landforms, deposits and habitats. The River Styles framework provides the user a set of guidelines on how to delimit and describe the structure and function of rivers based on patterns of river types and their biophysical linkages in a catchment context (Brierley and Fryirs 2005). It does this by characterizing rivers within their unique watersheds, a trait not shared with most existing river classification schemes (cf. Rosgen 1994, Montgomery and Buffington 1997). Within this method is a focus on the observation and interpretation of geomorphic forms and processes with which to assess river character and river behavior. Using these observations, a rigorous process for predicting future river condition is based on contemporary conditions, evidence of past conditions, and the recovery potential of any given reach with individual streams (e.g., Kellerhals et al. 1976, Frissell et al. 1986).

The basis for geomorphic river classification is the systematic categorization of physical attributes of a river flowing in its channel, the valley through which it flows, and the geomorphic features that comprise its floodplain and channel (Buffington and Montgomery 2013). Through a spectrum of bedrock and alluvial variants, these characteristics reflect a balance of sediment supply and channel transport capacity. A river's character is its unique river morphology, including valley, floodplain and in-stream geomorphic features; whereas river behavior is the tendency and capacity for adjustment within its valley setting and floodplain, tied to boundary conditions set by flow and sediment fluxes typical for that stream. River behavior drives the assembly of geomorphic units present within its channel—by form and process associations. Reach types are determined through analysis of four key physical parameters: valley setting, channel planform, floodplain and in-stream geomorphic units, and bed material texture. These parameters compile common sets of characteristics at the reach scale. Reach breaks are indicated by wholesale changes in any one of these parameters. Essentially, this is letting the river's behavior drive the interpretation of pattern and process.

The River Styles framework guides the process of Reach Typing and Geomorphic Condition assessment through a standard set of steps or stages. The first involves characterizing the watershed in terms of its regional setting and landscape components, and delineating the drainage network into reach types based on stream characteristics and behavioral attributes governed by landscape and lithologic controls (i.e., bedrock hardness and erodibility). In this stage, the emerging network of reach types develops into common downstream patterns that help to characterize forcing mechanisms for river attributes in the watershed. Given the assessment of river character and behavior accomplished in the first stage, the geomorphic condition of each reach type is then assessed based on the natural capacity for the system to adjust within its boundary conditions (i.e., valley setting, sediment supply and flow regime, catchment characteristics). These results allow an appraisal of the pathway of likely future adjustment and the recovery potential for each reach type.

Reach Typing and Geomorphic Condition assessment was done within the watersheds of the Columbia Habitat Monitoring Program (ChaMP; Figure 7.2). Reach Typing procedural trees and Geomorphic Condition geoindicator tables were developed for CHaMP monitoring watersheds (n = 11) and sites (n = 897). The procedural trees and geoindicators developed based from on-the-ground data collection were then applied to the entire Upper Salmon Spring/Summer Chinook MPG through photo interpretation and GIS-based spatial models.
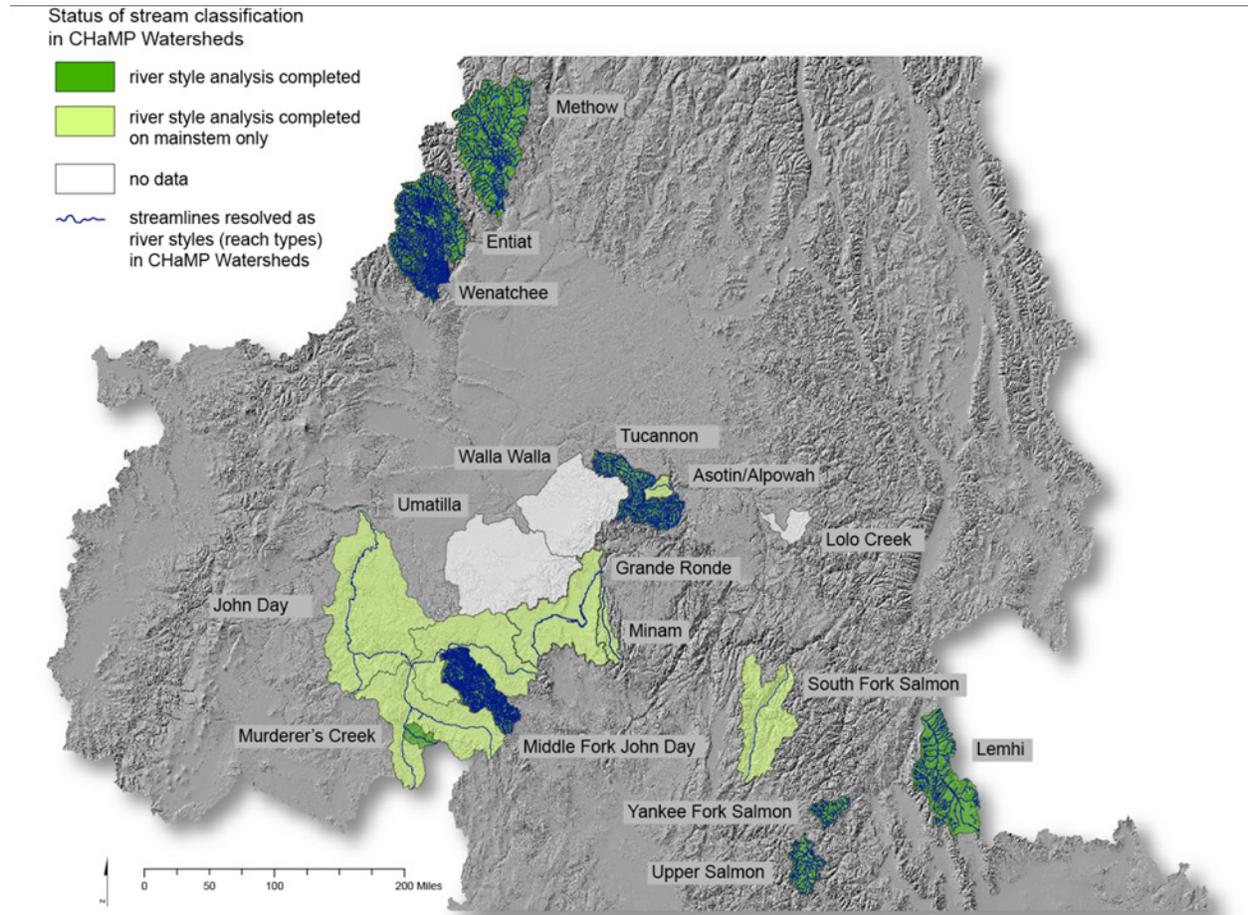
Figure 7.2. Extent of reach type analyses in CHaMP subbasins throughout the Columbia River basin.

7.3.2.1. Reach typing

Compilation of background information and designation of landscape was accomplished through aerial reconnaissance using Google Earth imagery (high-resolution satellite [SPOT image]) and aerial photography (Digital Globe, Google 2014). This involved mapping draft reach types onto a drainage network using the core criteria for identification of river types as put forth by Brierley and Fryirs (2005): recognizing the geomorphic attributes of valley setting or channel confinement, channel planform (the channel outline in map view), geomorphic units, and bed material texture. Fundamental change in any one of these physical attributes marks a reach break between one type and the next along a stream arc. Reach Types and their extent were corroborated at representative sites for each type

in the CHaMP watershed. Classification accuracy has not yet been determined for the application of the reach typing procedural trees to the portions of the Upper Salmon River Spring/Summer Chinook MPG outside of the Lemhi, Yankee Fork, and Upper Salmon River Mainstem population watersheds. Field validation in the East Fork and Panther Creek is planned for 2018–19 through collaboration with the Shoshone Bannock.

Once defined in terms of river character and behavior each reach is keyed to a series of river styles trees that display the criteria appropriate for individual settings. The "road map" for each river style tree is the river styles procedural tree shown in Figure 7.3. The procedural trees define the core criteria unique to each set of landscape controls within sub-catchments that drive valley and channel characteristics. The differences are most apparent in the sense of top-down controls, starting with the valley setting configuration. For example, floodplain and planform characteristics are important in laterally unconfined and partly confined valley settings but are not important in confined valleys. Conversely, bedrock channels are not generally a factor in laterally unconfined valleys.
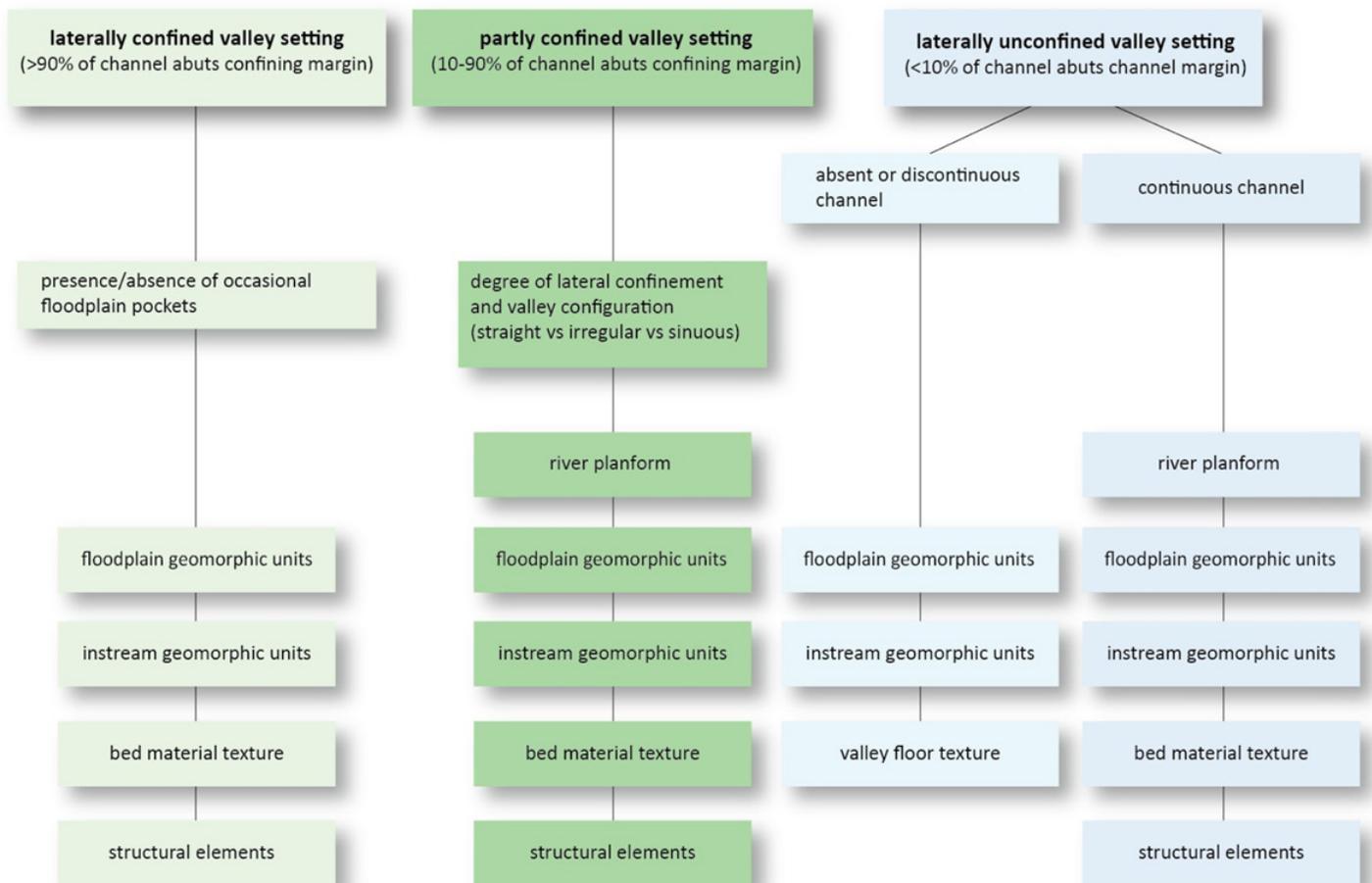
Figure 7.3. The river style procedural tree modified from Brierley and Fryirs (2005) for the interior Columbia River basin. Structural elements include any of the following: natural woody debris, large boulders, installed restoration structures, and engineered additions to the channel (roads, bridges, culverts, etc.).

The entry-level criteria for determining a reach type is valley setting, or degree of channel confinement. The degree of confinement is critical to understanding river behavior (the ability of a river to adjust laterally and to some extent, vertically, within its channel) because valley setting tends to dictate whether a river is storing sediment and maneuvering within its floodplain, or whether it is conveying sediment downstream over a steeper gradient with little room to adjust. It is an expression of the rate of bedrock incision relative to valley widening.

The valley setting describes the valley through which the river flows along with any other deposits or structures that impose a barrier to lateral adjustment of the river within its channel. Valley setting is determined through the interaction of the channel with confining margins imposed by a) the valley bottom margin, which is the trace of the alluvial floodplain defined by the valley walls or other deposits, and b) deposits such as alluvial or debris fans, coarse-grained abandoned floodplains (terraces), or bedrock outcrops (Figure 7.4). Together, the valley bottom margin and surficial deposits define the confining margin.
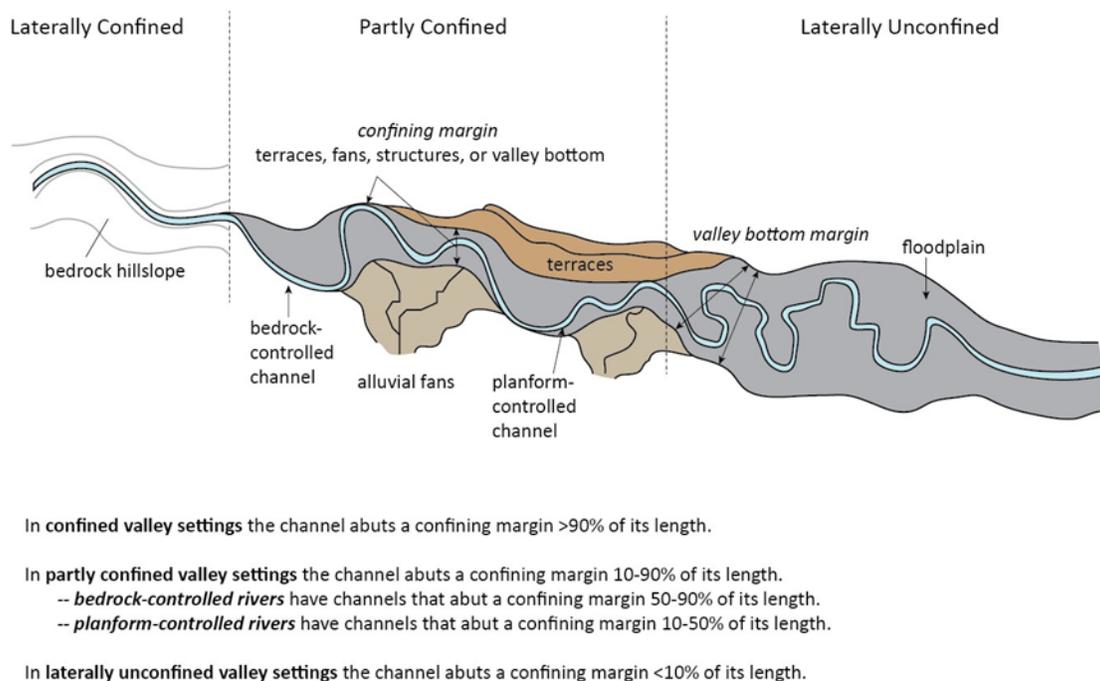
Figure 7.4. Conceptual schematic of valley setting along three distinct reaches. Gray background indicates floodplain. In confined valley settings, the channel is between narrow valley walls and contains little or no floodplain (see text for details). Modified from Wheaton et al. (2017).

If a channel flows within a confined valley setting, there generally is no floodplain or only short, discontinuous floodplain pockets. The channel abuts a confining margin >90% of its length in confined valley settings. In partly confined valley settings, the channel is restricted against the valley wall 10–90% of its course within that reach, and discontinuous floodplain segments may be observed as the river sweeps or scrolls between one valley wall and the next (Brierley and Fryirs 2005). A river flowing across a laterally unconfined valley is free to adjust laterally and downstream within its floodplain. It is in contact with confining margins 10% of its length or less. In this study, the approach to determining valley setting was subjective, based on visual estimates of channel contact, floodplain extent, and valley wall characteristics between reach breaks.

The next consideration is channel planform. The degree of channel sinuosity (channel length divided by downstream distance) and the number of channels present is noted, and whether the channel is bedrock-controlled or planform-controlled (able to laterally adjust or shift within its floodplain). Valley confinement and planform characteristics strongly influence the presence and character of floodplain and in-stream geomorphic units. These m- to km-scale features are the key indicators of flux boundary conditions (i.e.,

Figure 7.5. Conceptual map view of an in-stream, channel-spanning woody restoration structure. Features such as plunge/scour pools and structurally forced midchannel bars result from hydraulic modification of the channel bed material.

flow regime, flood history, and sediment flux through, or being stored within the reach). Bed material texture (sediment caliber or grain size) and sorting are strong indicators of system energy and proximity to source, transfer or accumulation process zones. In addition to the above criteria, we have added observations of structural elements given their importance in creating and maintaining fishery habitat (Wheaton et al. 2010). In-stream structural elements occur as naturally accumulated woody debris that are capable of forcing modification of bar forms (Wheaton et al. 2012), and as restoration structures that are installed to enhance channel form heterogeneity and habitat diversity (Figure 7.5).

Applying the procedural trees developed for a subset of the watersheds in the Upper Salmon River Chinook MPG to all of the population watersheds was done as a preliminary assessment of habitat typing (Figure 7.6).

7.3.2.2. Geomorphic condition

Stage Two of the River Styles framework is an assessment of the geomorphic condition of individual reaches of each river style. The geomorphic condition is the expected form and function of a river flowing in a particular valley setting, subject to boundary conditions of the physical setting and sediment/discharge conditions in the watershed, and constrained by limiting factors and pressures imposed by land use and development. Geomorphic condition is important to measure because it is a gauge of habitat quality, river health, and ties directly to the recovery potential of impacted stream reaches.

The assessment is accomplished by understanding the potential for a reach to modify its channel shape, in-stream geomorphic units, and floodplain, or its capacity for adjustment. Geomorphic features of the channel and floodplain are identified that have potential to change or respond to disturbances, and thereby provide indicators of the condition of each
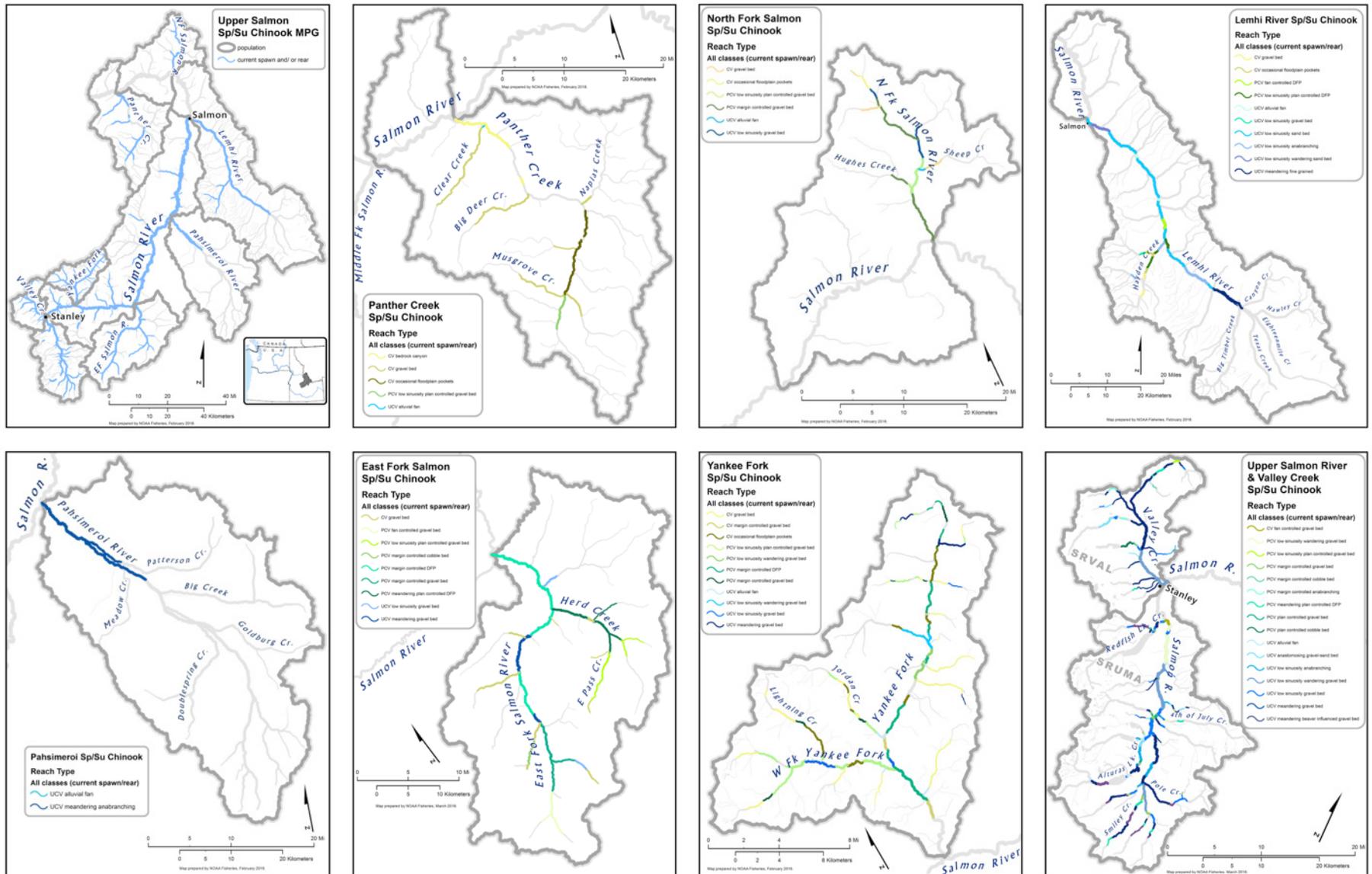
Figure 7.6. Reach typing for spawning/rearing habitat in eight of nine Chinook salmon populations in the Upper Salmon River MPG.

stream reach. Each river style has an explicit pattern of behavior, given its physical setting and boundary conditions. The condition of one reach of a river style relative to another (hereafter "variants") can be understood by comparing each one to a "reference reach," an example of that river style found in the watershed that is closest to pristine.

Geomorphic condition is assessed in three steps: first, the capacity for adjustment of each river style was assessed; next, the geomorphic evolution of each river style was investigated to identify irreversible geomorphic change and a "reference condition" (that of the reference reach); finally, the geomorphic condition of each reach was determined and explained.

The capacity for adjustment is potential morphological adjustments that do not result in a wholesale change in river style. Specifically, they are modifications of the floodplain, channel, and bed material characteristics controlled by the valley through which the river flows, the bedrock lithology and channel slope, and the sediment-discharge balance in the watershed (the physical and flux boundary conditions that define each river style). The adjustment potential of a stream is also a gauge of its sensitivity to local and systemwide disturbances in the watershed. For example, river styles possessing low adjustment potential are resistant to natural or anthropogenic disturbances, whereas those with significant adjustment potential are more susceptible to disturbances.

The confined valley river style (Figure 7.7, left) has very low capacity for adjustment and is considered "resistant" or "resilient" to disturbance. The low-to-moderate sinuosity gravel bed river style (Figure 7.7, right) has significant adjustment potential and is sensitive (susceptible) to direct and indirect disturbances. Arrows indicate the vertical and lateral adjustment possible for each valley setting. The confined-valley river is able to incise its bed, yet rates of bedrock incision are imperceptibly small compared to aggradation in systems where channel, floodplain, and bed characteristics are all controlled by sediment flux. Rivers of confined valley settings tend to have low capacity for adjustment because they flow within narrow bedrock walls and possess very limited or absent floodplains. The shape of confined

Confined valley setting   Laterally unconfined valley setting
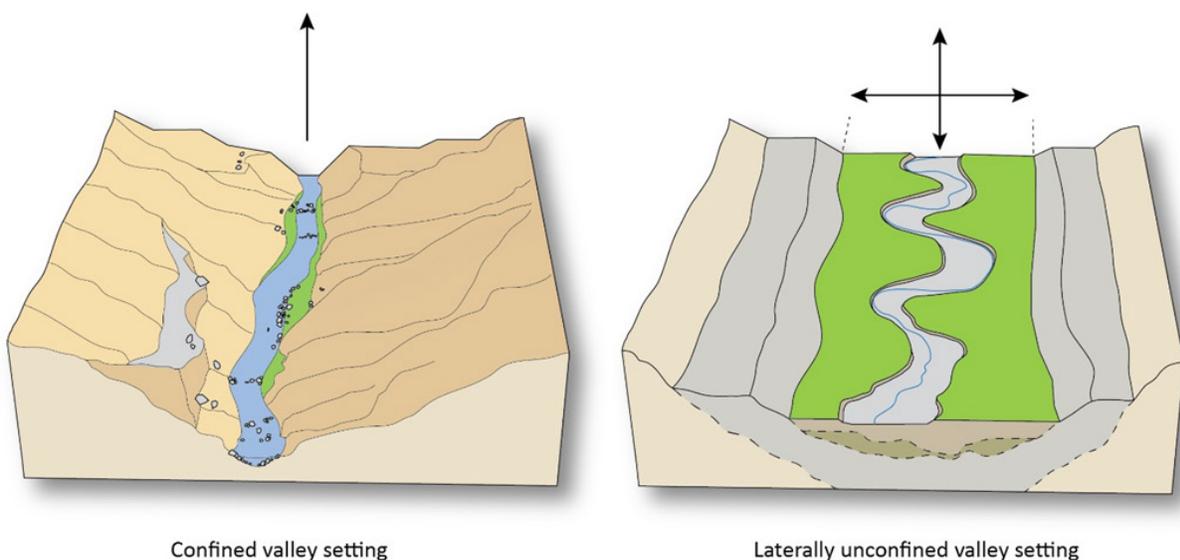
Figure 7.7. Differences in the natural capacity for adjustment of a river style in (left) a confined valley setting, and (right) a laterally unconfined valley setting. Floodplain extents are shown in green.

valley channels is restricted by intervening bedrock, leaving only the arrangement of coarse bed material as a mode of adjustment. River styles of laterally unconfined and partly confined valley settings, on the other hand, have moderate-to-high adjustment potential because their broad, fine-grained floodplains promote dynamically shifting, meandering planforms.

Reaches of every river style exist in varying stages of development, equilibrium and degradation in the interior Columbia River basin. These geomorphic variants occur through natural channel evolution (strongly controlled by watershed position and hydrology) and by local impacts and disturbances that affect their form and function (i.e., capacity for adjustment and reach sensitivity to disturbances). They are described in "evolutionary diagrams," a series of conceptual channel cross-sections that depict different reaches and their geomorphic attributes—including the type and timing of human impacts and modifications. Their purpose is to:

- Inventory the range of variants of every river style, and account for the differences in geomorphic controls.
- Assess river character and behavior prior to human settlement.
- Determine the nature of boundary conditions for that river style.
- Determine whether human disturbance has induced irreversible geomorphic change.
- Identify a reference condition for each river style.
- Predict future conditions and potential prioritized management reaches.

Evolutionary diagrams are constructed through analysis of aerial photographs, field notes and measurements collected during pro forma evaluations (including measured cross-sections and inventory of geomorphic attributes), and historical data. They include known changes to vegetation, land use, sediment dynamics, basin hydrology, and, in instances where available, sampling of key floodplain and hillslope deposits for precise age determination (e.g., radiocarbon and luminescence dating of sediments).

The channel, planform, and bed of a stream possess measurable components (geoindicators) such as channel shape and size, sinuosity of the planform, and stability and storage characteristics of the bed. Geoindicators that are a functional part of each river style were identified and assigned a diagnostic question designed to give a relevant and reliable signal for the condition of a reach for each river style. Table 7.3 illustrates this process for reaches in the Yankee Fork. Applying the geoindicator evaluation based on direct channel observations as well as compiled remote imagery allows the broad-scale estimation of geomorphic condition of the watersheds of the Upper Salmon River Chinook MPG (Figure 7.8).

Each river style and its geomorphic condition is assessed relative to some benchmark or reference condition that is a gauge of the extent to which human-induced change has influenced the long-term pattern of river form and function. Reference conditions chosen for river styles are generally the least-disturbed reaches, because pristine pre-settlement conditions do not exist for all reach types. Also, the preliminary reach type and geomorphic condition assessments done for the purposes of developing life cycle models in the Upper Salmon River Chinook MPG will be improved with additional on-the-ground validation across watersheds of the Upper Salmon, but also, more broadly, across the interior Columbia River basin.

Table 7.3. Example table of geoindicators diagnostic of geomorphic condition variants from Yankee Fork Salmon River. *Key*: A = bedrock-controlled gravel bed, B = bedrock canyon, C = boulder bed, D = occasional floodplain pockets, E = gravel bed, F = steep alpine headwater; Mod = moderate.

Degrees of freedom, relevant geoindicators	Questions used to assess geomorphic condition of each reach	A	B	C	D	E	F
Channel attributes	3 out of 4 questions must be answered YES for stream to be assessed in GOOD condition						
Size	Is channel size appropriate given the catchment area, the prevailing sediment regime, and the vegetation character? Is the channel functionally connected to floodplain pockets (i.e., is the channel overwidened/overdeepened, or does it have an appropriate width:depth ratio)?	✗	✓	✓	✓	✓	✓
Shape	Is the channel shape consistent with confined valley features (typically symmetrical)?	✓	✓	✓	✓	✓	✓
Bank	Is the bank morphology consistent with caliber of sediment? Are banks eroding in the correct places?	✗	✓	✓	✓	✓	✓
Woody debris loading	Is there woody debris in the channel or potential for recruitment of woody debris?	✗	✗	✓	✓	✓	✓
	Overall:	✗	✓	✓	✓	✓	✓
Channel planform	4 out of 5 questions must be answered YES						
Channel number	Is the channel appropriate for this reach type? Are there signs of change, such as avulsions or overbank channels forming on the floodplain?	✓	✓	✓	✓	✓	
Channel sinuosity	Is the channel sinuosity consistent with the sediment load/transport regime and the slope of the channel?	✗	✓	✓	✓	✓	✓
Lateral stability	Is the lateral stability consistent with the sediment texture and channel slope? Are there signs of degradation, such as local widening and atypical in-channel reworking of bed material?	✗	✗	✓	✓	✓	✓
Geomorphic unit (GU) assemblage	Are the number, type, and pattern of in-stream geomorphic units appropriate for the sediment regime, slope, bed material, and valley setting? Are key GUs of this reach type present?	✗	✓	✓	✓	✓	✓
Riparian vegetation	Are the appropriate types and density of riparian vegetation present on the banks and floodplain?	✗	✓	✗	✓	✗	✓
	Overall:	✗	✓	✓	✓	✓	✓
Bed character	3 out of 4 questions must be answered YES						
Grain size/sorting	Is the range of sediment throughout the channel and floodplain organized and distributed appropriately?	✗	✗	✗	✓	✗	✓
Bed stability	Is the bed vertically stable, such that it is not incising or aggrading inappropriately for the channel slope, sediment caliber, and sinuosity?	✓	✓	✓	✓	✓	✓
Sediment regime	Is the sediment storage and transport function of the reach appropriate for the catchment position (i.e., is it a sediment transfer or accumulation zone)?	✗	✓	✗	✓	✗	✓
Hydraulic diversity	Are roughness characteristics and the pattern of hydraulic diversity appropriate for the catchment position?	✗	✓	✓	✓	✓	✓
	Overall:	✗	✓	✗	✓	✗	✓
	Total checks and crosses are added for each stream reach:	✗	✗	✓	✓	✗	✓
Geomorphic condition	*Overall geomorphic condition of the reach type:*	Poor	Mod	Mod	Good	Mod	Good
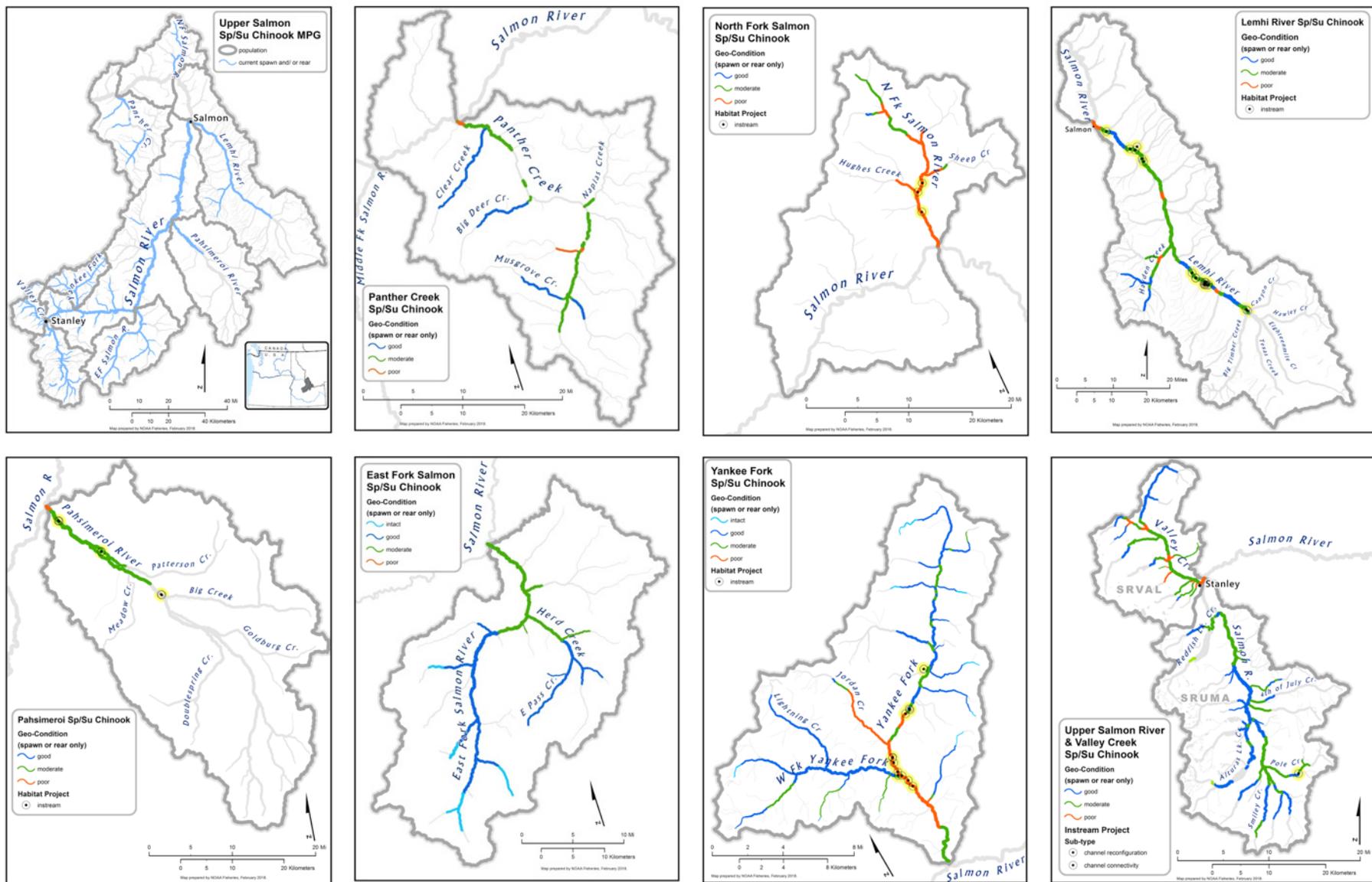
Figure 7.8. Geomorphic condition assessment of spawning/rearing habitat for eight of nine populations of Chinook salmon in the Upper Salmon River MPG.

7.3.3. Habitat capacity estimation

To estimate life stage-specific habitat capacity for spring Chinook (SPCH), models were developed to predict summer parr rearing and redd capacity estimates using paired fish and habitat data. Fish data were based on observations of juvenile summer parr density and abundance or redd observation data. Fish data were paired with habitat data collected using the Columbia Habitat Monitoring Program (CHaMP; BPA Project Number 2011-006-00) protocol.

Our assumption is that higher parr and redd densities correspond to better habitat. Observed densities at the survey-site scale (200–500 m) are rarely equal to a site's carrying capacity due to unmeasured or unaccounted-for variables. Quantile regression forest (QRF) models (Meinshausen 2006) are being used to address this. Random forest models have been shown to outperform more standard parametric models in predicting fish–habitat relationships in other contexts (Knudby et al. 2010). Quantile regression forests share many of the benefits of random forest models, such as the ability to capture non-linear relationships between the independent and dependent variables, naturally incorporate interactions between covariates, and work with untransformed data while being robust to outliers (Breiman 2001, Prasad et al. 2006). QRF models can also describe the entire distribution of predicted fish densities for a given set of habitat conditions, not just the mean expected density. Quantile regression models have been used in a variety of ecological systems to estimate the effect of limiting factors (Terrell et al. 1996, Cade and Noon 2003).

The habitat data used to develop the QRF models described here were largely collected by CHaMP (ISEMP/CHaMP 2015).[8] CHaMP sites are 200- to 500-m reaches within wadeable streams across select basins within the interior Columbia River Basin and were selected based on a spatially balanced Generalized Random Tessellation Stratified (GRTS) design (Stevens and Olsen 2004). CHaMP habitat data include, but are not limited to, measurements describing: channel units, channel complexity, fish cover, disturbance, riparian cover, size (depth, width, discharge), substrate, water quality, large woody debris, and temperature. Habitat data from the following CHaMP basins were used to develop the QRF models: Entiat, Grande Ronde (including Minam), John Day, Lemhi, Methow, Secesh, Tucannon, and Wenatchee. Additional habitat data collected beyond the scope of the CHaMP protocol (e.g., modeled temperature data) for each of the QRF models are described below.

Juvenile fish sampling was conducted for Spring/Summer Chinook parr during the summer at many of the same sites surveyed by CHaMP. Sampling was coordinated by a variety of entities depending on the location, including the Integrated Status and Effectiveness Monitoring Program (ISEMP), Oregon Department of Fish and Wildlife (ODFW), Columbia River Inter-Tribal Fish Commission (CRITFC), and the U.S. Fish and Wildlife Service (USFWS). Methods include mark–recapture, three-pass removal sampling, two-pass removal sampling, and single-pass electrofishing, as well as snorkeling.

These data were used to estimate summer parr abundance at all CHaMP sites where fish survey data were available. Three-pass removal estimates used the Carle–Strub estimator (Carle and Strub 1978), following advice from Hedger et al. (2013). Two-pass

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[8] https://www.champmonitoring.org

removal estimates used the estimator described by Seber and Schwarz (2002). Mark–recapture estimates used Chapman's modified Lincoln–Peterson estimator (Chapman 1951), and were deemed valid if they met the criteria described in Robson and Regier (1964). These estimates were made using the removal function from the FSA package (Ogle 2015) or the `closedp.bc` function from the `Rcapture` package (Rivest and Baillargeon 2014) in R software (R Core Team 2015).

Snorkel counts were transformed to abundance estimates using paired snorkel–electrofishing sites to calibrate snorkel counts.

Table 7.4. Number of unique sites in the initial dataset, by watershed, with paired fish–habitat data.

Watershed name	*n* sites
Entiat	32
John Day	21
Lemhi	13
Minam	10
South Fork Salmon	25
Upper Grande Ronde	77
Wenatchee	17
Total:	**189**

For sites with invalid estimates or that were sampled with a single electrofishing pass, a ratio estimator was developed to estimate the probability of capture. This estimator was based on a binomial generalized linear model that was developed separately for each subbasin and sampling entity. Possible covariates include how many fish were caught on the first pass, year, site length, and Julian day. After fitting all possible models with those covariates to data with valid abundance estimates, the model with the lowest AICc for each subbasin and sampling entity was chosen and used to predict abundances based on the number of fish caught on the first pass and any other covariates. Abundance estimates at all sites were then translated into linear and areal fish densities (fish/m and fish/m^2). For sites that were sampled in multiple years, only the fish and habitat data from the year with the highest observed fish density was retained to avoid possible pseudoreplication.

Table 7.4 shows the number of CHaMP sites with paired summer parr abundance estimates and habitat data used to develop the current parr summer capacity QRF model. Figure 7.9 shows the distribution of the log of fish density (fish/m^2) among those sites, colored by watershed.

Figure 7.9. Histogram showing the distribution of the log of fish density, colored by watershed.

The spring/summer Chinook redd data used to develop the current redd capacity QRF model were provided by the following groups:

- Idaho Department of Fish and Game (Lemhi).
- Nez Perce Tribe Department of Fisheries Resources (Secesh).
- Oregon Department of Fish and Wildlife (Minam, Upper Grande Ronde, John Day).
- U.S. Fish and Wildlife Service (Entiat).
- Washington Department of Fish and Wildlife (Tucannon, Wenatchee, Methow).

Data were compiled for 44,571 SPCH redds observed within CHaMP basins (Figure 7.10). Redd data span the years 1995–2016. For each redd, spatial coordinates (lat/long) were provided.
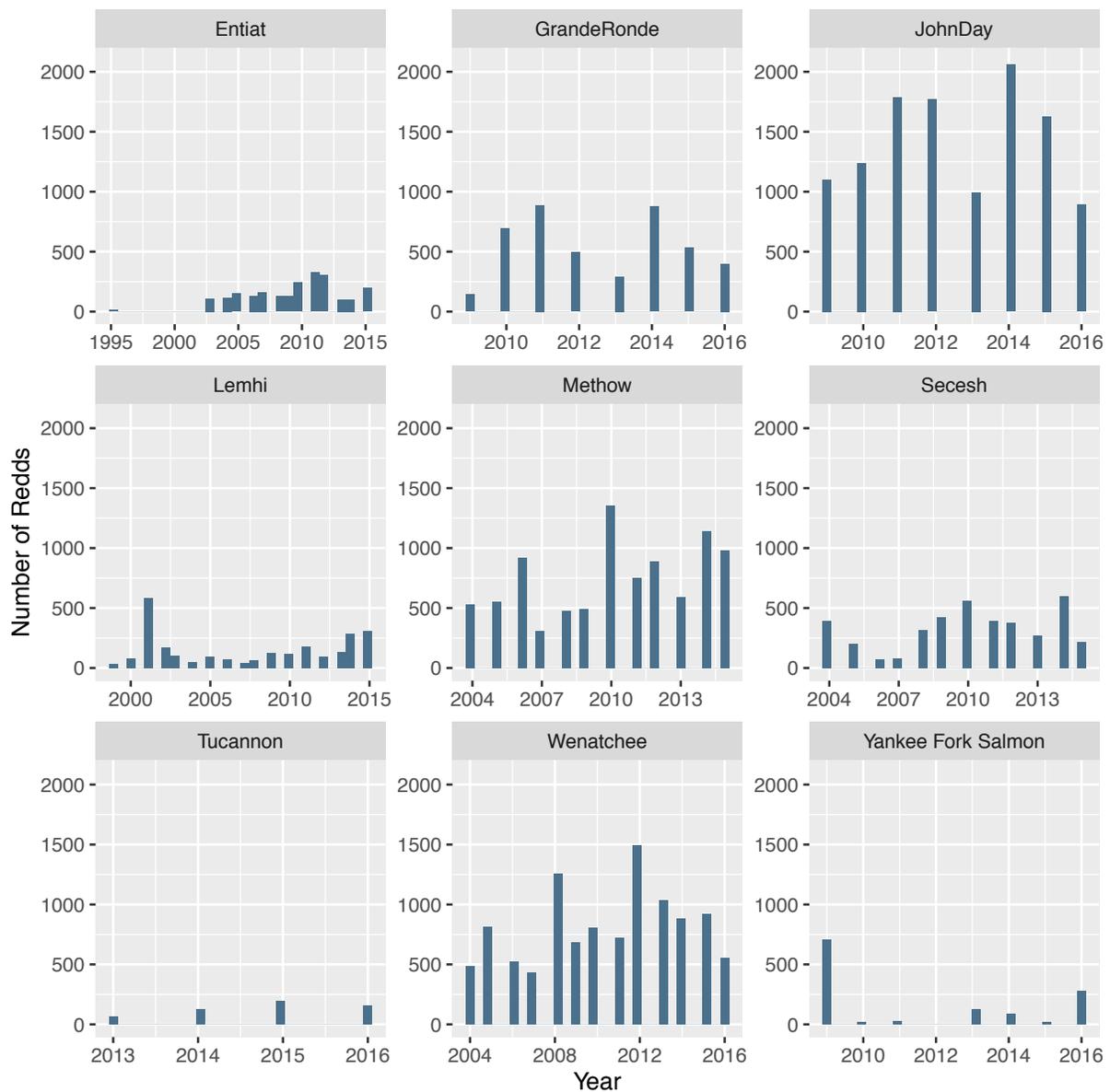
Figure 7.10. The number of SPCH redds observed, by year, for each of the CHaMP basins.

For the redd capacity QRF model, CHaMP habitat data were most recently downloaded on 7 Nov 2016. The full CHaMP dataset contains habitat data from a total of 2,430 CHaMP site visits. For CHaMP sites with multiple visits, the mean among site visit measurements was calculated for each habitat covariate. In total, habitat data for 816 unique CHaMP sites were available within CHaMP basins of interest (Figure 7.11).

Temperature data collected using in-stream temperature loggers were only available for a small portion of CHaMP survey sites. Therefore, modeled temperature data (McNyset et al. 2015) were used. Modelled temperature data summarizing the mean of eight-day means and the maximum of eight-day means for CHaMP sites for the period of 13 Aug through 14 Sep and for 2011–14 were developed. These dates were chosen as representative of the spring/summer Chinook spawning season in the interior Columbia River.

7.3.3.1. Variable selection

One of the key steps in developing a QRF model is selecting which habitat variables to use to predict fish capacities. Random forest models naturally incorporate interactions between correlated covariates, which is essential since nearly all habitat variables are considered correlated to one degree or another. However, redundant variables should be avoided. Further, including too many covariates may result in overfitting the model (e.g., including as many covariates as data points).

CHaMP collects more than 100 metrics describing the quantity and quality of salmonid spawning and rearing for each survey site. The Maximal Information-based Nonparametric Exploration (MINE) class of statistics (Reshef et al. 2011) were used to

Figure 7.11. The number of unique CHaMP sites within CHaMP basins of interest. In total, habitat data were compiled for 816 unique CHaMP sites within basins where observed redd data were also available. Note: CHaMP samples for the Minam River are lumped within the Grande Ronde.

determine those habitat characteristics (covariates) most highly associated with observed parr densities and observed redd abundance. MINE statistics are employed in the R package `minerva` (Albanese et al. 2013). Within the MINE class of statistics, the maximal information coefficient (MIC) was used to measure the strength of the linear or non-linear association between two variables (Reshef et al. 2011). The MIC value between each of the measured habitat covariates and the response variable (juvenile parr density or redd abundance) was used to inform decisions on which habitat covariates to include in the QRF capacity models.

7.3.3.2. Summer rearing capacity

Habitat metrics were first grouped into broad categories such as fish cover, large woody debris, complexity, riparian metrics, stream size, substrate, temperature, water quality, and channel unit makeup. Habitat metrics measuring volumes and areas were scaled to the wetted area of each site. Within each category, the association between each habitat metric and density of SPCH parr was calculated using MIC (Figure 7.12). We focused on areal fish density (fish/m^2) as the response variable. Our strategy was to select one or two variables with the highest MIC score within each category so that covariates describe different aspects of rearing habitat (e.g., substrate, riparian condition, large woody debris, cover, etc.). Based on the MIC statistics, the summer parr rearing capacity QRF model was fit using the habitat metrics described in Table 4.2.

7.3.3.3. Redd capacity

To determine which habitat metrics to include in the SPCH redd capacity model, the association between each habitat metric and maximum observed SPCH redd abundance (redds/km) was calculated using MIC (Figure 7.13). Moreover, pairwise correlations among the top ranked habitat covariates (per MIC) were considered to avoid using highly correlated or redundant metrics within the redd capacity model. Based on MIC results and the pairwise correlation among habitat covariates, five habitat covariates were included in the SPCH redd capacity QRF model (Table 7.5).
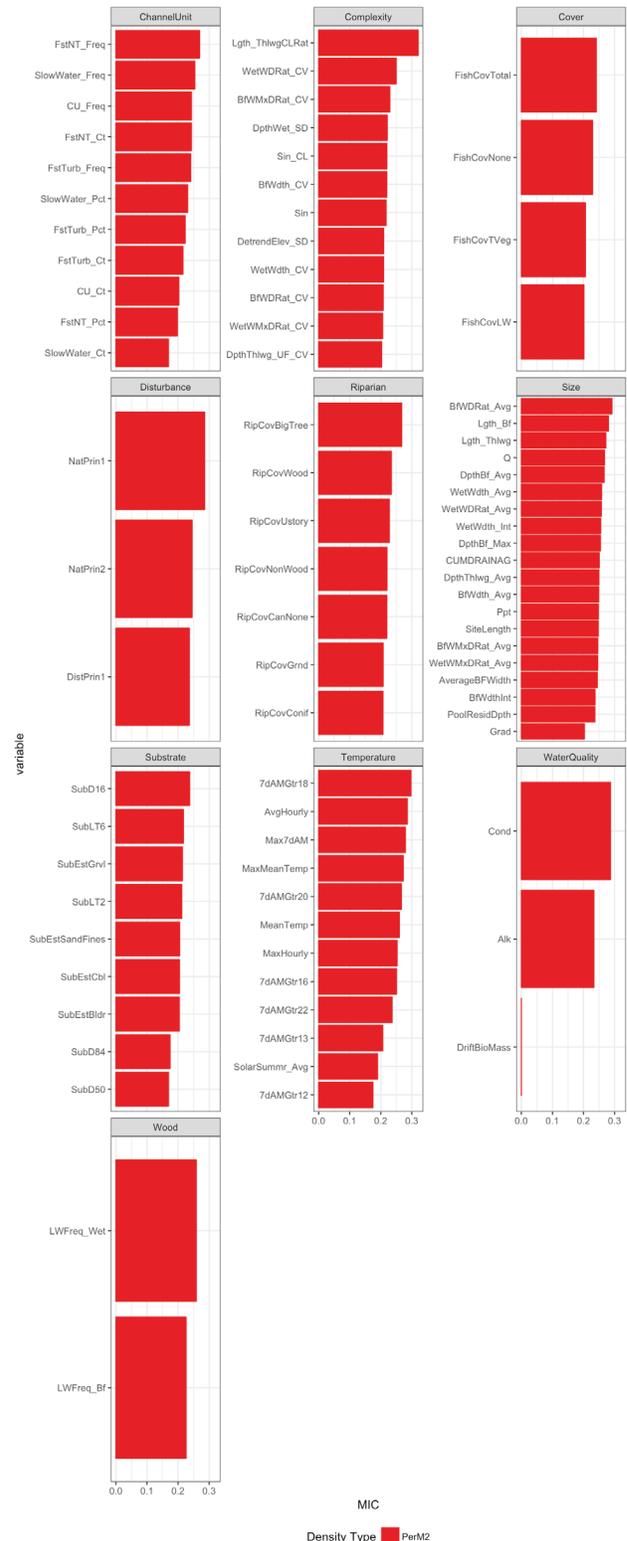
Figure 7.12. Bar plot showing the strength of association between each habitat metric and observed densities of SPCH parr (fish/m^2) during the summer, faceted by habitat categories. Results are used to determine habitat covariates to include in the summer parr capacity QRF model.

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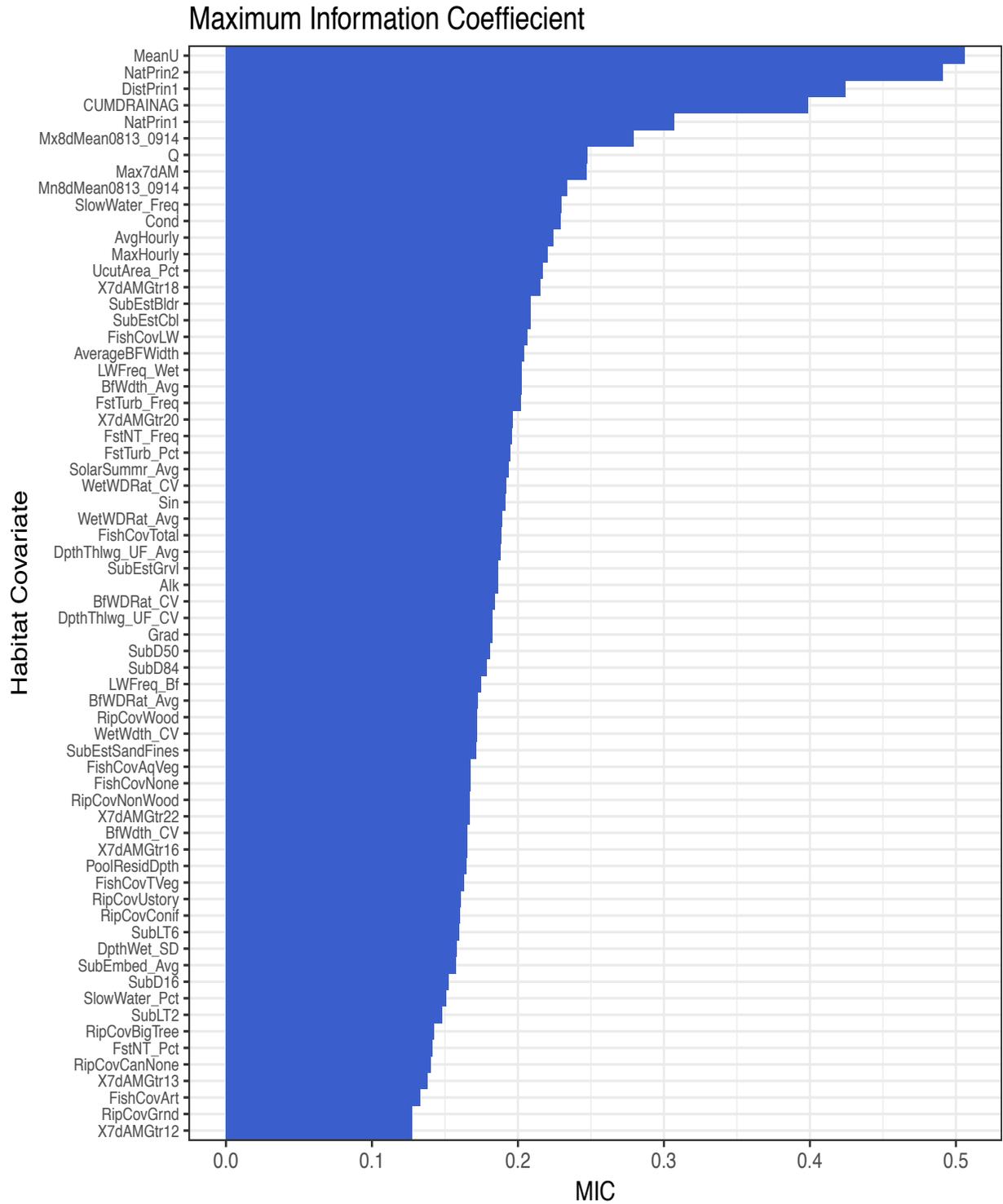
Figure 7.13. Maximal information coefficient (MIC) value for 66 habitat covariates considered, ranked to display the strength of association with the response variable (maximum number of redds observed).

Table 7.5. Habitat covariates selected for the current SPCH redd capacity QRF model.

Metric	Description
MeanU	Average annual discharge (cfs). Mean daily flow, averaged over one year, calculated from the <u>FLoWS model network</u>.[a]
Elev_M	Elevation (m).
DistPrin1	Disturbance index, including % urban, % agricultural, % impervious surface, and road density.
Mx8dMean0813_0914	Maximum of eight-day mean temperatures for the period of 13 Aug–14 Sept averaged across 2011–14.
SubEstGrvl	Percentage of coarse and fine gravel (2–64 mm) within the wetted site area.

[a] http://www.fs.fed.us/rm/boise/AWAE/projects/SpatialStreamNetworks.shtml

7.3.4. QRF models

QRF models allow one to visually examine the marginal effect of each habitat covariate on the quantile of interest through partial dependence plots (PDP). These plots show the marginal effect of changing a single habitat covariate while maintaining all other covariates at their mean values. QRF models can also predict habitat capacity at all sites where such habitat data is available (e.g., at CHaMP sites). Using the selected habitat covariates, QRF models (Meinshausen 2006) were fit for SPCH summer parr and redd capacity, respectively. QRF models combine the flexibility of random forest models (Breiman 2001) with the ability of quantile regression to extract relationships between quantiles of the data other than the mean (Cade and Noon 2003). Random forests can account for nonlinear relationships between the response and predictor variables, and naturally incorporate interactions between the predictor variables, two common features of ecological datasets (Liaw and Wiener 2002). After constructing a random forest, predictions of the mean response can be made by averaging the predictions of all the trees, similar to the expected value predictions from a statistical regression model. However, the individual predictions from each tree, viewed collectively, describe the entire distribution of the predicted response. Therefore, the random forest model can be used in the same way as other quantile regression methods to predict any quantile of the response. QRF models were fit using the `quantregForest` function from the `quantregForest` package (Meinshausen 2016) in R software (R Core Team 2015). For both models, the 90th quantile of the predicted distribution was used as a proxy for carrying capacity. One reason for the 90th quantile, instead of something higher, is to avoid using predictions that are aimed at the very upper tails of observed fish density, which may be influenced by sampling issues.

7.3.4.1. Summer rearing capacity

Summer parr abundance (and density) data and habitat data were paired up by site and year, and duplicate habitat visits within a year were removed. There were some missing values within the habitat dataset. Any site visit with more than three missing covariates was dropped from the analysis; the remaining missing values were imputed using the `missForest` R package (Stekhoven and Bühlmann 2012, Stekhoven 2013). In the end, we used 186 site visits and 14 habitat covariates (13.3 data points per covariate) to fit the summer parr capacity QRF model.
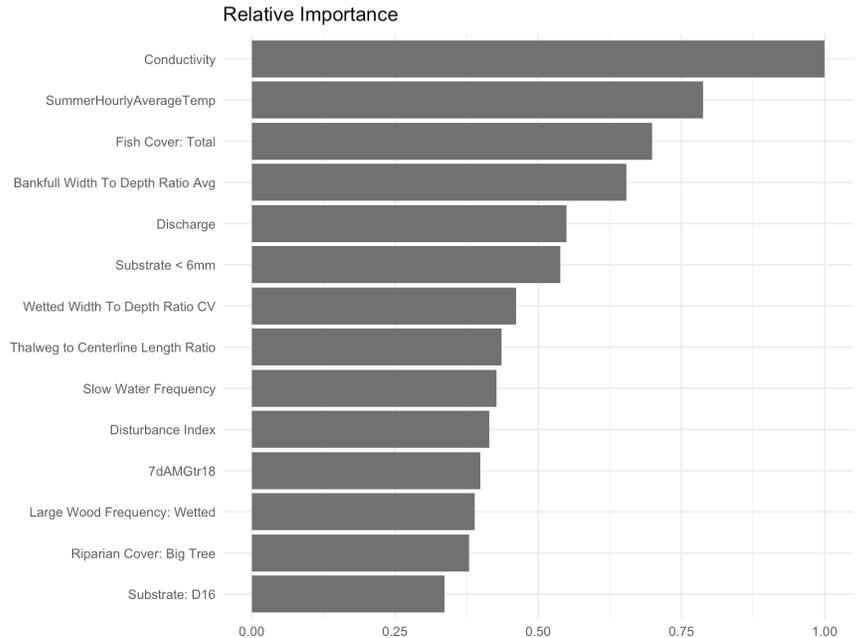
Figure 7.14. Relative importance of each of the habitat covariates included in the SPCH summer parr capacity model.

The results of the QRF summer parr capacity model match many biological expectations. The relative importance of the 14 habitat covariates within the model are shown in Figure 7.14. The general shape of partial dependence plots, which show the marginal effect of how predicted capacity changes as each habitat covariate changes, assuming all other covariates remain at their mean value, confirm that lower conductivity and more riparian ground cover (both of which suggest more undisturbed areas), deeper sites, higher bankfull width CV (an indicator of higher stream complexity), more disturbed areas, and more large woody debris lead to higher estimates of parr capacity (Figure 7.15). As the max temperature rises, so does capacity, until an upper threshold is reached and carrying capacity plummets. Other marginal fish–habitat relationships are not as straightforward, but many of these habitat metrics are correlated, and therefore, some marginal relationships may not depict anything seen in the field.

Partial Dependence Plots
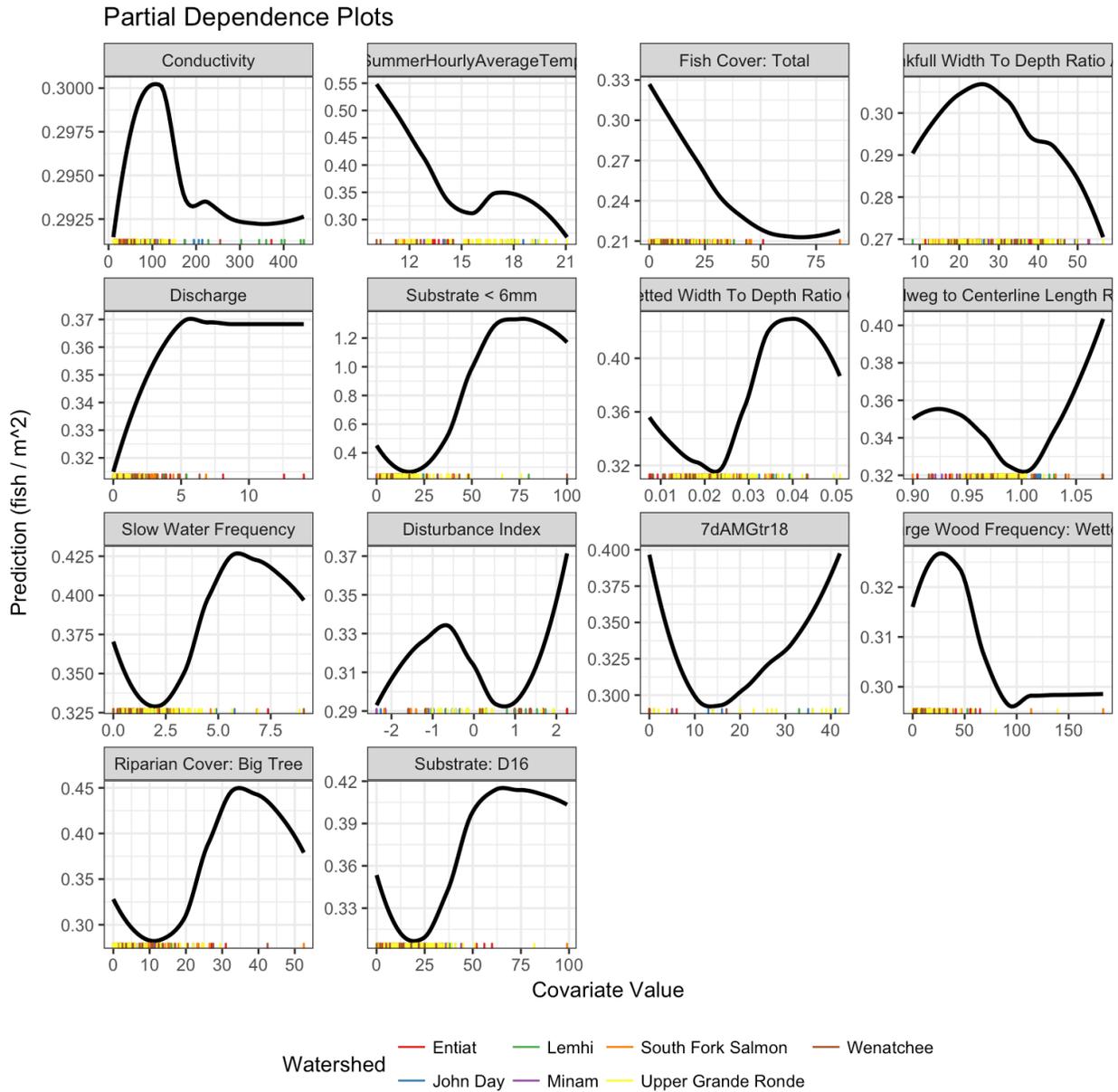
Figure 7.15. Partial dependence plots for the SPCH summer parr capacity QRF model, depicting how parr capacity shifts as the habitat metric changes, assuming all other habitat metrics remain at their mean values. Tick marks along the x-axis depict observed values and the subbasins they came from.

7.3.4.2. Redd capacity

Habitat data were initially available for 816 unique CHaMP sites; for each site, habitat measurements were averaged among site visits. Of those 816 unique CHaMP sites, 369 occurred within a stream in which redds have been observed and were used to fit the SPCH redd capacity QRF model. There were some missing values in the habitat dataset. Any site missing more than five covariates was removed from the analysis; the remaining missing values were imputed using the `missForest` R package (Stekhoven and Bühlmann 2012).

The relative importance of each of the five habitat covariates are shown in Figure 7.16. The QRF model allows one to examine the marginal effect of each habitat covariate on the quantile of interest of the response variable using PDPs. Figure 7.17 shows the relationship between each of the habitat covariates and a prediction of redd abundance while holding all other habitat covariates at their mean value.
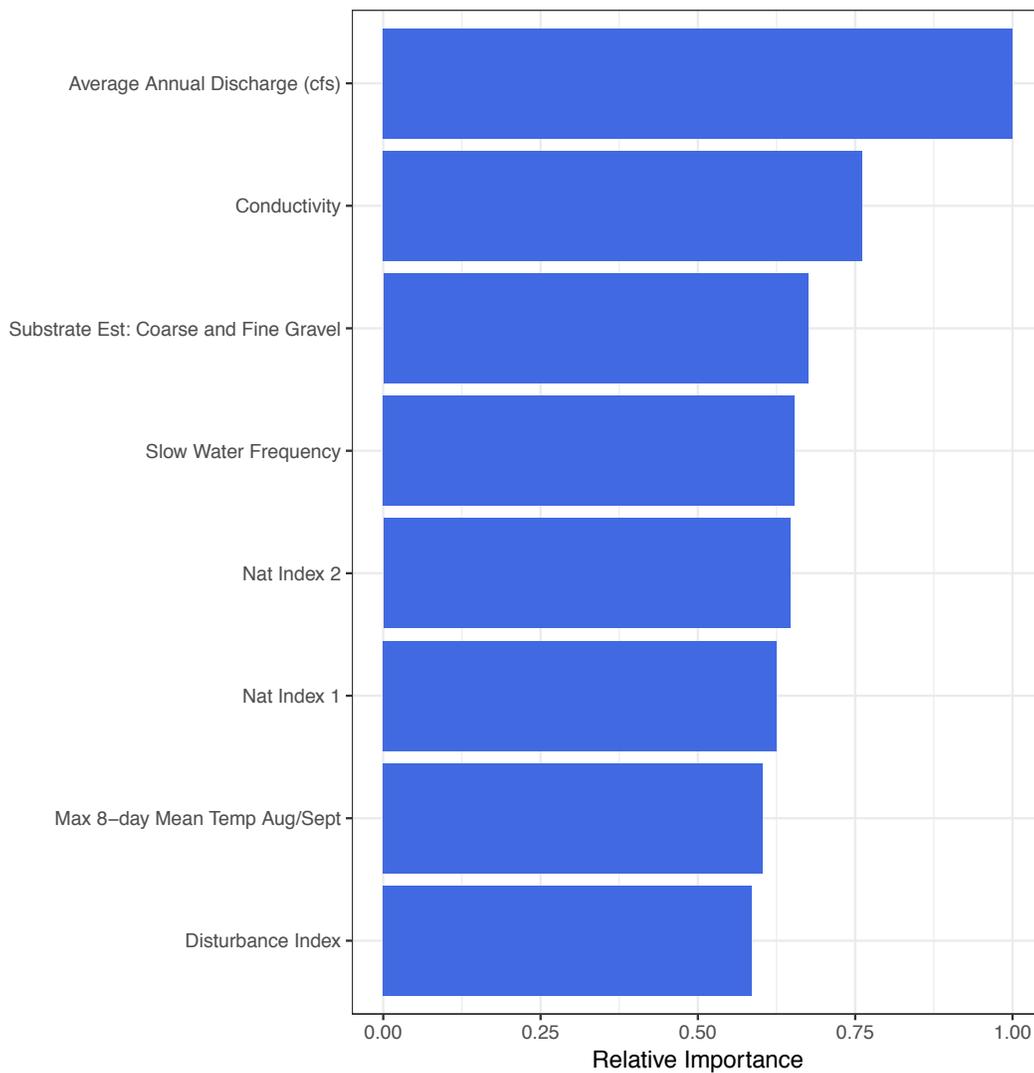
Figure 7.16. Relative importance values for each of the habitat covariates included in the redd capacity QRF model.
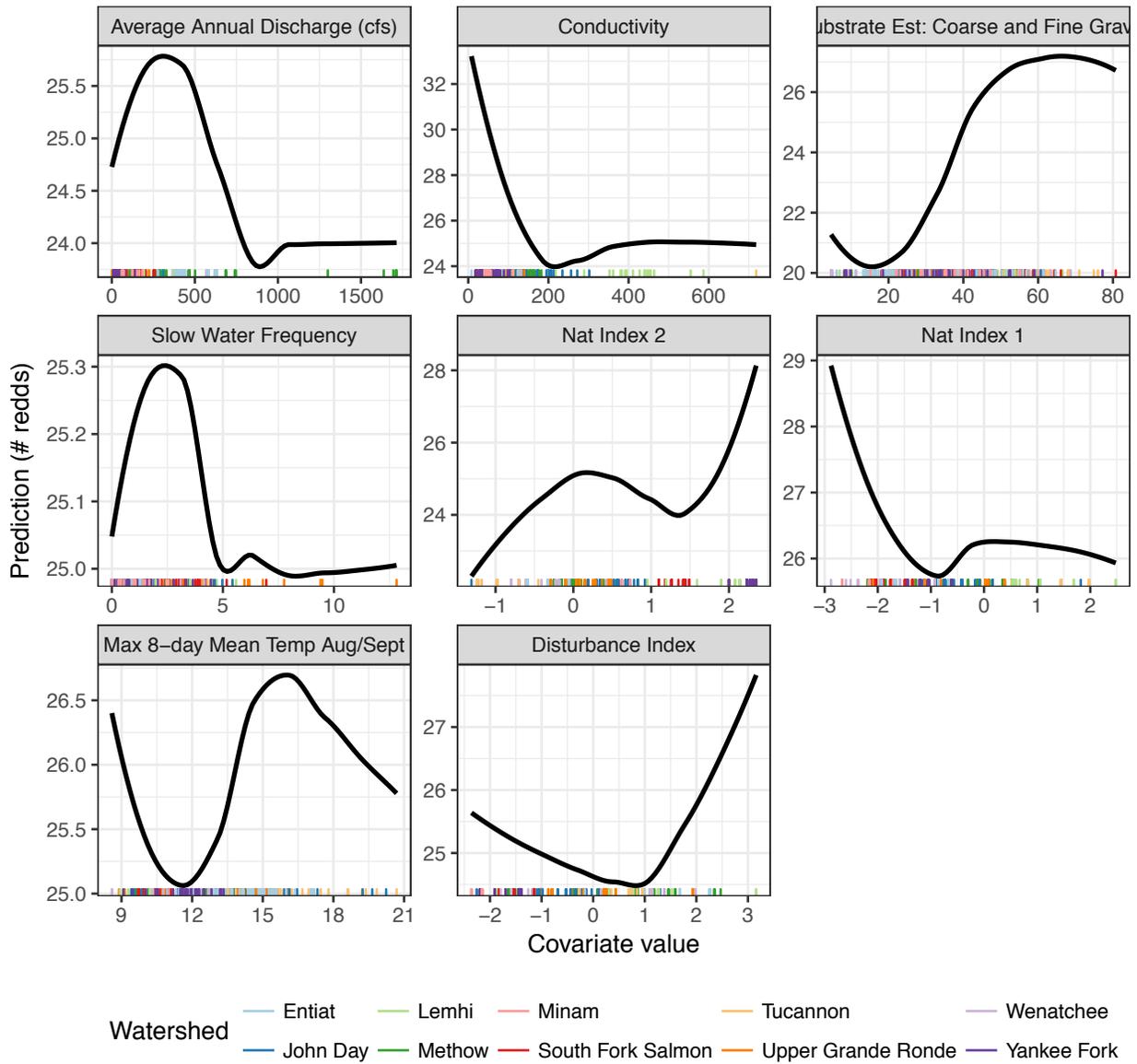
Figure 7.17. Partial dependence plots depicting how redd capacity shifts as the habitat covariate changes, assuming all other covariates remain at their mean values. Tick marks along the x-axis depict observed values and the subbasins they were recorded in.

7.3.4.3. Site-based predictions

After model fitting, QRF models can be used to predict capacity at all CHaMP sites using the habitat covariates that were used to fit the model. For CHaMP sites that have been sampled in multiple years, the mean of the habitat metrics among years was calculated to make predictions. The 90th quantile of predicted fish density was used as a proxy for carrying capacity. Using the SPCH summer parr capacity QRF model, predictions of parr capacity were made for CHaMP sites within the Lemhi River subbasin (Figure 7.18, left). Using the SPCH redd capacity QRF model, predictions of redd capacity were made for CHaMP sites within the Lemhi River (Figure 7.18, right).
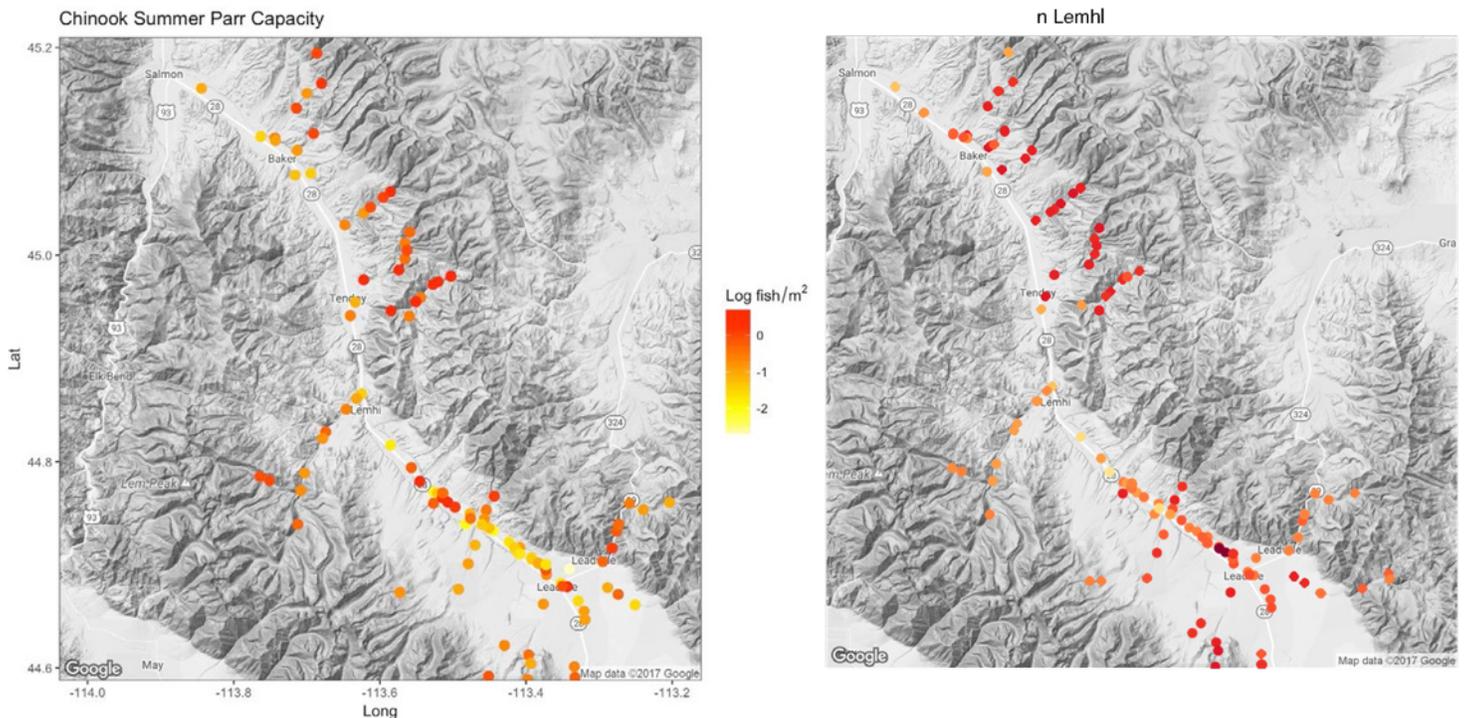
Figure 7.18. Predictions of carrying capacity at CHaMP sites in the Lemhi River. (left) Juvenile capacity (fish/m^2). (right) Redd capacities at CHaMP sites. Each prediction is for the 1 rkm surrounding the x-site for each of the 116 CHaMP sites.

7.3.5.   Extrapolation from site to watershed and application to unsampled watersheds

Predictions of habitat capacity have been made at all CHaMP sites within the interior Columbia River basin using the fitted quantile regression forest (QRF) models for both parr summer rearing and redd capacity for SPCH. To estimate capacity at larger scales (e.g., watershed, population), an extrapolation model was developed using globally available attributes (GAAs) from the list of master sample sites that CHaMP sites were originally selected from. The natural log of the CHaMP site predictions was used as the response variable for the extrapolation model. The extrapolation models use a multiple linear regression model that incorporates the design weights of the CHaMP sites using the `svyglm` function from the `survey` package (Lumley 2004, 2016) in R software (R Core Team 2015).

To summarize capacity at larger spatial scales, the mean linear capacity (e.g., fish/m or redds/m) of the master sample points within a given spatial scale is first determined. Only master sample points within the domain of SPCH (as determined by <u>StreamNet</u>[9]) are used. Mean estimates within that scale are then multiplied by the length of the stream within the SPCH domain. The GAAs used within each of the extrapolation models are shown in Table 7.6.

Table 7.6. GAA habitat covariates used for QRF model extrapolations.

Covariate	Scale	Unit	Summer rearing capacity CHaMP (per m)	Non-CHaMP (per m)	CHaMP (per m²)	Non-CHaMP (per m²)	Redd capacity CHaMP (per m)	Non-CHaMP (per m)
Mean Annual Velocity	Reach, 2 km	m/s	✗	✓	✓	✓	✗	✓
Slope	Reach, 2 km	m/m	✗	✗	✓	✓	✗	✓
Drainage Area (sqrt)	Reach, 2 km	sqrt km²	✗	✓	✓	✗	✓	✓
Stream Power	Reach, 2 km	N/s[a]	✗	✗	✗	✗	✓	✗
BFW	Site, 300 m	m/m	✓	✗	✓	✓	✗	✓
Channel Type	Site, 300 m	n/a	✓	✓	✗	✓	✓	✓
Temperature Range	Reach, 2 km	°C	✗	✗	✗	✓	✗	✓
Growing Degree Day	Reach, 2 km	GDU[b]	✗	✓	✓	✓	✓	✓
Precipitation	Reach, 2 km	cm	✗	✗	✗	✓	✗	✓
Elevation	Site, 300 m	m	✓	✓	✓	✓	✗	✓
CHaMP Watershed	Region	n/a	✓	✗	✓	✗	✓	✗
Disturbance Class PCA1	Watershed, HUC12	n/a	✓	✓	✗	✓	✗	✓
Natural Class PCA1	Watershed, HUC12	n/a	✗	✓	✓	✓	✗	✗
Natural Class PCA2	Watershed, HUC12	n/a	✓	✗	✗	✗	✓	✓

[a] Newtons per second.
[b] Growing degree unit.

To select the best extrapolation model, models with all possible combinations of GAAs were fit and the model with the lowest AIC score was selected. After model selection, the remaining attributes were used to predict SPCH parr capacity at all master sample points. For each response, two models were developed, one for sites within CHaMP watersheds, and one for all other sites, for a total of four extrapolation models for summer parr rearing. Summaries of extrapolation model fit are shown in Table 7.7. Predictions of fish/m² were translated into fish/m, and both responses were used separately to fit different extrapolation models. To estimate total capacity at the watershed scale, the mean capacity density (parr/m) for all master sample points in a watershed was multiplied by the stream length of that watershed. The master sample points and the stream length were filtered to only include areas within the range of SPCH, as defined by StreamNet.

Table 7.7. Summary of model fit for each of the SPCH summer parr rearing capacity QRF extrapolation models.

Model	Response	R^2	Adjusted R^2
CHaMP	per m	0.493	0.466
Non-CHaMP	per m	0.398	0.374
CHaMP	per m²	0.458	0.434
Non-CHaMP	per m²	0.407	0.382

7.3.5.1. Redd capacity

After selection of an extrapolation model, the remaining attributes were used to predict SPCH redd capacity (redds/m) at all master sample points. Two models were developed, one for sites within CHaMP basins, and one for other sites. Summaries of extrapolation model fit are shown in Table 7.8.

Table 7.8. Summary of model fit for the SPCH redd capacity QRF extrapolation models.

Model	R^2	Adjusted R^2
CHaMP	0.586	0.566
Non-CHaMP	0.497	0.477

To estimate total redd capacity at the watershed scale, the mean redd capacity (redds/m) for all master sample points in a watershed was multiplied by the stream length of that watershed. The master sample points and the stream length were filtered to only include areas within the range of SPCH, as defined by StreamNet.

7.4. Parsing QRF capacity estimates by reach type and geomorphic condition

Parr and redd capacity estimates where generated by the QRF modeling approach for the entire Spring/Summer Chinook spawning and rearing network within the Upper Salmon MPG. All reaches of the stream network in the Upper Salmon Spring/Summer Chinook MPG have been classified into Reach Type and Geomorphic condition. The stage-specific Beverton–Holt-based population life cycle models are based on a capacity and productivity estimate for each life stage. For the USAL Chinook populations modeled, stage-specific survival data have been generated in the Lemhi River basin.

Variation in stream habitat, both in terms of quality and quantity, impacts the degree of utilization by juvenile and adult salmonids. As such, stage-specific capacity and survival will vary along a natural gradient corresponding to the amount of habitat available on a reach-by-reach basis. These demographic terms with also vary along a gradient of habitat quality resulting from anthropogenic impacts. These gradients in habitat quality have been captured by the RT and GC descriptions of the USAL river network.

Linking the RT/GC and QRF was done over the entire USAL domain (entire stream network upstream from the confluence of Panther Creek and the mainstem Salmon River). The estimated parr and redd capacity values were summarized by RT × GC. That is, average parr and redd capacity was calculated for each combination of Reach Type ($n = 34$) and Geomorphic Condition ($n = 4$). Not all combinations of RT × GC are present in the USAL, but every reach had RT, GC, and capacity values. The capacity data were summarized by RT and GC in two manners, by RT and then as a departure from a GC of "good" for each RT. First, the RT-specific average and standard deviation of capacity for "good" RC reaches in the USAL domain was calculated (Table 7.9). The capacity for each RT in the "good" GC state forms the baseline for the value to parr and

Table 7.9. Reach type-specific average and standard deviation of capacity for "good" GC reaches in the USAL domain.

Geomorphic condition	Parr per m²	Redds per m²
Functioning	1.905	1.027
Good	1.000	1.000
Moderate	0.859	0.890
Poor	0.698	0.855

spawners for each RT. Since RT is highly unlikely to change, but the GC state of a reach evolves with restoration, the modeling framework must accommodate both the spatially explicit description of tributary habitat and its change through time. Just as the capacity of "good" GC state reaches can be evaluated from the reach typing and QRF capacity data sets, so can the value of "functioning," "moderate," and "poor" states. Using the entire USAL data set of 1,786 reaches across all 34 RTs, the relative capacity for either parr or redds per unit area was compared between GC states. Capacity values of each RT increase or decrease multiplicatively based on the relative capacity of all RTs by their GC state, normalized to a GC state of "good" (Table 7.10). That is, for "poor" or "moderate" GC states, the condition factor multiplier was less than 1, and for "functioning" GC states, it was greater than 1. Thus, the condition factor multiplier is not RT-specific; rather, it is generic for all state changes, with the RTs each having their own specific capacity for juveniles and redds per unit area.

Table 7.10. Capacity value of each reach type based on the relative capacity of all reach types by their geomorphic conditions, normalized to "good."

Reach type, GC = "good"	Parr		Redds	
	Parr per m^2	Standard error	Redds per m^2	Standard error
CV_BedrockCanyon	0.321	0.074	0.360	0.055
CV_BoulderBed	0.211	0.094	0.433	0.022
CV_FanControlled_GravelBed	0.271	0.060	0.315	0.042
CV_GravelBed	0.280	0.112	0.455	0.002
CV_OccFloodplainPockets	0.348	0.126	0.415	0.023
CV_SteepPerennialHeadwater	0.202	0.094	0.460	0.000
CV_SteepSubalpineHeadwater	0.178	0.121	0.460	0.000
CV_StepCascade	0.127	0.116	0.437	0.014
PCV_FanControlled_GravelBed	0.297	0.094	0.424	0.004
PCV_LowSinPlanControlled_DFP	0.461	0.104	0.456	0.009
PCV_LowSinPlanControlled_GravelBed	0.380	0.113	0.440	0.012
PCV_LowSinPlanControlledAnabranching	0.374	0.122	0.460	0.000
PCV_LowSinWandering_GravelBed	0.346	0.133	0.377	0.023
PCV_MarginControlled_Anabranching	0.571	0.125	0.460	0.000
PCV_MarginControlled_CobbleBed	0.358	0.048	0.393	0.023
PCV_MarginControlled_DFP	0.422	0.103	0.330	0.037
PCV_MarginControlled_GravelBed	0.383	0.088	0.396	0.011
PCV_MeanderingPlanControlled_DFP	0.588	0.136	0.446	0.010
PCV_PlanControlled_CobbleBed	0.840	0.185	0.460	0.000
PCV_PlanControlled_GravelBed	0.447	0.110	0.426	0.018
UCV_AlluvialFan	0.438	0.135	0.448	0.008
UCV_Anastomosing_GravelSandBed	0.651	0.155	0.446	0.024
UCV_LowSin_GravelBed	0.516	0.137	0.428	0.024
UCV_LowSin_SandBed	0.204	0.104	0.292	0.052
UCV_LowSinAnabranching	0.495	0.126	0.426	0.029
UCV_LowSinWandering_GravelBed	0.494	0.101	0.350	0.024
UCV_LowSinWandering_SandBed	0.150	0.089	0.267	0.063
UCV_Meandering_FineGrained	0.425	0.174	0.448	0.046
UCV_Meandering_GravelBed	0.590	0.141	0.426	0.023
UCV_MeanderingBeaverInfluenced_GravelBed	0.710	0.166	0.446	0.011

7.5. Incorporating recent tributary habitat restoration actions (2009–15)

Across the Pacific Northwest, both public and private groups are working to improve riverine habitat for a variety of reasons, including improving conditions for threatened and endangered salmon. Federal, tribal, state, and local efforts fund and collect project-level data on restoration actions. The goals of each of these groups are diverse, and this diversity has led to heterogeneity of data formats in use. In an attempt to make this diversity of effort accessible to management decision-makers, we created a standardized data dictionary of project types now being applied throughout the region and assemble project records into a database of restoration actions (Pacific Northwest Salmon Habitat Project Database, PNSHP; Barnas et al. 2015). The PNSHP was designed specifically to address the needs of regional monitoring programs that evaluate the effectiveness of restoration. Thus, minimum requirements for inclusion in the database are: project type, location, agency/ organization, and year or date. Large data contributors include both state and federal agencies, e.g.: the Washington State Salmon Recovery Funding Board, the Oregon Watershed Enhancement Board, the U.S. Forest Service, the U.S. Bureau of Land Management, and the Bonneville Power Administration. The database currently (2018) contains spatially referenced, project-level data on over 40,000 restoration actions initiated at over 100,000 locations in the last 25 years in the states of Washington, Oregon, Idaho, and Montana. Data sources include federal, tribal, state, local, and NGO contributors.

For the Upper Salmon Spring/Summer Chinook ESU, we spatially queried PNSHP for all project worksites in the area of interest, and based on location assigned each project worksite to one or more populations within the ESU for the time interval 2009–15. These projects, along with all available attributes, were then spatially joined to the RT/GC and capacity network data sets. Figure 7.19 is an example of the coregistered data displayed just for the Lemhi River basin.

While the PNSHP data system represents a spatially and temporally extensive picture of tributary restoration actions across the Pacific Northwest, individual records contain minimal restoration project specific information other than a location, start date and membership in broad project type categories. Therefore,
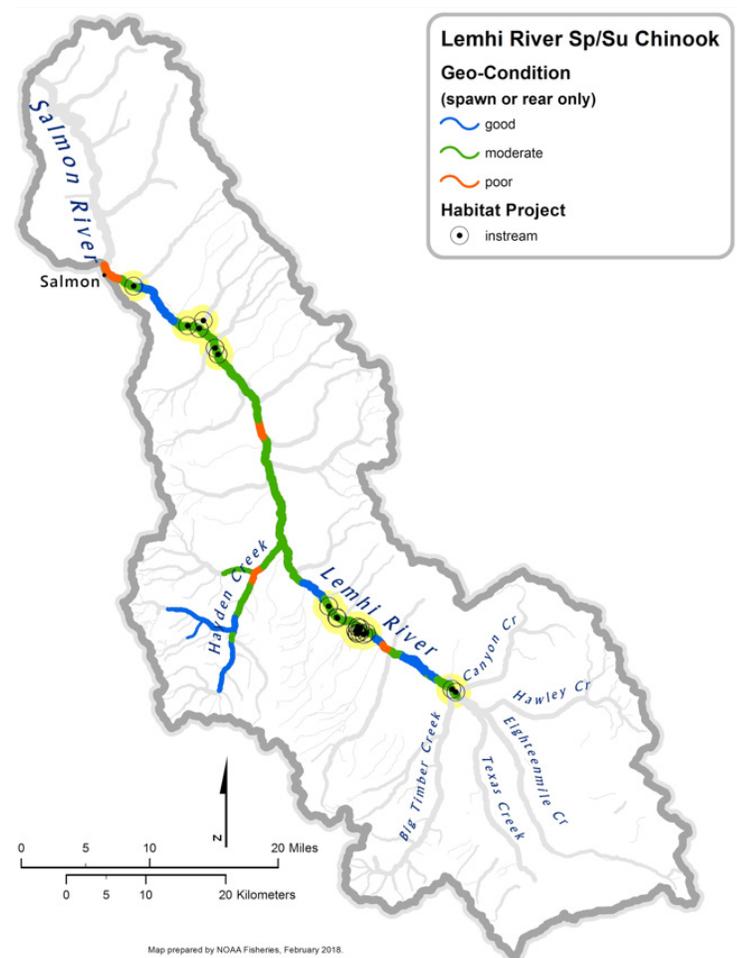
Figure 7.19. Lemhi River basin, showing in-stream habitat project locations, reach geomorphic condition, and spatial extent of spawn–rear habitat.

to use this rich data source as a driver of fish habitat condition change in the Upper Salmon LCMs required us to make several standardizing assumptions. First, only project types that could be expected to directly modify stream conditions were considered. Thus, in-stream habitat complexity actions and habitat access actions could be incorporated into an estimate of habitat change, while riparian planting, upland restoration, and water conservation actions could not be included. Secondly, since details of the extent and actual activities associated with each project were not available, all actions were standardized to have the same magnitude of impact in that each reach containing one or more projects was improved a single Geomorphic Condition step. For example, in Figure 7.19, each "in-stream" project location corresponds to both a specific project meeting the date and type considerations described above, and a reach within the spawning and rearing range of spring/summer Chinook that can be improved through a change in its geomorphic condition state.

7.6. Incorporating future actions in key Upper Salmon River Chinook populations

For Chinook populations in the Upper Salmon MPG, the potential benefit of future tributary actions was estimated based on distributing a similar level of effort during the recent past (2009–15) at the MPG level to four focal populations: Lemhi, Pahsimeroi, Upper Mainstem, and Yankee Fork. The USAL Chinook LCM incorporates the quality and quantity of tributary habitat with respect to extent (area) and geomorphic condition. Therefore, restoration actions that increase the extent (e.g., access) and geomorphic condition (in-stream complexity, floodplain reconnection) can be directly modeled. The forecast level of habitat restoration action at the scale of the entire USAL Chinook MPG was 10 miles of stream complexity improvement and 16 miles of habitat access. These levels of effort were distributed across the focal populations evenly, splitting the habitat access effort four ways but the habitat complexity improvement only three ways, as the current habitat status in the Pahsimeroi is of sufficiently high quality that additional in-stream work is not warranted (Table 7.11).

Because the increase in habitat quality (improving geomorphic condition) and the condition of the stream habitat made available by the access projects is not specified by the MPG total level of effort, several assumptions were applied to the distribution of effort in order to estimate the potential capacity benefit for both redd deposition and juvenile rearing. Habitat access projects were assumed to open habitat of representative quality, that is, additional

Table 7.11. Distribution of future habitat restoration (2019–21) actions across the four focal populations of the Upper Salmon River Chinook MPG.

	Complexity, channel length (km)	Complexity, area (m²)	Current available spawn/ rear area (m²)	Relative project area (%)	Access, channel length (km)	Access, spawn/ rear area (m²)	Relative project area (%)	Estimated increase in redd capacity (%)	Est. increase in rearing capacity (%)
SRLEM	4.0	44,000	1,500,711	2.9	6.4	70,400	4.7	5.0	5.5
SRPAH	0.0	0	594,315	0.0	6.4	81,280	13.7	13.7	13.7
SRUMA	4.0	18,400	730453	2.5	6.4	29,440	4.0	4.1	5.3
SRYFS	4.0	12,880	453497	2.8	6.4	20,608	4.5	4.6	6.4

habitat was added to the total available for spawning and rearing Chinook salmon in the same proportions of type and quality that are currently available. Therefore, the resultant change reflects a simple dilation of the current habitat in a watershed. The complexity actions, however, were applied to improve the quality of habitat only currently in moderate or good condition. That is, no improvement was made to reaches in poor condition. The rationale for this assumption was that greater biological benefit results from improving moderate and good habitat, and so strategic plans would be more likely to adopt project siting rules that maximize the benefit of in-stream actions. As such, the resultant change in redd and rearing capacity was greater than what could have been achieved by simply applying quality improvements at random across a watershed. The access and complexity improvements were treated independently, but first habitat quantity was added via simulated access actions, and then reach conditions were improved; the resulting habitat quality and quantity was then used to estimate the watershed redd and rearing juvenile capacity.

7.7.   Estimating population-level benefits of tributary habitat restoration actions

Across Chinook population watersheds in the Upper Salmon River basin, tributary restoration actions are generally meant to increase the quality and quantity of summer rearing habitat for parr. Spawning habitat is not thought to be limiting in any of the nine Chinook population watersheds in the Upper Salmon River MPG; however, significant habitat degradation due to mining activity in Panther Creek, North Fork, and Yankee Fork dramatically reduces potential high quality spawning areas, and extensive dewatering due to irrigation withdrawals in the Lemhi and Pahsimeroi also reduces the extent of accessible habitat. Several studies in the Lemhi (Bjorn et al. 1977) and Clearwater Rivers (Hillman et al. 1987) point to overwinter habitat availability as potentially limiting, in particular, impacting the proclivity of summer parr to overwinter in their natal tributary environment rather than migrating to the mainstem Salmon River six months before beginning the smoltification process and their downriver migration at a year post-emergence. Understanding the role tributary habitat quality and quantity may play in structuring the population dynamics of these populations through capacity or survival limitations of freshwater life stages, or the expression of life history diversity, is a component of the management and recovery strategy development for this MPG. While the status of these populations is monitored by state and tribal fisheries agencies, and the factors thought to be limiting population growth rates have been summarized as part of the Interior Columbia river Basin Technical Recovery Team's viability assessments (NMFS 2007) and the development of ESA Recovery Plans (NMFS 2007, 2009, 2017), currently, no broad-scale assessment of habitat condition as a determinant of salmonid population processes exists for the Upper Salmon River MPG. Therefore, as a means to estimate the potential biological benefit of changes to tributary habitat, life cycle models were applied to eight of the nine Chinook populations in the Upper Salmon River MPG. The goal of the work was to develop a management decision support platform that could be used to explore the potential population scale of reach-scale habitat management actions. The LCM framework acts to aggregate the impacts of habitat actions over time and space, but also is the formal structure though which stage-specific fish–habitat relationships are aggregated into a full life cycle impact by projecting population behavior through time.

7.8.   Life cycle model scenarios

Given the lack of consistent and comprehensive habitat status information for all Chinook population watersheds in the Upper Salmon River MPG, building spatially explicit life cycle models requires a series of assumptions and compromised. In this case, as was outlined in the methods sections above, habitat quality and quantity for the entire MPG represented through the application of river styles were based reach typing and geomorphic assessments. These reach classifications were based on detailed on-the-ground surveys in two watersheds (Lemhi and Yankee Fork), and the classification structure developed in these basins was then applied across the remainder of the MPG. Similarly, fish–habitat relationships were developed at locations where both detailed habitat data collection and adult and juvenile fish surveys were performed (a subset of CHaMP sites across the Columbia River basin) and then extended through quantile regression forest modeling to all reaches. An association between the habitat classification framework and the habitat capacity estimates was developed for the Upper Salmon River Chinook MPG, thereby allowing the development of spatially explicit life cycle models for all populations. Four tributary habitat-specific scenarios were developed across the eight Chinook populations: baseline condition, recent past restoration actions, random habitat quality improvements, and more directed habitat quality and quantity modifications. Note that the Lower Mainstem population was not done due to data quality issues—there is a mismatch between the habitat condition and fish capacity estimated having been developed from wadeable stream reaches only, and the Lower Mainstem being primarily a main channel-based population. Future work will develop equivalent habitat and fish metrics to allow the development of life cycle models for this population.

7.8.1.   Baseline habitat condition

The population-specific life cycle models for Upper Salmon River Chinook were developed to represent a baseline environmental condition existing in the late 2000s. The habitat assessments applied across the entire MPG were performed beginning in 2007, with the methodology being fully implemented in 2011. Base adult and juvenile capacity and survival relationships for the mainstem Snake and Columbia River and ocean rearing phases were developed from PIT tag-based mark-recapture data over the period of 1990–2010. The model's output of population abundance (e.g., natural-origin spawners) is constrained by the observations of Lower Granite Dam to Lower Granite Dam return rates and the redd surveys and juvenile outmigrant monitoring done in most of the population watersheds. As a result, calibration of the full life cycle model is straightforward. All populations with sufficiently long historical adult and juvenile abundance time series were used to calibrate the life cycle model output. Using pre-2012 adult and pre-2010 juvenile data as the "observed" data, an "estimated" data set was generated from a suite of model runs for each population by varying parr survival over a wide range of values. Estimated adult and juvenile abundance values that had a greater than 95% likelihood of being drawn from the same distribution as the observed data were noted; the parameter combinations that resulted in these model outputs were then used as the basis for all future model runs of that population—the rationale behind this pseudo-Bayesian parameter estimation method being that the model is a realistic approximation of a biological process, and thus, if the model output mimics the observed output of the natural biological process, a parsimonious conclusion is that the parameter values governing the approximated biological process are valid estimates of the vital rates governing the natural biological process.

7.8.2. Recent tributary habitat restoration actions

Tributary restoration actions, such as in-stream channel complexity projects, implemented in the Upper Salmon River Chinook MPG are meant to improve habitat quality. Thus, to estimate the population level benefit of tributary habitat quality improving actions by population, we incrementally improved the habitat quality of stream reaches containing relevant habitat actions during the time period 2009–15. Relevant habitat actions were those that from their type were reasonably assumed to have a positive impact on stream habitat condition. Only positive habitat quality change was included, such that if in the same time period natural disturbance events such as fire overlapped the restoration area, that impact was not included at this time. For example, all channel form actions and wood placement actions were assumed to have been implemented, implemented in the time frame reported in the PNSHP database, and implemented in a manner that resulted in positive biological benefit in terms of changes in capacity and productivity for the rearing juveniles and spawning adults that may utilize that portion of the watershed. No attempt was made to parse the habitat benefit by habitat action type, or reported extent and effort, as the project reporting data is not consistent enough to allow for such an analysis. Therefore, the attribution of benefit can be seen as a potential over-estimate in that some projects may not have been done as reported, or may not have been implemented in a manner that had any biological benefit. Conversely, lumping all in-channel actions into a single action type may be an underestimate of the action's benefit, since some action types have been demonstrated to be highly effective at increasing salmonid population processes, whereas others are thought to be beneficial, but have less scientific support (e.g., mimicking beaver-dominated stream and floodplain reaches versus large wood placement projects). Nonetheless, a simple, standardized method of attributing habitat benefit to restoration reaches was applied across all the of Upper Salmon River Chinook populations. Model estimate for the biological response from recent past actions ranged from no increase in median abundance for the East Fork Salmon River and Panther Creek, to greater than 25% increase in median abundance in the North Fork Salmon River, Valley Creek and the Lemhi River (Table 7.12). The range in population response is directly proportional to the magnitude of restoration effort. The capacity for stream restoration is not limitless, so regional priorities are reflected in the distribution of restoration actions across the populations in the MPG. The life cycle model approach can be useful in assessing regional action strategies, showing the relative benefit for implementing a fixed suite of actions with different spatial coverage across multiple populations.

7.8.3. Random habitat quality improvements

Future tributary action scenarios addressing habitat improvement across the entire Upper Salmon River Chinook MPG have not been developed yet. Some watersheds have restoration strategies in place (e.g., Lemhi River Conservation Plan), while others are currently being assessed for their potential (e.g., Panther Creek), and others have been the focus of aggressive actions in the past (e.g., Pahsimeroi River); so it is not unreasonable that a regional strategy has not yet emerged. However, developing life cycle model scenarios to explore the potential of future habitat restoration strategies requires spatially and temporally explicit plans for the project exercise to be most useful (McHugh et al. 2017).

Table 7.12. Upper Salmon River Chinook life cycle model output under six potential tributary habitat quantity/quality scenarios.

Population	Model output metric	Pre-2009 baseline	Pre-2018				Post-2018 potential actions	Pre-2018				Post-2018 potential actions RC
			Habitat actions (HA)	HA +1%	HA +5%	HA +10%		HA relative change (RC)	HA +1% RC	HA +5% RC	HA +10% RC	
East Fork Salmon River	Median natural-origin spawner	110	96	105	119	137	—	0%	0%	8%	25%	—
	Median pQET (@ n = 50, Yr 24)	0.77	0.82	0.79	0.74	0.68	—	0%	0%	-4%	-12%	—
Lemhi River	Median natural-origin spawner	223	289	285	306	325	443	30%	28%	38%	46%	99%
	Median pQET (@ n = 50, Yr 24)	0.26	0.18	0.21	0.18	0.14	0.05	-30%	-19%	-32%	-46%	-81%
North Fork Salmon River	Median natural-origin spawner	6	9	8	11	10	—	42%	33%	83%	67%	—
	Median pQET (@ n = 50, Yr 24)	1.00	1.00	1.00	1.00	1.00	—	0%	0%	0%	0%	—
Pahsimeroi River	Median natural-origin spawner	244	289	385	511	723	828	18%	58%	109%	196%	239%
	Median pQET (@ n = 50, Yr 24)	0.31	0.11	0.06	0.00	0.00	0.00	-64%	-80%	-100%	-100%	-100%
Panther Creek	Median natural-origin spawner	29	29	30	34	45	—	0%	3%	17%	55%	—
	Median pQET (@ n = 50, Yr 24)	1.00	1.00	1.00	1.00	1.00	—	0%	0%	0%	0%	—
Upper Mainstem Salmon River	Median natural-origin spawner	520	527	555	579	587	737	1%	7%	11%	13%	42%
	Median pQET (@ n = 50, Yr 24)	0.01	0.01	0.01	0.00	0.01	0.00	30%	84%	-42%	7%	-100%
Valley Creek	Median natural-origin spawner	51	64	62	68	73	—	25%	22%	33%	42%	—
	Median pQET (@ n = 50, Yr 24)	0.96	0.95	0.94	0.92	0.92	—	0%	-2%	-4%	-4%	—
Yankee Fork Salmon River	Median natural-origin spawner	64	67	65	82	87	151	4%	1%	28%	36%	136%
	Median pQET (@ n = 50, Yr 24)	0.96	0.97	0.93	0.90	0.91	0.61	0%	-4%	-7%	-6%	-37%

Lacking specific plans does not preclude the development of future scenarios for Upper Salmon River Chinook MPG populations, but it does mean that there is a strong disconnect between what may be likely to occur on the landscape and the designation of when and where habitat improvements are applied in a simulation. To overcome this mismatch, we took a randomization approach to the choice of reaches to be improve. Fixing the total possible area (reach length × reach width) as a fraction of the total spawning and rearing
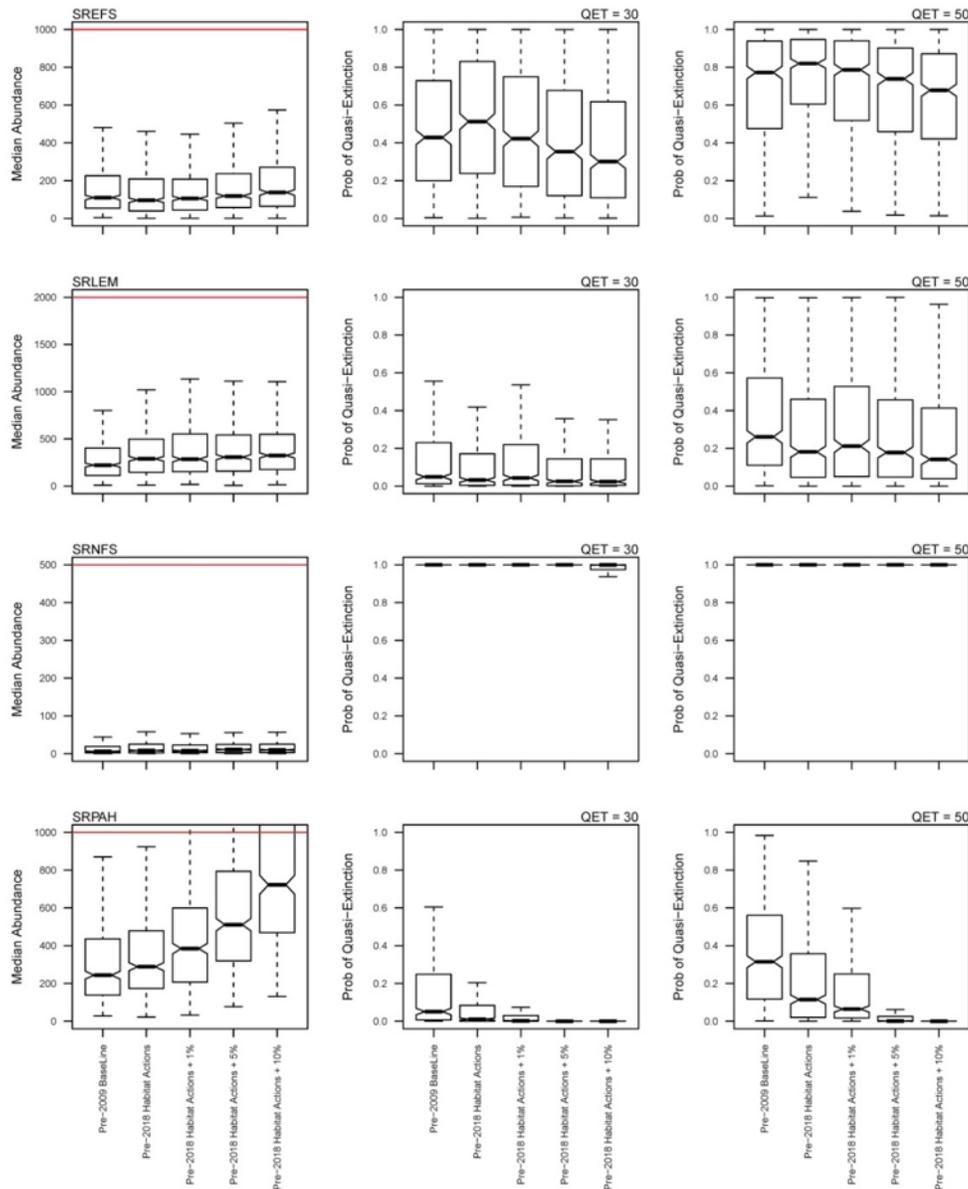
Figure 7.20. Upper Salmon River life cycle model output under five potential tributary habitat scenarios: 1) Baseline habitat condition pre-2009, 2) Baseline plus habitat restoration actions implemented 2009–15, 3) Baseline plus habitat restoration actions implemented 2009–15 plus an improvement of a randomly selected 1% of habitat area, 4) Baseline plus habitat restoration actions implemented 2009–15 plus an improvement of a randomly selected 5% of habitat area, and 5) Baseline plus habitat restoration actions implemented 2009–15 plus an improvement of a randomly selected 10% of habitat area. (top) East Fork Salmon River. (2nd row) Lemhi River. (3rd row) North Fork Salmon River. (bottom) Pahsimeroi River.

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habitat in each population watershed, we randomly applied habitat quality improvements to each population at three levels (1%, 5%, and 10%), replicating many times (<1,000) to capture the resulting variation in population response. In this case, the population response differed between replicate simulations both due to the uncertainty of around all biological parameters in the model, but also due to the variation in habitat quality improvement resulting from a spatially random application of improvement actions (Figures 7.20 and 7.21).
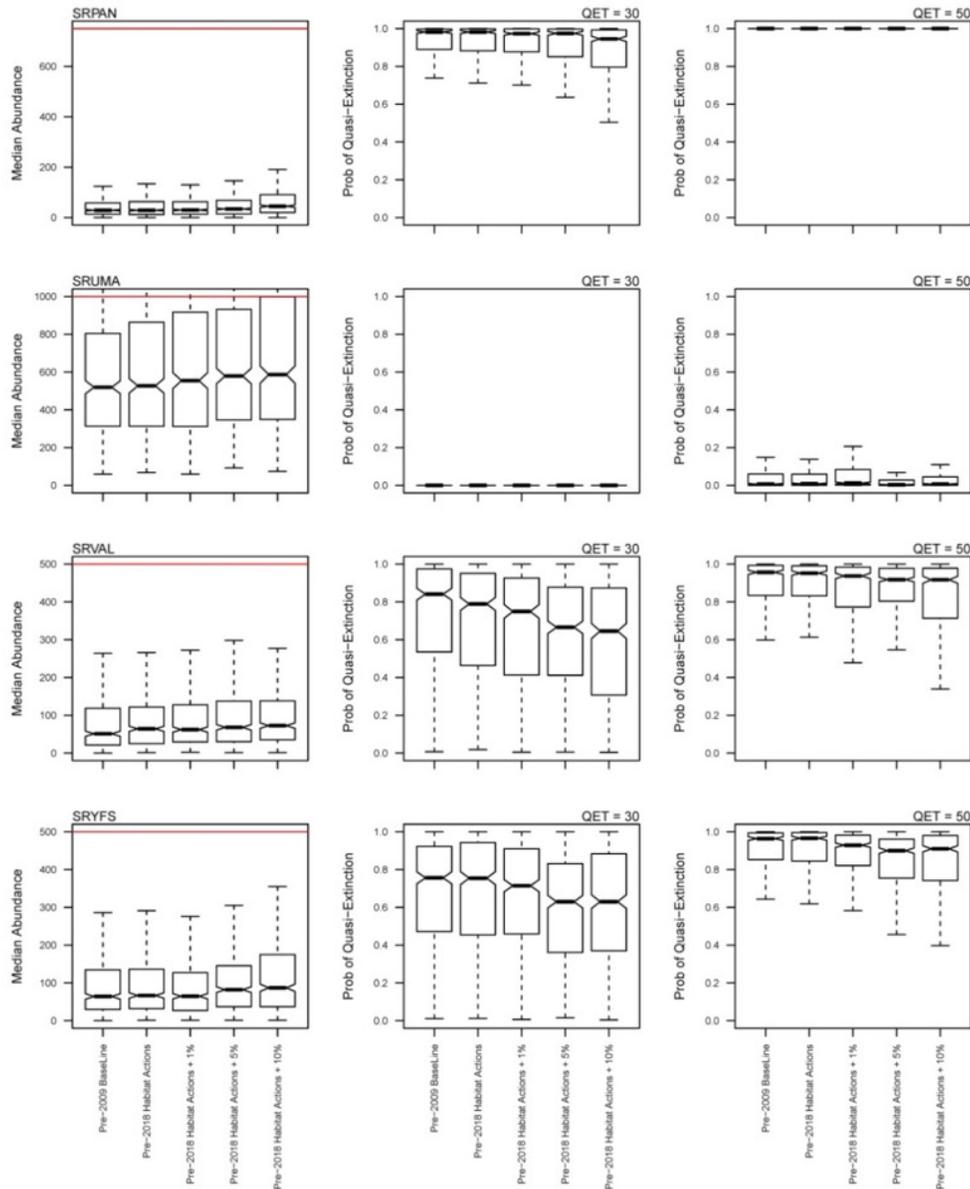
Figure 7.21. Upper Salmon River life cycle model output under five potential tributary habitat scenarios: 1) Baseline habitat condition pre-2009, 2) Baseline plus habitat restoration actions implemented 2009–15, 3) Baseline plus habitat restoration actions implemented 2009–15 plus an improvement of a randomly selected 1% of habitat area, 4) Baseline plus habitat restoration actions implemented 2009–15 plus an improvement of a randomly selected 5% of habitat area, and 5) Baseline plus habitat restoration actions implemented 2009–15 plus an improvement of a randomly selected 10% of habitat area. (top) Panther Creek. (2nd row) Upper Mainstem Salmon River. (3rd row) Valley Creek. (bottom) Yankee Fork Salmon River.

7.8.4. MPG-scale improvement of habitat quality and quantity for selected portions of each population

As part of the Proposed Action for the 2019 Biological Opinion on the operation and maintenance of the Federal Columbia River Hydropower System, the Action Agencies (the U.S. Army Corps of Engineers, the U.S. Bureau of Reclamation, and the Bonneville Power Administration) have suggested that the tributary habitat restoration action effort will be similar to that in the recent past years, but that effort would be targeted on specific types of restoration/conservation action and limited by MPG/ESU. In the Upper Salmon River Chinook MPG, the proposed actions focused on in-stream complexity and habitat access actions in a sub-set of the possible populations (Lemhi River, Pahsimeroi River, Upper Mainstem Salmon River and Yankee Fork Salmon River). The forecast level of habitat restoration action at the scale of the entire USAL Chinook MPG was 10 miles of stream complexity improvement and 16 miles of habitat access. These levels of effort were distributed across the focal populations evenly, splitting the habitat access effort four ways but the habitat complexity improvement only three ways, as the current habitat status in the Pahsimeroi is of sufficiently high quality that additional in-stream work is not warranted (Table 7.11). The estimated benefit of these restoration actions was large, with a 40% (Upper Mainstem) to 240% (Pahsimeroi) increase in median spawner abundance over pre-2009 baseline population levels (Table 7.12 and Figure 7.22). The resulting decrease in quasi-extinction risk (defined as probability that the population fell below 50 spawners for 4 successive years in the next 24 years) was also large (37% in the Yankee Fork Salmon River to 100% in the Pahsimeroi and Upper Mainstem Salmon River populations). In these scenarios, both habitat quality (in-stream complexity) and habitat quantity (habitat access) contributed to an overall increase in both spawning and rearing capacity (Table 7.11).

7.9. Conclusion

Overall, tributary habitat quality and quantity improvements resulted in improvements in population abundance and extinction risk metrics for all Upper Salmon River Chinook populations. Not surprisingly, the magnitude of the response scales directly with the magnitude of the change in habitat quality or quantity, with the smaller perturbations having no predicted effect on the population status. Population level benefits ranged from 0% to over 200% (Pahsimeroi population, potential post-2018 action scenario), with the largest impacts predicted for the future, directed, or intentional actions. Again, not surprisingly, random habitat improvement actions did result in estimated beneficial responses from the population models, but the variability in the outcome was large and the magnitude of the change was less than in comparably sized directed actions. From these preliminary explorations, it is clear that life cycle models are useful management-decision support tools, especially when constructed in a spatially explicit fashion that allows the development and comparison of specific habitat management scenarios.
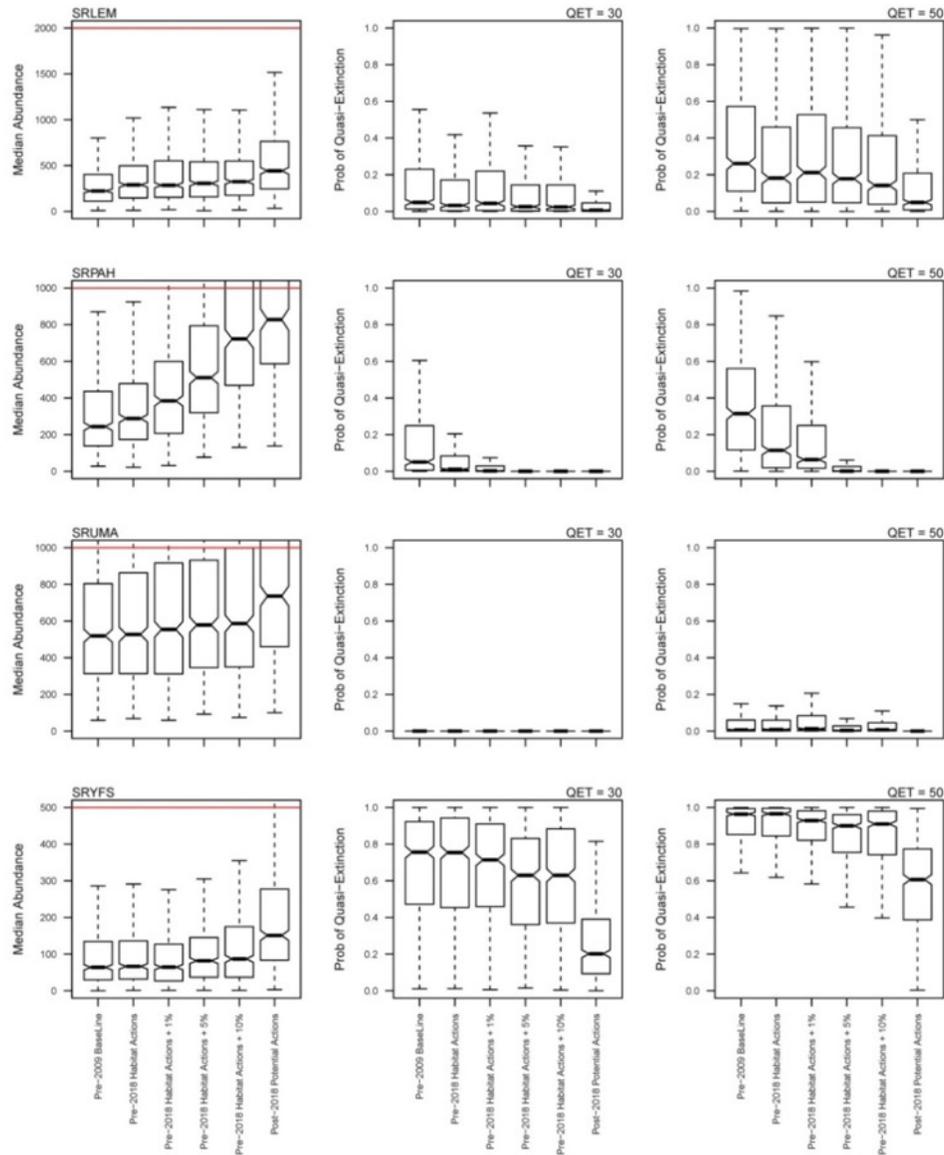
Figure 7.22. Upper Salmon River life cycle model output under six potential tributary habitat scenarios: 1) Baseline habitat condition pre-2009, 2) Baseline plus habitat restoration actions implemented 2009–15, 3) Baseline plus habitat restoration actions implemented 2009–15 plus an improvement of a randomly selected 1% of habitat area, 4) Baseline plus habitat restoration actions implemented 2009–15 plus an improvement of a randomly selected 5% of habitat area, 5) Baseline plus habitat restoration actions implemented 2009–15 plus an improvement of a randomly selected 10% of habitat area, and 6) a directed improvement of both quality and quantity, as specified in Table 7.11. (top) Lemhi River. (2nd row) Pahsimeroi River. (3rd row) Upper Mainstem Salmon River. (bottom) Yankee Fork Salmon River.

8. Summary

8.1. Why develop and use LCMs?

The initial questions we pose in the introduction assume that there is a basic link between salmon habitat and the associated life history requirements of target species and their respective populations. We attempt show the link between habitat and salmon in two ways throughout the document. First, through an understanding of fish–habitat relationships in relation to impacts and restoration of those impacts; and, secondly, through the use of life cycle models. LCMs typically predict the abundance at one life stage as a function of the previous life stage through a recruitment relationship that assumes density-dependence and employs transition functions that typically include productivity and capacity terms (e.g., Beverton–Holt, Ricker). The importance of this is that habitat/population capacity at one life stage may result in little or no gain in adult abundance if one or more successive life stages are at capacity. An accurate representation of capacity terms thus becomes vital as stream restoration projects typically seek to maximize abundance, growth, or survival at one or more stages. Therefore, LCMs can predict the potential for a restoration action to improve adult abundance, or determine the location and life stages limiting the population prior to restoration to guide restoration decision-making.

8.2. What methods can I use to develop LCMs?

Empirical models such as stock–recruitment (S–R), quantile regressions forest models (QRF), structural equation models (SEM), and habitat expansion models (HEM) can aid in quantifying the dynamics between successive life stages. S–R models typically require data-rich stage specific abundance data and can estimate contemporary capacity and population parameters. In addition, if data on ecological conditions are monitored throughout the time period of the life stage specific fish data, then they can be used to develop restoration scenarios. S–R models are of more limited use in data-poor scenarios if observation error in abundance estimates are large and the range of environmental conditions over the time period of the life stage-specific monitoring is not well quantified.

QRF models can also be very useful in data-rich environments. QRF models describe the entire distribution of predicted fish densities for a given set of habitat conditions, not just the mean expected density. In addition, they capture non-linear relationships between the independent and dependent variables, naturally incorporate interactions between covariates, and work with untransformed data while being robust to outliers, something common to biological data. Again, their primary weakness comes from the need of data-rich fish–habitat information such as the CHaMP/ISEMP programs in order to develop these relationships.

SEM models estimate the influence of predictor variables (e.g., habitat condition) on the average value of a response variable (e.g., fish abundance), and are a very useful approach for non-normal or nonlinear data, categorical responses, and hierarchical data structure. They also provide a flexible structure that allows for more data types and structures than habitat expansion or QRF methods. Again, they perform best in a data-rich fish–habitat environment, and many watersheds currently lack the habitat and fish data needed to utilize this approach.

Lacking habitat and fish data means that extrapolation is necessary in order to gain estimates of fish utilization at the appropriate scale for the development of watershed-wide restoration scenarios. Habitat expansion models (HEMs) directly extrapolate habitat capacity at any spatial scale by multiplying the amount of available habitat by the maximum density at which fish occur in each habitat, and summing all habitats of interest. HEMs are typically built with coarse-scale geomorphic controls, are best suited to estimate the effects of large-scale restoration scenarios, and require that habitat estimates be spatially extensive, but do not require further extrapolation. HEMs' primary weakness is the coarseness of their predictions, coupled with the need for further empirical analysis such as estimation of life stage-specific abundance estimates to verify the capacity estimates for specific life stages, such as watershed-scale smolt production. However, the benefit of HEMs are that they can be applied at very large spatial scales and require only widely available geomorphic and land-use inputs.

Empirical models, such as the preceding model types, typically lack the experimental manipulations needed to identify and validate causal mechanisms specifically related to restoration work. Thus, our understanding as to why an assemblage of variables interact to describe fish habitat requirements proves difficult with such models. Mechanistic models such as drift-foraging bioenergetics models are typically experimental or comparative studies confirming patterns described by mathematical models. These mathematical models are developed from ecological theory and thus can be more robust for predictions, as well as allow for the evaluation of alternative management or restoration scenarios. Such models are useful at a finer extent due to their complexity, but are data-intensive and can be difficult to calibrate and validate.

8.3.   Is scale, fitting, and calibration of LCMs an obstacle or an opportunity?

While there are two main scales of fish–habitat relationships (estimated stream reach or watershed), the most productive avenue to take is typically a combination of the two. Hybrid models typically use stream–reach-scale relationships to create indices and then establish relationships between these indices and watershed-scale fish data using a fitting or calibration process. The fitting process is typically a two-step process that includes initial parameter estimates based upon fish–habitat relationships at the stream–reach scale. The second step adjusts these values to agree with watershed-scale data, which typically focuses upon life stage-specific fish data. Calibration focuses on the LCMs and making sure their parameters are independently developed based on the literature and reach-scale data. Watershed-scale fish data can then be used to adjust these parameters to produce fish population predictions.

Important information can be gained from a hybrid approach. For example, in some cases a QRF or HEM estimate of habitat capacity may be very similar to a fitted estimate of population capacity from spawner and juvenile abundance. This agreement may indicate that freshwater habitat is limiting population growth. However, if current habitat capacity vastly exceeds population capacity in the same exercise, other processes or later life stages (e.g., smolt, spawner) may be limiting population growth. In these cases, the difference between the two estimates can provide guidance on the life stage and habitat of further research to identify limiting factors for the population of interest.

The use of LCMs to develop restoration scenarios examining how a suite of actions can change salmon habitat capacity and potential population size can only occur after a habitat change analysis is completed. The basis for the habitat change analysis is a comparison between historical habitat conditions, current habitat conditions, and potential habitat conditions developed from the geomorphic settings in a watershed (see for example the state-transition modeling of habitat condition done by Wondzell et al. [2007] and intrinsic potential models developed by Burnett et al. [2007]). The habitat conditions for each reference period or restoration scenario are then translated into a fish benefit (e.g., capacity, survival) for incorporation into the LCM process. Thus, because these estimates can be made at a scale that is relevant to populations (e.g., basin, subbasin), LCMs can be used to evaluate the efficacy of large restoration scenarios that have no practical experimental analog (e.g., widespread floodplain reconnection, widespread riparian plantings).

An additional benefit of the LCM approach is a sensitivity analysis, which is typically performed as part of the model development. Sensitivity analysis allows a user to evaluate the influence of each parameter in the model independent of the other parameters. This has several benefits to both model developers and practitioners. First, a developer can quickly identify which parameters are particularly influential on the resulting population outcome. Some parameters are not well known and are drawn from a wide distribution. If some parameters are both influential and their true value is not well known it can identify areas in need of further research. For developers, a sensitivity analysis can provide some information about what is limiting populations. For example, if the model is insensitive to the spawner capacity parameter it may indicate spawning habitat is not limiting, and that later stages in the model are the limiting factor.

8.4.   What comes first, the LCM or the understanding of habitat change?

Habitat change analyses typically include but are not limited to several categories including habitat quantity (i.e., stream channel area, pool frequency, floodplain condition), habitat quality (i.e., pool frequency, floodplain condition, wood loading, fine sediment levels, riparian condition), environmental conditions (i.e., stream temperature, streamflow), indicators of habitat quality (i.e., adjacent land use), and causes of habitat degradation (i.e., water diversions and barriers). Each of these variables can have an impact on salmon habitat capacity and their survival, and can affect one or more life stages. Some understanding of the changes that will result from restoration are needed to determine how they will be manifest in the LCM context. LCMs for some populations will be constrained by the available fish and habitat data. Therefore, first forming a general construct of the LCM with the locally available data will allow a rapid determination of which restoration types can be evaluated. For example, evaluating returning seasonal flow to a system as a restoration tool requires a hydraulic model of some sort (e.g., Physical Habitat Simulation [PHABSIM]) to determine how the habitat and stage-specific capacity will respond to additional flow. An LCM will not provide useful information about this restoration activity without system-specific flow–habitat information. However, in the absence of a hydraulic model, an LCM could inform whether the life stages flow addition is meant to help are, in fact, currently limiting population growth.

8.5.   What do I measure for habitat change?

Habitat quantity is one of the most important categories to quantify in developing restoration scenarios. A change in habitat quantity in its simplest form can be comparing the current vs. historical conditions of stream channel width (White et al. 2017). Pool frequency is another common metric used to gain a better understanding of both habitat quantity as well as habitat quality. Pool frequency can affect juvenile salmon rearing capacity, adult spawning preferences, and the overall quality of juvenile rearing habitat, which also allows for increases in densities and survivorship. Utilizing historical information, coupled with geomorphic classification and historical riparian condition, can allow for a comparison similar to stream channel width.

Floodplain habitat extent and condition is another habitat quantity metric important to salmonids. Floodplains are an important contributor to stream habitat complexity, and allow for the development and maintenance of multithreaded channels. This, even under altered conditions, give salmonids multiple options for each life stage and can result in greater growth and survival opportunities. In most cases, floodplains offer salmonids additional rearing capacity, especially during the wet months when juvenile fish seek rearing opportunities. These areas also facilitate increased growth and survival by offering abundant prey, optimal rearing temperatures, and refuge from predators. Comparing historical or potential to current condition, regardless of the method utilized, is an important component to changes in potential salmonid capacity and productivity and should be part of any analysis of freshwater habitat capacity.

Anthropogenic barriers to migration such as culverts, dams, levees, and dikes associated with floodplains and estuarine areas, and water diversions, are common causes of a reduction in the total area available to salmonids. Barriers, at a watershed scale, can reduce the amount of salmonid habitat anywhere from less than 1% to over 90% of historic capacity. Portions of tributaries and other habitats that are blocked from fish access can be mapped and estimates or inventories of habitat upstream of migration barriers can be quantified to estimate the amount of habitat which is disconnected. A key component of barrier removal analysis is the determination of a species-specific maximum upstream habitat in areas that are currently inaccessible. Accurate representation of upstream extent can have large implications for the capacity of a system, particularly for small tributary-occupying species (e.g., steelhead).

Water diversions and their impacts on salmonid habitat capacity and survival are important factors to consider with regards to habitat change analyses. Water diversions can reduce habitat capacity, survival, and have a deleterious effect on returning adult salmonids through increases in the magnitude and duration of summer low flows. In addition, water diversions, coupled with projected climate change scenarios can have negative additive effects with respect both juvenile salmonid capacity and survival. A survey of waterways, combined with a GPS receiver to identify locations of water diversion, combined with a physical description and photographs, is an effective method to identify water diversions. Staff at the NWFSC have been examining the potential impacts of water withdrawal in the interior Columbia River basins by identifying potential water rights relative to the estimated 1 Aug streamflow conditions. This is a potentially effective way to document areas where restorative actions in regards to water diversions would be important at a larger spatial extent.

Habitat quality is another important consideration in habitat change analyses. Riparian condition, stream temperature, in-stream channel complexity, and fine sediment intrusion are important aspects of watershed health to consider in evaluating habitat quality changes. Understanding the historic changes or potential to riparian condition is an important component to the development of restoration scenarios for streams. Riparian conditions effect a vast assortment of stream channel conditions and functions including but not limited to stream channel width, stream channel type, stream temperature, and wood loadings. In turn, these factors effect both habitat quantity and quality for salmonids at each life stage in the freshwater environment. Habitat change analyses of riparian vegetation can be determined either through historic reconstruction from historic datasets or through understanding the potential for growth based upon soil conditions, landscape attributes (i.e., elevation, ecological zonation, slope, aspect), or a combination of both historic and potential. Riparian vegetative cover has a primary and direct influence on one of the most important environmental factors associated with the salmonids—stream temperature.

Stream water temperature is widely recognized as one of the most important environmental factors naturally influencing the distribution, growth, and survival of salmonids and other aquatic organisms. Stream temperatures directly affect the physiological processes of salmonids, as well as their migration and spawn timing windows. Salmonids will seek out cold-water refuges as well as mobilize into relatively warmer water areas for metabolic and assimilative capacity purposes. Stream water temperature regimes can be modified by land-use activities including, but not limited to, water diversions, reduced or nonexistent riparian vegetation, excessive livestock grazing, and the simplification of streams due to channelization, levees, mining, and road construction. These activities can lead to increases in stream temperatures due to decreased streamflow, loss of subsurface streamflow and hyporheic exchange, and increased solar radiation reaching a stream. Understanding how stream temperature responds to the degradation and potential restoration of riparian vegetation and channel morphology is an important component to prioritizing freshwater salmon restoration actions. Where and how stream temperatures can be restored is thus directly linked to the preceding list of causes and effects that have occurred throughout a watershed. The questions then become, where and how much has stream temperature been altered and is there anything that could be implemented to restore stream temperatures? Once again, a comparison between historic and/or potential vs. current conditions becomes an analysis that can be used to determine these potential restoration locations as well as actions.

A second component linked to riparian zones and floodplains is in-stream channel complexity. In-stream channel complexity can be defined as obstructions associated with a stream channel such as individual pieces of wood, sediment substrate larger than the average diameter (i.e., boulders associated with a gravel streambed), and naturally accumulated (i.e., log jams) or naturally built (i.e., beaver dams) obstructions that alter the topography of the stream channel and adjacent landforms. In doing so, these structures create habitat complexity. Much has been published with regards to the natural functions of wood and other obstructions and the numerous ecosystem benefits associated with wood in particular (Gregory et al. 2003, Montgomery et al. 2003, Roni et al. 2014). As with the other aspects of stream ecosystems, understanding where and how in-stream

complexity restoration can occur correlates to the preceding list of cause and effects that have occurred throughout a watershed. The questions then become, where and how much has obstruction been altered and is there anything that could be implemented to restore stream channel complexity? Again, a comparison between historic and/or potential vs. current conditions becomes an analysis that can be used to determine these potential restoration locations as well as actions. The digital "reconstruction" of historic riverine landscapes, associated floodplains, and associated riparian and in-channel structures is a tool that helps us understand watershed restoration opportunities and constraints at multiple spatiotemporal scales. In addition, these historic reconstructions can and have been put into a geomorphic context, which allow for the identification of restoration opportunities that include land-use history, physical dynamics, and geologic settings to be considered. Another approach to gain a better understanding of restoration opportunities and the identification of restoration targets is the use of reference conditions to determine how much and potentially where restoration can occur with regards to in-stream channel complexity. Reference levels are useful as restoration targets for basin-scale wood loads to reestablish the central tendencies and functions associated with such obstructions.

8.6.   So I have an LCM and I know what habitat changed. Now what?

Once restoration scenarios are identified, each habitat change must be translated into a change in a life stage capacity or survival in the life cycle model. In general, habitat quantity or area changes tend to affect habitat capacity, whereas habitat quality change tends to affect life stage survival. The functional relationships between a habitat change and the change in capacity or survival are typically developed from literature values or from local empirical relationships.

The first step is translating habitat quantity into habitat capacity estimates for each life stage. Spawning capacity estimates occur two main ways. First, spawning gravel area can be translated into red capacity by dividing spawning gravel area by the average redd area of spawners for a species. Egg capacity is then the number of redds multiplied by fecundity for the species, and by number of females per redd if it is a number other than 1. Second, spawning capacity estimates can be a function of or changes in wood abundance in smaller streams. Redds/km have been quantified by channel type and landcover class, and egg capacity is number of redds multiplied by fecundity for the species, and by number of females per redd if it is a number other than 1. Rearing capacity for any life stage is typically estimated by summing all habitat areas of each habitat type, and then multiplying the total area of each habitat type by type-specific fish density.

The next step is to translate changes in habitat quality due to restorative actions into survival estimates. In general, habitat quality attributes tend to affect survival more than capacity, although to some extent it can be argued that they affect both. One challenge with estimating survival parameters is that the life cycle model requires a single value for a population, but habitat attributes vary by reach and cannot be summed to the basin scale. Some LCMs model major tributaries separately, which alleviates some of the challenges of averaging across habitats of varying quality. However, there are at least three ways to handle this problem. First, reach level survivals can be averaged across all reaches in a population, and weighted by habitat type if necessary. Second, survival can change as a function of a habitat quality change, such as a change in fine sediment in spawning gravels. Third, survival estimates can be scaled with a change in subbasin or population-scale capacity.

8.7. How do I take this information and make population-level outcomes from restoration?

Restoration alternatives can then be evaluated by developing a set of restoration scenarios, with each scenario representing either single or multiple restoration action types, and specified locations for each action type. For example, a restoration scenario may represent removal of passage barriers in specific locations, which reconnects spawning and rearing habitats above each barrier and increases spawning and rearing capacity for that scenario. For a more complex restoration scenario, the combined effects of multiple actions such as barrier removal, riparian planting, and floodplain habitat connection can be evaluated simultaneously. Each restoration action type influences specific life-stage capacities or survivals, and in most cases capacities and survivals of multiple life stages are increased. Locations may be reach-specific where there are data to do so, or they may be generalized to subbasins or subpopulations. The life cycle model then estimates the combined effects of the multiple life stage improvements on a salmon population.

8.8. Can you provide examples of how this is done and what it means?

We provided three examples of how LCMs, habitat change analyses, and the modeling of various restoration scenarios can help identify the magnitude of change in the habitat capacity, survival, and overall potential population size of several Chinook salmon populations in the Columbia River basin. Although in each case the implemented LCM was designed to accommodate the locally available data, the output was similar among regions to produce a comparable change in adult abundance and quasi-extinction risk with each restoration scenario.

In the Grande Ronde River basin there is an extensive record of juvenile and adult Chinook abundance, as well as a time series of stage-specific survival data. In addition, the Grande Ronde has spatially extensive inventories of tributary habitat from both the OAI and CHaMP habitat assessments. Moreover, several focused studies of Grande Ronde River habitat have sought to estimate the benefits of widespread stream habitat restoration actions with outputs that can be directly incorporated into the existing LCM framework. This rich dataset allowed for a detailed analysis that highlighted several outcomes of such potential actions. First, the Grande Ronde River model indicated the benefits on continued supplementation, particularly in the Upper Grande Ronde, although this benefit in the decline in QET comes at the cost of a reduction in natural spawners. We also assessed the potential for recent and future restoration to increase spawning and rearing capacity. For example, we estimate recent tributary habitat actions have resulted in a 21% increase in parr capacity, with additional benefits from those actions to increase over the next 50 years as a result of shading from riparian plantings. The benefit of the LCM framework, however, is to translate this stage-specific (i.e., parr) capacity into a change in adult abundance. In Grande Ronde River tributary Catherine Creek, recent habitat actions increase natural spawner abundance by 16%. LCMs also allow us to determine what effect other influences on the population might have. For example, our scenarios for the Grande Ronde River include both recent (i.e., high) and baseline (i.e., low) predation by pinnipeds on returning adults, which is strongly correlated with adult return timing that varies among populations in the Grande

Ronde River. Finally, we were able to include the benefits of several large-scale changes to capacity and survival, including the riparian plantings in key areas throughout the basin. For example, there are significant benefits in juvenile rearing and spawning capacities from the decrease in water temperature that would result from extensive riparian plantings. These benefits are manifested over the next 80 years, as the benefits of shading increase with increased tree growth. Although our results show that even with extensive restoration the QET in the Upper Grande Ronde would only experience a moderate improvement, the same restoration actions would have substantial benefits for Catherine Creek and Lostine/Wallowa populations. The richness of the dataset in the Grande Ronde River in both habitat and key demographic parameters has allowed for a flexible framework that can include many different types of actions simultaneously. The Grande Ronde River also benefits from a regional strategy of large-scale restoration that focuses on core "stronghold" areas to benefit habitats currently in use, followed by a downstream focus to increase the rearing, overwinter, and migration survival of those fish during outmigration.

Like the Grande Ronde LCM, the Wenatchee River spring Chinook population LCM benefits from a rich time series of several decades of juvenile and adult abundance and survival estimates. However, unlike the Grande Ronde, the Wenatchee River does not have the same level of habitat assessment that can be readily incorporated into the LCM. We were only able to include estimates of changes in rearing capacity from our relatively coarse Columbia River basinwide habitat model (Bond et al. 2019). While this model provides the resolution needed for the evaluation of a large-scale restoration strategy (e.g., basinwide floodplain reconnection), it is less adaptable to projects that are much smaller in scale (i.e., <200-m reach), or of a type that cannot be readily converted to capacity (e.g., changes in in-stream flow). As a result of this mismatch in project scale and projects occurring outside of the spring Chinook domain, we were only able to include two restoration projects in the Wenatchee River LCM for this analysis. The capacity change of these projects was modest enough that it did not provide a detectable change in the resulting adult abundance or QET. While this is clearly an underestimate of the potentially beneficial effects of many projects that could not be included in our model, it highlights the lack of a comprehensive restoration strategy that could be evaluated in our LCM framework. Although we could not produce an assessment analogous to the detailed scenario comparison of the Grande Ronde, the Wenatchee LCM framework is flexible and could incorporate changes in tributary-specific survival and capacity. Therefore, future habitat assessment and modeling should work to ensure that it occurs at a resolution that matches the current and proposed restoration.

Although the Upper Salmon River LCM does not have the benefit of as rich a set of abundance and survival data as our other examples, several tributaries have long time series (>15 yr) of outmigrants and spawners, and were used to calibrate models that were used throughout the Upper Salmon River. In addition, spatially extensive habitat models allowed for the direct modeling of the habitat condition of each reach in the Upper Salmon River. The Upper Salmon LCM can therefore incorporate any habitat project that increases the quality or quantity of habitat in a reach. We included four different assessments of habitat actions: baseline, recent habitat restoration, potential habitat improvements in randomly distributed reaches, and potential habitat improvements in reaches chosen for maximum benefit. Overall, tributary habitat quality and quantity improvements resulted

in improvements in population abundance and extinction risk metrics for all Upper Salmon River Chinook populations. Not surprisingly, the magnitude of the response scales directly with the magnitude of the change in habitat quality or quantity, with the smaller perturbations having no predicted effect on the population status. The largest population-level benefits were predicted for the future, directed, or intentional actions. Again, not surprisingly, random habitat improvement actions did result in estimated beneficial responses from the population models, but the variability in the outcome was large and the magnitude of the change was less than in comparably sized directed actions. Overall, the direct benefit of spawner abundance from improvements in habitat quality and quantity indicates that in the upper Salmon River, continued improvements in tributary habitats can benefit those populations and they are not limited by density-dependent processes that would dampen the effect of those actions.

Our goal was to develop a management decision support platform that could be used to explore the potential population-scale outcome of reach-scale habitat management actions. The LCM framework acts to aggregate the impacts of habitat actions over time and space, but also is the formal structure though which stage-specific fish–habitat relationships are aggregated into a full life cycle impact by projecting population behavior through time. In essence, the LCM approach takes the examples of fish response to habitat outlined in the sections on developing restoration scenarios, and translates them to population-level effects of adult abundance.

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