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Life Cycle Models of Interior Columbia River Basin Spring/Summer-Run Chinook Salmon Populations

May 2020

U.S. DEPARTMENT OF COMMERCE

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Life Cycle Models of Interior Columbia River Basin Spring/Summer-Run Chinook Salmon Populations

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Northwest Fisheries Science Center

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Abbreviations

AIC	Akaike Information Criterion
AMIP	adaptive management implementation plan
A/P	abundance/productivity
AQI	aquatic inventory
BH	Beverton–Holt
BiOp	Biological Opinion
BiOp-PA	Biological Opinion—Proposed Action
BNSF	Burlington Northern Santa Fe
BON	Bonneville Dam
BSR	biologically significant reach
CCFEG	Cascade Columbia Fisheries Enhancement Group
CCNRD	Chelan County Natural Resources Department
CHaMP	Columbia Habitat and Monitoring Program
COMPASS	comprehensive passage model
DPS	distinct population segment
EIS	environmental impact statement
ESA	Endangered Species Act
ESU	evolutionarily significant unit
FCRPS	Federal Columbia River Power System
GAA	globally available attributes
GAMM	generalized additive mixed models
GC	geomorphic condition
GCM	global climate model
GIS	geographic information system
GRTS	generalized random tessellation stratified
HCP	habitat conservation plan
HEC-RAS	Hydrologic Engineering Center's River Analysis System (USACE)
HEC-WAT	Hydrologic Engineering Center's Watershed Analysis Tool (USACE)
HGMP	hatchery genetic management plan
HOR	hatchery-origin returns
HOS	hatchery-origin spawner
HQI	habitat quality index
HSRG	Hatchery Scientific Review Group
HUC-6	Hydrologic Unit Code, Sixth Level
ICTRT	Interior Columbia Technical Recovery Team
IDFG	Idaho Department of Fish and Game
IP	intrinsic potential
IPCC	Intergovernmental Panel on Climate Change
ISAB	Independent Scientific Advisory Board
KS	Kolmogorov–Smirnov
LCM	life cycle model
LGR	Lower Granite Dam
LM	latent mortality
LWD	large woody debris
MACA	multivariate adaptive constructed analogs
MAT	minimum abundance threshold

Abbreviations (continued)

MLCM	matrix life cycle model
MPG	major population group
MWMT	maximum weekly maximum temperature
NAA	No-Action Alternative
NEPA	National Environmental Policy Act
NOAA	National Oceanic and Atmospheric Administration
NOR	natural-origin returns
NorWeST	Northwest Stream Temperatures database
NMFS	National Marine Fisheries Service
NWFSC	Northwest Fisheries Science Center
ODFW	Oregon Department of Fish and Wildlife
PA	Proposed Action
PDP	partial dependence plot
pHOS	proportion of hatchery-origin spawners
PIT-tag	passive integrated transponder tag
pNI	proportionate natural influence
PNSHP	Pacific Northwest Salmon Habitat Project
pQET	probability of falling below quasi-extinction threshold
PUD	public utilities district
QET	quasi-extinction threshold
QRF	quantile regression forest
RCP	representative concentration pathway
RESSIM	Reservoir System Simulation
RMJOC	River Management Joint Operating Committee
RT	radio tag
SAR	smolt-to-adult return rate
SOR	system operation review
SPCH	spring Chinook salmon
SS/D	spatial structure/diversity
SST	sea surface temperature
SWE	snow water equivalent
TDG	total dissolved gas
UCSRB	Upper Columbia Salmon Recovery Board
USACE	United States Army Corps of Engineers
USAL	Upper Salmon River
USFS	United States Forest Service
USGS	United States Geologic Survey
USOFR	U.S. Office of the Federal Register
VSP	viable salmonid population
WCR	West Coast Region (NOAA)
WDFW	Washington Department of Fish and Wildlife

Executive Summary

We developed stochastic stage-based life cycle models for populations of spring/summer-run Chinook salmon in the Snake River basin and for spring-run Chinook salmon in the Wenatchee River. The populations, nested within their evolutionarily significant unit (ESU) and major population group (MPG) designations, were as follows:

1. Snake River spring/summer-run Chinook Salmon ESU
    a. Upper Salmon River MPG
        - East Fork Salmon River
        - Lemhi River
        - North Fork Salmon River
        - Pahsimeroi River
        - Panther Creek
        - Salmon River upper mainstem
        - Valley Creek
        - Yankee Fork
    b. Middle Fork Salmon River MPG
        - Bear Valley Creek
        - Big Creek
        - Camas Creek
        - Loon Creek
        - Marsh Creek
        - Sulphur Creek
    c. South Fork Salmon River MPG
        - Secesh River
    d. Grande Ronde/Imnaha MPG
        - Catherine Creek
        - Upper Grande Ronde River
        - Lostine River
        - Minam River
        - Wenaha River

2. Upper Columbia River spring Chinook salmon (ESU)
    a. Upper Columbia/East Slope Cascades MPG
        - Wenatchee River

The population models were developed with retrospective data that described demographic rates for transitions between life stages throughout the life cycle. We also included environmental forcing functions for several of these transitions. We developed life stage-specific functions independently, and then linked them together into the life cycle models (LCMs).

After we developed the LCMs, we fit the models, using a pseudo-Bayesian routine, to data representing the adult and juvenile life stages. We fit both the means and variances to ensure that the models realistically represented the population dynamics.

Once the models were fit to data, we then ran them prospectively to examine a suite of alternatives, including the Proposed Action for the 2020 Federal Columbia River Power System (FCRPS) Biological Opinion (NMFS 2020). The alternatives included proposed changes to operations of the hydropower system and habitat actions in some of the populations. We also examined the effects of increased pinniped predation, primarily by California sea lions, in the Columbia River estuary.

To prospectively model the Proposed Action (PA) for hydrosystem operations, we used the COMPASS model, described in Chapter 2. A hypothesized benefit of the PA in the hydrosystem is a reduction in latent mortality. Latent mortality is any mortality due to passage through the hydrosystem that is expressed after individuals have passed through the hydrosystem. Because this reduction in mortality is not directly measurable, we examined a range of survival improvements due to a decrease in latent mortality.

To develop other stage-specific components of the LCMs, we relied on several peer-reviewed publications:

- The survival of adults through the hydrosystem is described in Crozier et al. (submitted b). This paper also projected the effects of climate on adult upstream survival.
- Pinniped predation, primarily by California sea lions, on adult salmon occurs in the estuary. We modeled population-specific mortality due to pinnipeds based on seasonally varying estimates of survival and on population-specific arrival timing. Pinniped predation has increased dramatically over the past few years, particularly for early-arriving populations. We describe our methods in Rub et al. (2019) and Sorel et al. (in review).
- Ocean survival is modeled based on PIT-tag data of juveniles detected at Bonneville Dam and returning as adults to Bonneville Dam. Survival is related to juvenile arrival date at Bonneville Dam and indicators of ocean conditions. The model is implemented for both the standard runs (PA) and for climate change runs. Details are provided in Chasco et al. (submitted).
- We also examined several scenarios of climate change. Details may be found in Crozier et al. (submitted a). We applied projected changes to environmental conditions at several life stages. We found that these populations were particularly sensitive to projected changes in ocean conditions.

We represented model output in terms of geometric mean abundance and probability of falling below a quasi-extinction threshold (pQET).

For a measure of abundance, we used the distribution of geometric mean abundance from many (~1,000) replicate simulations for each scenario and population. We used the geometric mean of Years 15–24 for each simulation and compared these estimates of abundance (median and four quantiles of the distribution) across scenarios within populations.

To calculate the probability of falling below QET, we used a multistep process. First, we assessed whether a single simulation fell below the quasi-extinction threshold within T years, where $T = 24$. QET was defined as the 4-year mean of spawners (on the spawning ground) falling below the specified QET. We examined two QET thresholds: 30 spawners and 50 spawners. As with abundance, we compared the distribution of the estimated quasi-extinction events, expressed as a probability of falling below QET, across scenarios within populations.

We found that, in general, the Proposed Action in the hydrosystem led to a slight decrease in direct survival. This was due to increased spill late in the migration season that prevented collection of juveniles for transportation. This decrease in direct mortality could be compensated for by the reduction in latent mortality. We believe this is an important uncertainty that merits further research.

Researchers with the U.S. Geological Survey, Western Fisheries Research Center, developed an analogous life cycle modeling process to describe the PA for the 2020 FCRPS Biological Opinion (NMFS 2020) for the natural-origin Snake River fall-run Chinook Salmon ESU (Tiffan and Perry 2020). Reporting similar performance metrics, Tiffan and Perry (2020) used a state-space modeling framework to project the impacts of the PA on fall Chinook salmon. No habitat actions are proposed for this ESU and future climate conditions were not modeled, but, similarly, sensitivity to summer spill and no appreciable change in extinction risk (QET 30 or 50 at Year 24) were forecast under the hydrosystem operations of the PA.

In populations that received major habitat restoration, we modeled substantial improvements in survival that increased projected abundance and decreased risk of extinction.

Climate change can potentially be a major negative influence on the population dynamics of these populations. Through a series of sensitivity analyses using an ensemble of climate change models, we project that climate change in the ocean will have a strongly detrimental effect on these populations. This will largely be due to increasing sea surface temperatures.

We believe that we can offset some of the impacts of climate change through management actions. As described in Chapter 7, we should establish which populations would both benefit the most from habitat restoration and have the likeliest positive effect on the ESUs as a whole. We plan to refine the process of identifying focal populations for restoration based on the sensitivity to management actions and the population's role in supporting its ESU.

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1 Introduction

Richard W. Zabel, Chris E. Jordan, and Thomas D. Cooney

1.1 Overview

Life cycle modeling has become an invaluable tool for managing at-risk populations (Doak et al. 1994, Beissinger 2002), particularly for species that have distinct life stages. A major feature of life cycle models (LCMs) is that they can translate changes in demographic rates (survival, capacity, or fecundity) in specific life stages into measures of population viability metrics (e.g., long-term abundance, productivity, or probability of extinction), which are more relevant for population management. In addition, LCMs allow for the examination of impacts across several life stages and in concert with other factors such as climate variability and change (Figure 1-1). In the Columbia River basin, researchers have used LCMs to address a broad range of questions in a variety of populations (Kareiva et al. 2000, Wilson 2003, Zabel et al. 2006, ICTRT and Zabel 2007, Crozier et al. 2008, Honea et al. 2009, Jorgensen et al. 2009). While early models were deterministic and density-independent (Kareiva et al. 2000), later efforts were more sophisticated, including stochasticity, density dependence, and climate variability and change (Zabel et al. 2006, ICTRT and Zabel 2007, Crozier et al. 2008).

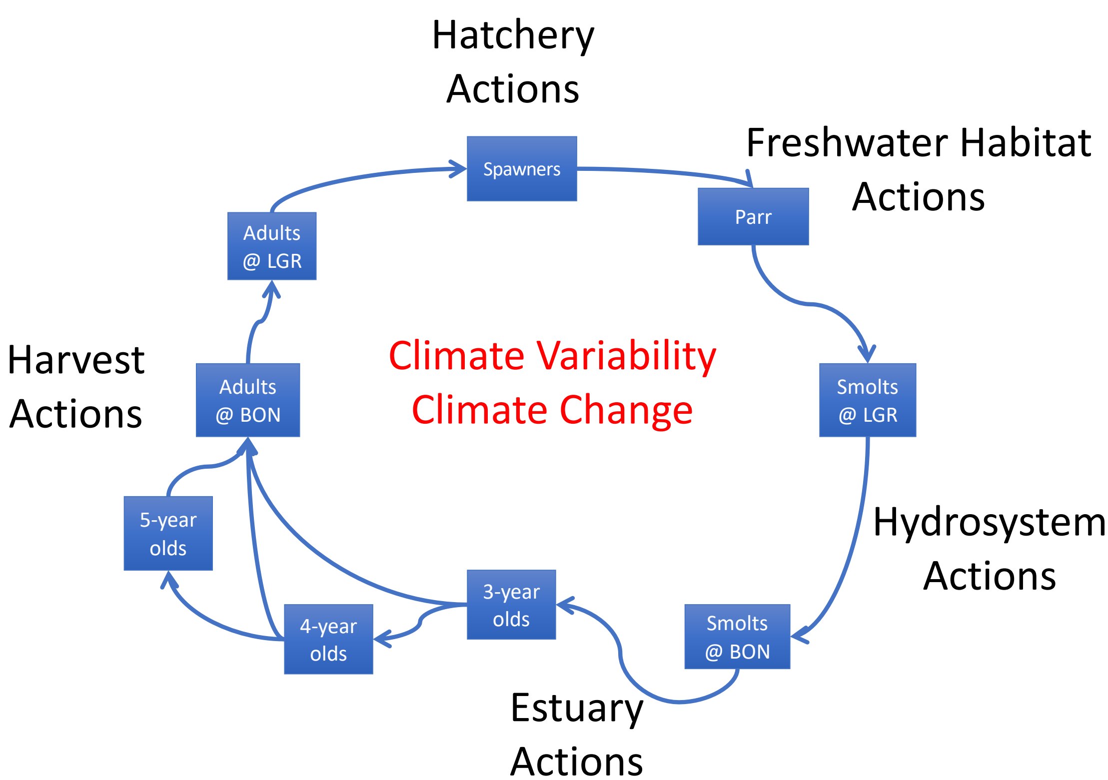

Figure 1-1. Typical interior Columbia River basin spring/summer-run Chinook salmon life cycle, with mitigation actions occurring at several life stages. In addition, climate variability and change can interact with actions to influence population performance.

This report presents life cycle models in support of the 2020 Federal Columbia River Power System (FCRPS) Biological Opinion (NMFS 2020). We focus on models of spring and summer Chinook salmon. We present models and results for several major population groups (MPGs) in the interior Snake River basin: Grande Ronde/Imnaha, Upper Salmon River, Middle Fork and South Fork Salmon River, and Upper Columbia/East Slope Cascades. In addition, we present supporting modules for all of the models: Survival through the hydrosytem, survival through the estuary and ocean life stages, mortality due to pinniped predation, and the effects of climate change.

Much of the focus for these models is on the benefits of habitat restoration. We have also produced a companion NOAA technical memorandum (Pess and Jordan 2019) that describes the methodology for converting habitat actions into benefits that can be incorporated into LCMs.

1.2 Population Models

1.2.1 Grande Ronde River basin

The Grande Ronde River basin in northeastern Oregon offers a good system for contrasts. Some of the tributaries have been heavily modified (Catherine Creek and Upper Grande Ronde River) and have been the focus of habitat restoration actions, and some of the tributaries are relatively pristine (Lostine/Wallowa, Minam, and Wenaha). In addition, some of the tributaries have supplementation, while others rely on natural production. These models are described in Chapter 3.

1.2.2 Upper Salmon River

We developed models for several populations in the Upper Salmon River MPG: East Fork Salmon River, Lemhi River, North Fork Salmon River, Pahsimeroi River, Panther Creek, Salmon River upper mainstem, Valley Creek, and Yankee Fork. These populations are impacted by water withdrawals and habitat degradation. Many of the habitat actions are focused on reconnecting habitat to make it accessible to salmon populations and increasing in-channel complexity for juvenile rearing habitat. We describe these models in Chapter 4.

1.2.3 Middle Fork/South Fork/Upper Salmon River

This suite of models covers three MPGs. In Middle Fork Salmon River, we developed models for Bear Valley Creek, Big Creek, Camas Creek, Loon Creek, Marsh Creek, and Sulphur Creek. For the South Fork Salmon River MPG, we developed a model for the Secesh River population. For the Upper Salmon River MPG, we developed a model for Valley Creek. Most of the populations lie within wilderness areas. In addition, these models are supported by a lengthy time series of PIT tag parr-to-smolt survival data, and we explored relationships between survival and tributary flow and temperature. Population models for these MPGs are presented in Chapter 5.

1.2.4 Wenatchee River

The Wenatchee River is a complex system that is supported by five production areas: Chiwawa River, Little Wenatchee River, Nason Creek, the mainstem Wenatchee River, and White River. We modeled the contributions of these areas separately, but combined the results from the production areas to produce population-level metrics. The area has also suffered from habitat degradation, and is the focus of many habitat actions. In addition, the population is heavily supplemented. We present the Wenatchee River model in Chapter 6.

1.2.5 Snake River

Researchers with the U.S. Geological Survey, Western Fisheries Research Center, developed an analogous life cycle modeling process to describe the Proposed Action (PA) for the 2020 FCRPS Biological Opinion (NMFS 2020) for the natural-origin Snake River fall-run Chinook Salmon ESU (Tiffan and Perry 2020). This work estimates the effects of covariates on key demographic parameters, and uses the fitted life cycle model to simulate population trajectories under the PA. Tiffan and Perry (2020) examined the effect of numerous environmental, hydrosystem, and ocean covariates on key demographic parameters. They used the fitted model to simulate population trajectories under the PA, showing that, overall, the probability of quasi-extinction was low, with only 1.6% of all simulations having a quasi-extinction probability >0.95.

1.3 Common Modules

Our strategy for developing the life cycle models was to produce two modules: one that was specific to the freshwater phase of specific populations, and another that was common at the evolutionarily significant unit (ESU) level. In these chapters, we describe modules that are common to ESUs.

1.3.1 Juvenile survival through the hydrosystem

Survival of juveniles through the hydrosystem is handled by the COMPASS model. A description of the model and the alternatives produced for NMFS (2020) is contained in Chapter 2.

1.3.2 Adult upstream survival

The survival of adults through the hydrosystem is described in Crozier et al. (submitted b).

1.3.3 Pinniped predation

Pinniped predation, primarily by California sea lions, on adult salmon occurs in the estuary. We modeled population-specific mortality due to pinnipeds based on seasonally varying estimates of survival and on population-specific arrival timing. Pinniped predation has increased dramatically over the past few years, particularly for early-arriving populations. We describe our methods in Rub et al. (2019) and Sorel et al. (in review).

1.3.4 Ocean survival

Ocean survival is modeled based on PIT-tag data of juveniles detected at Bonneville Dam and returning as adults to Bonneville Dam. Survival is related to juvenile arrival date at Bonneville Dam and indicators of ocean conditions. The model is implemented for both the standard runs (Proposed Action) and for climate change runs. Details are provided in Chasco et al. (submitted).

1.3.5 Calibration

The population models were fit to recent observations of the population and a recent no-action hydrosystem operation alternative with an approximate Bayes computation approach based on rejection-sampling. In rejection-sampling, approximations of parameters' posterior distributions can be constructed through repeated trials. The calibration procedure we used consisted of repeatedly drawing a set of parameter values from informative prior distributions (i.e., independent draws of parameters' values according to a random uniform distribution, from prespecified ranges for each parameter), running the model with the unique parameter sets, and comparing model outputs to empirical observations. Each unique parameter set was accepted (rejected) if it fell inside (outside) an acceptance level for deviation between model-generated and observed data. We defined the deviation as the Kolmogorov–Smirnoff (KS) statistic, D, which measured the degree to which the two distributions came from the same underlying distribution. We used time series of observations of spawner abundance from redd counts and estimates of smolt outmigrant abundance from smolt trapping as the two life stages with which to calibrate the LCM. We compared these recent observations to the LCM outputs so that the model would be calibrated to current conditions.

1.4 Other Sections

1.4.1 Climate change

We examined several scenarios of climate change. Details may be found in Crozier et al. (submitted a).

1.4.2 Population prioritization

The objective of Chapter 7 is to develop a standardized, quantitative method for identifying focal populations for near-term emphasis in habitat restoration. The basis for evaluation of populations is meant to be consistent with avoiding immediate (e.g., 24-year) losses in ESU capabilities to withstand demographic and localized catastrophic risk factors and for making progress toward longer-term goals for Endangered Species Act (ESA) and broad-sense recovery. The focal population concept will integrate into ongoing ESA recovery implementation and related activities (e.g., ESA consultations involving tributary habitat) in the Columbia River basin. The focal population identification will provide strategic guidance for sequencing future habitat restoration and protection at the population or MPG level. The focal population analysis is intended to be a tool for use in strategic planning initiatives such as the Grande Ronde Atlas and the Upper Salmon River MPG regional restoration planning effort involving the Idaho Governor's Office of Species Conservation, the Idaho Department of Fish and Game (IDFG), federal agencies, and tribal fisheries staff. The framework described in this chapter was initially developed for application to the Snake River spring/summer-run Chinook Salmon ESU's major population groups and their component populations; however, we have also developed a version for application to steelhead distinct population segments (DPSes). The process of identifying focal populations from a distillation of quantitative data is new and unique for the spatial extent and number of populations ranked here. Our aim in presenting this framework in Chapter 7 is to outline an example of moving prioritization away from a qualitative expert panel process to one that uses many disparate data sources to provide the most holistic, quantitative assessment possible.

1.5 Scenarios Evaluated

For all of the populations modeled in this exercise, we developed a standard set of scenarios to compare a range of management actions that capture the proposed actions considered in NMFS (2020). The scenarios are referenced to a no-action alternative (NAA) that acts as the starting physical and biological setting of impacts on Columbia River basin salmonid populations. The NAA is then further developed to represent the range of management options in the proposed action (PA). To facilitate comparing across scenarios and population modeling frameworks, we also established a standard analytical approach and a corresponding suite of output graphics.

1.5.1 No-Action Alternative

The No-Action Alternative (NAA) assumes that operations will continue as they have in the recent past. It also assumes that no further habitat actions will be taken, and that hatchery operations, ocean and in-river fisheries, and marine mammal predation remain at current levels.

1.5.2 Proposed Action

The Proposed Action (PA) assumes that hydrosystem operations will be altered to benefit fish passage. The main feature of the PA is the "Flex Spill" operation, where spill is increased until 125% nitrogen supersaturation is reached except during a 4-hour block in the morning and a 4-hour block in the early evening. The PA also includes proposed habitat actions by MPG—the modeled scenarios include assumptions on how the effort is distributed in time and space.

The Proposed Action is hypothesized to reduce latent mortality (LM), which is defined as any mortality due to passage through the hydrosystem that is not expressed until after fish pass through the hydrosystem. To reflect that LM is a hypothesis, we multiplied ocean survival by a range of factors: 1.0, 1.17, and 1.35.

We also examined several climate change scenarios, described in detail in Crozier et al. (submitted a). Briefly, we modeled climate change in four life stages: parr-to-smolt survival, juvenile migration through the hydrosystem, survival in the ocean stage, and adult survival through the hydrosystem. We first developed a Stationary climate scenario, where climate in the future is similar to climate in the past. Then we used an ensemble of climate change models from the Intergovernmental Panel on Climate Change (IPCC) to reflect uncertainty on climate change projections. We used downscaled projections of environmental factors in the relevant life stages. We reported results corresponding to all the IPCC climate change models as percentiles (25th, 50th, and 75th) across all climate change runs to capture low, medium, and high hypotheses on the severity of future climate. Details are in Crozier et al. (submitted a).

1.6 Standard Model Outputs

We represented model output in terms of mean abundance and probability of falling below a quasi-extinction threshold (pQET).

For mean abundance, we first ran many (e.g., 1,000) replicate simulations for each scenario and population. We then took the geometric means of Years 15–24 for each simulation. For each scenario and population, we reported the distribution of these geometric means (5th, 25th, 50th,75th, and 95th percentiles) in tables and figures (boxplots). For each boxplot, the solid bar in the middle of the plot represents the median of the metric across all replicate simulations. The box represents the middle 50% of the replicates, with the lower extreme of the box representing the 25th percentile and the upper extreme of the box representing the 75th percentile. The horizontal bars terminating the vertical dashed lines bound the middle 99% of the replicates, with the lower bar representing the 1st percentile, and the upper bar representing the 99th percentile.

To calculate the probability of falling below QET, we used a multistep process. First, we assessed whether a single simulation fell below the quasi-extinction threshold within T years, where $T = 24$. QET was defined as the 4-year mean of spawners (on the spawning ground) falling below the specified QET. We examined two QET thresholds: 30 spawners and 50 spawners. If a population fell below this threshold, it scored as a quasi-extinction event. We estimated the probability of falling below QET for each population and scenario as the proportion of quasi-extinction events in a set of simulated futures.

As with the abundance metric, we also report the quasi-extinction risk as the distribution of the resulting estimates of pQET. To generate a distribution of pQET, we constructed 100 bootstrap samples (sampling with replacement) of 100 replicate simulations from the 1,000 simulations for each scenario. We estimated pQET for each of the 100 samples of 100 replicates and report the 5th, 25th, 50th, 75th, and 95th percentile of the pQET distribution.

1.7 Adaptive Management

1.7.1 Portfolio of life stage-specific actions

One of the advantages of life cycle modeling is the ability to assess impacts at multiple life stages by translating changes in life-stage demographic rates to changes in viability metrics. In this way, we can put together a portfolio of actions to compare across different portfolios. We are proposing an adaptive management strategy where we use life cycle models to design and assess alternative suites of actions. Prospective LCMs are used to develop alternative portfolios of actions. These portfolios can be compared with a variety of performance metrics, such as in a cost–benefit or extinction risk framework. The life cycle models also play a critical role in an adaptive management context, as they make testable, quantitative predictions. These predictions are treated as hypotheses, and an appropriately designed monitoring program can assess the predicted outcome and can be used to evaluate and improve the analytical framework when the outcomes differ from expected.

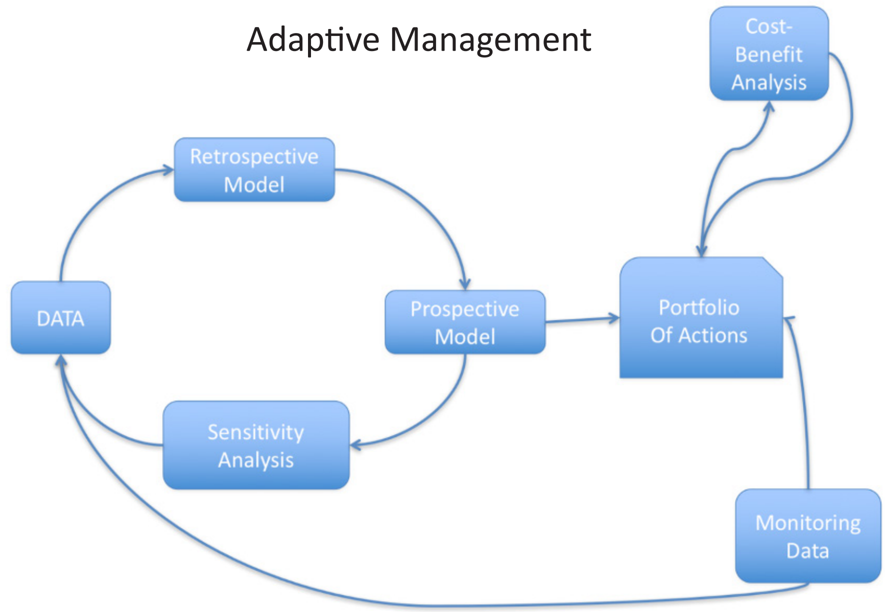

Figure 1-2. Adaptive Management scheme. Prospective life cycle models are used to develop alternative portfolios of actions, which can be compared in a cost–benefit analysis. Once a portfolio is chosen, we use monitoring data to assess whether actions are effective.

In the context of Adaptive Management (Figure 1-2), life cycle models form both the analytical framework for making quantitative, testable predictions of management action outcomes, as well as the basis for the data or monitoring needs. The data needs of an LCM-based decision support system are both to parameterize the population processes represented in the model (e.g., stage-specific abundance, survival, and capacity), and to test the population response to management actions (e.g., fish–habitat relationships, mainstem project survival, or hatchery–wild interactions). In either case, the life cycle model is the use-case for the monitoring data and, as such, should be used to set the spatial and temporal resolution of sampling, choice of monitoring metrics, and ultimately the data quality in terms of sampling and measurement uncertainty. Having an analytical tool as the consumer of monitoring data allows direct assessments of the consequence of variation in data quality, since the impact of data quality can be immediately translated into the quality of decision-making in terms of the risk of making an incorrect decision.

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2 The COMPASS Model for Assessing Juvenile Salmon Passage through the Hydropower Systems on the Snake and Columbia Rivers

James R. Faulkner, Daniel L. Widener, and Richard W. Zabel

2.1 Introduction

The Comprehensive Passage Model (COMPASS) was developed as a tool for investigating the passage experience of migrating juvenile salmon and steelhead under various environmental conditions and management scenarios (Zabel et al. 2008, COMPASS 2019). COMPASS was reviewed by ISAB in 2008 and has been used to inform a variety of management decisions concerning juvenile salmon since then.

COMPASS contains physical descriptions of the Snake and Columbia Rivers and their main tributaries, which include spatial representations with widths, depths, and elevations to allow volume and velocity calculations. The hydroelectric dams in the system are also represented, and algorithms are used to route flow through the set of passage routes unique to the configurations at each dam. This allows dam operations such as spill and surface collector operation to be accounted for on daily or finer timesteps.

Flow is input at the river headwaters or at the dams, either as measured observations or as predictions from hydrological models. Other possible environmental inputs include temperature, turbidity, and dissolved gas. COMPASS can also take spill proportions and reservoir elevations as inputs and can take surface weir volumes and operation schedules. Schedules and rates of smolt transportation on barges are also taken as inputs for operation of collector dams.

COMPASS contains a set of biological models we developed for: a) arrival timing at the head of the hydropower system, b) reservoir migration rate, c) reservoir survival, d) dam passage routing for various species, and e) dam survival. These submodels were all fitted to observed data and are functions of the set of variables describing environmental conditions and dam operations that are available to COMPASS, including flow, velocity, temperature, and spill. When combined together, these submodels allow predictions of the passage experience of population releases through the system to Bonneville Dam tailrace.

The model runs on a subdaily timestep, and uses environmental inputs on a daily level to update the predictions of the submodels for each timestep. Fish are added at the top segment of the system (head of Lower Granite reservoir for Snake River stocks) according to the arrival timing submodel (a). The model then advances sequentially via timesteps, moving the fish downstream using the migration rate submodel (b) and applying mortality in each timestep according to the reservoir survival (c) or dam passage and survival (d and e) submodels. The final results returned at the completion of a COMPASS model run include the total proportion of fish that survived to Bonneville Dam tailrace and the daily proportion of survivors reaching Bonneville Dam tailrace.

COMPASS also models the smolt transportation program. COMPASS takes the collection start date and separation probability as inputs for each of the three collector dams: Lower Granite, Little Goose, and Lower Monumental Dams. After the collection start date, all fish predicted by the dam passage submodel (d) to enter the juvenile bypass system are potentially subject to transportation. The proportion of fish specified by the separation probability will be returned to the river, but all remaining fish will be transported to the tailrace of Bonneville Dam. COMPASS assumes a uniform travel time of 2 days from the collection date and a survival of 0.98 during transportation for all transported fish. For a COMPASS model run with transportation enabled, the results returned by the model include the overall proportion of fish surviving to Bonneville Dam tailrace as well as separate estimates for in-river migrants and transported fish, separate vectors of the daily proportion of surviving in-river migrants and transported fish reaching Bonneville Dam tailrace, and the overall proportion of fish that were transported. These results are returned to the overarching life cycle model and serve as inputs to the smolt-to-adult return rate model and other models.

Here we describe the application of the model to a set of simulated data representing a management scenario, and present the results of the COMPASS runs. This scenario represents a set of rules for the operation of hydroelectric dams and storage reservoirs, including both continuations of current activities used from 2017–19 and modifications to certain management actions and new actions, described in more detail in the next section.

2.2 Methods

2.2.1 Model updates

Since the most recent documentation of COMPASS (COMPASS 2019), we have made some updates to the submodels and to the general functionality of COMPASS. The only change relevant to the COMPASS model runs for NMFS (2020) is a new ability to use timestep-specific spill inputs. This modification allows the COMPASS model to accurately match within-day "flex spill" operations that modify spill percentages for specific periods of each day.

2.2.2 Prospective modeling

A prospective management scenario was investigated. The scenario, labeled the Proposed Action (PA), comprises a prospective management regime that includes the following actions:

- Increased flexibility in the operations of storage reservoirs in the upper Columbia River basin.
- Increased flexibility of operations and increased water releases from Lake Roosevelt in the upper Columbia River basin.
- Increased spill at FCRPS dams, up to 125% tailrace TDG. Spill operations also include "Flex Spill" within-day modifications of spill, which comprise two four-hour blocks per day of reduced spill to Performance Standard levels.
- Increased operating range of reservoir elevation at John Day Dam and in the Snake River reservoirs.

- Modified timing of flow releases from Dworshak Reservoir.
- Earlier (20 April) start to transportation operations, relative to the 1 May start in previous alternatives.
- Installation of fish-friendly turbines at Ice Harbor, McNary, and John Day Dams; these turbines are assumed to reduce fish mortality by 50% relative to the current turbines.

Aside from these actions, the PA carries forward the general management regime from 2017–19.

We changed the timestep settings used in the COMPASS model to 12 timesteps per day (each 120 minutes long) for the Proposed Action model runs. This allowed us to implement the "Flex Spill" within-day changes in spill exactly as specified in the Proposed Action. These subdaily changes comprise reduced rates of spill during two four-hour blocks per day, one block from 06:00 to 10:00 and the second block from 16:00 to 20:00. The other environmental variables of the PA—flow, temperature, and reservoir elevation—are constant for any given day.

The U. S. Army Corps of Engineers (USACE) generated the Proposed Action using a suite of hydrologic and management models, including RESSIM, HEC-RAS, and HEC-WAT. These models accurately account for power generation and spill and associated hydrology in the hydropower system, and output daily predictions of flow, total dissolved gas, and reservoir elevation along with hourly predictions of spill associated with each dam. This was done for a set of 80 water years (representing headwater inputs for the years 1929–2008). USACE also used a model to predict water temperature during five representative years, and developed an algorithmic process to map the resulting temperatures onto the 80 water years. We used the environmental values predicted by the USACE models for the 80 water years as inputs to COMPASS.

We constructed models of the arrival distribution at Lower Granite Dam for various populations of wild Snake River Chinook salmon and steelhead. These models are based on data for PIT-tagged wild smolts, and use quantile regression to predict the probability distribution of fish arrival using flow and temperature in Lower Granite Lake. Separate arrival models were fitted for multiple populations of fish originating in the following rivers: Grande Ronde, Imnaha, South Fork Salmon, Middle Fork Salmon, and Upper Salmon. We applied these models to the 80 water years for the Proposed Action, and then combined the predicted arrival distributions for the individual populations into an overall distribution based on the average number of spawners for each population. These predicted population distributions were used as release profiles in COMPASS, where each water year had the same number of fish released.

For upper Columbia River stocks, we constructed an arrival distribution at Rock Island Dam based on observed passage of smolts (hatchery and wild, tagged and untagged). We created a multiyear average of daily proportion of smolts passing Rock Island Dam using data from 1998–2013. We then shifted this arrival distribution earlier based on the average observed travel time of smolts between Wells Dam and Rock Island Dam, and changed the release location to Wells Pool. This predicted distribution was held constant and used as the release distribution for all 80 water years in all scenarios for COMPASS runs with upper Columbia River fish.

We ran the COMPASS model for each of the 80 water years for the Proposed Action. We produced separate results for Snake River spring/summer-run Chinook and upper Columbia River spring Chinook salmon. We collected several summary measures of passage experience for each year, including in-river survival from Lower Granite Dam to Bonneville Dam for both in-river migrants, proportion of fish transported, and daily arrival distributions at Bonneville Dam tailrace for both in-river and transported fish. Since there are no collector dams on the Columbia River, no fish from upper Columbia River stocks were transported, and we did not produce any outputs related to transportation for those runs.

We also ran the Monte Carlo version of COMPASS for the Proposed Action to estimate uncertainty in predicted in-river survival. We drew 500 random parameter sets for the reservoir survival submodel and predicted in-river survival for each scenario with each parameter draw. The full results of all 500 survival estimates were provided to the life cycle modeling group for use with Monte Carlo runs of the overall life cycle model.

2.3 Results

Here we present results from prospective model runs for the Proposed Action (Tables 2-1, 2-2). For Snake River spring/summer-run Chinook salmon, 80-year average in-river survival was 50.1%, average proportion transported was 25.4%, and the average Julian date of arrival at Bonneville Dam was day 137.4 for in-river migrants and day 136.8 for transported migrants (Table 2-1, Figures 2-1, 2-2). On average, 17.2% of dam passages were via powerhouse routes (the juvenile bypass system or turbines).

For upper Columbia River spring Chinook salmon, all migrants were in-river. Average 80-year in-river survival was 51% and the average Julian date of arrival at Bonneville Dam was day 147.3 (Table 2-2). On average, 42.3% of dam passages were via powerhouse routes.

Table 2-1. Mean COMPASS statistics predicted for Snake River spring Chinook salmon for the Proposed Action management scenario.

Scenario	Mean in-river survival	Mean day at BON (in-river)	Mean day at BON (transport)	Proportion transported	Proportion powerhouse passage
Proposed Action	0.5085	137.4	136.8	0.2537	0.172

Table 2-2. Mean COMPASS statistics predicted for upper Columbia River spring Chinook salmon for the Proposed Action management scenario.

Scenario	Mean in-river survival	Mean day at BON (in-river)	Proportion powerhouse passage
Proposed Action	0.5104	147.3	0.423

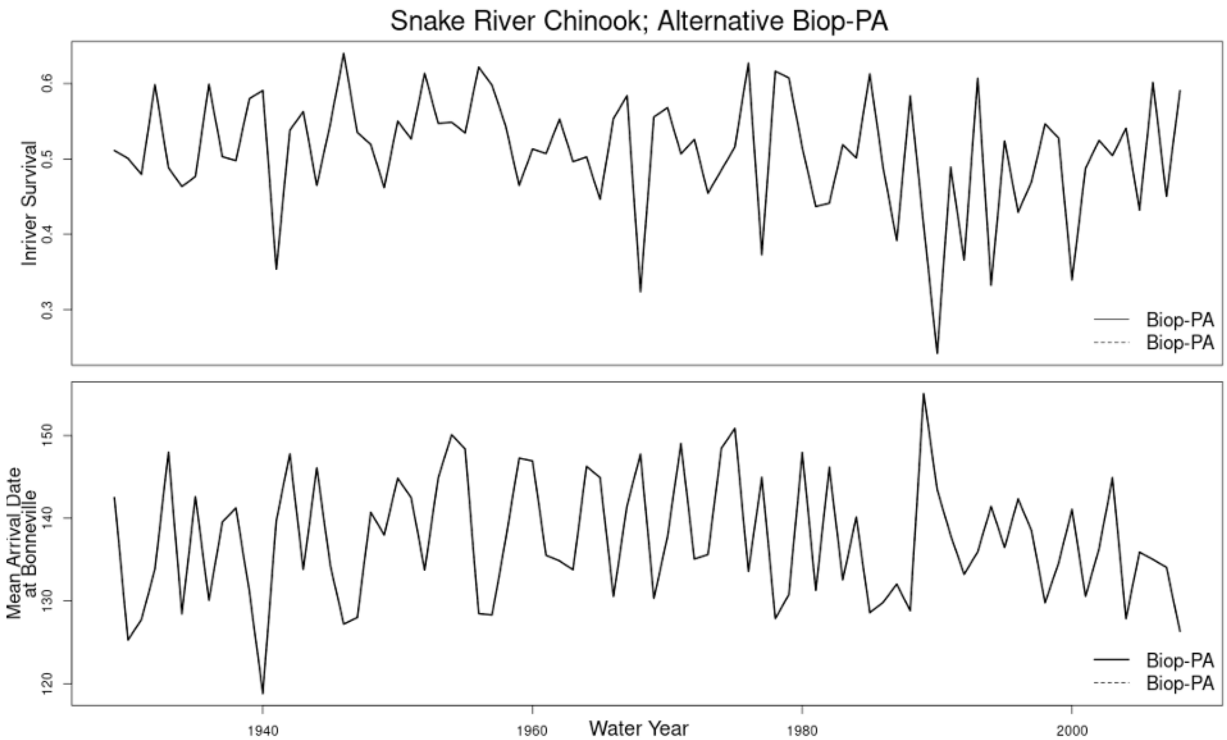

Figure 2-1. Plots of the 80 individual yearly COMPASS predictions for in-river survival and mean Julian date of arrival at Bonneville Dam for Snake River spring Chinook salmon in the Proposed Action management scenario.

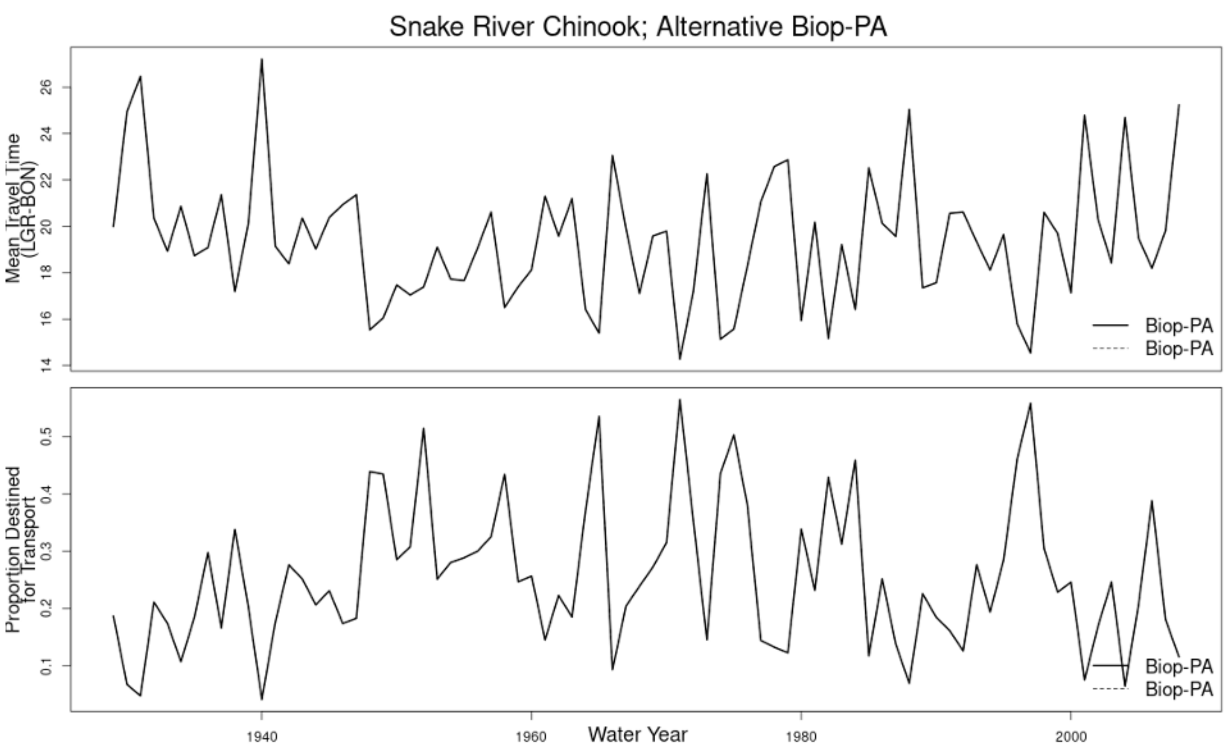

Figure 2-2. Plots of the 80 individual yearly COMPASS predictions for travel time between Lower Granite Dam and Bonneville Dam and proportion of smolts transported for Snake River spring Chinook salmon in the Proposed Action management scenario.

We produced 500 Monte Carlo estimates of survival of Snake River Chinook salmon for each of the 80 water years in the Proposed Action (Figure 2-3). Uncertainty in COMPASS survival varied by year, but generally the 95% confidence band extended around ten percentage points in survival about the deterministic estimate.

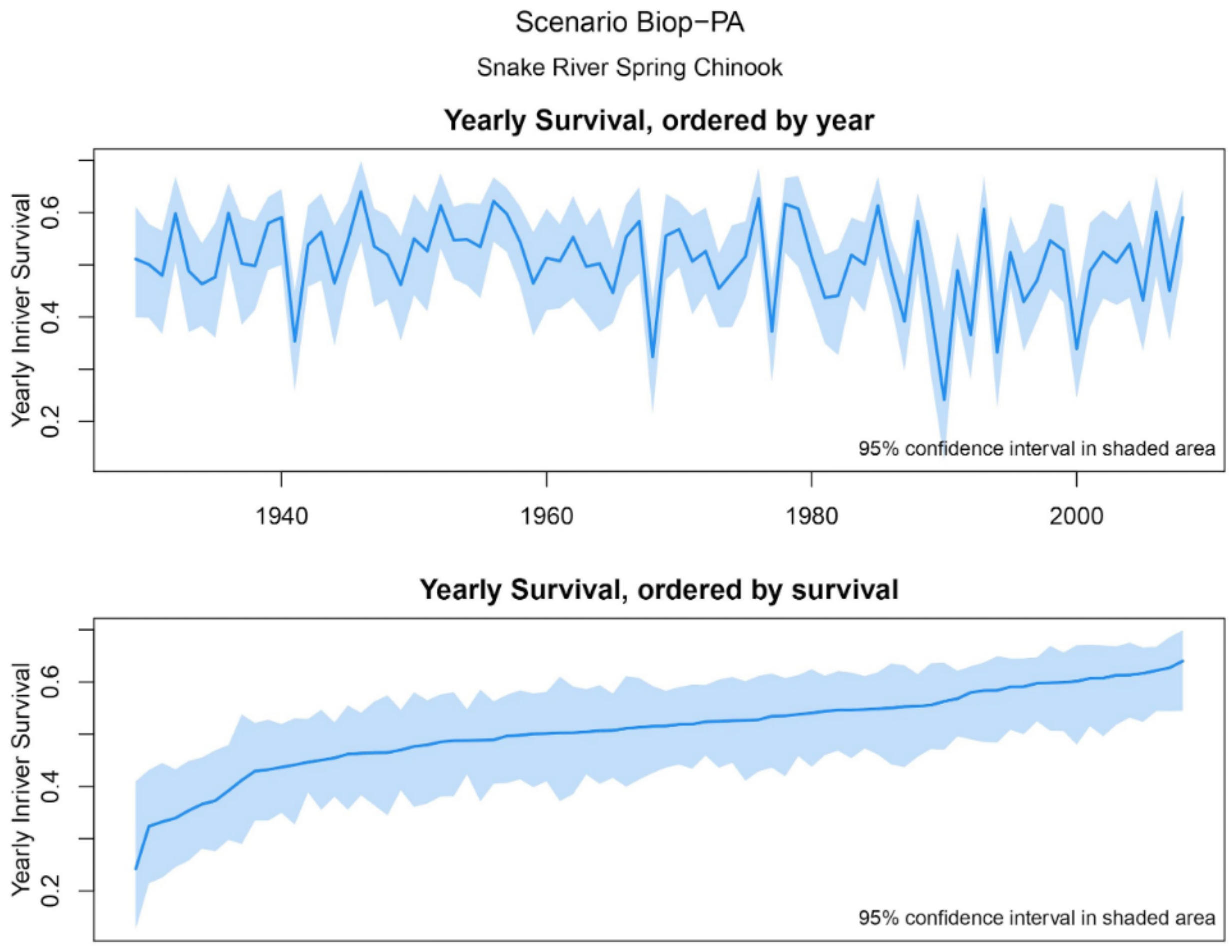

Figure 2-3. Results of the 500 Monte Carlo runs for the Proposed Action management scenario for Snake River spring Chinook salmon. The top panel shows predicted survival across the 80 water years, with the dark blue line in the center the deterministic survival estimate for that year and the shaded band containing 95% of the resulting Monte Carlo survival estimates using random survival parameter draws for that year. The bottom panel shows the same data, but with the years reordered by the deterministic survival estimate.

2.4 Discussion

The results from the COMPASS runs for the Proposed Action generally show that survival under that management regime will be in the same ballpark as other recent years (observed survival in the FCRPS for Snake River Chinook salmon has averaged almost exactly 50% from 2006 to the present). A base-case alternative directly comparable to the Proposed Actionwas not tested, so the exact magnitude of difference in survival between current operations and the PA cannot be determined with certainty, but it is most likely small.

The COMPASS model predicts that the average proportion transported will be very low for Snake River Chinook salmon under the Proposed Action. The average transportation rate from 2006 to the present has been 33.6%, with a usual start date of 1 May; the Proposed Action is predicted to have a lower transportation rate despite starting transportation operations earlier, on 20 April. The cause of the low transportation rates in the PA is the high rates of spill at FCRPS dams in the alternative.

These high rates of spill result in most fish being predicted to pass dams via the spillway, which is also the cause of the low rates of powerhouse passage seen in the PA. The predicted powerhouse passage rate for upper Columbia River Chinook salmon is substantially higher than the rate for Snake River Chinook salmon. This is because the COMPASS model estimates for upper Columbia River Chinook salmon include passage at Rock Island, Wanapum, and Priest Rapids Dams. These mid-Columbia dams are not part of FCRPS and are not operating to the same high levels of spill as the FCRPS dams, resulting in much higher powerhouse passage rates at those dams. Powerhouse passage rates at FCRPS dams are similar between both upper Columbia River Chinook and Snake River Chinook salmon.

We did not attempt to account for the negative effects of increased spill related to increased production of saturated gas and possible trauma induced by passage through highly turbulent spillways. Spill level and pattern can also create eddies in the tailraces of some dams, depending on flow and turbine operations. Fish trapped in eddies are more vulnerable to predation and are subject to longer travel times. Such conditions are not modeled in COMPASS, and effects on survival are not explicitly accounted for.

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3 Estimating Population-Level Outcomes of Restoration Alternatives in Data-Rich Watersheds: An Example from the Grande Ronde River Basin Focusing on Spring Chinook Salmon Populations

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The Grande Ronde River Basin includes six historical populations of spring Chinook salmon (Figure 3-1). 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 (Catherine Creek, Grande Ronde River upper mainstem, Lostine River, and Minam River), and adult spawning ground surveys in these populations plus Wenaha River. These five spring Chinook salmon populations represent a range of habitat conditions. The Minam and Wenaha Rivers are relatively pristine basins, although there were historical mining impacts in some parts of the drainage, and hatchery strays are present in both. 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 Grande Ronde River upper mainstem 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.

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 midsummer in-stream tagging with passive integrated transponders (PIT-tags) 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 are used to estimate juvenile rearing capacity in a state-space model.

3.1 Overview/Summary

The five Grande Ronde/Imnaha MPG spring Chinook salmon population LCMs are framed in the matrix life cycle modeling format originally described in Zabel et al. (2006) and fully developed in Chapter 5 of Pess and Jordan (2019). We used information generated from the spawner-to-smolt life stage monitoring as the basis for incorporating detailed juvenile life

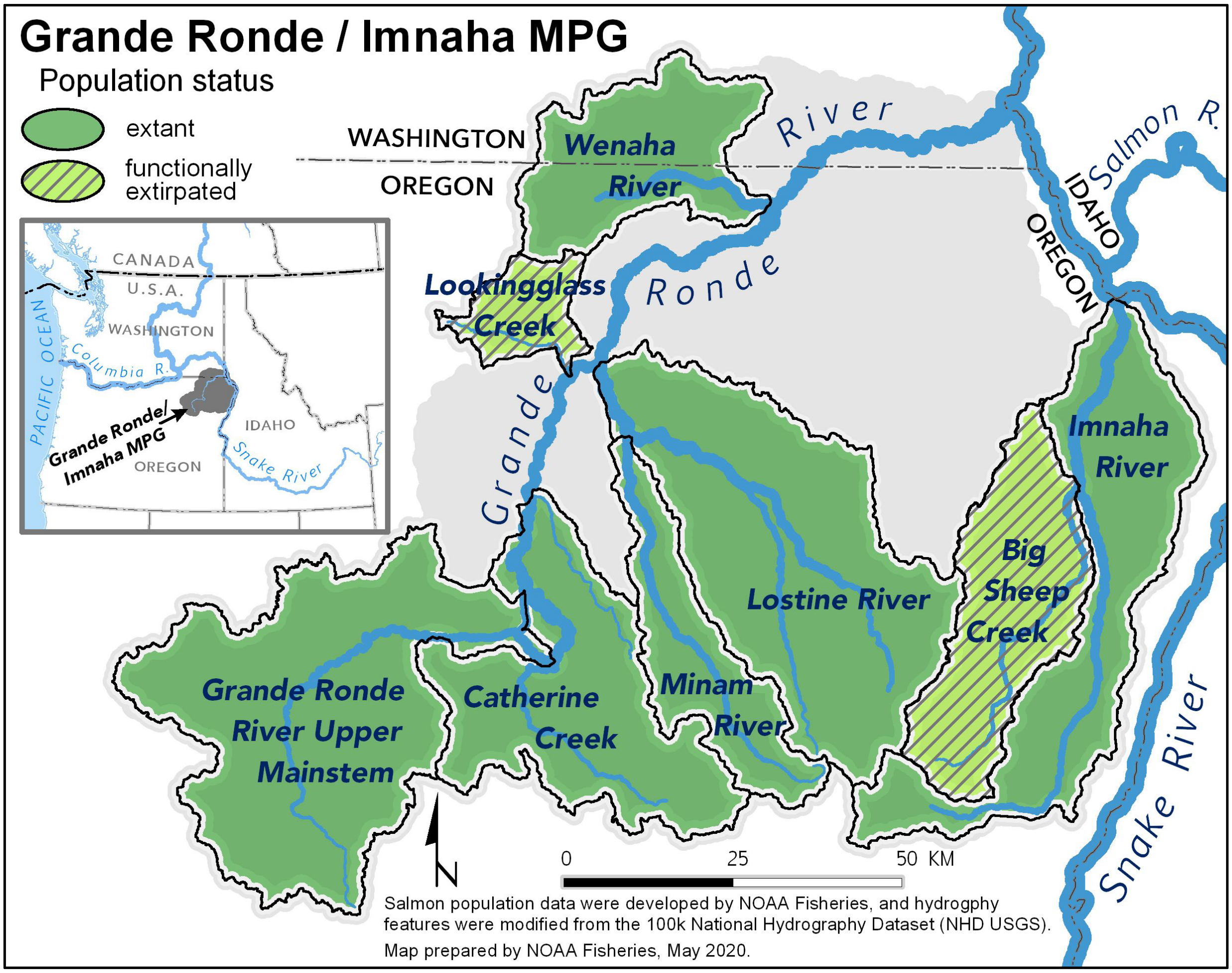

Figure 3-1. Map of the spring/summer-run Chinook salmon populations in the Grande Ronde/ Imnaha MPG. Of the six extant populations, all but Imnaha River are modeled with the Grande Ronde simulation life cycle model.

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: 1) estimation of annual spawning escapements (mid-1950s to present), 2) presmolt emigration (1992–2016 migration years) estimates of late-summer parr densities at sample sites within each population, and 3) 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, fastwater, and run habitat units in Grande Ronde River 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/Imnaha MPG matrix life cycle models (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 modeled a range of scenarios that encompass habitat restoration, mainstem hydrosystem operations, and climate change projections. The scenarios represent a base case with a stationary (current) climate, current FCRPS operations, and tributary habitat conditions reflecting restoration actions through 2021. The management action scenarios include the NMFS (2020) Proposed Action hydrosystem operation and habitat restoration plan, and two levels of projected benefit through reduced latent mortality. The management action scenario is also applied across a range of future climate conditions impacting mainstem and ocean survival.

3.2 Grande Ronde River LCM Structure

Our five Grande Ronde/Imnaha MPG spring Chinook salmon population LCMs are framed in the MLCM format originally described in Zabel et al. (2006) and similar to other LCMs for several Salmon River basin populations (Jordan et al., this volume, Chapter 4, Crozier et al., this volume, Chapter 5) and the Wenatchee River (Chapter 6 of Pess and Jordan 2019, Jorgensen and Bond, this volume, Chapter 6)—although here, 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/Imnaha MPG populations.

A detailed description of the freshwater tributary life stage elements of the model can be found in Chapter 5 of Pess and Jordan (2019). 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 (Faulkner et al., this volume, Chapter 2, Chasco et al. submitted, and Crozier et al. submitted a). Snake River spring/summer-run Chinook salmon 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/Imnaha MPG populations have active local broodstock supplementation programs. Broodstocking for each of those programs is managed with population-specific schedules. We include modules in the population models that mimic the schedules and recent performances of the supplementation programs, including survivals to release and smolt-to-adult return rates.

The modeled scenario set represents the Proposed Action of NMFS (2020). In total, 13 scenarios were run for each population—a no-action alternative (NAA) and four climate scenarios (stationary, low, medium, and high), each with a range of latent mortality reduction (none, low, high) to test the potential benefit of mainstem hydrosytem operations. The same patterns of hatchery and harvest actions were applied to all scenarios, and we applied the same pattern of freshwater habitat action benefit to all Proposed Action scenarios (except NAA).

The Grande Ronde/Imnaha MPG 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.

3.2.1 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 extrapolate 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.

Juvenile spring/summer-run 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 all populations but Wenaha River, 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, fastwater, and run habitat units in Grande Ronde River 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 rearing habitat with summer maximum weekly maximum stream temperatures (MWMT) 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 traps. We developed production relationships for the reaches above the weir sites, standardized to a common unit of habitat (10,000 m^2 of pool-equivalent area) using the habitat datasets described above.

3.2.1.1 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 (R Core Team 2016). We assumed a lognormal error structure and weighted five-year-old parent spawners by 1.26 (NMFS 2007) to account for higher fecundity of these females. The Beverton–Holt model, with its density-dependent term, was a better fit to the data series for each population.

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.

3.2.1.2 Summer parr-to-spring tributary outmigrant stage

The combination of life stage PIT-tag groups available for the Grande Ronde/Imnaha MPG populations represents 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. Summer parr estimates are generated based on sampling in August, while fall downstream migrants passing the smolt traps generally peak in mid-October. 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 and 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/Imnaha MPG, including survival estimates derived from winter tagging above the smolt traps after fall emigration.

3.2.1.3 Spring outmigrant-to-Lower Granite Dam stage

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

3.2.1.4 Lower Granite Dam to Lower Granite Dam

The Grande Ronde/Imnaha MPG population models include common mainstem and ocean survival modules shared by the Upper Salmon River (Jordan et al., this volume, Chapter 4) and Snake River (Crozier et al., this volume, Chapter 5) models. Models for survival in these stages are described completely in associated chapters and journal publications (mainstem COMPASS modeling: Faulkner et al., this volume, Chapter 2; ocean survival: Chasco et al. submitted); upstream survival: Crozier et al. submitted a). The three life stages covered by the common mainstem/ocean module are: downstream migration through the hydropower system, Bonneville-to-Bonneville SARs, and upstream migration.

The included hydrosystem operations scenarios are described in Faulkner et al. (this volume, Chapter 2). In brief, flow, spill, reservoir elevation, water temperature, and dissolved gas for the Proposed Action were all modeled by the U.S. Army Corps of Engineers (USACE). These models assume turbine replacements at Ice Harbor, McNary, and John Day Dams that have substantially lower fish mortality than the existing turbines.

These runs used a universal transportation start date of 20 April at all three transporter dams: Lower Granite, Little Goose, and Lower Monumental Dams. After this date, all fish predicted to enter the bypass system at these dams were treated as transported fish by the COMPASS model; they are removed from the river at the transport dam, and added to the tailrace of Bonneville Dam two days later. COMPASS assumes uniform 0.98 survival during transportation. In each simulation, the COMPASS model produced distributions of arrival times for in-river and transported smolts at Bonneville Dam, which were then input into the SAR model.

Chasco et al. (submitted) used a mixed-effects logistic regression model for wild fish to determine the effect of the date of ocean entry (from COMPASS) and environmental covariates (specified by the climate scenario) on the probability that an individual fish would return as an adult to Bonneville Dam. The model includes random effects for day and for the day by year interaction, which follow an autoregressive process over time. We developed separate models for fish that had migrated through the mainstem in the river and for fish that had been transported downstream in barges. The downstream survival models were structured as follows: the model for transported fish included only a single covariate, a large-scale measure of sea surface temperature (SST), and a model for in-river migrants that included two covariates, *SSTarc* in winter and a more local measure of SST along the Washington coast.

To assess the implications of speculative reductions in delayed mortality for in-river migrating fish, a multiplicative factor was include to increase aggregate hydrosystem survival. In these latent mortality (LM) scenarios, we increased marine survival rates of in-river migrants by 17% and 35%.

To account for adult upstream survival, Crozier et al. (submitted b) used generalized additive mixed models (GAMMs) to evaluate the effects of both anthropogenic and environmental covariates on spring/summer Chinook salmon survival. To develop simulation output, all of the nonenvironmental covariates (fisheries catch, the proportion of fish that had been transported in barges as juveniles, etc.) had similar distributions to the baseline period (2004–16), and survival from the hydrosystem to spawning was treated as constant due to the lack of appropriate data for most populations with which to fit a relationship.

To compare population trajectories in a climate experiencing historical levels of variability but no directional trends (a "stationary" climate), with population trajectories in a climate responding to anthropogenic greenhouse gases, Crozier et al. (submitted a) generated a suite of simulated scenarios. They created a stationary climate scenario by fitting a covariance matrix for all the freshwater and marine environmental covariates used in the life cycle model. The covariance matrix was then incorporated into a multivariate state-space model that accounts for temporal correlations across environmental variables. Autoregression was further incorporated into the random effects within the SAR model to account for additional temporal patterns that were not captured in the raw environmental time series included in the selected covariates. The state-space model was used to simulate natural variability in all covariates in a stationary climate.

To represent changing climate scenarios, Crozier et al. (submitted a) added trends to the stationary simulations. To simulate a temporal signal in these scenarios, they extracted trends from global climate model (GCM) projections of representative concentration pathway (RCP) 8.5, the "business-as-usual" scenario. To extract the relevant trends from GCMs, Crozier et al. (submitted a) calculated the variable mean over a baseline period centered on 2015 (2005–25), created monthly anomalies from the 2005–25 period for each time series, created a 20-year running mean of anomalies for each time series, and calculated the 25th, 50th, and 75th quantiles for each month from the smoothed anomalies across all of the time series. These roughly linear trends represent the spread across climate models of low, medium, and high rates of change in each covariate.

3.2.2 Output metrics

We report model output across the scenarios with two performance metrics: abundance and risk of quasi-extinction. Abundance is calculated as the geometric mean of spawner abundance over Years 15 to 24 and reported as the 5th, 25th, 50th, 75th, and 95th percentiles of the distribution of abundances across the 1,000 replicate simulations run for each scenario. Risk of quasi-extinction is calculated as the fraction of replicate simulation abundance levels that fall to either 30 or 50 spawners for four consecutive years by Year 24 (pQET). As with the abundance metric, the quasi-extinction risk is also reported as the distribution of the resulting estimates of pQET. To generate a distribution of pQET, we constructed 100 bootstrap samples (sampling with replacement) of 100 replicate simulations from the 1,000 simulations for each scenario. We estimated pQET for each of the 100 samples of 100 replicates and report the 5th, 25th, 50th, 75th, and 95th percentiles of the pQET distributions.

3.3 Developing Restoration Scenarios

Current spawning and juvenile rearing habitats for each of the five populations extend from higher-elevation moderate-gradient forested valleys downstream through lower-gradient alluvial fan and Grande Ronde Valley habitats. The Catherine Creek and Grande Ronde River upper mainstem populations, along with the Wallowa and Lower Lostine River reaches in the Lostine River population, have been substantially altered by human impacts, including channel straightening, diking, large woody debris (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. 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.

3.3.1 Estimating habitat change inputs for the LCMs

The Grande Ronde/Imnaha 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 models, overwintering survival is linked to summer parr density, reflecting the strong patterns in the empirical datasets 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 datasets are related to the estimates of available habitat generated using the Oregon AQI datasets. 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.

3.3.2 Current habitat conditions

The current distribution of redds in Catherine Creek is largely restricted to reaches upstream of the ODFW weir site. Less than 5% of redds counted in annual surveys from 2009–16 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 Grande Ronde/Imnaha MPG are in the upper sections above Sheep Creek. Current redd surveys do not cover the mainstem reach passing through Vey Meadows. The Vey Meadows reach was 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 average pool proportion above the AQI survey reach was roughly 50% of the gravel survey estimate 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 surveys. 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.

In recent years (2009–19+), ODFW has included georeferencing of individual redd counts in their annual spring Chinook salmon redd surveys in the Grande Ronde basin. ODFW complemented their summer parr snorkel surveys (ISEMP–CHaMP 2018) by sampling contiguous reaches from near La Grande upstream to the upper reaches of the East Fork Grande Ronde River. We contrasted the resulting adult spawning and parr density patterns with reach-specific NorWeST-derived August stream temperatures 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 Grande Ronde/Imnaha, 95% of the georeferenced redds were upstream of the reach where average NorWeST stream temperatures exceed 17.5°C.

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 Grande Ronde River. 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 (Catherine Creek and Grande Ronde River upper mainstem) exhibited relatively high temperatures at the downstream end of current use as defined by ODFW. Other variables quantified by ODFW in the Grand Ronde River basin include reach-level longitudinal surveys summarized by habitat type, sediment characteristics, and estimates of LWD.

3.3.3 Current strategy-specific restoration scenarios

The action agencies have committed to pursue additional actions within the Grande Ronde/ Imnaha MPG, targeting the same strategic priorities as in past Biological Opinions (NMFS 2000, 2010, 2019). 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 MPG, 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 NMFS (2000) tributary habitat strategy. Assuming that they accomplish the minimum levels of habitat improvement they identify over the 15-year term of the Biological Opinion, the estimated benefit could be a 50% improvement in average habitat capacity across the treated watersheds. Allowing the potential for restoration actions to continue to accrue benefit could result in a 70% improvement after 24 years.

3.3.4 Proposed Action 15-year action plan

Pess and Jordan (2019), Chapter 5, describes the analytical process by which past tributary actions, mechanistic models, and detailed future action plans (e.g., the Grande Ronde Atlas[1]) were used to develop projected benefits to spawning and rearing Chinook salmon in the Grande Ronde River basin. Using the projected 2019–24 benefits projections developed

[1] https://grmw.org/

Table 3-1. Proposed Action (NMFS 2019, 2020) for habitat restoration actions in the Grande Ronde/Imnaha MPG. Actions are assumed to be applied only to the Catherine Creek and Grande Ronde River upper mainstem populations.

	Spawn/rear habitat length (km)	NMFS (2019) PA (2019–21)		NMFS (2020) PA (2021–36)	
		Habitat access (km/% of total)	Stream complexity (km/% of total)	Habitat access (km/% of total)	Stream complexity (km/% of total)
Grande Ronde/Imnaha MPG (Catherine Creek and Grande Ronde River upper mainstem populations only)	340	37/11	13/4	139/41	38/11

fully in Chapter 5 of Pess and Jordan (2019), we extend that level of effort for two additional 5-year action plans as the framework for estimating the habitat action benefit in the Grande Ronde River basin under NMFS (2020). We assume that the capacity and survival benefits estimates presented in Chapter 5 of Pess and Jordan (2019) are achievable under two additional implementation instances (Years 5–10 and 10–15), and that the effects accumulate in a simple additive manner (Table 3-1). No reach-scale plans have been developed for this time horizon, so some assumptions are necessary in order to project potential restoration activities into the future (Year 15).

3.3.5 Habitat capacity projections

Figure 3-2 depicts the projected increases in juvenile rearing capacity for the range of scenarios run for the Catherine Creek and Grande Ronde River upper mainstem populations. The projections clearly illustrate some of the key assumptions behind the model inputs for habitat restoration actions. We assumed that each proposed action set (three planning/implementation blocks of five years each) 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 habitat changes with associated survival/production changes 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.

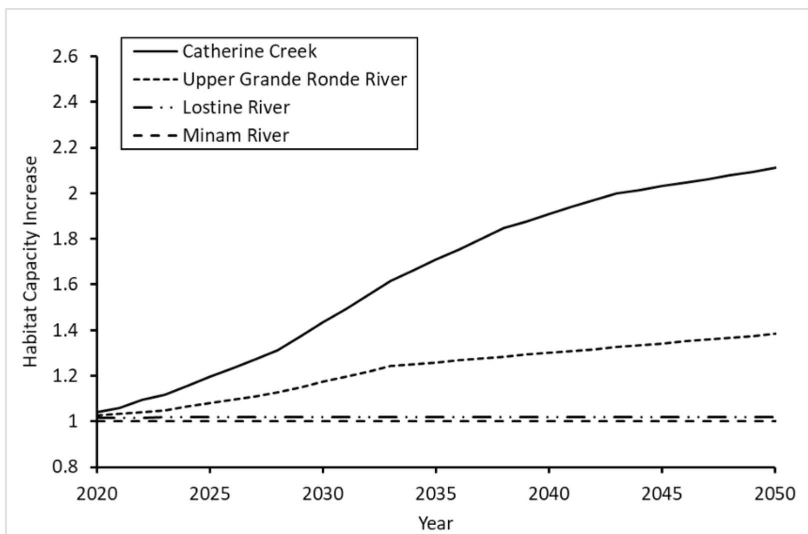

Figure 3-2. Habitat capacity increase model input resulting from proposed restoration actions. The habitat capacity increase multiplier increases from 1.0 in the two treated populations: Catherine Creek (solid line) and Grande Ronde River upper mainstem (dotted line). The parameter values for Lostine River (dash/dot line) and Minam River (dashed line) are shown for comparison—no planned actions result in no modeled increase in tributary habitat capacity.

For both populations, the projected habitat response of implementing the proposed 2021–36 projects results in larger proportional increases than those associated with the past actions. The 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 Grande Ronde River upper mainstem for the long-term scenario (which includes substantial additional riparian restoration) is large, reflecting the importance of reducing temperatures for this population (e.g., Justice et al. 2017).

3.4 LCM 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 summarized results over 1,000 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 and a range of future climate impacts on survival. 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 salmon populations (Figure 3-3, Tables 3-2, 3-3, and 3-4). The box outline in each graph illustrates the middle 50% of modeled outcomes across the 1,000 runs for each scenario; the whiskers capture 99% of the outcomes.

Projecting forward the impacts of the tributary habitat improvements results in a large improvement in natural-origin spawner abundance for Catherine Creek and a negligible change for Grande Ronde River upper mainstem. It is important to note that for the supplemented populations (Catherine Creek, Grande Ronde River upper mainstem, and Lostine River), adult returns from natural-origin broodstock hatchery releases also contribute to spawning (Figure 3-4, Tables 3-5, 3-6, and 3-7). For example, the median projections for total spawners (natural-origin plus hatchery supplementation returns) increased to over 500 each for the Catherine Creek, Grande Ronde River upper mainstem, and Lostine River populations. 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 Grande Ronde River upper mainstem populations.

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 or 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 or 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 2 contiguous km 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.

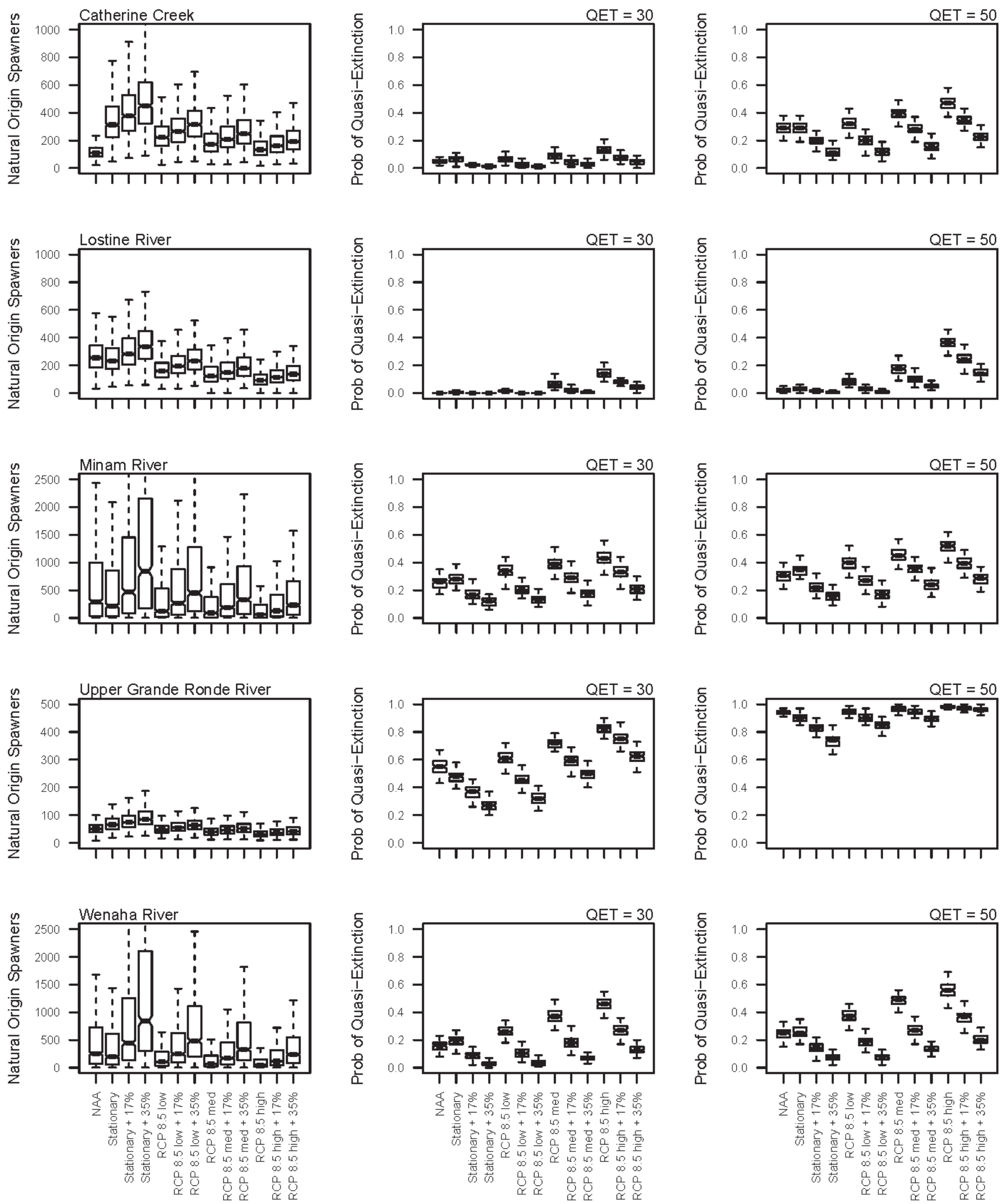

Figure 3-3. Estimates of natural-origin spawners and the quasi-extinction risk (QET 30 and 50) for the five populations and the 13 scenarios used in the Grande Ronde simulation life cycle model.

Table 3-2. Estimates of natural-origin spawners for the five populations and 13 scenarios output from the Grande Ronde simulation life cycle model. The climate scenarios are *Stationary* and a range of global climate model (GCM) projections for the representative concentration pathway (*RCP 8.5*) emissions scenarios. *Low*, *mean*, and *high* reflect the 25th, 50th, and 75th quantiles across GCM time series. A range of percentiles (5%, 25%, 50%, 75%, 95%) is shown for abundance across simulations for each scenario in each population.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Catherine Creek					
NAA	53	83	110	145	210
Stationary	126	226	313	446	679
Stationary + 17%	166	268	376	526	835
Stationary + 35%	196	320	451	620	958
RCP 8.5 low	88	160	222	302	469
RCP 8.5 low + 17%	120	190	266	358	602
RCP 8.5 low + 35%	141	228	315	415	689
RCP 8.5 mean	72	122	172	247	416
RCP 8.5 mean + 17%	89	150	207	300	487
RCP 8.5 mean + 35%	105	178	247	348	565
RCP 8.5 high	53	93	134	194	298
RCP 8.5 high + 17%	67	117	162	231	359
RCP 8.5 high + 35%	81	137	191	272	437
Lostine River					
NAA	116	186	254	342	534
Stationary	102	173	233	325	513
Stationary +17%	129	206	281	394	612
Stationary + 35%	146	246	336	448	701
RCP 8.5 low	66	114	159	219	351
RCP 8.5 low +17%	89	143	193	269	422
RCP 8.5 low + 35%	108	171	232	313	505
RCP 8.5 mean	45	84	123	188	292
RCP 8.5 mean +17%	60	102	148	221	357
RCP 8.5 mean + 35%	74	124	178	259	406
RCP 8.5 high	34	62	92	135	224
RCP 8.5 high +17%	41	77	114	166	267
RCP 8.5 high + 35%	51	95	137	195	323
Minham River					
NAA	1	38	293	1,000	3,032
Stationary	1	29	217	853	2,999
Stationary +17%	2	86	465	1,453	3,779
Stationary + 35%	7	170	843	2,155	4,762
RCP 8.5 low	1	18	123	535	1,752
RCP 8.5 low +17%	3	52	268	881	2,563
RCP 8.5 low + 35%	8	119	450	1,275	3,293
RCP 8.5 mean	1	11	91	374	1,410
RCP 8.5 mean +17%	2	30	190	610	1,904
RCP 8.5 mean + 35%	6	68	331	934	2,473
RCP 8.5 high	1	8	61	237	990
RCP 8.5 high +17%	1	25	127	424	1,399
RCP 8.5 high + 35%	4	58	230	664	1,806

Table 3-2 (continued). Estimates of natural-origin spawners for the five populations and 13 scenarios output from the Grande Ronde simulation life cycle model.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Grande Ronde River (um)[a]					
NAA	28	39	50	64	89
Stationary	34	50	66	87	123
Stationary + 17%	39	57	74	99	142
Stationary + 35%	46	66	85	114	161
RCP 8.5 low	26	37	48	61	93
RCP 8.5 low + 17%	30	43	55	72	104
RCP 8.5 low + 35%	34	49	63	80	120
RCP 8.5 mean	20	29	39	52	81
RCP 8.5 mean + 17%	23	34	46	60	92
RCP 8.5 mean + 35%	26	39	51	68	101
RCP 8.5 high	16	24	31	42	61
RCP 8.5 high + 17%	19	28	36	48	70
RCP 8.5 high + 35%	21	32	42	56	78
Wenaha River					
NAA	11	68	255	722	2,698
Stationary	5	58	195	616	2,248
Stationary +17%	22	140	446	1,254	3,533
Stationary + 35%	56	301	844	2,105	5,345
RCP 8.5 low	6	38	106	286	1,308
RCP 8.5 low +17%	22	94	247	627	2,077
RCP 8.5 low + 35%	49	199	482	1,109	3,056
RCP 8.5 mean	4	24	68	219	973
RCP 8.5 mean +17%	12	63	170	459	1,701
RCP 8.5 mean + 35%	31	131	328	812	2,455
RCP 8.5 high	2	15	50	152	657
RCP 8.5 high +17%	7	41	116	316	1,107
RCP 8.5 high + 35%	18	86	240	544	1,793

[a] *(um)* = upper mainstem

Table 3-3. Estimates of probability of quasi-extinction at Year 24 with a QET of 30 spawners for the five populations and 13 scenarios output from the Grande Ronde simulation life cycle model. The climate scenarios are *Stationary* and a range of global climate model (GCM) projections for the representative concentration pathway (*RCP 8.5*) emissions scenarios. *Low*, *mean*, and *high* reflect the 25th, 50th, and 75th quantiles across GCM time series. A range of percentiles (5%, 25%, 50%, 75%, 95%) is shown for abundance across simulations for each scenario in each population.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Catherine Creek					
NAA	0.02	0.04	0.05	0.06	0.08
Stationary	0.03	0.05	0.07	0.08	0.10
Stationary + 17%	0.01	0.02	0.02	0.03	0.05
Stationary + 35%	0.00	0.01	0.01	0.02	0.04
RCP 8.5 low	0.03	0.05	0.06	0.08	0.09
RCP 8.5 low + 17%	0.00	0.01	0.02	0.04	0.05
RCP 8.5 low + 35%	0.00	0.01	0.01	0.02	0.04
RCP 8.5 mean	0.05	0.07	0.09	0.10	0.14
RCP 8.5 mean + 17%	0.02	0.03	0.05	0.06	0.09
RCP 8.5 mean + 35%	0.00	0.02	0.02	0.04	0.05
RCP 8.5 high	0.08	0.11	0.13	0.15	0.19
RCP 8.5 high + 17%	0.03	0.06	0.07	0.09	0.12
RCP 8.5 high + 35%	0.01	0.03	0.05	0.06	0.08
Lostine River					
NAA	0.00	0.00	0.00	0.00	0.01
Stationary	0.00	0.00	0.00	0.01	0.01
Stationary +17%	0.00	0.00	0.00	0.00	0.00
Stationary + 35%	0.00	0.00	0.00	0.00	0.01
RCP 8.5 low	0.00	0.01	0.01	0.02	0.03
RCP 8.5 low +17%	0.00	0.00	0.00	0.00	0.01
RCP 8.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean	0.03	0.04	0.06	0.08	0.10
RCP 8.5 mean +17%	0.00	0.01	0.02	0.03	0.05
RCP 8.5 mean + 35%	0.00	0.00	0.01	0.01	0.02
RCP 8.5 high	0.10	0.12	0.14	0.17	0.19
RCP 8.5 high +17%	0.04	0.07	0.08	0.09	0.12
RCP 8.5 high + 35%	0.01	0.03	0.04	0.05	0.07
Minham River					
NAA	0.20	0.22	0.26	0.28	0.31
Stationary	0.22	0.25	0.28	0.31	0.35
Stationary +17%	0.12	0.14	0.17	0.20	0.23
Stationary + 35%	0.07	0.09	0.12	0.14	0.16
RCP 8.5 low	0.27	0.31	0.34	0.38	0.41
RCP 8.5 low +17%	0.15	0.18	0.20	0.23	0.26
RCP 8.5 low + 35%	0.09	0.11	0.14	0.15	0.18
RCP 8.5 mean	0.32	0.36	0.39	0.42	0.45
RCP 8.5 mean +17%	0.23	0.26	0.29	0.32	0.37
RCP 8.5 mean + 35%	0.13	0.15	0.18	0.20	0.25
RCP 8.5 high	0.35	0.40	0.43	0.47	0.52
RCP 8.5 high +17%	0.26	0.30	0.33	0.37	0.41
RCP 8.5 high + 35%	0.14	0.18	0.21	0.23	0.29

Table 3-3 (continued). Estimates of probability of quasi-extinction at Year 24 with a QET of 30 spawners for the five populations and 13 scenarios output from the Grande Ronde simulation life cycle model.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Grande Ronde River (um)[a]					
NAA	0.47	0.51	0.55	0.59	0.61
Stationary	0.40	0.45	0.48	0.50	0.54
Stationary + 17%	0.29	0.33	0.37	0.40	0.43
Stationary + 35%	0.21	0.24	0.27	0.29	0.34
RCP 8.5 low	0.53	0.58	0.61	0.65	0.69
RCP 8.5 low + 17%	0.38	0.43	0.45	0.48	0.53
RCP 8.5 low + 35%	0.26	0.29	0.32	0.35	0.40
RCP 8.5 mean	0.66	0.69	0.72	0.74	0.78
RCP 8.5 mean + 17%	0.53	0.56	0.59	0.62	0.68
RCP 8.5 mean + 35%	0.42	0.47	0.50	0.52	0.57
RCP 8.5 high	0.77	0.80	0.83	0.85	0.88
RCP 8.5 high + 17%	0.69	0.72	0.75	0.78	0.82
RCP 8.5 high + 35%	0.55	0.59	0.63	0.65	0.69
Wenaha River					
NAA	0.10	0.13	0.16	0.18	0.21
Stationary	0.14	0.17	0.20	0.22	0.25
Stationary +17%	0.05	0.07	0.09	0.10	0.14
Stationary + 35%	0.01	0.02	0.03	0.04	0.07
RCP 8.5 low	0.19	0.24	0.26	0.29	0.33
RCP 8.5 low +17%	0.06	0.08	0.11	0.13	0.16
RCP 8.5 low + 35%	0.01	0.02	0.03	0.05	0.07
RCP 8.5 mean	0.31	0.34	0.37	0.41	0.46
RCP 8.5 mean +17%	0.12	0.15	0.18	0.21	0.23
RCP 8.5 mean + 35%	0.04	0.06	0.07	0.08	0.11
RCP 8.5 high	0.39	0.43	0.46	0.49	0.53
RCP 8.5 high +17%	0.21	0.24	0.27	0.30	0.34
RCP 8.5 high + 35%	0.08	0.11	0.13	0.15	0.18

[a] *(um)* = upper mainstem

Table 3-4. Estimates of probability of quasi-extinction at Year 24 with a QET of 50 spawners for the five populations and 13 scenarios output from the Grande Ronde simulation life cycle model. The climate scenarios are *Stationary* and a range of global climate model (GCM) projections for the representative concentration pathway (*RCP 8.5*) emissions scenarios. *Low*, *mean*, and *high* reflect the 25th, 50th, and 75th quantiles across GCM time series. A range of percentiles (5%, 25%, 50%, 75%, 95%) is shown for abundance across simulations for each scenario in each population.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Catherine Creek					
NAA	0.21	0.26	0.29	0.32	0.35
Stationary	0.24	0.26	0.29	0.32	0.36
Stationary + 17%	0.14	0.18	0.20	0.22	0.26
Stationary + 35%	0.07	0.09	0.11	0.14	0.18
RCP 8.5 low	0.25	0.29	0.32	0.35	0.39
RCP 8.5 low + 17%	0.14	0.17	0.20	0.23	0.27
RCP 8.5 low + 35%	0.07	0.10	0.12	0.14	0.18
RCP 8.5 mean	0.31	0.37	0.40	0.42	0.47
RCP 8.5 mean + 17%	0.22	0.26	0.28	0.31	0.34
RCP 8.5 mean + 35%	0.09	0.13	0.15	0.18	0.22
RCP 8.5 high	0.40	0.44	0.47	0.50	0.54
RCP 8.5 high + 17%	0.28	0.32	0.34	0.37	0.42
RCP 8.5 high + 35%	0.17	0.21	0.23	0.25	0.29
Lostine River					
NAA	0.00	0.01	0.02	0.03	0.04
Stationary	0.01	0.02	0.03	0.04	0.05
Stationary +17%	0.00	0.01	0.01	0.02	0.03
Stationary + 35%	0.00	0.00	0.01	0.01	0.03
RCP 8.5 low	0.04	0.06	0.08	0.10	0.13
RCP 8.5 low +17%	0.01	0.02	0.03	0.04	0.05
RCP 8.5 low + 35%	0.00	0.00	0.01	0.01	0.03
RCP 8.5 mean	0.11	0.15	0.18	0.20	0.23
RCP 8.5 mean +17%	0.06	0.08	0.10	0.12	0.14
RCP 8.5 mean + 35%	0.02	0.04	0.05	0.06	0.09
RCP 8.5 high	0.29	0.34	0.37	0.39	0.44
RCP 8.5 high +17%	0.19	0.22	0.24	0.27	0.32
RCP 8.5 high + 35%	0.10	0.13	0.14	0.17	0.20
Minham River					
NAA	0.24	0.27	0.31	0.33	0.37
Stationary	0.29	0.31	0.35	0.37	0.43
Stationary +17%	0.16	0.20	0.22	0.25	0.29
Stationary + 35%	0.10	0.13	0.16	0.18	0.21
RCP 8.5 low	0.31	0.36	0.40	0.43	0.48
RCP 8.5 low +17%	0.19	0.24	0.27	0.30	0.34
RCP 8.5 low + 35%	0.11	0.14	0.17	0.20	0.22
RCP 8.5 mean	0.38	0.42	0.45	0.49	0.54
RCP 8.5 mean +17%	0.28	0.33	0.36	0.38	0.41
RCP 8.5 mean + 35%	0.18	0.21	0.24	0.27	0.33
RCP 8.5 high	0.44	0.49	0.52	0.55	0.59
RCP 8.5 high +17%	0.32	0.36	0.40	0.43	0.46
RCP 8.5 high + 35%	0.22	0.25	0.29	0.31	0.35

Table 3-4 (continued). Estimates of probability of quasi-extinction at Year 24 with a QET of 50 spawners for the five populations and 13 scenarios output from the Grande Ronde simulation life cycle model.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Grande Ronde River (um)[a]					
NAA	0.90	0.93	0.94	0.95	0.97
Stationary	0.86	0.88	0.91	0.92	0.95
Stationary + 17%	0.77	0.81	0.83	0.85	0.89
Stationary + 35%	0.67	0.70	0.74	0.76	0.81
RCP 8.5 low	0.90	0.93	0.95	0.96	0.98
RCP 8.5 low + 17%	0.86	0.88	0.90	0.92	0.95
RCP 8.5 low + 35%	0.79	0.83	0.85	0.87	0.90
RCP 8.5 mean	0.94	0.95	0.97	0.98	0.99
RCP 8.5 mean + 17%	0.91	0.93	0.94	0.96	0.98
RCP 8.5 mean + 35%	0.85	0.88	0.90	0.91	0.94
RCP 8.5 high	0.96	0.98	0.98	0.99	1.00
RCP 8.5 high + 17%	0.95	0.96	0.97	0.98	1.00
RCP 8.5 high + 35%	0.93	0.95	0.96	0.97	0.99
Wenaha River					
NAA	0.18	0.22	0.25	0.27	0.31
Stationary	0.20	0.23	0.25	0.29	0.33
Stationary +17%	0.08	0.12	0.14	0.17	0.21
Stationary + 35%	0.03	0.06	0.07	0.09	0.11
RCP 8.5 low	0.30	0.35	0.38	0.41	0.44
RCP 8.5 low +17%	0.13	0.16	0.19	0.21	0.25
RCP 8.5 low + 35%	0.04	0.06	0.08	0.09	0.12
RCP 8.5 mean	0.41	0.46	0.49	0.51	0.56
RCP 8.5 mean +17%	0.20	0.24	0.27	0.30	0.34
RCP 8.5 mean + 35%	0.09	0.12	0.13	0.15	0.18
RCP 8.5 high	0.49	0.52	0.56	0.60	0.63
RCP 8.5 high +17%	0.30	0.33	0.37	0.39	0.43
RCP 8.5 high + 35%	0.15	0.18	0.20	0.23	0.27

[a] *(um)* = upper mainstem

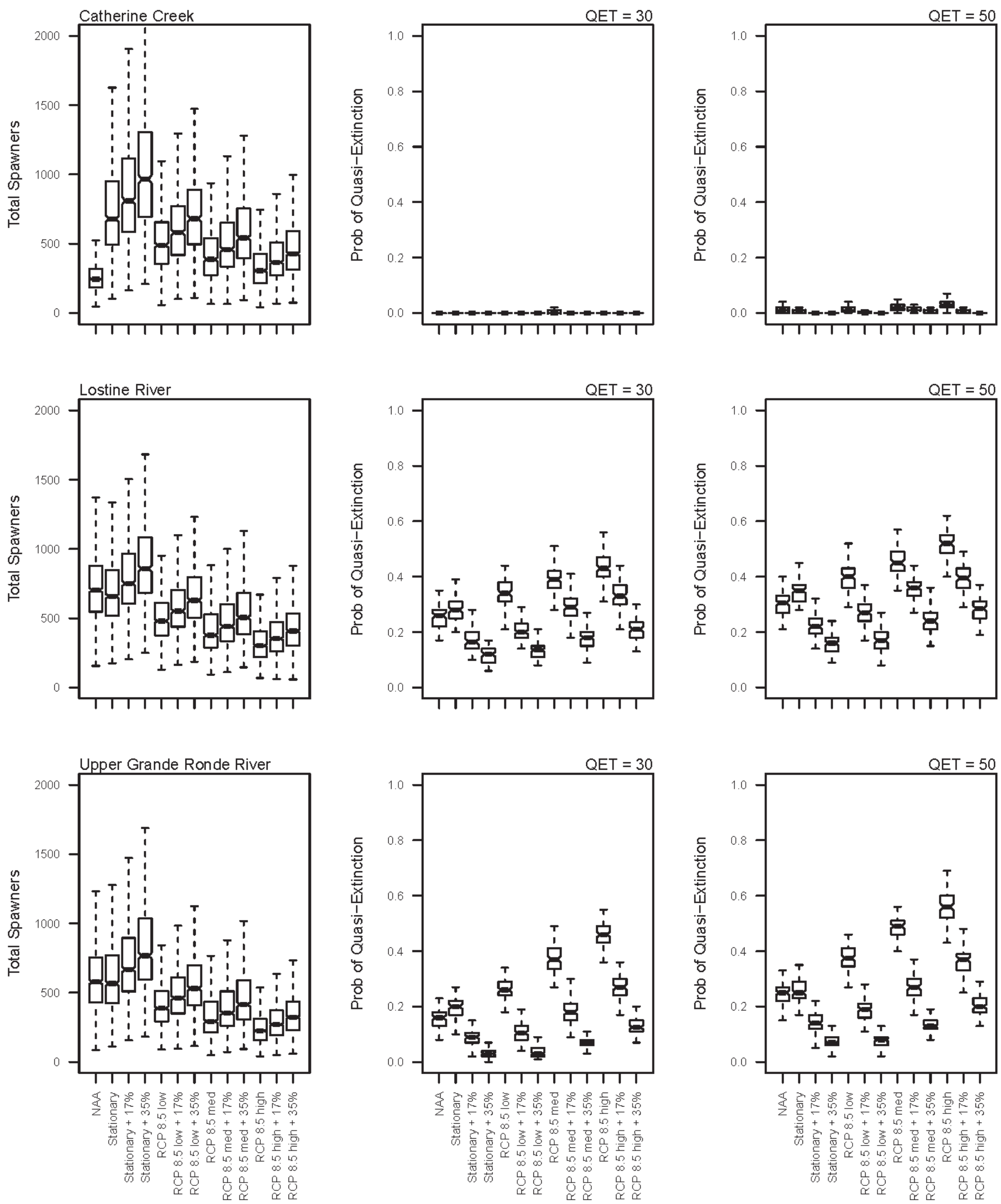

Figure 3-4. Estimates of total spawners, sum of hatchery- and natural-origin adults spawning in the wild, and the quasi-extinction risk (QET 30 and 50) for the three supplemented populations and the 13 scenarios used in the Grande Ronde simulation life cycle model.

Table 3-5. Estimates of total spawners (hatchery- and natural-origin spawners) for the three supplemented populations and 13 scenarios output from the Grande Ronde simulation life cycle model. The climate scenarios are *Stationary* and a range of global climate model (GCM) projections for the representative concentration pathway (*RCP 8.5*) emissions scenarios. *Low*, *mean*, and *high* reflect the 25th, 50th, and 75th quantiles across GCM time series. A range of percentiles (5%, 25%, 50%, 75%, 95%) is shown for abundance across simulations for each scenario in each population.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Catherine Creek					
NAA	120	184	245	320	461
Stationary	280	491	679	950	1,429
Stationary + 17%	366	584	810	1,115	1,746
Stationary + 35%	429	692	965	1,307	1,991
RCP 8.5 low	205	354	489	655	999
RCP 8.5 low + 17%	269	419	582	773	1,273
RCP 8.5 low + 35%	315	496	680	889	1,448
RCP 8.5 mean	170	274	388	542	890
RCP 8.5 mean + 17%	202	335	459	653	1,038
RCP 8.5 mean + 35%	239	396	543	753	1,192
RCP 8.5 high	130	216	306	428	649
RCP 8.5 high + 17%	158	270	366	509	775
RCP 8.5 high + 35%	189	312	428	593	935
Lostine River					
NAA	390	547	703	877	1,167
Stationary	355	519	658	846	1,181
Stationary +17%	415	605	751	965	1,328
Stationary + 35%	475	681	858	1,083	1,483
RCP 8.5 low	254	375	480	610	883
RCP 8.5 low +17%	309	439	552	704	1,017
RCP 8.5 low + 35%	355	505	629	795	1,148
RCP 8.5 mean	187	289	377	529	740
RCP 8.5 mean +17%	224	335	440	602	861
RCP 8.5 mean + 35%	261	384	505	684	962
RCP 8.5 high	139	222	302	404	595
RCP 8.5 high +17%	161	261	353	473	681
RCP 8.5 high + 35%	193	304	408	534	783
Grande Ronde River (um)[a]					
NAA	293	428	578	753	1,074
Stationary	265	426	567	768	1,164
Stationary +17%	317	510	668	895	1,333
Stationary + 35%	379	595	767	1,035	1,545
RCP 8.5 low	185	292	387	513	817
RCP 8.5 low +17%	229	351	462	612	927
RCP 8.5 low + 35%	270	409	530	699	1,059
RCP 8.5 mean	126	212	293	434	654
RCP 8.5 mean +17%	162	260	353	510	770
RCP 8.5 mean + 35%	194	305	415	592	885
RCP 8.5 high	94	158	226	310	496
RCP 8.5 high +17%	111	193	271	372	584
RCP 8.5 high + 35%	139	230	324	434	684

[a] *(um)* = upper mainstem

Table 3-6. Estimates of probability of quasi-extinction at Year 24 with a QET of 30 for the three supplemented populations and 13 scenarios output from the Grande Ronde simulation life cycle model. The climate scenarios are *Stationary* and a range of global climate model (GCM) projections for the representative concentration pathway (*RCP 8.5*) emissions scenarios. *Low*, *mean*, and *high* reflect the 25th, 50th, and 75th quantiles across GCM time series. A range of percentiles (5%, 25%, 50%, 75%, 95%) is shown for abundance across simulations for each scenario in each population.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Catherine Creek					
NAA	0.00	0.00	0.00	0.00	0.01
Stationary	0.00	0.00	0.00	0.00	0.01
Stationary + 17%	0.00	0.00	0.00	0.00	0.00
Stationary + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low	0.00	0.00	0.00	0.00	0.01
RCP 8.5 low + 17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean	0.00	0.00	0.00	0.01	0.01
RCP 8.5 mean + 17%	0.00	0.00	0.00	0.00	0.01
RCP 8.5 mean + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 high	0.00	0.00	0.00	0.00	0.01
RCP 8.5 high + 17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 high + 35%	0.00	0.00	0.00	0.00	0.00
Lostine River					
NAA	0.00	0.00	0.00	0.00	0.00
Stationary	0.00	0.00	0.00	0.00	0.00
Stationary +17%	0.00	0.00	0.00	0.00	0.00
Stationary + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low +17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean +17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 high	0.00	0.00	0.00	0.00	0.00
RCP 8.5 high +17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 high + 35%	0.00	0.00	0.00	0.00	0.00
Grande Ronde River (um)[a]					
NAA	0.00	0.00	0.00	0.00	0.00
Stationary	0.00	0.00	0.00	0.00	0.00
Stationary +17%	0.00	0.00	0.00	0.00	0.00
Stationary + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low +17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean +17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 high	0.00	0.00	0.00	0.00	0.00
RCP 8.5 high +17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 high + 35%	0.00	0.00	0.00	0.00	0.00

[a] *(um)* = upper mainstem

Table 3-7. Estimates of probability of quasi-extinction at Year 24 with a QET of 50 for the three supplemented populations and 13 scenarios output from the Grande Ronde simulation life cycle model. The climate scenarios are *Stationary* and a range of global climate model (GCM) projections for the representative concentration pathway (*RCP 8.5*) emissions scenarios. *Low*, *mean*, and *high* reflect the 25th, 50th, and 75th quantiles across GCM time series. A range of percentiles (5%, 25%, 50%, 75%, 95%) is shown for abundance across simulations for each scenario in each population.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Catherine Creek					
NAA	0.00	0.00	0.01	0.02	0.03
Stationary	0.00	0.00	0.01	0.01	0.02
Stationary + 17%	0.00	0.00	0.00	0.00	0.01
Stationary + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low	0.00	0.01	0.01	0.02	0.03
RCP 8.5 low + 17%	0.00	0.00	0.00	0.00	0.01
RCP 8.5 low + 35%	0.00	0.00	0.00	0.00	0.01
RCP 8.5 mean	0.00	0.01	0.02	0.03	0.04
RCP 8.5 mean + 17%	0.00	0.01	0.01	0.02	0.03
RCP 8.5 mean + 35%	0.00	0.00	0.01	0.01	0.02
RCP 8.5 high	0.01	0.02	0.03	0.04	0.06
RCP 8.5 high + 17%	0.00	0.00	0.01	0.01	0.03
RCP 8.5 high + 35%	0.00	0.00	0.00	0.00	0.01
Lostine River					
NAA	0.00	0.00	0.00	0.00	0.00
Stationary	0.00	0.00	0.00	0.00	0.00
Stationary +17%	0.00	0.00	0.00	0.00	0.00
Stationary + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low +17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean	0.00	0.00	0.00	0.00	0.01
RCP 8.5 mean +17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 high	0.00	0.00	0.01	0.01	0.02
RCP 8.5 high +17%	0.00	0.00	0.00	0.01	0.02
RCP 8.5 high + 35%	0.00	0.00	0.00	0.00	0.01
Grande Ronde River (um)[a]					
NAA	0.00	0.00	0.00	0.00	0.00
Stationary	0.00	0.00	0.00	0.00	0.00
Stationary +17%	0.00	0.00	0.00	0.00	0.00
Stationary + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low +17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean	0.00	0.00	0.00	0.00	0.01
RCP 8.5 mean +17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 high	0.00	0.01	0.02	0.03	0.04
RCP 8.5 high +17%	0.00	0.00	0.01	0.01	0.02
RCP 8.5 high + 35%	0.00	0.00	0.00	0.01	0.02

[a] *(um)* = upper mainstem

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 cross-correlation characteristics as the baseline timeframe. The Grande Ronde River upper mainstem 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. Thus, restoring riparian shading and natural channel form in this degraded reach is an example of a key action linked explicitly to declining marine survivals forecast with climate change.

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4 Upper Salmon River MPG Spring/Summer-run Chinook Salmon Life Cycle Models

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4.1 Introduction

For the Upper Salmon River MPG (USAL), we have developed life cycle models (LCMs) for eight of the nine populations (the Salmon River lower mainstem population was not modeled at this time due to a lack of habitat and population data). The life cycle models are stage-specific, with Beverton–Holt-based stage transitions covering the spawner-to-egg, egg-to-fry, fry-to-parr, parr-to-smolt, and smolt-to-spawner stages. All of the stage transitions are density-dependent, with the exception of the smolt-to-spawner component. The redd and juvenile rearing capacities are estimated as functions of stream habitat quality and quantity from Quantile Regression Forest models: 90th quantile regression based on a random forest model 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 Beverton–Holt function) estimates were generated for each population with adult or juvenile abundance time series.

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 2009–15 interval, 2) a projection of future actions (post-2018) based on random project locations, and 3) a projection of future actions (post-2021) based on applying the level of effort specified for USAL in the Proposed Action across three focal populations within the MPG.

Stream habitat restoration actions were estimated to impact carrying capacity for spawning, rearing, and juvenile stage transition, or survival. Since the basis of the freshwater habitat in these models is the reach type and geomorphic condition of the reaches, we only modeled in-stream complexity actions (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 one step.

The population-level outcomes of restoration alternatives were modeled by running population simulations for 48 years, replicated 1,000 times each. The performance metrics from these simulation sets were the median and quantiles of the size of natural-origin spawner population 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.

4.2 Background

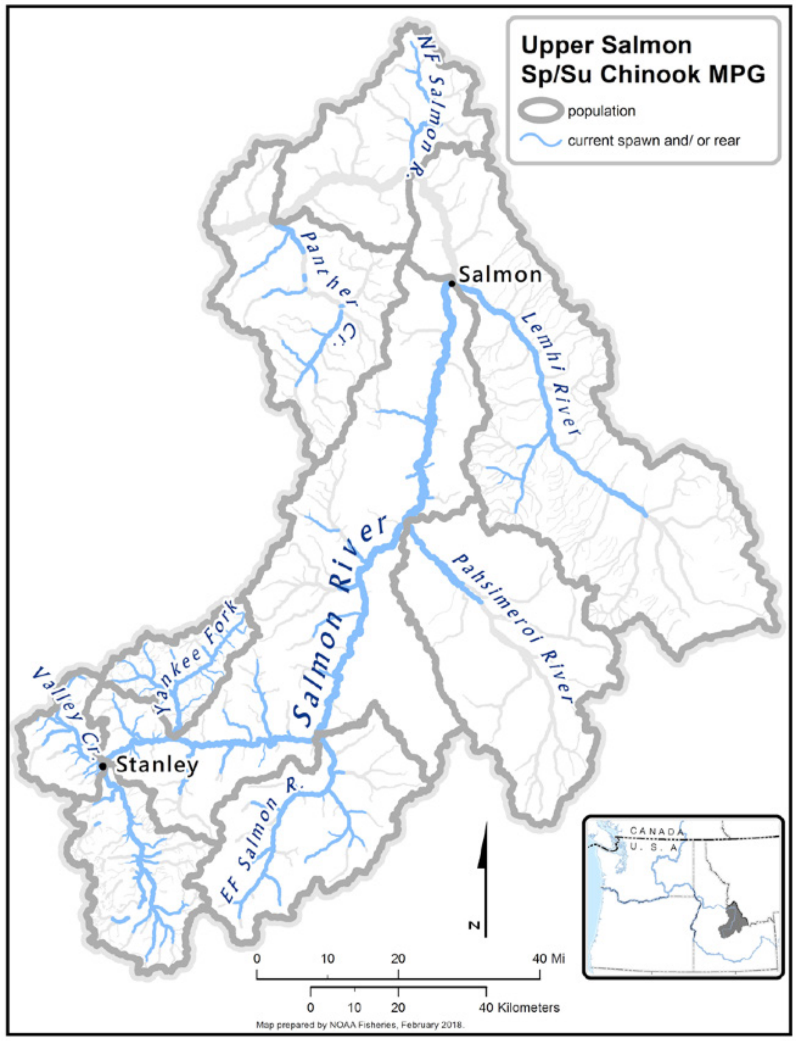

Figure 4-1. Spring/summer Chinook salmon populations in the Upper Salmon River MPG.

The Upper Salmon River MPG includes nine independent populations (Figure 4-1): East Fork Salmon River, Lemhi River, North Fork Salmon River, Pahsimeroi River, Panther Creek (extirpated), Salmon River lower mainstem (below Redfish Lake Creek), Salmon River upper mainstem (above Redfish Lake Creek), Valley Creek, and Yankee Fork. All four population size-classes, based on historic intrinsic production potential, are represented in the MPG.

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 Salmon River upper mainstem populations are included in integrated hatchery programs based on indigenous stocks. These two hatchery programs also operate a segregated production program. Currently, outplanting of eggs (Panther Creek) and adults (Yankee Fork) occurs when returns to the segregated program allow it. The East Fork Salmon River, Lemhi River, Yankee Fork, and Valley Creek populations have some history of hatchery supplementation with Upper Salmon River, local, and Rapid River stocks, but are considered to be persisting because of natural reproduction of the local stocks at present.

All extant populations in this MPG were at high risk for the integrated viability rating 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 and abundance levels being 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; they could therefore achieve viable status if A/P risk were 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 Salmon River upper mainstem population has the highest relative abundance and 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 River 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 East Fork Salmon River, Pahsimeroi River, and Yankee Fork 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-run Chinook Salmon ESU, long-term trend and population growth rate estimates have been less than 1 for all natural production datasets, 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. 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 MPG populations.

4.3 Upper Salmon River 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 structure and its implementation in the Upper Salmon River MPG has been described fully in a previous NOAA technical memorandum (Pess and Jordan 2019); a simplified description is included below.

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.

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. For the simulations presented here, the habitat action benefits were implemented as three-step increases in habitat quality (complexity) and quantity (access), each separated by five years.

4.3.1 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). Sequential, or stage-based, Beverton–Holt relationships are applied to represent the potential nonlinear (density-dependent) transition probabilities.

Productivity at each stage transition, for each potential site in the model, is a function of the density-independent productivity for each stage, dependent on the relative importance/relationship between productivity and reach type. The function is a simple product of a scalar governing the relative value of habitat condition, a scalar governing the relative value of reach type, and the 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 for each life stage in each site is modeled as the maximum density-based habitat quality and quantity. Each population is represented in the model as an area-based measure of each reach type present (reach length × bankfull width). Each reach type/geomorphic condition combination has an underlying expected life stage-specific density. Thus, the overall capacity for each life stage Beverton–Holt relationship is an area-weighted sum of the density habitat type/condition product for each stage and site.

4.3.2 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.

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, measurements collected during pro forma evaluations (including measured cross-sections and inventories 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. 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 MPG.

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 presettlement 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 MPG will be improved with additional on-the-ground validation across watersheds of the Upper Salmon River, but also, more broadly across the interior Columbia River basin.

4.3.3 Habitat capacity estimation

To estimate life stage-specific habitat capacity for spring Chinook salmon (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 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 nonlinear 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 2018) and were obtained from the CHaMP website.[1] 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 watersheds were used to develop the QRF models: Entiat, Upper Grande Ronde, Minam, John Day, Lemhi, Methow, 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.

4.3.4 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 (PDPs). 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 are 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 2016). 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.

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

4.3.4.1 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 2011). 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.

4.3.4.2 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 2011).

4.3.4.3 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. Using the SPCH redd capacity QRF model, predictions of redd capacity were made for CHaMP sites within the Lemhi River.

4.3.5 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 or 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 and Scott 2017) in R software (R Core Team 2016).

To summarize capacity at larger spatial scales, we first determine 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 StreamNet[2]) are used. Mean estimates within that scale are then multiplied by the length of the stream within the SPCH domain.

4.3.6 Parsing QRF capacity estimates by reach type and geomorphic condition

Parr and redd capacity estimates were generated by the QRF modeling approach for the entire spring/summer Chinook salmon spawning and rearing network within the Upper Salmon River MPG. All reaches of the stream network in USAL have been classified into reach type (RT) and geomorphic condition (GC). 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 salmon populations modeled, stage-specific survival data were 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 will 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—the 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 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* GC reaches in the USAL domain were calculated. The capacity for each RT in the *good* GC state forms the baseline for the value to parr and 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 datasets, so can the value of "*functioning*," "*moderate*," and "*poor*" states. Using the entire USAL dataset 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*. 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.

[2] http://www.streamnet.org

4.4 Incorporating Tributary Habitat Restoration Actions

4.4.1 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. State, tribal, federal, 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 decisionmakers, we created a standardized data dictionary of project types now being applied throughout the region and assembled project records into a database of restoration actions—the Pacific Northwest Salmon Habitat Project Database (PNSHP; Barnas et al. 2015, NMFS 2019). 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, including 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 (2020) contains spatially referenced, project-level data on over 43,000 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, state, local, NGO, and tribal contributors.

For the Snake River spring/summer-run Chinook Salmon 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 datasets.

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, to use this rich data source as a driver of fish habitat condition change in the Upper Salmon River 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. Second, 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 by a single GC step.

4.4.2 Incorporating future actions in key Upper Salmon River Chinook salmon populations

For Chinook populations in the Upper Salmon River MPG, the potential benefit of future tributary actions was estimated based on distributing a similar level of effort during the recent past (2009–15) and the level of effort assumed from the Proposed Action of NMFS (2019) at the MPG level to three focal populations: Lemhi River, Pahsimeroi River, and Salmon River upper mainstem. The Upper Salmon River simulation 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 (e.g., in-stream complexity or floodplain reconnection) can be directly modeled. The forecast level of habitat restoration action at the scale of the entire Upper Salmon River MPG in the Proposed Action of NMFS (2020) was 30.4 km of stream complexity improvement and 85.5 km of habitat access. These levels of effort were distributed across the focal populations evenly. The habitat access effort was split three ways, but the habitat complexity improvement only two ways, as the current habitat status in the Pahsimeroi is of sufficiently high quality that additional in-stream work is not warranted (Table 4-1). As the term of NMFS (2020) is 15 years, the potential benefit of the proposed habitat actions was modeled as being applied in three increments five years apart. Each increment was ⅓ of the potential action, with ⅓ of the aggregate potential benefit. The benefit was assumed to accrue immediately, and the actions applied at successive time periods were additive.

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

Table 4-1. NMFS (2020) Proposed Action habitat actions in the Upper Salmon River MPG. Actions are proposed of multiple types, but only “Habitat Access” and “In-stream Complexity” are incorporated into the life cycle modeling. Each action type is divided equally across the target population watersheds and three implementation cycles of five years each.

Habitat Actions	Lemhi River	Pahsimeroi River	Upper Mainstem Salmon River
Proposed Action (PA) In-stream Complexity, total (15 yr)	15.2 km	0.0 km	15.2 km
PA In-stream Complexity per interval (5 yr)	5.0 km	0.0 km	5.0 km
PA In-stream Complexity area, total (15 yr)	167,200.0 m^2	0.0 m^2	69,920.0 m^2
PA In-stream Complexity area per interval (5 yr)	55,176.0 m^2	0.0 m^2	23,074.0 m^2
Current spawn/rear area	1,500,711.0 m^2	594,315.0 m^2	730,453.0 m^2
Relative increase in area per interval (5 yr)	3.7%	0.0%	3.2%
Proposed Action (PA) Habitat Access, total (15 yr)	28.5 km	28.5 km	28.5 km
PA Habitat Access per interval (5 yr)	9.4 km	9.4 km	9.4 km
PA Habitat Access area, total (15 yr)	103,499.0 m^2	119,494.0 m^2	43,281.0 m^2
PA Habitat Access area per interval (5 yr)	2.3%	6.6%	2.0%
Estimated increase in redd capacity per interval (5 yr)	2.6%	6.6%	2.3%
Estimated increase in rearing capacity per interval (5 yr)	4.1%	6.6%	3.5%

access projects were assumed to open habitat of representative quality, that is, additional 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, 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 habitat quantity was added first 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.

4.4.3 Estimating population-level benefit of tributary habitat restoration actions

Across Chinook salmon 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 salmon population watersheds in the Upper Salmon River MPG; however, significant habitat degradation due to mining activity in North Fork Salmon River, Panther Creek, and Yankee Fork dramatically reduces potential high-quality spawning areas, and extensive dewatering due to irrigation withdrawals in the Lemhi and Pahsimeroi Rivers 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 one 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. 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 salmon 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.

4.5 Life Cycle Model Scenarios

4.5.1 Hydrosystem survival

The Upper Salmon River simulation LCM includes common mainstem and ocean survival modules shared by the Grande Ronde (Cooney et al., this volume, Chapter 3) and Snake River models. Models for survival in these stages are described in Crozier et al. (this volume, Chapter 5), Chasco et al. (submitted), and Crozier et al. (submitted b). The three life stages covered by the common mainstem/ocean module are: downstream migration through the hydropower system, Bonneville-to-Bonneville smolt-to-adult return rates (SARs), and upstream migration.

The included hydrosystem operations scenarios are described in Faulkner et al. (this volume, Chapter 2). In brief, flow, spill, reservoir elevation, water temperature, and dissolved gas for the Proposed Action were all modeled by the U.S. Army Corps of Engineers (USACE). These models assume turbine replacements at Ice Harbor, McNary, and John Day Dams that have substantially lower fish mortality than the existing turbines.

These runs used a universal transportation start date of 20 April at all three transporter dams: Lower Granite, Little Goose, and Lower Monumental Dams. After this date, all fish predicted to enter the bypass system at these dams were treated as transported fish by the COMPASS model; they are removed from the river at the transport dam, and added to the tailrace of Bonneville Dam two days later. COMPASS assumes uniform 0.98 survival during transportation. In each simulation, the COMPASS model produced distributions of arrival times for in-river and transported smolts at Bonneville Dam, which were then input into the SAR model.

Chasco et al. (submitted) used a mixed-effects logistic regression model for wild fish to determine the effect of the date of ocean entry (from COMPASS) and environmental covariates (specified by the climate scenario) on the probability that an individual fish would return as an adult to Bonneville Dam. The model includes random effects for day and for the day by year interaction, which follow an autoregressive process over time. We developed separate models for fish that had migrated through the mainstem in the river and for fish that had been transported downstream in barges. The downstream survival models were structured as follows: the model for transported fish included only a single covariate, a large-scale measure of sea surface temperature (SST), and a model for in-river migrants that included two covariates, *SSTarc* in winter and a more local measure of SST along the Washington coast.

To assess the implications of speculative reductions in delayed mortality for in-river migrating fish, a multiplicative factor was include to increase aggregate hydrosystem survival. In these latent mortality (LM) scenarios, we increased marine survival rates of in-river migrants by 17% and 35%.

To account for adult upstream survival, Crozier et al. (submitted b) used generalized additive mixed models (GAMMs) to evaluate the effects of both anthropogenic and environmental covariates on spring/summer Chinook salmon survival. To develop simulation output, all of the nonenvironmental covariates (fisheries catch, the proportion of fish that had been transported in barges as juveniles, etc.) had similar distributions to the baseline period (2004–16), and survival from the hydrosystem to spawning was treated as constant due to the lack of appropriate data for most populations with which to fit a relationship.

4.5.2 Climate scenarios

To compare population trajectories in a climate experiencing historical levels of variability but no directional trends (a "stationary" climate), with population trajectories in a climate responding to anthropogenic greenhouse gases, Crozier et al. (submitted a) generated a suite of simulated scenarios. They created a stationary climate scenario by fitting a covariance matrix for all the freshwater and marine environmental covariates used in the life cycle model. The covariance matrix was then incorporated into a multivariate state-space model that accounts for temporal correlations across environmental variables. Autoregression was further incorporated into the random effects within the SAR model to account for additional temporal patterns that were not captured in the raw environmental time series included in the selected covariates. The state-space model was used to simulate natural variability in all covariates in a stationary climate.

4.5.3 Climate trends

To represent changing climate scenarios, Crozier et al. (submitted a) added trends to the stationary simulations. To simulate a temporal signal in these scenarios, they extracted trends from global climate model (GCM) projections of representative concentration pathway (RCP) 8.5, the "business-as-usual" scenario. To extract the relevant trends from GCMs, Crozier et al. (submitted a) calculated the variable mean over a baseline period centered on 2015 (2005–25), created monthly anomalies from the 2005–25 period for each time series, created a 20-year running mean of anomalies for each time series, and calculated the 25th, 50th, and 75th quantiles for each month from the smoothed anomalies across all of the time series. These roughly linear trends represent the spread across climate models of low, medium, and high rates of change in each covariate.

4.5.4 Tributary habitat condition

Given the lack of consistent and comprehensive habitat status information for all Chinook salmon population watersheds in the Upper Salmon River MPG, building spatially explicit life cycle models requires a series of assumptions and compromises. In this case, as was outlined in the methods above, habitat quality and quantity for the entire MPG were represented through the application of river styles based on reach typing and geomorphic condition assessments. These reach classifications were based on detailed on-the-ground surveys in two watersheds (Lemhi River 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 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 salmon populations: baseline condition, recent past restoration actions, random habitat quality improvements, and more directed habitat quality and quantity modifications. Note that the Salmon River lower mainstem population was not modeled due to data quality issues—there is a mismatch between the habitat condition and fish capacity estimations having been developed from wadeable stream reaches only, while the Salmon River lower mainstem is 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.

4.5.5 Baseline habitat condition

The population-specific life cycle models for Upper Salmon River Chinook salmon 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 Rivers 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" dataset 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 is 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.

4.5.6 Model scenarios and output metrics

In total, 13 scenarios were run for each population: a no-action alternative (NAA) and four climate scenarios (stationary, low, medium, and high), each with a range of latent mortality reduction (1.0, 1.17, and 1.35) to test the potential benefit of mainstem hydrosytem operations. We applied the same pattern of freshwater habitat action benefit to all scenarios.

We report model output across the scenarios with two performance metrics: abundance and risk of quasi-extinction. Abundance is calculated as the geometric mean of spawner abundance over Years 15 to 24 and reported as the 5th, 25th, 50th, 75th, and 95th percentiles of the distribution of abundances across the 1,000 replicate simulations run for each scenario. Risk of quasi-extinction is calculated as the fraction of replicate simulation abundance levels that fall to either 30 or 50 spawners for four consecutive years by Year 24 (pQET). As with the abundance metric, the quasi-extinction risk is also reported as the distribution of the resulting estimates of pQET. To generate a distribution of pQET, we constructed 100 bootstrap samples (sampling with replacement) of 100 replicate simulations from the 1,000 simulations for each scenario. We estimated pQET for each of the 100 samples of 100 replicates and report the 5th, 25th, 50th, 75th, and 95th percentiles of the pQET distributions.

4.6 Results

As part of the Proposed Action for NMFS (2020) on the operation and maintenance of the Federal Columbia River Hydropower System, the Action Agencies (USACE, USBOR, and BPA) have suggested that the tributary habitat restoration action effort will be similar to that in 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 MPG, the proposed actions focused on in-stream complexity and habitat access actions in a subset of the possible populations (Lemhi River, Pahsimeroi River, and the Salmon River upper mainstem). The forecast level of habitat restoration action at the scale of the entire Upper Salmon River MPG was 30.4 km of stream complexity improvement and 85.5 km of habitat access. These levels of effort were distributed across the focal populations evenly. The habitat access effort was split three ways, but the habitat complexity improvement only two ways, as the current habitat status in Pahsimeroi River is of sufficiently high quality that additional in-stream work is not warranted (Table 4-1). The estimated benefit of these restoration actions was large relative to the NAA scenario, with a 15% (Lemhi River), 32% (Salmon River upper mainstem), and 113% (Pahsimeroi River) increase in median spawner abundance (Table 4-2, Figure 4-2). The resulting decrease in quasi-extinction risk (defined as the probability that the population falls below 50 spawners for four successive years in the next 24 years) was also large (60% in Pahsimeroi River). 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 (Tables 4-3, 4-4, Figure 4-2). All sets of scenarios across all populations respond positively to latent mortality reductions (17% and 35% increases in survival), but are negatively impacted by the estimated downward trend in ocean survival due to climate change.

Hatchery returns in the Upper Salmon River MPG result from two facilities, each with two separate programs. Both the Pasimeroi and the Sawtooth hatcheries have out-of-basin-origin production, or segregation programs, as well as integrated conservation programs. The life cycle models consider both programs, tracking natural-origin smolts separately from both the integrated and segregated production programs. To be consistent with NMFS (2020), only natural-origin and total spawners (natural- and hatchery-origin adults spawning in the wild) are reported here (Figure 4-3, Tables 4-5, 4-6, 4-7).

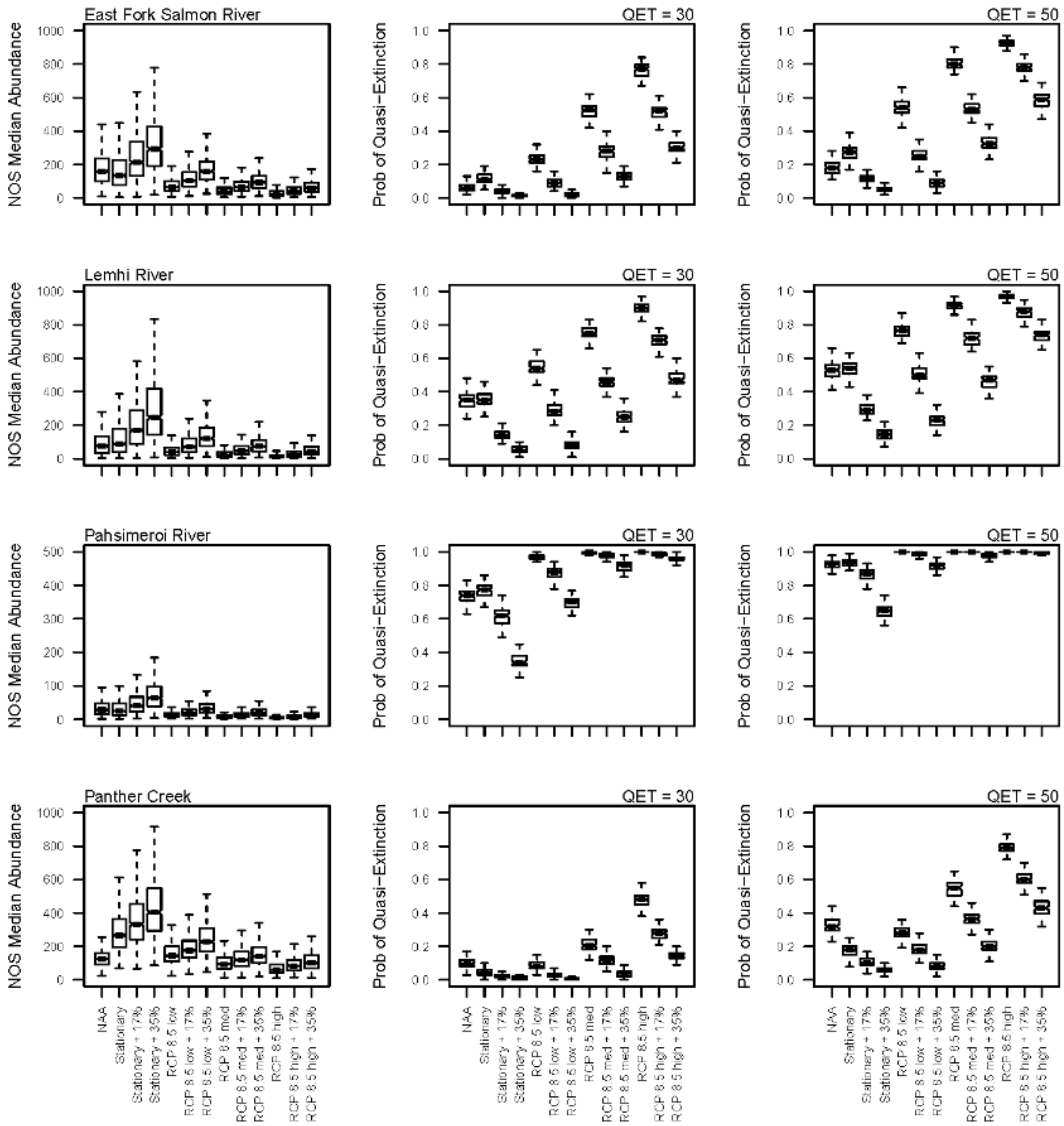

Figure 4-2. Geometric mean natural-origin spawner abundance as a function of simulation scenario for Upper Salmon River MPG populations. The box–whisker plots show the distribution of the geometric means (left) and the probability of quasi-extinction (pQET) with a threshold of 30 (center) and 50 (right) across the 1,000 replicate simulations. The geometric mean is calculated over simulation Years 15 to 24, while pQET is calculated over simulations Years 1 to 24. In the box and whisker plots, the dark bar is the median value, the box shows the inner quartiles (25th to 75th percentile), and the whiskers mark the 2.5th and 97.5th.

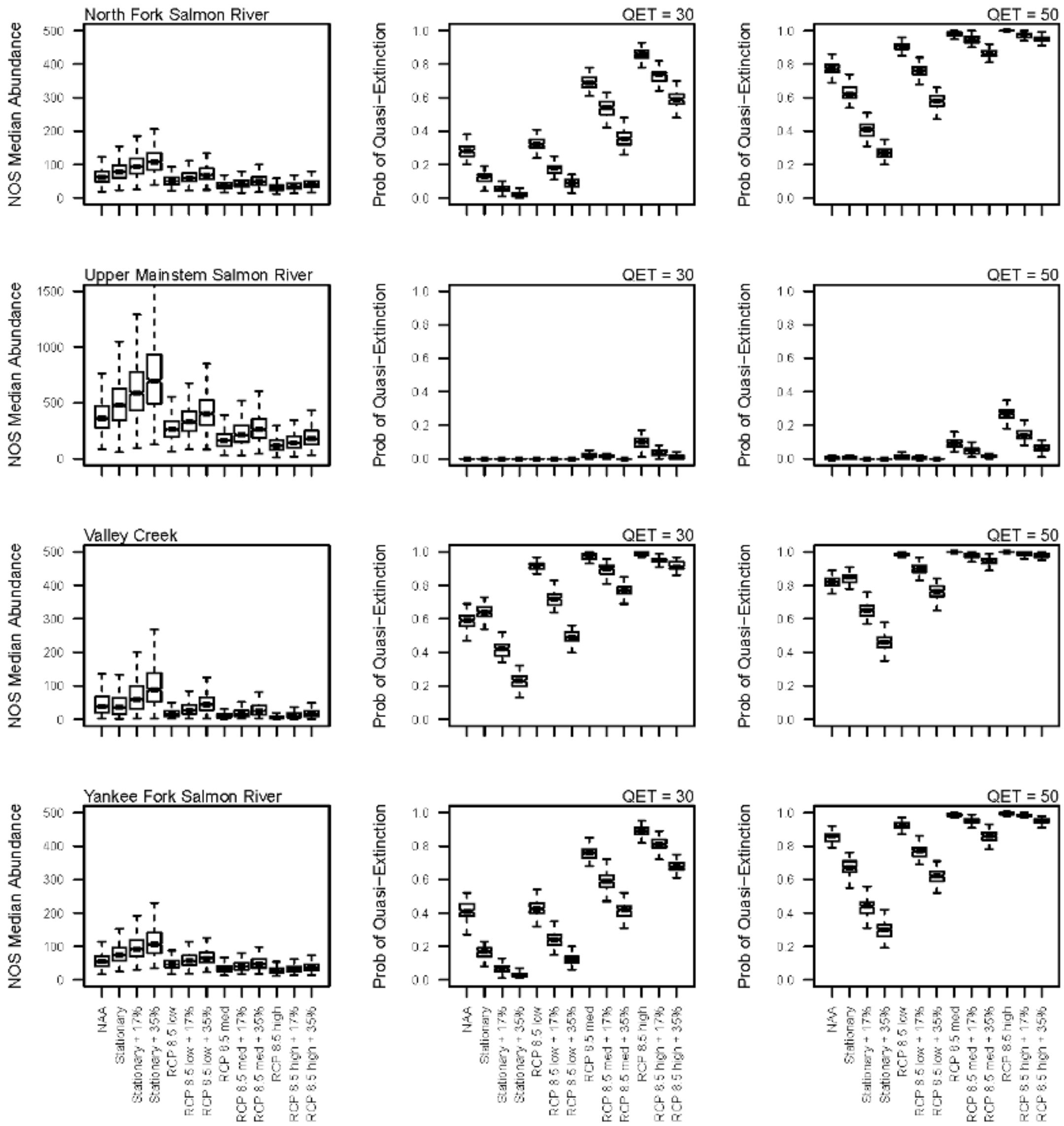

Figure 4-2 (continued). Geometric mean natural-origin spawner abundance as a function of simulation scenario for Upper Salmon River MPG populations.

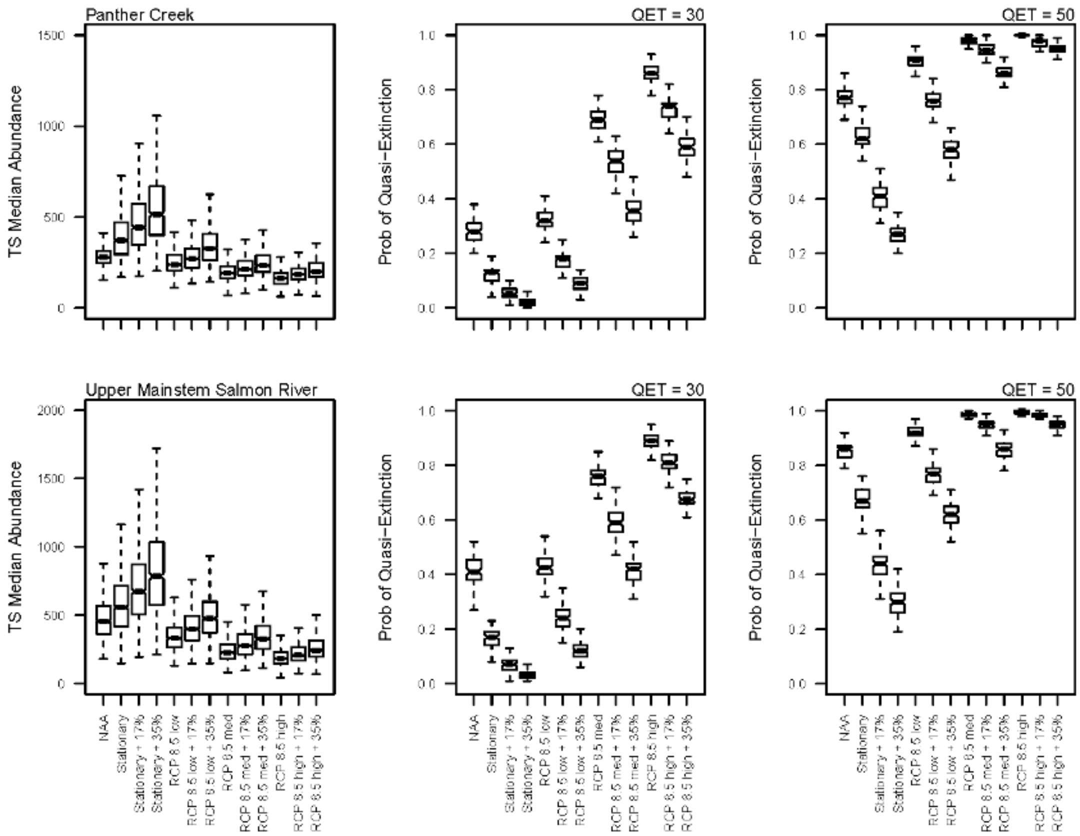

Figure 4-3. Geometric mean total spawner (natural-and hatchery-origin adults spawning in the wild) abundance as a function of simulation scenario for Upper Salmon River MPG populations. The box–whisker plots show the distribution of the geometric means (left) and the probability of quasi-extinction (pQET) with a threshold of 30 (center) and 50 (right) across the 1,000 replicate simulations. The geometric mean is calculated over simulation Years 15 to 24, while pQET is calculated over simulations Years 1 to 24. In the box–whisker plots, the dark bar is the median value, the box shows the inner quartiles (25th to 75th percentile), and the whiskers mark the 2.5th and 97.5th.

Table 4-2. Geometric mean natural-origin spawner abundance as a function of simulation scenario. Geometric mean is calculated over simulation Years 15 to 24, and the percentiles represent the distribution of geometric mean abundance metrics across the 1,000 replicate simulations. The climate scenarios are *Stationary* and a range of global climate model (GCM) projections for the representative concentration pathway (*RCP 8.5*) emissions scenarios. *Low*, *mean*, and *high* reflect the 25th, 50th, and 75th quantiles across GCM time series. A range of percentiles (5%, 25%, 50%, 75%, 95%) is shown for abundance across simulations for each scenario in each population.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
East Fork Salmon River					
NAA	39	101	158	237	386
Stationary	32	80	135	228	400
Stationary + 17%	56	132	213	335	537
Stationary + 35%	94	190	292	426	676
RCP 8.5 low	21	43	65	102	179
RCP 8.5 low + 17%	35	70	104	153	254
RCP 8.5 low + 35%	57	109	159	220	363
RCP 8.5 mean	11	24	39	62	111
RCP 8.5 mean + 17%	21	40	65	97	169
RCP 8.5 mean + 35%	32	60	94	133	226
RCP 8.5 high	7	14	24	40	78
RCP 8.5 high + 17%	12	26	41	65	122
RCP 8.5 high + 35%	20	39	61	93	168
Lemhi River					
NAA	9	35	76	134	247
Stationary	9	40	87	180	377
Stationary +17%	29	88	170	286	535
Stationary + 35%	57	142	246	419	746
RCP 8.5 low	8	22	39	68	141
RCP 8.5 low +17%	16	40	71	121	224
RCP 8.5 low + 35%	37	77	120	185	326
RCP 8.5 mean	5	12	22	40	87
RCP 8.5 mean +17%	10	25	45	74	128
RCP 8.5 mean + 35%	18	42	74	113	213
RCP 8.5 high	3	7	13	23	55
RCP 8.5 high +17%	6	14	26	45	102
RCP 8.5 high + 35%	11	24	42	70	135
North Fork Salmon River					
NAA	5	16	28	48	85
Stationary	4	12	25	47	92
Stationary +17%	8	23	41	67	113
Stationary + 35%	16	39	64	98	167
RCP 8.5 low	4	7	12	19	36
RCP 8.5 low +17%	6	13	20	30	57
RCP 8.5 low + 35%	10	20	31	46	83
RCP 8.5 mean	2	4	7	11	22
RCP 8.5 mean +17%	4	8	12	19	34
RCP 8.5 mean + 35%	6	12	19	29	54
RCP 8.5 high	2	3	4	7	16
RCP 8.5 high +17%	2	5	8	12	25
RCP 8.5 high + 35%	4	8	12	19	39

Table 4-2 (continued). Geometric mean natural-origin spawner abundance as a function of simulation scenario.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Pahsimeroi River					
NAA	52	93	125	159	228
Stationary	119	197	266	365	560
Stationary + 17%	149	243	332	456	692
Stationary + 35%	191	294	404	548	819
RCP 8.5 low	66	109	144	199	290
RCP 8.5 low + 17%	77	132	176	235	360
RCP 8.5 low + 35%	109	170	228	310	481
RCP 8.5 mean	36	67	94	134	212
RCP 8.5 mean + 17%	45	82	120	167	267
RCP 8.5 mean + 35%	62	103	141	198	315
RCP 8.5 high	22	41	61	92	157
RCP 8.5 high + 17%	31	56	84	119	192
RCP 8.5 high + 35%	40	71	102	147	245
Panther Creek					
NAA	34	49	62	79	111
Stationary	42	61	78	98	136
Stationary +17%	51	73	94	118	176
Stationary + 35%	61	84	108	134	197
RCP 8.5 low	30	40	50	62	85
RCP 8.5 low +17%	34	49	59	74	98
RCP 8.5 low + 35%	40	56	69	88	121
RCP 8.5 mean	22	30	37	46	66
RCP 8.5 mean +17%	25	34	42	53	73
RCP 8.5 mean + 35%	28	39	49	63	88
RCP 8.5 high	18	24	31	38	54
RCP 8.5 high +17%	21	28	35	44	67
RCP 8.5 high + 35%	24	31	40	51	73
Salmon River upper mainstem					
NAA	166	277	361	472	674
Stationary	215	343	478	629	919
Stationary +17%	262	432	587	777	1,148
Stationary + 35%	319	494	697	932	1,344
RCP 8.5 low	118	194	262	339	498
RCP 8.5 low +17%	157	248	330	425	622
RCP 8.5 low + 35%	182	303	402	524	785
RCP 8.5 mean	69	112	163	224	351
RCP 8.5 mean +17%	85	151	212	298	461
RCP 8.5 mean + 35%	109	190	262	356	558
RCP 8.5 high	41	78	116	165	282
RCP 8.5 high +17%	56	99	140	200	353
RCP 8.5 high + 35%	77	129	179	252	413

Table 4-2 (continued). Geometric mean natural-origin spawner abundance as a function of simulation scenario.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Valley Creek					
NAA	7	20	38	67	127
Stationary	5	16	35	63	135
Stationary + 17%	11	31	59	99	198
Stationary + 35%	21	51	88	138	250
RCP 8.5 low	4	9	15	26	50
RCP 8.5 low + 17%	7	17	27	44	84
RCP 8.5 low + 35%	13	28	44	67	118
RCP 8.5 mean	2	6	9	16	32
RCP 8.5 mean + 17%	4	10	16	26	52
RCP 8.5 mean + 35%	7	14	25	41	74
RCP 8.5 high	2	3	5	10	23
RCP 8.5 high + 17%	3	6	10	18	37
RCP 8.5 high + 35%	4	9	15	25	51
Yankee Fork					
NAA	28	42	55	71	111
Stationary	40	57	74	97	142
Stationary +17%	47	71	92	119	178
Stationary + 35%	57	80	106	140	211
RCP 8.5 low	26	38	46	58	79
RCP 8.5 low +17%	31	45	57	73	100
RCP 8.5 low + 35%	37	52	65	82	122
RCP 8.5 mean	20	26	34	42	61
RCP 8.5 mean +17%	23	31	40	51	76
RCP 8.5 mean + 35%	26	36	47	61	89
RCP 8.5 high	16	22	27	35	51
RCP 8.5 high +17%	19	25	32	40	61
RCP 8.5 high + 35%	21	29	36	47	74

Table 4-3. Probability of quasi-extinction across all eight populations and the 13 model scenarios, with a quasi-extinction threshold of 30. The quasi-extinction probability for the natural-origin spawners was calculated over simulation Years 1 to 24. The climate scenarios are *Stationary* and a range of GCM projections for the *RCP 8.5* emissions scenarios. *Low*, *mean*, and *high* reflect the 25th, 50th, and 75th quantiles across GCM time series. A range of percentiles (5%, 25%, 50%, 75%, 95%) is shown for abundance across simulations for each scenario in each population.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
East Fork Salmon River					
NAA	0.03	0.05	0.06	0.08	0.11
Stationary	0.07	0.10	0.11	0.14	0.17
Stationary + 17%	0.01	0.03	0.04	0.05	0.07
Stationary + 35%	0.00	0.01	0.01	0.02	0.03
RCP 8.5 low	0.18	0.21	0.23	0.25	0.28
RCP 8.5 low + 17%	0.04	0.07	0.09	0.11	0.13
RCP 8.5 low + 35%	0.00	0.01	0.02	0.03	0.04
RCP 8.5 mean	0.46	0.49	0.53	0.55	0.60
RCP 8.5 mean + 17%	0.21	0.25	0.28	0.31	0.34
RCP 8.5 mean + 35%	0.09	0.11	0.13	0.15	0.18
RCP 8.5 high	0.69	0.73	0.77	0.79	0.83
RCP 8.5 high + 17%	0.45	0.49	0.52	0.54	0.60
RCP 8.5 high + 35%	0.23	0.28	0.30	0.33	0.37
Lemhi River					
NAA	0.26	0.31	0.35	0.38	0.42
Stationary	0.29	0.33	0.35	0.38	0.42
Stationary +17%	0.10	0.13	0.14	0.16	0.20
Stationary + 35%	0.02	0.04	0.05	0.07	0.09
RCP 8.5 low	0.47	0.52	0.54	0.58	0.63
RCP 8.5 low +17%	0.22	0.26	0.29	0.32	0.38
RCP 8.5 low + 35%	0.04	0.06	0.08	0.10	0.13
RCP 8.5 mean	0.69	0.73	0.75	0.78	0.81
RCP 8.5 mean +17%	0.39	0.44	0.46	0.48	0.53
RCP 8.5 mean + 35%	0.19	0.22	0.25	0.28	0.33
RCP 8.5 high	0.86	0.88	0.90	0.92	0.95
RCP 8.5 high +17%	0.64	0.68	0.71	0.73	0.77
RCP 8.5 high + 35%	0.42	0.45	0.47	0.51	0.57
North Fork Salmon River					
NAA	0.66	0.71	0.74	0.76	0.81
Stationary	0.71	0.74	0.78	0.80	0.83
Stationary +17%	0.54	0.58	0.62	0.65	0.68
Stationary + 35%	0.27	0.32	0.34	0.38	0.43
RCP 8.5 low	0.95	0.96	0.97	0.98	0.99
RCP 8.5 low +17%	0.81	0.85	0.88	0.90	0.92
RCP 8.5 low + 35%	0.63	0.67	0.70	0.72	0.75
RCP 8.5 mean	0.98	0.99	1.00	1.00	1.00
RCP 8.5 mean +17%	0.95	0.97	0.98	0.99	1.00
RCP 8.5 mean + 35%	0.87	0.89	0.92	0.93	0.96
RCP 8.5 high	0.99	1.00	1.00	1.00	1.00
RCP 8.5 high +17%	0.98	0.98	0.99	0.99	1.00
RCP 8.5 high + 35%	0.92	0.95	0.96	0.97	0.99

Table 4-3 (continued). Probability of quasi-extinction across all eight populations and the 13 model scenarios, with a quasi-extinction threshold of 30.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Pahsimeroi River					
NAA	0.05	0.08	0.10	0.12	0.15
Stationary	0.01	0.03	0.04	0.06	0.08
Stationary + 17%	0.00	0.02	0.02	0.03	0.05
Stationary + 35%	0.00	0.00	0.01	0.02	0.03
RCP 8.5 low	0.05	0.07	0.08	0.10	0.12
RCP 8.5 low + 17%	0.01	0.02	0.03	0.04	0.06
RCP 8.5 low + 35%	0.00	0.00	0.01	0.01	0.03
RCP 8.5 mean	0.15	0.19	0.21	0.24	0.28
RCP 8.5 mean + 17%	0.07	0.10	0.12	0.14	0.18
RCP 8.5 mean + 35%	0.01	0.02	0.04	0.05	0.07
RCP 8.5 high	0.40	0.45	0.48	0.50	0.55
RCP 8.5 high + 17%	0.22	0.25	0.28	0.30	0.34
RCP 8.5 high + 35%	0.10	0.13	0.15	0.16	0.20
Panther Creek					
NAA	0.21	0.25	0.28	0.31	0.35
Stationary	0.07	0.10	0.13	0.14	0.18
Stationary +17%	0.02	0.04	0.05	0.07	0.10
Stationary + 35%	0.00	0.01	0.02	0.03	0.05
RCP 8.5 low	0.26	0.30	0.32	0.35	0.39
RCP 8.5 low +17%	0.12	0.15	0.18	0.19	0.24
RCP 8.5 low + 35%	0.04	0.07	0.09	0.11	0.14
RCP 8.5 mean	0.63	0.66	0.69	0.72	0.76
RCP 8.5 mean +17%	0.45	0.50	0.54	0.57	0.61
RCP 8.5 mean + 35%	0.30	0.32	0.36	0.39	0.43
RCP 8.5 high	0.80	0.84	0.86	0.88	0.91
RCP 8.5 high +17%	0.66	0.70	0.74	0.75	0.79
RCP 8.5 high + 35%	0.51	0.56	0.59	0.62	0.67
Salmon River upper mainstem					
NAA	0.00	0.00	0.00	0.00	0.00
Stationary	0.00	0.00	0.00	0.00	0.01
Stationary +17%	0.00	0.00	0.00	0.00	0.00
Stationary + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low +17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean	0.00	0.01	0.02	0.03	0.04
RCP 8.5 mean +17%	0.00	0.01	0.02	0.02	0.04
RCP 8.5 mean + 35%	0.00	0.00	0.00	0.00	0.01
RCP 8.5 high	0.05	0.07	0.10	0.12	0.14
RCP 8.5 high +17%	0.01	0.02	0.03	0.05	0.06
RCP 8.5 high + 35%	0.00	0.00	0.01	0.02	0.03

Table 4-3 (continued). Probability of quasi-extinction across all eight populations and the 13 model scenarios, with a quasi-extinction threshold of 30.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Valley Creek					
NAA	0.51	0.56	0.59	0.62	0.66
Stationary	0.57	0.62	0.64	0.67	0.71
Stationary + 17%	0.35	0.38	0.42	0.45	0.48
Stationary + 35%	0.17	0.20	0.23	0.26	0.29
RCP 8.5 low	0.87	0.90	0.92	0.93	0.95
RCP 8.5 low + 17%	0.66	0.69	0.72	0.74	0.79
RCP 8.5 low + 35%	0.43	0.47	0.49	0.52	0.55
RCP 8.5 mean	0.94	0.96	0.98	0.99	1.00
RCP 8.5 mean + 17%	0.85	0.87	0.90	0.92	0.94
RCP 8.5 mean + 35%	0.71	0.75	0.77	0.79	0.84
RCP 8.5 high	0.97	0.98	0.99	1.00	1.00
RCP 8.5 high + 17%	0.92	0.94	0.95	0.96	0.98
RCP 8.5 high + 35%	0.87	0.90	0.91	0.94	0.96
Yankee Fork					
NAA	0.32	0.38	0.41	0.45	0.50
Stationary	0.11	0.14	0.17	0.19	0.21
Stationary +17%	0.04	0.05	0.07	0.08	0.11
Stationary + 35%	0.01	0.02	0.03	0.04	0.06
RCP 8.5 low	0.36	0.40	0.43	0.46	0.50
RCP 8.5 low +17%	0.17	0.21	0.24	0.27	0.30
RCP 8.5 low + 35%	0.07	0.10	0.12	0.14	0.17
RCP 8.5 mean	0.70	0.73	0.76	0.78	0.81
RCP 8.5 mean +17%	0.51	0.56	0.59	0.62	0.67
RCP 8.5 mean + 35%	0.33	0.38	0.42	0.44	0.47
RCP 8.5 high	0.83	0.87	0.89	0.91	0.94
RCP 8.5 high +17%	0.73	0.79	0.81	0.84	0.87
RCP 8.5 high + 35%	0.60	0.66	0.68	0.70	0.74

Table 4-4. Probability of quasi-extinction across all eight populations and the 13 model scenarios, with a quasi-extinction threshold of 50. The quasi-extinction probability for the natural-origin spawners was calculated over simulation Years 1 to 24. The climate scenarios are *Stationary* and a range of GCM projections for the *RCP 8.5* emissions scenarios. *Low*, *mean*, and *high* reflect the 25th, 50th, and 75th quantiles across GCM time series. A range of percentiles (5%, 25%, 50%, 75%, 95%) is shown for abundance across simulations for each scenario in each population.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
East Fork Salmon River					
NAA	0.13	0.15	0.18	0.21	0.25
Stationary	0.20	0.24	0.28	0.30	0.35
Stationary + 17%	0.07	0.10	0.11	0.13	0.16
Stationary + 35%	0.02	0.04	0.05	0.06	0.09
RCP 8.5 low	0.46	0.51	0.54	0.57	0.63
RCP 8.5 low + 17%	0.18	0.23	0.25	0.28	0.31
RCP 8.5 low + 35%	0.06	0.07	0.09	0.11	0.14
RCP 8.5 mean	0.76	0.78	0.80	0.83	0.88
RCP 8.5 mean + 17%	0.46	0.51	0.53	0.56	0.59
RCP 8.5 mean + 35%	0.27	0.30	0.33	0.36	0.41
RCP 8.5 high	0.88	0.91	0.93	0.94	0.96
RCP 8.5 high + 17%	0.72	0.76	0.78	0.80	0.84
RCP 8.5 high + 35%	0.49	0.55	0.59	0.61	0.66
Lemhi River					
NAA	0.44	0.49	0.53	0.56	0.61
Stationary	0.45	0.51	0.54	0.57	0.61
Stationary +17%	0.24	0.27	0.29	0.32	0.36
Stationary + 35%	0.09	0.12	0.15	0.17	0.21
RCP 8.5 low	0.69	0.74	0.77	0.79	0.83
RCP 8.5 low +17%	0.43	0.48	0.50	0.54	0.58
RCP 8.5 low + 35%	0.17	0.20	0.23	0.25	0.30
RCP 8.5 mean	0.87	0.90	0.92	0.93	0.95
RCP 8.5 mean +17%	0.65	0.68	0.72	0.75	0.80
RCP 8.5 mean + 35%	0.37	0.43	0.47	0.49	0.54
RCP 8.5 high	0.95	0.96	0.97	0.98	1.00
RCP 8.5 high +17%	0.83	0.85	0.88	0.90	0.93
RCP 8.5 high + 35%	0.67	0.71	0.74	0.76	0.80
North Fork Salmon River					
NAA	0.88	0.91	0.93	0.94	0.97
Stationary	0.90	0.92	0.94	0.95	0.97
Stationary +17%	0.82	0.84	0.87	0.89	0.91
Stationary + 35%	0.60	0.62	0.65	0.67	0.70
RCP 8.5 low	0.99	1.00	1.00	1.00	1.00
RCP 8.5 low +17%	0.97	0.98	0.99	0.99	1.00
RCP 8.5 low + 35%	0.87	0.90	0.92	0.93	0.95
RCP 8.5 mean	1.00	1.00	1.00	1.00	1.00
RCP 8.5 mean +17%	0.99	1.00	1.00	1.00	1.00
RCP 8.5 mean + 35%	0.95	0.97	0.98	0.99	1.00
RCP 8.5 high	1.00	1.00	1.00	1.00	1.00
RCP 8.5 high +17%	1.00	1.00	1.00	1.00	1.00
RCP 8.5 high + 35%	0.98	0.99	0.99	1.00	1.00

Table 4-4 (continued). Probability of quasi-extinction across all eight populations and the 13 model scenarios, with a quasi-extinction threshold of 50.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Pahsimeroi River					
NAA	0.26	0.30	0.32	0.36	0.41
Stationary	0.11	0.15	0.18	0.20	0.24
Stationary + 17%	0.07	0.09	0.10	0.13	0.16
Stationary + 35%	0.03	0.05	0.06	0.07	0.09
RCP 8.5 low	0.20	0.26	0.28	0.31	0.34
RCP 8.5 low + 17%	0.13	0.16	0.18	0.21	0.24
RCP 8.5 low + 35%	0.04	0.07	0.08	0.10	0.13
RCP 8.5 mean	0.47	0.51	0.55	0.58	0.62
RCP 8.5 mean + 17%	0.29	0.34	0.37	0.39	0.45
RCP 8.5 mean + 35%	0.14	0.18	0.20	0.23	0.28
RCP 8.5 high	0.73	0.77	0.79	0.81	0.85
RCP 8.5 high + 17%	0.55	0.58	0.60	0.63	0.68
RCP 8.5 high + 35%	0.34	0.40	0.43	0.47	0.50
Panther Creek					
NAA	0.70	0.75	0.77	0.79	0.83
Stationary	0.56	0.60	0.62	0.66	0.71
Stationary +17%	0.34	0.37	0.41	0.44	0.47
Stationary + 35%	0.20	0.25	0.27	0.29	0.33
RCP 8.5 low	0.86	0.89	0.91	0.92	0.95
RCP 8.5 low +17%	0.71	0.74	0.76	0.78	0.81
RCP 8.5 low + 35%	0.49	0.55	0.58	0.61	0.65
RCP 8.5 mean	0.96	0.97	0.98	0.99	1.00
RCP 8.5 mean +17%	0.91	0.93	0.95	0.96	0.98
RCP 8.5 mean + 35%	0.81	0.85	0.86	0.88	0.91
RCP 8.5 high	0.99	1.00	1.00	1.00	1.00
RCP 8.5 high +17%	0.95	0.96	0.98	0.98	0.99
RCP 8.5 high + 35%	0.92	0.94	0.95	0.96	0.98
Salmon River upper mainstem					
NAA	0.00	0.00	0.00	0.01	0.02
Stationary	0.00	0.00	0.01	0.01	0.03
Stationary +17%	0.00	0.00	0.00	0.00	0.00
Stationary + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low	0.00	0.00	0.01	0.02	0.03
RCP 8.5 low +17%	0.00	0.00	0.00	0.01	0.01
RCP 8.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean	0.06	0.07	0.09	0.11	0.14
RCP 8.5 mean +17%	0.02	0.03	0.05	0.06	0.07
RCP 8.5 mean + 35%	0.00	0.01	0.02	0.02	0.04
RCP 8.5 high	0.22	0.25	0.27	0.29	0.35
RCP 8.5 high +17%	0.09	0.12	0.14	0.16	0.19
RCP 8.5 high + 35%	0.03	0.05	0.06	0.08	0.09

Table 4-4 (continued). Probability of quasi-extinction across all eight populations and the 13 model scenarios, with a quasi-extinction threshold of 50.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Valley Creek					
NAA	0.75	0.80	0.82	0.84	0.88
Stationary	0.79	0.82	0.85	0.86	0.89
Stationary + 17%	0.59	0.62	0.65	0.68	0.72
Stationary + 35%	0.36	0.43	0.46	0.49	0.53
RCP 8.5 low	0.97	0.98	0.99	0.99	1.00
RCP 8.5 low + 17%	0.85	0.88	0.90	0.92	0.94
RCP 8.5 low + 35%	0.70	0.74	0.77	0.79	0.83
RCP 8.5 mean	0.99	1.00	1.00	1.00	1.00
RCP 8.5 mean + 17%	0.95	0.97	0.98	0.99	1.00
RCP 8.5 mean + 35%	0.91	0.93	0.94	0.96	0.98
RCP 8.5 high	1.00	1.00	1.00	1.00	1.00
RCP 8.5 high + 17%	0.97	0.98	0.99	1.00	1.00
RCP 8.5 high + 35%	0.96	0.97	0.98	0.99	1.00
Yankee Fork					
NAA	0.80	0.83	0.86	0.87	0.90
Stationary	0.59	0.65	0.67	0.71	0.74
Stationary +17%	0.35	0.40	0.44	0.46	0.50
Stationary + 35%	0.23	0.26	0.30	0.33	0.37
RCP 8.5 low	0.89	0.91	0.92	0.94	0.96
RCP 8.5 low +17%	0.71	0.74	0.77	0.79	0.82
RCP 8.5 low + 35%	0.55	0.59	0.62	0.65	0.69
RCP 8.5 mean	0.97	0.98	0.99	0.99	1.00
RCP 8.5 mean +17%	0.91	0.94	0.95	0.96	0.98
RCP 8.5 mean + 35%	0.80	0.84	0.86	0.88	0.91
RCP 8.5 high	0.98	0.99	1.00	1.00	1.00
RCP 8.5 high +17%	0.96	0.98	0.98	0.99	1.00
RCP 8.5 high + 35%	0.91	0.94	0.95	0.96	0.97

Table 4-5. Geometric mean of total spawner (natural- and hatchery-origin spawning in the wild) abundance. Geometric mean is calculated over simulation Years 15 to 24. The climate scenarios are *Stationary* and a range of GCM projections for the *RCP 8.5* emissions scenarios. *Low*, *mean*, and *high* reflect the 25th, 50th, and 75th quantiles across GCM time series. A range of percentiles (5%, 25%, 50%, 75%, 95%) is shown for abundance across simulations for each scenario in each population.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Pahsimeroi River					
NAA	206	247	279	314	388
Stationary	219	295	372	470	678
Stationary + 17%	246	347	443	570	831
Stationary + 35%	294	401	515	667	970
RCP 8.5 low	166	204	238	292	389
RCP 8.5 low + 17%	176	221	270	327	463
RCP 8.5 low + 35%	198	264	326	408	589
RCP 8.5 mean	127	164	191	229	303
RCP 8.5 mean + 17%	136	178	212	258	365
RCP 8.5 mean + 35%	159	198	234	290	411
RCP 8.5 high	100	134	163	192	252
RCP 8.5 high + 17%	116	156	184	218	282
RCP 8.5 high + 35%	126	170	200	245	343
Salmon River upper mainstem					
NAA	254	363	455	570	777
Stationary	285	418	559	717	1,019
Stationary +17%	335	507	672	872	1,267
Stationary + 35%	396	573	785	1,033	1,478
RCP 8.5 low	186	263	332	409	574
RCP 8.5 low +17%	215	316	399	497	708
RCP 8.5 low + 35%	251	373	476	599	870
RCP 8.5 mean	137	180	227	289	420
RCP 8.5 mean +17%	151	216	277	362	538
RCP 8.5 mean + 35%	171	254	326	423	646
RCP 8.5 high	109	146	181	229	345
RCP 8.5 high +17%	125	168	207	264	418
RCP 8.5 high + 35%	142	195	243	318	488

Table 4-6. Estimates of probability of quasi extinction at 24 years with a QET of 30 spawners for the two supplemented populations and 13 scenarios output from the Upper Salmon River simulation life cycle model. The climate scenarios are *Stationary* and a range of GCM projections for the *RCP 8.5* emissions scenarios. *Low*, *mean*, and *high* reflect the 25th, 50th, and 75th quantiles across GCM time series. A range of percentiles (5%, 25%, 50%, 75%, 95%) is shown for abundance across simulations for each scenario in each population.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Pahsimeroi River					
NAA	0.00	0.00	0.00	0.00	0.00
Stationary	0.00	0.00	0.00	0.00	0.00
Stationary + 17%	0.00	0.00	0.00	0.00	0.00
Stationary + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low + 17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean + 17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 high	0.00	0.00	0.00	0.00	0.01
RCP 8.5 high + 17%	0.00	0.00	0.00	0.00	0.01
RCP 8.5 high + 35%	0.00	0.00	0.00	0.00	0.00
Salmon River upper mainstem					
NAA	0.00	0.00	0.00	0.00	0.00
Stationary	0.00	0.00	0.00	0.00	0.00
Stationary +17%	0.00	0.00	0.00	0.00	0.00
Stationary + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low +17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean +17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 high	0.00	0.00	0.00	0.00	0.01
RCP 8.5 high +17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 high + 35%	0.00	0.00	0.00	0.00	0.01

Table 4-7. Estimates of probability of quasi extinction at 24 years with a QET of 50 spawners for the two supplemented populations and 13 scenarios output from the Upper Salmon River simulation life cycle model. The climate scenarios are *Stationary* and a range of GCM projections for the *RCP 8.5* emissions scenarios. *Low*, *mean*, and *high* reflect the 25th, 50th, and 75th quantiles across GCM time series. A range of percentiles (5%, 25%, 50%, 75%, 95%) is shown for abundance across simulations for each scenario in each population.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Pahsimeroi River					
NAA	0.00	0.00	0.00	0.00	0.00
Stationary	0.00	0.00	0.00	0.00	0.00
Stationary + 17%	0.00	0.00	0.00	0.00	0.00
Stationary + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low + 17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean	0.00	0.00	0.00	0.01	0.02
RCP 8.5 mean + 17%	0.00	0.00	0.00	0.00	0.01
RCP 8.5 mean + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 high	0.00	0.00	0.01	0.01	0.02
RCP 8.5 high + 17%	0.00	0.00	0.00	0.01	0.01
RCP 8.5 high + 35%	0.00	0.00	0.00	0.00	0.00
Salmon River upper mainstem					
NAA	0.00	0.00	0.00	0.00	0.00
Stationary	0.00	0.00	0.00	0.00	0.00
Stationary +17%	0.00	0.00	0.00	0.00	0.00
Stationary + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low +17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 low + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean +17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 mean + 35%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 high	0.00	0.00	0.01	0.01	0.02
RCP 8.5 high +17%	0.00	0.00	0.00	0.00	0.00
RCP 8.5 high + 35%	0.00	0.00	0.00	0.00	0.01

4.7 Conclusion

Overall, tributary habitat quality and quantity improvements resulted in improvements in population abundance and extinction risk metrics for all Upper Salmon River MPG Chinook salmon 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 of habitat actions alone ranged from 15% to 113% (maximum seen in the Pahsimeroi River population relative to the NAA). However, positive benefits from management actions, to the extent proposed in NMFS (2020), are small relative to the projected decreases in ocean survival. 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 environmental-management scenarios.

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5 Middle Fork and South Fork Salmon River MPGs of the Snake River Spring/Summer-Run Chinook Salmon ESU

Lisa G. Crozier and Richard W. Zabel

5.1 Introduction

This document presents life cycle model (LCM) results for eight populations of Snake River spring/summer-run Chinook salmon that spawn in tributaries of the Salmon River in Idaho. The analyzed populations span three major population groups (MPGs) and were selected because they have extensive passive integrated transponder (PIT) data, particularly in the parr rearing stage prior to movement past smolt traps (e.g., Lamb et al. 2018). Bear Valley Creek, Big Creek, Camas Creek, Loon Creek, Marsh Creek, and Sulphur Creek are in the Middle Fork Salmon River MPG (Figure 5-1). Secesh River is in the South Fork Salmon River MPG, and Valley Creek is in the Upper Salmon River MPG. They are all wild populations, with no direct supplementation from hatcheries.

These populations spawn in or adjacent to protected wilderness areas, where habitat is mostly in good condition. Therefore, these populations have received few habitat restoration actions. Instead, we focused on how population viability responds to changes in climate, in combination with proposed actions for the hydrosystem. We used the extensive data available specific to these populations to model the effects of climatic drivers on five separate life stages: spawner to parr, parr to smolt, juvenile migration through the hydropower system, smolt to adult returns (SAR), and adult migration through the hydrosystem. This model builds on earlier versions by Zabel et al. (2006), Crozier et al. (2008), and Crozier et al. (2017b), and was recently published in Crozier et al. (submitted a).

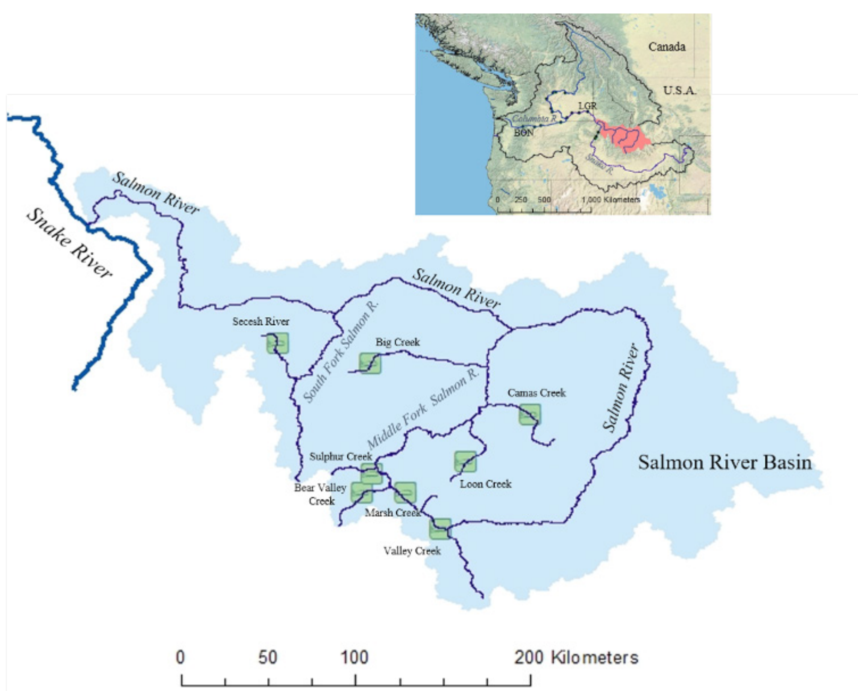

Figure 5-1. Map of the Salmon River basin showing modeled populations within their respective MPGs and the outline of the Columbia River basin.

A particular advance in this version of the model was to account for the correlations among environments that salmon encounter as they move through their life cycle. For migratory species and species with complex life cycles, it is especially important to account for the correlation structure of climate effects in different habitats and life stages, because large-scale climate forcing could have delayed or multiplicative effects over the life cycle. Population-level effects are also sensitive to climate events that affect multiple cohorts simultaneously. Snake River spring/summer-run Chinook salmon travel from their headwater habitat in central Idaho, past eight major hydroelectric projects in the Snake and Columbia Rivers, to grow and mature in the northeastern Pacific Ocean. They return to freshwater from one to four years later for a single spawning opportunity. Survival in the early marine stage, in particular, is very sensitive to carry-over effects from freshwater, such as arrival timing and body size.

Here we integrate individual- and population-level data with current climate projections from global climate models (GCMs) for covariates that affect each life stage. The methods and sources for adding climate projections to the LCM are described in Crozier et al. (submitted a) and Crozier et al. (submitted b), and details regarding modeling of the proposed action are described in Faulkner et al. (this volume, Chapter 2) and Appendix A. In this chapter, we describe the population responses to scenarios conducted specifically for the Biological Opinion (NMFS 2020).

There is a great deal of uncertainty regarding how ocean ecosystems will respond to the combination of climate change, ocean acidification, and other drivers, and how salmon will cope with consequent community reorganization. Here we employ the best statistical models that are available for quantitative projection to characterize our current understanding of the relationship between environmental conditions and salmon survival.

5.2 Methods

5.2.1 Life history overview

The populations studied here primarily follow the characteristic life history of stream-type Chinook salmon (Healey 1991). Adults enter freshwater from April to June, with individual populations clustering into early ("spring") and late ("summer") groups (Crozier et al. 2016), hence the joint designation of the ESU as "spring/summer-run." Adults spend one to three months migrating or holding in deep pools in the Salmon River basin prior to spawning in August and September (Crozier et al. 2017a). Females usually reproduce at four or five years old, with a fecundity advantage for five-year-olds. Males are more likely than females to return at a younger age. Both sexes are considered "adults" at four or older. Eggs hatch the following spring, and juveniles rear for a full year in freshwater before migrating through the mainstem Snake and Columbia Rivers in April and May. These populations typically migrate relatively quickly through the Columbia River estuary and head northward in early summer to the Gulf of Alaska (Peterson et al. 2010, Weitkamp 2010, Fisher et al. 2014). Salmon from this ESU are rarely caught in ocean fisheries, but they are an important component of fisheries in the lower Columbia River. Harvest quotas in the lower river stem from federal treaties and cooperative agreements with states and other parties (United States v. Oregon 2018).

5.2.2 Data informing survival estimates

Spawner age and abundance estimates were compiled through large-scale collaborative efforts between states, tribes, and coordinating bodies (StreamNet 2010, IDFG et al. 2018, Nez Perce Tribe 2019). Stage-specific survival estimates were obtained from multiple sources (Crozier et al. 2016, DART 2017, Faulkner et al. 2018, Lamb et al. 2018, NMFS 2019, Chasco et al. submitted), originating from tagging and detection records downloaded from the Columbia Basin PIT Tag Information System (PTAGIS; Table S1 in Crozier et al. submitted a).[1] We only used detection records for fish identified as wild from known population sources. Environmental covariates used in the model include air temperature, stream flow and stream temperature, sea surface temperatures (SST), and coastal upwelling (Table S2 in Crozier et al. submitted a).

Parr survival estimates reflect data from fish that were PIT-tagged over summer as juveniles in high-elevation rearing habitat and the following spring along the migration route (Achord et al. 2011, Lamb et al. 2018). Hydrosystem survival has been closely monitored and modeled since 1993 (Zabel et al. 2008, Faulkner et al. 2018), and is covered in detail in this volume, Chapter 2. From 1993–2016, between 12 and 95% of smolts were transported in barges past six to eight dams in the Snake and Columbia Rivers and released in the estuary below Bonneville Dam ("transported fish"), while the remainder migrated downstream in the river ("in-river fish"). Transported fish were assumed to have high survival in the barges (0.98), but tend to have lower adult return rates than in-river fish. Survival from the smolt-to-adult stage for both groups was estimated from PIT-tag detections at Bonneville Dam (Chasco et al. submitted). Upstream survival is also based on PIT-tag detections from Bonneville Dam to Lower Granite Dam from 2004–16, and includes harvest in the mainstem rivers (Crozier et al. 2016, Crozier et al. submitted b). Survival from Lower Granite Dam to the spawning grounds (i.e., prespawn mortality) was treated as a constant (0.9), based on data summarized by Bowerman et al. (2016).

5.2.3 Model structure

We employed a stochastic, age-structured model modified from earlier publications (Kareiva et al. 2000, Zabel et al. 2006, Crozier et al. 2008). The model, as depicted in Figure 5-2, has five annual time steps based on the five-year generation time of Snake River spring/summer-run Chinook salmon, which correspond approximately to life stage transitions. The first time step (from *Spawners* to *Parr*) spans fall spawning, incubation, and early parr rearing. Survival through the second time step (S_2) includes both tributary rearing ($S_{tributary}$) from summer (July or August) to the following spring, when they pass Lower Granite Dam, and migration ($S_{mainstem}$) through the Snake and Columbia River hydrosystem. The third time step includes ocean entry and the first winter and spring in the ocean. Some Chinook salmon return to spawn in their third year (jacks), but most females and adult males stay in the ocean for one or two more years, during which ocean survival is represented as S_o. The number of fish in the ocean each year is a latent variable,

[1] https://ptagis.org/

fit by detections of survivors when they re-enter freshwater (S_{SAR}). Upstream migration survival through the hydrosystem ($S_{upstream}$) and from Lower Granite Dam to spawning ($S_{prespawn}$) are captured in the fourth or fifth time step, following the propensity to return at a given age. Older females tend to lay more eggs, which is reflected in the fecundity parameter, F_5. To calculate "effective spawners," we combined different age classes that returned to spawn in the same year as the weighted sum of three- (weight = 0), four- (weight = 1), and five-year-old (weight = F_5) fish, which is estimated by an expansion of the number of redds (nests) counted during spawning surveys (ICTRT and Zabel 2007).

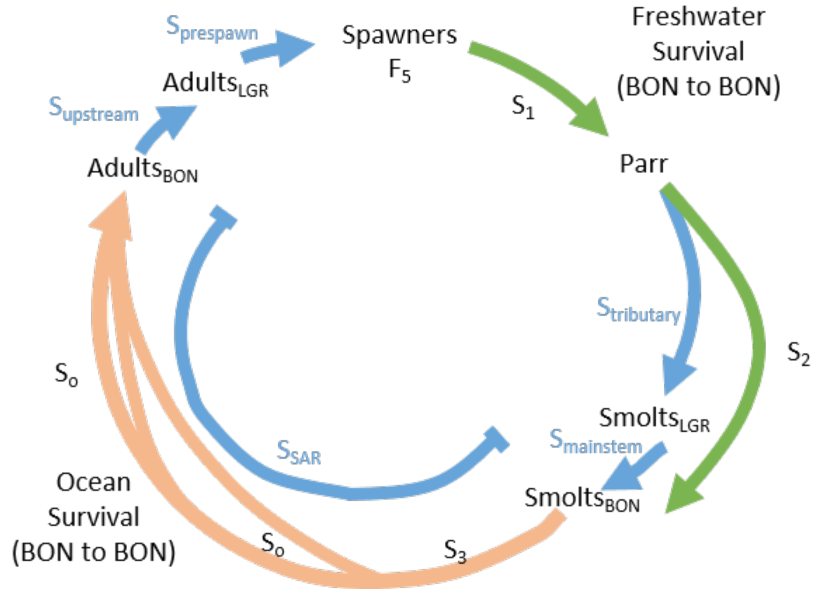

Figure 5-2. Diagram of life cycle model for Snake River spring/summer-run Chinook salmon. Environmental covariate survival models that were fit directly to PIT-tagged data are shown in blue ($S_{tributary}$, $S_{mainstem}$, S_{SAR}, $S_{upstream}$). Life stages and fitted transition parameters are in black.

5.2.4 Model fitting

We fit the life cycle model in two steps. We first fit individual life stage relationships to recent population data. However, the life cycle model introduces some additional parameters that could not be directly fit to data, and also involves linking submodels that were developed independently. We therefore calibrated the full life cycle model in a second step using a modified Approximate Bayesian Computing approach (Csillery et al. 2010, Hartig et al. 2011). After running the life cycle model over 500,000 "prior" parameter sets under historical conditions (1998–2015), we selected the top 1,000 parameter sets (the top 0.2%) for each population ranked by deviance from a Kolmogorov-Smirnov test for projection runs.

Spawner-to-parr (S_1) and parr-to-smolt ($S_{tributary}$) stages: We fit adult recruits per spawner for eight populations in a hierarchical Bayesian framework using multiple likelihood equations that reflected stages that could be directly compared with data. We used a two-stage Gompertz function (Gompertz 1825) to solve the two stages simultaneously, combined with independently estimated survivals for later stages (Crozier et al. submitted a). Individual population coefficients (productivity and capacity parameters for both stages, as well as coefficients for temperature and flow) were assumed to be random samples from an underlying normal distribution (Gelman et al. 2004).

After ensuring that seasonal mean flows on the mainstem Salmon River and seasonal mean air temperature over the Middle Fork Salmon River were strongly correlated with higher-resolution dynamics (e.g., within-tributary maximum daily and mean seasonal flows , and mean and maximum daily stream temperatures measured in each tributary), we used seasonal basin-scale environmental indices in further modeling. Pearson's correlation coefficients in pairwise comparisons between basin seasonal mean and tributary maximum daily flows ranged from 0.75 to 0.96. Correlations between seasonal mean air temperature and seasonal mean or August maximum stream temperatures measured at tagging sites ranged from 0.5 to 0.96. Model comparisons across seasonal indices confirmed that summer air temperature and fall flow, as shown in Crozier et al. (2008), were still among the best seasonal indices for predicting parr-to-smolt survival (Crozier et al. submitted a). We therefore used fall (October–December) mean flows at the Salmon, Idaho, stream gage (13302500, USGS 2019) and summer (July–September) mean air temperatures for the upper Middle Fork Salmon River region as modeled by PRISM Climate Group (2017) in simulations.

5.2.4.1 Lower Granite-to-Lower Granite

The Lower Granite-to-Lower Granite module encompasses three life stages: downstream migration through the hydropower system, Bonneville-to-Bonneville SARs, and upstream migration. Models for survival in these stages are described in this volume, Chapter 2, Chasco et al. (submitted), and Crozier et al. (submitted b), respectively. The process of integrating these models into the LCM is explained in Crozier et al. (submitted a). This module is common to all Snake River spring/summer-run Chinook models.

The hydrosystem operation scenarios we considered are described in this volume, Chapter 2. In brief, flow, spill, reservoir elevation, water temperature, and dissolved gas for this study were all modeled by the U.S. Army Corps of Engineers as either the Proposed Action (PA) or the No-Action Alternative (NAA) considered in the Environmental Impact Statement (USACE et al. 2020). The PA models assume that turbine replacements at Ice Harbor, McNary, and John Day Dams will substantially lower fish mortality compared with the existing turbines. In the NAA scenario, turbine replacements only occurred at McNary and Ice Harbor Dams.

The PA runs used a universal transportation start date of 20 April at all three transporter dams: Lower Granite Dam, Little Goose Dam, and Lower Monumental Dam. After this date, all fish predicted to enter the bypass system at these dams were treated as transported fish by the COMPASS model; they are removed from the river at the transport dam, and added to the tailrace of Bonneville Dam two days later. The transportation start date was 1 May in the NAA scenario. COMPASS assumes uniform 0.98 survival during transportation. In each simulation, the COMPASS model produced distributions of arrival times for in-river and transported smolts at Bonneville Dam which were then input into the SAR model.

Smolt-to-adult return (S_{SAR}): We used a mixed-effects logistic regression model for wild fish (Chasco et al. submitted) to determine the effect of the date of ocean entry (from COMPASS) and environmental covariates (specified by the SAR model and the climate scenario) on the probability that an individual fish would return as an adult to Bonneville Dam. The SAR model includes random effects for day and for the day by year interaction, which follow an autoregressive process over time. We developed separate models for fish that migrated through the mainstem in the river and for fish that had been transported downstream in barges. Through model selection by Akaike information criterion (AIC), we identified the top models for each migration type. The top model for transported fish included a variable for alongshore flow that we could not extract from GCMs, but models without this variable performed similarly (ΔAIC <4). We therefore selected a model for transported fish that included only a single covariate, a large-scale measure of sea surface temperature (*SSTarc* in winter), and a model for in-river migrants that included two covariates, *SSTarc* in winter and a more local measure of SST along the Washington coast (*SSTwa* in summer).

We also assessed the implications of speculative reductions in delayed mortality for in-river migrating fish. In the delayed mortality scenarios, we increased marine survival rates of in-river migrants by 17% and 35%.

Adult upstream survival ($S_{upstream}$): For the adult upstream survival model, we used generalized additive mixed models (GAMMs) to evaluate the effects of both anthropogenic and environmental covariates on spring/summer-run Chinook salmon survival (Crozier et al. submitted b). To run the model in simulation mode, we assumed fisheries catch had similar distributions to the baseline period, 2004–16. The proportion of returning adults that had been transported in barges as juveniles was an output from the COMPASS model followed by the SAR model. We assumed transportation history did not affect the propensity to return at a given age. We held survival from the hydrosystem to spawning ($S_{prespawn}$) constant due to the lack of appropriate data for most populations with which to fit a relationship.

5.2.4.2 Ocean age at maturity

After fitting initial parameter values using the point estimates for all stage-specific survivals, we performed a pseudo-Bayesian analysis to better account for uncertainty in and correlations among all parameter estimates. The parameters that were calibrated partition total smolt-to-adult survival (S_{SAR}) output from the SAR submodel to age-specific survivals (S_3) for the first year, then S_0 for the second and third years, combined with a propensity to return at a given age (jacks: b_3; 4-year-olds: b_4; 5-year-olds: $1 - b_4$) as follows:

$$S_{SAR} = \frac{b_3 \times N_3 + b_4 \times N_4 + N_5}{N_2}$$

Where $N_3 = S_3 \times N_2$,
$N_4 = (1 - b_3) \times N_3 \times S_0$, and
$N_5 = (1 - b_4) \times N_4 \times S_0$.

Fish that stay in the ocean longer have additional mortality (S_0), but older fish have higher fecundity (F_5). In the model, the effective number of spawners reflects the age distribution of female spawners, which return as either four- or five-year-olds, and the five-year-olds have the fecundity advantage. A very small percentage of fish return as six-year-olds; these fish were lumped with five-year-olds. We further limited the maximum number of effective spawners in simulations to 2,500, to limit population growth that goes well beyond anything seen since the beginning of the spawner database in 1957.

We calibrated the model following a method described in Jorgensen and Bond (this volume, Chapter 6) for Wenatchee River population models. Briefly, we used a pseudo-Bayesian approach, in which the model is run with 500,000 sets of parameter values ("priors"), and then selected the top 1,000 parameter combinations ("posteriors") that produced the distributions of spawner counts and parr-to-smolt survival estimates that were most similar to the observed estimates. Because recent population sizes did not always provide sufficient information to constrain production at very large escapements, we further restricted the parameter set to values that produced fewer than twice the estimated historical production of smolts under mean conditions from twice the historical maximum spawner count. Priors for the age-at-maturity parameters reflected the mean and range specified in Table 1 in Zabel et al. (2006). We set the mean of each prior at the mean from Zabel et al. (2006), and adjusted the variance of either a beta distribution (for b_3, b_4, and S_0) or a normal distribution (for F_5) until we encompassed the range specified in Zabel et al. (2006).

We used a Kolomogorov-Smirnoff test to rank model fit by comparing model predictions to observations for both spawner estimates and parr-to-smolt survival. We selected the top 0.2% of parameter combinations as those parameter sets that produced the smallest sum of the squared D statistics for spawners and parr-to-smolt survival estimates.

$$D_{calibration} = D_{spawner}^{\ 2} + D_{parr\text{-}to\text{-}smolt}^{\ 2}$$

This approach retains any correlation structure in the parameter values that is consistent with the data. We then used these posterior parameter sets to simulate population time series under various climate and latent mortality scenarios.

5.2.5 Simulation scenarios

We designed simulation scenarios that would compare population trajectories in a climate experiencing historical levels of variability but no directional trends (a "stationary" climate) with population trajectories in a climate responding to anthropogenic greenhouse gases. We created the stationary climate scenario by fitting a covariance matrix for all the freshwater and marine environmental covariates used in the life cycle model using the `TMB` libraries for R (Kristensen et al. 2016). This covariance matrix was then incorporated into a multivariate state-space model that accounts for temporal correlations across environmental variables. Autoregression was further incorporated into the random effects within the SAR model to account for additional temporal patterns that were not captured in the raw environmental time series included in the selected covariates. The state-space model was used to simulate natural variability in all covariates in a stationary climate. In the climate change scenarios, we simply added trends to the stationary simulations. The resulting time series retained similar levels of variability as the historical time series. The NAA scenario was only run for the stationary climate, whereas the PA scenario was run for the stationary climate and a set of climate change scenarios.

5.2.6 Climate trends

To account for anthropogenic carbon emissions, we extracted trends from global climate model (GCM) projections of representative concentration pathway (RCP) 8.5. This is considered a business-as-usual scenario for carbon emissions. The climate scenario was modelled using the ensemble approach, as advocated by the Intergovernmental Panel on Climate Change (IPCC 2014). This approach addresses uncertainty in GCM model assumptions by using as many different models as possible. There are 26 GCMs available from Coupled Model Intercomparison Project CMIP5, provided by NOAA's Earth Systems Research Laboratory (Alexander et al. 2018). Representative GCM projections were selected for relatively slow warming, relatively fast warming, and the ensemble mean. We used output directly from the CMIP5 effort for marine variables. For freshwater variables, more steps were involved in generating each trend. Scientists at the University of Washington downscaled output from 10 of those GCMs using the Multivariate Adaptive Constructed Analogs (MACA) downscaling method, and processed the output through four different hydrological models to project 40 different time series for naturalized flow (RMJOC 2018, Chegwidden et al. in preparation). Our intent was to capture the range of uncertainty across as many of these different projections as possible within RCP 8.5.

To represent the impact of climate change within each of these projections, we calculated monthly anomalies within each time series from a historical reference period, defined as the 2005–25 period. Then, a 20-year running mean of differences was calculated for each time series. At each monthly time step, we then calculated the 25th, 50th, and 75th quantiles across all projections. These quantiles represent trajectories that are relatively low (the 25th quantile), the ensemble mean (the 50th quantile), and relatively high (the 75th quantile) rates of change under the RCP 8.5 scenario.

The average change in temperature (°C) over the first 24 years of our simulations was 0.2–0.5°C from the low- to high-change scenarios at Bonneville Dam and Lower Granite Dam, approximately 1°C in air temperature in the Salmon River basin, and 0.6–0.65°C in both ocean temperature metrics. Changes in mean flow were negligible compared to the standard deviation in flows.

5.2.7 Response metrics

We quantified population responses to the perturbations in terms of 1) geometric mean population abundance in simulation Years 15–24, and 2) the probability of exceeding a quasi-extinction threshold within in the next 24 years. We summarized abundance as the four-year running mean spawner count in a given year, which averages over all cohorts in a generation. We calculated the quasi-extinction probability as the proportion of 1,000 simulations in which the running mean spawner count drops below either 30 (QET 30) or 50 (QET 50) within a given time frame. We characterized uncertainty in the abundance estimate as the 5th, 25th, 50th, 75th, and 95th quantiles across individual simulations of geomean abundance during 15–24 years into the simulation.

We characterized uncertainty in QET 30 and QET 50 by the following steps. We first ran the 5,000 replicates of the LCM. We then randomly sampled 100 replicates at a time, and calculated the QET 30 and QET 50 of each set. We repeated the resampling process 1,000 times to generate 1,000 estimates of each of the two metrics. We reported the 5th, 25th, 50th, 75th, and 95th quantiles of this distribution of estimates.

5.3 Results

In comparing the NAA results with the PA results for a detrended (stationary) climate, mean population abundance was slightly higher in the NAA scenario while latent mortality remained constant across scenarios. However, both the 17.5% and 35% improvements to latent mortality reversed the relationship (Figure 5-3, Table 5-1). Similarly, quasi-extinction risk for both threshold levels was higher in the PA scenarios under the null hypothesis of no change in latent mortality, but not in the other scenarios (Tables 5-2 and 5-3).

In comparing climate scenarios, population abundance declined by roughly 50% from the stationary climate to the ensemble mean climate projection in 24 years, when comparing results at any given level of release from latent mortality (Figure 5-3, Table 5-1). From the more optimistic projection to the more pessimistic climate projection, population abundance declined by about 40–60%. Probabilities of surpassing the quasi-extinction thresholds remained relatively low in the larger populations (<15% for QET 30 across climate scenarios for Bear Valley Creek, Big Creek, Marsh Creek, and Secesh River; Figure 5-3). However, the smaller populations reached quite high probabilities (e.g., QET 30 was 40–88% across most scenarios for Camas Creek, Loon Creek, and Sulphur Creek). Release from latent mortality lowered the probability of extinction for all populations, although QET 30 probabilities were still over 50% for those smaller populations in the more pessimistic climate scenario (RCP 8.5 High, latent mortality = 35%).

Large declines due to climate trends occurred in the marine stage, as shown in Figure 5-4. These trends were primarily responsible for the general population declines across all Snake River spring/summer-run Chinook salmon populations.

5.4 Conclusion

In this study, we compared population abundance and quasi-extinction probabilities under the No-Action Alternative developed for the EIS and the proposed action for NMFS (2020) with speculative ranges of change in marine survival rates of in-river migrants. The NAA scenario had slightly higher abundances and lower extinction probabilities in the stationary climate compared with the PA scenario when there was no improvement in marine survival, but release from latent mortality reversed these relationships.

We also compared population abundance and quasi-extinction probabilities under the Proposed Action across a range of climate scenarios. Abundance declined about 40–60% across climate scenarios, and extinction rates went up under all hypotheses regarding latent mortality.

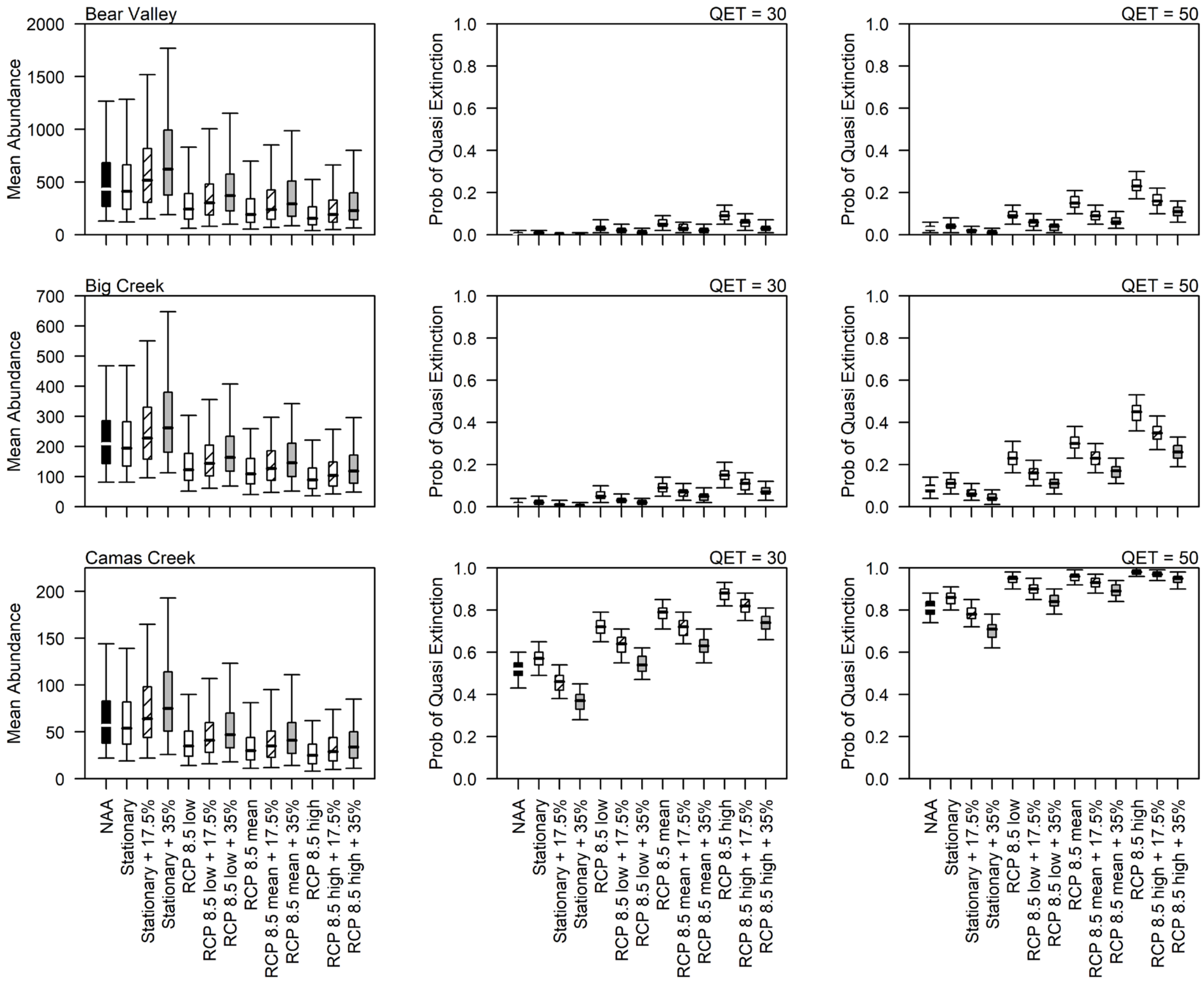

Figure 5-3a. Population responses to the proposed action and climate scenarios, including several assumptions about release from latent mortality (no change, or increase in survival of 17% or 35%). Box edges represent the 25th and 75th percentiles, and the bar in the middle represents the 50th percentile across simulations. The whiskers extend to the 5th and 95th percentiles. The left column shows geometric mean abundance in simulation Years 15 to 24. Other columns show the frequency at which populations dropped below the QET of 30 spawners (middle) or 50 spawners (right) within 24 years of initiating the simulations. The line indicates the ratio of populations that met the criterion acoss 1,000 simulations. The boxes and whiskers reflect the frequencies from the bootstrapping process. These graphs show results for these Middle Fork Salmon River MPG populations: Bear Valley Creek, Big Creek, and Camas Creek.

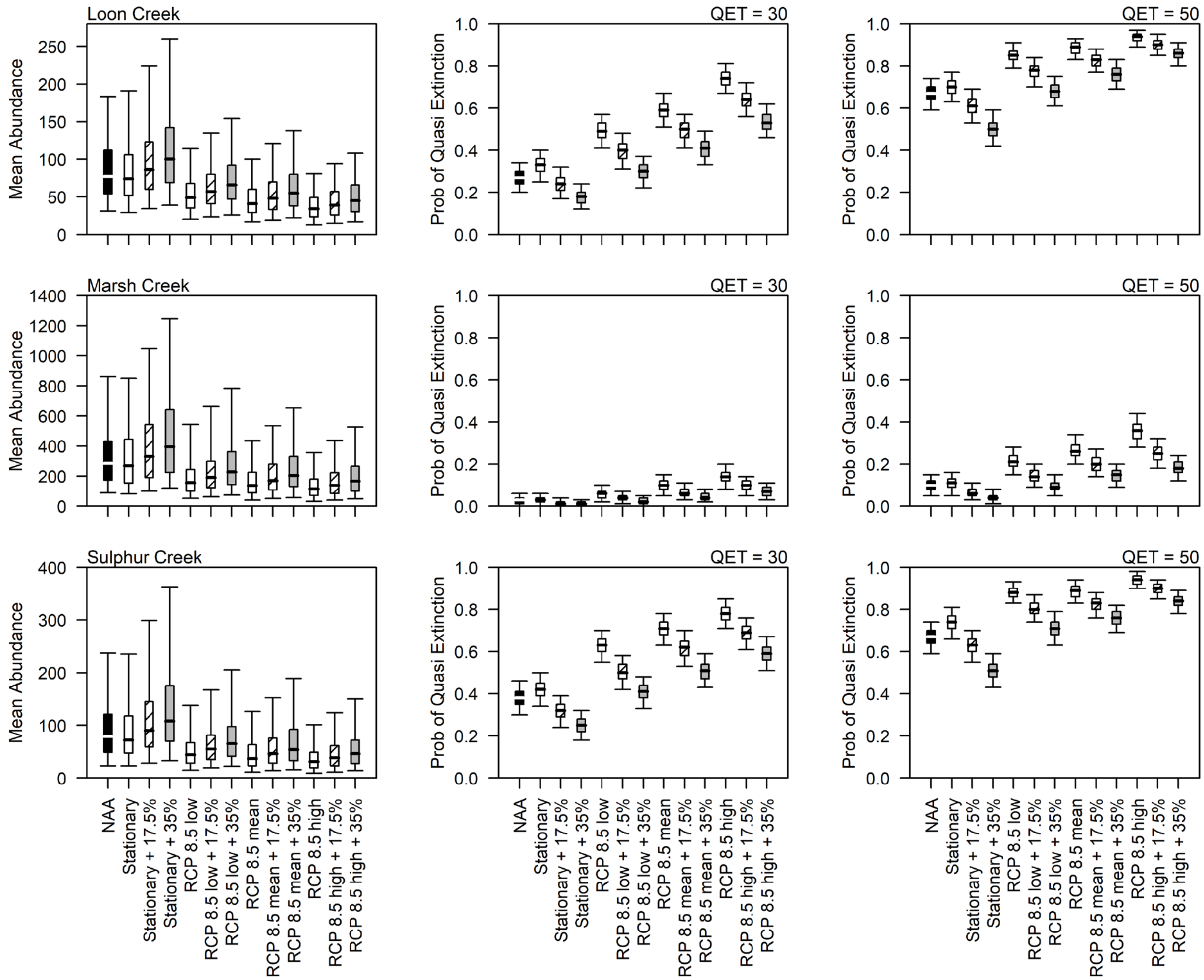

Figure 5-3b. As in Figure 5-3a, for these Middle Fork Salmon River MPG populations: Loon Creek, Marsh Creek, and Sulphur Creek.

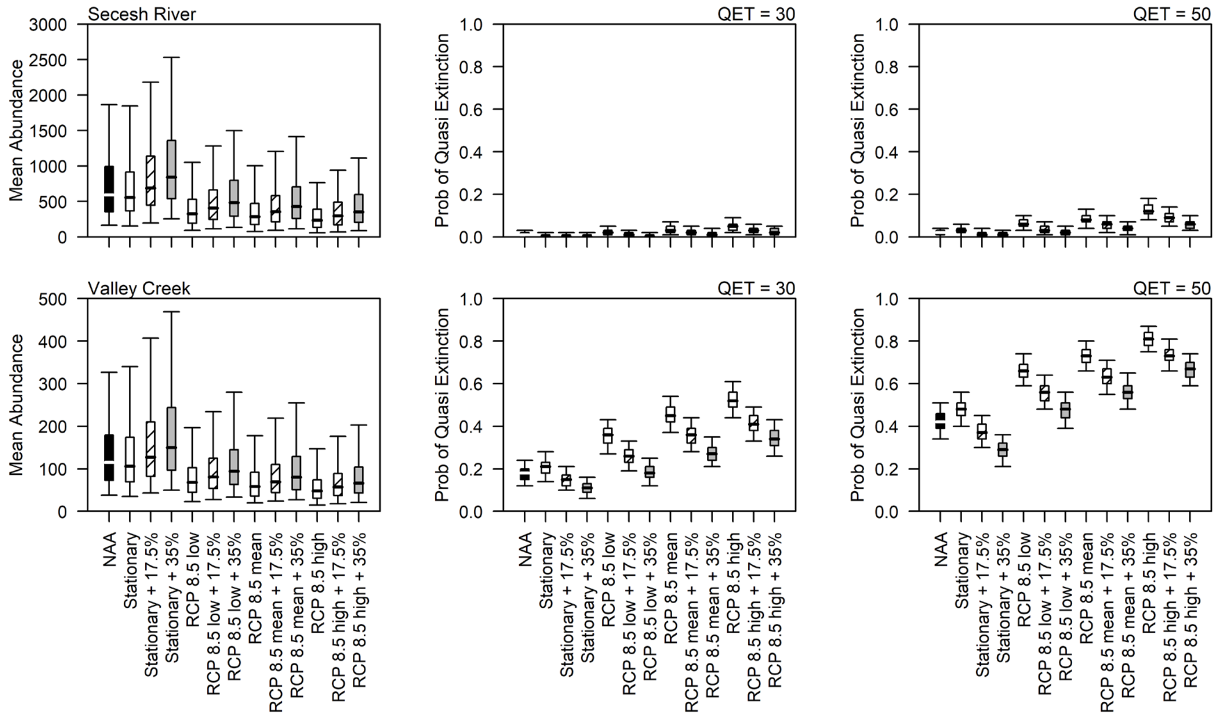

Figure 5-3c. As in Figure 5-3a, for the South Fork Salmon River MPG Secesh River population, and for the Upper Salmon River MPG Valley Creek population.

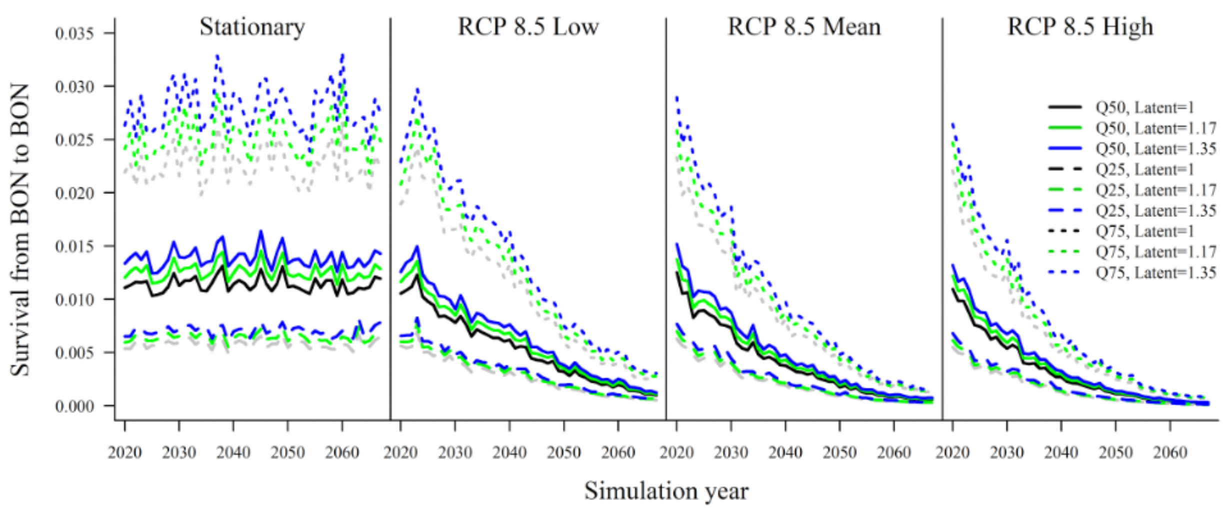

Figure 5-4. Survival of smolts from Bonneville Dam (BON) to adults at Bonneville Dam showing the median across simulations within each scenario (Q50) and the interquartile range (Q25 and Q75) across simulations for all three latent mortality scenarios.

Table 5-1. Abundance estimates for four climate scenarios with alternative assumptions of release from latent mortality. The climate scenarios are *Stationary* and three possibilities from global climate model (GCM) projections for the representative concentration pathway (RCP) 8.5 emissions scenarios. *Low*, *Mean*, and *High* reflect the 25th, 50th, and 75th quantiles across GCM time series. A range of percentiles (5%, 25%, 50%, 75%, 95%) is shown for abundance across simulations for each scenario in each population.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Bear Valley Creek					
NAA	128	269	432	684	1,266
Stationary	121	242	412	663	1,284
Stationary + 17%	150	306	518	818	1,519
Stationary + 35%	190	375	622	993	1,770
RCP 8.5 low	61	149	243	391	829
RCP 8.5 low + 17%	79	188	303	480	1,005
RCP 8.5 low + 35%	99	226	371	575	1,151
RCP 8.5 mean	53	116	193	340	698
RCP 8.5 mean + 17%	69	145	238	423	852
RCP 8.5 mean + 35%	85	176	292	510	986
RCP 8.5 high	38	95	156	265	525
RCP 8.5 high + 17%	49	120	192	329	661
RCP 8.5 high + 35%	62	142	229	397	800
Big Creek					
NAA	82	143	209	286	468
Stationary	82	135	194	282	469
Stationary +17%	96	158	228	330	551
Stationary + 35%	112	181	262	380	647
RCP 8.5 low	52	88	123	177	303
RCP 8.5 low +17%	61	102	144	205	356
RCP 8.5 low + 35%	69	118	164	234	407
RCP 8.5 mean	41	76	109	160	259
RCP 8.5 mean +17%	47	88	127	186	297
RCP 8.5 mean + 35%	52	100	146	211	342
RCP 8.5 high	36	60	89	129	221
RCP 8.5 high +17%	42	69	104	148	257
RCP 8.5 high + 35%	48	78	118	171	296
Camas Creek					
NAA	22	38	57	83	144
Stationary	19	37	54	82	139
Stationary +17%	22	44	64	98	165
Stationary + 35%	26	51	75	114	193
RCP 8.5 low	14	24	35	51	90
RCP 8.5 low +17%	16	28	41	60	107
RCP 8.5 low + 35%	18	33	47	70	123
RCP 8.5 mean	11	20	30	44	81
RCP 8.5 mean +17%	12	23	35	51	95
RCP 8.5 mean + 35%	14	27	41	60	111
RCP 8.5 high	8	16	25	37	62
RCP 8.5 high +17%	10	19	29	44	74
RCP 8.5 high + 35%	11	22	34	50	85

Table 5-1 (continued). Abundance estimates for three climate scenarios with alternative assumptions of release from latent mortality.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Loon Creek					
NAA	31	54	77	112	183
Stationary	29	52	74	106	191
Stationary +17%	34	60	86	123	224
Stationary + 35%	39	69	100	142	260
RCP 8.5 low	20	35	49	68	114
RCP 8.5 low +17%	23	41	57	80	135
RCP 8.5 low + 35%	26	47	66	92	154
RCP 8.5 mean	17	29	41	60	100
RCP 8.5 mean +17%	19	33	48	70	121
RCP 8.5 mean + 35%	22	38	55	80	138
RCP 8.5 high	13	23	34	49	81
RCP 8.5 high +17%	15	26	39	57	94
RCP 8.5 high + 35%	17	30	45	66	108
Marsh Creek					
NAA	90	173	285	431	862
Stationary	83	154	269	445	850
Stationary +17%	101	191	330	542	1,046
Stationary + 35%	121	225	395	642	1,247
RCP 8.5 low	52	101	157	244	544
RCP 8.5 low +17%	62	123	191	298	664
RCP 8.5 low + 35%	75	145	228	362	783
RCP 8.5 mean	41	90	138	226	435
RCP 8.5 mean +17%	50	109	170	280	535
RCP 8.5 mean + 35%	57	131	204	330	654
RCP 8.5 high	32	71	115	181	357
RCP 8.5 high +17%	41	85	140	223	436
RCP 8.5 high + 35%	48	102	166	265	527
Sulphur Creek					
NAA	23	49	79	121	237
Stationary	23	47	72	118	235
Stationary +17%	28	59	90	145	299
Stationary + 35%	33	70	108	175	363
RCP 8.5 low	15	28	44	67	138
RCP 8.5 low +17%	19	35	55	81	167
RCP 8.5 low + 35%	22	41	65	98	205
RCP 8.5 mean	11	23	37	63	126
RCP 8.5 mean +17%	14	28	46	76	152
RCP 8.5 mean + 35%	16	33	54	92	189
RCP 8.5 high	9	19	31	49	101
RCP 8.5 high +17%	11	23	38	61	124
RCP 8.5 high + 35%	14	27	46	72	150

Table 5-1 (continued). Abundance estimates for three climate scenarios with alternative assumptions of release from latent mortality.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Secesh River					
NAA	164	354	593	990	1,864
Stationary	154	364	556	916	1,843
Stationary +17%	198	446	688	1,139	2,180
Stationary + 35%	256	539	840	1,357	2,528
RCP 8.5 low	89	194	323	529	1,049
RCP 8.5 low +17%	112	243	406	663	1,281
RCP 8.5 low + 35%	136	292	484	796	1,495
RCP 8.5 mean	74	173	286	471	1,001
RCP 8.5 mean +17%	92	213	353	583	1,203
RCP 8.5 mean + 35%	114	258	426	707	1,412
RCP 8.5 high	57	136	235	391	763
RCP 8.5 high +17%	70	168	295	488	939
RCP 8.5 high + 35%	85	205	352	594	1,112
Valley Creek					
NAA	38	73	115	179	327
Stationary	35	69	106	174	340
Stationary +17%	43	82	127	210	407
Stationary + 35%	50	97	150	244	469
RCP 8.5 low	23	45	68	103	197
RCP 8.5 low +17%	28	54	81	125	234
RCP 8.5 low + 35%	33	63	94	145	280
RCP 8.5 mean	20	36	58	92	178
RCP 8.5 mean +17%	24	44	69	110	219
RCP 8.5 mean + 35%	27	51	80	129	255
RCP 8.5 high	15	31	48	74	147
RCP 8.5 high +17%	18	37	57	89	176
RCP 8.5 high + 35%	21	43	66	104	203

Table 5-2. Probability of exceeding the quasi-extinction threshold of 30 spawners in the running mean over four-year periods (QET 30). Estimates are shown for four climate scenarios with alternative assumptions of release from latent mortality. The climate scenarios are *Stationary* and a range of global climate model (GCM) projections for the representative concentration pathway (RCP) 8.5 emissions scenarios. *Low*, *Mean*, and *High* reflect the 25th, 50th, and 75th quantiles across GCM time series. A range of percentiles (5%, 25%, 50%, 75%, 95%) is shown for abundance across simulations for each scenario in each population.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Bear Valley Creek					
NAA	0.00	0.00	0.00	0.01	0.02
Stationary	0.00	0.00	0.01	0.01	0.02
Stationary + 17%	0.00	0.00	0.00	0.01	0.01
Stationary + 35%	0.00	0.00	0.00	0.00	0.01
RCP 8.5 low	0.01	0.02	0.03	0.04	0.07
RCP 8.5 low + 17%	0.00	0.01	0.02	0.03	0.05
RCP 8.5 low + 35%	0.00	0.00	0.01	0.02	0.03
RCP 8.5 mean	0.02	0.04	0.05	0.07	0.09
RCP 8.5 mean + 17%	0.01	0.02	0.03	0.05	0.06
RCP 8.5 mean + 35%	0.00	0.01	0.02	0.03	0.05
RCP 8.5 high	0.05	0.07	0.09	0.11	0.14
RCP 8.5 high + 17%	0.02	0.04	0.06	0.07	0.10
RCP 8.5 high + 35%	0.01	0.02	0.03	0.04	0.07
Big Creek					
NAA	0.00	0.01	0.01	0.02	0.04
Stationary	0.00	0.01	0.02	0.03	0.05
Stationary +17%	0.00	0.00	0.01	0.01	0.03
Stationary + 35%	0.00	0.00	0.00	0.01	0.02
RCP 8.5 low	0.02	0.04	0.05	0.07	0.10
RCP 8.5 low +17%	0.00	0.02	0.03	0.04	0.06
RCP 8.5 low + 35%	0.00	0.01	0.02	0.03	0.04
RCP 8.5 mean	0.05	0.07	0.09	0.11	0.14
RCP 8.5 mean +17%	0.03	0.05	0.07	0.08	0.11
RCP 8.5 mean + 35%	0.02	0.03	0.05	0.06	0.09
RCP 8.5 high	0.09	0.13	0.15	0.17	0.21
RCP 8.5 high +17%	0.06	0.08	0.11	0.13	0.16
RCP 8.5 high + 35%	0.03	0.06	0.07	0.09	0.12
Camas Creek					
NAA	0.43	0.49	0.52	0.55	0.60
Stationary	0.49	0.54	0.57	0.60	0.65
Stationary +17%	0.38	0.42	0.46	0.49	0.54
Stationary + 35%	0.28	0.33	0.37	0.40	0.45
RCP 8.5 low	0.65	0.69	0.72	0.75	0.79
RCP 8.5 low +17%	0.55	0.60	0.64	0.67	0.71
RCP 8.5 low + 35%	0.47	0.51	0.54	0.58	0.62
RCP 8.5 mean	0.71	0.76	0.79	0.81	0.85
RCP 8.5 mean +17%	0.64	0.68	0.72	0.75	0.79
RCP 8.5 mean + 35%	0.55	0.60	0.63	0.66	0.71
RCP 8.5 high	0.82	0.85	0.88	0.90	0.93
RCP 8.5 high +17%	0.75	0.79	0.82	0.85	0.88
RCP 8.5 high + 35%	0.66	0.71	0.74	0.77	0.81

Table 5-2 (continued). Probability of falling below quasi-extinction thresholds for QET = 30 for three climate scenarios with alternative assumptions of release from latent mortality.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Loon Creek					
NAA	0.20	0.24	0.27	0.30	0.34
Stationary	0.25	0.30	0.33	0.36	0.40
Stationary +17%	0.17	0.21	0.24	0.27	0.32
Stationary + 35%	0.12	0.15	0.18	0.20	0.24
RCP 8.5 low	0.41	0.46	0.49	0.53	0.57
RCP 8.5 low +17%	0.31	0.36	0.40	0.43	0.48
RCP 8.5 low + 35%	0.22	0.27	0.30	0.33	0.37
RCP 8.5 mean	0.51	0.56	0.59	0.62	0.67
RCP 8.5 mean +17%	0.41	0.46	0.50	0.53	0.57
RCP 8.5 mean + 35%	0.33	0.37	0.41	0.44	0.49
RCP 8.5 high	0.67	0.71	0.74	0.77	0.81
RCP 8.5 high +17%	0.56	0.61	0.64	0.67	0.72
RCP 8.5 high + 35%	0.46	0.50	0.53	0.57	0.62
Marsh Creek					
NAA	0.00	0.01	0.03	0.04	0.06
Stationary	0.00	0.02	0.03	0.04	0.06
Stationary +17%	0.00	0.01	0.01	0.02	0.04
Stationary + 35%	0.00	0.00	0.01	0.02	0.03
RCP 8.5 low	0.02	0.04	0.06	0.07	0.10
RCP 8.5 low +17%	0.01	0.03	0.04	0.05	0.07
RCP 8.5 low + 35%	0.00	0.01	0.02	0.04	0.05
RCP 8.5 mean	0.05	0.08	0.10	0.12	0.15
RCP 8.5 mean +17%	0.03	0.05	0.06	0.08	0.11
RCP 8.5 mean + 35%	0.02	0.03	0.04	0.06	0.08
RCP 8.5 high	0.08	0.12	0.14	0.16	0.20
RCP 8.5 high +17%	0.05	0.08	0.10	0.12	0.14
RCP 8.5 high + 35%	0.03	0.05	0.07	0.09	0.11
Sulphur Creek					
NAA	0.30	0.35	0.38	0.41	0.46
Stationary	0.34	0.39	0.42	0.45	0.50
Stationary +17%	0.24	0.29	0.32	0.35	0.39
Stationary + 35%	0.18	0.22	0.25	0.28	0.32
RCP 8.5 low	0.55	0.60	0.63	0.66	0.70
RCP 8.5 low +17%	0.42	0.47	0.50	0.54	0.58
RCP 8.5 low + 35%	0.33	0.38	0.41	0.44	0.48
RCP 8.5 mean	0.63	0.68	0.71	0.74	0.78
RCP 8.5 mean +17%	0.53	0.58	0.62	0.65	0.70
RCP 8.5 mean + 35%	0.43	0.47	0.51	0.54	0.59
RCP 8.5 high	0.71	0.75	0.78	0.81	0.85
RCP 8.5 high +17%	0.61	0.66	0.69	0.72	0.76
RCP 8.5 high + 35%	0.51	0.56	0.59	0.62	0.67

Table 5-2 (continued). Probability of falling below quasi-extinction thresholds for QET = 30 for three climate scenarios with alternative assumptions of release from latent mortality.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Secesh River					
NAA	0.00	0.00	0.01	0.02	0.03
Stationary	0.00	0.00	0.00	0.01	0.02
Stationary +17%	0.00	0.00	0.00	0.01	0.02
Stationary + 35%	0.00	0.00	0.00	0.01	0.02
RCP 8.5 low	0.00	0.01	0.02	0.03	0.05
RCP 8.5 low +17%	0.00	0.00	0.01	0.02	0.03
RCP 8.5 low + 35%	0.00	0.00	0.00	0.01	0.02
RCP 8.5 mean	0.01	0.02	0.03	0.05	0.07
RCP 8.5 mean +17%	0.00	0.01	0.02	0.03	0.05
RCP 8.5 mean + 35%	0.00	0.01	0.01	0.02	0.04
RCP 8.5 high	0.02	0.03	0.05	0.06	0.09
RCP 8.5 high +17%	0.01	0.02	0.03	0.04	0.06
RCP 8.5 high + 35%	0.00	0.01	0.02	0.04	0.05
Valley Creek					
NAA	0.12	0.15	0.18	0.20	0.24
Stationary	0.14	0.18	0.21	0.23	0.28
Stationary +17%	0.10	0.12	0.15	0.17	0.21
Stationary + 35%	0.06	0.09	0.11	0.13	0.16
RCP 8.5 low	0.27	0.32	0.36	0.39	0.43
RCP 8.5 low +17%	0.19	0.23	0.26	0.29	0.33
RCP 8.5 low + 35%	0.12	0.16	0.18	0.21	0.25
RCP 8.5 mean	0.37	0.42	0.45	0.49	0.54
RCP 8.5 mean +17%	0.28	0.32	0.36	0.39	0.44
RCP 8.5 mean + 35%	0.21	0.24	0.27	0.30	0.35
RCP 8.5 high	0.44	0.49	0.52	0.56	0.61
RCP 8.5 high +17%	0.33	0.38	0.41	0.45	0.49
RCP 8.5 high + 35%	0.26	0.31	0.34	0.38	0.43

Table 5-3. Probability of exceeding the quasi-extinction threshold of 50 spawners in the running mean over four-year periods (QET 50). Estimates are shown for four climate scenarios with alternative assumptions of release from latent mortality. The climate scenarios are *Stationary* and a range of global climate model (GCM) projections for the representative concentration pathway (RCP) 8.5 emissions scenarios. *Low*, *Mean*, and *High* reflect the 25th, 50th, and 75th quantiles across GCM time series. A range of percentiles (5%, 25%, 50%, 75%, 95%) is shown for abundance across simulations for each scenario in each population.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Bear Valley Creek					
NAA	0.01	0.02	0.03	0.04	0.06
Stationary	0.01	0.03	0.04	0.05	0.08
Stationary + 17%	0.00	0.01	0.02	0.02	0.04
Stationary + 35%	0.00	0.00	0.01	0.02	0.03
RCP 8.5 low	0.05	0.08	0.09	0.11	0.14
RCP 8.5 low + 17%	0.02	0.04	0.06	0.07	0.10
RCP 8.5 low + 35%	0.01	0.02	0.04	0.05	0.07
RCP 8.5 mean	0.10	0.13	0.15	0.18	0.21
RCP 8.5 mean + 17%	0.05	0.07	0.09	0.11	0.14
RCP 8.5 mean + 35%	0.03	0.05	0.06	0.08	0.11
RCP 8.5 high	0.17	0.21	0.23	0.26	0.30
RCP 8.5 high + 17%	0.10	0.14	0.16	0.19	0.22
RCP 8.5 high + 35%	0.06	0.09	0.11	0.13	0.16
Big Creek					
NAA	0.04	0.07	0.09	0.10	0.14
Stationary	0.06	0.09	0.11	0.13	0.16
Stationary +17%	0.03	0.05	0.06	0.08	0.11
Stationary + 35%	0.01	0.03	0.04	0.06	0.08
RCP 8.5 low	0.16	0.20	0.23	0.26	0.31
RCP 8.5 low +17%	0.10	0.13	0.16	0.18	0.22
RCP 8.5 low + 35%	0.06	0.09	0.11	0.13	0.16
RCP 8.5 mean	0.23	0.28	0.30	0.33	0.38
RCP 8.5 mean +17%	0.16	0.20	0.23	0.26	0.30
RCP 8.5 mean + 35%	0.11	0.14	0.17	0.19	0.23
RCP 8.5 high	0.36	0.41	0.45	0.48	0.53
RCP 8.5 high +17%	0.27	0.32	0.35	0.38	0.43
RCP 8.5 high + 35%	0.19	0.23	0.26	0.29	0.33
Camas Creek					
NAA	0.74	0.78	0.81	0.84	0.88
Stationary	0.80	0.83	0.86	0.88	0.91
Stationary +17%	0.72	0.76	0.78	0.81	0.85
Stationary + 35%	0.62	0.67	0.71	0.73	0.78
RCP 8.5 low	0.90	0.93	0.95	0.96	0.98
RCP 8.5 low +17%	0.85	0.88	0.90	0.92	0.95
RCP 8.5 low + 35%	0.78	0.82	0.84	0.87	0.90
RCP 8.5 mean	0.92	0.94	0.96	0.97	0.99
RCP 8.5 mean +17%	0.88	0.91	0.93	0.95	0.97
RCP 8.5 mean + 35%	0.84	0.87	0.89	0.92	0.94
RCP 8.5 high	0.96	0.97	0.98	0.99	1.00
RCP 8.5 high +17%	0.94	0.96	0.97	0.98	0.99
RCP 8.5 high + 35%	0.90	0.93	0.95	0.96	0.98

Table 5-3 (continued). Probability of falling below quasi-extinction thresholds for QET = 50 for three climate scenarios with alternative assumptions of release from latent mortality.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Loon Creek					
NAA	0.59	0.64	0.67	0.70	0.74
Stationary	0.63	0.67	0.70	0.73	0.77
Stationary +17%	0.53	0.58	0.61	0.64	0.69
Stationary + 35%	0.42	0.47	0.50	0.53	0.59
RCP 8.5 low	0.79	0.83	0.85	0.87	0.91
RCP 8.5 low +17%	0.70	0.75	0.78	0.80	0.84
RCP 8.5 low + 35%	0.61	0.65	0.68	0.71	0.75
RCP 8.5 mean	0.83	0.86	0.89	0.91	0.93
RCP 8.5 mean +17%	0.77	0.80	0.83	0.85	0.88
RCP 8.5 mean + 35%	0.69	0.73	0.76	0.79	0.83
RCP 8.5 high	0.89	0.92	0.94	0.95	0.97
RCP 8.5 high +17%	0.85	0.88	0.90	0.92	0.95
RCP 8.5 high + 35%	0.80	0.84	0.86	0.88	0.91
Marsh Creek					
NAA	0.05	0.08	0.10	0.12	0.15
Stationary	0.05	0.09	0.11	0.13	0.16
Stationary +17%	0.03	0.05	0.06	0.08	0.11
Stationary + 35%	0.01	0.03	0.04	0.05	0.08
RCP 8.5 low	0.15	0.19	0.21	0.24	0.28
RCP 8.5 low +17%	0.09	0.12	0.14	0.17	0.20
RCP 8.5 low + 35%	0.05	0.08	0.09	0.11	0.15
RCP 8.5 mean	0.20	0.24	0.26	0.29	0.34
RCP 8.5 mean +17%	0.14	0.17	0.20	0.23	0.27
RCP 8.5 mean + 35%	0.09	0.12	0.15	0.17	0.20
RCP 8.5 high	0.28	0.32	0.36	0.39	0.44
RCP 8.5 high +17%	0.18	0.22	0.25	0.28	0.32
RCP 8.5 high + 35%	0.12	0.16	0.18	0.21	0.24
Sulphur Creek					
NAA	0.59	0.64	0.67	0.70	0.74
Stationary	0.66	0.71	0.74	0.77	0.81
Stationary +17%	0.55	0.60	0.63	0.66	0.70
Stationary + 35%	0.43	0.48	0.51	0.54	0.59
RCP 8.5 low	0.83	0.86	0.88	0.90	0.93
RCP 8.5 low +17%	0.74	0.78	0.80	0.83	0.87
RCP 8.5 low + 35%	0.63	0.68	0.71	0.74	0.79
RCP 8.5 mean	0.83	0.86	0.89	0.91	0.94
RCP 8.5 mean +17%	0.76	0.80	0.83	0.85	0.88
RCP 8.5 mean + 35%	0.69	0.73	0.76	0.79	0.82
RCP 8.5 high	0.90	0.92	0.94	0.96	0.98
RCP 8.5 high +17%	0.85	0.88	0.90	0.92	0.94
RCP 8.5 high + 35%	0.78	0.82	0.84	0.86	0.89

Table 5-3 (continued). Probability of falling below quasi-extinction thresholds for QET = 50 for three climate scenarios with alternative assumptions of release from latent mortality.

	Percentiles				
Population, scenario	**5%**	**25%**	**50%**	**75%**	**95%**
Secesh River					
NAA	0.00	0.01	0.02	0.03	0.04
Stationary	0.00	0.02	0.03	0.04	0.06
Stationary +17%	0.00	0.01	0.01	0.02	0.04
Stationary + 35%	0.00	0.00	0.01	0.02	0.03
RCP 8.5 low	0.03	0.05	0.06	0.08	0.10
RCP 8.5 low +17%	0.01	0.02	0.03	0.05	0.07
RCP 8.5 low + 35%	0.00	0.01	0.02	0.03	0.05
RCP 8.5 mean	0.04	0.07	0.08	0.10	0.13
RCP 8.5 mean +17%	0.02	0.04	0.06	0.07	0.10
RCP 8.5 mean + 35%	0.01	0.03	0.04	0.05	0.07
RCP 8.5 high	0.08	0.11	0.12	0.15	0.18
RCP 8.5 high +17%	0.05	0.07	0.09	0.11	0.14
RCP 8.5 high + 35%	0.03	0.04	0.06	0.07	0.10
Valley Creek					
NAA	0.34	0.39	0.42	0.46	0.51
Stationary	0.40	0.45	0.48	0.51	0.56
Stationary +17%	0.30	0.34	0.37	0.41	0.45
Stationary + 35%	0.21	0.26	0.29	0.32	0.36
RCP 8.5 low	0.59	0.63	0.66	0.69	0.74
RCP 8.5 low +17%	0.48	0.52	0.56	0.59	0.64
RCP 8.5 low + 35%	0.39	0.44	0.48	0.51	0.56
RCP 8.5 mean	0.66	0.70	0.73	0.76	0.80
RCP 8.5 mean +17%	0.55	0.60	0.63	0.67	0.71
RCP 8.5 mean + 35%	0.48	0.53	0.56	0.59	0.65
RCP 8.5 high	0.75	0.78	0.81	0.84	0.87
RCP 8.5 high +17%	0.66	0.71	0.73	0.76	0.81
RCP 8.5 high + 35%	0.59	0.63	0.67	0.70	0.74

Chinook salmon, as well as other salmon species, are very sensitive to ocean conditions (Zabel et al. 2006). This analysis showed their effects on abundance and extinction risk within the next 24 years. Although these results are not predictions of a certain future, they do demonstrate the most likely direction of population trajectories for this cold-water fish. Sea surface temperature has been negatively correlated with adult returns in these populations for many decades, and this basic relationship is unlikely to change.

However, there are important caveats to the specific results presented here. Climate has strong decadal and other patterns that can obscure the long-term trends caused by greenhouse gas forcing (Johnstone and Mantua 2014), and the North Pacific might not show the modeled trends in sea surface temperature in the near term. Furthermore, other climate indices, such as the North Pacific Gyre Oscillation, also show strong relationships to salmon survival (Kilduff et al. 2015, Ohlberger et al. 2016), and it is not clear how climate change will affect other modes of variability in the ocean. Finally, salmon survival is not directly related to sea surface temperature, but rather to a suite of ecosystem changes that cause changes in the salmon's prey and predators (Holsman et al. 2012). It is possible that the food web will respond in an unexpected, novel way to future ocean conditions. Nonetheless, the results presented here reflect strong patterns that emerge from existing data and the best climate projections available. Additional research monitoring interactions between salmon and their predators and prey would help to resolve existing uncertainties.

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6 A Life Cycle Modeling Framework for Estimating Impacts to Wenatchee River Spring-run Chinook Salmon from Effects of Proposed Hydropower Operations and Potential Future Habitat Actions

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6.1 Introduction

Estimating the effects of multiple stressors and alternative environmental conditions on endangered populations is an ever-present challenge to natural resource managers. The challenges for Pacific salmonids are compounded because of the diversity of habitats they use: they are found in tributary and mainstem river freshwater habitats at the beginning and end of their lives, and in the open ocean in between. The sources of stressors can be broad in spatial and temporal scale—for example, basin-level conditions in the northeastern Pacific Ocean—and acute and localized, such as conditions at the tributary reach level. Stressors can be sequential and applied at each relevant life stage, and multiple stressors can occur simultaneously within a single life stage.

Life cycle models represent one method for evaluating and integrating effects from multiple stressors across space and time. The life cycle model framework can be used to apply isolated effects acting on a particular life stage, and it can accommodate cumulative effects across several life stages.

The following is an application of a life cycle simulation model framework developed to evaluate population-level responses to environmental and management actions of Endangered Species Act-listed spring-run Chinook salmon in the Wenatchee River basin (USOFR 1999). This modeling benefits from and builds upon other life cycle modeling efforts directed in this basin (ICTRT and Zabel 2007, Honea et al. 2009, 2016). Development of this model occurs through a collaboration between state and federal biologists and local recovery planners.

6.2 Methods

We describe below the methods used in our analysis. First, we describe the life cycle model. Next, we discuss translation of a set of completed and anticipated future habitat projects into parameter changes to estimate their potential effects, and then explore the parameter set in the life cycle model. Then, we briefly outline the calibration process, and follow that with an explanation of several scenarios included in this analysis.

6.2.1 Life cycle model

The Wenatchee River spring-run Chinook salmon life cycle model (LCM) is an age-structured, stage-based, spatially explicit population viability model with stochastic elements (Figure 6-1). Much of the following overview comes from several reports (Jorgensen et al. 2013, 2017) describing the model that have been reviewed by the Northwest Power and Conservation Council's Independent Scientific Advisory Board (ISAB 2013, 2017).

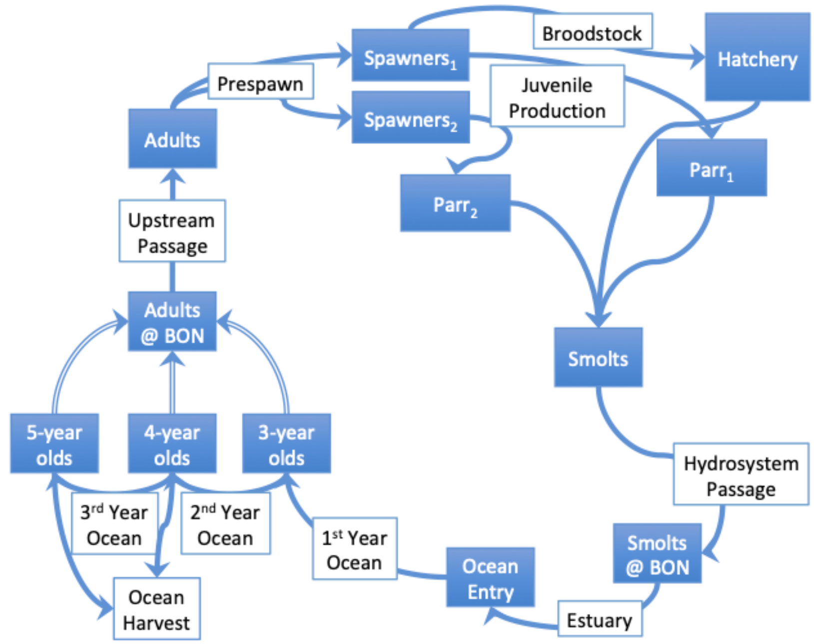

Figure 6-1. Diagram of the Wenatchee River spring-run Chinook salmon life cycle model. Boxes with numbered subscripts denote representation in the model of multiple spatial units (i.e., tributaries) contributing to fish production in this population.

This LCM functions similarly to the Leslie-style matrix structure (Leslie 1945). In that traditional formulation, there are two main elements—an array of abundance and a transition matrix. The abundance array for a simple LCM following this type of framework would be:

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

The 5×1 abundance array 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 abundance array includes the number of fish at each life stage for each time step, t. The number of individuals at the next time step, $t + 1$, would be calculated by multiplying $\mathbf{N}(t)$ by a 5×5 transition matrix, $\mathbf{A}(t)$:

$$\boldsymbol{N}(t+1) = \mathbf{A}(t) \cdot \mathbf{N}(t).$$

The dimensions (5×5) of the transition matrix, $\mathbf{A}(t)$, reflect the five age classes incorporated into the model; its entries can be fixed values or change with each time step, t. The transition matrix, $\mathbf{A}(t)$, for this Leslie matrix model would take this form:

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

It 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 (s_u; see Table 6-1 for parameters and their values), survival after harvest ($1 - h_r$), and survival to spawning (s_{sb}). Fertility is denoted by the F_i terms. s_2 is the survival of parr (moving from one-year-old fish to two-year-olds), which includes overwinter 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.

This simplified LCM form was the basis for the ICTRT and Zabel (2007) life cycle model and from which the below-described life cycle model comes. In the following sections, we highlight some changes made to the Leslie matrix format and its inputs.

6.2.1.1 Spatial structure

To represent major fish production areas as distinct entities with their own unique characteristics and to account for hatchery production (Figure 6-1; Jorgensen et al. 2013), the abundance array, $\mathbf{N}(t)$, has been modified to include fish production as discrete spatial units,

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

where each $n_{x,y}$ element reflects ages (row) of fish originating from a specific subbasin production area (column). Maturing fish that originated in a particular subbasin (column) return to that subbasin (a few exceptions to this rule are detailed in sections below). Hatchery programs (subscript h) are included and tracked by program type and objective for up to k hatchery program types (conservation or safety-net objectives). 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, additional parameters transition fish across life stages. The additional parameters are applied to each subbasin, j, or hatchery, h. In some cases (e.g., adult maturation rates, upstream survival, fertility, and hydrosystem and ocean survivals), these are the same and shared among subbasins. In other cases (e.g., the unique characteristics of fish production areas), these are different to capture the unique characteristics of a subbasin or hatchery objective.

Table 6-1. Parameters used for the Wenatchee River spring-run Chinook salmon life cycle model for major production areas, which included Chiwawa River, Nason Creek, and 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] (included in s_2)	Drawn from a distribution	Drawn from a distribution	Drawn from a distribution
Hydrosystem survival (included in s_2)	Proposed action	Proposed action	Proposed action
s_3 (first ocean year)	Stochastic variable	Stochastic variable	Stochastic variable
s_o (ocean survival for years after s_3)	Drawn from a distribution	Drawn from a distribution	Drawn 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_{pinn} (predation by pinnipeds)	Scenario-dependent	Scenario-dependent	Scenario-dependent
s_u (Bonneville Dam-to-basin survival rate)	Drawn from a distribution	Drawn from a distribution	Drawn from a distribution
s_{sb} (in-basin prespawning survival rate)	Drawn from a distribution	Drawn from a distribution	Drawn from a distribution
Initial abundance of 4- and 5-year-old tributary spawners used to initialize the LCM (geomean of 2008–14)	406	148	38

[a] Chiwawa River production estimates included a Box–Cox transformation as a way to deal with the heteroscedasticity in the data (Zabel et al. 2006, ICTRT and Zabel 2007).
[b] Parr–smolt survival accounts for the period from the parr stage to the smolt stage upon exiting the Wenatchee River basin.

Table 6-2. Parameters used in the Wenatchee River basin spring-run Chinook salmon life cycle model that simulated proportionate natural influence (pNI) guidance for operations of the hatchery programs in the Chiwawa River and Nason Creek.

Parameter	Chiwawa River	Nason Creek
Maximum number of hatchery fish allowed to spawn in the wild in the absence of natural-origin fish (NOR); decreases linearly with increasing NOR abundance to the NOR cutoff threshold	200	200
NOR cutoff threshold: Abundance at which no hatchery fish are allowed to spawn in the wild	1,000	1,000
Hatchery-origin fish (HOR) domestication discount: Annual domestication discount as calculated from a 25-yr running mean of pNI; decreases linearly with increasing pNI	0.60–0 (pNI-dependent)	0.60–0 (pNI-dependent)

6.2.2 Life cycle model inputs and Proposed Action elements

In the previous sections we outlined the model structure and its spatial scope. In this section we provide additional detail about some of the other parameters used in the life cycle model (Tables 6-1 and 6-2), including hatchery effects.

6.2.2.1 Parr capacity

NWFSC's Watershed Program has developed methods to characterize summer parr capacity as a function of geomorphic habitat classes (Bond et al. 2019). Below, we briefly describe that method and how parr capacities were changed as a consequence of implementing completed habitat restoration projects and as a consequence of proposed future habitat projects. More exhaustive detail of parr capacity estimation is in Pess and Jordan (2019).

Habitat projects

We attempted to quantify the biological benefits to Wenatchee River basin ESA-listed spring-run Chinook salmon from freshwater habitat restoration actions completed in 2009–18 (Figure 6-2). The process consisted of linking projects completed during this period to estimated changes in physical habitat, which, in turn, influenced changes in capacity and were used as inputs to the life cycle model (Pess and Jordan 2019). Next, we estimated potential effects from proposed future projects associated with the proposed action (Table 6-3).

Completed projects

The Upper Columbia Salmon Recovery Board (UCSRB) maintains a searchable online database that includes all upper Columbia River basin restoration projects (the Habitat Work Schedule[1]), and has verified project data and information in the database for projects completed up through 2018 (G. Maier, UCSRB, personal communication). The list of projects includes not only those directed at changing habitats (e.g., 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 (e.g., 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 our modeling, we focused on projects from the Habitat Work Schedule completed between 2009 and 2018—which we included in the new baseline—located in areas that contributed to the production of Wenatchee River basin spring-run Chinook salmon. For example, we excluded projects that self-reported that they targeted benefit for spring-run Chinook salmon if they were located in areas with essentially no contemporary occurrence of spring-run Chinook salmon, such as Chumstick and Peshastin Creeks. We also did not consider the effects of projects located in the mainstem Wenatchee River. While the mainstem is important for spring-run Chinook salmon, the focus of this analysis was to assess the 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

[1] http://hws.ekosystem.us/search

limited spawning in the upper Wenatchee mainstem (A. Murdoch, WDFW, unpublished data), and there is uncertainty about whether Wenatchee River mainstem juvenile rearing capacity in any season is limiting. In the absence of quantifiable evidence, we assumed for this analysis that the mainstem Wenatchee River was 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. Changes to streamflow, riparian plantings, conservation easements, and land purchases to prevent further development—projects designed to protect intact habitats—are all important for species conservation; however, because they were not currently quantifiable as capacity changes, these types of projects fell outside of the scope of this analysis.

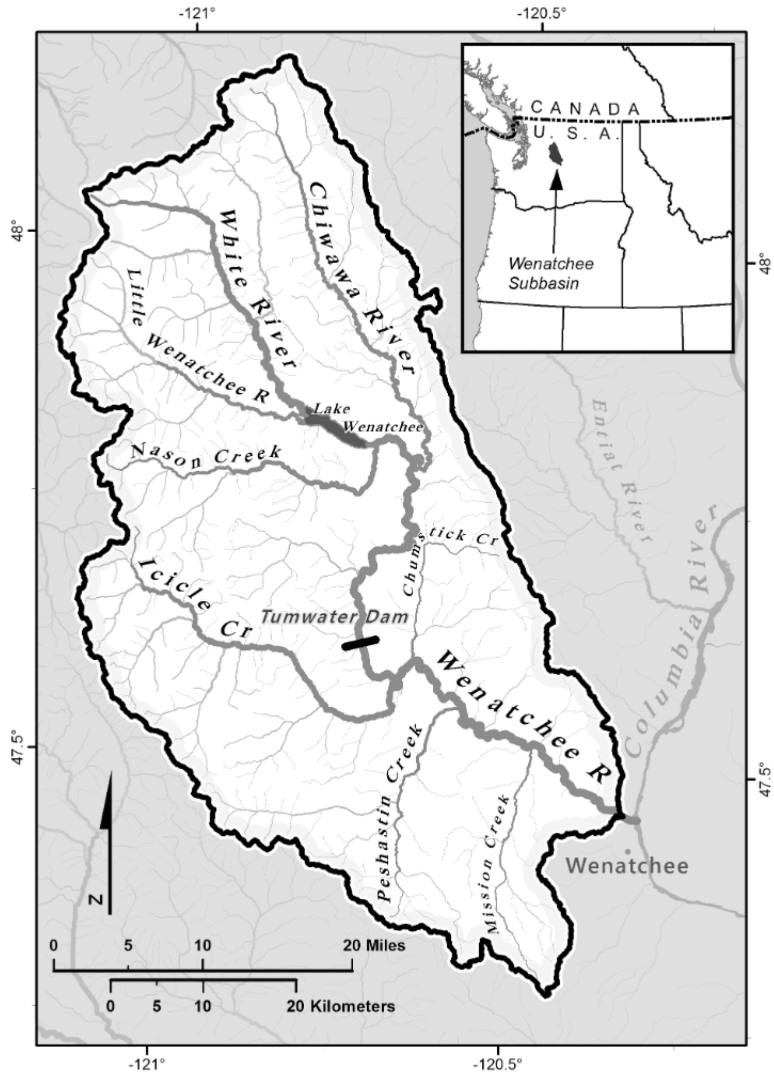

Figure 6-2. Map of the Wenatchee River basin. Natural production of spring-run Chinook salmon occurs primarily in the main tributaries above Tumwater Dam: Chiwawa, White, and Little Wenatchee Rivers, and Nason Creek. Map by D. Holzer, NMFS/NWFSC.

Completed projects with estimable benefits

To estimate the benefits of selected projects, we focused on physical habitat changes resulting from projects completed during the 2009–18 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 was targeted—because overall life stage survivals are composed of incremental survivals across the spatial domain occupied throughout a life stage. It would be difficult to partition how any one particular location or moment contributes to that survival within the timeframe of a life stage, making the influence of a project's benefits on survival difficult to assess.

However, physical changes to the landscape can be quantified in terms of physical space available and its quality or suitability. Through the framework of Bond et al. (2019), we can assign assumed capacity parr densities to restored and unrestored hydrogeomorphological features throughout the Wenatchee River basin. We made the assumption that a project's benefits can be estimated to affect capacity through Bond et al.'s (2019) estimation of capacity from the summation of fish density per habitat type, and the expected resulting habitat type as a consequence of implementation of each project. In addition, we reiterate that if a project had benefits resulting in changes in survival, we do not currently have a method to capture survival changes. Thus, we may not have captured all of the potential benefits attributable to a habitat project or combination of projects.

Given our approach to ascribe project benefits based on geomorphic changes and given the 2009–18 time window of project completion, we identified four projects that qualified:

1. CCFEG Large Wood Atonement White River, 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 in-river wood retention rates.[2]
2. CCNRD Nason Creek Lower White Pine Reconnection Project, an oxbow reconnection in Nason Creek. This is 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.[3]
3. CCNRD Nason RM 2.3 Side Channel Reconnection, a side channel/oxbow reconnection in lower Nason Creek; it was previously only connected to the main channel at high flow, but is now estimated to be activated at 1–2-yr flows.[4]
4. CCNRD Nason Creek Upper White Pine Floodplain Restoration RM 13.3–13.8, converting a diversion channel in Nason Creek into a meandering stream channel and connecting the channel to adjacent areas modified to function as floodplain.[5]

Below, we describe the process of ascribing benefits to capacity by the four projects (Table 6-3).

CCFEG Large Wood Atonement White River

To estimate potential change in Chinook salmon parr rearing capacity resulting from the CCFEG Large Wood Atonement White River project, we used an existing model of Columbia River basin floodplain habitat (Bond et al. 2019) to estimate juvenile capacity change . Briefly, the Bond et al. (2019) model was 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. To estimate capacity change for the projects, including this one, we used existing estimates of parr densities for geomorphic habitats found in each project location,

[2] http://waconnect.paladinpanoramic.com/project/290/16940
[3] http://waconnect.paladinpanoramic.com/project/290/14462
[4] http://waconnect.paladinpanoramic.com/project/290/80392
[5] http://waconnect.paladinpanoramic.com/project/290/14463

Table 6-3. Estimated (prebaseline; Bond et al. 2019) and additional estimated parr capacity for Wenatchee River tributaries (number of parr and percent change) as a consequence of completed actions and adding to the baseline period (2009–18) and for the Wenatchee River basin's estimated share (Table 6-4) of future anticipated projects (2021–36) under NMFS (2020)'s proposed actions for the Upper Columbia River Chinook Salmon ESU allocated to three Wenatchee River tributaries.

	Projects, 2009–18			Proposed projects, 2021–36		
Tributary	**Prebaseline capacity**	**Project change**	**Change incorporated into baseline**	**Complexity**	**Access (75th %ile)**	**Change from baseline**
White River	440,007	7,605	1.7%	4,427	11,166	3.5%
Chiwawa River	438,756	—	—	2,204	11,134	3.0%
Nason Creek	252,630	19,188	7.6%	7,688	6,411	5.2%

including for the White River. These estimates assigned parr capacity densities to bar edge, bank edge, and midchannel habitat areas separately. Each of these habitat areas were estimated from the modeled wetted width of each 200-m stream segment. We assumed a linear relationship between the width of edge habitats and stream width.

The potential effect of large wood additions on juvenile capacity of the lower White River was calculated from a multistep process. We estimated the area of the wood installations (consisting of either pile arrays 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 parr/m^2) compared to wood-free banks (0.33 parr/m^2).[6] Based on the project plan's specification for the structures, we estimated that each of 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. 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 specifications—we examined the sites using 2014 satellite imagery (roughly a year after project completion). We could clearly view 11 of the sites, each of which had accumulated wood, and were able to calculate their areas. The mean area of the 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 7,605 additional parr capacity from the full project post-implementation.

CCNRD Nason Creek Lower White Pine Reconnection Project

To assess the potential change in Chinook salmon parr rearing capacity resulting from this project, we used the same Bond et al. (2019) capacity model to estimate the change in wetted floodplain habitat resulting from the reconnection. 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 project, we estimated that an additional 5,058 m^2 of usable Nason Creek side channel floodplain habitat could be created—3,035 in additional parr capacity.

[6] T. Beechie, NMFS/NWFSC, unpublished data from Skagit River assessments.

CCNRD Nason RM 2.3 Side Channel Reconnection
Capacity change was estimated in a similar manner as the previous projects, and we estimated that it could result in approximately 6,116 m^2 of side channel habitat from the mainstem creek. We estimated the increase of summer parr capacity to be 3,670.

CCNRD Nason Creek Upper White Pine Floodplain Restoration RM 13.3–13.8
For the final completed project, which was to add a meander to a confined channel and reconnect to adjacent areas modified as floodplain habitat, we estimated that ten 200-m reaches would be reconnected with their floodplain as a result of this project. We estimated that side channel habitat created from this reconnection could potentially increase summer parr capacity by an additional 12,483.

The habitat capacity modeling did 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 found in other Columbia River basin reaches. Finally, we did not model any potential change to mainstem Nason Creek habitats that may result from this side channel reconnection, although there may be associated improvements in mainstem habitat suitability.

Proposed future projects
In addition to estimating effects from completed projects for the new baseline, we estimated additional effects anticipated for future projects (the 2021–36 period) in the proposed action (Table 6-3). We apportioned anticipated future project effort in the Upper Columbia River Chinook Salmon ESU to the three extant populations according to their proportions of total capacity (Table 6-4).

The proposed upper Columbia River habitat actions for 2021–36 included protected flow, flow enhancement, entrainment screening, access, stream complexity, and riparian habitat (NMFS 2020). The Bond et al. (2019) habitat model can only assess the benefits of access and increased complexity as they pertain to increased juvenile rearing capacity. Although the other proposed action types are intended to provide benefits for salmonids, they were not quantifiable as changes to juvenile capacity and were excluded from this analysis. The proposed actions' effort was not specified to particular populations in the Upper Columbia River Chinook Salmon ESU in NMFS (2020). Therefore, to assess benefits from access and increased complexity potentially directed toward the Wenatchee River basin, we made several assumptions. First, we allocated lengths of improved stream in each population-specific basin in proportion to the lengths of streams used by spring Chinook salmon in each of three basins (Entiat, Methow, and Wenatchee Rivers) within the Upper Columbia/East Slope Cascades MPG. The Wenatchee River basin comprises an estimated 45% of the currently available extant spring Chinook salmon habitat in the Upper Columbia River Chinook Salmon ESU (Table 6-4). Therefore, 45% of the stream length listed for proposed actions in the upper Columbia River were included in our Wenatchee River habitat change assessment. Second, based on locations of recent past projects, we assumed that future projects to benefit spring Chinook salmon would occur among three tributaries: White River, Chiwawa River, and Nason Creek. We also assumed that habitat capacity of reaches with new or improved access would be similar to currently accessible habitats in those tributaries. A further assumption was that increasing complexity would increase habitat capacity by the creation of new side channel or floodplain habitats, and that restoration would take place in areas that can increase floodplain width of habitats without the removal of urbanized land or paved roads. Therefore, habitat improvements were limited to reaches where floodplain could be increased by the removal of unimproved roads, rangeland, or cropland, and that no restoration would take place in reaches where additional floodplain gain would displace paved roads or urbanized areas.

Table 6-4. Proportion of spring-run Chinook salmon habitat in the three extant Upper Columbia River Chinook Salmon ESU populations' subbasins as measured by the proportions of stream reaches.

Population	Length of rearing habitat (m)	Proportion of ESU
Entiat River	109,295	0.12
Methow River	391,565	0.43
Wenatchee River	415,999	0.45

To estimate the Wenatchee River basin-specific benefits of improved access (Table 6-3), we estimated the total length in miles of access allocated to the Wenatchee River basin (i.e., 45% of all upper Columbia River improved access). Access lengths were converted from miles of improved access to the number of 200-m reaches, the smallest unit of habitat in the Bond et al. (2019) capacity model, by dividing the access length in meters by 200 and rounding to the nearest reach number (i.e., 5 miles of improved access for the extant Upper Columbia River Chinook Salmon ESU converts to 19 reaches in the Wenatchee River basin). We then randomly sampled the appropriate number of 200-m reaches from currently accessible tributaries of the White River, Chiwawa River, and Nason Creek, and summed their estimated current capacities to estimate a total benefit from the improved access. This random draw and summation process was repeated 500 times, and the average benefit of all draws was calculated to estimate the average benefit for each tributary. The draws of 200-m reaches were restricted to smaller tributaries, assuming opportunities to increase habitat access are in primarily smaller tributaries to the White River, Chiwawa River, and Nason Creek.

Similarly, we estimated the benefit from increased complexity (Table 6-3) from 500 random draws of a commensurate number of 200-m reaches totaling the length of habitat improvement. The estimated potential side channel habitat area for each randomly drawn reach and the resulting habitat capacity were added to the existing tributary capacity for a total benefit, summed by tributary. However, in contrast to estimation of improved access, when estimating the benefit of increasing complexity, we assumed that projects with a better-than-average benefit would be chosen instead of randomly selected, as restoration project site choices are guided by the Biological Strategy (RTT 2017), a process by which projects are chosen based on a synthesis of best-available information about fish needs and hypothesized basin impairments. Therefore, we chose the 75th percentile of benefit from the distribution of 500 draws from increased complexity, rather than the mean.

Incorporation of habitat effects in the LCM

Estimated effects from proposed future projects were added in three discrete stages, assuming that project implementation would occur in a stepped manner. We applied the first set of the estimated projects' effects at simulation Year 5, the second at Year 10, and the remaining estimated effects at Year 15. We attempted several other stepped applications of effects, such as all effects at simulation Year 1, or all at Year 15. However, because the magnitude of the estimated effects was relatively small and had no detectable effect on model outputs, there was no observable difference in any of the stepped-application methods.

6.2.2.2 Parr–smolt

The parr-to-smolt transition, s_2, includes three elements, parr-smolt overwinter survival (s_{ps}), migration survival through the mainstem Columbia River Public Utility District (PUD) and federal dams past Bonneville Dam, and the potential for avian predation. Overwinter survival to the smolt stage was drawn yearly from a distribution determined through a model parameter calibration routine, hydrosystem smolt migration through Bonneville Dam survival was determined from the COMPASS model (Zabel et al. 2008), and avian predation was assumed to be at the level observed in recent years.

As fish complete the smolt stage, they pass through the mainstem Columbia River. We applied the proposed action juvenile hydropower survival for LCM runs in this study. More assumptions about this alternative are described in Scenarios.

6.2.2.3 Ocean and pinnipeds

The ocean phase of salmon in the life cycle model encompasses estuary entry and life at sea. When smolts pass Bonneville Dam, they reach the estuary and can spend a variable number of years in the ocean. Survival during the first year in the ocean (s_3) was estimated from a model fitted to Wenatchee River 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 (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; Burke et al. 2017). First-year ocean survival was calculated by removing mortality estimated for subsequent ocean years from SAR. All subsequent survival in ocean years (s_0) was drawn from a distribution as determined through a parameter calibration process.

Maturation rates from the marine to the adult return stage were 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. These rates were set to match the observed age structure of adult returns.

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_{pinn}), from which the resulting mortality rates appear to have increased since 2012 (Sorel et al. 2017, in review). The survival of migrating adults from pinniped predation was calculated from the outcome of the ratio of the estimated survival rate from the more recent period divided by the estimated preceding survival rate (2013–14/2010–12).

6.2.2.4 Upstream

Survival from Bonneville Dam to the mouth of the Wenatchee River (s_u) was 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 was set to a constant value of 9% and represents an average of harvest rates estimated in recent years.

6.2.2.5 Spawners

Several life history events are applied in the life cycle model to adults that have migrated to the mouth of the Wenatchee River have not yet become spawners on the spawning grounds. First, a small number of fish migrating upstream in the Columbia River bypass the Wenatchee River, and some fish stray or disperse to nonnatal tributaries within the Wenatchee River basin above and below Tumwater Dam (5% and <3%, respectively; Murdoch, unpublished; Pearsons and O'Connor 2020). Those below Tumwater Dam were not considered to contribute to the population and are removed from the life cycle model. The rates of bypass and below-Tumwater Dam dispersal were applied only to hatchery-origin fish (HOR) 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, hatchery program directives stipulate that not all hatchery-origin returns are allowed to spawn in the wild. The yearly number of HORs passed upstream through Tumwater Dam and allowed to spawn is determined each year by established proportionate natural influence (pNI) guidance targets, natural-origin fish (NOR) abundance, and hatchery broodstock composition. The pNI guidance target governs the year-to-year proportion of hatchery-origin returns that spawn in the wild (pHOS; see next section for hatchery effects and Table 6-2 for parameters related to the hatchery programs). Third, all fish that are on the spawning grounds experience some level of prespawn mortality (s_{sb}), which was drawn yearly from a distribution as determined through a model parameter calibration process.

6.2.2.6 Hatchery effects

There are several hatchery programs operating in the Wenatchee River basin. For life cycle modeling purposes, we considered only the effects from conservation and mitigation-related hatcheries directed at supporting the ESA-listed spring-run Chinook salmon.

Chinook salmon conservation and mitigation hatchery programs in the Wenatchee River basin supplement the population and provide a safety net for years of very low returns. Relicensing of the mid-Columbia River PUD hydropower dams included a Habitat Conservation Plan (HCP). That plan allows for agreements for hatchery production for populations affected by the dams to be used to offset the impact of mortality caused by the PUD dams (HCP reports for PUD dam projects: Wells Hydroelectric Project FERC License No. 2149,[7] Rocky Reach FERC License No. 2145,[8] Rock Island FERC License No. 943[9]).

[7] https://douglaspud.org/wells-project/hcp
[8] https://www.chelanpud.org/docs/default-source/default-document-library/rr_hcp.pdf
[9] https://www.chelanpud.org/docs/default-source/default-document-library/ri_hcp.pdf

The LCM includes domestication effects as a consequence of the hatchery supplementation, to be able to evaluate population-level consequences of hatchery management operation strategies. The LCM tracks the annual numbers of natural- and hatchery-origin adults. All natural-origin adults are allowed to spawn, except for those collected for hatchery broodstock. Some hatchery-origin adults are allowed to spawn naturally, some are collected for broodstock, and the rest are removed from the system. Of the naturally spawning adults, in each annual model timestep, the LCM sets the proportions of total spawners that are of hatchery origin (pHOS) based on draft management guidelines (Jorgensen et al. 2017). The hatchery program guidelines are set with the goal to minimize, to the largest extent possible, adverse ecological and evolutionary impacts to natural-origin fish from supplementation actions (HGMP Chiwawa 2009, HGMP Nason 2009, HGMP Addendum 2010).

There appears to be a domestication effect on the population as a consequence of hatchery supplementation in the Wenatchee River basin: data from a long-term Wenatchee River-based study of relative reproductive success show that offspring from hatchery-origin fish that spawned naturally in the wild had decreased survival compared to offspring from natural-origin fish that spawned in the wild (Ford et al. 2014). A domestication penalty in the LCM is applied based on the findings from that study on the progeny of HORs as a function of the estimated 25-year (approximately five generations) running mean of an annual metric that approximates the strength of domestication selection as measured by the estimated pNI (HSRG 2009, Jorgensen et al. 2017).

6.2.3 Calibration

Before conducting prospective model runs, we calibrated the LCM to recent observations of the population and a recent no-action hydrosystem operation alternative. In response to a review of a prior LCM calibration routine (ISAB 2017), we switched to a rejection-sampling procedure which is among a class of methods referred to as Approximate Bayes Computation (Beaumont 2010, Csilléry et al. 2010, Hartig et al. 2011). In rejection-sampling, approximations of parameters' posterior distributions can be constructed through repeated LCM trials. In prospective simulations of a calibrated LCM, parameter values are drawn from these posterior distributions. What follows is a brief description of our calibration procedure.

The calibration procedure consisted of repeatedly drawing a set of parameter values from informative prior distributions (i.e., independent draws of parameters' values according to a random uniform distribution, from prespecified ranges for each parameter), running the model with the unique parameter sets, and comparing model outputs to empirical observations. Each unique parameter set was accepted (rejected) if it fell inside (outside) an acceptance level for deviation between model-generated and observed data. We defined the deviation as the Kolmogorov-Smirnoff (KS) statistic, D, which measured the degree to which the two distributions came from the same underlying distribution (Conover 1971). Because we had time series of observations of spawner abundance from redd counts and estimates of smolt outmigrant abundance from smolt trapping, there were two life stages with which to calibrate the LCM. We compared these recent observations to the LCM outputs so that the model would be calibrated to current conditions. The KS tests consisted of calculating the two-sample, two-sided D statistic of a comparison of two distributions:

1) the distribution of spawning adults from a 100-year simulation, and 2) the distribution of recent (2005–14) estimates of spawner abundance from the Salmon Population Summary database.[10] We repeated this procedure of calculating a *D* statistic for the comparison of the distributions also for the LCM-generated smolts and for the distribution of estimated Wenatchee River basin smolt abundance from recent years (Murdoch, unpublished). These two *D* statistics were calculated for each of the 10,000 model iterations. Because we had two *D* statistics from each model iteration (one for the spawner distribution comparison and one for the smolt distribution comparison), we combined them in each model iteration by calculating the sum of the squares of the two statistics. We chose as our acceptance criterion the top 1% of the combined KS *D* statistics from the 10,000 iterations. This selection criterion allowed only those parameter sets that generated model outputs that were most closely aligned to observations.

6.2.4 Scenarios

The analysis in this report included the No-Action Alternative (NAA) developed for the recent NEPA EIS process, and variations on the hydrosystem operations Proposed Action (PA). The PA included a downriver juvenile migration survival through the hydrosystem for a preferred alternative as outlined in NMFS (2020). We note that the PA applied to the mainstem federal dam projects in the Columbia and lower Snake Rivers and not to the mid-Columbia River PUD dams, which were assumed to operate under their status-quo guidelines.

All of the variations of the PA included assumptions, detailed above, about effects of habitat projects in the baseline period (2009–18) and potential estimated effects from anticipated future projects. The variations of the PA consisted of differing assumptions about the potential carry-over effects of hydrosystem operations under the PA. As a way of expressing some uncertainty about the potential for the PA operations of the federal dams to influence potential carry-over effects as a consequence of juveniles migrating through the system to the estuary, we ran two additional PA variants where we reduced potential latent mortality of juveniles during their ocean-entry period. The percentages of release from latent mortality (17.5% and 35% reductions) reflected differing assumptions about the experience of juveniles navigating through the federal dams that were involved with the PA. Because Wenatchee River fish encounter four of the eight mainstem Columbia and Snake River federal dams, we applied half the amounts of reduced latent mortality levels attributable to the PA, but we kept the variant-naming convention (17.5% and 35% latent mortality reductions) to be consistent with reporting for other salmonid populations in both this report and NMFS (2020).

6.3 Results

The following describes results from our analyses of estimated effects of the Proposed Action, which included effects from habitat projects and hydrosystem operations with alternative assumptions of latent mortality (Figure 6-3; Tables 6-5 and 6-6). Compared to the NAA, the PA increased the number of natural-origin spawners. As release from latent

[10] https://www.webapps.nwfsc.noaa.gov/apex/f?p=261:HOME::::::

mortality increased, natural-origin abundance also increased. There was a substantial increase in the number of spawners when we added the number of hatchery-origin fish allowed to spawn naturally into the total spawner count.

Extinction risk, as measured via probability of falling below the quasi-extinction threshold (pQET), was very low at both QET levels across all scenarios, including the latent mortality assumptions we evaluated. Hatchery fish were not included in the estimated probabilities of quasi-extinction (Figure 6-3; Tables 6-6 and 6-7).

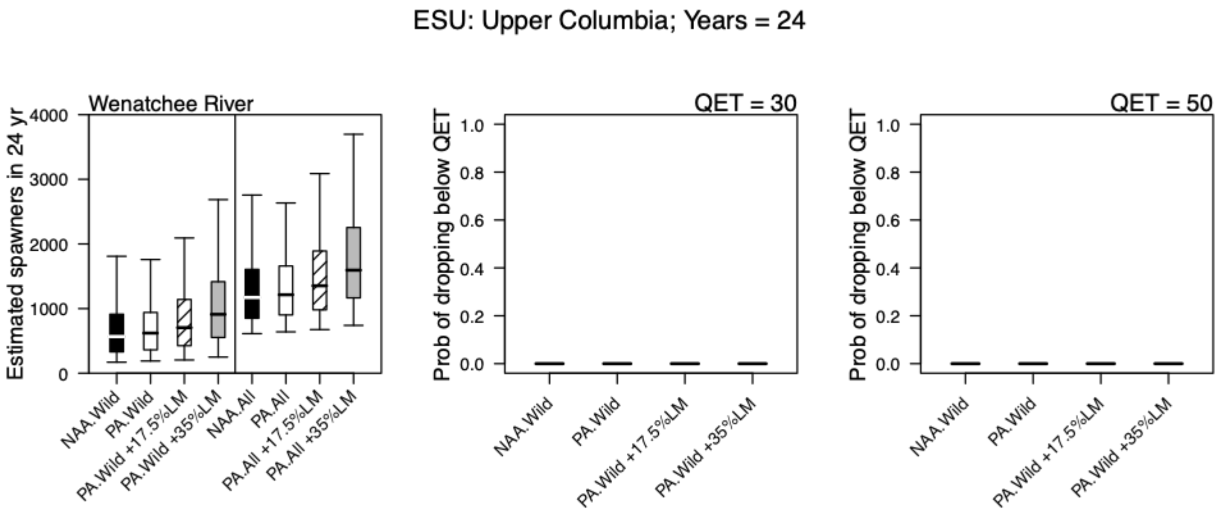

Figure 6-3. Population responses to the Proposed Action, including several assumptions about release from latent mortality (*LM*), for natural origin (*Wild*), and a combination of natural and hatchery origin (*All*), for the 10–24-year simulation period. Box edges represent the 25th and 75th percentiles, the bar in the middle represents the 50th, and the whiskers extend to the 5th and 95th percentiles.

Table 6-5. Abundance estimates for the No-Action (*NAA*) and Proposed Action (*PA*) alternatives, with several assumptions of release from latent mortality (*LM*). Abundance is shown for natural-origin (*Wild*) fish and for combined natural- and hatchery-origin fish (*All*).

	Percentiles						
Scenario	**2.5%**	**5%**	**25%**	**50%**	**75%**	**95%**	**97.5%**
NAA Wild	146	171	330	567	913	1,809	2,167
PA Wild	154	189	362	621	940	1,759	2,307
PA Wild + 17.5% LM	176	205	426	705	1,141	2,092	2,533
PA Wild + 35% LM	210	250	553	911	1,416	2,686	3,266
NAA All	561	612	850	1,172	1,607	2,756	3,142
PA All	594	640	902	1,214	1,659	2,634	3,299
PA All +17.5%LM	613	675	981	1,353	1,891	3,089	3,631

Table 6-6. Probability of falling below quasi-extinction thresholds for QET = 30 for the No-Action (*NAA*) and Proposed Action (*PA*) alternatives for simulation years 10–24, with alternative assumptions of release from latent mortality (*LM*).

	Percentiles						
Scenario	**2.5%**	**5%**	**25%**	**50%**	**75%**	**95%**	**97.5%**
NAA Wild	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PA Wild	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PA Wild + 17.5% LM	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PA Wild + 35% LM	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Table 6-7. Probability of falling below quasi-extinction thresholds for QET = 50 for the No-Action (*NAA*) and Proposed Action (*PA*) alternatives for simulation years 10–24, with alternative assumptions of release from latent mortality (*LM*).

	Percentiles						
Scenario	**2.5%**	**5%**	**25%**	**50%**	**75%**	**95%**	**97.5%**
NAA Wild	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PA Wild	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PA Wild + 17.5% LM	0.0	0.0	0.0	0.0	0.0	0.0	0.0
PA Wild + 35% LM	0.0	0.0	0.0	0.0	0.0	0.0	0.0

6.4 Discussion

The estimated magnitude of the benefits from the hydropower operations and anticipated future habitat projects as measured by LCM-estimated spawner abundance were slightly larger than recent observations of spawner abundance (621 vs. 535; LCM median of geometric mean number of natural origin spawners, and geometric mean of observed natural spawners 1998–2013, respectively). Spawner abundance of the Proposed Action was often more than the recently observed number of spawners in repeated LCM iteration trials, but the amount by which it was greater varied. Under alternative assumptions of release from latent mortality, the number of adult spawners could potentially increase substantially from current observations. Estimated effects from habitat projects alone were not detectable in repeated LCM trials. Furthermore, it did not matter whether all future habitat project effects were inserted at the beginning of a simulation (Year 1), staggered through the initial period of a simulation (Years 5, 10, and 15), or if they were inserted near the end of the initial simulation period (Year 15). The projected future actions were not large enough in magnitude to have any measurable effects on LCM outcomes.

Extinction risk, as measured by the probability of falling below the quasi-extinction threshold (pQET), was low regardless of the latent mortality assumptions and QET levels evaluated in this analysis, and it bears consideration in its interpretation. First, pQET was calculated for simulation Years 10–24. Had the time window been increased, extinction risk would be higher. Second, the low pQET can be largely attributed to the life cycle model's representation of the operations of the supplementation programs currently active in this basin. The input of conservation hatchery production substantially buffered extinction risk, because progeny of hatchery-origin fish spawning in the wild contributed to natural-origin spawners. In addition, the safety-net supplementation programs provided additional support when the conservation programs potentially fell short of broodstock targets. This added the potential for backup fish production (even while still subject to a domestication decrement), substantially reducing the natural-origin spawners' probability of falling below the quasi-extinction thresholds. Extinction risk would be considerably higher without ongoing supplementation from the hatcheries, as it is set up in the LCM to simulate the current rules and guidelines set forth by the co-managers of those programs.

Several aspects of anticipated effects as a consequence of the habitat projects required assumptions about their potential effects. These assumptions are explained in detail in Pess and Jordan (2019), and we list these ideas briefly here. We assumed the scope of the completed and future anticipated projects matched on-the-ground site design and project magnitude. Also, we assumed that the projects had the intended biological benefit of increased capacity per area changed. A further assumption was that the projects led strictly to benefits with no potential deleterious effects. Additionally, we assumed that projects sited above Lake Wenatchee had the intended effects without any potential for lake effects which could dampen projects' effectiveness (i.e., lake predators, capacity of the lake for rearing juveniles, etc.). While some projects were intended for enhancing Chinook salmon, they were directed in tributaries that are currently outside their domain of occupancy and were not included. If fish expand their range, these projects may provide some benefits in the future. Lower mainstem Wenatchee River projects that were directed toward reducing small water diversions, providing incremental increases in flow, were not included in the analysis. Lastly, the modeling framework did not account for projects that protect existing functioning habitats, such as conservation easements, nor the potential for habitat degradation through encroachment of future urban development. Natural riverscapes are dynamic, and both restored and unrestored stream reaches will continually change in their life stage-specific suitability at varying timescales, challenging our ability to capture a useful snapshot of rearing capacity under our current habitat modeling framework. Nonetheless, we feel that at the scale of the LCM, our habitat modeling provides an adequate assessment of the relative merit of larger (~km) stream habitat restoration projects.

Another important assumption implicit in the life cycle modeling results was reliance on stationarity of other factors. This analysis does not include potential effects as a consequence of climate change, which would be manifested in all habitats and during multiple life stages, including both freshwater and ocean conditions. Because ocean conditions are a major driver of LCM population dynamics, any changes in conditions for salmon—such as those modeled for Snake River spring/summer Chinook salmon populations (Crozier et al. submitted)—could have substantial deleterious impacts on upper Columbia River salmonid populations, including Wenatchee River spring-run Chinook salmon. Freshwater climate impacts could have potentially mixed effects on juvenile rearing (Crozier et al. submitted), and increased summer water temperatures during the upstream migration and spawning period could have severe consequences on survival (e.g., Bowerman et al. 2018). Another important stationarity factor was an assumption that there would be status-quo continuation of current operations of the mid-Columbia River PUD dams. Changes to PUD dam operations could have complementary or countering effects from operational changes at the lower Columbia River federal dam projects. All of these factors could have a substantial influence on Wenatchee River spring-run Chinook salmon, and demand further investigation.

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7 Identifying Interior Columbia River Basin ESU Focal Populations for Near-term Tributary Habitat Recovery Efforts

Thomas D. Cooney, Morgan H. Bond, Damon M. Holzer, Chris E. Jordan, Jeffrey C. Jorgensen, Martin Liermann, and George R. Pess

7.1 Introduction

Objective: *Identify focal populations for near-term emphasis in habitat restoration action planning. Goals of management actions are: 1) to avoid immediate (e.g., 24-year) losses in evolutionarily significant unit (ESU)/major population group (MPG) capabilities to withstand demographic and localized catastrophic risk factors, and 2) to make progress toward longer-term goals for Endangered Species Act (ESA) and broad-sense recovery.*

The focal population concept is intended to complement or integrate ongoing ESA recovery implementation and related activities (e.g., ESA consultations involving tributary habitat) in the Columbia River basin. The main purpose of the focal population exercise is to provide strategic guidance for the sequencing of future habitat restoration and protection efforts at the population or MPG level. The focal population analysis is intended as a tool for use in strategic planning initiatives such as the Grande Ronde Atlas[1] and the Upper Salmon River MPG regional restoration planning effort. The framework described in this chapter was initially developed for application to the Snake River spring/summer-r Chinook Salmon ESU MPGs and their component populations; however, we have also developed a version for application to Snake River steelhead distinct population segments (DPSes).

The importance of sequencing or prioritizing restoration and recovery efforts over time has increasingly been gaining attention in the conservation literature. Examples include approaches to prioritize among sites in biological reserve planning (McBride et al. 2010, Wilson et al. 2011), considerations for maximizing the preservation and enhancement of inherent genetic diversity among populations varying in size (Willi et al. 2006, Aitken et al. 2013), and population size vs. environmental variation in metapopulation frameworks (Drechsler and Wissel 1998). These examples highlight the importance of explicitly considering how to maximize gains toward long-term objectives in light of starting conditions and inherent limitations on annual resources available for restoration activities in a defined period of time (e.g., 1–5 years). Another important consideration is the time for restoration actions to achieve desired improvements in habitat conditions, and the associated lags in benefits to fish. In many ways, the basic principles for these multipopulation-level sequential planning strategies parallel advice regarding within-population protection and restoration (Beechie et al. 2010).

The Snake River Recovery Plan (NMFS 2017) describes a starting point (current status) and a desired end condition (ESU/DPS viability scenarios) in terms of individual populations. The recovery plan also catalogues key limiting factors, and identifies corresponding potential actions for each population. Status evaluations and ESA recovery objectives for

[1] https://grmw.org/

interior Columbia River ESUs and DPSes are organized around populations grouped into MPGs. This basic framework for hierarchical assessment at the ESU/DPS level is adapted and employed by regional technical recovery teams in all U.S. West Coast salmonid recovery domains. Evaluating ESUs and DPSes in this context supports consideration of not only the collective individual status of each population, but also allows for understanding the contribution each population makes to the next hierarchical level.

The Interior Columbia Technical Recovery Team (ICTRT) recommended MPG-level recovery criteria that were explicitly designed to provide for resilience against annual variation in environmental influences, opportunities for exchange with nearby populations in the event of short-term localized catastrophic impacts, the maintenance of major patterns of life history diversity, and adaptability to changing environmental conditions (ICTRT 2007b). At the MPG level, each set of population-specific recovery plans collectively contain the basic information needed to identify populations for immediate focus to support progress from current status toward long-term viability goals. Each management unit plan adopts an MPG recovery scenario that identifies target levels for component populations, whether they are considered viable or maintained. For each population, the management unit plans also outline key opportunities for tributary habitat protection and restoration that would contribute to improving populations toward those objectives.

The Snake River Recovery Plan also acknowledges that employing strategic approaches to implementing actions will enhance the potential for success in achieving and moving beyond long-term ESA recovery objectives. Opportunities to implement habitat protection and restoration actions will vary across populations, depending on the geomorphic setting and land ownership patterns. In many cases, restoration implementation will need to consider short-term limitations on available logistic or monetary resources. For some populations, there may be important sequencing considerations, e.g., particular habitat improvement opportunities that, if adequately addressed, would increase the potential benefits of subsequent actions aimed at other factors. As recovery progresses, the emphasis would be expected to broaden or shift to include the additional populations required to improve in status to meet or exceed their assigned viability objectives.

Considering short-term priorities for immediate focus of restoration activities is especially important for ESUs where all MPGs and their component populations are well below viability objectives—such as Snake River spring/summer Chinook salmon. Although almost all Snake River spring/summer Chinook salmon populations are rated at overall High Risk, the gaps to reduced risk status vary. Some of those populations may be exhibiting levels of natural production that, while below long-term targets, retain a substantial component of ESU-specific genetic diversity relative to much lower average level populations. Combined with habitat size/complexity and current abundance, the spatial arrangement of populations within MPGs is also an important consideration to targeting near-term actions. In the near term, assigning higher priorities to restoration/protection activities in current or potential “source” populations would benefit overall ESU recovery. Those populations could serve to bolster or even recolonize nearby populations (in the case of prolonged downturns in survival or chance localized catastrophic events) before their own recovery actions have a chance to take effect.

Another important consideration in sequencing the application of restoration resources is the relative vulnerability of populations to potential climate change impacts manifested in freshwater or ocean habitats. Information on the relative distribution of adult and juvenile life stages, combined with projected climate-induced changes in stream temperatures and river discharge (streamflows), can be used to identify relative vulnerabilities of freshwater habitats. Projected impacts of potential changes in ocean environmental conditions are much more uncertain, but sensitivity analyses suggest a wide range of potential impacts that would be common across most if not all populations within a particular ESU.

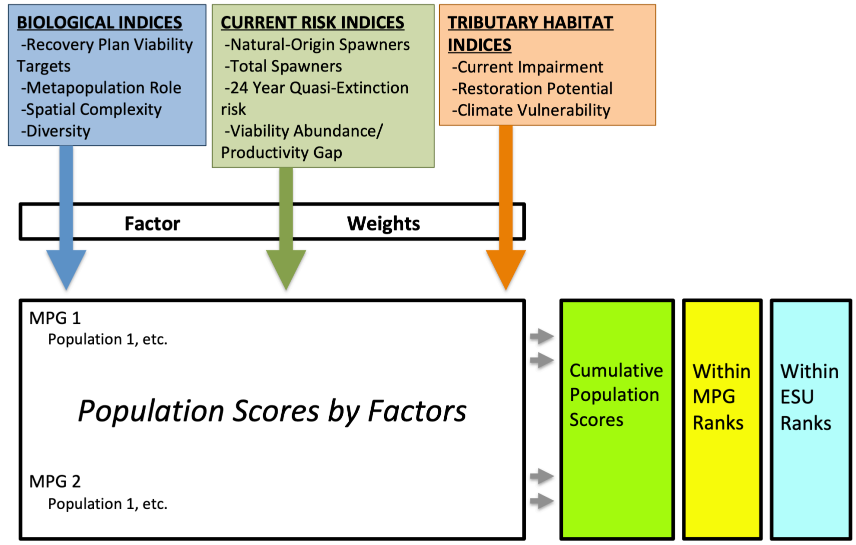

Figure 7-1. Focal population index components and output summaries (populations ranked within MPGs, across ESUs/DPSes).

7.2 Methods

Based on discussions in a joint NOAA West Coast Region (WCR)/NWFSC workgroup, criteria for determining focus populations should include:

- Biological and ecological indices (also identified as Viable Salmonid Population [VSP] characteristics): life history patterns, genetic characteristics, intrinsic potential (includes population size and complexity), and metapopulation characteristics.
- Current population status: quasi-extinction risks, current abundance relative to minimum thresholds, and supplementation contributions and recovery gaps.
- Relative habitat improvement potential.
- Climate change vulnerabilities.

Based on these recommendations, factors relevant to assigning a relative priority ranking to individual populations, within MPGs and across ESUs, are organized into three basic categories: biological and ecological indices, current population status, and habitat restoration potential (which includes climate change considerations). Within each category, sets of component indices feed into the identification of focal populations for tributary habitat efforts (Figure 7-1). The indices within these categories each comprise one or more metrics, weighted according to their relative uncertainty or direct relevance to current conditions. Indices included in this analysis are based on information that is either preexisting or anticipated to be available in the near future. The individual factor indices contribute to the overall score for a population. Output population scores are ranked at the MPG and overall ESU levels.

7.2.1 Index scoring

Populations are assigned a score on a scale ranging from 1 to 5 (where 5 represents a high potential for near-term restoration) for each of the indices, with the scores for a particular index being determined by applicable metrics (see Tables 7-1 through 7-3 for details). Indices for current status and habitat improvement potential are based on assigning higher relative weights to populations with moderately degraded tributary habitat rather than highly degraded; these populations have greater potential for improvements to meet viability targets in shorter time frames.

7.2.2 Component categories: Viable Salmonid Population indices

This category includes three factors that directly capture the considerations for MPGs recommended by ICTRT. These factors were adopted through Recovery Plans (Table 7-1), and are intended to result in a recovery strategy targeting a suite of populations within each ESU that would provide for protection against catastrophic and adaptive risk factors.

The Recovery Plan Viability Target index assigns higher weights via the target viability rating to populations selected from alternative MPG-level viability scenarios in current Recovery Plans (Table 7-1). These populations fulfill the ICTRT recommendations for having a sufficient number of populations spatially distributed across MPGs (the Metapopulation Role index), both to ensure that opportunities for recolonization are present and to increase the potential for continued adaptations to changing environments. This target includes a score for population size (the Spatial Complexity index), historical intrinsic potential (IP), which is assigned based on the ICTRT size category for a given population (example in Figure 7-2). The score increases from 2 for a Basic-size group population to 5 for Large or Very Large populations. The scoring reflects that populations with higher historical IPs tend to occupy more subwatersheds (i.e., Hydrologic Unit Code [HUC] 6s) and are less vulnerable to localized catastrophic loss and persistence risk. The target also includes a score for life history diversity (the Diversity index), assigning higher values to major patterns that were present historically, but that may be reduced across extant populations. These ratings thus reflect the determinations of major life history types.

Table 7-1. Biological and ecological index factors, data sources, and scoring criteria.

Index	Metric(s)	Source	Scoring
Recovery Plan Viability Targets	Target viability rating	ICTRT (2007b)	5: Highly Viable or Viable 3: Maintained 2: Reintroduction (extirpated)
Metapopulation Role	Population functional categories	Fullerton et al. (2016)	4: Source 2: Connector –1: Pseudo-sink
Spatial Complexity	Interior Columbia Technical Recovery Team (ICTRT) population size categories	ICTRT (2007b)	5: Very large 4: Large 3: Intermediate 2: Basic
Diversity	ICTRT major life history patterns	ICTRT (2007b)	4: Unique 3: Common

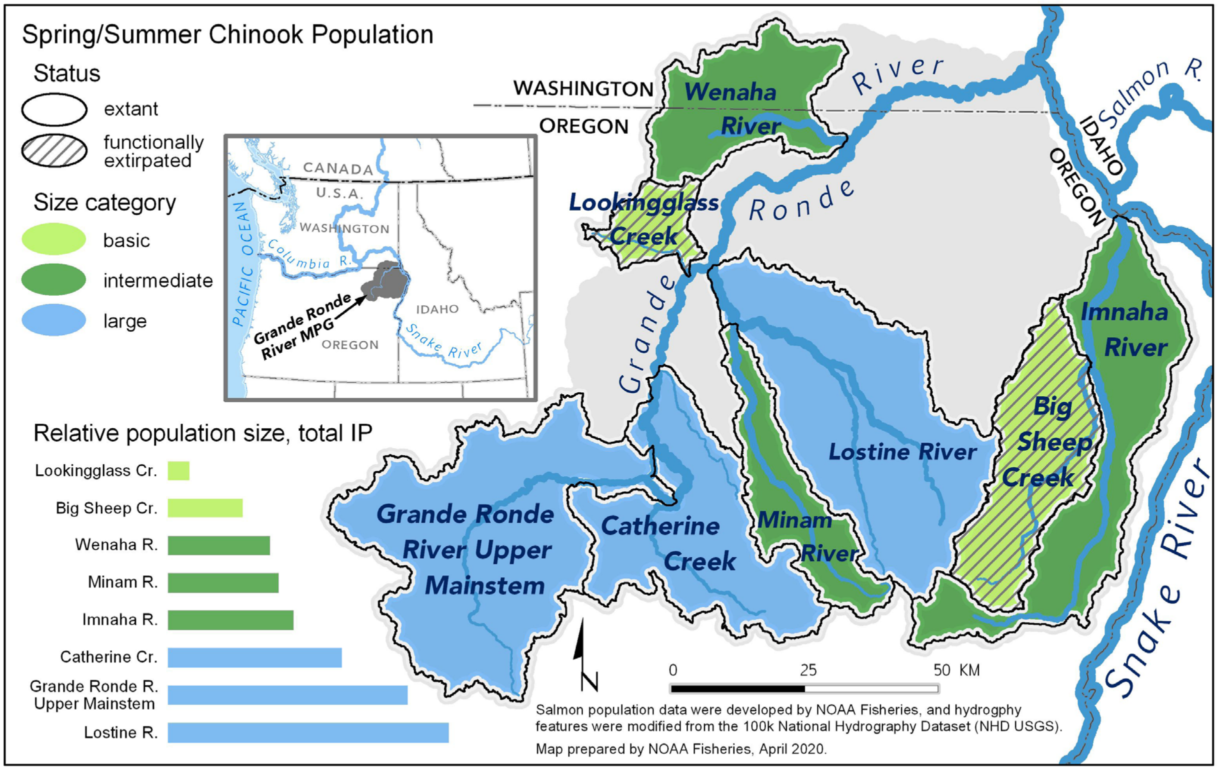

Figure 7-2. Map of the of the Grande Ronde River basin, associated Chinook salmon spawning populations, and their population size relative to their intrinsic potential (IP).

A second index, Metapopulation Role, captures results from a recent analysis of alternative metapopulation scenarios which highlighted particular populations based on genetic patterns, alternative distance/dispersal assumptions, or demographic patterns (Fullerton et al. 2016). Based on applying a set of metapopulation models, several MPGs in the Snake River spring/summer-run Chinook Salmon ESU were highlighted due to potential roles in a metapopulation framework. This includes large populations that may be acting as source populations, populations that could be important interconnecting stepping stones within a metapopulation framework, and populations that may be "pseudo-sinks" within such a framework. The populations highlighted by this effort were generally consistent with the application of ICTRT criteria, but resulted in increased emphasis on a subset of the populations based on the additional index.

7.2.3 Component categories: Current abundance, risk, and recovery indices

This category includes two indices that reflect current population status: natural-origin spawners and total spawners (Table 7-2). Current spawning abundance relative to ICTRT-recommended minimum abundance thresholds (MATs) determines each index. Recent geometric mean natural-origin abundance is expressed as a proportion of the MAT for a particular population. A variation on this index is also included for populations currently subject to ongoing local-origin broodstock supplementation programs that use the total adult spawning estimates (natural-origin + supplementation returns) in the geometric mean.

Table 7-2. Current abundance risk factors, data sources, and scoring criteria.

Index	Metric(s)	Source(s)	Scoring
Natural-origin Spawners (available for most populations)	Current ÷ MAT	SPS Database (2019)[a]	5: 0.50 to 0.90 × MAT
Total Spawners (available for direct supplementation populations)	Current ÷ MAT	SPS Database (2019)[a]	4: 0.25 to 0.49 3: <0.25 or >1.5 2: Insufficient data
24-year Quasi-extinction Risk (average across available values)	Detailed LCM (no supplementation, 14 populations)		5: 0 .06 to 0.29 4: 0.30 to 0.49 3: >0.50 or <−0.50
	Detailed LCM (with supplementation, 3 populations)	Zabel et al. (2013, 2017)	
	Simple Model (18 of 27 extant populations)	Buhle et al. (2018)	
	ICTRT hockey stick	ICTRT (2007b)	
Viability A/P Gap	ICTRT gap calculation	ICTRT (2007a)	5: 0.25 to 0.74 4: 0.75 to 0.99 3: 0.0 to 0.24, >1.0 2: >2.0 1: <0.0

[a] https://www.webapps.nwfsc.noaa.gov/sps

Current risk indices (Table 7-2) include projections of short-term quasi-extinction risk. The index for 24-year Quasi-extinction Risk reflects a composite of the most recent updates of population-level risk. For a given population, the projected risk level used in the index would be taken from the following list in order of priority: projections assuming current stage-specific survivals from detailed life cycle models, projections from the NWFSC intermediate population model, and projections from the NWFSC simple population model.

A number of different methods for projecting short-term extinction risks have been used or developed. The current set of indices includes existing metric sets and two variations of multiple population state-space modeling techniques. We anticipate that after the development and testing of an intermediate-level life cycle modeling exercise, short-term extinction risk assessments will be able to rely on a combination of detailed LCMs and the intermediate model results. There are several advantages to using LCMs to determine population status, including incorporating recent documented changes in life stage survivals into risk projections, as well as avoiding potential bias associated with fitting stock production functions only using adult-to-adult return data (e.g., estimating current low abundance productivity and capacity).

An index for the current abundance and productivity gap (Viability A/P Gap), as defined by ICTRT, compares the most recent paired estimates of current natural-origin abundance and ICTRT productivity against the viability curve corresponding to the designated role of the population in its recovery plan delisting scenario. This includes both a 5% risk for populations targeted to achieve viable or higher status, and a 25% risk curve for maintained status. This index roughly translates into the proportional change in either survival or limiting life stage capacity required to meet the viability target, assuming recent average environmental and ocean variations.

7.2.4 Component categories: Tributary habitat indices

Indices in this category (Table 7-3) include one assessment of the current level of tributary habitat impairment vs. full restoration potential, as well as indices that explicitly capture short- and long-term benefit potential. An index of relative vulnerability to climate change is also described in this category. Current tributary habitat impairment is expressed as an index based on empirically derived GIS assessments of current vs. historical conditions such as in-stream flow, average stream temperature, and stream structure across population reaches. Stream structure impacts are an important limiting factor on spawning and rearing stages in many Chinook salmon and steelhead populations in the interior Columbia River basin (Justice et al. 2017). The index for this element is based on metrics for two related conditions: floodplain connectivity/condition, and stream structure corresponding to pool habitat frequency and characteristics (e.g., reach proportion and pool depth statistics).

Restoration potential includes a habitat quality index, channel planform, and floodplain availability (Table 7-3). All three indices compare the current condition to a high-end or optimum condition in order to determine relative potential for recovery (Table 7-3). For example, floodplain availability is derived from land use and geomorphic indexing into classes (Bond et al. 2019). Sampling aerial photographs from sites randomly selected within each class was used to extrapolate estimates at the HUC 6 level across populations (Bond et al. 2019). For each population, we subtracted current from restored floodplain area and indexed the resulting differences, assigning a score from 0 (no potential increase) to 5, the maximum across populations.

Table 7-3. Tributary habitat impairment factors, data sources, and scoring criteria.

Index	Metric(s)	Source(s)	Scoring	Notes
Current Impairment				
Intrinsic potential	ICTRT Viability Report	ICTRT (2007b)	—	Population average
Flow	Summer minimum flow	Isaak et al. (2011)	—	Weighted by rearing areas
Temperature	Current average	Isaak et al. (2011)	—	Intrinsic potential weighted within current distribution
Stream structure	Floodplain connectivity	Moore et al. (2007)	—	Weighted by spawning and rearing
	Pool proportions	ODFW[a]		
Restoration Potential				
Expert panel habitat quality index	Current/high bookend	NMFS (2014)	—	Highest index value to midrange impairments
Channel planform	Current/optimum	Beechie et al. (2006)	—	—
Floodplain availability	Current/optimum	Bond et al. (2019)	—	—
Climate Vulnerability				
Temperature change	NorWeST current/2040	ICTRT (2007b)	5: GT 0.9 4: 0.66–0.89 3: LT 0.65	Index includes weighting with intrinsic potential and Justice et al. (2017) parr density vs. temperature
Flow change	—	Isaak et al. (2011)	—	—

[a] E. R. Sedell, ODFW, unpublished data.

7.2.5 Component categories: Climate vulnerability

We used the NorWeST extrapolated stream temperature estimates as the basis for climate vulnerability (Isaak et al. 2017). For each population, we calculated a weighted temperature corresponding to current-use reaches using intrinsic potential area. Potential temperature impacts associated with climate change were generated from NorWeST's projected 2040 maximum weekly maximum stream temperatures (MWMT; 1-km reaches) and the MWMT Chinook salmon parr density function from Justice et al. 2017. We assigned the highest scores to populations where there was either no change in the proportion of current area exceeding 18°C, or where change in area was less than 15%. The index represents the relative change from current in weighted parr intrinsic potential for current-use areas. We assigned a score from 1 to 5 based on the resulting index for each population. If the 2040 area weighted temperatures were less than 55% of current, the index value was 2. The 2040 temperature change proportions between 0.55 and 0.69 were assigned a value of 3. Temperature change proportions between 0.7 and 0.74 were assigned a value of 4, and indices above 0.75 were rated as 5.

7.2.6 Prioritizing focal salmon populations

Each focal salmon spawning population has an aggregate tributary habitat score that combines biological and ecological indices, current population status, the current and potential habitat improvement opportunities associated with tributary habitat, and the associated climate vulnerability (Table 7-4). The aggregate score can then be used to rank each population within an MPG as well as throughout an entire ESU (Figures 7-3, 7-4).

Table 7-4. Combined indices to determine tributary habitat aggregate scores for ranking Chinook salmon populations within an MPG or ESU.

Index	Equation(s)	Comments
Biological and Ecological	Recovery plan viability role + spatial complexity + diversity contribution	Spatial complexity incorporates size of spawning population.
Current Status	Natural-origin abundance + current short-term risk + recovery gap	Current short-term risk is based upon average short-term extinction risk. Recovery gap is based upon applied recovery plan scenarios. Values for each are based upon a lookup conversion table.
Tributary Habitat Opportunity	Current habitat impairment + Short-term habitat potential + Long-term habitat potential Current habitat impairment = Expert panel habitat quality index (HQI; high bookend vs. current) Short-term habitat potential = Flow impairment + (0.5 × stream structure impairment) Long-term habitat potential = Temperature impairment + (0.5 × stream structure impairment)	Values for HQI are based upon a lookup conversion table. Restoration is not included in score.
Climate Vulnerability	—	—
Total Score	Biological and Ecological + Current Status + Tributary Habitat Opportunity + Climate Vulnerability	

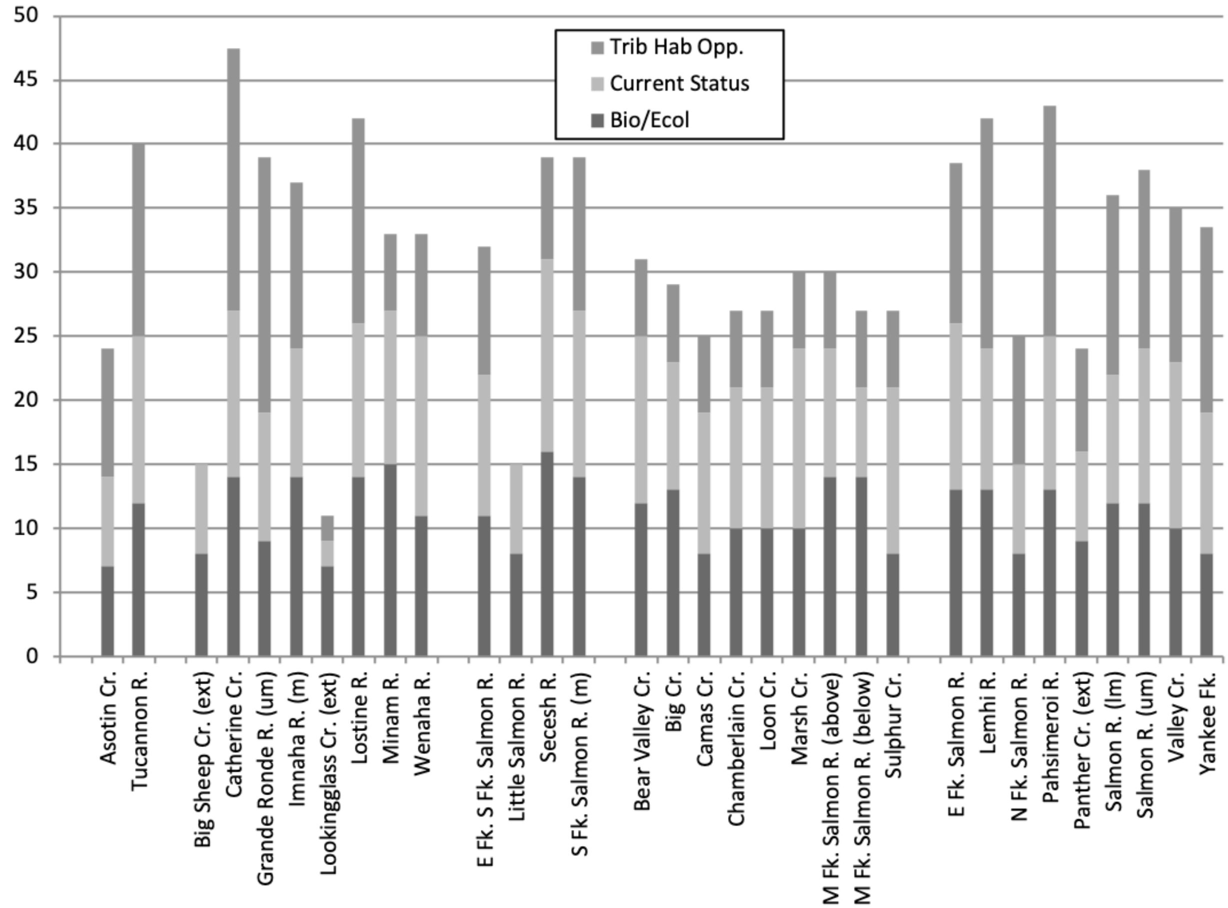

Figure 7-3. Population index values in the Lower Snake, Grande Ronde/Imnaha, South Fork Salmon River, Middle Fork Salmon River, and Upper Salmon River MPGS. Component factors from dark to light gray: biological and ecological; habitat restoration potential composite; 24-yr QET risk. Populations are organized by MPG. Key: *(above)* = above Indian Creek, *(below)* = below Indian Creek, *Cr.* = creek, *E* = east, *(ext)* = extirpated, *Fk.* = fork, *(lm)* = lower mainstem, *M* = middle, *(m)* = mainstem, *N* = north, *R.* = river, *S* = south, *(um)* = upper mainstem.

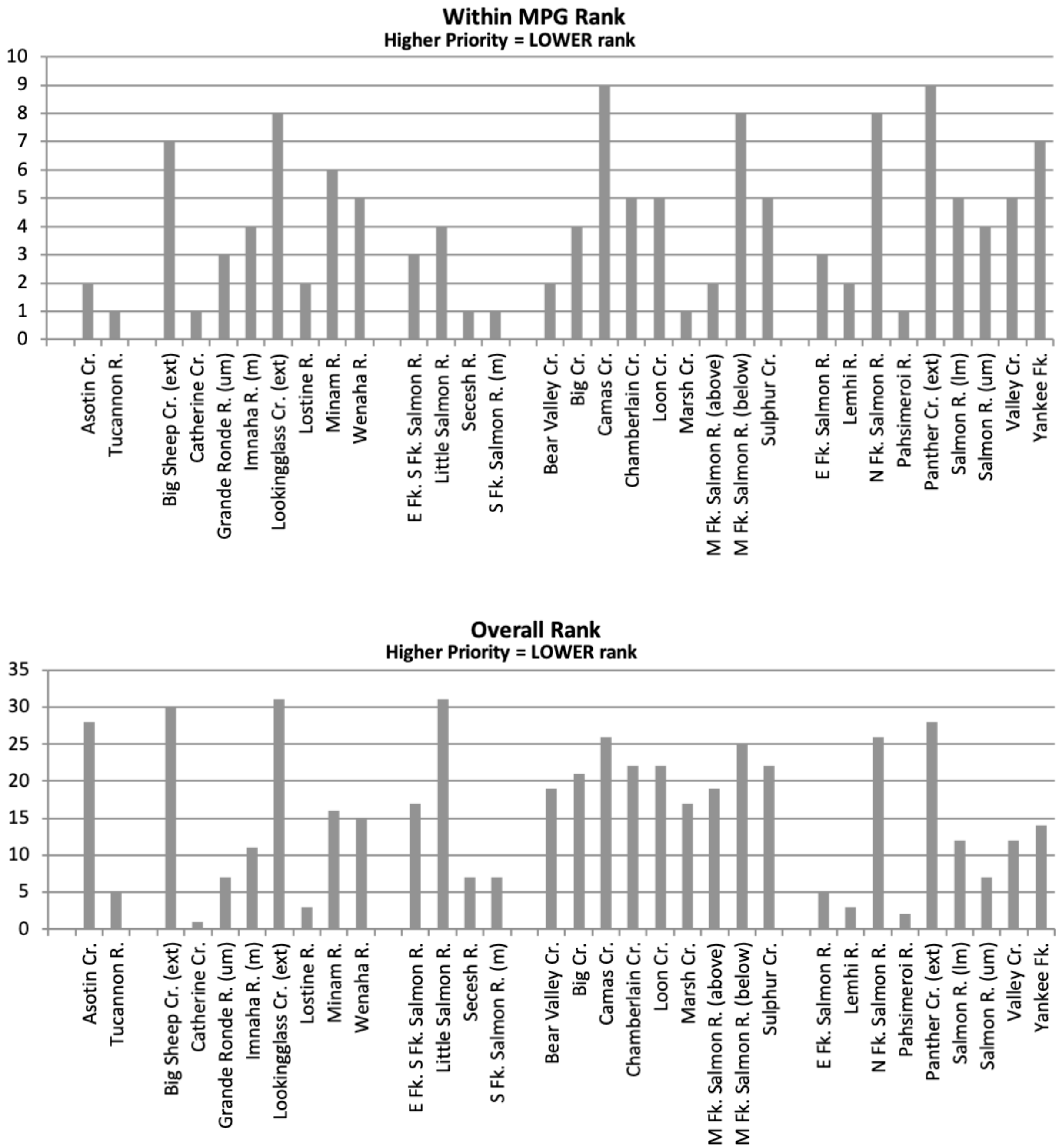

Figure 7-4. Snake River spring/summer-run Chinook Salmon ESU, focal population index rankings. Top panel: Within-MPG rank. Bottom panel: Within-ESU rank. Key: *(above)* = above Indian Creek, *(below)* = below Indian Creek, *Cr.* = creek, *E* = east, *(ext)* = extirpated, *Fk.* = fork, *(lm)* = lower mainstem, *M* = middle, *(m)* = mainstem, *N* = north, *R.* = river, *S* = south, *(um)* = upper mainstem.

Specific metrics used to quantify each of the general indices are identified in Tables 7-1 through 7-3, which also indicate their associated scoring. As with any semi-quantitative scoring system, there are instances when multiple populations have the same scores. However, such a system does help differentiate the most important populations and can be used to determine their overall resiliency to differences in how scores are aggregated under varying assumptions associated with the equations.

7.3 Results and Discussion

Outputs of applying this framework to the Snake River spring/summer-run Chinook Salmon ESU populations, organized by MPG, are provided in Tables 7-5 and 7-6. The summary results include composite population scores (Figure 7-3). Relative rankings at the MPG level (for illustration, the top scoring population and the next three highest-ranked populations for immediate restoration potential are highlighted) and the “top 10” ranked populations across the ESU are summarized in the tables and depicted in Figure 7-4. Populations ranked 1 through 4 in each MPG generally reflect larger, more spatially complex populations with moderate to high habitat restoration potential.

The rankings within MPGs were relatively insensitive to dropping individual component indices from the analysis (Tables 7-7 and 7-9) or doubling the weights for individual components (Tables 7-8 and 7-10). Recalculating the indices after dropping individual components or combinations did not change the population identified as top-rated in 4 out of 5 of the MPGs. Under that same iteration, the top-rated population in the Middle Fork dropped in ranking to #4, and both Big Creek and Marsh Creek populations improved in rankings. Removing the influence of the Recovery Plan Viability Target and Metapopulation Role indices did result in a switch between the populations ranked 2 and 3 in the Grande Ronde MPG (Table 7-7). With one exception, the sensitivity analyses doubling each component one at a time did not change the populations ranked 1–4 within each MPG, although there were some shifts in relative rankings within the top 4 (Tables 7-8 and 7-10).

The process of identifying focal populations from a distillation of quantitative data is new and unique for the spatial extent and number of populations ranked here. Our aim is to move prioritization away from a qualitative, expert-panel process to one that uses many disparate data sources to provide the most holistic, quantitative assessment possible. A primary advantage of this process is that it can easily be updated as new data become available or new populations are added. Ultimately, this process can be expanded to include all other ESUs (e.g., Upper Columbia River Chinook Salmon) in the Columbia River basin. However, even within the existing Snake River basin rankings, continued improvement in spatially extensive estimates of habitat quality, impairments, and restoration opportunities will continue to improve the utility of these priority population rankings.

Table 7-5. Focal population index and ranks. Lower Snake, Grande Ronde/Imnaha, and South Fork Salmon River MPGs. Key: *Cr.* = creek, *E* = east, *(ext)* = extirpated, *Fk.* = fork, *(m)* = mainstem, *R.* = river, *S* = south, *(um)* = upper mainstem.

Population	Bio. & Ecol.	Status	Subtot.	Status: MPG Rank	Status: ESU Rank	Tributary Opp.	Climate Vuln.	Overall Score	Within-MPG Rank	ESU Rank
Asotin R.	7	7	14	2	12	10	3	27	2	28
Tucannon R.	12	13	25	1	6	15	3	43	1	5
Big Sheep Cr. (ext)	8	7	15	7	11	—	3	18	7	30
Catherine Cr.	14	13	27	1	2	20.5	3	50.5	1	1
Grande Ronde R. (um)	9	10	19	6	10	20	3	42	3	7
Imnaha R. (m)	14	10	24	5	8	13	3	40	4	11
Lookingglass Cr. (ext.)	7	2	9	8	13	2	3	11	8	31
Lostine R.	14	12	26	3	5	16	3	45	2	3
Minam R.	15	12	27	1	2	6	3	36	6	16
Wenaha R.	11	14	25	4	6	8	3	37	5	15
E Fk. S Fk. Salmon R.	11	11	22	3	9	10	3	35	3	17
Little Salmon R.	8	7	15	—	—	0	3	14	4	31
Secesh R.	16	15	31	1	1	8	3	42	1	7
S Fk. Salmon R. (m)	14	13	27	2	2	12	3	42	1	7

Table 7-6. Focal population index and ranks. Middle Fork Salmon River and Upper Salmon River MPGs. Key: *(above)* = above Indian Creek, *(below)* = below Indian Creek, *Cr.* = creek, *E* = east, *(ext)* = extirpated, *Fk.* = fork, *(lm)* = lower mainstem, *M* = middle, *N* = north, *R.* = river, *(um)* = upper mainstem.

Population	Bio. & Ecol.	Status	Subtot.	Status: MPG Rank	Status: ESU Rank	Tributary Opp.	Climate Vuln.	Overall Score	Within-MPG Rank	ESU Rank
Bear Valley Cr.	12	13	25	1	2	6	3	34	2	19
Big Cr.	13	10	23	4	8	6	4	33	4	21
Camas Cr.	8	11	19	9	15	6	4	29	9	26
Chamberlain Cr.	10	11	21	5	11	6	4	31	5	22
Loon Cr.	10	11	21	5	11	6	4	31	5	22
Marsh Cr.	10	14	24	2	4	6	5	35	1	17
M Fk. Salmon R. (above)	14	10	24	2	4	6	4	34	2	19
M Fk. Salmon R. (below)	14	7	21	5	11	6	3	30	8	25
Sulphur Cr.	8	13	21	5	11	6	4	31	5	22
E Fk. Salmon R.	13	13	26	1	1	12.5	4	43	3	5
Lemhi R.	13	11	24	3	4	18	3	45	2	3
N Fk. Salmon R.	8	7	15	9	18	10	4	29	8	26
Pahsimeroi R.	13	12	25	2	2	18	3	46	1	2
Panther Cr. (ext)	9	7	16	8	17	8	3	27	9	28
Salmon R. (lm)	12	10	22	6	10	14	3	39	5	12
Salmon R. (um)	12	12	24	3	4	14	4	42	4	7
Valley Cr.	10	13	23	5	8	12	4	39	5	12
Yankee Fk.	8	11	19	7	15	14.5	4	38	7	14

Table 7-7. Sensitivity analysis of the within-MPG rank scores for Lower Snake, Grande Ronde/Imnaha, and South Fork Salmon River MPGs, after zeroing out component indices (weight = 0). Key: *Cr.* = creek, *E* = east, *(ext)* = extirpated, *Fk.* = fork, *(m)* = mainstem, *R.* = river, *S* = south, *(um)* = upper mainstem.

		Sensitivity Analysis of Within-MPG Rank: Weight = 0									
Population	**Within-MPG Rank**	**Metapop. Role & MPG (2×)**	**Diversity & Spatial Complex.**	**Natural-Origin Abund.**	**24-yr QET Risk**	**ICTRT Viability Gap**	**Habitat Impair.**	**Short-term Potential**	**Long-term Potential**	**Climate Vuln.**	**All Habitat**
Asotin R.	2	2	2	2	2	2	2	2	2	2	2
Tucannon R.	1	1	1	1	1	1	1	1	1	1	1
Big Sheep Cr. (ext)	7	7	7	7	7	7	7	7	7	7	7
Catherine Cr.	1	1	1	1	1	1	1	1	1	1	1
Grande Ronde R. (um)	3	2	3	3	3	2	3	5	5	3	6
Imnaha R. (m)	4	4	4	4	4	4	5	3	4	4	5
Lookingglass Cr. (ext.)	8	8	8	8	8	8	8	8	8	8	8
Lostine R.	2	3	2	2	2	2	2	2	2	2	2
Minam R.	6	6	6	6	6	6	6	6	6	6	2
Wenaha R.	5	4	4	5	5	5	4	3	3	5	2
E Fk. S Fk. Salmon R.	3	3	3	3	3	3	3	3	3	3	3
Secesh R.	1	2	1	2	2	2	1	1	2	2	1
S Fk. Salmon R. (m)	1	1	1	1	1	1	2	2	1	1	2

Table 7-8. Sensitivity analysis of the within-MPG rank scores for Lower Snake, Grande Ronde/Imnaha, and South Fork Salmon River MPGs, after weighting individual component indices by a factor of 2 (weight = 2). Key: *Cr.* = creek, *E* = east, *(ext)* = extirpated, *Fk.* = fork, *(m)* = mainstem, *R.* = river, *S* = south, *(um)* = upper mainstem.

		Sensitivity Analysis of Within-MPG Rank: Weight = 2									
Population	**Within-MPG Rank**	**Metapop. Role & MPG (2×)**	**Diversity & Spatial Complex.**	**Natural-Origin Abund.**	**24-yr QET Risk**	**ICTRT Viability Gap**	**Habitat Impair.**	**Short-term Potential**	**Long-term Potential**	**Climate Vuln.**	**All Habitat**
Asotin R.	2	2	2	2	2	2	2	2	2	2	2
Tucannon R.	1	1	1	1	1	1	1	1	1	1	1
Big Sheep Cr. (ext)	7	7	7	7	7	7	7	7	7	7	7
Catherine Cr.	1	1	1	1	1	1	1	1	1	1	1
Grande Ronde R. (um)	4	3	3	3	3	3	3	2	3	2	4
Imnaha R. (m)	3	4	4	5	4	4	4	4	4	4	3
Lookingglass Cr. (ext.)	8	8	8	8	8	8	8	8	8	8	8
Lostine R.	2	2	2	2	2	2	2	3	2	3	2
Minam R.	6	6	6	6	6	6	6	6	6	6	6
Wenaha R.	5	5	5	4	5	5	5	5	5	5	5
E Fk. S Fk. Salmon R.	3	3	3	3	3	3	3	3	3	3	3
Secesh R.	1	2	2	1	2	2	2	2	2	2	1
S Fk. Salmon R. (m)	2	1	1	2	1	1	1	1	1	1	2

Table 7-9. Sensitivity analysis of the within-MPG rank scores for Middle Fork Salmon River and Upper Salmon River MPGs, after zeroing out component indices (weight = 0). Key: *(above)* = above Indian Creek, *(below)* = below Indian Creek, *Cr.* = creek, *E* = east, *(ext)* = extirpated, *Fk.* = fork, *(lm)* = lower mainstem, *M* = middle, *N* = north, *R.* = river, *(um)* = upper mainstem.

		Sensitivity Analysis of Within-MPG Rank: Weight = 0									
Population	**Within-MPG Rank**	**Metapop. Role & MPG (2×)**	**Diversity & Spatial Complex.**	**Natural-Origin Abund.**	**24-yr QET Risk**	**ICTRT Viability Gap**	**Habitat Impair.**	**Short-term Potential**	**Long-term Potential**	**Climate Vuln.**	**All Habitat**
Bear Valley Cr.	2	4	2	2	1	3	2	2	2	1	1
Big Cr.	4	2	4	4	2	3	4	4	4	4	4
Camas Cr.	9	7	8	8	6	8	8	8	8	8	8
Chamberlain Cr.	5	4	3	3	2	2	3	3	3	2	2
Loon Cr.	5	4	4	5	6	5	5	5	5	5	5
Marsh Cr.	1	1	1	1	2	1	1	1	1	2	2
M Fk. Salmon R. (above)	2	7	4	5	2	6	5	5	5	5	5
M Fk. Salmon R. (below)	8	9	9	9	9	9	9	9	9	9	9
Sulphur Cr.	5	2	4	7	8	7	7	7	7	7	7
E Fk. Salmon R.	3	3	2	2	3	2	3	1	1	3	1
Lemhi R.	2	1	2	2	1	2	1	4	4	2	3
N Fk. Salmon R.	8	8	8	8	8	8	8	8	8	8	8
Pahsimeroi R.	1	1	1	1	2	1	1	2	1	1	2
Panther Cr. (ext)	9	9	9	9	9	9	9	9	9	9	8
Salmon R. (lm)	5	5	7	5	5	5	4	7	6	5	5
Salmon R. (um)	4	4	4	4	3	4	4	2	3	4	3
Valley Cr.	5	7	5	5	7	6	6	4	5	6	5
Yankee Fk.	7	5	6	7	5	7	7	6	7	7	7

Table 7-10. Sensitivity analysis of the within-MPG rank scores for Middle Fork Salmon River and Upper Salmon River MPGs, after weighting individual component indices by a factor of 2 (weight = 2). Key: *(above)* = above Indian Creek, *(below)* = below Indian Creek, *Cr.* = creek, *E* = east, *(ext)* = extirpated, *Fk.* = fork, *(lm)* = lower mainstem, *M* = middle, *N* = north, *R.* = river, *(um)* = upper mainstem.

		Sensitivity Analysis of Within-MPG Rank: Weight = 2									
Population	**Within-MPG Rank**	**Metapop. Role & MPG (2×)**	**Diversity & Spatial Complex.**	**Natural-Origin Abund.**	**24-yr QET Risk**	**ICTRT Viability Gap**	**Habitat Impair.**	**Short-term Potential**	**Long-term Potential**	**Climate Vuln.**	**All Habitat**
Bear Valley Cr.	1	1	1	2	1	1	1	1	2	2	1
Big Cr.	4	2	4	4	4	4	4	4	4	4	4
Camas Cr.	7	8	8	8	7	8	8	8	8	8	7
Chamberlain Cr.	2	3	3	3	3	3	3	3	3	3	2
Loon Cr.	6	5	5	5	7	5	5	5	5	5	6
Marsh Cr.	3	3	2	1	2	2	2	2	1	1	3
M Fk. Salmon R. (above)	5	5	5	7	4	5	5	5	5	5	5
M Fk. Salmon R. (below)	9	9	9	9	9	9	9	9	9	9	9
Sulphur Cr.	7	7	7	6	6	7	7	7	7	7	7
E Fk. Salmon R.	2	3	3	2	3	2	3	3	2	3	2
Lemhi R.	2	1	2	3	1	2	2	1	2	2	2
N Fk. Salmon R.	8	8	8	8	8	8	8	8	8	8	8
Pahsimeroi R.	1	1	1	1	1	1	1	1	1	1	1
Panther Cr. (ext)	9	9	9	9	9	9	9	9	9	9	9
Salmon R. (lm)	5	4	5	5	6	7	3	5	5	5	5
Salmon R. (um)	4	4	4	4	4	4	3	4	4	3	4
Valley Cr.	5	7	5	5	7	6	6	4	5	6	5
Yankee Fk.	7	5	6	7	5	7	7	6	7	7	7

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A Appendix: The COMPASS Stream Temperature Model at Lower Granite Dam

Daniel L. Widener

A.1 Introduction

This report describes the functional form and calibrated parameters of the NWFSC stream temperature model to predict daily tailwater temperature at Lower Granite Dam for use in the COMPASS model. This report also describes the specific application of the stream temperature model to future climate data used in the prospective climate change analyses for the 2020 NOAA Biological Opinion (NMFS 2020).

A.1.1 Rationale for the model

This stream temperature model was constructed with a specific purpose in mind: to predict water temperature needed by the COMPASS model in prospective environmental scenarios where other environmental factors are available but water temperature is not. The COMPASS model runs on a yearly dataset and requires daily water temperature for the reaches of the river that are modeled, typically spanning the reaches of the Snake River and Columbia River between Lower Granite Dam and Bonneville Dam. The COMPASS model can take water temperature inputs at the location of any dam on the mainstem Snake or Columbia Rivers.

To meet these input requirements, the stream temperature model was designed to predict daily water temperatures for an entire calendar year at a time at the location of one of the major dams. Separate models were fitted for the four lower Snake River dams and the four lower Columbia River dams. This report documents only the model fitted for Lower Granite Dam, as it was the only model used in the climate change analyses for the 2020 NOAA Biological Opinion (NMFS 2020). The stream temperature model for each dam is calibrated to temperatures in the tailwater. We chose to use the tailrace as our calibration point, rather than the forebay, because the tailwater is mixed and provides a good estimate of the overall average water temperature. In contrast, the forebay of most dams is thermally stratified during summer; thus, forebay temperature readings are partially a function of monitor depth and may not reflect the overall average water temperature.

In order for the stream temperature model to be usable in prospective analyses, the inputs to the stream temperature model were confined to the types of data expected to be available in typical prospective environmental scenarios produced by NWFSC and by collaborators such as the U.S. Army Corps of Engineers and the University of Washington. These data include flow within the mainstem Snake River and Columbia River, air temperature within the Snake River basin, and snowpack within the Snake River basin. We also included precipitation as an input to the stream temperature model, even though it is usually not available in prospective environmental scenarios. Flow releases from Dworshak Dam are also a very important factor influencing water temperature in the Snake River; releases of cold water to cool the Snake River are a primary consideration of summer operations at Dworshak Dam. Dworshak Dam operations are usually not modeled in prospective environmental scenarios; however, their influence on water temperature at Lower Granite Dam is strong enough that they must be included in the stream temperature model anyway.

A.1.2 Future climate environmental data

As mentioned above, the stream temperature model for Lower Granite Dam was used in the prospective climate change analyses for NMFS (2020). These analyses were based on a set of prospective environmental scenarios produced by the University of Washington (Chegwidden et al. 2019), which we refer to as the RMJOC climate change trends. The outputs of these RMJOC trends include the majority of the environmental data needed by the stream temperature model. We applied the stream temperature model to a number of these trends to predict water temperature at Lower Granite Dam (LGR); this process is described in detail in Section A.5.

A.2 Formulation of the Stream Temperature Model

The model for daily LGR tailwater temperature, *WT(day)*, is a linear combination of four submodels:

$$WT(day) = f(day) - SWE(day) - DWE(day) + T_{air}(day)$$

The first submodel, *f(day)*, is a sine curve with a period of 365 days, which captures the overall yearly pattern in temperature. This submodel is an expansion of a model by Beer and Anderson (2011, 2013). We used the sine structure of the original Beer and Anderson model, but added environmental predictors to the *y*-intercept and phase components. The formulation of the *f(day)* submodel is as follows:

$$f(day) = (A_0 + A_1 \times F_{spr} + A_2 \times AT_{spr} + A_3 \times SW_{apr}) + B \times sin\left(\frac{2\pi}{365}(day + C_0 + C_1 \times F_{spr} + C_2 \times AT_{spr} + C_3 \times SW_{apr})\right)$$

Parameters, *f(day)*:

- A_0, A_1, A_2, A_3: Parameters that collectively describe a linear model of the *y*-intercept of the sine function.
- *B*: Amplitude parameter of the sine function.
- C_0, C_1, C_2, C_3: Parameters that collectively describe a linear model of the phase of the sine function.

Coefficients, *f(day)*:

- F_{spr}: Mean flow, in kcfs, at Lower Granite Dam between 1 March and 31 May.
- AT_{spr}: Mean air temperature, in °C, at Lewiston, Idaho, between 1 March and 31 May.
- SW_{apr}: Mean snow water equivalent, in inches, in the Snake River basin, 1–30 April.
- *day*: Day of the year.

The second submodel, *SWE(day)*, is a model of the daily excursion in water temperature stemming from spring snowmelt. This submodel was also a component of the original Beer and Anderson (2011, 2013) model, but we modified the submodel by adding a logistic relationship between snowpack and the maximum snow water excursion (SWE). This logistic relationship results in diminishing returns in the effect on water temperature when snowpack is very high. The formulation of the *SWE(day)* submodel is as follows:

$$SWE(day) = \begin{array}{ll} day < X_{be} & |\ 0 \\ X_{be} \le day \le X_{en} & |\ \left(\dfrac{K_0 - K_1 \times F_{spr}}{\left(1 + Q \times e^{(-B \times SW_{apr})}\right) \times 2}\right)\left(1 - sin\left(2\pi\left(\dfrac{day - X_{be}}{X_{en} - X_{be}}\right) + \dfrac{1}{4}\right)\right) \\ X_{en} < day & |\ 0 \end{array}$$

Parameters, *SWE(day)*:

- X_{be}, X_{en}: First and last day that snow water excursion is non-zero.
- K_0, K_1: Upper asymptote parameters of the logistic function describing maximum SWE.
- Q: Phase parameter of the logistic function describing maximum SWE.
- B: Slope parameter of the logistic function describing maximum SWE.

Coefficients, *SWE(day)*:

- F_{spr}: Mean flow, in kcfs, at Lower Granite Dam between 1 March and 31 May.
- SW_{apr}: Mean snow water equivalent, in inches, in the Snake River basin, 1–30 April.
- *day*: Day of the year.

The third submodel, *DWE(day)*, is a model of the daily excursion in water temperature stemming from the influence of cold water released from Dworshak Dam. This submodel is structured similarly to the *SWE(day)* submodel, but there is an ordinary linear relationship between Dworshak flow proportion and the maximum excursion, rather than a logistic relationship. The formulation of the *DWE(day)* submodel is as follows:

$$DWE(day) = \begin{array}{ll} day < DX_{be} & |\ 0 \\ DX_{be} \le day \le DX_{en} & |\ \dfrac{FDW_{day-6}}{FLGR_{day}}\left(D_0 \times \dfrac{1}{2}\right)\left(1 - sin\left(2\pi\left(\dfrac{day - DX_{be}}{DX_{en} - DX_{be}}\right) + \dfrac{1}{4}\right)\right) \\ DX_{en} < day & |\ 0 \end{array}$$

Parameters, *DWE(day)*:

- DX_{be}, DX_{en}: First and last day that Dworshak water excursion is non-zero.
- D_0: Maximum scale of Dworshak water excursion.

Coefficients, *DWE(day)*:

- FDW_{day-6}: Daily flow, in kcfs, at Dworshak Dam tailrace, lagged six days.
- $FLGR_{day}$: Daily flow, in kcfs, at Lower Granite Dam on the current day.
- *day*: Day of the year.

The fourth and final submodel, $T_{air}(day)$, is a combination of daily deviations in temperature stemming from air temperature, instantaneous flow, and precipitation. The formulation of this submodel is as follows:

$$T_{air}(day) = \frac{M_0 + M_1 \times FLGR_{day}}{1 + M_Q \times e^{(-M_B \times AT_{day})}} + M_L \times ATL_{day-7} + M_P \times P_{day} + M_F \times FLGR_{day}$$

Parameters, $T_{air}(day)$:

- M_0, M_1: Upper asymptote parameters of the logistic model of instantaneous daily air temperature effect.
- M_Q: Phase parameter of the logistic model of instantaneous daily air temperature effect.
- M_B: Slope parameter of the logistic model of instantaneous daily air temperature effect.
- M_L: Linear parameter of lagged daily air temperature effect.
- M_P: Linear parameter of daily precipitation effect.
- M_F: Linear parameter of daily flow effect.

Coefficients, $T_{air}(day)$:

- $FLGR_{day}$: Daily flow, in kcfs, at Lower Granite Dam on the current day.
- AT_{day}: Daily air temperature, in °C, at Lewiston, Idaho, on the current day.
- ATL_{day-7}: Five-day mean air temperature at Lewiston, lagged seven days.
- P_{day}: Precipitation, in inches, on the current day.

A.3 Calibration of the Stream Temperature Model

The stream temperature model for Lower Granite Dam was calibrated to historical water temperature data from the tailrace temperature monitor at Lower Granite Dam. Daily data from 1995 through 2015 were used for calibration; because fish are not generally present during the winter in COMPASS model runs, only data between Julian Day 50 and Day 300 were used. Any days with missing data were ignored and did not contribute to model fitting. Maximum likelihood was used to calibrate the stream temperature model parameters; we assumed a normal error distribution, and all data points were assumed independent.

Model selection was performed via a backwards Akaike information criterion (AIC) selection process. We began with the full model, and in a stepwise process dropped the parameter that would result in the largest improvement in penalized AIC. When a point was reached where dropping any parameter would result in a degradation in penalized AIC, we selected that model as the final model (Table A-1).

Table A-1. Fitted coefficients of the stream temperature model for Lower Granite Dam. Parameters with *n/a* were dropped during model selection and are not part of the final model.

Submodel	Parameter	Fitted Value
f(day)	A_0	10.0539
	A_1	−0.0220
	A_2	n/a
	A_3	n/a
	B	−7.4286
	C_0	34.8012
	C_1	1.2143
	C_2	0.1648
	C_3	n/a
SWE(day)	X_{be}	74.0
	X_{en}	244.0
	K_0	6.0959
	K_1	0.0159
	Q	4.7663
	B	0.1962
DWE(day)	DX_{be}	164.8
	DX_{en}	302.8
	D_0	−4.4105
$T_{air}(day)$	M_0	1.9256
	M_1	−0.0102
	M_Q	5.1329
	M_B	0.2454
	M_L	0.1691
	M_P	0.3050
	M_F	−0.0001922

A.4 Model Performance

The final stream temperature model for Lower Granite Dam does a good job of capturing overall yearly patterns in water temperature in the calibration dataset. The stream temperature model is also able to capture many fine-scale fluctuations in temperature (Figures A-1 and A-2).

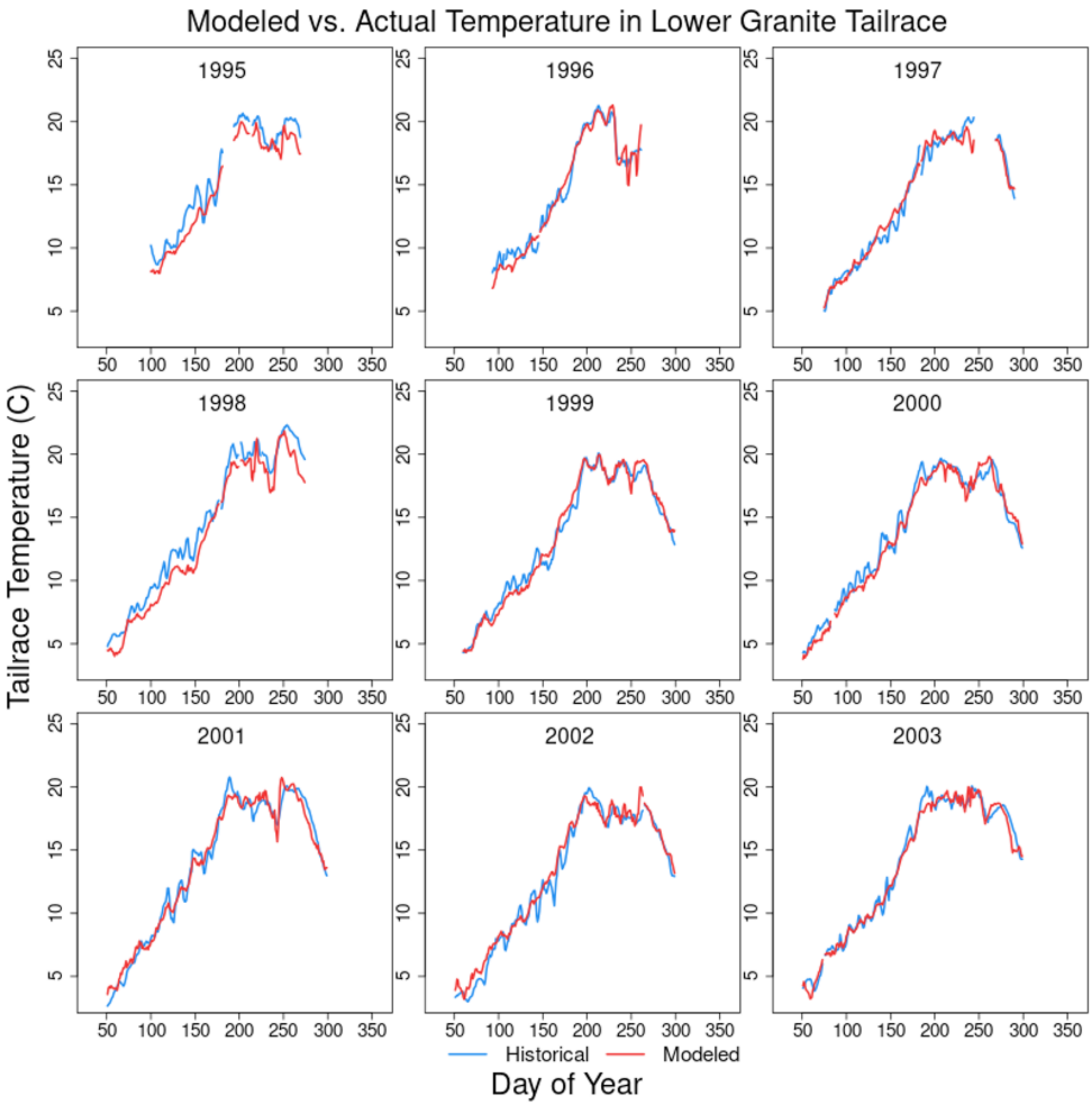

Figure A-1. Historical water temperature (in blue) in the tailrace of Lower Granite Dam, and water temperature predicted by the stream temperature model (in red), for the years 1995–2003.

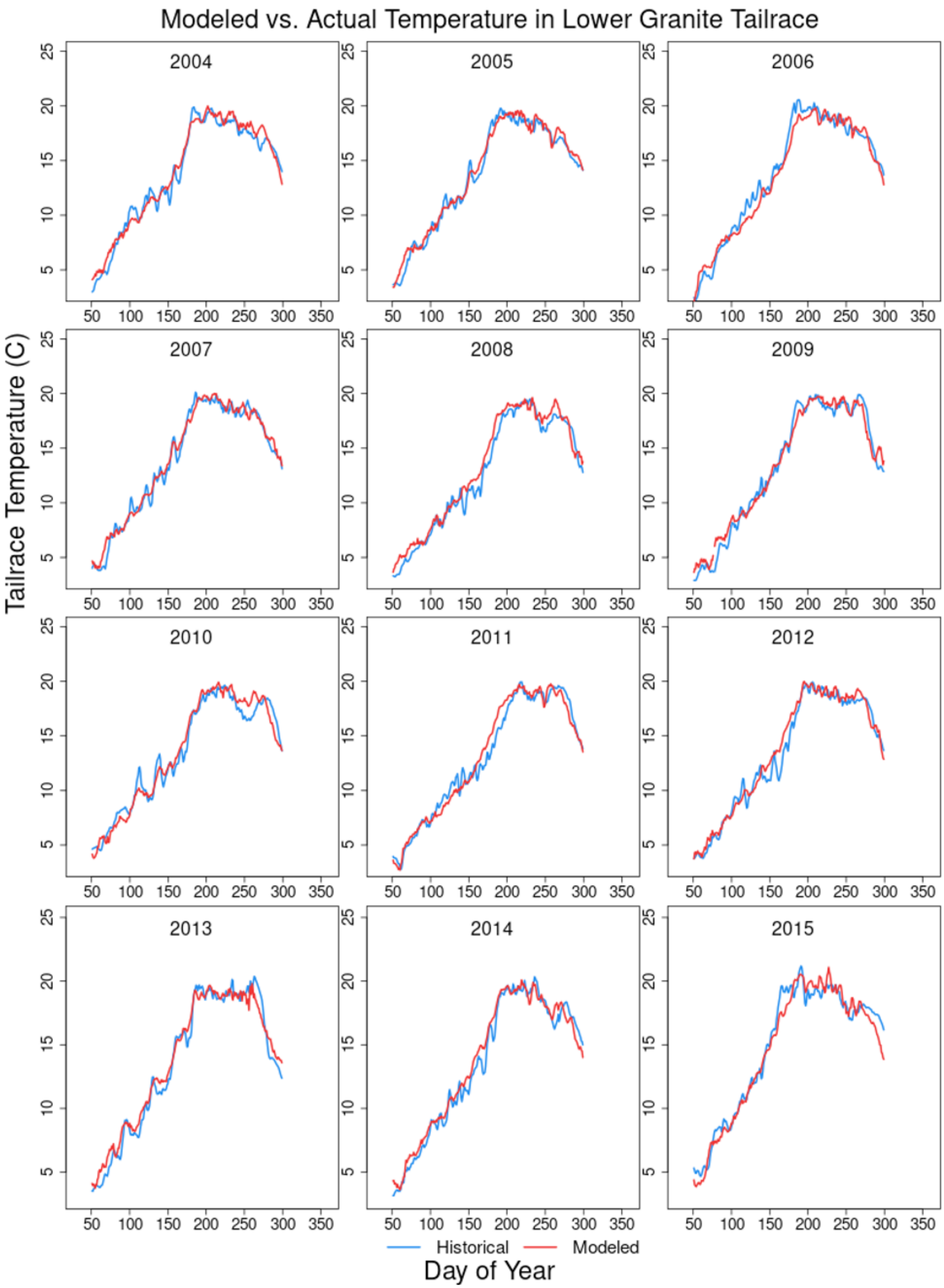

Figure A-2. Historical water temperature (in blue) in the tailrace of Lower Granite Dam, and water temperature predicted by the stream temperature model (in red), for the years 2004–15.

A.5 Prospective Application of the Stream Temperature Model

We applied the stream temperature model at Lower Granite Dam to a set of prospective environmental scenarios produced by the Columbia River Climate Change[1] modeling group at the University of Washington for the River Management Joint Operating Committee (RMJOC; Chegwidden et al. 2019). We collectively refer to these prospective scenarios as the RMJOC trends. We used 80 of these trends of past and future environmental conditions; the trends we chose included a variety of different climate change models and hydrologic models (Crozier et al. submitted).

We modeled daily water temperature at Lower Granite Dam for each RMJOC trend using the stream temperature model. We took the environmental data produced from the RMJOC trends and formatted them for use in the prospective stream temperature model; all 150 years, 1950–2099, were used for each trend. Here we describe how the stream temperature input requirements were met for the five main types of input data the model requires:

1. **Flow data at Lower Granite Dam:** Daily flow was modeled at Lower Granite Dam by Columbia River Climate Change (data received from Oriana Chegwidden and Bart Nijssen). Flow was modeled separately for five different scenarios; these five scenarios were each used in multiple RMJOC climate trends. For each flow scenario, the daily flow values were used directly in the prospective stream temperature model; we also created yearly averages of flow between 1 March and 31 May for use in the prospective stream temperature model.
2. **Flow data at Dworshak Dam:** No model or data exists to predict what Dworshak Dam outflow would be in the various prospective climate trends. Therefore, we made a daily average of Dworshak outflows from 2006 to 2015. This average Dworshak flow trajectory was used for all years in the prospective stream temperature model for the various climate trends, and did not vary between years or trends.
3. **Air temperature data at Lewiston, Idaho:** Daily air temperature was modeled at Lewiston (lat 46.3747°N, long 117.0156°W) in the Snake River basin by Columbia River Climate Change (data from Chegwidden and Nijssen). Maximum and minimum daily air temperatures were modeled, and each prospective climate trend was modeled separately. We averaged the maximum and minimum daily air temperature values for each trend. These daily average vectors were used in the prospective stream temperature model, as well as five-day rolling means of the daily average vectors. Additionally, we created yearly averages of all daily values between 1 March and 31 May for use in the prospective stream temperature model.
4. **Snowpack data:** Daily snow water equivalent (SWE) was modeled at the locations of SNOTEL[2] monitoring sites in the Snake River basin by Columbia River Climate Change (data from Chegwidden and Nijssen). Each prospective climate trend was modeled separately. Each trend averaged the SWE across all modeled sites in the Snake River Basin for the full month of April for each year. The resulting basinwide average April SWE values were used in the prospective stream temperature model.

[1] https://www.hydro.washington.edu/CRCC/documentation/introduction/
[2] https://www.wcc.nrcs.usda.gov/snow/snotel-wereports.html

5. **Precipitation data:** No model or data exists to predict daily precipitation in the various RMJOC climate trends. Daily precipitation was assumed to be zero for all days in the prospective stream temperature model.

Once daily water temperature at Lower Granite Dam had been predicted for the full time series of each RMJOC trend, we then produced seasonal averages of water temperature. For each year in each trend, we averaged the predicted daily water temperatures between 1 April and 30 June. The resulting April–June average water temperatures at Lower Granite Dam were then used in the climate change modeling for NMFS (2020).

In this specific case, the daily water temperatures produced by the stream temperature model were not used in COMPASS model runs. Instead, COMPASS model runs for NMFS (2020) used water temperature predicted by the U.S. Army Corps of Engineers. The stream temperature model was used only to predict seasonal average temperature at Lower Granite Dam for each of the RMJOC climate trends.

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