

Evaluating Salmonid and Stream Ecosystems Response to Conservation Measures and Environmental Stressors in the Columbia River Basin: Annual Report 2021-2022

publication date: May 1, 2023

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Technical Report

23-03

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Evaluating salmonid and stream ecosystem response to conservation measures and environmental stressors in the Columbia River basin

BPA Project # 2009-004-00

Report covers work performed under BPA contract #(s) 73354 REL 55 and 90582

Report covers work performed from:

January 2021– December 2022

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May 2023

This report was funded by the Bonneville Power Administration (BPA), U.S. Department of Energy, as part of BPA's program to protect, mitigate, and enhance fish and wildlife affected by the development and operation of hydroelectric facilities on the Columbia River and its tributaries. The views in this report are the author's and do not necessarily represent the views of BPA.

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List of common abbreviations

Abbreviation	Description
AqI	Aquatic Inventories Project
BPA	Bonneville Power Administration
BSR	Biologically Significant Reach
CC or CAT	Catherine Creek
CHaMP	Columbia Habitat Monitoring Program
CRB	Columbia River basin
CRITFC	Columbia River Inter-Tribal Fish Commission
CTUIR	Confederated Tribes of the Umatilla Indian Reservation
CTWSRO	Confederated Tribes of the Warm Springs Reservation of Oregon
DEQ	Oregon Department of Environmental Quality
EPA	Environmental Protection Agency
FL	Fork Length
GR	Grande Ronde River
GRMW	Grande Ronde Model Watershed
(SS)LCM	(State-Space) Life Cycle Model
LOS	Lostine River
MFJD	Middle Fork John Day River
MIN	Minam River
NOAA	National Oceanic and Atmospheric Administration
NPCC	Northwest Power and Conservation Council
ODFW	Oregon Department of Fish and Wildlife
OSU	Oregon State University
OWRD	Oregon Water Resources Department
PNAMP	Pacific Northwest Aquatic Monitoring Program
RM&E	Research, Monitoring, and Evaluation
UAS	Unmanned Aircraft System
UGR	upper Grande Ronde River
USBR	U.S. Bureau of Reclamation
USFS	U.S. Forest Service
USGS	U.S. Geological Survey
USWCD	Union Soil and Water Conservation District

Executive Summary

Background and Objectives

The Columbia River Inter-Tribal Fish Commission is conducting a research, monitoring, and evaluation study designed to determine the effectiveness of aggregate restoration actions in improving freshwater habitat conditions and viability of ESA-listed spring Chinook Salmon (*Oncorhynchus tshawytscha*) populations. A critical uncertainty for fisheries managers in the Columbia River basin is whether freshwater habitat restoration actions will improve basin-wide habitat quantity/quality and thereby salmon productivity to a level sufficient to offset human-caused survival impairments elsewhere in the life cycle. Geographically, this project is focused on the Grande Ronde River basin (tributary of the Snake River in the Columbia River basin), but with applications and testing of research, monitoring, and evaluation approaches also occurring in other Columbia River tributaries.

We have hierarchically structured our work into four primary goals and associated objectives in our current and proposed work plan (Figure 1):

Goal 1: Assess status and trends of key fish habitat limiting factors in the Grande Ronde basin.

- *Objective A: Describe habitat conditions of the Grande Ronde basin as compared to historical and/or reference target values.*
- *Objective B: Evaluate annual and decadal trends of key limiting factors in the Grande Ronde basin.*

Goal 2: Evaluate effectiveness of aggregate restoration actions in the Grande Ronde basin.

- *Objective C: Evaluate if the pace of restoration can counteract habitat and temperature degradation.*

Goal 3: Relate biological responses to habitat change.

- *Objective D: Relate life stage-specific fish response to habitat change in the Grande Ronde basin and nearby basins.*
- *Objective E: Relate population-level fish response to habitat change in the Grande Ronde basin and potentially other basins.*

Goal 4: Apply lessons learned from RM&E to Grande Ronde basin salmon recovery efforts and other emerging concerns or locations.

- *Objective F: Address needs of CRITFC tribal and other partners evaluating emerging concerns in the Grande Ronde basin and other geographic locations.*

Sub-objectives are separated into three phases (Table 1) corresponding to the current work plan of FYs 2021-2022 (Phase 0), and proposed work in FYs 2023-2025 (Phase 1) and FYs 2026-2027 (Phase 2). Progress made in 2021 and 2022 towards sub-objectives associated with Phase 0 are presented in this report.

Progress and Key Findings

Goal 1: Assess status and trends of key fish habitat limiting factors in the Grande Ronde basin

Objective A: Describe habitat conditions of the Grande Ronde basin as compared to historical and/or reference target values

Objective A-1: Finalize tributary habitat protocol

- This sub-objective was completed and presented in the FY 2021 annual report (White et al. 2022).

Objective A-2: Collect habitat data using the Tributary Habitat Assessment Protocol (TribAP) in Atlas Tier I-II priority areas within the current Chinook extent in the Grande Ronde basin.

- In 2021, we worked collaboratively with the Grande Ronde Model Watershed (funded by USDA Forest Service) to survey a total of 37.1 km (mainstem length) and 0.26 km² of stream habitat in Tier 1 priority areas of the upper Grande Ronde River basin using the Tributary Habitat Assessment Protocol.
- For each survey segment, we calculated a suite of metrics describing key characteristics of habitat quantity and quality/diversity for both mainstem and floodplain/side channel habitats.
- Riverscape conditions varied widely across the survey area, coincident with a large degree of variability in underlying physical setting (e.g., geology, gradient, streamflow) as well as land use and restoration impacts. These data will be used in future analyses to assess how key habitat limiting factors have changed over time and how they relate to target values for restoration and recovery.
- Habitat surveys conducted in Tier 2 areas of the upper Grande Ronde basin during 2022 will be described in the 2023 annual report.

Objective A-3: Set target values for restoration

- We performed an analysis of CHaMP data across the Columbia River basin to quantify potential target values for restoration with respect to two variables: frequency of medium-sized pools and frequency of large wood pieces.
- We selected the 90th percentile of all CHaMP sites, stratified by a channel type classification, as a preliminary target value.
- The current median values of these variables in the Minam and upper Grande Ronde rivers are far below the target value, indicating that intensive restoration activities may be required to meet these targets for the average site within these subbasins.

Objective A-4: Habitat status assessment (LiDAR)

- In summer 2020, CRITFC, in collaboration with GRMW and BPA, acquired topobathymetric (“green”) LiDAR across 76,188 acres of the Grande Ronde River and Wallowa River

watersheds, including tributaries currently or historically occupied by Chinook Salmon and steelhead.

- We are in the process of using this LiDAR data to describe spatially continuous instream fish habitat metrics and floodplain conditions and to compare how habitats have changed since our previous LiDAR acquisition in 2009. We expect to complete this task by FY2024.

Goal 2: Evaluate effectiveness of aggregate restoration actions in the Grande Ronde basin

Objective C: Evaluate if the pace of restoration can counteract habitat and temperature degradation

Objective C-1: Develop list of restoration scenarios

- CRITFC staff hosted a collaborative workshop in November of 2021 with Grande Ronde basin partners to develop a suite of riverscape restoration and management scenarios intended to be used as inputs to the Grande Ronde spring Chinook life cycle model described in Objective D.
- The group developed a draft set of broad management scenarios addressing instream and floodplain restoration, riparian vegetation restoration, food webs, and other in-basin factors (i.e., landowner access, lower valley mortality, hatchery supplementation) and discussed specific management actions that could be implemented to achieve recovery objectives.
- The set of management scenarios is still in development with the goal of finalizing and running through the life cycle model during FY2023 and FY2024.

Objective C-2: Quantify change in habitat limiting factors relative to restoration actions

- To address a critical gap in our ability to translate restoration actions into habitat change across large geographic extents (e.g., biologically significant reaches [BSRs], Chinook population areas), we plan to quantify observed changes in habitat limiting factors before and after restoration at selected sites within the study area.
- While no progress was made on this objective in FY2022, we plan to complete this objective in late FY2023 or early FY2024.

Objective C-3: Extrapolate impacts of restoration scenarios across historical Chinook extent in upper GR basin

- Using observed changes in habitat characteristics at selected restoration sites from Objective C-2, we plan to extrapolate the expected uplift to habitat conditions throughout the stream network in the study area. This objective is planned to be completed in FY2024.

Goal 3: Relate biological responses to habitat change

Objective D: Relate life stage-specific response to habitat change in the Grande Ronde and nearby basins

Objective D-1: Collect data on fish and macroinvertebrate distribution

- We conducted snorkel count surveys in 22 monitoring sites subsampled from habitat segments surveyed using the Tributary Habitat Assessment Protocol (TribAP) during summer of 2021.
- We collected benthic macroinvertebrate samples at 12 monitoring sites using a nationally standardized, reach-based methodology.
- Results indicated an overall decrease in juvenile salmonid density (fish · 100m⁻¹) with increasing downstream distance, with a much more substantial decrease in juvenile Chinook density than juvenile Steelhead density, likely due to differences in thermal tolerance between species.
- Benthic macroinvertebrate metrics derived from collected samples indicated a similar pattern, with decreases in density, biomass, O/E, aquatic species composition (ASC_{Rel}) and other metrics with further distance downstream on the Grande Ronde.
- Snorkel and benthic macroinvertebrate surveys conducted in Tier 2 areas of the upper Grande Ronde basin during 2022 will be described in the 2023 annual report.

Objective D-2: Index of Chinook Salmon rearing capacity

- This work was completed and described in the 2021 annual report.

Objective D-3: Finalize pre-spawn mortality analysis

- This work was completed and described in the 2021 annual report.

Objective D-4: Complete study of juvenile fish emergence timing and floodplain inundation

- We are working with partners to develop a framework for evaluating juvenile Chinook Salmon floodplain habitat use across watersheds that links dynamics of floodplain accessibility with aspects of juvenile salmon ecology that can influence their ability to utilize floodplain habitat.
- We completed preliminary modeling of floodplain inundation dynamics across 30-50 km of three NE Oregon subbasins and over multiple years. Our approach involves using a deep-learning model to classify wetted surfaces from multi-spectral imagery obtained over a range of flows.
- We are in the process of linking these floodplain predictions to factors influencing juvenile salmon habitat use, including emergence phenology, proximity of spawning locations to floodplain habitat, and dispersal patterns.

Objective D-5: Complete study of juvenile Chinook dispersal and floodplain use

- We are evaluating juvenile Chinook Salmon dispersal patterns from spawning locations to summer rearing habitats in the Middle Fork John Day River using genetic-based parentage assignments.

- We sampled adults ($n = 141$) in 2020, sampled parr ($n = 3,388$) in 2021, genotyped adults and parr, and paired parr to female adults using parentage-based tagging ($n = 1,326$ parr-female pairs).
- Results indicate that 1) dispersal was downstream-biased, 2) parr dispersed farther distances than most studies indicate, 3) larger individuals dispersed farther, and 4) dispersal patterns were not consistent across the basin, with parr generally moving from warmer areas to cooler areas.
- A draft manuscript has been included in Appendix B: Juvenile Salmon Dispersal using Parentage-Based Tagging.

Objective E: Relate population-level fish response to habitat change in the Grande Ronde basin and potentially other basins

Objective E-1: Grande Ronde Phase 1 – Development of spring Chinook statistical estimation Life Cycle Model

- We performed a suite of exploratory analyses using the state-space model input data intended to guide how to properly account for the effects of intrinsic population factors (e.g., density-dependence) on population dynamics.
- We found evidence for density-dependent survival rates, both in the pre-recruitment (egg-to-parr) and post-recruitment (parr-to-smolt) freshwater phases.
- We found evidence for density-dependent pre-recruitment growth rates (indexed by mean length at end of summer), and that post-recruitment survival was positively related to mean length; we further found that little variability in survival was explained by density after controlling for growth rates.
- We now have a hypothesized mechanism to capture density effects on freshwater juvenile survival variability in the state-space model: pre-recruitment survival and growth are both density-dependent; growth in the pre-recruitment phase acts as a delayed effect on smolt migration survival and potentially parr overwinter rearing survival.
- The analyses we conducted suffered from many statistical assumption violations (e.g., variables measured without error, years are independent replicates) – a key role of the state-space model will be to alleviate, or at least better account for, these violations to facilitate more robust inferences.

Objective E-2: LCM Phase II – management scenarios

- Restoration scenarios to impose on the SSLCM model output have not yet been formally developed, although several planning meetings, including a well-attended workshop devoted to brainstorming with partners in the basin, have occurred.
- We plan to assess combinations of restoration and other management actions (e.g., changes to supplementation) of various intensities. More details will be forthcoming in the 2023 annual report.

Objective E-3: LCM Phase III – simulation of outcomes

- We have not yet built the simulation model based on the SSLCM output because the SSLCM model structure has been in flux and the management scenarios are not yet finalized. Some

progress on constructing the simulation model is anticipated in 2023, and some preliminary results may be available for the 2023 report.

Goal 4: Apply lessons learned from RM&E to Grande Ronde salmon recovery efforts and other emerging concerns or locations

Objective F: Address needs of CRITFC tribal and other partners evaluating emerging concerns in the Grande Ronde and other geographic locations

Objective F-1: Continue developing/applying Adaptive Management framework with GRMW and Grande Ronde basin partners

- In 2022, we participated in the Grande Ronde State of the Science Adaptive Management Workshop hosted by GRMW to discuss restoration progress, RM&E findings, and emerging uncertainties/questions pertinent to management efforts in the basin, all of which were formally documented. Important highlights from the workshop are discussed under Objective F-1.
- Coordination with other entities involved in fish habitat restoration and associated data collection in the Grande Ronde, Upper Columbia, Mid-Columbia, and Snake River watersheds is a critical component of this project. Numerous examples of our collaborative efforts are described within including: 1) participation in Atlas review and updates, 2) collaboration with USDA Forest Service on Meadow Creek restoration and RM&E, 3) Coordination with OSU, CTUIR, NOAA, and ODFW on juvenile salmon dispersal study, 4) Hosting a multi-agency training on fish habitat and snorkel survey methods, and 5) Collaborating closely with NOAA and ODFW on development of a spring Chinook statistical estimation life cycle model for the Grande Ronde basin.
- We identified protocol updates and additional research needs related to some of our RM&E methods and designs based on lessons learned from previous work in the Grande Ronde basin including 1) minor updates to the Tributary Habitat Assessment Protocol (e.g., adding marsh habitats), 2) additional paired mark-recapture and snorkel survey work in 2023 to better estimate snorkel detection efficiency in deep or complex habitat types, and 3) training needed for identification of imperiled freshwater mussels.

Objective F-2: Represent tribal concerns in Columbia River basin tributary habitat RM&E policy discussions

- CRITFC's River Ecology Group participated in regular meetings and workshops over the past several years focusing on development and review of a regional tributary habitat research, monitoring, and evaluation (RM&E) strategy in coordination with tribes, Northwest Power and Conservation Council (NPCC), NOAA, and BPA.
- We also participated in regular meetings with the Columbia Basin Collaborative's (CBC) Estuary, Tributary & Mainstem Habitat workgroup to develop regional recommendations to the Integration/Recommendations Group (IRG) regarding best management practices for habitat and salmon recovery in the Columbia River basin.

Dissemination of project findings

- Staff from CRITFC's River Ecology group delivered 24 scientific presentations during 2021 and 2022 describing project findings.
- We produced 5 peer-reviewed publications during 2021 and 2022.

Introduction

The Columbia River Inter-Tribal Fish Commission (CRITFC) is conducting a research, monitoring, and evaluation (RM&E) study designed to determine the effectiveness of aggregate restoration actions in improving freshwater habitat conditions and viability of ESA-listed spring Chinook Salmon (*Oncorhynchus tshawytscha*) populations. A critical uncertainty for fisheries managers in the Columbia River basin (CRB) is whether freshwater habitat restoration actions will improve basin-wide habitat quantity/quality and thereby salmon productivity to a level sufficient to offset human-caused survival impairments elsewhere in the life cycle. Geographically, this project is focused on the upper Grande Ronde River, Catherine Creek, and Minam River watersheds (tributaries of the Snake River in the Columbia River basin), but with recent applications and testing of approaches in other Columbia River tributaries.

Many studies in recent years have examined the current condition of fish habitat in Columbia River subbasins and how these habitat conditions influence salmon survival and productivity. Some of the most common impediments to the survival of salmon include high water temperatures, increased concentrations of fine sediment in spawning gravel, loss of riparian vegetation, channelization and diminished channel and floodplain complexity, loss of large wood in the channel, loss of large pools for adult fish holding and juvenile rearing, and depletion of summertime streamflow. More recent studies have additionally identified food webs (e.g., nutrient limitation, primary productivity, prey availability, or predation) as limiting factors for salmonids. Climate change presents an additional threat, as it can lead to changes in the quantity, timing, and type (i.e., snow vs rain) of precipitation as well as increased summer air and water temperatures.

Habitat restoration in the upper Grande Ronde River and Catherine Creek basins is being conducted by agencies including the USDA Forest Service, Confederated Tribes of the Umatilla Indian Reservation (CTUIR), Oregon Department of Fish and Wildlife (ODFW), Union Soil and Water Conservation District (USWCD), Grande Ronde Model Watershed (GRMW), Trout Unlimited and U.S. Bureau of Reclamation (USBR). However, it remains unclear how these collective restoration actions affect salmon habitat quality and quantity in the freshwater tributary life stages, let alone how they impact salmon populations in the context of the complete life cycle. Fish-habitat relationships are inherently complex as they are influenced by interactions among intrinsic watershed factors (e.g., geology, valley form, natural streamflow regime), biological factors (e.g., predation, prey availability) and anthropogenic factors (e.g., land use, climate change, restoration). These in turn affect ecological conditions and ultimately drive changes in fish abundance and productivity. This project incorporates several of these interacting factors in a holistic analytical framework.

This project serves the needs of the CRITFC 1855 Treaty Tribes in their obligations related to the 2008 Columbia Basin Fish Accords to ensure implemented habitat projects are “linked to biological benefits based on limiting factors for ESA-listed fish” (Accords 2008, p. 10). Among other general provisions for non-hatchery projects, the Accords states the Tribes shall “provide estimated habitat quality improvement and survival benefits from the project (or suite of projects) to a population or populations of listed salmon and steelhead based on key limiting factors” (p. 14). This project provides the knowledge basis to ensure benefits to populations from habitat projects can be estimated with reasonable precision and accuracy. Our project additionally serves the mission laid out in Wy-Kan-Ush-Mi Wa-Kish-Wit (Spirit of the Salmon Plan) developed by CRITFC and the Tribes, which takes a holistic approach to salmon and aquatic ecosystem recovery over multi-generational time scales.

Our project directly supports the goals of the Habitat sub-strategy of the Council’s 2014 Columbia River Basin Fish and Wildlife Program, namely in terms of developing “assessments of current physical and biological conditions and also identify factors that limit the productivity and capacity of focal species in priority reaches.” This includes evaluating how climate change may affect fish population responses to habitat restoration emphasized in the Program’s 2020 Addendum. More specific to the study area, the Grande Ronde Subbasin Plan describes the need for RM&E efforts to address both project-level (“bottom-up”) habitat assessments but also large-scale landscape and ecosystem-level (“top-down”) monitoring; the latter is a gap our project strives to fill.

Beyond monitoring and evaluation of limiting factors, our project has a strong research emphasis consistent with the Council's 2017 Research Plan by evaluating whether “improvements in tributary habitat conditions not only boost survival and productivity of fish in the tributaries but also contribute to survival benefits at the population scale” (p. 6). In our project this is primarily addressed using life cycle models, but we also conduct research on several critical uncertainties (e.g., climate change, food webs, land use) so their influence on the expected benefits to fish populations from habitat projects can be understood and shared with the broader scientific community. These combined efforts are intended to provide guidance on whether and how restoration implementers and basin managers can meet important biological targets, such as the minimum abundance and productivity thresholds for natural origin spawners expressed in NOAA’s Recovery Plans for Grande Ronde/Imnaha River populations of Snake River Spring Chinook Salmon.

Overall goals of the project include the following:

Goal 1: Assess status and trends of key fish habitat limiting factors in the Grande Ronde basin—

Objectives towards this goal involve describing habitat conditions as compared to historical and/or target values (Objective A) and evaluating annual and decadal trends of key limiting factors (Objective B).

Goal 2: Evaluate effectiveness of aggregate restoration actions in the Grande Ronde basin—

Objectives towards this goal involve evaluating if the pace of restoration (current or projected) can counteract habitat and temperature degradation (Objective C).

Goal 3: Relate biological responses to habitat change—Objectives towards this goal involve collecting biological data and developing life stage-specific (Objective D) and population-level (Objective E) models relating habitat change to fish response in the Grande Ronde and other basins.

Goal 4: Apply lessons learned from our research, monitoring, and evaluation (RM&E) to Grande Ronde salmon recovery efforts and other emerging concerns or locations—Objectives towards this goal involve remaining engaged in the Grande Ronde Atlas adaptive management framework (Objective F-1), representing tribal concerns in development of a regional tributary habitat evaluation framework (Objective F-2), and developing a new work plan adapting our program to other emerging concerns, locations, or focal species (Objectives F-3, F-4, F-5). It was the intention in the original 2009 proposal that lessons learned in the Grande Ronde subbasin would be applied to other geographic areas in the Columbia River basin.

Goals, objectives, and tasks corresponding to finalizing the current work plan (Phase 0; FY 2021-22) and for the proposed Phase 1 (FY 2023-2025) and Phase 2 (FY 2026-27) plans are mapped in Figure 1 below, with timelines in the corresponding Gantt chart (Table 1).

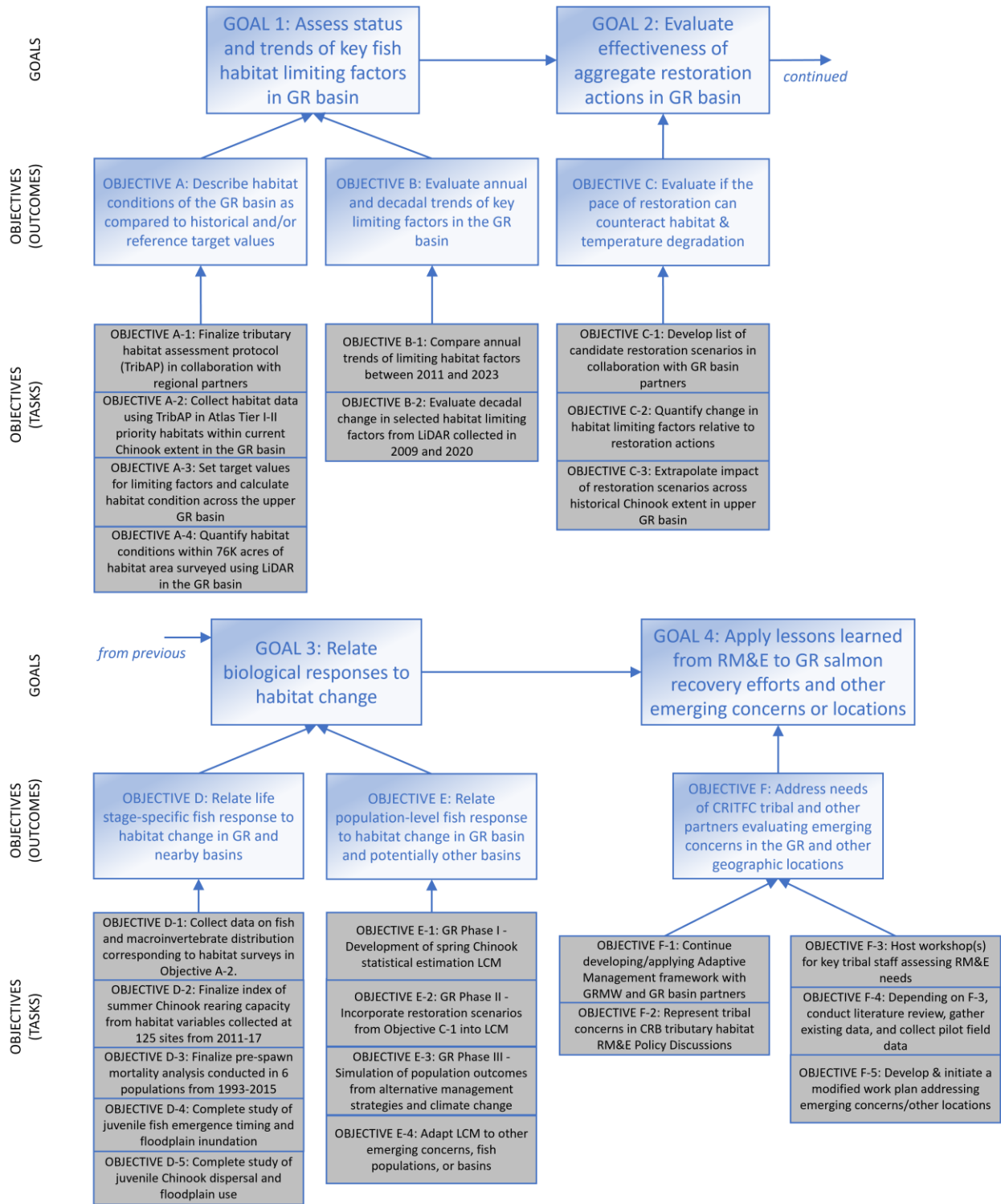


Figure 1. Project goals, objectives, and tasks.

Table 1. Gantt chart showing projected timelines of tasks for the current and proposed work plans. Fiscal year (FY) for this project is April 1 – March 31.

Objective			Phase 0:		Phase 1:			Phase 2:	
			Current work plan		Three-year proposal			Five-year proposal	
Goal	(Task)	Short description	FY 2021	FY 2022	FY 2023	FY 2024	FY 2025	FY 2026	FY 2027
1	A-1	Finalize tributary habitat protocol	■	■	■				
	A-2	Apply habitat protocol in GR basin		■	■	■			
	A-3	Target habitat values analysis			■	■			
	A-4	Habitat status assessment (LiDAR)		■	■	■			
	B-1	Annual habitat trends analysis (ground)				■	■		
	B-2	Decadal habitat trends analysis (LiDAR)			■	■	■		
2	C-1	Develop candidate restoration scenarios	■	■	■	■			
	C-2	Quantify restoration impact on habitat		■	■	■			
	C-3	Spatial extrapolation of restoration			■	■	■		
3	D-1	Collect fish and macroinvertebrate data	■	■	■	■			
	D-2	Index of Chinook rearing capacity		■	■	■			
	D-3	Finalize pre-spawn mortality analysis			■	■			
	D-4	Emergence phenology of rearing Chinook		■	■	■			
	D-5	Juvenile Chinook dispersal study (MFJD)			■	■			
	E-1	LCM Phase I: Statistical estimation	■	■	■	■			
	E-2	LCM Phase II: Management scenarios		■	■	■	■		
	E-3	LCM Phase III: Simulation of outcomes			■	■	■	■	
	E-4	Adapt LCM for other factors/populations					■	■	■
4	F-1	Adaptive management with GRMW	■	■	■	■	■	■	■
	F-2	Trib Habitat RM&E Framework for CRB	■	■	■	■	■	■	■
	F-3	Needs assessment with CRITFC tribes				■	■		
	F-4	Draft workplan and pilot fieldwork					■	■	
	F-5	Final work plan and implementation						■	■

Study Area

This project is occurring primarily in the Grande Ronde River and its tributaries, which originates in the Blue Mountains of NE Oregon and flows 334 km to its confluence with Snake River (Figure 2). Focal study watersheds include the upper Grande Ronde River (UGR) upstream of the town of La Grande, Catherine Creek (CAT), and to a lesser extent the Minam River (MIN). Spring Chinook Salmon populations in these basins were listed as threatened under the Endangered Species Act in 1992. Population declines over the past century were due in part to overharvest, hydropower impacts, and degraded habitat conditions resulting from intensive anthropogenic disturbances including timber harvest, cattle grazing, levee and road construction, stream diversions for irrigation, and removal of beaver populations (*Castor canadensis*). Specifically, stream temperature, streamflow, habitat diversity, large wood structures, and quantity of key habitats such as large pools, have been identified as key limiting factors for recovery of salmonid populations in these basins. The Minam River is a designated wilderness area and represents a minimally impacted reference stream. We additionally conduct limited research in nearby basins with similar biophysical conditions and land use history, such as Lookingglass Creek (Grande Ronde basin) and the Middle Fork John Day River.

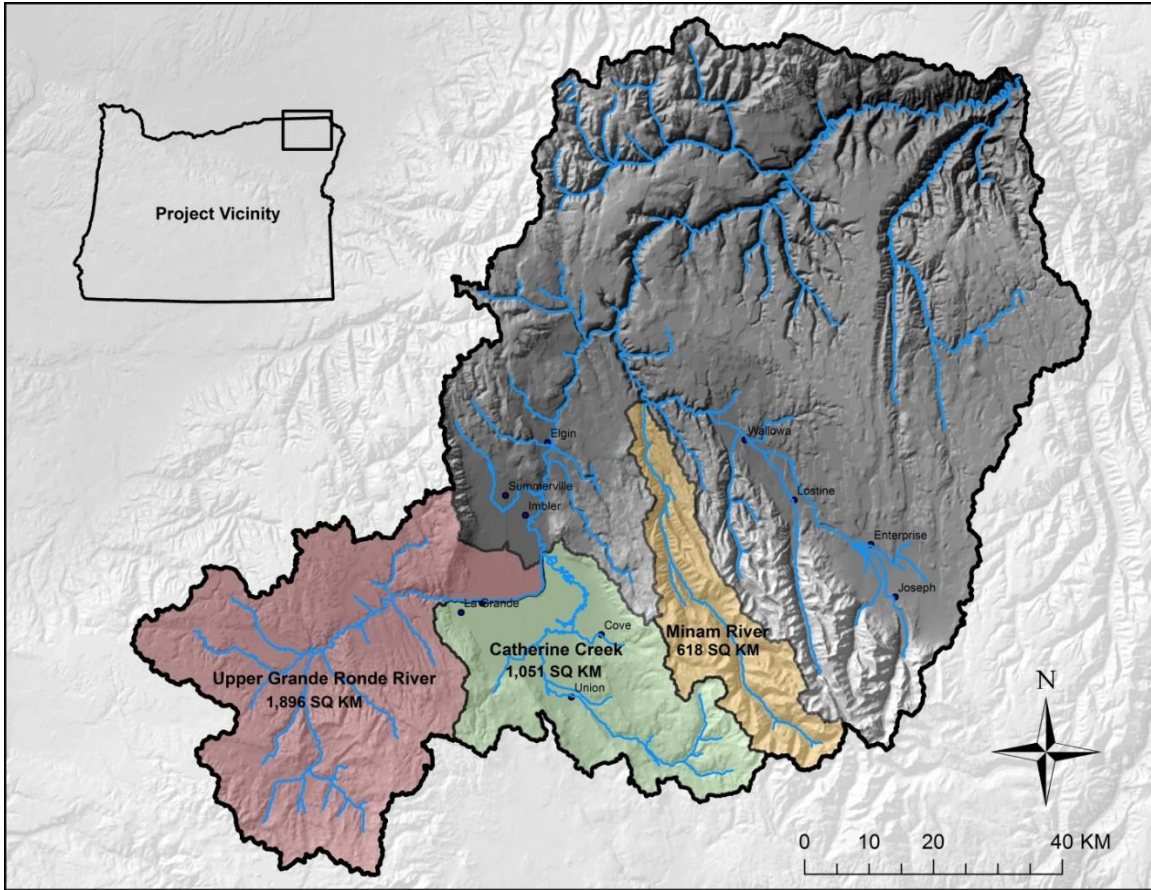


Figure 2. Study area in the Grande Ronde River basin, NE Oregon. Focal watersheds include the upper Grande Ronde River, Catherine Creek, and Minam River. The upper Grande Ronde and Catherine Creek are basins with significantly impacted habitat, currently undergoing restoration in various locations. The Minam River basin is the local reference area that has far less anthropogenic impact.

Progress on project goals

Goal 1: Assess status and trends of key fish habitat limiting factors in the Grande Ronde basin

Objective A: Describe habitat conditions of the Grande Ronde basin as compared to historical and/or reference target values

Objective A-1: Finalize tributary habitat protocol

This sub-objective was completed and presented in the FY 2021 report.

Objective A-2: Collect habitat data using the Tributary Habitat Assessment Protocol (TribAP) in Atlas Tier I-II priority areas within the current Chinook extent in the Grande Ronde basin.

Habitat Surveys

Background

CRITFC has consistently monitored fish habitat conditions in the upper Grande Ronde River, Catherine Creek, and Minam River since 2009. After initially drafting our own agency stream monitoring protocol, CRITFC supported the development and implementation of the Columbia Habitat Monitoring Program (CHaMP 2016). Data collected from these programs (2011-2017) provided the basis for describing status and trends of limiting habitat factors for Chinook Salmon in the study basins. In 2017, Bonneville Power Administration (BPA) commissioned a review of the Columbia Habitat Monitoring Program (CHaMP) which highlighted potential problems or shortcomings concerning repeatability, efficiency, and validity of some monitoring methods, and with extrapolation of metrics to unsampled portions of the basin. In response to these concerns and feedback from the Independent Science Review Panel (ISRP), our project reassessed the components that would ensure the success and longevity of a new monitoring approach and allow for the continuation of status and trends analysis. We identified four major considerations when reconceptualizing a new monitoring strategy: 1) the methods used should be based on regionally accepted practices which are both repeatable and could reliably be used for comparison to previously derived metrics from other protocols, 2) it should incorporate measures of long-term impacts of land use and climate change, 3) it increases efficiency of previous fish habitat survey methodologies by reducing the intensity of ground-based measurements and integrating data collected by unmanned aircraft systems (UAS), and 4) it promotes partnership and garners regional support as a means for funding and continued long-term monitoring.

Using these considerations as a guide, we developed the Tributary Habitat Assessment Protocol (Justice et al. 2020). This protocol includes a pared-down list of metrics identified as having minimal observer bias, clear linkage to common ecological concerns in Columbia basin tributaries (i.e., water temperature, channel and floodplain complexity, pool habitats, fine sediment, etc.), and consistency (i.e., cross-walkability) with previously collected habitat monitoring data. This protocol was tested during the 2018 field season and implemented during summer of 2021 and 2022. The methods outlined in this protocol are based on widely accepted and previously implemented monitoring methods used throughout the Pacific Northwest (i.e., CHaMP, ODFW Aquatic Inventories Project (AqI)). Acknowledging previous criticisms, the monitoring

approach outlined within the CRITFC Tributary Habitat Assessment Protocol aims to cover more ground with less effort and cost, while obtaining high resolution aerial imagery using UAS that will provide a rich, spatially-referenced dataset with numerous current and future analytical opportunities. Our application of this monitoring protocol corresponds to biologically significant reaches (BSRs) in the project areas, with a specific focus on areas designated as high or medium priority for habitat restoration (i.e., Tier 1 and 2 BSRs; Atlas 2015). However, the protocol is flexible enough to be tailored to specific limiting factors or needs of a particular basin such as prioritization of stream segments with active or planned restoration.

The Tributary Habitat Assessment Protocol was designed to provide a comprehensive and continuous riverscape perspective of the status and trends in fish habitat by merging datasets from multiple spatial scales (channel unit ~ 1-100m, reach ~ 100-1000 m, segment ~ 1000-10,000m, watershed) and components of the riverscape (hydrology, geomorphology, biology). The major components of this protocol are split into ground and aerial-based methods. The ground-based methods are a fusion of two widely used and accepted protocols within the CRB including AqI (Moore et al 2019) and CHaMP (CHaMP 2016), while the aerial-based portion of the protocol utilizes drones. Drones have become ubiquitous in monitoring throughout a range of disciplines within the CRB. Drones are used in this protocol to collect imagery of the stream channel and floodplain and to develop georeferenced orthomosaics, digital surface models, and digital terrain models. To increase efficiency and repeatability, we reduced the frequency and total number of measurements collected within habitat units (i.e., channel units) by ground crews compared with previous CHaMP surveys. We attempted to reduce the reliance on qualitative or visually-estimated metrics to the degree possible with the intention of producing metrics that are robust enough to provide meaningful evaluations of habitat change over time.

Methods

Prior to conducting field surveys, the stream network was classified into segments to set the spatial boundaries for measurements of fish habitat and biota and to help organize the survey workflow into units of manageable size. We delineated stream segments using the National Hydrography Dataset High Resolution flowlines (NHDPlus HR, 1:24K scale; USGS 2016) as a starting point. Similar to the U.S. Forest Service (USFS) Region 6 Level I Stream Inventory Handbook (USDA 2018), segment boundaries were based on the presence of large tributary junctions (Strahler order ≥ 4) or significant changes in valley confinement and gradient. Stream reaches falling between these break points were lumped together into a 'segment' and assigned a unique identification number. With a few exceptions, we used a minimum segment length of 1 km, consistent with the spacing of Generalized Random Tessellation Stratified (GRTS) master sample points used by CHaMP and other programs to characterize broad-scale status and trends in fish habitat conditions. Segments were grouped by 12-digit hydrologic unit code (HUC) watershed boundaries, consistent with the National Watershed Boundary Dataset (WBD; USGS et al. 2015) to facilitate data tracking and management.

In 2021, we worked collaboratively with the GRMW (funded by USFS) to survey a total of 37.1 km (mainstem length) and 0.26 km² of stream habitat in the upper Grande Ronde River basin using the Tributary Habitat Assessment Protocol. Surveys were intended to cover the current extent of Chinook Salmon spawning and rearing habitat within the Grande Ronde Atlas Tier 1 area (Figure 3; Atlas 2015) which was comprised of 21 unique river segments. Unfortunately, some locations could not be surveyed due to landowner denial—most notably the Vey Meadows portions of Sheep Creek and the Grande Ronde River (~ 19.8 km). Additionally, portions of some segments (notably GR0007, GR0014, and GR0020) were

surveyed by drone but the corresponding ground-based habitat data was lost or was lacking sufficient clarity to tie it to the drone imagery.

Within each segment, we collected aerial imagery (standard red/green/blue [RGB] and multispectral) using a DJI Matrice 600 Pro drone outfitted with a dual payload including a Zenmuse X5 RGB camera (16 MP resolution) and Micasense Rededge MX multispectral sensor (1.2 MP) flown at an altitude of 80-90 m (262-295 ft) above ground level. We used DJI Ground Station Pro on an Apple iPad mini for flight planning and control. Front and side image overlap was set to 80% and flight speed was generally below 8 m/s. Aerial targets consisting of bright spray paint or painted bucket lids were surveyed with an EOS Arrow 100 GNSS receiver (sub-meter accuracy) and used for georeferencing the aerial imagery.

Ground based measurements such as channel unit number, type (pool, fast turbulent, fast non-turbulent, etc.), large wood count, water depth, etc. were recorded using ArcGIS Survey123 on iPad mini or Samsung Galaxy tablets. To tie ground-based measurements to aerial imagery, we surveyed the boundaries of channel units and the edge of water using an EOS Arrow 100 GNSS receiver (Figure 4). The Arrow 100 typically achieved submeter accuracy except in rare cases with deep canyon walls (e.g., Dark Canyon Creek). Data collection with the Arrow 100 receiver was managed using ArcGIS Field Maps on an iPad mini or Samsung Galaxy tablet. Detailed survey methods are available on [monitoringresources.org](http://www.monitoringresources.org) at <http://www.monitoringresources.org/Document/Protocol/Details/3554>.

For each survey segment, we calculated a suite of metrics describing key characteristics of habitat quantity and quality/diversity for both mainstem and floodplain/side channel habitats (Table 2). For each metric, we computed the length-weighted mean within each BSR to account for differences in segment length where the weighting factor was given by the mainstem channel length of each segment. Some image processing methods are still in the process of being finalized (e.g., riparian vegetation and large wood) and therefore the metrics that rely on these methods are not included in this report. Additionally, while the original protocol described the development of streamflow metrics such as mean annual flow, mean summer flow and center of flow mass, we have since decided that deriving accurate streamflow metrics across a diverse set of river segments within a watershed using a limited number of gauging stations requires a sophisticated modeling approach that is beyond the scope of this protocol.

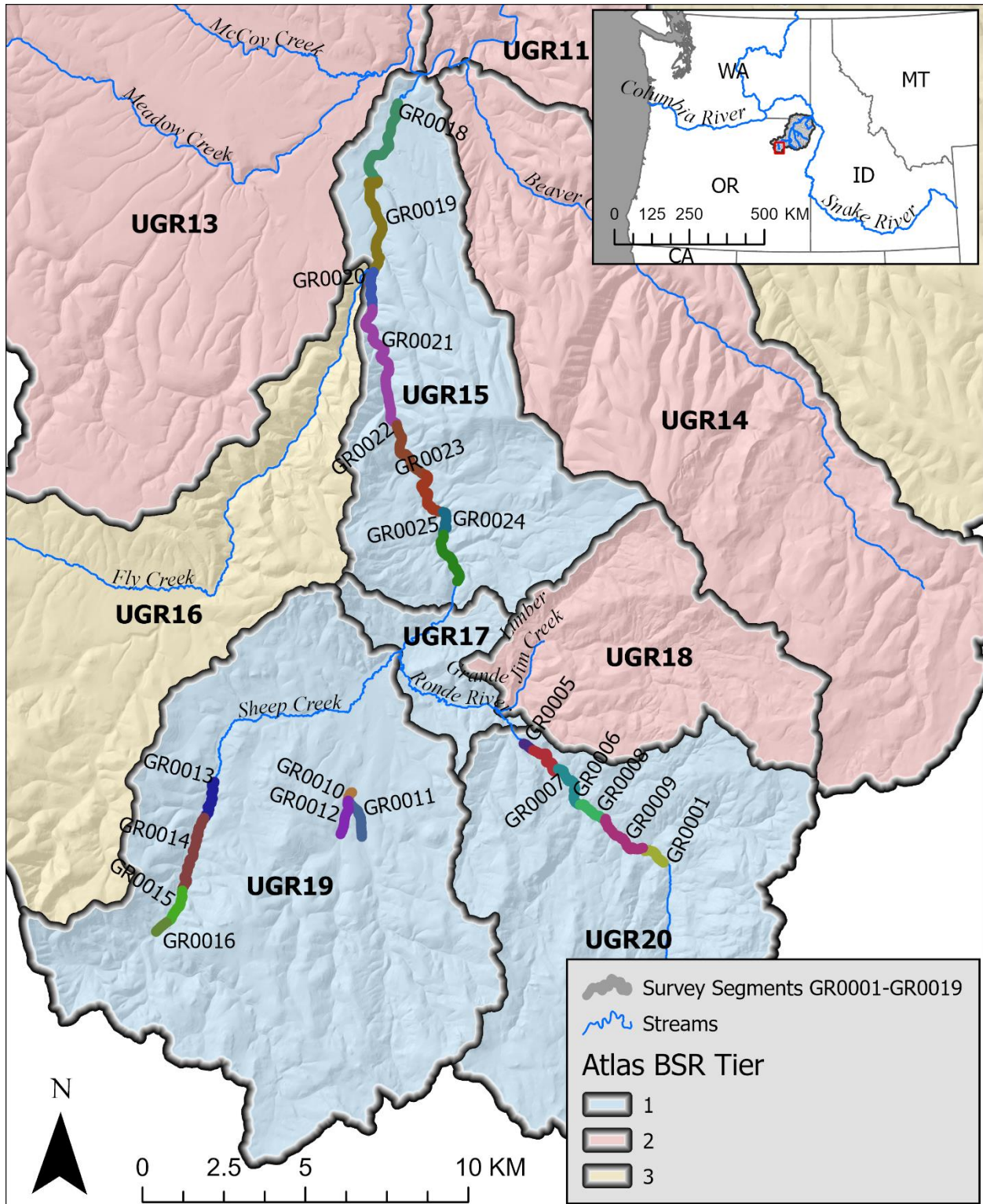


Figure 3. Survey area in the upper Grande Ronde River basin in NE Oregon

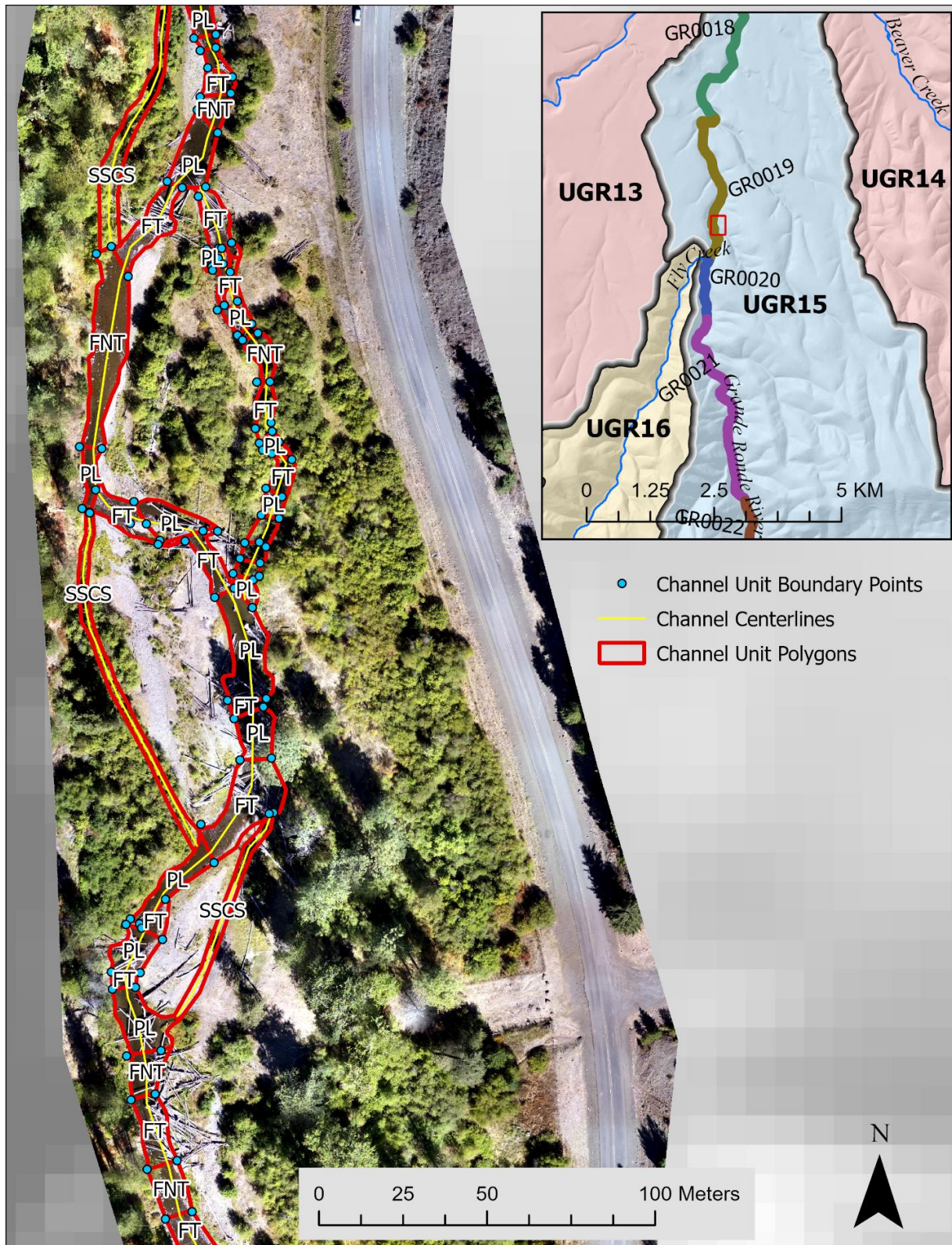


Figure 4. Example of spatial data collected within each survey segment including the unmanned aircraft system (UAS) orthomosaic, channel unit boundary points and polygons and channel centerlines.

Table 2. Description of stream habitat metrics derived from the Tributary Habitat Assessment Protocol.

Primary metric type	Secondary metric type	Metric	Description	Data source
Habitat quality/diversity	Floodplain/side channels	River complexity index (<i>RCI</i>)	River complexity index ($RCI = S*(1+J)$ where S = stream sinuosity, J = # of side channel junctions (Brown 2002)	Field/LiDAR/UAS
	Floodplain/side channels	Side channel ratio (<i>SCRatio</i>)	Length of side channels divided by length of main channel during base flow (Beechie et al. 2017)	Field/LiDAR/UAS
	Riparian condition	Riparian tree cover (<i>RipTreeCov</i>)	Average percent tree canopy cover in the riparian zone (50 m stream buffer)	UAS/LiDAR
	Riparian condition	Riparian tree height (<i>RipTreeHt</i>)	Average tree height (m) in the riparian zone (50 m stream buffer)	UAS/LiDAR
	Riparian condition	*Riparian vegetation departure index (<i>RVD</i>)	Average percentage departure of current vegetation from simulated historical vegetation reference conditions in the riparian zone (Macfarlane et al. 2017)	Satellite/Modeled
	Riparian condition	Normalized difference vegetation index (<i>NDVI</i>)	Average NDVI index in the riparian zone (50 m stream buffer) calculated as the ratio between visible and near-infrared reflectance of vegetation cover (Bhandari et al. 2012). NDVI is used as an index of vegetation greenness or health.	UAS/Satellite
	River channel (cover)	Large wood area percentage (<i>LWAreaPct</i>)	Percentage of stream surface area covered by large wood during base flow.	UAS/LiDAR
	River channel (cover)	Large wood frequency bankfull (<i>LWFreqBF</i>)	Number of large wood pieces (> 3m length and 0.15 m diameter) within the bankfull channel per 100 m stream length (Moore et al. 2019)	Field
	River channel (cover)	Large wood frequency wetted (<i>LWFreqWet</i>)	Number of large wood pieces (> 3m length and 0.15 m diameter) within the wetted channel during base flow per 100 m stream length	Field
	River channel (cover)	Overhanging vegetation cover (<i>OverVegCov</i>)	Percentage of stream surface area covered by vegetation during base flow	UAS/LiDAR
River channel (cover)	Undercut bank percentage (<i>UcutBankPct</i>)	Percentage of the total bank length that is undercut	Field/UAS	

Primary metric type	Secondary metric type	Metric	Description	Data source
	River channel (pools)	Residual pool depth (<i>PoolResidDpth</i>)	Mean residual pool depth (max depth – pool tail depth in meters; Mossop and Bradford 2006)	Field/LiDAR
	River channel (pools)	Large pool frequency (<i>PoolFreqLg</i>)	Number of large pools (> 20 m ² area and > 0.80 m max depth) per km stream length (McIntosh et al. 2000)	Field/UAS/LiDAR
	River channel (pools)	Medium pool frequency (<i>PoolFreqMd</i>)	Number of medium- or larger-sized pools (> 20 m ² area and > 0.50 m max depth) per km stream length	Field/UAS/LiDAR
	River channel (substrate)	Median sediment particle size (<i>D50</i>)	Median sediment particle size on the streambed surface in riffles (Wolman 1954)	Field
	Water quality	*Coldwater refuge density (<i>ColdRefDen</i>)	Number of cold-water refuges per km stream length (Dugdale et al. 2015)	FLIR
	Water quality	Maximum weekly maximum temperature (<i>MWMT</i>)	Maximum 7-day running average of daily maximum temperature (EPA 2003)	Field/Model
	Water quality	*Observed/Expected benthic macroinvertebrates (<i>O/E</i>)	Ratio of observed to expected (O/E) benthic macroinvertebrate taxa as predicted by the River Invertebrate Prediction and Classification System (RIVPACS, Hawkins et al. 2000)	Field
Habitat quantity	Floodplain/side channels	Off-channel habitat base flow (<i>OCHabBase</i>)	Surface area (m ²) of connected off-channel habitat (alcoves, backwaters, and side channels) during base flow	LiDAR/UAS
	Floodplain/side channels	Side channel length (<i>SCLength</i>)	Length (m) of side channels during base flow	Field/UAS
	River channel (fast water)	Fast water area (<i>FastArea</i>)	Surface area (m ²) of fast water habitat (e.g., fast turbulent, fast non-turbulent, fast small side channels) during base flow	Field/UAS
	River channel (total length)	Main channel length (<i>MCLength</i>)	Length (m) of main channel habitat during base flow	Field/UAS
	River channel (pools)	Slow water area (<i>SlowArea</i>)	Surface area (m ²) of slow water habitat (e.g., pools, off-channel units, slow small side channels) during base flow	Field/UAS

Results

The total quantity of habitat surveyed within Tier 1 priority areas of the upper Grande Ronde basin is summarized in Table 3. UGR15, which comprises the furthest downstream portion of the Chinook spawning extent in the upper Grande Ronde River from Meadow Creek upstream to the Vey Meadows boundary, held the largest amount of overall stream habitat (total mainstem length = 18.9 km, total area [mainstem plus side channels] = 0.18 km²; Table 3). In comparison, UGR20, which includes the primary spawning grounds for Chinook Salmon and stretches upstream from the upper Vey Meadows boundary to the headwaters above East Fork Grande Ronde comprised only 7.4 km of mainstem length and 0.05 km² total habitat area. UGR19, which includes the smaller tributary streams of Sheep Creek and Chicken Creek, comprised approximately 10.7 km of mainstem length and 0.027 km² total habitat area.

The percentage of slow water habitat (i.e., pools, off-channel units, slow small side channels) across all segments ranged from as little as 1% up to 94% (mean = 32%; Table 3). The percentage of slow water habitat was typically lowest in UGR15 (length weighted mean = 17%) followed by UGR20 and finally UGR19. Despite being dominated by fast-water habitat types (i.e., riffles and fast non-turbulent units), off-channel habitats at base flow (i.e., off-channel units and side channels) were most abundant in UGR15 (total off-channel area = 10,055 m²) compared with the other BSRs (510 m² in UGR19 and 5,210 m² in UGR20).

Riverscape metrics related to habitat quality/diversity varied widely across the survey area, coincident with a large degree of variability in underlying physical setting (e.g., geology, gradient, streamflow) as well as land use and restoration impacts. River complexity index, a dimensionless metric that captures both the sinuosity of the primary channel and the frequency of side channel nodes (Brown 2002), ranged from 1.2 to 13.8 (mean = 6.9), and was highest in UGR15 (weighted mean = 9.3) and lowest in UGR20 (weighted mean = 5.2). While the overall amount of off-channel habitat was highest in UGR15, the side channel ratio, which takes into account differences in overall length of the mainstem channel, did not differ substantially between UGR15 and UGR20 (weighted mean = 0.14 and 0.13, respectively; Table 4), but was substantially lower in UGR 19 (weighted mean = 0.03).

River channel cover as represented by the frequency of large wood pieces per 100 m in the bankfull channel was similar in UGR19 and UGR20 (weighted mean = 32.1 and 29.7, respectively) and relatively low in UGR15 (weighted mean = 16.2). The percentage of undercut bank was low across all segments (range = 0 – 4.5 %, mean = 0.98 %) and did not differ substantially across BSRs.

The frequency of large pools (i.e., count of pools with max depth > 0.8 m and area > 20 m² per kilometer) was highest in the upper portion of the Grande Ronde River (UGR20; weighted mean = 5.2) and substantially lower in UGR15 and UGR19 (weighted mean in both = 2.4) on average. There were some notable locations where large pool frequency was substantially higher than other survey areas including segment GR0008 in the upper Grande Ronde mine tailings reach (10.4 large pools per km) and GR0011 in Chicken Creek (12.3 large pools per km), where recent restoration work by the USDA Forest Service has enhanced pool habitat.

Peak summer water temperature throughout the study area was above the Environmental Protection Agency (EPA) temperature standard of 16 °C MWMT for core Chinook rearing areas (EPA 2003; Table 4). MWMT across all segments ranged from 18.1 to 27.7 °C (mean = 23. °C). Peak temperatures were highest and very consistent in the lower portion of the watershed (UGR15; weighted mean = 27.2 °C). In comparison, weighted mean MWMT was 21.8 °C in UGR19, and 19.9 °C in UGR20, though due to data loss, only a single temperature site was available for UGR15 in 2021. Cold-water refuges, defined as patches of water

with minimum temperatures at least 2 degrees colder than the ambient river temperature and having a surface area > 1 m², were most abundant in UGR15 (weighted mean refuge density = 27.8 refuges per km), were not detected in UGR19, and were sparsely abundant in UGR20 (weighted mean = 1.8). It should be noted that the cold-water refugia data were derived from 2010 thermal imagery, and have likely changed over time.

Next Steps

This report provides a basic summary of habitat metrics computed from the Tributary Habitat Assessment Protocol surveys conducted in 2021. As noted above, the metrics reported here do not yet include those related to riparian vegetation (cover, height, overhanging vegetation, riparian vegetation departure index) or aerial estimates of large wood surface area. We expect to complete metric calculations by September, 2023. Metric calculations for 2022 surveys are underway and will be described in the 2023 annual report. While calculating habitat metrics is an important first step, more useful to our tribes and basin partners is an assessment of how these metrics have changed over time and how they relate to target values for restoration and recovery (i.e., how much have we moved the needle towards achieving restoration objectives). Analyses addressing these questions will be forthcoming in the 2023 annual report.

Caveats

The metrics reported here may change slightly following a final series of QA checks and updates to our metrics calculation database.

Table 3. Habitat quantity metrics from 2021 surveys summarized by biologically significant reach (BSR) and segment.

BSR	Segment ID	River Channel				Floodplain/Side Channels		
		MCLength	FastArea	SlowArea	% SlowArea	OCHabBase	SCLength	
UGR15	GR0018	3223	26592	6602	20	2416	644	
	GR0019	3473	35244	7226	17	3997	974	
	GR0020	1423	534	1026	66	0	247	
	GR0021	4207	28997	3928	12	0	129	
	GR0022	2267	19938	4281	18	2416	417	
	GR0023	1820	19749	1753	8	881	163	
	GR0024	812	7442	612	8	194	48	
	GR0025	1742	19212	272	1	151	54	
	Total	18,967	157,708	25,701		10,055	2,675	
	Mean	2,371	19,713	3,213	19	1,257	334	
	Wt Mean ¹	2,864	23,805	4,124	17	1,538	407	
	SD	1,156	11,297	2,714	20	1,505	328	
UGR19	GR0010	477	323	1398	81	37	21	
	GR0011	1544	1628	4141	72	17	10	
	GR0012	1625	99	1543	94	0	0	
	GR0013	1777	2958	2516	46	63	60	
	GR0014	3473	4790	639	12	103	109	
	GR0015	1308	4106	1269	24	194	72	
	GR0016	487	1671	385	19	96	67	
		Total	10,690	15,575	11,892		510	339
	Mean	1,527	2,225	1,699	50	73	48	
	Wt Mean	2,097	2,891	1,694	43	76	60	
	SD	1,008	1,801	1,277	33	66	39	
UGR20	GR0001	935	3099	2923	49	1042	291	
	GR0005	483	3444	329	9	0	0	
	GR0006	1449	10263	1433	12	1328	274	
	GR0007	1547	5658	2857	34	0	0	
	GR0008	1061	5558	3624	39	2430	375	
	GR0009	1958	7355	4298	37	410	86	
		Total	7,432	35,377	15,463		5,210	1,025
		Mean	1,239	5,896	2,577	30	868	171
	Wt Mean	1,420	6,522	2,912	31	845	166	
	SD	520	2,653	1,457	16	938	163	
All BSRs	Total	37,089	208,660	53,056		15,775	4,039	
	Min	477	99	272	1	0	0	
	Max	4,207	35,244	7,226	94	3,997	974	
	Mean	1,766	9,936	2,526	32	751	192	
	SD	1,044	10,552	2,007	27	1,132	245	

* Weighted mean where weights were given by the mainstem channel length within each segment.

Table 4. Habitat quality/diversity metrics from 2021 surveys summarized by biologically significant reach (BSR) and segment.

BSR	Segment ID	Floodplain/Side Channels		River Channel (Cover)			River Channel (Pools)			Substrate	Water Quality	
		RCI	SCRatio	LWFreqBF	LWFreqWet	UCutBankPct	PoolResidDpth	PoolFreqLg	PoolFreqMd	D50	ColdRefDen ²	MWMT
UGR15	GR0018	11.8	0.20	1.8	1.0	1.46	0.52	3.1	6.2	65	9.7	27.7
	GR0019	13.8	0.28	25.7	10.7	0.67	0.42	2.3	5.2	68	14.9	27.0
	GR0020	11.1	0.17	60.0	26.4	4.45	0.47	6.6	33.0	67	0.0	
	GR0021	6.0	0.03	11.7	5.8	0.00	0.46	1.6	5.5	69	7.6	27.0
	GR0022	10.2	0.18	13.1	6.6	1.33	0.39	4.0	11.0	70	113.8	27.0
	GR0023	10.2	0.08	13.7	7.1	0.42	0.34	0.5	5.5	75	81.3	
	GR0024	2.4	0.05	11.5	5.7	0.00	0.37	1.2	9.9	103	8.4	
	GR0025	3.6	0.03	8.5	4.2	0.15	0.22	0.0	1.7	82	0.0	27.7
	Mean	8.6	0.13	18.2	8.4	1.06	0.40	2.4	9.7	75	29.5	27.3
	Wt Mean ¹	9.3	0.14	16.2	7.5	0.92	0.42	2.4	8.1	71	27.8	27.2
SD	4.1	0.09	18.1	7.7	1.48	0.09	2.1	9.8	13	43.2	0.4	
UGR19	GR0010	3.6	0.04	47.4	28.3	0.35	0.42	6.3	16.8	45	0.0	
	GR0011	6.7	0.01	43.5	29.6	0.47	0.70	12.3	25.3	35	0.0	
	GR0012	1.6	0.00	17.5	5.5	0.11	0.26	0.6	1.2		0.0	
	GR0013	9.6	0.03	38.9	30.5	0.47	0.28	0.7	13.4	45	0.0	21.8
	GR0014	7.3	0.03	29.0	18.8	0.07	0.30	0.0	2.1	50	0.0	
	GR0015	8.6	0.05	30.7	21.0	2.97	0.30	0.8	6.1	39	0.0	
	GR0016	6.9	0.13	31.0	21.6	2.65	0.27	0.0	2.1	46	0.0	
	Mean	6.3	0.04	34.0	22.2	1.01	0.36	3.0	9.6	44	0.0	21.8
	Wt Mean	6.7	0.03	32.1	21.1	0.68	0.35	2.4	8.3	44	0.0	21.8
	SD	2.8	0.04	10.1	8.7	1.24	0.16	4.7	9.2	5	0.0	
UGR20	GR0001	4.8	0.31	29.9	19.0	1.35	0.43	7.5	26.7	54	0.0	
	GR0005	2.1	0.00	8.1	4.1	0.40	0.59	2.1	6.2	105	0.0	20.9
	GR0006	11.9	0.18	37.1	14.5	0.45	0.38	4.2	11.9	96	9.2	21.0
	GR0007	1.2	0.00	39.9	27.0	0.05	0.52	4.8	5.7	71	0.0	20.8
	GR0008	8.1	0.35	37.2	27.4	1.15	0.44	10.4	26.4	46	0.0	
	GR0009	2.7	0.04	17.5	13.9	1.58	0.43	3.1	20.4	45	0.0	18.1
	Mean	5.1	0.15	28.3	17.7	0.83	0.47	5.3	16.2	70	1.5	20.2
	Wt Mean	5.2	0.13	29.7	18.7	0.87	0.45	5.2	16.4	65	1.8	19.9
	SD	4.1	0.16	12.8	8.8	0.61	0.08	3.1	9.6	26	3.8	1.4
	All BSRs	Min	1.2	0.00	1.8	1.0	0.00	0.22	0.0	1.2	35	0.0
Max		13.8	0.35	60.0	30.5	4.45	0.70	12.3	33.0	105	113.8	27.7
Mean		6.9	0.10	26.4	15.7	0.98	0.41	3.4	11.5	64	11.7	23.9
SD		3.8	0.11	15.3	10.0	1.16	0.12	3.5	9.6	21	29.3	3.7

¹ Weighted mean where weights were given by the mainstem channel length within each segment. ² Cold-water refugia derived from 2010 thermal imagery.

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Water Temperature

Background

CRITFC has maintained an extensive network of year-round water temperature loggers in the Grande Ronde basin since 2009 with the goal of tracking long-term trends in water temperature related to land use, restoration, and climate change. These data have also been used to develop fish-habitat relationships, water temperature models such as Heat Source, and have been shared widely with basin partner and others to aid in assessment of site-specific restoration effectiveness or development of large-scale temperature models (e.g., NorWeST, DEQ TMDL).

Methods

CRITFC actively maintains 61 water temperature sites in the upper Grande Ronde, Catherine Creek, and Minam River watersheds, but has monitored up to 174 sites in past years (Figure 5). In 2021 and 2022, we downloaded data from 52 and 50 sites, respectively. Additionally, we compiled temperature data collected by 10 other agencies between 1988 and 2017, totaling 455 unique sites. External agencies included the Bureau of Land Management (BLM), CTUIR, Oregon Department of Environmental Quality (DEQ), GRMW, Nez Perce Tribe, ODFW, Oregon Water Resources Department (OWRD), Union Soil and Water Conservation District (USWCD), USFS, and U.S. Geological Survey (USGS).

Raw hourly temperature measurements were checked for errors using a combination of automated and manual/visual inspection QAQC routines. Any hourly measurements that exceeded 30 °C, were less than -1 °C, had an absolute hourly change > 3 °C, or had an absolute field audit temperature difference > 0.5 °C were flagged as potential errors and were subsequently verified manually by CRITFC staff.

For each temperature monitoring site, we generated a suite of daily, weekly, and annual water temperature metrics using SQL scripts within CRITFC's centralized database management system (CDMS) that describe key components of stream thermal regimes (e.g., magnitude, variability, frequency, duration, and timing) using guidance from Heck et al. (2018) (Table 5). To avoid errors associated with missing data, metrics were only computed for sites that had valid measurements for at least 90% of the total possible records for a given time period as per Isaak et al. (2017).

We summarized broad spatial and temporal patterns in water temperature across four streams of primary interest to our life cycle modeling work including Catherine Creek, Minam River, Lostine River, and upper Grande Ronde River (Figure 6 - Figure 9). Temperature metrics were presented as boxplots by river kilometer, with boxplots representing the variability across all years of available data for a given site (e.g., horizontal line = median, box = 25th – 75th percentile [interquartile range], and whiskers = min and max values within 1.5 times the interquartile range). This summary analysis included sample years 1991 – 2022 for Catherine Creek and upper Grande Ronde, and 1995 – 2022 for Lostine and Minam. The year range differed among monitoring sites and metrics depending on the presence of gaps in the annual temperature record. While the period of record varied across sites, this approach was considered appropriate to capture broad-scale temperature patterns across each basin. Temperature metrics at the mouths of tributaries were also shown to provide spatial orientation and information about potential thermal refugia along the mainstem.

We conducted an exploratory analysis of temporal trends in peak water temperatures (*MWMT*; see Table 5 for list of metric definitions) at selected sites in the upper Grande Ronde River that had relatively long time series to get a general sense for how summer temperatures have been changing over time. We used locally estimated scatterplot smoothing (LOESS) regression to visualize temporal trends, recognizing that a rigorous and statistically defensible analysis of trends will require incorporation of temporal autocorrelation—an approach that we plan to implement in the future.

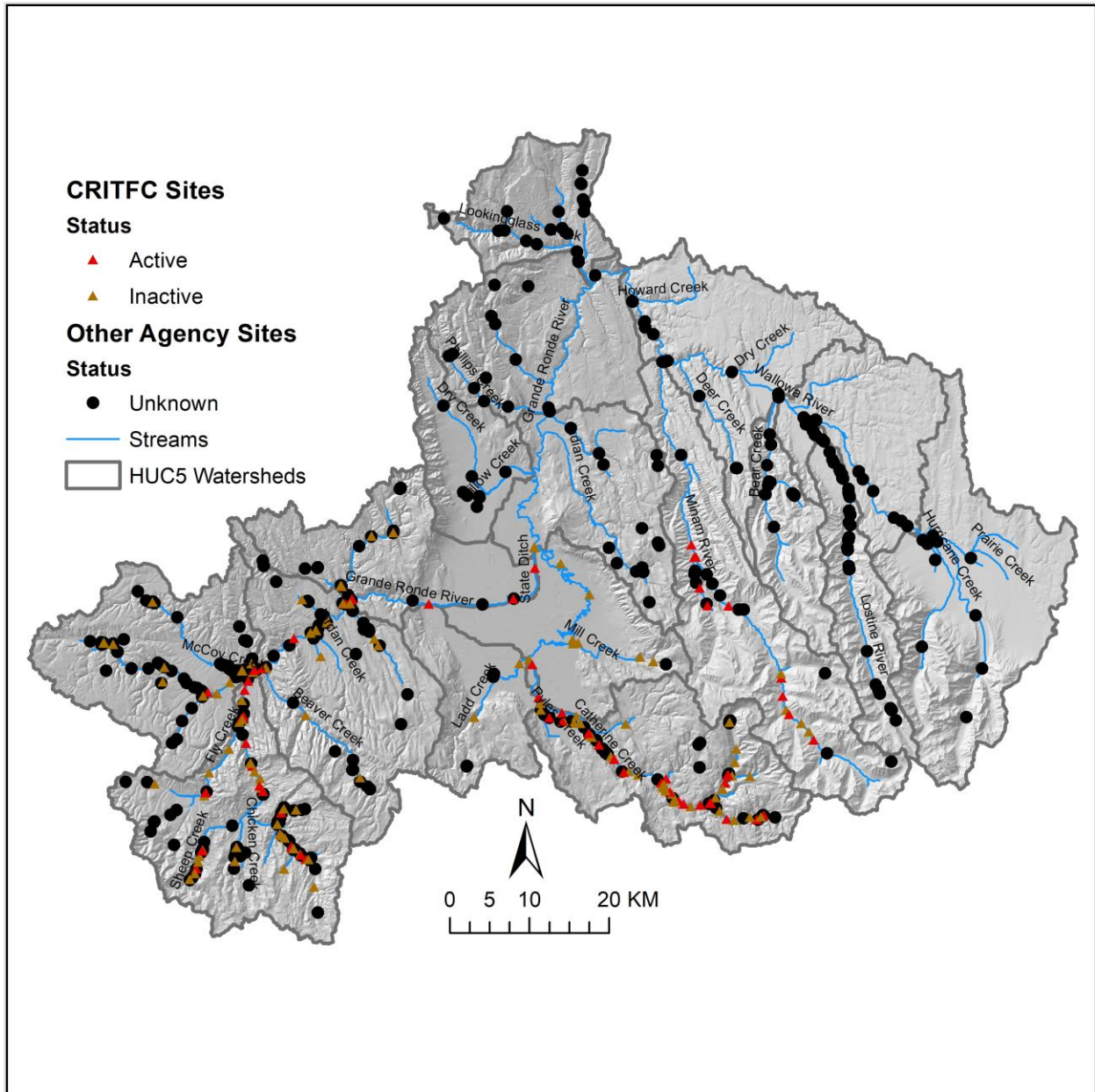


Figure 5. Water temperature monitoring sites in the Grande Ronde River basin from 1988 to 2022.

Table 5. Annual water temperature metrics calculated in the CRITFC water temperature database from Heck et al. (2018).

Category	Metric	Definition
Magnitude (°C)	<i>Maximum</i>	Warmest temperature of the year
	<i>MWMT</i>	Maximum Weekly Maximum Temperature (i.e., highest 7-day average of daily maximum temperatures in summer [Jul. 1 - Aug. 31])
	<i>MWAT</i>	Maximum Weekly Average Temperature (i.e., highest 7-day average of daily mean temperatures in summer [Jul. 1 - Aug. 31])
	<i>DegreeDays</i>	Sum of daily mean temperatures > 0 °C in a year
	<i>JuneAvg</i>	Average of daily mean temperatures in June
	<i>JulyAvg</i>	Average of daily mean temperatures in July
	<i>AugAvg</i>	Average of daily mean temperatures in August
	<i>SeptAvg</i>	Average of daily mean temperatures in September
Variability (°C)	<i>MeanRange</i>	Difference between the highest and lowest daily mean temperature in a year
	<i>MaxRange</i>	Difference between the highest and lowest daily maximum temperature in a year
	<i>MeanVariance</i>	A statistical measure of deviations from the mean among daily mean temperatures in a year
	<i>MaxVariance</i>	A statistical measure of deviations from the mean among daily maximum temperatures in a year
Frequency (n)	<i>Days>16</i>	Number of days in a year where the daily maximum temperature exceeded 16 °C
	<i>Days>18</i>	Number of days in a year where the daily maximum temperature exceeded 18 °C
	<i>Days>20</i>	Number of days in a year where the daily maximum temperature exceeded 20 °C
Duration (n)	<i>CD>16</i>	Consecutive number of days in a year where daily maximum temperature exceeded 16 °C
	<i>CD>18</i>	Consecutive number of days in a year where daily maximum temperature exceeded 18 °C
	<i>CD>20</i>	Consecutive number of days in a year where daily maximum temperature exceeded 20 °C
Timing	<i>CTD50_Date</i>	Date of attaining 50% of the cumulative degree days in a given year
	<i>CTD75_Date</i>	Date of attaining 75% of the cumulative degree days in a given year

Results

Maximum weekly maximum temperatures (*MWMT*) followed a typical warming trend in the downstream direction among the four rivers we examined (Figure 6). The Lostine River was coolest overall (range =

13.6 – 21.7 °C, mean = 18.3 °C), followed by the Minam River (range = 13.1 – 26.8 °C, mean = 20 °C), Catherine Creek (range = 17.2 – 27.2 °C, mean = 21.9 °C) and finally the upper Grande Ronde River (range = 12.4 – 30.4 °C, mean = 23.9 °C). Peak temperatures in the upper Grande Ronde River exhibited particularly rapid warming from the headwaters (~ rkm 290) downstream about 30 km to the outlet of Vey Meadows (~ rkm 264) below which temperatures were near or exceeded the upper incipient lethal limit for salmonids of 25 °C (McCullough et al. 2001). Many of the tributaries entering the upper Grande Ronde River and Catherine Creek provided substantially cooler temperatures than the adjoining mainstem such as Little Catherine Creek, Rock Creek, Jordan Creek, Beaver Creek, Fly Creek, and Clear Creek, though *MWMT* tended to be more variable among years in these tributaries.

Variance of daily maximum temperatures within a year was notably higher in the upper Grande Ronde compared with the other rivers of interest, particularly downstream of Vey Meadows (~ rkm 264; Figure 7). In contrast, daily max temperatures were much less variable in the Lostine River and the Minam River, and moderately variable in Catherine Creek. Similar to *MWMT*, temperature variance generally increased in a downstream direction, although there was some deviation from this trend, especially in the Grande Ronde River.

The frequency of days in which water temperatures exceeded 18 °C—a stressful threshold for rearing salmonids (EPA 2003)—followed similar patterns as *MWMT* (e.g., high in upper Grande Ronde (range = 0 – 114 days; mean = 61.2 days), moderate in Catherine Creek (range = 0 – 107 days, mean = 51.4 days) and low in Lostine (range = 0 – 62 days, mean = 20.6 days) and Minam (range = 0 – 87 days, mean = 30.7 days; Figure 8). As with the other temperature metrics, the number of days exceeding 18 °C increased in a downstream direction and was 0 or near 0 in the upper portions of Lostine and Minam Rivers.

In contrast with the other temperature metrics we examined, the day of year that the cumulative degree days achieved the 50th percentile (*CTD50_Date*) didn't exhibit a consistent spatial trend across rivers (Figure 9). The *CTD50_Date* in the upper Grande Ronde and Lostine rivers appeared to arrive earlier in a downstream direction, consistent with earlier snowmelt in lower elevation portions of the watersheds. However, while *CTD50_Date* in Catherine Creek and Minam River varied somewhat across sites, there was no apparent longitudinal trend along the river corridor. Water temperatures warmed earlier on average in the upper Grande Ronde River (*CTD50_Date* range = 193 – 221 [July 12 – Aug 9], mean = 206 [July 25]), compared with Catherine Creek (range = 192 – 248 [July 11 – September 5], mean = 208 [July 27]), Minam River (range = 202 – 220 [July 21 – August 8], mean = 210 [July 29]), and Lostine River (range = 200 – 224 [July 19 – August 12], mean = 212 [July 31]), though the differences were minor.

Temporal patterns in peak summer temperatures across selected sites in the upper Grande Ronde River mainstem since 1992 were very similar across sites but no overall trend (positive or negative) was apparent (Figure 10). *MWMT* reached a high point around 2005-2007, declined to a low point around 2010-2011 and then steadily increased to another high point around 2015. Temperatures have remained near that high level since 2015. It is likely that changes in water temperature in the Grande Ronde basin over last 30 years will differ substantially by river and the history of land use and degradation/restoration in a given area. Additionally, some aspects of a river's thermal regime (i.e., magnitude, variation, timing, frequency, and duration) may vary substantially over time, while others may not. We intend to do a more in-depth exploration of these temporal trends in the 2023 annual report.

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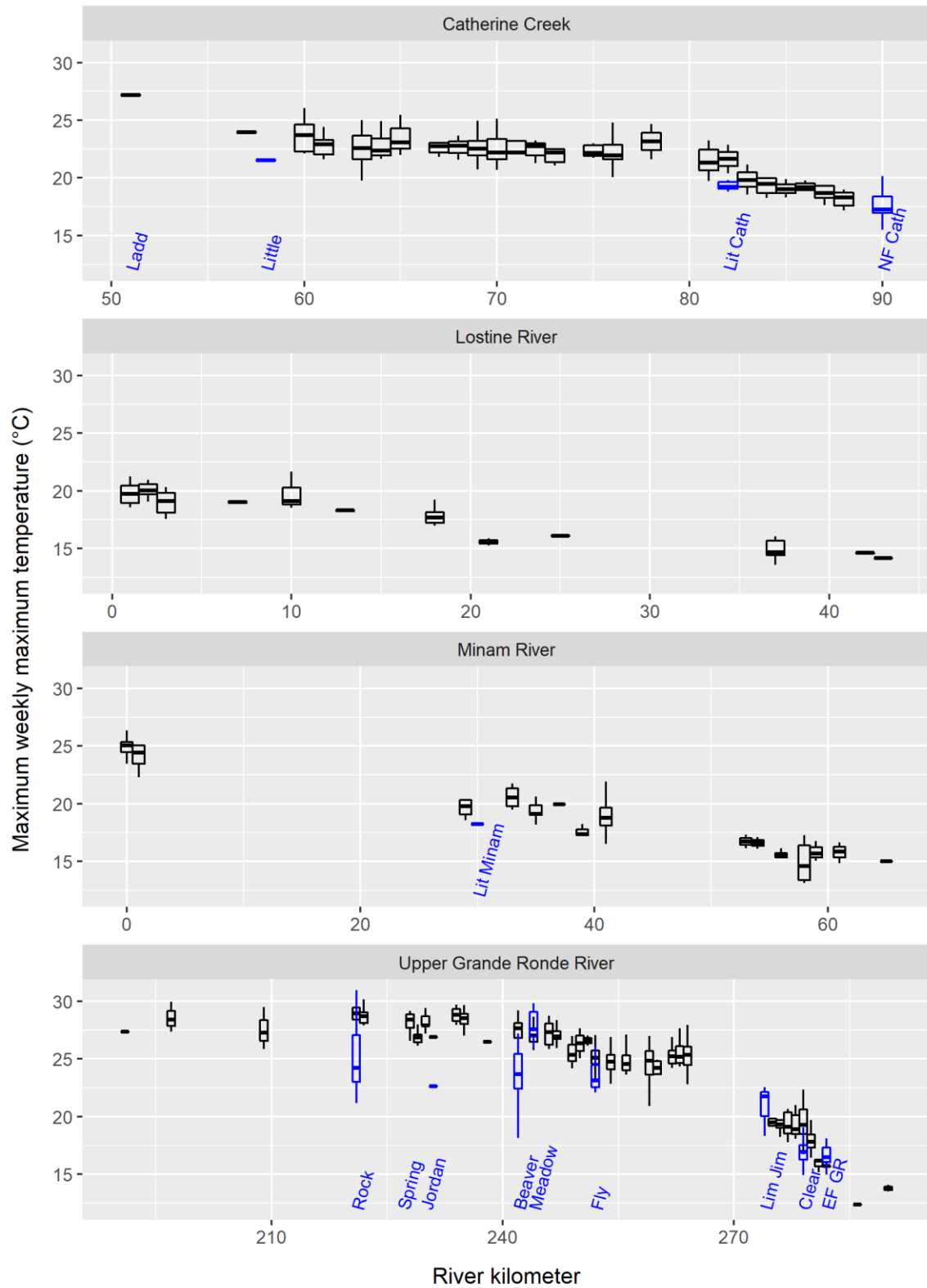


Figure 6. Maximum weekly maximum water temperature (°C) in Catherine Creek (1992-2022), Lostine River (1999-2021), Minam River (1997-2021), upper Grande Ronde River (upstream of Catherine Creek; 1992-2022) and mouths of associated tributaries (shown in blue).

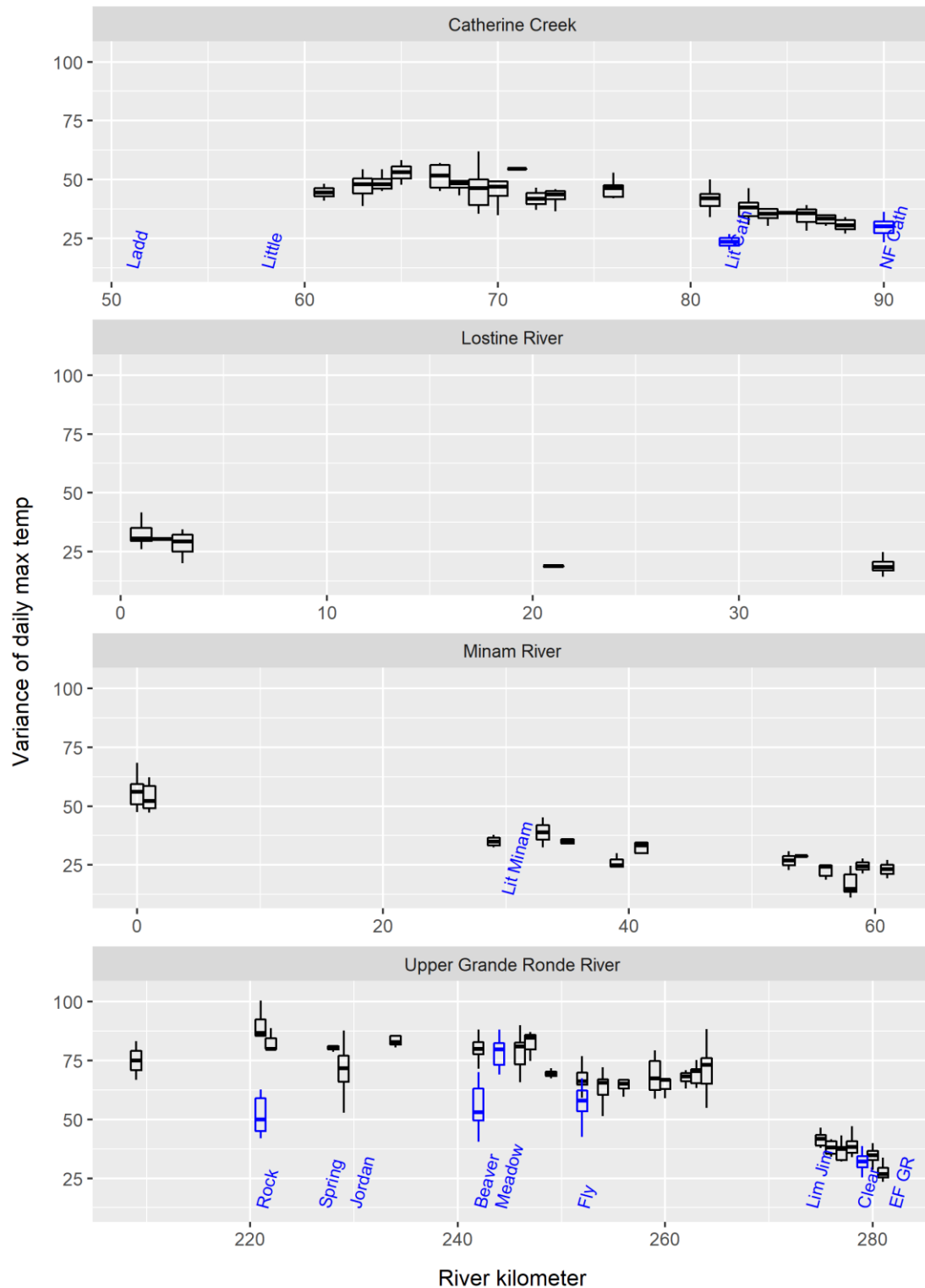


Figure 7. Variance of daily maximum water temperatures in Catherine Creek (1999-2021), Lostine River (2001-2021), Minam River (2003-2021), upper Grande Ronde River (upstream of Catherine Creek; 1999-2021) and mouths of associated tributaries (shown in blue).

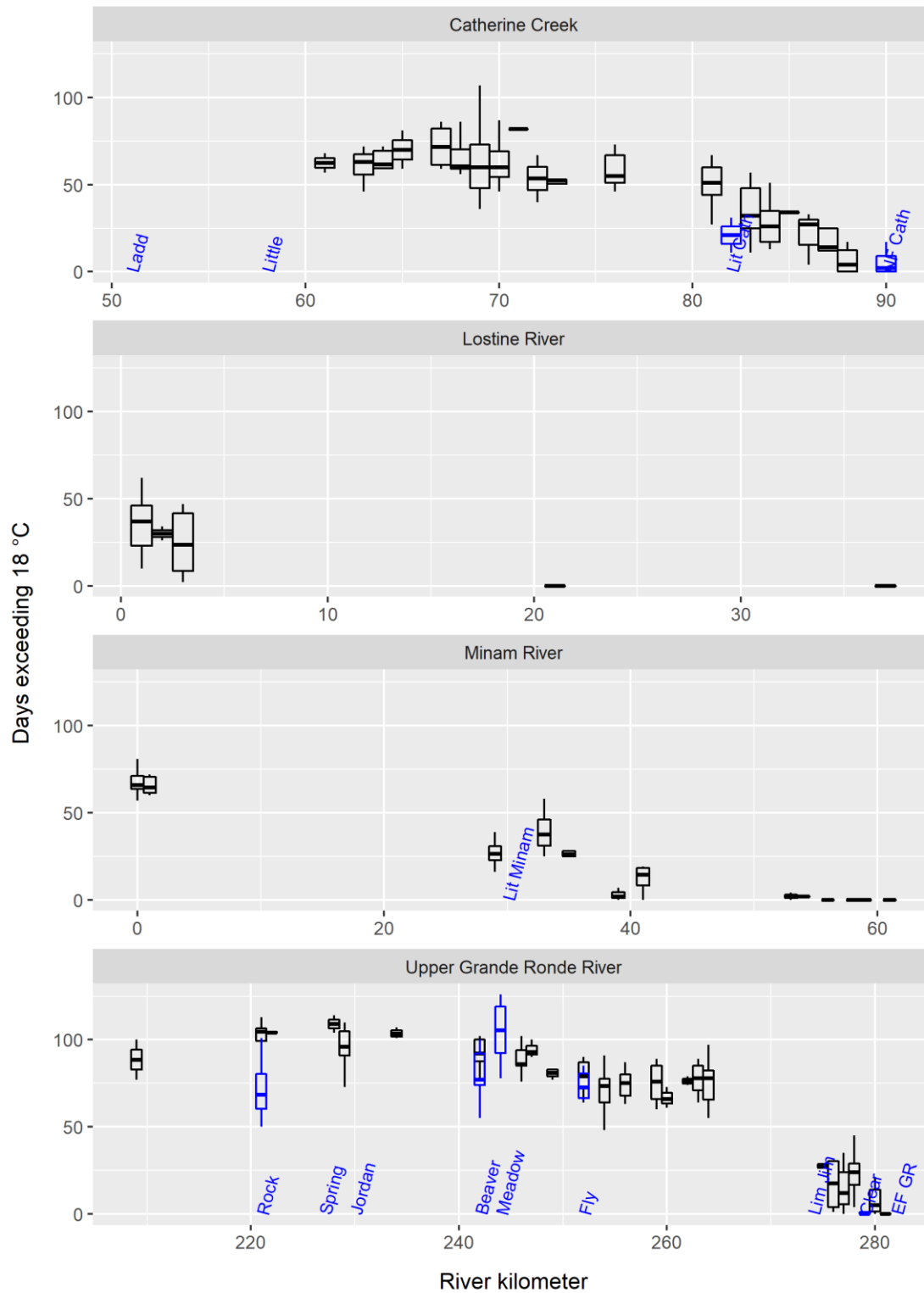


Figure 8. Number of days exceeding 18 °C in Catherine Creek (1999-2021), Lostine River (2001-2021), Minam River (2003-2021), upper Grande Ronde River (upstream of Catherine Creek; 1999-2021) and mouths of associated tributaries (shown in blue).

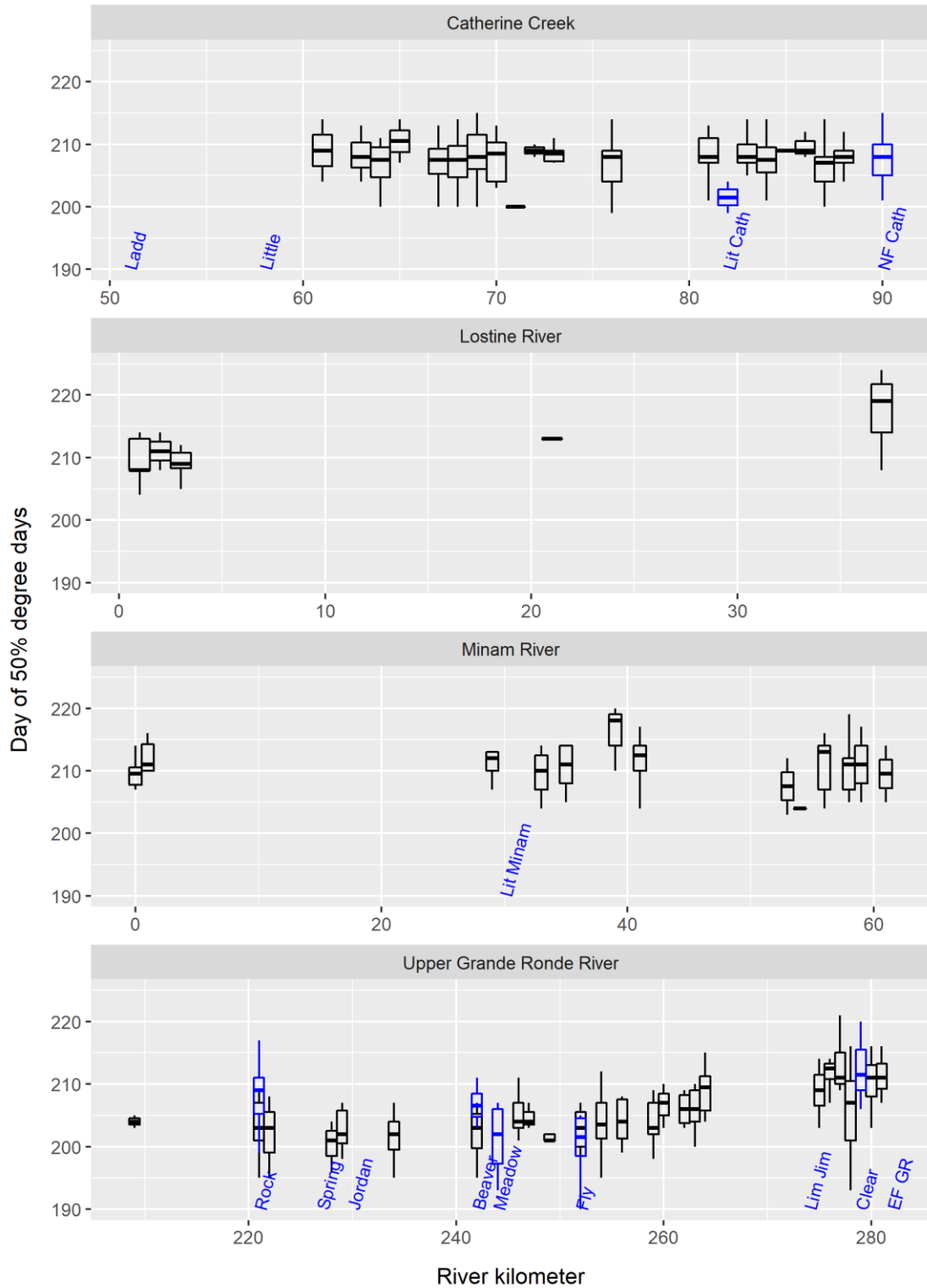


Figure 9. Day of the year that 50% of the cumulative degree days was attained in Catherine Creek (1999-2021), Lostine River (2001-2021), Minam River (2003-2021), upper Grande Ronde River (upstream of Catherine Creek; 1999-2021) and mouths of associated tributaries (shown in blue).

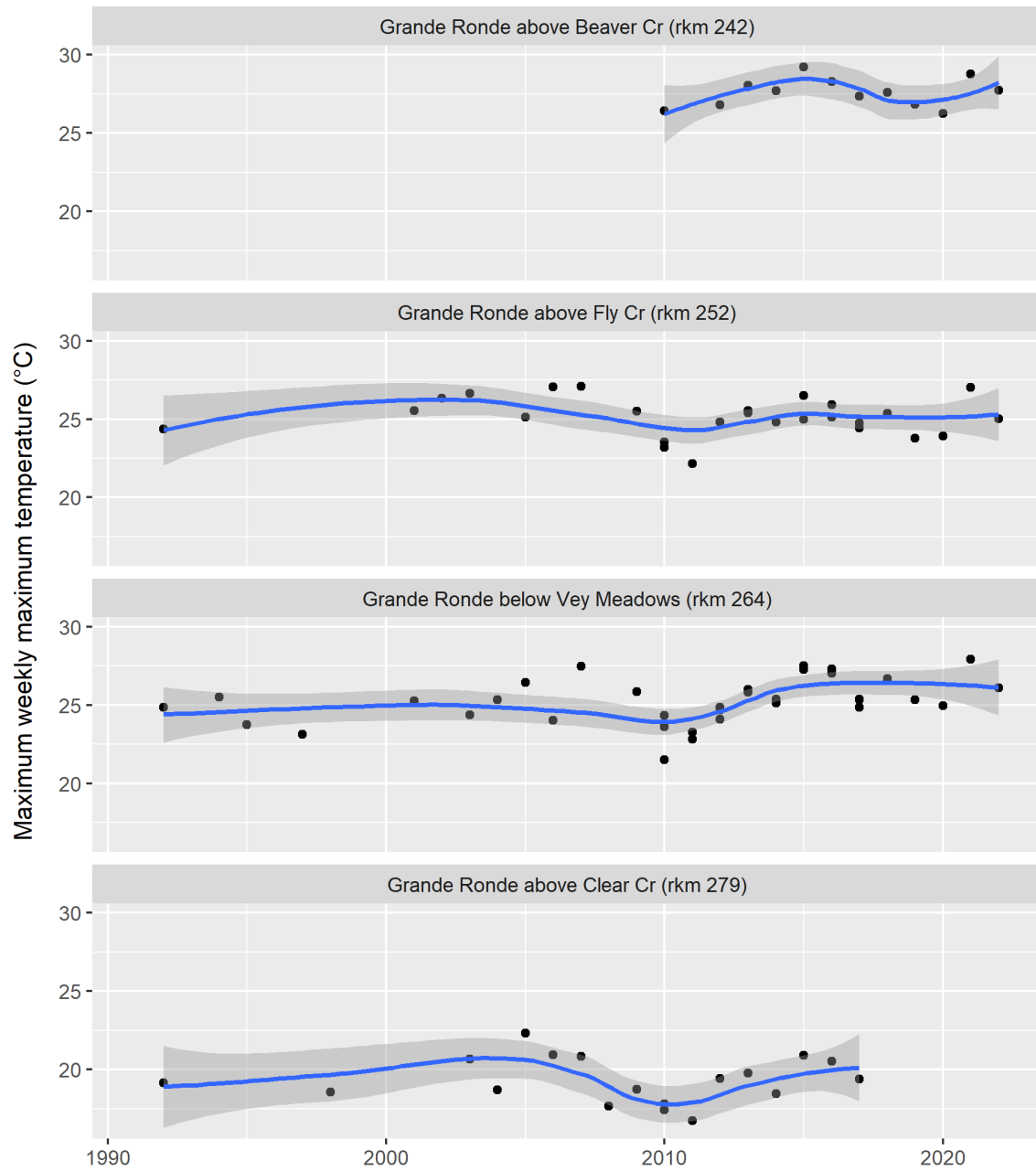


Figure 10. Temporal trends in maximum weekly maximum water temperature (°C) at selected locations along the upper Grande Ronde River. Blue lines represent locally estimated scatterplot smoothing (LOESS) lines and gray ribbons represent 95% confidence intervals. Temperature data for these plots was collected by the USFS and CRITFC.

Objective A-3: Set target values for limiting factors and calculate habitat condition across the upper Grande Ronde basin

This analysis was conducted by former CRITFC Senior Fisheries Scientist and project leader, Seth White, who has since left this role. This chapter contains analytical results and text paraphrased from his writing while employed by CRITFC, used with permission. The analytical methods, results, and inferences presented here are preliminary.

Background

The output of the life cycle model (Objective E) will be used to parameterize simulation models of the populations it includes. These simulations will be designed to evaluate which types of restoration actions will be most likely to result in population recovery. The LCM will estimate the relationship(s) between available habitat and population dynamics parameters (e.g., growth rates, parr rearing capacity). We can then use these relationships to predict the effect of changing habitat availability resulting from restoration on population dynamics, recovery goals, and future status.

To reflect restoration actions in such a simulation model, we will need to have an indication of how much the habitat will change under restoration – summarizing our analyses to date on this topic is the purpose of this chapter. While habitat change is exceedingly difficult to predict, we do have the ability to summarize habitat features throughout the Columbia River basin collected as part of the CHaMP (CHaMP 2016). Summarizing this distribution, stratified by either a land-use or geomorphic classification, and selecting an upper percentile would yield estimates of plausible values of habitat variables that could be achieved through restoration activities.

Methods

The objective is to quantify the value of habitat variables associated with high quality habitat found within the Columbia River basin across a range of river geomorphologies and/or land-use classifications. The original plan was to use multivariate analyses to find the types of classifications that best discriminate habitat conditions. We evaluated multiple classification types, including:

1. Riparian Condition Assessment (Macfarlane et al. 2018) – a composite classification incorporating land use intensity, floodplain accessibility, and riparian vegetation departure.
2. Historical-based Reference (White et al., unpublished) – A site-based classification of watershed road density thresholds influencing whether large pools had increased or decreased from the 1930s-1940s to the present.
3. Landscape Classification of PNW Hydraulic Units (Whittier et al. 2011) – A composite classification incorporating natural characteristics (landform, geology, and stream form) and human disturbance (urban and agricultural land use, impervious surface, and road density).
4. Protected Areas – Classification of protected areas in the U.S. from the USFS Gap Analysis Project
5. Channel Type – A combination of two geomorphic classifications (Montgomery and Buffington 1997; Beechie and Imaki 2014)

CHaMP data were analyzed in the context of each of the above classifications alone, and each in combination with the channel type geomorphic classification. The CHaMP-measured habitat conditions we chose to focus on were large wood density (pieces in bankfull channel/100m) and the frequency of medium-sized pools (>20m² surface area, >0.5m maximum depth; number of pools/km; McIntosh 2000). We chose

these variables for now since they are two key physical habitat variables in our habitat capacity model (Justice et al., unpublished).

Performing the original discrimination of habitat conditions analysis as planned did not yield useful results. Preliminary analyses indicated that none of the above classifications helped to describe variability in the habitat conditions. We thus chose to perform a simpler, more focused analysis: to summarize the distribution of these variables grouped by channel type by their 90th percentile. We selected the 90th percentile as the target value within channel types to ensure we are targeting restoration to return the habitat in the Grande Ronde basin to be among the top found within the Columbia River basin. This distribution was taken over the CHaMP survey data collected across the 2,231 site visits that occurred between 2011-2017 and spanned 11 Columbia River subbasins as well as Region 17.

To place the existing conditions in two Grande Ronde subbasins (upper Grande Ronde and Minam) in the context of the target values, we also calculated how far the median is (relatively speaking) from the target value, by channel type. This calculation was performed as: $(\text{CRB 90th} - \text{subbasin median}) / \text{subbasin median}$; we used the median for comparison rather than the 90th percentile because we wanted to express, relative to the current average site, how much habitat change may be needed to bring the average site up to the target value – this also explains why the subbasin current median is in the denominator.

Results

There was substantial variability in the pool (Figure 11) and large wood (Figure 12) variables both across sites within subbasins, and in the shape of the distribution across subbasins. For both variables (and large wood in particular, Figure 12), most sites had values on the lower end of the distribution in all populations; notable exceptions are for pool frequency in South Fork Salmon, Tucannon, and Yankee Fork subbasins which had less skewed distributions (Figure 11).

There was also substantial variability among sites when stratified by channel type, with most values focused on lower values and a long tail of rare but large values; this was true of both the pool (Figure 13) and large wood (Figure 14) variables. There was less variability among channel types in the frequency of large wood than of pools.

The 90th percentile for the pool variable was largest (unsurprisingly) in pool/riffle channel types (33.8 medium pools/km) and lowest for cascade/step channel types (17.1 medium pools/km); other channel types with high frequency of medium pools included meandering, straight, and plane bed (Table 6). Regardless of channel type, the median value of the pool variable for the Minam and upper Grande Ronde subbasins were quite a bit lower than the 90th percentile target value, indicating much restoration effort may be needed to bring the average site up to the target value with respect to pool frequency in most channel types. For example, the median pool frequency in pool/riffle channel types for upper Grande Ronde must increase by 334% if the average site is to be equal to the target (Table 6).

The 90th percentile for the wood variable was largest in cascade/step channel types (33.4 medium pools/km) and lowest for pool/riffle channel types (18.4 medium pools/km); this is the opposite pattern as was found for pool frequency (Table 6). Regardless of channel type, the median value of the pool variable for the Minam and upper Grande Ronde subbasins were again quite a bit lower than the 90th percentile target value.

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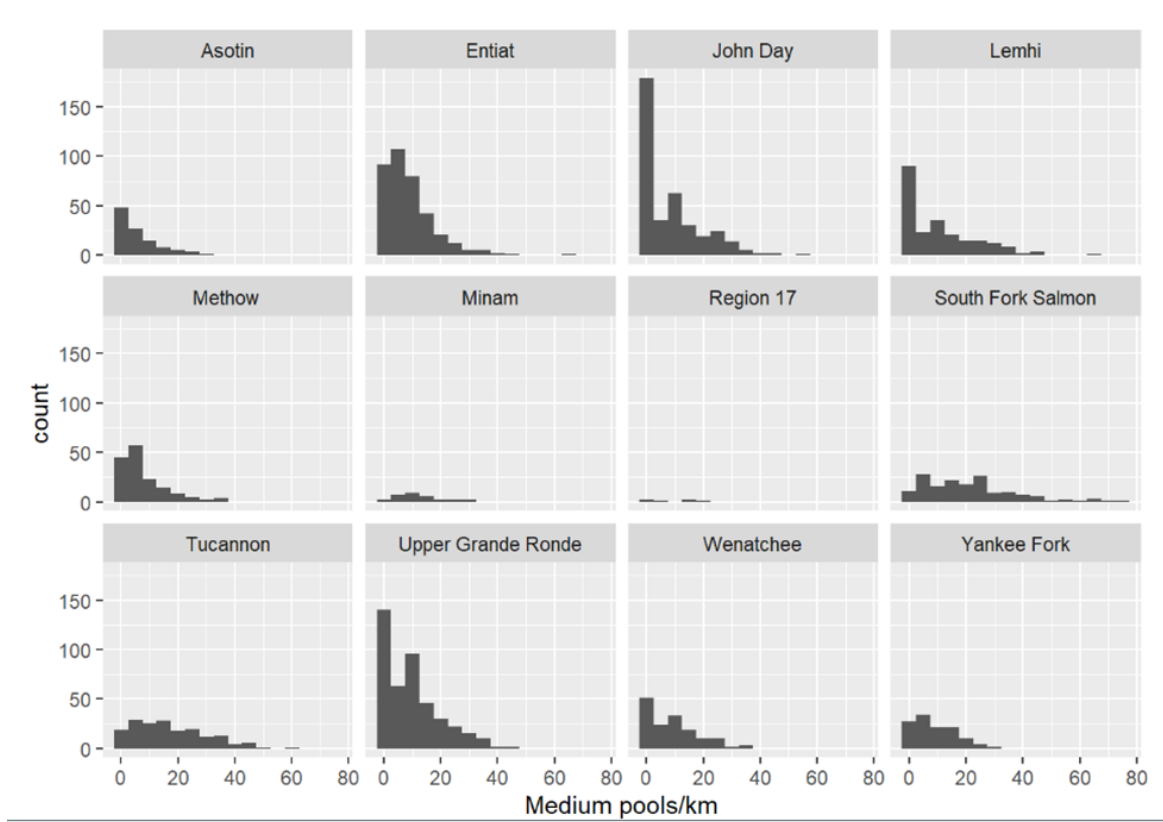


Figure 11. Basin-specific distributions showing frequency of medium-sized pools/km at CHaMP sites aggregated across all channel types.

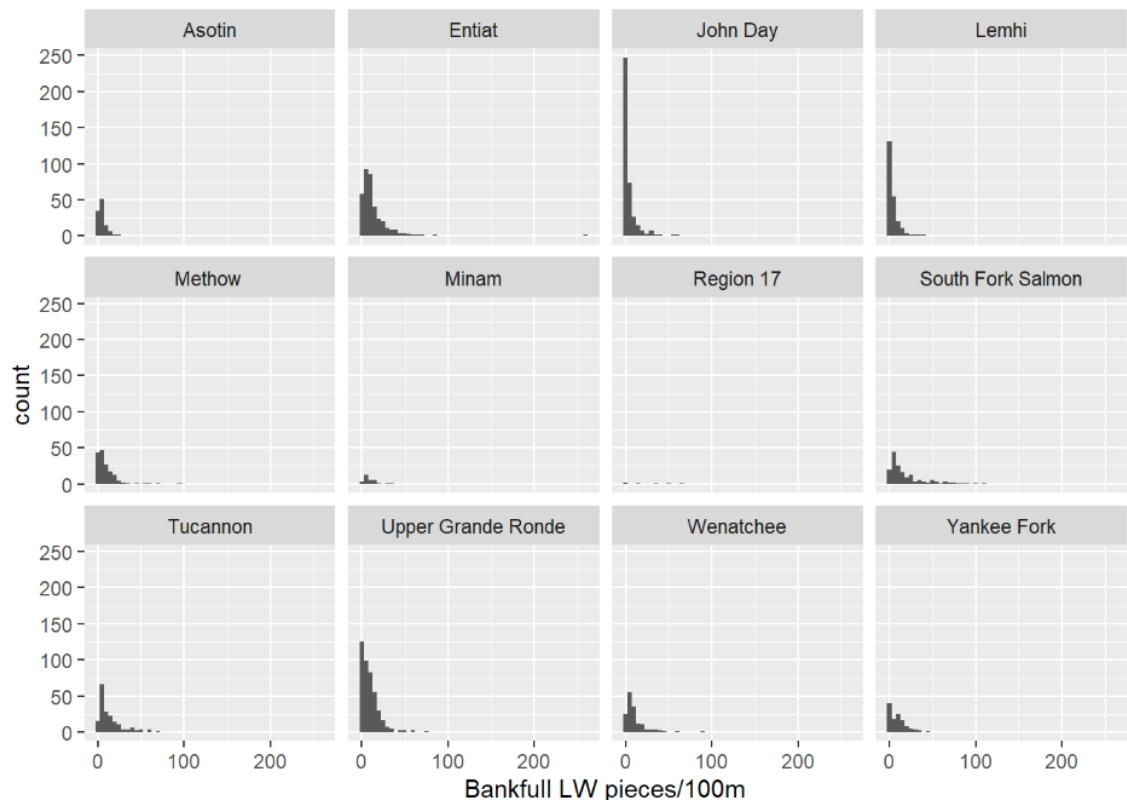


Figure 12. Basin-specific distributions showing the frequency of large wood/100m at CHaMP sites aggregated across all channel types.

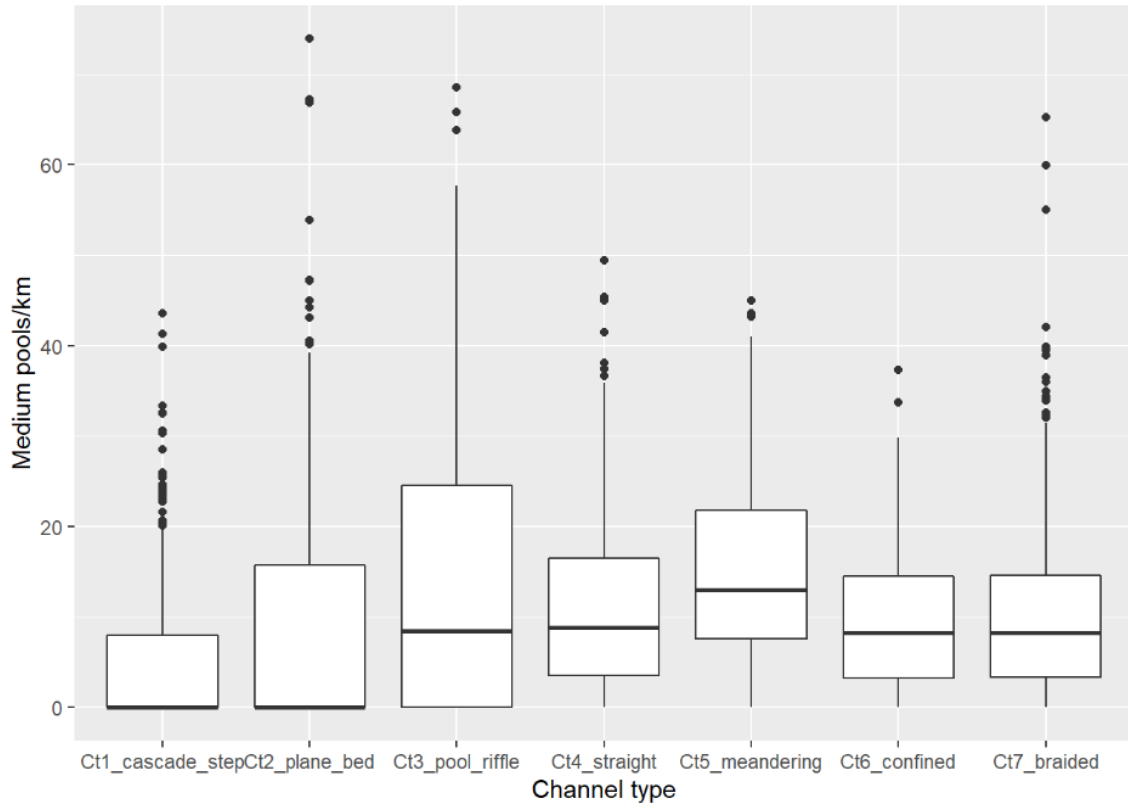


Figure 13. Channel type-specific distributions of the number of medium-sized pools/km aggregated across all subbasins.

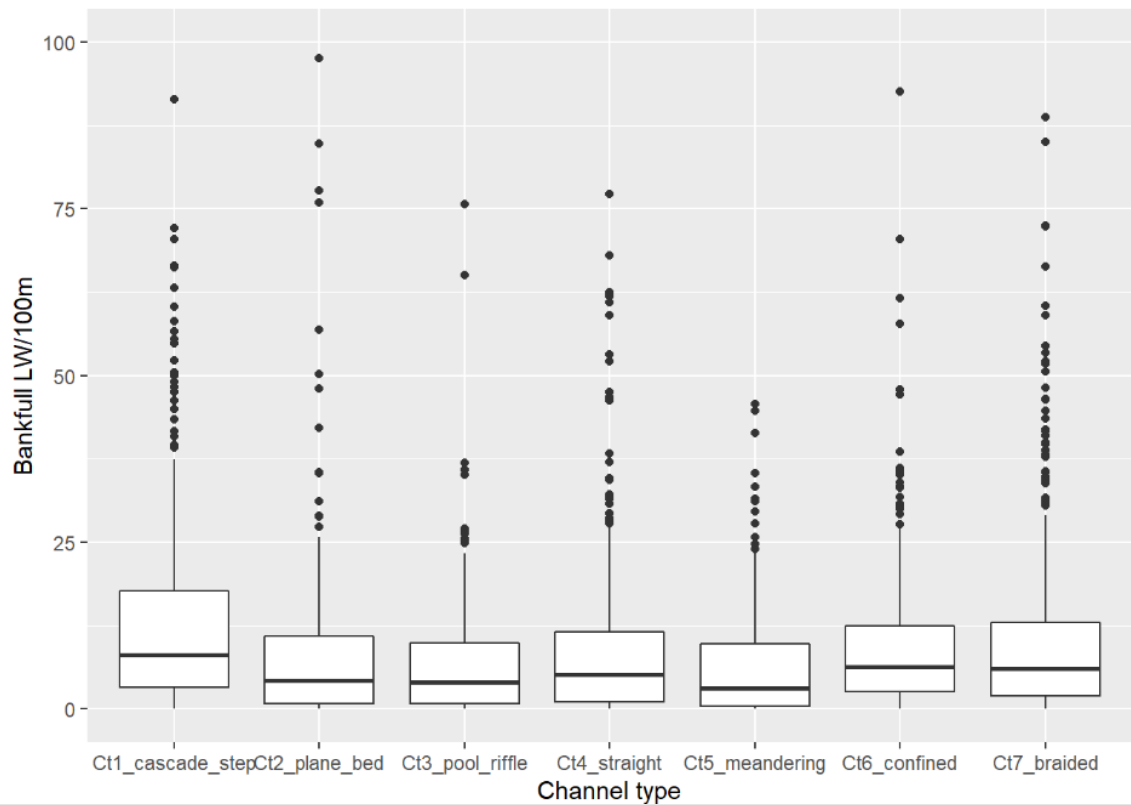


Figure 14. Channel type-specific distributions of the number of large wood peices/100m aggregated across all subbasins.

Table 6. The 90th percentile of two CHaMP survey-derived variables aggregated across all subbasins, but separated by channel type (CRB), and where the median of two Grande Ronde subbasins fall in relation. "CRB" represents the 90th percentile of all available CHaMP records and the "%Δ from Median" show the result of the calculation: $[(CRB\ 90^{th} - \text{subbasin median})/\text{subbasin median}]$. Cells with a "—" indicate no sites with that channel type in that subbasin were available for analysis.

Channel Type	Medium Pools/km			Large Wood Pieces/100m		
	CRB	%Δ from Median		CRB	%Δ from Median	
		Minam	Upper GR		Minam	Upper GR
Cascade/Step	17.1	—	589%	33.4	—	145%
Plane Bed	26.6	—	—	21.0	—	120%
Pool/Riffle	33.8	—	334%	18.4	—	71%
Straight	28.3	102%	249%	26.5	99%	392%
Meandering	30.5	—	239%	24.0	—	1000%
Confined	22.9	237%	156%	27.3	439%	254%
Braided	23.5	140%	55%	28.39	594%	421%

Objective A-4: Habitat status assessment (LiDAR)

In summer 2020, CRITFC, in collaboration with GRMW and BPA, acquired topobathymetric (“green”) LiDAR across 76,188 acres of the Grande Ronde River and Wallowa River watersheds, including tributaries currently or historically occupied by Chinook Salmon and steelhead. Data were collected by the contractor Quantum Spatial, Inc. (QSI, formerly Watershed Sciences, Inc.) and detailed results of imagery acquisition, QA/QC, image processing, and a list of post-processed deliverables are available in a report (NV5 Geospatial 2021). We are in the process of developing procedures and a workflow to describe spatially continuous instream fish habitat metrics and floodplain conditions, including developing and refining methods to classify large wood from LiDAR points. We expect to complete this task by FY2024.

Goal 2: Evaluate effectiveness of aggregate restoration actions in the Grande Ronde basin

Objective C: Evaluate if the pace of restoration can counteract habitat and temperature degradation

Objective C-1: Develop list of restoration scenarios

CRITFC staff hosted a collaborative workshop in November of 2021 with Grande Ronde basin partners to develop a suite of riverscape restoration and management scenarios intended to be used as inputs to the Grande Ronde spring Chinook life cycle model described in Objective D. The group developed a draft set of broad management scenarios addressing instream and floodplain restoration, riparian vegetation restoration, food webs, and other in-basin factors (i.e., landowner access, lower valley mortality, hatchery supplementation). Workshop participants discussed a more detailed list of actions that could be implemented to address each of these broad management categories. At this stage, these restoration scenarios are still in development and require additional feedback from basin partners and are therefore not included in this report. Our goal is to finalize the list of management scenarios and apply them to the life cycle model during FY2023 and early FY2024.

Objective C-2: Quantify change in habitat limiting factors relative to restoration actions

To address a critical gap in our ability to translate restoration actions into habitat change across large geographic extents (e.g., biologically significant reaches [BSRs], Chinook population areas), we plan to quantify the observed change in habitat conditions before and after restoration at restoration sites within the study area. This analysis will focus on changes in key habitat limiting factors (e.g., water temperature, pool frequency, large wood frequency, and floodplain/side channel habitat) in response to restoration actions across a range of restoration categories (e.g., barrier removal, large wood placement, riparian planting, floodplain enhancement; Roni et al. 2021). Expert opinion or literature-based estimates of habitat change may be used for some restoration categories if local empirical data is insufficient. While no progress was made on this objective in FY2022, we plan to complete this objective in late FY2023 or early FY2024.

References

Roni, P., M. Krall, K. Ross, and D. Arterburn. 2021. Action Effectiveness Monitoring 2020 Annual Report. Page 121. Cramer Fish Sciences, Annual Report Project Number 2016-001-00, Issaquah, WA.

Objective C-3: Extrapolate impacts of restoration scenarios across historical Chinook extent in upper GR basin

Using observed changes in habitat characteristics at selected restoration sites from Objective C-2, we plan to extrapolate the expected uplift to habitat conditions throughout the stream network in the study area. The resulting product will be a spatially explicit map of habitat uplift that can be used independently in restoration planning, and also incorporated via simulated restoration scenarios in the life cycle model (Objective E-2, E-3). This objective is planned to be completed in FY2024.

Goal 3: Relate biological responses to habitat change

Objective D: Relate life stage-specific response to habitat change in the Grande Ronde and nearby basins

Objective D-1: Collect data on fish and macroinvertebrate distribution

Juvenile salmonid abundance and distribution

Background

Snorkel density surveys were conducted across a subset of habitat monitoring survey segments to assess fish response to habitat quality, and to evaluate the effectiveness of aggregate restoration activities in improving freshwater habitat conditions and viability of ESA-listed fish species at the population scale. Snorkel surveys conducted during the summer of 2021 contribute to large-scale monitoring efforts across the upper Grande Ronde basin, with the intent of assessing status and trends in fish habitat conditions, particularly for Chinook Salmon (*Oncorhynchus tshawytscha*).

Methods

During the summer of 2021, snorkel surveys were conducted between August 11th and August 17th to assess the abundance and distribution of juvenile salmonids in summer rearing habitat in the upper Grande Ronde and Sheep Creek (Figure 15). Snorkel surveys were conducted at 22 monitoring reaches in the upper Grande Ronde River, with reach lengths of approximately twenty times the bankfull channel width, ranging from 158 – 732 m (mean = 365 m). Surveys were conducted following a protocol developed by CRITFC, drawing heavily from the methods of Thurow (1994) and integrated with the Pacific Northwest Aquatic Monitoring Program (PNAMP) methods. Details about the snorkel survey methodology can be found in White et al. (2012).

The resulting data from snorkel surveys, corresponding ground-based habitat data, and unmanned aircraft systems (UAS)-based spatial data were post-processed and metrics relating to habitat characteristics influencing snorkel detection efficiency were calculated. Derived habitat metrics include (at the channel unit scale) the density of large wood pieces within the wetted channel and average depth. These habitat metrics, paired with juvenile snorkel count data for Chinook Salmon and Steelhead/Rainbow Trout (hereafter referred to as *O. mykiss*), were used to estimate the probability of detection at the channel unit scale with channel units categorized as either slow (pools) or fast (fast turbulent and fast non-turbulent) using a Bayesian detection probability model that accounts for uncertainty in both snorkel count and mark-recapture estimates, as well as mechanistic links between local conditions and detectability for each species (Staton et al. 2022). Snorkel counts were then expanded using the predicted detection probability to estimate true abundance.

The CRITFC snorkel survey protocol samples only 25% of riffle or fast turbulent channel units within a sample reach, requiring that estimates of channel unit abundance be further expanded to account for unsampled channel units as well as for any channel units that were subsampled (i.e., only 80% of the channel unit area snorkeled) as the protocol allows for the subsampling of channel units that are excessively long in length and impractical to sample in their entirety (White et al. 2012). After this further expansion to account for fish present in unsurveyed fast turbulent channel units, estimates were converted to linear density (no.·100m⁻¹) to account for differences in stream size across sites.

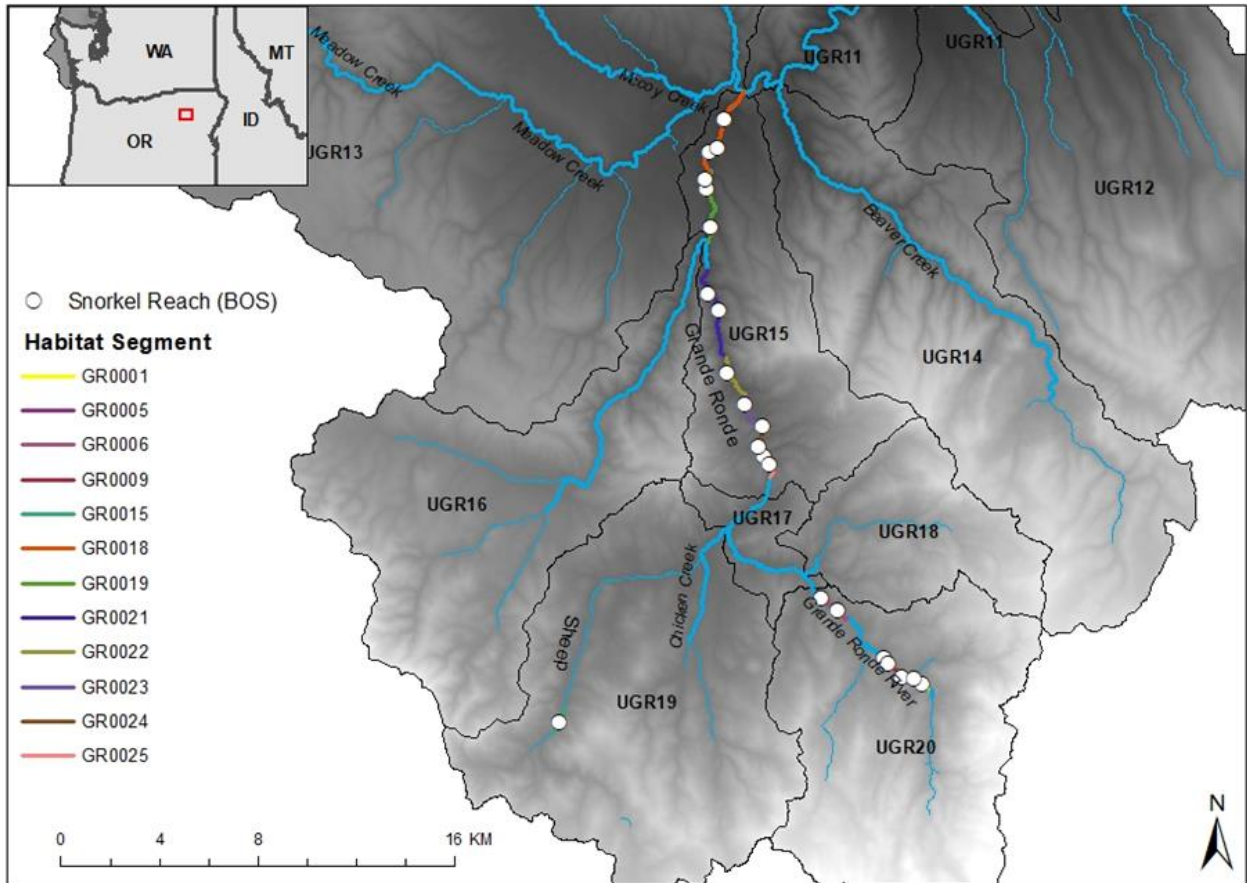


Figure 15. Snorkel reaches (bottom of site) and habitat segment locations for 2021 fish density and distribution sampling.

Results

Estimated juvenile Chinook densities (fish · 100m⁻¹) throughout the reaches monitored in 2021 ranged widely from 0 to 364 (mean = 62), with higher densities occurring in monitoring sites located farther upstream on the Grande Ronde River (Table 5; Figure 16). No juvenile Chinook were observed in two of six reaches surveyed downstream of the confluence with Fly Creek, and no Chinook were observed in the single reach snorkeled in Sheep Creek (Table 8).

Juvenile Steelhead/Rainbow Trout densities ranged from 5 to 203 (mean = 65), with slightly more variable patterns in estimated density relative to juvenile Chinook Salmon (Table 8; Figure 17). Juvenile *O. mykiss* snorkel counts categorize fish observed as either less than 80 mm or greater than 80 mm, which allows for monitoring of young of the year/age one individuals as well as an overall count of juvenile fish. One caveat, however, is that this methodology necessitates that adult resident Rainbow Trout count data be included with juvenile Steelhead. The extent to which the inclusion of adult resident individuals influences reach level estimates of juvenile *O. mykiss* density is presumed to be minimal due to the high rate of anadromy observed in *O. mykiss* populations in the Grande Ronde River. *O. mykiss* less than 80 mm in length were observed less frequently than individuals greater than 80 mm in length, with a mean density of 37 per 100 m (Table 10).

Table 7. Biologically significant reach (BSR) scale mean estimated fish density (fish ·100m⁻¹) and fish density range for reaches snorkeled during the 2021 field season for juvenile Chinook Salmon and all size classes of Steelhead/Rainbow Trout.

BSR	Stream	Reaches Sampled		Mean Density (fish/100 m)	SD	Density Range (fish/100 m)
UGR15	Grande Ronde	14	<i>Chinook</i>	7.1	7.87	0 – 20.5
			<i>Steelhead</i>	33.0	28.84	5.3 – 79.6
UGR19*	Sheep Creek	1	<i>Chinook</i>	0	NA	NA
			<i>Steelhead</i>	68.4	NA	NA
UGR20	Grande Ronde	7	<i>Chinook</i>	182.2	155.46	6.8 – 364.7
			<i>Steelhead</i>	101.2	60.36	30.1 – 203.2

* Only one snorkel density reach was sampled in BSR19, and no Chinook were observed.

Snorkel reaches surveyed during the 2021 field season were distributed across three biologically significant reaches (BSR), two of which encompass sections of the upper Grande Ronde River (UGR15 and UGR20) while the third encompasses Sheep Creek (UGR19; Figure 15). UGR15, which comprises the stretch between the confluence with Meadow Creek and below the confluence with Chicken Creek, had the lowest overall density of juvenile salmonids of these three BSRs with ranges of 0 to 21 and 5 to 80 individuals per 100 m for Chinook and *O. mykiss*, respectively (Table 7). UGR20 had the highest densities of both juvenile Chinook and *O. mykiss*, with a mean density of 182 Chinook and 101 Steelhead/Rainbow Trout per 100 m (Figure 16 and Figure 17, respectively). UGR19, located in Sheep Creek, had only one sample site, and as such, no further summary statistics are available beyond those already reported at the snorkel reach scale.

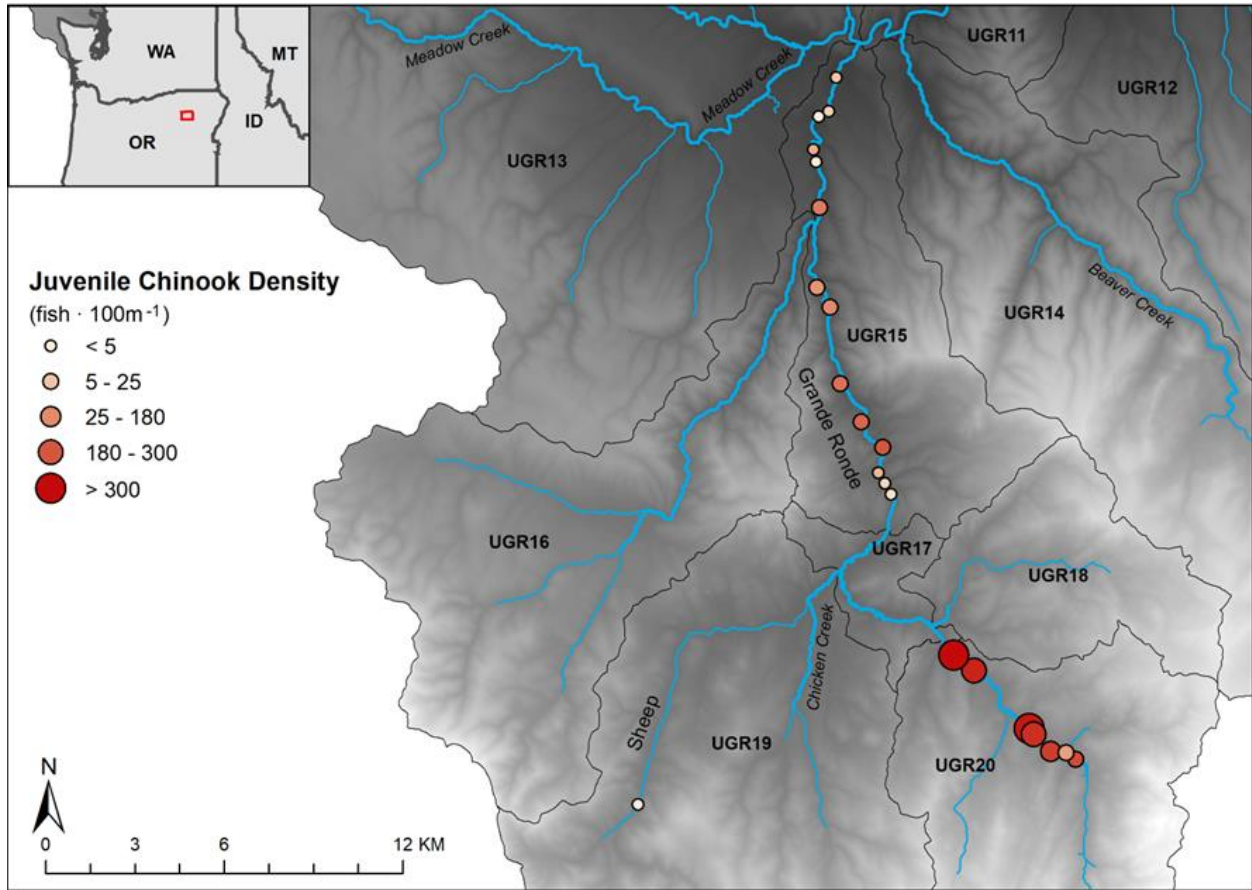


Figure 16. Distribution of juvenile Chinook Salmon density (fish · 100m⁻¹) values in the upper Grande Ronde River and Sheep Creek. Increasing symbol size and shade indicate increasing density.

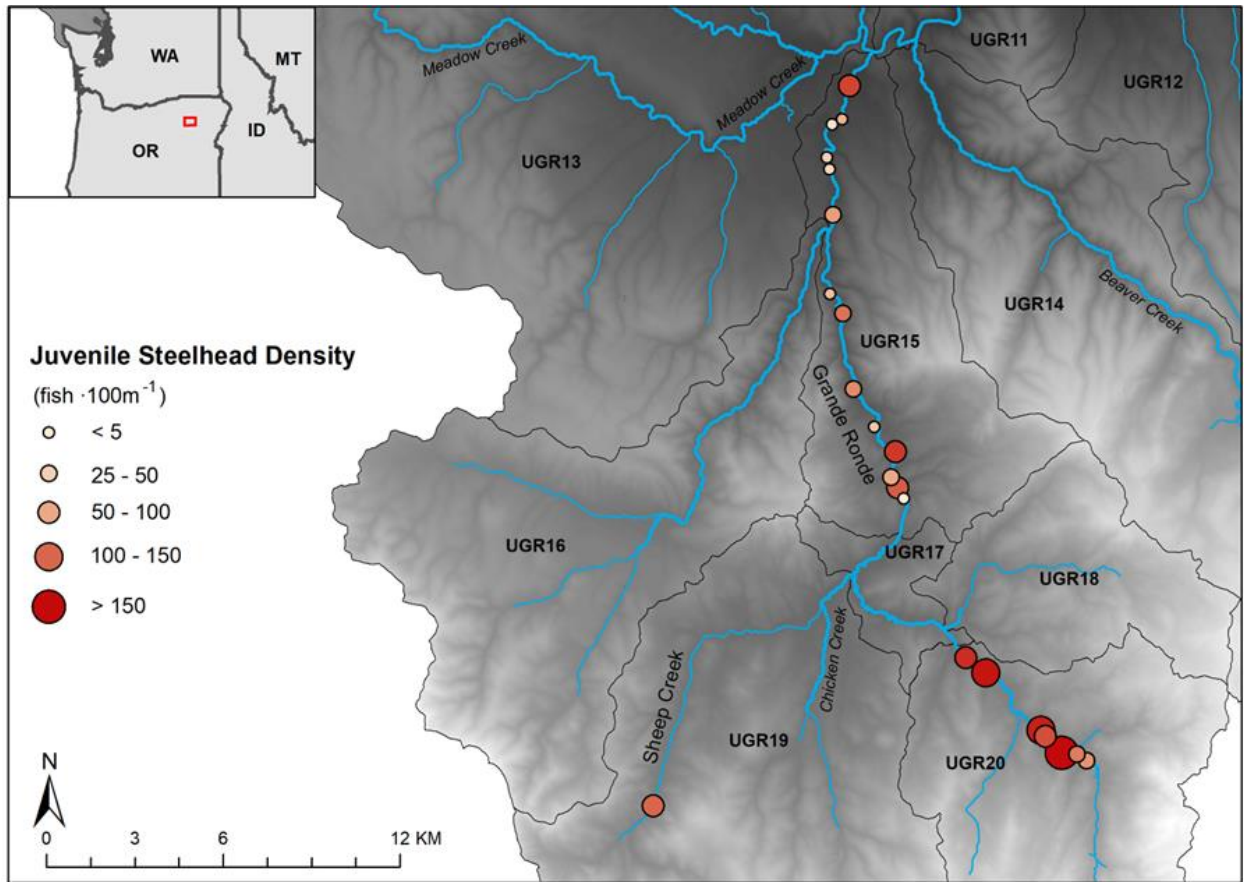


Figure 17. Distribution of juvenile *O. mykiss* density (fish · 100m⁻³) values in the upper Grande Ronde River (21 reaches) and Sheep Creek (1 reach). Increasing symbol size and shade indicate increasing density.

Table 8. Juvenile Chinook Salmon estimated abundance and density (fish ·100m-1) derived from 2021 snorkel density surveys.

Reach	Stream	Total Length (m)	Total Area (m ²)	N	SD	Density (fish/100 m)	SD
CBW05583-031546	UGR	514	19591	69	1.59	13.41	0.31
CBW05583-060218	UGR	274	8775	26	0.41	9.48	0.15
CBW05583-063978	UGR	185	5178	40	0.59	21.53	0.32
CBW05583-099818	UGR	169	2919	59	3.91	34.85	2.31
CBW05583-190266	UGR	290	6541	2	0	0.69	0
CBW05583-203578*	UGR	619	26876	0	NA	0	NA
CBW05583-206314	UGR	196	5387	577	14.91	293.25	7.58
CBW05583-269114	UGR	565	28102	8	0.04	1.42	0.006
CBW05583-280042	UGR	164	3601	520	18.37	316.11	11.16
CBW05583-321338	UGR	448	22685	92	0.47	20.54	0.11
CBW05583-361962	UGR	279	6302	19	0.17	6.80	0.06
CBW05583-366394	UGR	533	37318	86	0.30	16.13	0.06
CBW05583-370490	UGR	490	31866	96	1.47	19.58	0.32
CBW05583-419642	UGR	318	9965	1	0	0.31	0
CBW05583-453434*	UGR	280	8453	0	NA	0	NA
CBW05583-453946*	Sheep	158	3417	0	NA	0	NA
CBW05583-457530	UGR	732	34366	15	0.14	2.05	0.02
CBW05583-468458	UGR	222	8084	811	24.91	364.66	11.20
CBW05583-486202	UGR	420	13154	55	1.71	13.07	0.41
dsgn4-000009	UGR	166	5083	396	9.72	238.41	5.85
dsgn4-000202	UGR	648	30264	8	0.01	1.23	0.002
dsgn4-000277	UGR	367	14404	7	0.04	1.90	0.01

*No juvenile Chinook observed in sample reach.

Table 9. *O. mykiss* estimated abundance and density (fish ·100m⁻¹) derived from 2021 snorkel density count data. Estimates represent total abundance and density across all size classes rather than a single life history stage.

Reach	Stream	Total Length (m)	Total Area (m ²)	N	SD	Density (fish/100 m)	SD
CBW05583-031546	UGR	514	19591	143	11.73	27.79	2.28
CBW05583-060218	UGR	274	8775	50	6.57	18.23	2.40
CBW05583-063978	UGR	185	5178	56	2.79	30.14	1.51
CBW05583-099818	UGR	169	2919	344	89.34	203.19	52.77
CBW05583-190266	UGR	290	6541	226	113	77.74	38.87
CBW05583-203578	UGR	619	26876	35	1.85	5.65	0.30
CBW05583-206314	UGR	196	5387	288	30.99	146.37	15.75
CBW05583-269114	UGR	565	28102	450	25.07	79.61	4.44
CBW05583-280042	UGR	164	3601	192	35.99	116.72	21.88
CBW05583-321338	UGR	448	22685	392	22.15	87.5	4.95
CBW05583-361962	UGR	279	6302	117	9.54	41.85	3.41
CBW05583-366394	UGR	533	37318	191	5.93	35.83	1.11
CBW05583-370490	UGR	490	31866	64	5.73	13.05	1.17
CBW05583-419642	UGR	318	9965	17	0.96	5.35	0.30
CBW05583-453434	UGR	280	8453	23	8.63	8.19	3.07
CBW05583-453946	Sheep	158	3417	108	8.05	68.35	5.09
CBW05583-457530	UGR	732	34366	60	5.77	8.19	0.79
CBW05583-468458	UGR	222	8084	205	30.77	92.18	13.85
CBW05583-486202	UGR	420	13154	189	29.47	44.90	7.00
dsgn4-000009	UGR	166	5083	130	12.39	78.27	7.46
dsgn4-000202	UGR	648	30264	149	9.43	22.97	1.45
dsgn4-000277	UGR	367	14404	101	15.23	27.48	4.14

Table 10. Juvenile *O. mykiss* estimated abundance and density (fish · 100m⁻¹) derived from 2021 snorkel density count data. Estimates represent total abundance and density for *O. mykiss* less than 80 mm in length to provide information regarding young of the year and age one fish.

Reach	Stream	Total Length (m)	Total Area (m ²)	N	SD	Density (fish/100 m)	SD
CBW05583-031546	UGR	514	19591	69	5.66	13.41	1.10
CBW05583-060218*	UGR	274	8775	0	NA	0	NA
CBW05583-063978	UGR	185	5178	20	0.99	10.76	0.54
CBW05583-099818	UGR	169	2919	28	7.27	16.54	4.30
CBW05583-190266	UGR	290	6541	61	30.5	20.98	10.49
CBW05583-203578	UGR	619	26876	5	0.26	0.81	0.04
CBW05583-206314	UGR	196	5387	176	18.94	89.45	9.62
CBW05583-269114	UGR	565	28102	28	1.56	4.95	0.28
CBW05583-280042	UGR	164	3601	32	5.99	19.45	3.65
CBW05583-321338	UGR	448	22685	32	7.81	7.14	0.40
CBW05583-361962	UGR	279	6302	47	3.83	16.81	1.37
CBW05583-366394	UGR	533	37318	118	3.67	22.14	0.69
CBW05583-370490	UGR	490	31866	7	0.63	1.43	0.13
CBW05583-419642	UGR	318	9965	4	0.23	1.25	0.07
CBW05583-453434*	UGR	280	8453	0	NA	0	NA
CBW05583-453946	Sheep	158	3417	61	4.54	38.61	2.88
CBW05583-457530	UGR	732	34366	36	3.46	4.91	0.47
CBW05583-468458	UGR	222	8084	24	3.60	10.79	1.62
CBW05583-486202	UGR	420	13154	8	1.24	1.9	0.30
dsgn4-000009	UGR	166	5083	17	1.62	10.23	0.98
dsgn4-000202	UGR	648	30264	34	2.15	5.24	0.33
dsgn4-000277	UGR	367	14404	10	1.5	2.72	0.41

*No juvenile Steelhead/Rainbow Trout less than 80 mm observed.

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Benthic macroinvertebrates

Background

Aquatic macroinvertebrate communities are important ecological components of river ecosystems; however, these communities have experienced significant declines in species richness, biomass, and distribution as the result of anthropogenic stressors including climate change (Domisch et al. 2011). Assessments of benthic macroinvertebrate diversity and density provide information about water quality and overall river health as well as the integrity of aquatic food webs, which may present a bottleneck for salmonid populations in the Columbia River basin (Independent Scientific Advisory Board 2011; Naiman et al. 2012). Both river basin management and restoration methods typically focus heavily on physical habitat conditions, which may neglect to address the influence of management and restoration actions on aquatic food webs and the corresponding impact to salmonid populations occupying habitat in which invertebrate communities are in flux as the result of restoration activities, climate change or land use practices (Bellmore et al. 2017; Sullivan and White 2017).

Since 2010, CRITFC has monitored fish habitat conditions in the Grande Ronde basin, including benthic macroinvertebrate sampling to assess salmonid prey resources as well as overall habitat condition to evaluate the effectiveness of aggregate restoration activities. Benthic macroinvertebrate (BMI) samples collected during the 2021 field season contribute to long-term BMI monitoring efforts in the Grande Ronde basin conducted by CRITFC as well as partnering tribal, state, and federal agencies. The widespread spatiotemporal scope of this dataset provides the opportunity to evaluate trends in BMI density and diversity in relation to restoration effectiveness and climate change-related variables like streamflow and stream temperature as well as the continued development of BMI metrics as indicators of food availability for salmonids (White et al. 2019).

Methods

Aquatic macroinvertebrate sampling was conducted in 2021 by crews from CRITFC and the Grande Ronde Model Watershed (GRMW) at nine sites in the upper Grande Ronde River and three sites in Sheep Creek using a spatially balanced random survey design (Stevens and Olsen 2004; Figure 18). Prior to sampling, each reach was divided into 11 equidistant transects across the river channel. Beginning at the farthest downstream transect, the first sample location was randomly selected as the right, center or left of the

transect (25%, 50% and 75% of the wetted width, respectively). Samples were collected using a D-framed kick net with 500 µm mesh, in which substrate within the quadrat (0.3 x 0.3 m) was thoroughly brushed prior to vigorously kicking the substrate to a depth of 10 cm for 30 seconds. In instances where transects fell within slow water habitats such as pools and glides, kick nets were dragged repeatedly through the kicked area to ensure that all organisms were collected. The process was completed at each transect, resulting in a composite sample which was then transferred into one or more sample jars and preserved in 95% ethanol.

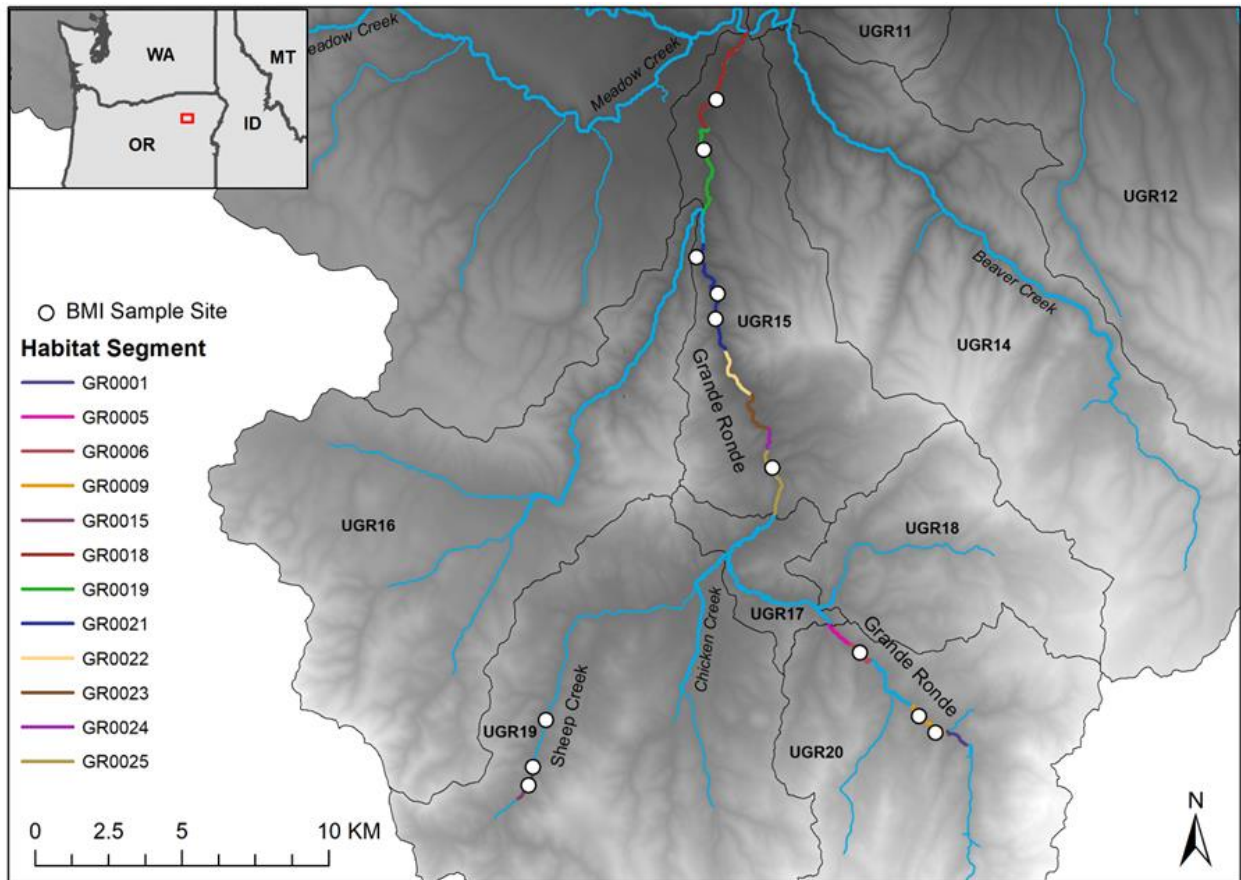


Figure 18. Map of biological sample reaches subsampled for benthic macroinvertebrates in the summer of 2021 in the upper Grande Ronde (9 sites) and Sheep Creek (3 sites).

Following the completion of benthic macroinvertebrate sampling, samples were delivered to a professional taxonomic lab (Rhithron Associates, Missoula, MT) for subsampling and taxonomic analysis. In addition to standard indices typically derived from BMI data such as indices of biotic integrity (IBIs), BMI samples were also used to calculate novel metrics including descriptive indices of ecological networks (Cohen et al. 2003) and food availability for salmonids based on life history characteristics, propensity to enter the water column, palatability to salmonids, and other characteristics (Rader 1997). Full details of metric development are found in Sullivan and White (2017).

Results

Benthic macroinvertebrate density (individuals/m²) varied considerably across reaches sampled during the 2021 field season, with the highest density observed in Sheep Creek; however, the other two reaches

sampled in Sheep Creek exhibited substantially lower BMI densities, indicating that the reach with an exceptionally high (relative) density is likely more reflective of localized conditions rather than an indicator of the overall BMI density in Sheep Creek (Table 11). Densities in reaches sampled in 2021 ranged from 1,149 to 15,764 (mean = 5,570 individuals/m²), with similar variability in biomass, which ranged from 0.79 to 12.87 g/m² (Table 11).

At the biologically significant reach (BSR) scale, UGR15, which is the furthest downstream reach of those sampled in 2021, had both the lowest mean density (3,649 individuals/m²) and the highest biomass (7.36 g/m²), suggesting a greater proportion of large individuals (Table 10). UGR20, located on the Grande Ronde River upstream of the confluence with Limber Jim Creek, had a mean density of 5,916 individuals/m² and the lowest biomass among the three BSRs (1.66 g/m²). UGR19, which encompasses Sheep Creek, had the highest density (10,475 individuals/m²); however, as mentioned above, one of the three reaches sampled within this BSR had an exceptionally high density relative to other reaches sampled during 2021 monitoring efforts.

Table 11. Benthic macroinvertebrate indices derived from summer 2021 sampling in the upper Grande Ronde River and Sheep Creek.

Stream	Reach	Density (ind. m ²)	Biomass (g m ²)	O/E (obs./exp.)	E. OR. IBI Score	Temp. Sens. Taxa (#)	Rader ASC _{Rel}
Sheep	CBW05583-335162	7721	1.51	0.35	30	1	3821.8
Sheep	CBW05583-453946	15764	9.32	0.50	30	2	3878.4
Sheep	CBW05583-228666	7941	1.47	0.50	40	2	3676.9
UGR	CBW05583-206314	3112	0.79	0.64	40	2	3949.6
UGR	dsgn4-000009	7721	2.59	0.78	44	5	4225.5
UGR	CBW05583-099818	6915	1.59	0.78	46	7	4272.6
UGR	dsgn4-000202	4022	1.81	0.35	40	0	3970.1
UGR	CBW05583-453434	3194	12.87	0.28	30	0	3753.5
UGR	CBW05583-355130	1149	0.91	0.43	44	1	4013.9
UGR	CBW05583-104250	2667	3.00	0.43	44	1	3860.0
UGR	CBW05583-486202	1409	7.67	0.35	36	0	3422.4
UGR	CBW05583-190266	5235	3.24	0.43	36	0	3967.5

Table 12. Benthic macroinvertebrate summary statistics averaged by biologically significant reach (BSR).

BSR	Stream	Reaches Sampled	Mean Density (ind. m ²)	SD	Mean Biomass (g m ²)	SD	Mean O/E	SD
UGR15	UGR	6	3649	2341.4	7.36	7.79	0.37	0.05
UGR19	Sheep	3	10,475	4581.4	4.10	4.52	0.49	0.04
UGR20	UGR	3	5916	2461.5	1.66	0.90	0.73	0.04

The macroinvertebrate Observed/Expected Taxa Loss model (O/E) quantifies the ratio of taxa loss or gain by comparing a list of “Expected” taxa based on reference locations with the BMI sample collected at a monitoring location (the “Observed”) (Hubler 2008). Sites with scores of less than one have fewer taxa at a site than were predicted by the model, while scores greater than one are either equivalent to reference locations or may even surpass reference locations as the result of management or restoration actions (i.e., carcass supplementation, etc.) All reaches sampled in 2021 had O/E scores lower than one, ranging from 0.28 to 0.78, indicating that fewer taxa were observed across all monitoring sites than would be expected based on reference locations in the same region (Table 11). At the BSR scale, UGR15 had a mean O/E score of 0.37, while UGR20 had a mean O/E score of 0.73, indicating a decrease in the integrity of the benthic macroinvertebrate assemblage with increasing distance downstream (Table 12). UGR19 (Sheep Creek) had a mean O/E score of 0.49, which also suggests degraded stream conditions relative to reference areas.

Aquatic species composition (ASC_{Rel}), a score which incorporates the expected or actual abundance of taxa for each site, is weighted to account for the potential contribution of each taxon to the salmonid prey base determined by life history and ecological traits related to drift propensity (Sullivan and White 2017; Rader 1997). As a BMI metric, ASC_{Rel} indicates the proportional abundance of taxa in a sample likely to be contributing to the concentration of macroinvertebrate drift, making it a pertinent metric for assessing habitat suitability for drift foraging fish species such as juvenile Chinook (Hayes et al. 2007). Overall, sample reaches located higher in the drainage had higher ASC_{Rel} scores, with higher scores observed in the farthest upstream reaches on the Grande Ronde River than those observed in Sheep Creek (Figure 19). These results suggest that although BMI biomass was relatively low in the upper reaches of the Grande Ronde River, the invertebrate community may be more susceptible as prey for juvenile salmonids.

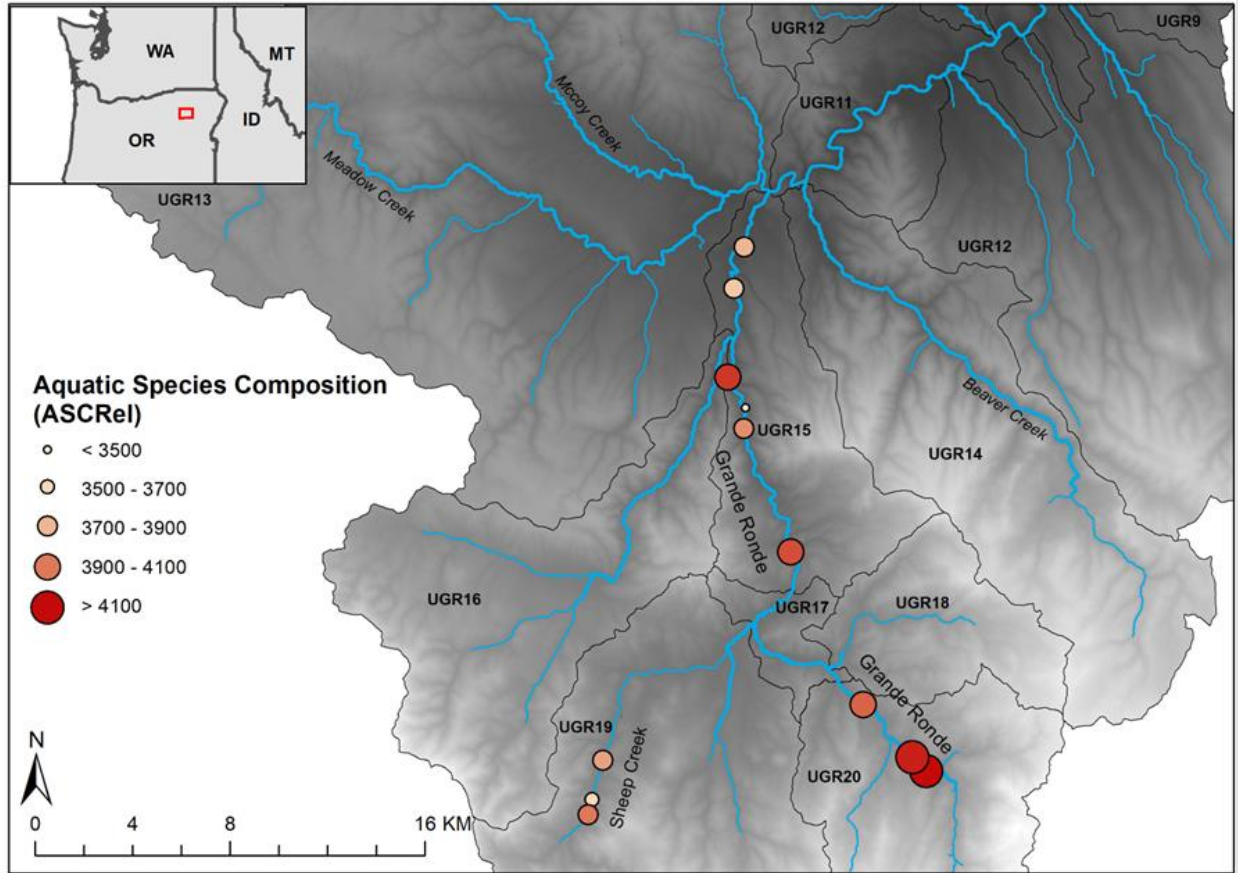


Figure 19. Distribution of drift propensity metric (ASC_{Rel}) values in the upper Grande Ronde River and Sheep Creek. Increasing symbol size and shade indicate increasing ASC_{Rel} scores.

The Eastern Oregon Benthic Index of Biological Integrity (IBI) score provides an index value which describes the condition of stream macroinvertebrate communities and is calculated based on a multi-metric approach utilizing a suite of metrics (taxa richness, percent pollution-tolerant taxa, percent sensitive taxa, etc.) to describe various aspects of assemblage composition (Hubler 2008). IBI scores range from 30-50, with 30 indicating poor biotic integrity and 50 indicating conditions equivalent to what would be found in a reference stream. IBI scores in reaches sampled during summer 2021 ranged from 30 to 46, with higher scores observed at reaches located higher in the basin than those observed farther downstream in the Grande Ronde River and in Sheep Creek (Table 11).

The number of temperature-sensitive indicator taxa observed in sample reaches ranged from 0 to 7, with all occurrences of zero temperature-sensitive in UGR15, the furthest downstream BSR, while the highest numbers of sensitive taxa were observed in UGR20 (Table 11; Brandt 2001). The mean number of sensitive taxa in UGR15 was 0.29 (N=6), while the mean in UGR20 was 4.67 (N=3). The three reaches located in Sheep Creek (BSR UGR19) had a mean of 1.67 temperature-sensitive taxa (Table 11). The distribution of temperature sensitive taxa corresponds with the other BMI metrics calculated, with more intact assemblages observed at reaches located higher in the drainage.

Next Steps

Next steps include further analysis relating food availability metrics and other BMI derived metrics to fish abundance, diet, growth, and productivity as well as an evaluation of spatiotemporal trends in the diversity and abundance of macroinvertebrates using spatial stream-network (SSN) models.

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Objective D-2: Index of Chinook Salmon rearing capacity

This work was completed and described in the 2021 annual report (White et al. 2022).

References

White, S., C. Justice, B. Staton, L. Burns, and M. Kaylor. 2022. Evaluating salmonid and stream ecosystem response to conservation measures and environmental stressors in the Columbia River basin. Page 74. Columbia River Inter-Tribal Fish Commission, BPA Project # 2009-004-00, Portland, OR.

Objective D-3: Finalize pre-spawn mortality analysis

This work was completed and described in the 2021 annual report.

See references from Objective D-2.

Objective D-4: Complete study of juvenile fish emergence timing and floodplain inundation

Introduction

Over the last three decades, hundreds of millions of dollars in mitigation funds have been allocated to habitat protection and restoration programs throughout the Columbia River basin to recover depressed salmon populations. Much of this effort has focused on enhancing the quantity, quality, and diversity of rearing habitats to increase juvenile salmon productivity. However, restoration efficacy depends in part on the degree to which fish can access and benefit from these enhanced habitats. The ability to move to and from habitats during early life stages is critically important to salmon population dynamics and productivity (Schlosser 1995) and may depend on the proximity of adjacent habitat types, the dispersal patterns juveniles exhibit, and the timing and duration that habitats are accessible.

There is increasing recognition that floodplain habitats can be uniquely important to juvenile salmon during early life stages by providing velocity refuge and enhanced growth opportunities (Sommer et al. 2001; Jeffres et al. 2008). Human development has fragmented and degraded floodplain habitat worldwide, making it a critical target for restoration efforts (Bond et al. 2019). However, floodplain inundation is dynamic, and the accessibility of floodplains may only occur within a specific range of flows or above a threshold (Baldock et al. 2015). These dynamics of inundation set the template for when floodplains are theoretically accessible, but realized use will depend on the ability of juveniles to disperse to these habitats and the phenology of juvenile emergence from redds relative to floodplain inundation. The use of floodplains may also influence dispersal patterns to summer rearing locations, as lateral connection to slower velocity habitats during high flows may reduce downstream dispersal. Thus, a key challenge is to understand where floodplains can provide the greatest potential use and benefit to salmon. Incorporating these dynamic habitats into effective conservation planning and implementation requires explicit consideration of how they align in space and time with the habitats they complement (e.g., spawning locations, summer rearing habitats) and with the biology, phenology, and life stage requirements of target species.

In collaboration with the NOAA's Northwest Science Center, the Confederated Tribes of the Warm Springs Reservation of Oregon, Oregon State University, and the Oregon Department of Fish and Wildlife, we are developing a framework for predicting juvenile Chinook Salmon floodplain habitat use across watersheds that links dynamics of floodplain accessibility with aspects of juvenile salmon ecology. More specifically, we are 1) modeling floodplain inundation across space and time (when and where floodplains are theoretically accessible), 2) modeling emergence phenology across watersheds (when juvenile salmon can access floodplains), and 3) evaluating the distribution of spawning locations relative to accessible floodplain habitats. We are applying this approach to three NE Oregon basins: the upper Grande Ronde River (UGR), Catherine Creek (CC), and the Middle Fork John Day River (MFJD). Lastly, we are quantifying juvenile salmon dispersal patterns from spawning locations to floodplain habitats in MFJD using genetics-based parentage assignments to assess how the proximity of floodplains to spawning locations is likely to influence habitat utilization. Quantifying dispersal was not logistically feasible for all three subbasins and we focused on MFJD to leverage other genetics sampling efforts (see Appendix B: Juvenile Salmon Dispersal using Parentage-Based Tagging).

This is a large project with multiple sub-components, each involving extensive data collection, modeling, and data analysis, and we have made substantial progress on each sub-component. We completed preliminary predictions of floodplain inundation dynamics across all three basins. However, preliminary estimates for CC and UGR were completed in spring 2023, and we are still in the process of refining predictions and performing data QAQC. We have completed emergence timing estimates for water years 2012-2019 across all three basins (see Kaylor et al. 2022), and we are in the process of acquiring water temperature and spawning survey data for water years 2020-2022 to predict emergence phenology in additional years. Lastly, we collected adult (2020 and 2021) and fry (2021 and 2022) tissue samples from the MFJD to evaluate dispersal patterns from spawning locations to floodplains using parentage-based tagging. We received parentage assignments for 2021 fry and have explored preliminary results of fry dispersal patterns to floodplains. We anticipate receiving parentage assignments for 2022 fry in summer 2023.

Methods

The three sub-basins – CC (1051 km²), UGR (1896 km²), and MFJD (2051 km²) – exhibit similar geomorphology, habitat conditions, and biota. The subbasins are close in proximity relative to the scale of the Columbia River basin, but CC and UGR are tributaries of the Grande Ronde River, which flows northward to its confluence with the Snake River, whereas the MFJD flows northwest into the John Day River, meeting the Columbia River farther downstream. The drainages of MFJD (maximum elevation: 2478 m) and UGR (maximum elevation: 2414 m) originate from the lower elevation Blue Mountains, whereas CC (maximum elevation: 2640 m) drains slightly higher elevations of the Wallowa Mountains. The timing of snowmelt and associated peak flows is earlier for MFJD compared to UGR and CC, which exhibit similar timing of peak flows. In contrast with UGR and CC, spring Chinook Salmon are not listed within the MFJD, and no hatchery supplementation has occurred.

Floodplain inundation is highly dynamic across watersheds and through time, largely regulated by the flow regime and valley geomorphology. We developed a process to capture this complexity (Figure 20) by utilizing high-resolution, multi-spectral, satellite imagery and image classification to identify wetted surfaces across images spanning a range of flows and dates. A deep-learning model was trained to identify wetted surfaces through an iterative process of image classification, manual correction and retraining, and new image classification resulting in rapid and efficient processing of large spatial extents. Stream networks

in each basin were split into 200 m segments and floodplain polygons were generated for each 200 m segment. For each satellite image, wetted surfaces were classified, total wetted area was calculated within each polygon, and floodplain wetted area was calculated by subtracting bankfull stream surface area within each polygon. We used flow measurements from in-basin gauging stations (CC - USGS 13320000; UGR - OWRD 13317850; MFJD - USGS 14044000) to develop relationships between flow and floodplain wetted area across all image dates. These relationships were then applied to continuous gauging station flow data to produce daily estimates of floodplain area for each 200 m segment.

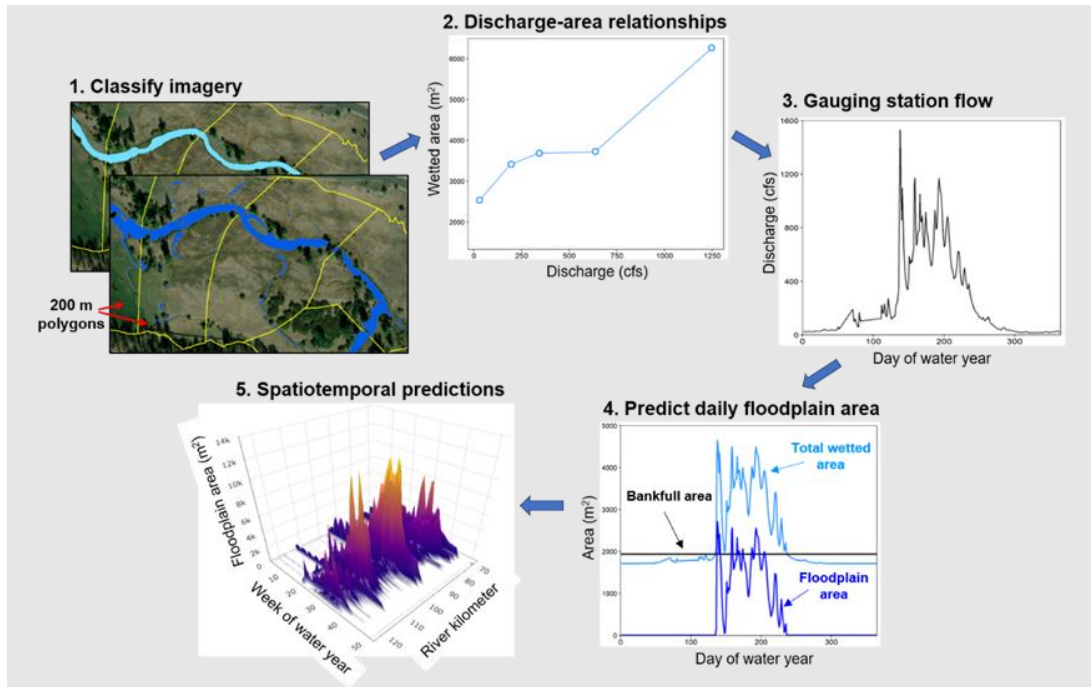


Figure 20: Simplified diagram of our floodplain area modelling process: 1) classify wetted area within each 200 m polygon for each image; 2) develop relationships between discharge and wetted area for each image date and polygon; 3-4) use flow-area relationships

To develop robust relationships between flow and floodplain wetted area, a sufficient number of satellite images encompassing a range of flows is needed. Satellite imagery was generally sufficient across the spatial extent of the MFJD, but images in CC and UGR often had incomplete watershed coverage. As a result, MFJD estimates were produced first and CC and UGR modeling required additional effort and approaches to supplement imagery. In CC and UGR, bathymetric LiDAR was used to create a relative elevation model (REM), which can be used to estimate flooded surfaces at different water levels. We used the REM to extend wetted area estimates from imagery into floodplain areas not covered by imagery by choosing wetted elevation values that matched the wetted area in each image. Estimates for CC and UGR were completed recently, and we are currently performing QA/QC on the data and refining our modeling approach.

We predicted emergence timing across each basin using spawn timing and annual water temperature data. A complete description of our methodological approach is described in Kaylor et al. (2022), but we provide a brief overview below. We modeled spawn timing using spawning survey data collected by partners in each basin in which 2-5 surveys were repeated every 7-10 days in each river and each year. We fit linear

models to redd observation dates with spawn year and river kilometer as covariates. We then predicted emergence timing from spawn dates using the relationship between water temperature and incubation development rates developed by Beacham and Murray (1990), but with modifications suggested by Sparks et al. (2019) to better account for variable temperature exposure. We simulated 1000 emergence estimates for each year and each temperature sensor site by drawing random spawn dates from the predicted mean spawn date and model residual. Lastly, to evaluate longitudinal patterns of emergence among basins and years, we fit linear models with river kilometer and spawn year as covariates to median emergence timing estimates.

We are quantifying juvenile Chinook Salmon dispersal using genetic parentage-based tagging (Bravington et al. 2016) to assign fry sampled from floodplains to adults sampled from spawning locations, and thus an approximate location of origin. We sampled adults in 2020 and 2021 and fry the following years in 2021 and 2022 to evaluate dispersal in two years. We sent all samples to the Columbia River Inter-Tribal Fish Commission's Hagerman Genetics laboratory for genotyping and parentage-based tagging analysis. We sampled 141 post-spawn adults in September 2020 (162 redds total) and 1110 fry from floodplain and off-channel habitats in spring 2021 (March-May). Of the 141 adult samples, 113 met the genotyping standards of the lab and were retained for parentage assignments, and 397 of the fry sampled in 2021 were paired to a female adult (fry-male pairings not included; see Appendix B). In contrast to 2020 adult sampling, poor adult returns, unusually warm temperatures, and intensive otter predation/scavenging resulted in far fewer redds and spawning adults in September 2021. We were able to sample only 24 carcasses of which only half were retained by the lab for parentage assignments. With fewer redds in 2021 (and thus presumably fry offspring), fry were more challenging to capture in 2022, but we obtained fin clips from 386 fry. Most 2022 fry were sampled from mainstem channel margins and off-channel habitats (e.g., alcoves) as floodplains were rarely inundated after emergence due to relatively early and low spring flows. We have conducted preliminary analyses exploring dispersal patterns of 2021 fry and we anticipate receiving parentage assignments for 2022 fry in summer 2023.

Preliminary results and discussion

We are in the process of performing QAQC on CC and UGR floodplain area predictions. Therefore, the preliminary results presented in this section are specific to the MFJD where we have made the most progress relating floodplain inundation predictions to juvenile salmon ecology. These findings are an example of how we may link these complex spatiotemporal data to inform floodplain habitat utilization, but our approach, findings, and associated conclusions are likely to change as we continue analysis, make comparisons among basins, and further discuss the project with basin partners.

The timing, duration, and extent of MFJD floodplain habitat accessibility varied considerably across space and among years. As an example, Figure 21 shows the extent of floodplain area over time (week of water year) and across space (river kilometer) for 2018 and 2019 – the extent of wetted floodplain habitat was greater in 2019 and floodplain habitats remained wetted later into the spring relative to 2018. Across all years, the magnitude of floodplain area – reflecting the extent of floodplain connectivity – was most pronounced between rkms 92-97 (i.e., the Oxbow Conservation Area) and 105-111 (the Forrest Conservation Area) – two low-gradient, unconfined valleys where extensive restoration has occurred in the last decade – as well as between rkms 70-85 and 119-122.

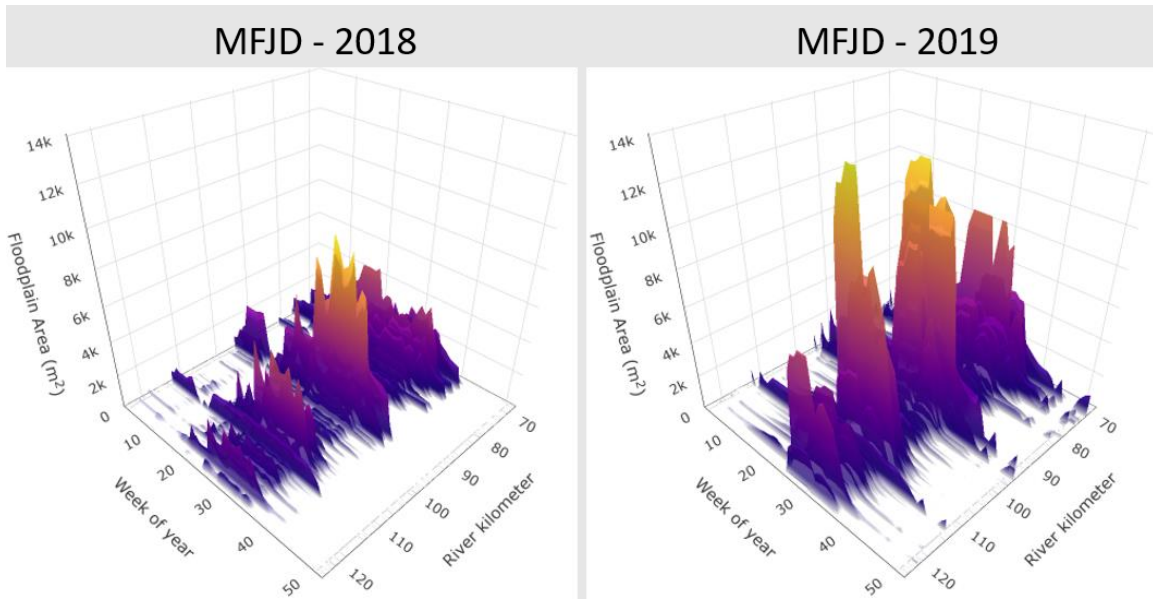


Figure 21: Spatiotemporal patterns of floodplain wetted area throughout the year (week of water year) across the Middle Fork John Day River basin (river kilometer). Warmer colors indicate greater wetted floodplain area and white indicates wetted surfaces were within the main channel.

Floodplain wetted area predictions also reveal spatiotemporal variation in the duration of theoretical floodplain accessibility (blue bars and lines; Figure 22 & Figure 23) – defined here as the number of days in which total wetted area exceeded 1.25 times the area of the bankfull channel. The duration of floodplain accessibility was generally greatest between rkms 92-97 and 105-112. In contrast, floodplain habitats were accessible for shorter durations or not at all in some locations (rkms 85-92, 97-102) and years (e.g., 2013, 2015, 2018). Some areas with limited durations of floodplain accessibility are in confined valleys with small historic floodplains (e.g., rkms 97-102), whereas other sections are characterized by low-gradient, unconfined valleys (e.g., rkms 85-92) where restoration may be more effective in reconnecting floodplains.

In the MFJD, estimated emergence timing was latest in middle sections (~ rkm 90) and progressively earlier moving upstream, with approximately six weeks difference in median emergence dates at the farthest upstream section relative to the middle section (Figure 22). Consequently, the proportion of days that floodplains were accessible after emergence relative to the total number of days floodplains were theoretically accessible was lowest in middle sections (e.g., rkms 92-97) and greatest in the farthest upstream sections (rkms 105-112). Yet because the middle section between rkms 92-97 exhibited the longest durations of floodplain accessibility, the total number of days of accessibility after emergence were comparable to well-connected upstream floodplains (e.g., rkms 105-112).

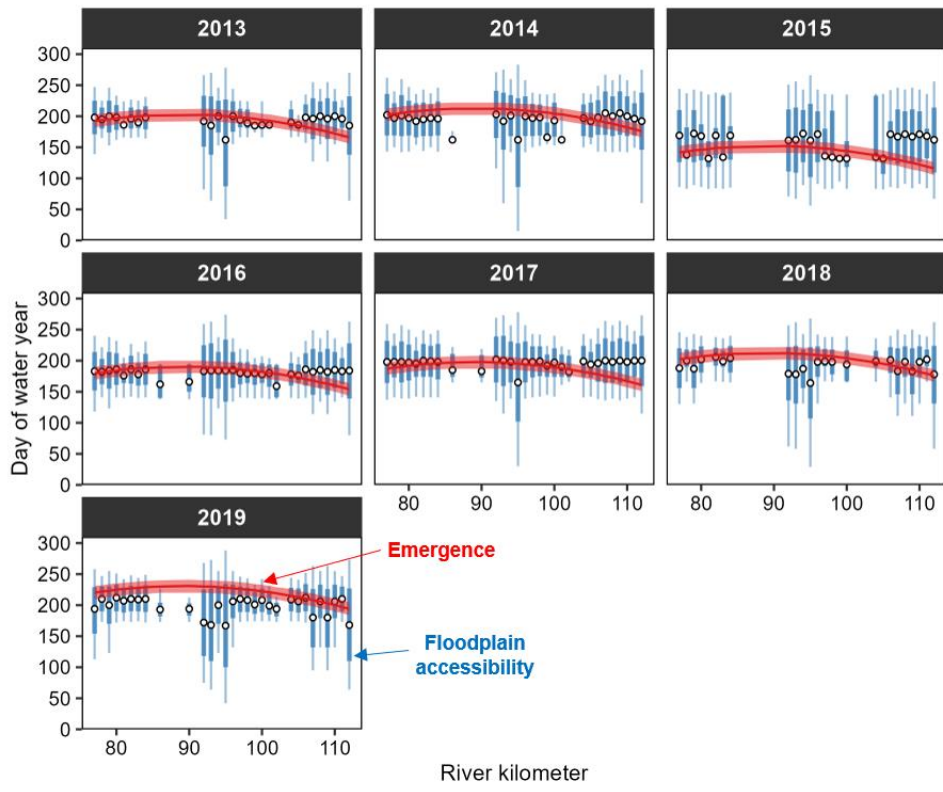


Figure 22: Predicted emergence timing across the MFJD in each year (red; ribbon represents 95% prediction interval) relative to the timing and duration of floodplain accessibility (blue lines and white points). 200 m floodplain polygons were grouped into 1 km bins for presentation. Floodplain accessibility is quantified as days with total floodplain area within each bin exceeding 1.25 times total bankfull area of that 1 km section. Rkms without points or bars indicate that floodplain area did not exceed 1.25x bankfull width (i.e., confined channels).

There was also considerable variation in the overlap between theoretically accessible floodplain habitat and emergence timing among years. For example, predicted fry emergence was earliest in 2015 and a greater proportion of the days in which floodplains were accessible occurred after emergence compared to years such as 2019, in which emergence was latest and floodplains were only accessible for a short duration after emergence.

These results suggest that the duration floodplains are accessible to fry depends both on the timing of inundation and emergence. If the timing of floodplain inundation was consistent across the watershed, fry upstream would have up to six weeks longer to access floodplains due to earlier emergence. But given patterns of emergence timing, floodplain habitats downstream, where emergence is later, can provide similar durations of accessibility if they remain inundated later in the spring. Collectively, these results suggest that floodplain restoration in upstream sections may provide greater benefits to Chinook Salmon fry (e.g., flow refuge, prey abundance, favorable temperatures) relative to downstream sections, assuming equal floodplain characteristics, restoration effort, and restoration approach.

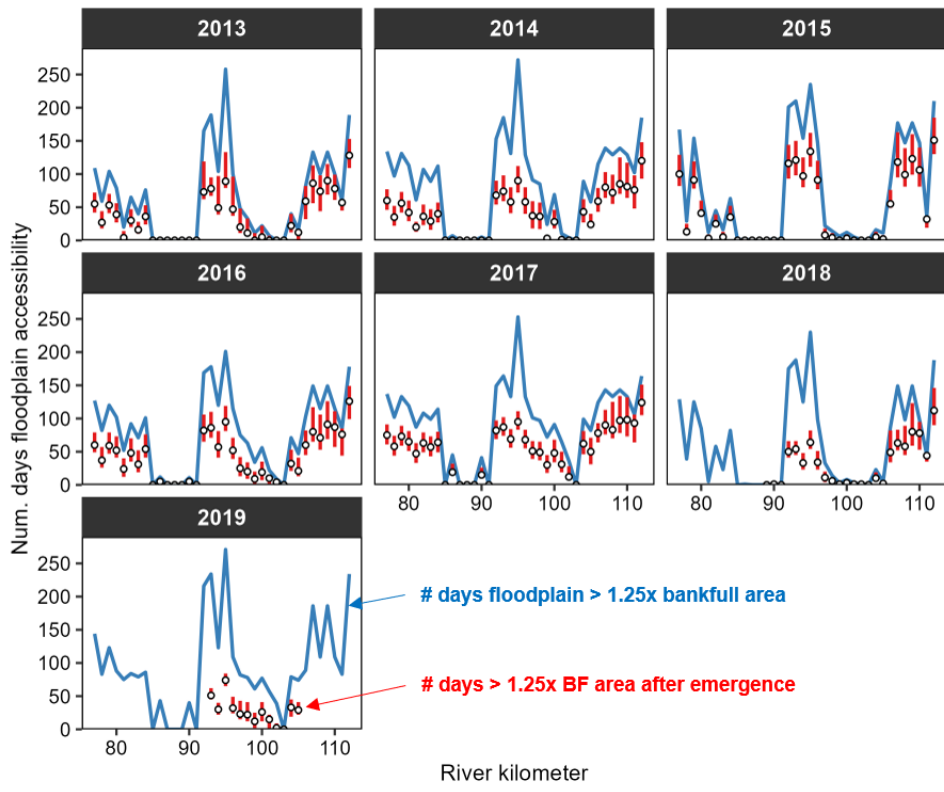


Figure 23: The estimated number of days floodplains exceeded 1.25x bankfull within 1 km bins in each water year (blue line) vs the number of days floodplain area exceeded 1.25x bankfull width after the median emergence date (points and red lines). Data shown for days after emergence only includes estimates between the farthest downstream and upstream temperature loggers for each year. Note that differences are smaller during some years such as 2015 and larger during others such as 2018-2019.

All 397 fry captured in spring 2021 and paired to a female dispersed downstream (Figure 24). The median dispersal distance was approximately 0.8 km downstream, but ~25% dispersed more than 5.0 km downstream and ~5% dispersed more than 12.6 km downstream. These results suggest that the relationship of floodplain habitats relative to spawning locations depends not just on proximity, but whether floodplains are located upstream or downstream of spawning locations. Recognizing that there are many benefits of restoring floodplain connectivity, these results suggest that restoring floodplain habitats upstream of spawning locations is likely to result in low habitat utilization by Chinook Salmon fry originating downstream. In contrast, results suggest that floodplain habitats located several or more kilometers downstream of spawning locations may be utilized by fry, although to a lesser extent than habitats located closer to spawning locations.

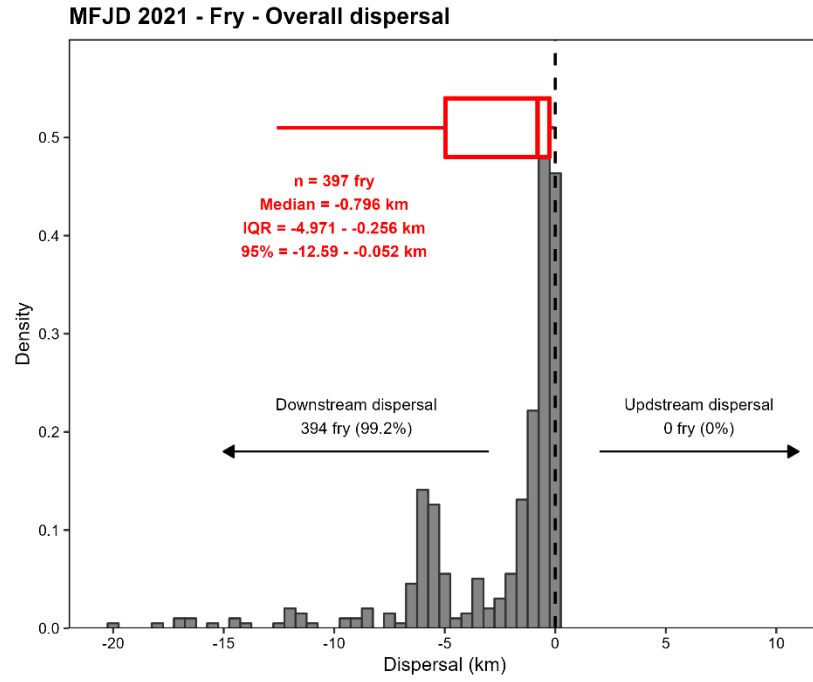


Figure 24: The distribution of fry dispersal estimates from spawning locations to floodplain and off-channel habitats in spring 2021.

Compared to the MFJD, spawning distributions are more clustered within relatively short core spawning areas within CC and UGR. This has the potential to result in greater discrepancies between spawning locations and well-connected floodplain habitats. Consequently, understanding dispersal patterns provides an important link between habitat proximity and potential floodplain habitat use.

Next steps

Once we have performed QA/QC on floodplain area predictions for CC and UGR, we will integrate emergence timing estimates and spawning distributions in each basin to assess the spatial overlap of accessible floodplain habitat and spawning locations and the temporal overlap of floodplain accessibility and emergence timing. We expect substantially different results in CC and UGR compared to MFJD: the timing of flows peaks earlier in MFJD relative to CC and UGR (Figure 25, upper-left panel), emergence patterns exhibit opposite longitudinal patterns (Figure 25, upper-right and bottom panels), and spawning distributions are more clustered in CC and UGR. Comparisons among these three basins will provide important insight into how emergence timing, including longitudinal patterns within basins, relate to the duration and ability of Chinook Salmon fry to utilize floodplain habitats.

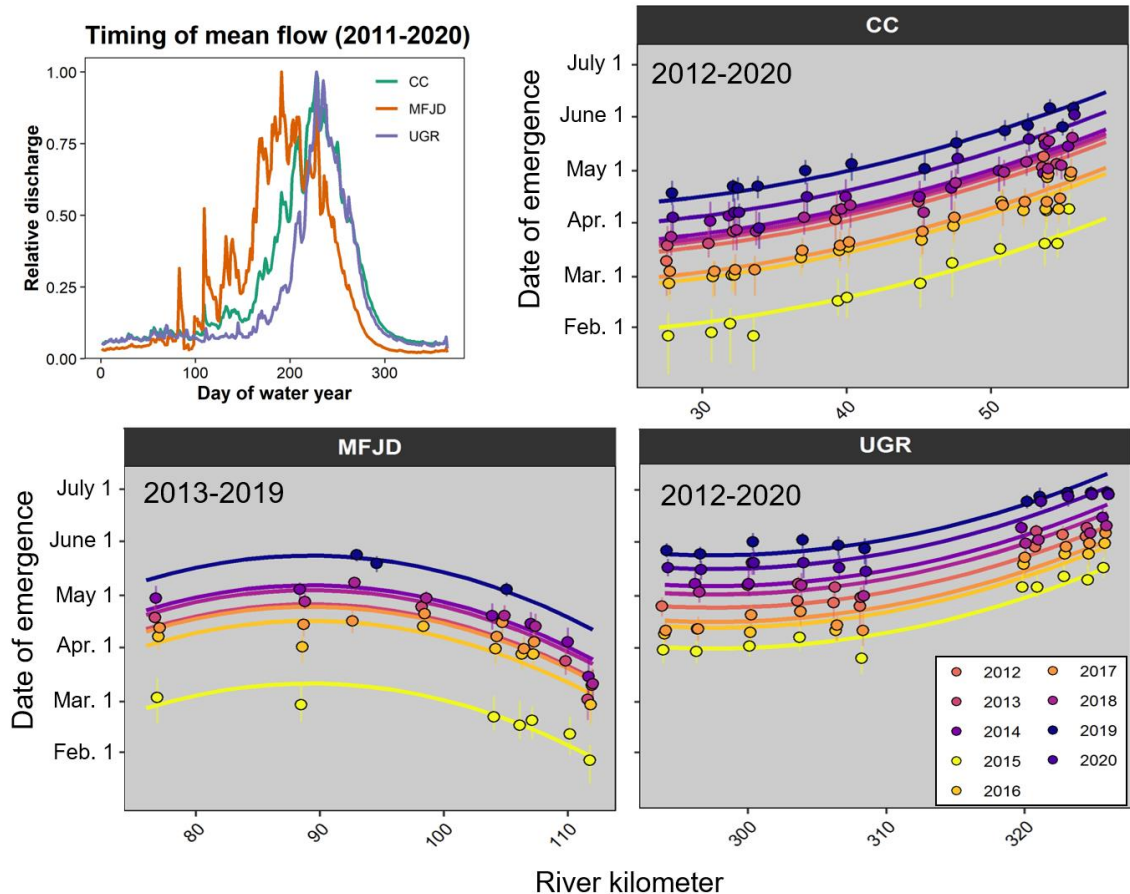


Figure 25: Mean annual relative discharge over the course of a water year for each basin (upper left panel), demonstrating a pattern of generally earlier timing of peak flows and earlier onset of summer baseflows in MFJD compared to CC and UGR. The upper right and bottom panels show longitudinal patterns of emergence timing estimates in each basin and over multiple years. Emergence timing was predicted to occur progressively later moving upstream in CC and UGR, but not in MFJD where emergence was progressively earlier with distance upstream (upstream of rkm 90).

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Objective D-5: Complete study of juvenile Chinook dispersal and floodplain use

A draft manuscript associated with this study is attached to this report (Appendix B: Juvenile Salmon Dispersal using Parentage-Based Tagging). We included an abstract overviewing the study and findings below. Please note that the results and analyses presented in this draft are preliminary and subject to change.

Abstract

For Pacific Salmon species, the dispersal of juveniles from spawning locations to rearing habitats is a critical process influencing individual fitness and population productivity. Studies evaluating juvenile salmonid dispersal have found that dispersal is generally limited (within 0.5 km of origin) and downstream biased, but that numerous biophysical factors can influence dispersal patterns. The expression of factors that restrict or promote dispersal likely varies across river networks, potentially resulting in differences in dispersal patterns reflective of local conditions. We utilized a riverscape sampling approach and genetics-based parentage assignments to evaluate dispersal patterns of a wild population of spring-run Chinook Salmon in NE Oregon. We sampled post-spawn female adults from spawning locations in 2020 and their offspring (parr) from summer rearing habitats across nearly 40 km of the mainstem and nine tributaries in 2021. Female adults ($n = 67$) and parr ($n = 3,388$) were genotyped, parr were assigned to female parents ($n = 1,326$), and dispersal was calculated using the location of females and paired parr. We also estimated parr abundance across the river network to correct for sampling bias effects, and to relate the spawning distribution and dispersal patterns to subsequent parr distribution. Overall dispersal estimates were downstream biased (median = -0.77 km) with 68% of parr dispersing downstream. Dispersal distances were high relative to those found in other studies, with 25% of parr dispersing more than 0.9 km upstream and 25% dispersing more than 3.7 km downstream (range = 28.6 km downstream - 10.6 km upstream). At finer spatial scales, dispersal patterns varied throughout the river network. Parr originating lower in the watershed exhibited upstream dispersal bias and less variability in dispersal patterns, whereas parr originating higher in the network had downstream dispersal bias, greater variability in dispersal, and generally dispersed

farther. Combining dispersal patterns with estimates of parr distribution revealed that parr originating in areas with higher maximum July temperatures generally dispersed to cooler sections of the mainstem or tributaries, whereas parr originating in cooler sections of the mainstem were more likely to remain near their origin and within the mainstem. Our results suggest greater dispersal at early life-stages than prior published estimates and that heterogeneity in biophysical conditions across watersheds may translate to variable patterns of dispersal. These findings provide guidance for prioritizing the types and locations of restoration efforts to increase habitat utilization through better alignment with juvenile salmon ecology.

Objective E: Relate population-level fish response to habitat change in the Grande Ronde basin and potentially other basins

Objective E-1: Grande Ronde Phase 1 – Development of spring Chinook statistical estimation Life Cycle Model

Background

We are developing a state-space life cycle model (SSLCM) for Grande Ronde spring Chinook salmon that estimates population dynamics parameters (e.g., average survival, rearing capacity, inter-annual variability) from historical monitoring data. The purpose of fitting the SSLCM is to estimate posterior distributions of population dynamics parameters that can be sampled from to populate forward simulations of the populations under various habitat restoration and climate change scenarios. The SSLCM tracks cohorts of fish as they transition through various life stages, links these “state variables” over time (e.g., parr recruit abundance is a function of spawner abundance the previous year), and separates the variability in data sets into process (i.e., biological) and observation noise. The model integrates many different data sources and fits to them simultaneously in a single joint likelihood; these data sources include rotary screw trap passage estimates, a variety of PIT tag-derived survival estimates, mean length data, hatchery smolt releases, adult return abundance, and the composition of adult returns by age and origin. Four spawning populations within the Grande Ronde basin are simultaneously analyzed: Catherine Creek (CAT), Lostine River (LOS), Minam River (MIN) and the upper Grande Ronde River (UGR), and where relevant, the model stratifies life stages by juvenile migratory strategy (fall vs. spring migrants), rearing origin type (spawned in hatchery vs. natural setting), and adult return age (total age of 3, 4, or 5 years).

For the SSLCM posterior distributions to be reliable in a simulation framework, the model should capture at least some critical drivers of population dynamics, i.e., the model should quantify effects of either intrinsic or extrinsic factors on population responses. For example, density-dependence (DD) is a known intrinsic feature of salmonid population dynamics (Grossman and Simon 2020), including Snake River spring Chinook salmon populations (Walters et al. 2013), and failing to appropriately account for density-dependent juvenile survival will produce unrealistic expectations of future population status (Rose et al. 2001). Our approach to building the SSLCM was to focus on capturing the intrinsic factors through process model equations while treating extrinsic factors as a combination of (a) fixed habitat capacity effects on population parameters and (b) “process noise” attributable to environmental stochasticity – while being grounded to the extent possible in empirical biological monitoring data.

This chapter summarizes the questions and analyses that motivated our approach in structuring intrinsic factors occurring between egg deposition and arrival at Lower Granite Dam (LGR) as smolt. In these

analyses, we use only information generated/derived from monitoring programs – none of the variables we use here are output from the SSLCM.

Research Questions

The research questions were motivated by our desire to capture DD population dynamics via process model equations to the extent justified by the data. With respect to “population dynamics”, we refer specifically to two types of rates: survival and growth. Survival is the fraction of fish alive at one stage also alive at the next stage, whereas growth is the ratio in body size (expressed here as population mean length, mm fork length [FL]) between two life stages. Our research questions (summarized in Table 13) were informed by relationships between density, growth (size), and survival described in the literature (see references in specific questions below); if we uncover similar relationships for Grande Ronde spring Chinook salmon populations, this suggests that linking these quantities using process model equations in the SSLCM may have utility for capturing intrinsic population variability.

Question #1: Does Parr Recruitment Appear DD?

Recruitment is frequently viewed as a DD process, being a default assumption of spawner-recruit equations (Adkison 2022). That is, not only is the number of recruits a function of egg production (more eggs, more potential surviving offspring), but so too is the survival from egg to recruit (more eggs, reduced fitness). This is commonly expressed using a Beverton-Holt recruitment function between the number of individuals at one stage and the number of individuals that recruit to another (e.g., Moussalli and Hilborn 1986). Here we have defined “recruits” as parr that reach the end of the summer when they are first sampled for active-capture PIT tagging. Detecting a negative relationship between egg-to-parr survival (denoted $\phi_{E \rightarrow P_{\text{summer}}}$, derivation below) and total egg production would provide evidence of pre-recruitment DD survival that should be accounted for when linking generations in the SSLCM (i.e., spawners produce eggs which become parr).

Question #2: Does Pre-Recruitment Growth Appear DD?

Juvenile salmonid growth rates are widely understood to be affected by the density of conspecifics (Grant and Imre 2005; Copeland and Venditti 2009; Walters et al. 2013; Myrvold and Kennedy 2015; Grossman and Simon 2020). Given its frequency in the literature and the documented linkage between both density/size and survival (e.g., Zabel and Achord 2004), we thought it worthwhile to explore pre-recruitment growth as being an early-life DD process that may affect survival outcomes later in life. Detecting a negative relationship between the mean size of parr at the end of the summer ($L_{P_{\text{summer}}}$, standardized for sample date) and total egg production (E) would provide evidence of pre-recruitment DD growth that may be useful in explaining inter-annual variability in survival later.

Question #3: Does Parr Survival to LGR Appear DD?

Several authors have found DD survival relationships for Snake River juvenile salmonids (Achord et al. 2003; Walters et al. 2013), including past analyses of Grande Ronde spring Chinook salmon (Cooney et al. 2017). The findings of Question #1 will establish how critical it is to account for pre-recruitment DD survival, but we should also investigate the extent to which it occurs following parr recruitment and prior to arrival at LGR. Detecting a negative relationship between the survival rate from recruitment as summer parr to arrival at LGR as smolt ($\phi_{P_{\text{summer}} \rightarrow S_{\text{LGR}}}$) and the number of parr recruits (P_{summer}) would provide evidence of post-recruitment DD survival that may need to be accounted for in the SSLCM.

Question #4: Do Survival Outcomes Correlate with Individual Size?

Fish size has been found to be related to survival in salmonids in multiple life stages (Ebersole et al. 2006; Connor and Tiffan 2012) and for Snake River Chinook salmon populations in particular (Zabel and Achord 2004; Hostetter et al. 2015; Feldhaus et al. 2016; Cooney et al. 2017), motivating us to investigate size-dependent survival as a key driver of Grande Ronde juvenile population dynamics. Questions #5 and #7 focus on relating mean length of the population to population survival rates; in contrast, this question focuses on investigating whether size is related to survival at the individual level, and whether it is consistent across years and among the four focal Grande Ronde spring Chinook populations. If we find that individual parr size is positively related to their (apparent) survival probability, then we may conclude that size-based survival is a feature of within-year, among-individual survival dynamics.

Question #5: Do Years of Faster Parr Growth Coincide with Higher Survival?

Question #4 investigates individual-level size effects on survival, however the SSLCM is not an individual based model – it models processes on the population scale and models variability that occurs on the inter-annual time scale within a life-stage. Thus, knowledge that size is related to survival within individual years is not terribly useful to the SSLCM on its own. To model growth dynamics within the SSLCM, simpler population aggregate values that represent population-level growth rates are needed to investigate whether size-based survival is a useful mechanism to explain inter-annual variability in survival. As in Question #2, we chose to use the mean parr size (FL, mm; standardized for capture timing) as the indicator of population aggregate summer growth rates, although in this case as the predictor variable in relationships with parr to LGR smolt survival ($\phi_{P_{\text{summer}} \rightarrow S_{\text{LGR}}}$) as the response variable. If we find consistently positive associations between population mean length and population survival rates, then we may conclude that growth occurring over the summer may act as a delayed effect (Beckerman et al. 2002; Mebane and Arthaud 2010; Walters et al. 2013) on post-recruitment survival rates, either during overwinter rearing, the migration to LGR, or both.

Question #6: Does There Appear to be Additional DD Survival After Controlling for Growth?

If we find that the answer to Question #2 (‘Does pre-recruitment growth appear DD?’) is “yes”, and we further find in Questions #4 and #5 that growth/size is related to survival, then it follows that linking density to growth and growth to survival could provide a mechanism to capture post-recruitment DD survival dynamics in the SSLCM. However, perhaps accounting for only DD growth affecting survival does not fully capture the effect of density on survival. In that case, we would be under-accounting DD and we would thus expect the model to give overly optimistic predictions of future population status. We approached this question by examining the relationship between the expected (apparent) survival to LGR of parr of a fixed size and total parr recruits across years. If we find weak or no relationships between these variables, that can provide confidence that we have sufficiently captured post-recruitment DD survival dynamics through growth and its effect on survival.

Question #7: Does Population Smolt Length Correlate with Survival?

Recall that if the answer to Question #5 (‘Do years of faster parr growth coincide with higher survival?’) is “yes”, then we need to know which life stage(s) we should use to account for size-based survival: in the overwinter rearing stage or in the stage involving migration to LGR. The latter of these is monitored directly via screw trap in the spring when smolt are tagged and measured upon out-migration. We thus chose to focus on investigating the relationship between survival of smolt migrating from the screw trap in the spring

to LGR ($\phi_{S_{\text{spring}} \rightarrow S_{\text{LGR}}}$) and the mean smolt size (FL, mm). Finding a positive association here suggests that we should at least model a size effect on natural origin fish making this migration. It would not, however, indicate whether we should also account for size-based survival in the overwinter rearing stage.

Question #8: Is Parr to Smolt Growth Related to Parr Size?

If we find in Question #7 that modeling size-based survival is important for smolt out-migration, then we will need a process model component to generate smolt size to serve as a predictor variable. It is reasonable to think of modeling smolt size based on parr size and a growth factor, however, how the growth factor should be modeled is uncertain. Investigating the relationship between the ratio in mean size for smolt:parr ($\Delta_{P_{\text{summer}} \rightarrow S_{\text{spring}}}$) and parr mean size could be useful for informing whether the growth factor should be treated as independent or as a function of initial parr size (e.g., Ebersole et al. 2006).

Data Sources

Population-Level Survival

Survival data sources for most analyses were from annual PIT tagging efforts conducted by Oregon Department of Fish and Wildlife (ODFW). Fish from all four populations are tagged in four main pulses: as parr in summer, as parr upon fall migration past the screw trap, as parr in winter in the headwaters rearing area, and as smolt upon spring migration past the screw trap. The annual survival rates of fish tagged in each of these events to their arrival at LGR as smolt is estimated independently via Cormack-Jolly-Seber methods; similar methods are used for each of the four populations. None of our research questions relied on winter survival estimates and we denote the other three survival terms by: parr tagged in the summer ($\phi_{P_{\text{summer}} \rightarrow S_{\text{LGR}}}$), parr tagged in the fall ($\phi_{P_{\text{fall}} \rightarrow S_{\text{LGR}}}$), and smolt tagged in the spring ($\phi_{S_{\text{spring}} \rightarrow S_{\text{LGR}}}$).

In addition to these directly monitored survival terms, we also calculated egg-to-parr survival as:

$$\phi_{E \rightarrow P_{\text{summer}}} = \frac{P_{\text{summer}}}{E}$$

Where E represents total egg production and P_{summer} represents total parr recruitment – both quantities are derived below.

Abundance

We used two abundance indices to represent population density in these analyses: total egg production and total parr recruitment. Neither of these quantities are directly observed or estimated by monitoring programs, but we have reconstructed their values based on quantities that are monitored and with several assumptions.

To reconstruct total egg production (E), we obtained the total adult return-to-river estimates produced by ODFW and apportioned them to age based on age composition sampling conducted at weirs and carcass surveys. We then subtracted the number of fish known to have been removed at the weirs (e.g., for broodstock) and due to tributary harvest, assumed 50% of age-4 and age-5 fish were female (0% of age-3), multiplied these resulting females-at-age-and-year by fecundity-at-age-and-year (estimated from females spawned in hatcheries), and summed across ages.

To reconstruct total parr recruiting to the end of summer (P_{summer}), we expanded the number of smolt reaching LGR (S_{LGR}) by the estimated survival of summer parr to LGR as smolt ($\phi_{P_{\text{summer}} \rightarrow S_{\text{LGR}}}$):

$$P_{\text{summer}} = \frac{S_{\text{LGR}}}{\phi_{P_{\text{summer}} \rightarrow S_{\text{LGR}}}}$$

To obtain S_{LGR} , we multiplied the estimated passage by the screw trap of parr in the fall (P_{fall}) and of smolt in the spring (S_{spring}) by their respective estimated survivals to LGR and summed the result:

$$S_{\text{LGR}} = P_{\text{fall}} \times \phi_{P_{\text{fall}} \rightarrow S_{\text{LGR}}} + S_{\text{spring}} \times \phi_{S_{\text{spring}} \rightarrow S_{\text{LGR}}}$$

Note that these calculations were performed for each year and population separately – the indices are dropped here for simplicity.

Mean Length & Growth Index

Upon capture for PIT tagging, fish are measured for body size (FL, mm) – we calculated mean length for all measured (tagged and untagged) parr in the summer tagging event ($L_{P_{\text{summer}}}$) and smolt in the spring tagging event ($L_{S_{\text{spring}}}$). As an index of population growth (in body size) between the summer parr stage and spring smolt stage, we also calculated the ratio of smolt:parr mean length ($\Delta_{P_{\text{summer}} \rightarrow S_{\text{spring}}}$).

Capture in the summer tagging event is conducted using active methods (snorkel-herding, electrofishing), and therefore the measured population mean length varies inter-annually due to sample timing. We discovered that sampling has occurred earlier in the summer (indexed by median day of year [DOY] of capture) over the years, which resulted in spuriously strong negative time trends in parr mean length for some populations. We thus devised a standardization method to correct for inter-annual variability in sample timing for parr mean length data – all analyses presented in this chapter use the standardized version.

The standardization method used these four steps for each population independently:

1. *Approximate the daily growth function* by fitting a linear regression model with observed mean length as the response variable and median capture DOY as the predictor variable.
2. *Quantify the year-specific mean length deviates* at the time of sampling by dividing the observed mean length by the mean length predicted by the regression model.
3. *Obtain the expected mean length at the average median sampling DOY* to represent the baseline size of fish in any given year if sampling always occurred with the same timing.
4. *Apply the mean length deviate* for each year by multiplying the deviate obtained in step (2) by the expected mean length in step (3).

Although timing varies for spring length sampling, we did not correct for it. The passive screw trap capture method for smolt length sampling means that this variability is related to migratory behavior, not sample timing.

Individual Length & Apparent Survival

For the analysis of individual outcomes, we queried PTAGIS for fish tagged (in years after 2000) during the summer event and retained (or derived) several key variables including: brood year, population of origin, size at tagging, and fate. The “fate” variable ($D_{i,t,j}$) is a binary indicator of apparent survival at the individual-level: 1 indicates that the fish was detected (i.e., known alive) at LGR or any point later and 0 if not. This is an index of apparent survival because we did not account for fish that survived to LGR but were not detected (i.e., false negatives), thus calculating the fraction of all tagged fish that were detected will under-estimate survival. We deemed that using apparent survival is appropriate for investigating a size

effect on individual survival rates based on the assumption that detection is independent of fish size, or at least not positively related. Hostetter et al. (2015) investigated survival and detection probabilities of spring Chinook salmon in the Columbia River hydrosystem and found that although survival was positively related to fish size, detection probability was negatively related to fish size. Thus, this directional selection would result in dampening, not exacerbating, any positive size effects on survival we quantify by using only apparent survival.

Analyses

We performed one analysis per research question that involved relating a response variable to a predictor variable using regression methods. Most analyses (all excluding those for Question #4) fitted one simple linear regression model per population with individual years serving as replicate data points. For Question #4, we fitted logistic regression models to each year separately treating individual outcomes as replicates. No analyses accounted for measurement error in the data and all assumed independence among populations and years (and in the case of Question #4, individuals), indicating that we will under-represent the true degree of uncertainty in any relationships. Although we present and display quantities like p -values and 95% confidence intervals, we focus much more of our interpretive attention on the directionality of relationships and their consistency across populations rather than these uncertainty-related outputs. These analyses were exploratory in nature – a key role of the SSLCM is to relax these unrealistic assumptions.

Linear Regression Fits

Most analyses used simple linear regression with the appropriate transformations applied for normality approximation of residuals. Let $y_{t,j}$ and $x_{t,j}$ be the values of response and predictor variables, respectively, for population j in year t – for example, x or y may represent the mean length of the population at a given life stage, the abundance at a life stage, or the survival [expressed as a probability on the interval (0,1)] from one life stage to another. For each question-specific analysis, we fitted four regression models (one per population) of the form:

$$f(y_{t,j}) = \beta_{0,j} + \beta_{1,j} \times g(x_{t,j}) + \varepsilon_{t,j}$$

$$\varepsilon_{t,j} \stackrel{\text{iid}}{\sim} N(0, \sigma_j^2)$$

where $f()$ and $g()$ are (optional) transformations on the response and predictor variables, respectively, $\beta_{0,j}$ and $\beta_{1,j}$ are the intercept and slope for population j , and $\varepsilon_{y,j}$ is the residual term with variance σ_j^2 . All models included fixed-effects only (i.e., all $\beta_{0,j}$, $\beta_{1,j}$, and σ_j^2 were independent).

Inference was based primarily on two outputs for each fitted model: the visual relationship and a two-sided hypothesis test against $H_0: \beta_{1,j} = 0$. For visualization purposes, the model predicted value $\hat{y}_{i,j}$ for some arbitrary value i at $\hat{x}_{i,j}$ (where $\hat{x}_{i,j}$ takes on values within the range observed for population j)

$$\hat{y}_{i,j} = f^{-1}[\hat{\beta}_{0,j} + \hat{\beta}_{1,j} \times g(\hat{x}_{i,j})]$$

was plotted (with 95% confidence band) over top of the scatterplot of observed values. In drawing the conclusion about H_0 , we used $\alpha = 0.05$ and a Bonferonni correction for the 28 hypotheses tested (7 questions using this approach \times 4 populations), which required that the p -value of a given test must be <0.002 (rather than 0.05) for H_0 to be rejected.

Logistic Regression Fits

Let $D_{i,t,j}$ represent the detection status of tagged fish i in year t for population j , coded as $D_{i,t,j} = 1$ if the fish was detected alive at LGR or a downstream location, and $D_{i,t,j} = 0$ otherwise. We fitted models of the form:

$$\text{logit}(\pi_{i,t,j}) = \alpha_{0,t,j} + \alpha_{1,t,j} \times L_{i,t,j}$$

where $\pi_{i,t,j}$ is the apparent survival probability for individual i , $\alpha_{0,t,j}$ and $\alpha_{1,t,j}$ are population- and year-specific intercepts and slopes (i.e., log odds and log odds ratio), and $L_{i,t,j}$ is the length of individual i upon tagging. The models estimated the coefficients ($\hat{\alpha}_{i,t,j}$ and $\hat{\alpha}_{1,t,j}$) using maximum likelihood methods assuming that all $D_{i,t,j} \sim \text{Bernoulli}(\pi_{i,t,j})$.

Inference was also primarily based on two outputs: the visual collection of year-specific fitted survival vs. length relationships and the proportion of $\hat{\alpha}_{i,t,j}$ terms that were positive (indicating that survival probability increases with increasing length). For visualization purposes, the model predicted value ($\hat{\pi}_{k,t,j}$) at some arbitrary length value ($\hat{x}_{k,t,j}$, takes on values within the range observed for population j in year t)

$$\hat{\pi}_{k,t,j} = \text{logit}^{-1}(\hat{\alpha}_{0,t,j} + \hat{\alpha}_{1,t,j} \times \hat{x}_{k,t,j})$$

was plotted for all years together.

For Question #6, we wished to control for the effect of size on survival. We thus calculated the model-fitted apparent survival value of a 65mm fish ($\hat{\pi}_{65,t,j}$):

$$\hat{\pi}_{65,t,j} = \text{logit}^{-1}(\hat{\alpha}_{0,t,j} + \hat{\alpha}_{1,t,j} \times 65)$$

to serve as the response variable in that regression analysis.

Results

Parr Mean Length Standardization

There was evidence (for all populations except UGR) of systematic variability in the parr mean length data due to sample timing (Figure 26a-d). Additionally, there appeared to have been a trend of sampling earlier in the summer in more recent years than in past years for some populations, evidenced by the sharply decreasing time trends in observed parr mean length (Figure 26, e.g., CAT, panel e). The ultimate result of our standardization was an overall flattening of mean length time trends and reducing the magnitude of extreme values (Figure 26e-h).

Question #1: Does Parr Recruitment Appear DD?

All four populations displayed decreasing relationships between egg-to-parr survival rates $\phi_{E \rightarrow P_{\text{summer}}}$ and total egg production (E ; Figure 27, row #1), although only one population (CAT) had a slope term that was statistically significant after Bonferonni correction. Still, this implies evidence for early-life DD survival occurring after egg deposition and before the end of summer.

Question #2: Does Pre-Recruitment Growth Appear DD?

Years with lower egg production were associated with years of larger parr on average upon recruitment ($L_{P_{\text{summer}}}$; Figure 27, row #2), indicating evidence of early-life DD effects on growth rates. The relationships were estimated to be highly non-linear such that the effect of increasing E on $L_{P_{\text{summer}}}$ was much greater (i.e., steeper slope on natural scale variables) at small values of E than at larger values,

consistent with DD growth patterns for stream-dwelling salmonids described in the literature (Grant and Imre 2005).

Question #3: Does Parr Survival to LGR Appear DD?

In addition to DD survival occurring prior to parr recruitment (Question #1), there was also evidence of DD survival occurring between the end of summer and arrival to LGR (Figure 27, row #3). All populations showed a declining relationship with average survival near 0.2 and 0.05 at the lowest and highest parr abundances, respectively. This survival term ($\phi_{P_{\text{summer}} \rightarrow S_{\text{LGR}}}$) encompasses all post-recruitment processes occurring in-basin, prior to and including the outmigration to LGR, and is estimated directly from PIT tag data.

Question #4: Do Survival Outcomes Correlate with Individual Size?

Unlike the other questions, the analysis for this question analyzed individual detection outcomes. The majority of the population/year combinations (93%) displayed positive effects of size ($L_{i,t,j}$) on apparent survival probability (π ; Figure 27, row #4). Additionally, in many population/year combinations (64%), the length effect was larger than 0.025, implying that for every 1cm increase in length, the odds of apparent survival increased by a factor of 1.28. Since we assume that detection probability is not positively related to fish size, we interpret this as a minimum effect size for the majority of years.

Question #5: Do Years of Faster Parr Growth Coincide with Higher Survival?

Parr mean length was positively related to population-level survival to LGR for all populations – indicating that years with faster early-life growth on average coincide with higher survival following recruitment (Figure 27, row #5).

Question #6: Does there Appear to be Additional DD Survival After Controlling for Growth?

Apparent survival rates standardized to be applicable to fish of a common size (65mm, $\hat{\pi}_{65}$) showed non-existent or very weak relationships with parr abundance (Figure 27, row #6), indicating that much of the variability in the Question #4 relationship can be explained by size-based survival following recruitment.

Question #7: Does Population Smolt Length Correlate with Survival?

Similar to the finding from Question #5 of a positive relationship between $\phi_{P_{\text{summer}} \rightarrow S_{\text{LGR}}}$ and $L_{P_{\text{summer}}}$, we also found consistently positive relationships between $\phi_{S_{\text{spring}} \rightarrow S_{\text{LGR}}}$ and $L_{S_{\text{spring}}}$ (Figure 27, row #7).

Question #8: Is Parr to Smolt Growth Related to Parr Size?

The relative change in mean length from parr to smolt ($\Delta_{P_{\text{summer}} \rightarrow S_{\text{LGR}}}$) was negatively related to parr mean length for all four populations and the realized values were distributed fairly tightly around the mean relationship (Figure 27, row #8).

Ramifications for SSLCM

Below we have summarized these findings and their implications for how we will structure the freshwater juvenile dynamics in the SSLCM (numbers in parentheses indicate which answered research question[s] led to each conclusion).

The reproductive link from spawners to parr must be DD. (1)

The finding that egg-to-parr survival declines with increasing egg production implies that we cannot simply model parr recruitment as egg production multiplied by some constant conversion factor. Failing to account for DD survival pre-recruitment could lead the model to give wildly optimistic predictions.

Post-recruitment survival dynamics must be DD. (3)

Not only did we find that the fraction of all eggs that recruit to the parr stage is a DD process, but we also found that post-recruitment survival in FW (i.e., $\phi_{P_{\text{summer}} \rightarrow S_{\text{LGR}}}$) also appears to be DD. Similarly, unrealistic predictions may well occur as a result of failing to account for the post-recruitment DD dynamics, compelling us to capture it somehow using process model equations.

It should be noted that the strength of this relationship may be exacerbated in the analyses here, given that $\phi_{P_{\text{summer}} \rightarrow S_{\text{LGR}}}$ (the response variable) was used to derive P_{summer} (the predictor variable). A key role of the SSLCM will be to quantify relationships based on latent states that must jointly explain all data sources, thus removing the sort of issues potentially caused by our analysis of variables derived from one another. No other analyses in this chapter require this specific caveat.

Growth/size may be a useful mechanism to reflect density effects on survival. (2, 4, 5, 6, 7)

In addition to post-recruitment survival to LGR ($\phi_{P_{\text{summer}} \rightarrow S_{\text{LGR}}}$), we found that mean parr size ($L_{P_{\text{summer}}}$) was also negatively related to density. As has been previously reported (e.g., Zabel and Achord 2004), we found that individual size was positively related to (apparent) survival. Further still, when controlling for fish size, we found that density did not explain remaining variability in survival rates. This coupling of relationships (and non-relationships) indicates that pre-recruitment DD growth is a plausible mechanism to explain post-recruitment DD survival effects, and that it is worth exploring in the SSLCM.

Overwinter survival could possibly be modeled as size-dependent. (5, 7)

We found that survival during the entire post-recruitment FW period ($\phi_{P_{\text{summer}} \rightarrow S_{\text{LGR}}}$) and the portion only involving outmigration ($\phi_{S_{\text{spring}} \rightarrow S_{\text{LGR}}}$) were both related to mean length at the start of each period. So we know that it may be useful to model $\phi_{S_{\text{spring}} \rightarrow S_{\text{LGR}}}$ as a function of smolt mean length, however, we do not know whether survival in the period between recruitment and outmigration (i.e., overwinter rearing from parr to smolt) should also be modeled as size-dependent. Given this uncertainty, and that size-dependent overwinter survival has been reported previously (Ebersole et al. 2006), we plan to evaluate two versions of the model in this regard: one that assumes no size effect on over-winter survival and one that estimates the effect of parr mean length on overwinter survival. Both models will have process noise terms to account for unexplained variability – if the survival process variance is much larger in the model without size effect modeled, that implies utility in modeling size-dependent overwinter survival.

Changes in mean size should be modeled relative to the initial size. (8)

We had reason to suspect that the change in mean length from parr to smolt would be related to parr mean length (albeit in coho salmon; Ebersole et al. 2006), and indeed we found strong negative relationships here. This implies that the SSLCM should model the change in mean length between parr and smolt in a specific fashion that has a dependence on parr mean length.

Table 13. Summary of the research questions and variables used in their respective analyses. Symbols are described in Table 14, but broadly: survival (ϕ), apparent survival (π), fish length (L), length ratio (Δ), eggs (E), parr (P) and smolt (S).

#	Question	Variable		Transformation	
		Response	Predictor	$f()$	$g()$
1	Does parr recruitment appear DD?	$\phi_{E \rightarrow P_{\text{summer}}}$	E	logit()	–
2	Does pre-recruitment growth appear DD?	$L_{P_{\text{summer}}}$	E	ln()	ln()
3	Does parr survival to LGR appear DD?	$\phi_{P_{\text{summer}} \rightarrow S_{\text{LGR}}}$	P_{summer}	logit()	–
4	Do survival outcomes correlate with individual size?	$\pi_{i,t,j}$	$L_{i,t,j}$	logit()	–
5	Do years of faster parr growth coincide with higher survival?	$\phi_{P_{\text{summer}} \rightarrow S_{\text{LGR}}}$	$L_{P_{\text{summer}}}$	ln()	–
6	Does there appear to be additional DD survival after controlling for growth?	$\hat{\pi}_{65,t,j}$	P_{summer}	logit()	–
7	Does population smolt length correlate with survival?	$\phi_{S_{\text{spring}} \rightarrow S_{\text{LGR}}}$	$L_{S_{\text{spring}}}$	ln()	–
8	Is parr to smolt growth related to parr size?	$\Delta_{P_{\text{summer}} \rightarrow S_{\text{spring}}}$	$L_{P_{\text{summer}}}$	ln()	–

Table 14. Definitions of all symbols used in describing the analyses presented in this chapter. The "quantity" column denotes the type of variable (monitored = directly estimated by routine monitoring programs vs. derived) or model type it pertains to.

Quantity	Symbol	Description
Monitored	$\phi_{P_{\text{summer}} \rightarrow S_{\text{LGR}}}$	Estimated survival to LGR for parr tagged at end of summer
	$\phi_{P_{\text{fall}} \rightarrow S_{\text{LGR}}}$	Estimated survival to LGR for parr tagged in fall
	$\phi_{S_{\text{spring}} \rightarrow S_{\text{LGR}}}$	Estimated survival to LGR for smolt tagged in spring
	P_{fall}	Estimated passage of parr past the screw trap in fall
	S_{spring}	Estimated passage of smolt past the screw trap in spring
	$L_{P_{\text{summer}}}$	Mean length of parr captured at end of summer
	$L_{S_{\text{spring}}}$	Mean length of smolt captured in spring
Derived	E	Total egg production
	S_{LGR}	Abundance of smolt arriving at LGR
	P_{summer}	Abundance of parr recruiting to end of summer
	$\Delta_{P_{\text{summer}} \rightarrow S_{\text{spring}}}$	Ratio of smolt to parr mean length
Linear Reg.	t, j	Year (t) and population (j) combination
	$y_{t,j}$	Generic response variable
	$\hat{y}_{i,j}$	Model-fitted generic response variable at arbitrary covariate value ($\hat{x}_{i,j}$)
	$x_{t,j}$	Generic predictor variable
	$\beta_{0,j}, \beta_{1,j}$	Intercept and slope of linear regression
	$\hat{\beta}_{0,j}, \hat{\beta}_{1,j}$	Model-fitted intercept and slope of linear regression
	$\varepsilon_{t,j}$	Residual error term
σ_j^2	Residual error variance	
Logistic Reg.	i, t, j	Individual (i) by year (t) by population (j) combination
	$D_{i,t,j}$	1 if detected alive at LGR or any downstream location, 0 otherwise
	$\pi_{i,t,j}$	Apparent survival conditioned on fish length
	$\hat{\pi}_{k,t,j}$	Model-fitted apparent survival at arbitrary length value k
	$L_{i,t,j}$	Individual length at summer tagging
	$\alpha_{0,t,j}, \alpha_{1,t,j}$	Intercept and slope of logistic regression
	$\hat{\alpha}_{0,t,j}, \hat{\alpha}_{1,t,j}$	Model-fitted intercept and slope of logistic regression
$\hat{\pi}_{65,t,j}$	Model-fitted apparent survival at length of 65mm	

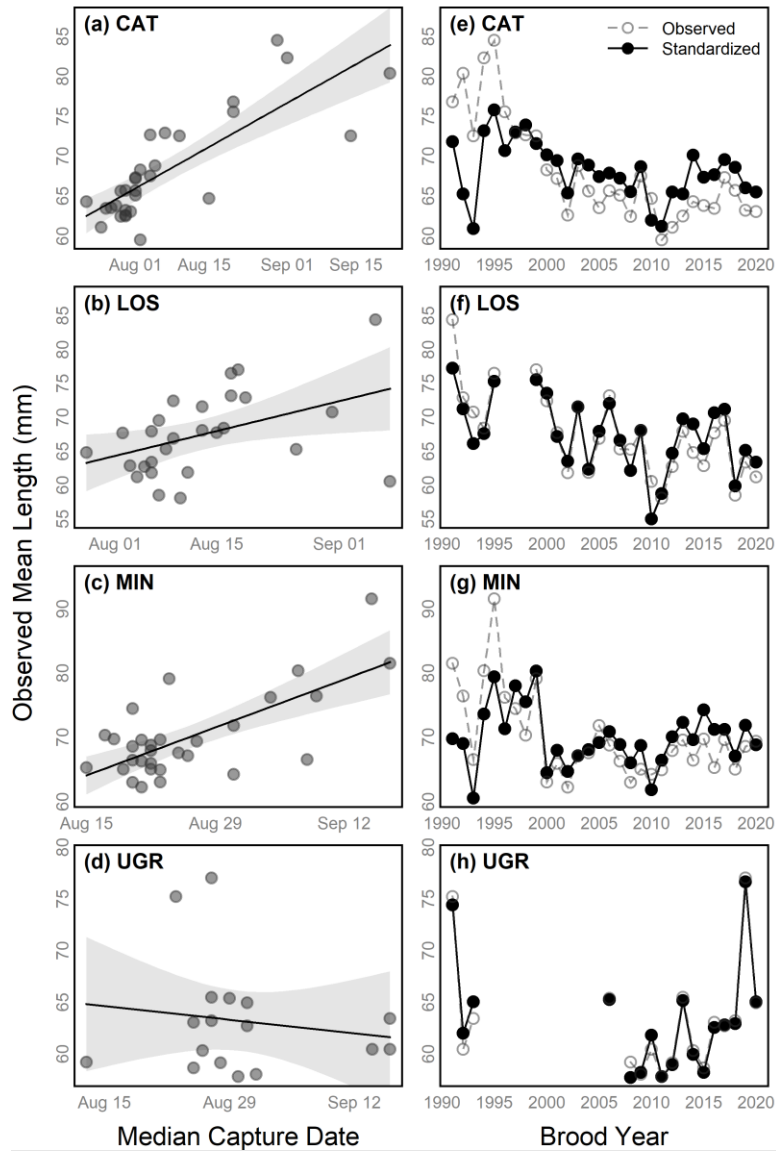


Figure 26. Population-specific relationships between mean parr length (fork length, mm) and the median date of tagging (panels (a-d) and population-specific comparisons of observed and standardized mean parr length (panels e-h). The relationships (a-d) show the need for the temporal standardization in most populations, and the time series (e-h) show the magnitude of standardization that occurred.

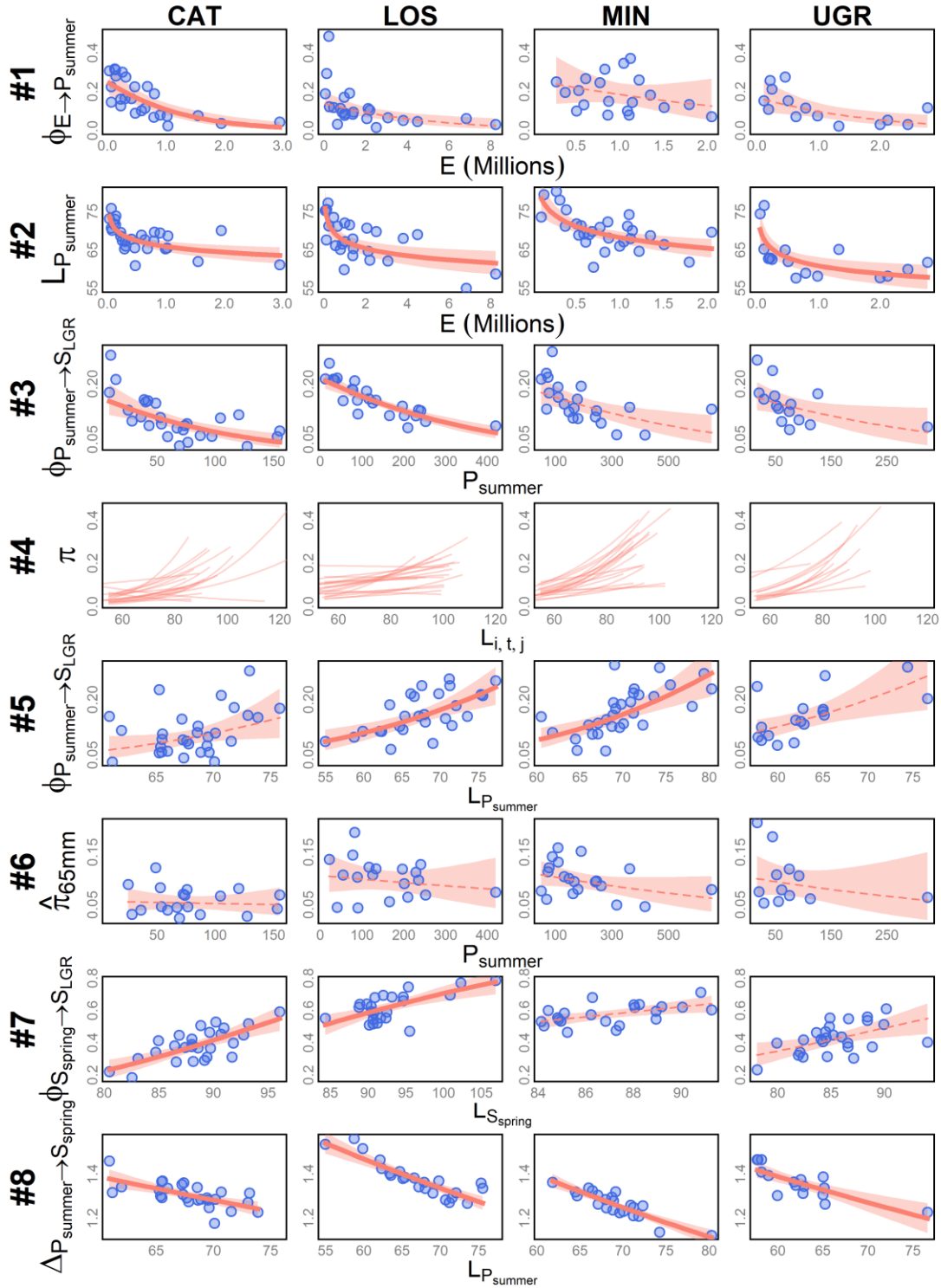


Figure 27. Regression relationships (red; best-fit with 95% confidence band) based on monitoring data (blue; points are individual years). Dashed best-fit lines indicate a failure to reject H_0 ($\beta_{1,j} = 0$) after Bonferonni correction. Panels in a row show the population-specific relationship for each question and have identical y-axis limits. Question #4 relationships are shown as year-specific logistic regressions. All length measurements are mm. Symbols are described in Table 14, but broadly: survival (ϕ), apparent survival (π), fish length (L), length ratio (Δ), eggs (E), parr (P), and smolt (S).

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Objective E-2: LCM Phase II – management scenarios

Restoration scenarios to impose on the SSLCM model output have not yet been formally developed, although several planning meetings, including a well-attended workshop devoted to brainstorming with partners in the basin, have occurred and a loose plan has been developed. Briefly, we plan to assess combinations of restoration and other management actions (e.g., changes to supplementation) of various intensities. The most interesting of these scenarios will be crossed with additional scenarios that involve out-of-basin factors, such as changes to the hydrosystem or ocean mortality. More details will be forthcoming in the 2023 annual report.

Objective E-3: LCM Phase III – simulation of outcomes

We have not yet built the simulation model based on the SSLCM output because the SSLCM model structure has been in flux and the management scenarios are not yet finalized. Some progress on constructing the simulation model is anticipated in 2023, and some preliminary results may be available for the 2023 report.

Goal 4: Apply lessons learned from RM&E to Grande Ronde salmon recovery efforts and other emerging concerns or locations

Objective F: Address needs of CRITFC tribal and other partners evaluating emerging concerns in the Grande Ronde and other geographic locations

Objective F-1: Continue developing/applying Adaptive Management framework with GRMW and Grande Ronde basin partners

Adaptive Management Plan

We have continued to work closely with colleagues at the Grande Ronde Model Watershed (GRMW) and other Grande Ronde basin partners on formalizing and documenting an Adaptive Management Plan for the Grande Ronde basin. An initial version of the Adaptive Management Plan was presented in White et al. (2021), a collaborative publication assessing progress to date in habitat and salmon restoration in the Grande Ronde basin and was further refined in 2022 (Figure 28). In 2022, we participated in the Grande Ronde State of the Science Adaptive Management Workshop hosted by GRMW. The workshop provides a forum to present and discuss restoration progress, RM&E findings, and emerging uncertainties/questions pertinent to management efforts in the basin, all of which were formally documented. Some important highlights from the workshop included 1) a recent sediment supply study (Wolf Water Resources 2022) revealed the extent of sediment loss from the basin, the long-time scales associated with sediment recovery, and reach-scale estimates of sediment supply dynamics, which provide a highly useful tool to prioritize the locations of floodplain restoration projects, 2) steelhead productivity appears to be declining and more monitoring of steelhead populations is needed, 3) additional PIT tag arrays positioned at key locations are needed to better understand reach- or site-specific mortality, 4) restoration projects have increased in intensity and capacity at the site scale, but cumulative restoration impacts are probably not enough to see measurable changes in population-scale carrying capacity, 5) dissolved oxygen in the lower Grande Ronde valley was below water

quality standards at certain locations and times and should be investigated further. Detailed notes on the outcome of the adaptive management workshop are provided in Appendix C.

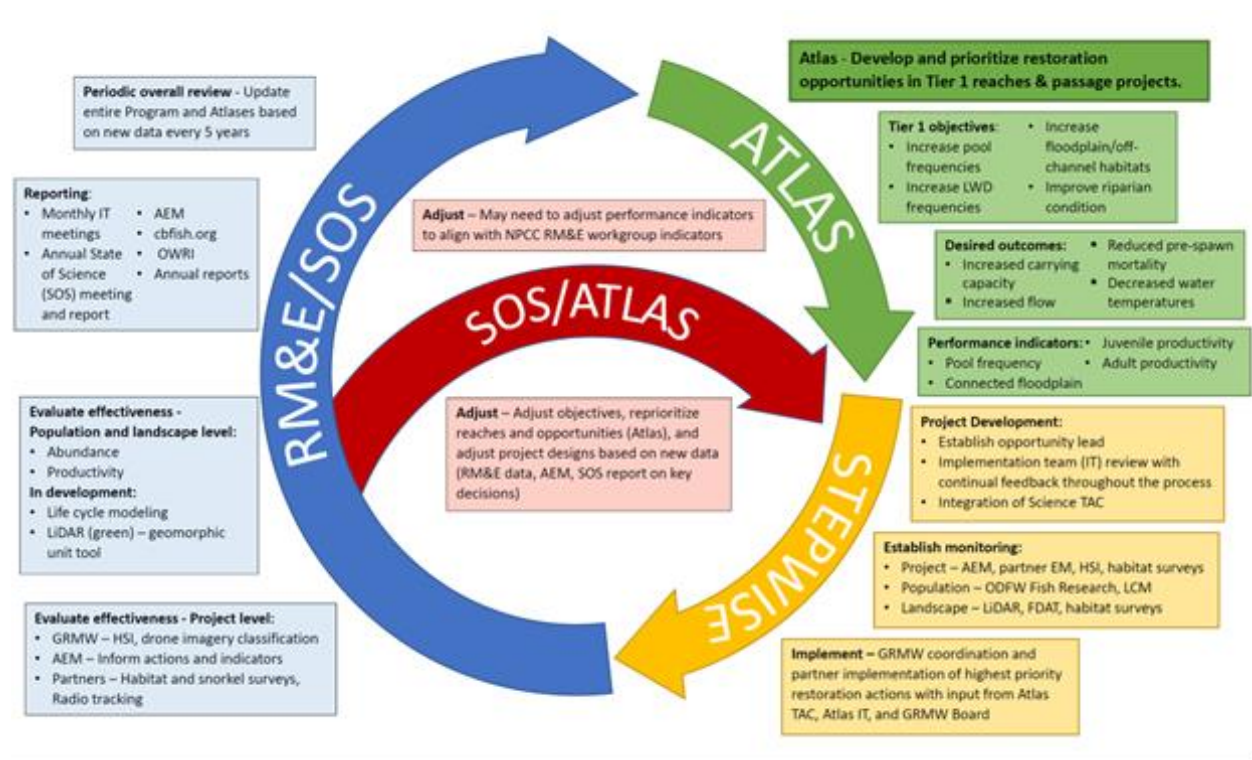


Figure 28. The adaptive management framework established by GRMW and GR basin partners.

Watershed Coordination

Coordination with other entities involved in fish habitat restoration and associated data collection in the Grande Ronde, Upper Columbia, Mid-Columbia, and Snake River watersheds is a critical component of this project. There are numerous other agencies (e.g., BPA and NOAA) evaluating related questions and implementing similar projects in other basins (e.g., Upper Columbia, S.F. Salmon, and the John Day). Close coordination and communication are essential to ensure that data being collected across various watersheds are similar enough that they can be used to draw inferences about the broader Columbia River basin (e.g., Is the habitat adequate for salmonid survival to specific life-cycle stages?; Are key limiting factors for habitat quality improving with time and as an aggregate or site-specific expression of restoration actions taken?). Coordination among agencies also helps to reduce duplication of effort and allows pooling of resources and knowledge to answer broader and more impactful research questions. A list of key partner projects that are closely related to the proposed work are listed in Table 15.

CRITFC is a partner in the Grande Ronde Atlas process and provides basin-scale analysis of restoration effects which feed into restoration management decisions (e.g., Figure 29). Additionally, CRITFC staff participated in the Grande Ronde Restoration Atlas review workshop led by GRMW to ensure that CRITFC and the tribes have a voice in restoration planning efforts in the basin.

In 2022, the USDA Forest Service spearheaded a multi-agency project to implement and monitor a whole-watershed, process-based restoration effort in Meadow Creek, a tributary to the upper Grande Ronde River. As a partner in this collaborative restoration project, we participated in regular meetings with USDA Forest Service staff and other basin partners (CTUIR, ODFW, BPA), and contributed to development of a comprehensive restoration plan including assembling available data and content describing the current status of physical characteristics, biota and land use history across the watershed. Beginning in 2023, the USDA Forest Service PNW Research station and CRITFC will enter into a formalized cooperative agreement to collaborate on research and monitoring of restoration projects in the upper Grande Ronde River and its tributaries including a Forest Service funded post-doc position supervised by CRITFC to focus on this work.

We participated in several meetings with partners from OSU, CTUIR, NOAA and ODFW to discuss and plan a project evaluating Chinook Salmon dispersal in Catherine Creek, including how restoration, location within the watershed and adult origin may influence juvenile salmon dispersal patterns, distribution, and habitat use. This collaborative research project is expected to begin in summer of 2023 and continue through 2024.

We hosted salmon habitat monitoring and snorkel survey training workshops in July of 2021 and 2022 involving participants from multiple agencies including CTUIR, USDA Forest Service, ODFW, and GRMW. The training focused on CRITFC's Tributary Habitat Assessment Protocol (TribAP), fish identification, and standardized snorkel survey methods. Following the training, we collaborated closely with GRMW and USFS staff to implement these monitoring protocols within the Grande Ronde basin.

The development of the state-space life cycle model for Grande Ronde spring Chinook salmon has been a highly collaborative process between CRITFC, NOAA, and ODFW. Staff from each organization have contributed to the development following their individual strengths. NOAA staff developed an early version of the state-space model and have been instrumental in providing advice and feedback on changes to the model structure. ODFW staff, being most intimately familiar with the biological monitoring programs in the basin, have been primarily in charge of decisions around how to treat the various data sources and for compiling and maintaining the data sets into a format usable by the state-space model. CRITFC staff, which includes a mixture of salmon population modelers and salmon habitat and ecology experts, have done the model development involved with revising the original state-space model and have compiled estimates of weighted usable rearing habitat that will serve a key role in all phases of this work. All parties have been consulted prior to making major developmental changes to the model and will be invited to serve as co-authors on manuscripts that are produced as result of this collaboration.

Table 15: List of key partners and projects related to the proposed work in the Grande Ronde (GR) basin and beyond.

Organization	Related project(s)	Relationship to proposed project
Bonneville Power Administration	Project Action Effectiveness Monitoring (AEM) Programmatic (BPA 2016-001-00)	Findings from the BPA AEM project will be useful in developing and evaluating CRITFC's models of aggregate restoration impacts on limiting factors
Bureau of Indian Affairs	Climate Change Threats to Salmonid Food Webs (BIA A19AV00480); Resilient Aquatic Food Webs for Tribal Communities (BIA A19AP00024)	Provided funding for development and analysis of benthic macroinvertebrate metrics related to salmonid food webs; funded expansion of benthic macroinvertebrate sampling to tribal partners in the Columbia basin
Confederated Tribes of the Umatilla Indian Reservation	Grande Ronde Watershed Restoration (BPA 1996-083-00); Grande Ronde Supplementation M&E (BPA 2007-083-00); Biomonitoring of Fish Habitat Enhancement (BPA 2009-014-00)	GR Atlas Partner; implements habitat projects that CRITFC's surveys of habitat and biological monitoring characterize; has adopted CRITFC M&E methodologies (e.g., snorkel surveys, benthic macroinvertebrate collections); leads weir sampling of adult Chinook which produces data used in our life cycle model; Chinook Salmon supplementation program will be assessed using a life cycle model
Grande Ronde Model Watershed Foundation	Grande Ronde Model Watershed (BPA 1992-026-01)	Leads coordination of adaptive management framework (Atlas) in GR basin; uses CRITFC's limiting factors assessments to guide restoration prioritization; documents restoration activities in basin that CRITFC will use for modeling restoration impacts on limiting factors; co-funded collection of topobathymetric LiDAR in 2020
National Oceanic and Atmospheric Administration	Various	Key partner with CRITFC and ODFW in developing the life cycle model; co-funding CRITFC's research on emergence phenology, floodplain use, and early life history of Chinook Salmon; co-funded a public outreach film with GRMW, CRITFC, and USFS on restoration in the GR basin
Nez Perce Tribe	Protect & Restore NE OR & SE WA Watershed Habitat (BPA 2007-393-00)	GR Atlas associate; utilized GIS products developed by CRITFC for restoration planning in the Wallowa basin; CRITFC collected topobathymetric LiDAR in Wallowa basin in 2020 that will be used for limiting factors assessment; provided water temperature data used in recent analysis of Chinook pre-spawn survival
Oregon Department of Fish and Wildlife	Grande Ronde Fish Habitat Improvement (BPA 1984-025-00); Grande Ronde Salmonid Life Cycle Monitoring (BPA 1992-026-04)	GR Atlas partner; implements habitat projects that CRITFC's surveys of habitat and biological monitoring characterize; has adopted CRITFC M&E methodologies (e.g., snorkel surveys, benthic macroinvertebrate collections); collects population level fish data (e.g., spawner abundance) used in life cycle model; key partner with CRITFC and NOAA in developing the life cycle model

Organization	Related project(s)	Relationship to proposed project
Oregon State University & University of Oregon	Long-term ecological effects of passive restoration in the Middle Fork John Day (OWEB 218-6041); CTUIR John Day Watershed Restoration (BPA 2007-397-00); ODFW John Day Habitat Enhancement (BPA 1984-021-00)	OSU & UO Initiated study of long-term effects of cattle grazing management on river channel, riparian, and floodplain conditions in Middle Fork John Day; CRITFC contributed benthic macroinvertebrate sampling and analysis
U.S. Bureau of Reclamation	Various	Implements and conducts RM&E on various restoration projects in GR basin, implements habitat projects that CRITFC's surveys of habitat and biological monitoring characterize
U.S. Forest Service	Various	GR Atlas partner; conducts M&E (Level II surveys) and implements habitat projects that CRITFC's surveys of habitat and biological monitoring opportunistically characterize
Union Soil and Water Conservation District	Various	GR Atlas partner; implements habitat projects that CRITFC's surveys of habitat and biological monitoring opportunistically characterize

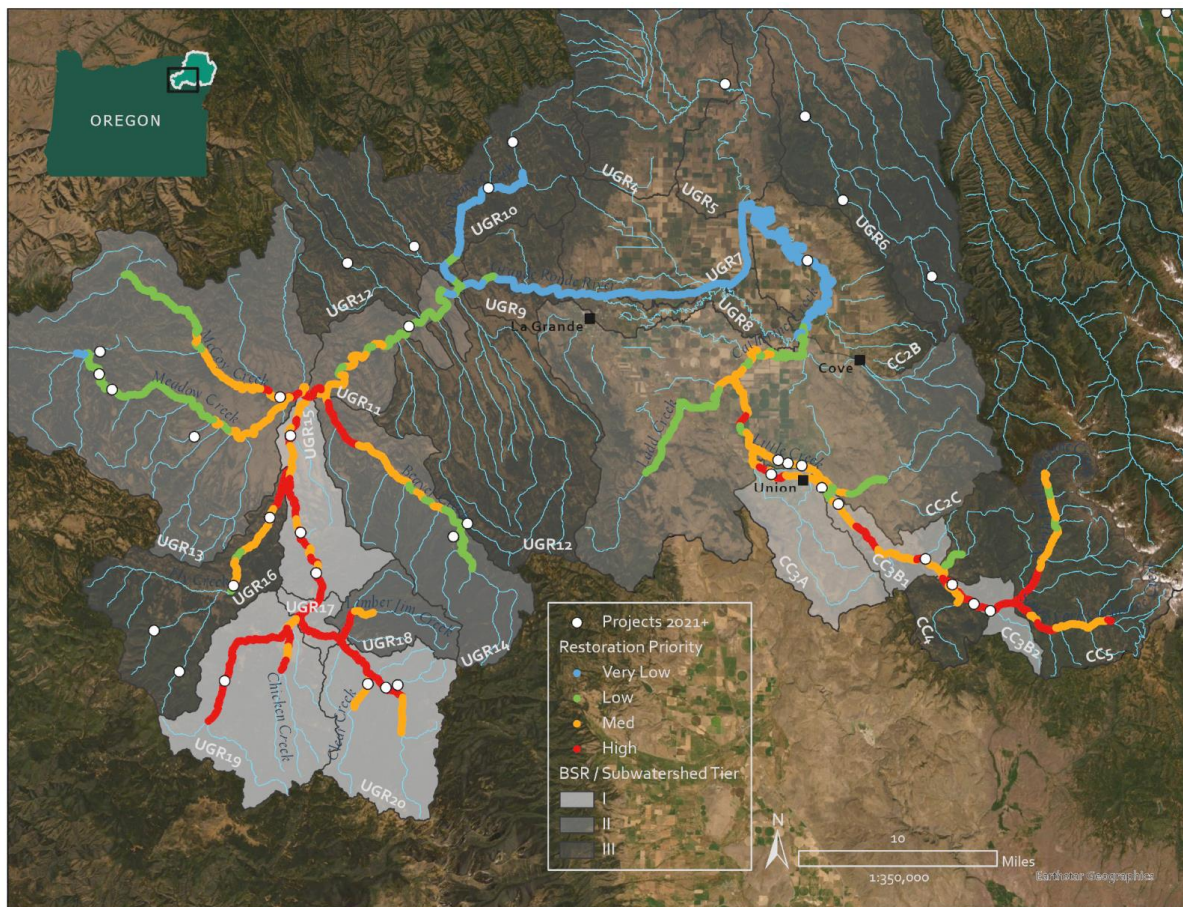


Figure 29. Restoration projects planned in the upper Grande Ronde River and Catherine Creek in 2021 and later, in relation to Atlas Tier I-III biological significant reaches (BSRs) and the restoration prioritization from Justice et al. 2017 developed in BPA project 2009-004-00. Most projects outside of the high and medium priority areas are either passage improvements or small, headwater stream meadow restoration projects by the USFS.

RM&E Methods and Designs

Tributary Habitat Monitoring Protocol Updates

Some updates were made to CRITFC's Tributary Habitat Assessment Protocol based on lessons learned from the 2021 field season. Most notably, we included an additional "marsh" channel unit type to account for inundated floodplain features or wetland habitats that were not adequately described by typical pool, fast turbulent, or fast non-turbulent channel unit classes. These habitat types are more commonly observed in locations that have been recently restored using stage-0 or other intensive floodplain restoration treatments, or habitats influenced by beaver activity or restoration to mimic beaver dams. Other minor edits were made to some of the language to improve clarity and encourage crews to collect additional edge of water boundary points. All protocol changes were made in monitoringresources.org using suggested versioning procedures.

Snorkel Survey Protocol Updates

ODFW snorkel surveys conducted at Wallowa River restoration sites in 2022 indicated that the existing snorkel detection model developed by CRITFC and used by Grande Ronde partners to estimate abundance from snorkel counts (Staton et al. 2022) was poorly suited to estimate snorkel detection probability in very deep (e.g., pools > 0.5 m average depth) or complex channel units (e.g., > 0.2 pieces of large wood per m²) because these values exceed covariate ranges in the initial dataset used to develop the model. Additionally, the original snorkel detection dataset was heavily weighted towards steelhead habitat, resulting in a smaller sample size for Chinook Salmon and likely increased uncertainty around the snorkel probability estimates and covariate effects for Chinook. In response to this shortcoming, CRITFC and ODFW plan to collect additional paired snorkel count and mark-recapture data during 2023, targeting deeper and more complex habitats that are within the known rearing area for Chinook Salmon.

Freshwater Mussel Survey Methods

Xerces Society for Invertebrate Conservation in collaboration with CTUIR are developing a regional survey protocol for freshwater mussels. Freshwater mussels are an important component of tribal first foods and are among the most threatened freshwater species globally. In order to assist with data collection needed to test and validate this protocol, we attended the freshwater mussel identification and sampling training held at the Oregon Chapter of the American Fisheries Society (AFS) meeting in Eugene, OR during spring, 2023. We plan to utilize the draft survey protocol during summer of 2023 in the Minam River watershed.

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- White, S. M., S. Brandy, C. Justice, K. A. Morinaga, L. Naylor, J. Ruzycski, E. R. Sedell, J. Steele, A. Towne, J. G. Webster, and I. Wilson. 2021. Progress Towards a Comprehensive Approach for Habitat Restoration in the Columbia Basin: Case Study in the Grande Ronde River. *Fisheries:fsh.10562*.

Objective F-2: Represent tribal concerns in Columbia River basin tributary habitat RM&E policy discussions

CRITFC staff participated in regular meetings and workshops over the past several years focusing on development and review of a regional tributary habitat research, monitoring, and evaluation (RM&E) strategy in coordination with tribes, Northwest Power and Conservation Council (NPCC), NOAA, and BPA. The primary goal of the draft strategy was to provide guidance for habitat RM&E actions funded by BPA to achieve better coordination among agencies conducting habitat RM&E, and to ensure that key management questions can be answered with the data collected. The final draft of the RM&E strategy document was completed in October of 2022 and the final draft was posted to the NPCC website (<https://www.nwcouncil.org/fish-and-wildlife/forums-and-workgroups/rme/>).

Staff from CRITFC's River Ecology group also participated in the Columbia Basin Collaborative (CBC) Estuary, Tributary & Mainstem Habitat meetings to develop regional recommendations to the Integration/Recommendations Group (IRG) regarding best management practices for habitat and salmon recovery in the Columbia River basin. This work with the CBC has provided a good opportunity to transfer lessons learned from extensive habitat-related RM&E in the Grande Ronde basin to a broader regional level and to ensure that tribal values and perspectives on habitat restoration and salmon recovery are considered in regional RM&E planning and funding.

Dissemination of Project Findings

Presentations

Burns, L.A. 2021. Pacific Northwest Aquatic Monitoring Partnership Remote Sensing Forum Reboot.

Burns, L.A. et al. 2021. Lessons from the Field: Project updates from the first season of Tributary Habitat Assessment in the Grande Ronde River basin. CRITFC Brown Bag Seminar. Virtual meeting.

Justice, C. et al. 2021. Habitat capacity index for Chinook Salmon parr in the Grande Ronde basin. Grande Ronde Life Cycle Modeling Workshop and Grande Ronde State of the Sciences Meeting. Virtual meetings.

Justice, C. et al. 2021. Potential for Stream Restoration to Mitigate Climate Change Impacts to Salmon Populations: A Case Study in the Grande Ronde River. American Fisheries Society Climate Change Webinar.

Justice, C., S. White, B. Staton, D. Graves, L. Burns, M. Kaylor. 2022. Habitat capacity index for Chinook Salmon parr in a Columbia Basin tributary. Presentation to the American Fisheries Society 152nd Annual Meeting, Spokane, WA.

Justice, C., S. White, D. McCullough. November 15, 2022. Stream temperature in the upper Grande Ronde and Meadow Creek. Available data, models, and potential applications for restoration planning. Presented to Meadow Creek Restoration and Research Planning Meeting work group. La Grande, OR.

Kaylor et al. 2021. Evaluating spatial and temporal patterns of chinook salmon growth, size, and emergence timing across NE Oregon rivers. Eastern Oregon University's Eastside Ecology Forum.

Kaylor, M.J. et al. 2021. Patterns of Chinook Salmon emergence timing and implications for size and growth. Grande Ronde Basin State of the Science annual meeting. La Grande, OR.

Kaylor, M., et al. February 7, 2022. Riverscape patterns of Chinook Salmon emergence phenology, growth, and dispersal in NE Oregon. OSU Department of Fish, Wildlife, and Conservation Science Monday Morning Meeting series.

Kaylor, M. et al. May 11, 2022. Assessing juvenile Chinook floodplain habitat accessibility and dispersal from redds to summer rearing habitats. Middle Fork John Day IMW and John Day Basin Partnership joint meeting.

Kaylor, M. and M. Bond., et al. May 19, 2022. Developing a framework for floodplain habitat utilization in the Middle Fork John Day River and Grande Ronde River Basin. Grande Ronde Model Watershed Implementation Team.

Kaylor, M., L. Ciepiela, J. Lemanski, M. Feden, I. Tattam, S. Kelly, C. Justice, S. White, E. Collins, S. Narum, and J. Armstrong. Juvenile Chinook Salmon dispersal patterns revealed using genetics as a tracer. Presentation to the American Fisheries Society 152nd Annual Meeting, Spokane, WA. August 23, 2022.

Kaylor, M., M. Bond, C. Justice, S. White, J. Lemanski, G. Pess and J. Armstrong. Integrating floodplain inundation dynamics and Chinook Salmon emergence phenology across riverscapes. Presentation to the American Fisheries Society 152nd Annual Meeting, Spokane, WA. August 23, 2022.

- Kaylor, M., M. Bond, C. Justice, J. Lemanski, G. Pess, and S. White. Evaluating floodplain habitat accessibility for juvenile Chinook Salmon: Linking floodplain inundation and emergence timing. Presented to Idaho Fish and Game regional meeting. October 25, 2022.
- Staton, B.A. et al. 2021. Grande Ronde spring Chinook life cycle model: model update/overview. Grande Ronde Life Cycle Modeling Virtual Workshop.
- Staton, B.A., Burns, L.A., Gibson, P., Justice, C., Kaylor, M., Liermann, M. Sedell, T., Sharma, R., and White, S. 2021. Grande Ronde Spring Chinook Life Cycle Model Update/Overview. Requested presentation at the February meeting of the Atlas Implementation Team.
- Staton, B.A., C. Justice, S. White, E. Sedell, L. Burns, M. Kaylor. 2022. An integrated model for quantifying covariates of detectability: Illustrated with snorkel surveys for riverine juvenile salmonids. Presentation to the American Fisheries Society 152nd Annual Meeting, Spokane, WA.
- Staton, B.A., P. Gibson, C. Justice, M. Kaylor, M. Liermann, T. Sedell, R. Sharma, and S.M. White. 2022. Grande Ronde Spring Chinook Life Cycle Model: Modeling Framework and Description of Freshwater Juvenile Dynamics. Adaptive Management Implementation Program meeting.
- White et al. 2021. Evaluating salmonid and stream ecosystem response to conservation measures and environmental stressors in the Columbia River basin. Proposal review for Northwest Power & Conservation Council and Independent Science Review Panel, Portland, OR.
- White et al. 2021. Progress towards a comprehensive approach for habitat restoration in the Columbia basin: Case study in the Grande Ronde River. CRITFC State of the Basin meeting, March 31, Portland, OR.
- White, S.M. 2021. Progress Towards a Comprehensive Approach for Habitat Restoration in the Columbia Basin: Case Study in the Grande Ronde River. Lower Snake River Compensation Plan meeting, Boise, ID.
- White, S.M. et al. 2021. Selected examples of life cycle model scenario output. Grande Ronde Life Cycle Modeling virtual meeting.
- White, S.M. et al. 2021. Setting targets for habitat conditions. Grande Ronde Life Cycle Modeling virtual meeting.
- White, S.M., B. Staton, P. Gibson, C. Justice, M. Kaylor, M. Liermann, T. Sedell, R. Sharma. 2022. Grande Ronde spring Chinook life cycle model: Collaborative development of management scenarios. Adaptive Management Implementation Program meeting.

Publications or Draft Manuscripts

- Kaylor, M.J., C. Justice, J.B. Armstrong, B.A. Staton, L.A. Burns, E. Sedell and S.M. White. 2021. Temperature, emergence phenology, and consumption drive seasonal shifts in fish growth and production across riverscapes. *Journal of Animal Ecology*, 00:1-15. <https://doi.org/10.1111/1365-2656.13491>

- Kaylor, M.J., J.B. Armstrong, J.T. Lemanski, C. Justice, and S.M. White. 2022. Riverscape heterogeneity in estimated Chinook Salmon emergence phenology and implications for size and growth. *Ecosphere* 13:e4160. <https://doi.org/10.1002/ecs2.4160>
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Appendix A: Data Storage and Access

Data collected under this project is currently managed in different ways depending on the data type. General spatial data such as monitoring sites, stream layers, land ownership, and various other datasets that are used for analyses but are not deliverable end products are stored in a geodatabase at CRITFC and managed for internal use by CRITFC GIS specialists and project managers. Physical habitat data collected by CRITFC using the CHaMP protocol from 2011 to 2017 is now managed by the StreamNet Program and is available to the public at <https://www.streamnet.org/home/data-maps/champ/>. Habitat data collected using CRITFC's Tributary Habitat Assessment protocol including drone imagery is stored on CRITFC servers and is available upon request. Once metric calculations are completed for 2022 habitat data, our plan is to migrate all habitat survey data to CRITFC's Centralized Database Management System (CDMS) with data available for download by the public upon request. Fish abundance data was recently uploaded to StreamNet's Data Store for use in expanding the Fish Density Analysis Tool (FDAT) developed by Dan Isaak and his team at the Rocky Mountain Research Station. These data are available for download at https://app.streamnet.org/datastore_search_classic.cfm?id=844. Water temperature data collected by CRITFC is stored on CRITFC's CDMS and is available for download upon request.

The code for the Grande Ronde spring Chinook state-space life cycle model and the data it fits to are stored in two separate GitHub repositories. We chose to organize it this way to enable tracking data-related changes separately from model-related changes, and this has proven to be quite useful. Further, all collaborators have access to the most current main and development versions and GitHub includes a useful issue tracker where needed changes can be discussed; these aspects have greatly facilitated collaboration. The data sets are not very large (relatively speaking; total of all files <5MB), which makes GitHub a workable solution for housing and updating the data sets for this endeavor. If the data were stored outside of basic text files (e.g., .csv format) or in very large files, we would need to find something different, as GitHub was not designed to track/host these sorts of files. Both repositories are private currently (only collaborators have access), however, we plan to make both completely public as supplements to the manuscript that presents this work.

Appendix B: Juvenile Salmon Dispersal using Parentage-Based Tagging

Abstract

For Pacific Salmon species, the dispersal of juveniles from spawning locations to rearing habitats is a critical process influencing individual fitness and population productivity. Studies evaluating juvenile salmonid dispersal have found that dispersal is generally limited (within 0.5 km of origin) and downstream biased, but that numerous biophysical factors can influence dispersal patterns. The expression of factors that restrict or promote dispersal likely varies across river networks, potentially resulting in differences in dispersal patterns reflective of local conditions. We utilized a riverscape sampling approach and genetics-based parentage assignments to evaluate dispersal patterns of a wild population of spring-run Chinook Salmon in the Middle Fork John Day River, NE Oregon. We sampled post-spawn female adults from spawning locations in 2020 and their offspring (parr) from summer rearing habitats across nearly 40 km of the mainstem and nine tributaries in 2021. Female adults ($n = 67$) and parr ($n = 3,388$) were genotyped, parr were assigned to female parents ($n = 1,326$), and dispersal was calculated using the location of females and paired parr. We also estimated parr abundance across the river network to correct for sampling bias effects, and to relate the spawning distribution and dispersal patterns to subsequent parr distribution. Overall dispersal estimates were downstream biased (median = -0.77 km) with 68% of parr dispersing downstream. Dispersal distances were high relative to those found in other studies, with 25% of parr dispersing more than 0.9 km upstream and 25% dispersing more than 3.7 km downstream (range = 28.6 km downstream - 10.6 km upstream). At finer spatial scales, dispersal patterns varied throughout the river network. Parr originating lower in the watershed exhibited upstream dispersal bias and less variability in dispersal patterns, whereas parr originating higher in the network had downstream dispersal bias, greater variability in dispersal, and generally dispersed farther. Combining dispersal patterns with estimates of parr distribution revealed that parr originating in areas with higher maximum July temperatures generally dispersed to cooler sections of the mainstem or tributaries, whereas parr originating in cooler sections of the mainstem were more likely to remain near their origin and within the mainstem. Our results suggest greater dispersal at early life-stages than prior published estimates and that heterogeneity in biophysical conditions across watersheds may translate to variable patterns of dispersal. These findings provide guidance for prioritizing the types and locations of restoration efforts to increase habitat utilization through better alignment with juvenile salmon ecology.

Introduction

Riverine fish experience a heterogeneous environment in which variability in physical habitat conditions, food resources, and biotic interactions (e.g., competition, predation) form a dynamic landscape of habitat quality (Stanford et al. 2005). Movement allows individuals to seek favorable habitats as conditions change to enhance individual fitness; for example, to minimize competition (Einum et al. 2008), avoid sub-optimal or lethal environmental conditions (Hahlbeck et al. 2022), and track food resources and thermal conditions to maximize growth opportunities (Armstrong et al. 2010; Baldock et al. 2016). However, movement may be limited by individual characteristics and biophysical conditions including life stage- or size-specific swimming capacity, landscape constraints (e.g., physical or thermal barriers), and risks associated with moving (e.g., predation). Across watersheds, heterogeneity in biophysical conditions may translate to variation in the expression of factors promoting and constraining movement, and thus fine-scale patterns of movement that are dependent on local conditions.

The movement of juvenile salmon from spawning nests (redds) to rearing habitats, hereafter termed “dispersal”, is a critical process affecting individual growth and survival (Einum and Nislow 2005), which collectively influence population dynamics through effects on juvenile distribution, habitat utilization, and production (Teichert et al. 2011; Einum et al. 2011). Anadromous salmon are highly fecund and the spawning locations of adults within a population are typically clustered within small (e.g., multiple redds within a pool tail-out) and large (i.e., core reaches within a basin) spatial scales (Beechie et al. 2008), resulting in high localized densities of recently emerged juveniles (Flitcroft et al. 2014). Juveniles that disperse to lower-density habitats typically exhibit greater growth and survival (Einum and Nislow 2005; Brunson et al. 2017; Aparicio et al. 2018), and collectively, these individual dispersal patterns can influence population-level density-dependent effects. Further, spatial patterns in juvenile rearing habitat quality may not align with spawning distributions, and dispersal facilitates juvenile habitat selection and rearing range expansion, including into tributaries and headwaters not utilized by spawning adults (Anderson et al. 2013; Scheu 2022). However, the spatial distribution of juvenile salmonids often mirrors adult spawning distributions (Foldvik et al. 2010; Atlas et al. 2015), suggesting limited overall dispersal, or alternatively, a high degree of habitat complementation between spawning and rearing habitats.

Empirical evaluations of juvenile salmon dispersal generally suggest that dispersal is limited (< 0.5 km of origin) and downstream biased (reviewed by Eisenhauer et al. 2021), conforming with the Restricted Movement Paradigm (RMP; Gerking 1959) which argues that most individuals in a population are sedentary. Yet empirical dispersal estimates are limited and there is increasing evidence challenging the RMP in juvenile salmon as dispersal is evaluated across a wider range of environmental conditions (Eisenhauer et al. 2021). The majority of studies evaluating dispersal have done so by out-planting eggs or fry to streams (reviewed by Eisenhauer et al. 2021), typically at small spatial scales and with low variability in biophysical conditions, whereas few studies have evaluated dispersal in naturally spawning populations (but see Anderson et al. 2013). Wild populations exposed to greater variability in inter- and intra-specific competition, environmental conditions, habitat quality, and emergence timing (Kaylor et al. 2022), may exhibit more variable dispersal patterns across watersheds that reflect responses and adaptation to local conditions. Indeed, studies evaluating juvenile salmonid dispersal in naturally spawning populations have reported large-scale dispersal of tens or hundreds of kilometers associated with alternative life-history strategies (Bradford and Taylor 1997; Scheu 2022). Few studies have evaluated fine-scale dispersal of juvenile salmonids from their origin to rearing habitats (but see Anderson et al. 2013), and to our

knowledge, no studies have evaluated population-level dispersal across the entirety of the adult spawning and juvenile rearing extents.

In this study, we utilized a riverscape sampling approach and genetics-based parentage assignments to evaluate juvenile dispersal patterns of a wild population of spring-run Chinook Salmon in NE Oregon. We sampled post-spawn adults from spawning locations and juveniles from mainstem and tributary rearing habitats the following summer. Adults and juveniles were genotyped, parent-offspring pairs were assigned, and dispersal was calculated as the stream distance between juvenile and adult locations for each parent-offspring pair. Further, we estimated juvenile salmon abundance across the watershed and related spatial patterns of dispersal to the resulting juvenile salmon distribution. We tested the following predictions: 1) overall, population-level dispersal patterns will reflect those observed in other studies, with downstream bias and the majority of individuals remaining within 0.5 km of their origin, 2) at finer spatial scales, dispersal patterns will vary across the watershed, reflecting differences in biophysical conditions promoting or inhibiting dispersal, and 3) dispersal distances will be associated with size (positive or negative relationship), which could arise from greater swimming ability of larger individuals (positive), from growth benefits associated with moving to lower density or higher quality areas (positive), or competitive interactions (e.g., feeding hierarchies) stimulating smaller individuals to disperse (negative). Understanding dispersal patterns in wild populations has important implications for prioritizing the locations and types of restoration efforts to increase habitat use and restoration efficacy through better alignment with juvenile salmon ecology.

Methods

Study Area and species description

The study was conducted in the Middle Fork John Day River (MFJD), a tributary of the John Day River in northeast Oregon. The MFJD watershed encompasses 2,051 km² and flows northwest from its origins in the Blue Mountains to its confluence with the North Fork John Day River (Figure 1). The study area is approximately 300 km from the Columbia River estuary and anadromous fish encounter three dams on the mainstem Columbia River en route to and from the MFJD. The region is characterized by hot summers with little precipitation and cold winters in which most of the precipitation is snow. Streamflow peaks during spring snowmelt – typically March to May – and is lowest during mid-to-late summer, a period coinciding with maximum water temperature. In summer, mainstem temperatures are warm across the juvenile rearing extent (Figure S1), but with slightly cooler temperatures in areas associated with tributary inputs (e.g., rkms 90-95 and 103-107). All nine tributaries were cooler than adjacent sections of the mainstem (Figure S1) in summer 2021 but varied in temperature and discharge (Table 1). Streams with the highest summer discharge include Granite Boulder Creek, Vinegar Creek, and Clear Creek and streams with coolest summer temperatures include Beaver Creek, Granite Boulder Creek, Dead Cow Gulch, and Clear Creek.

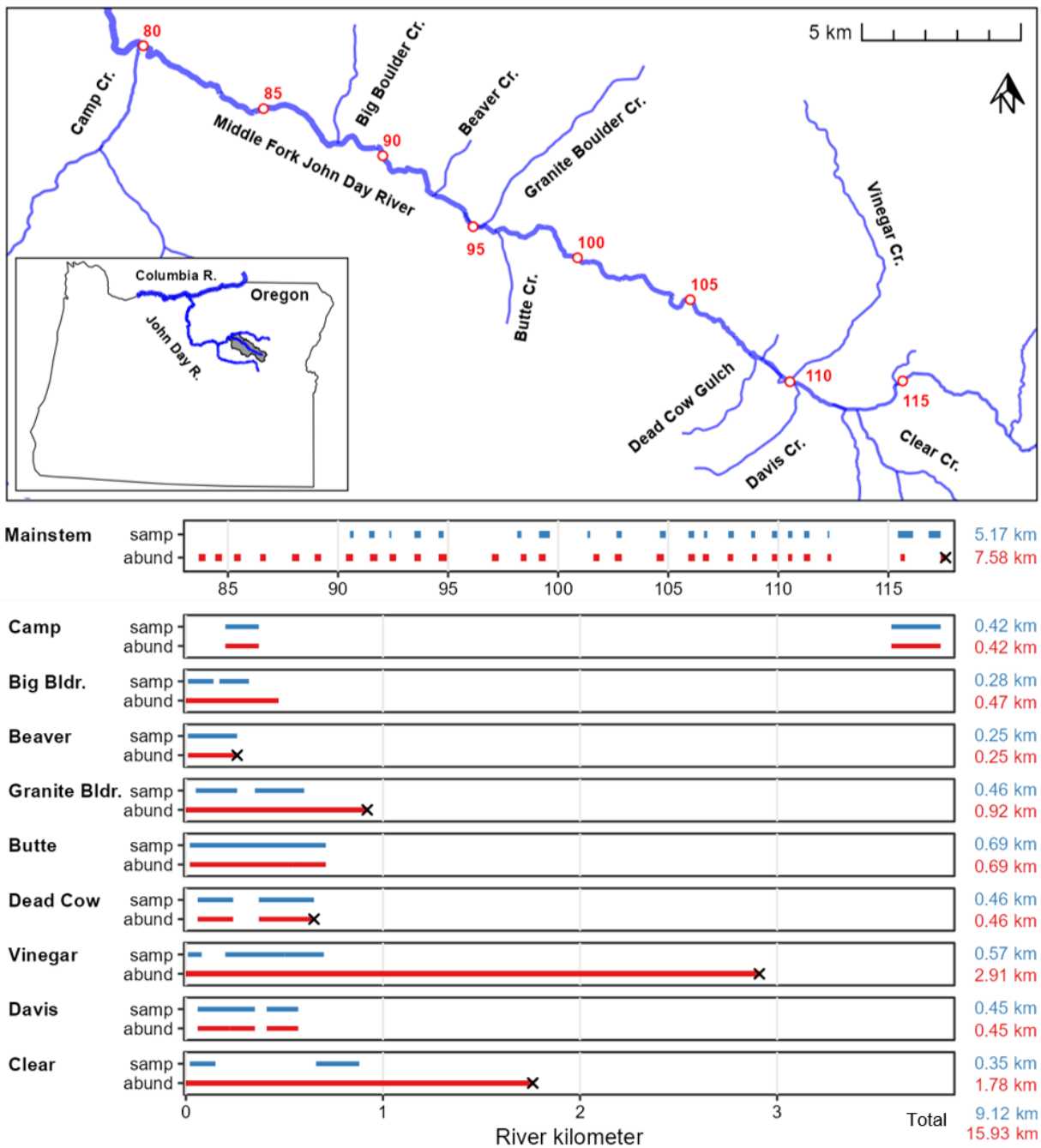


Figure 1: Study extent within the Middle Fork John Day River Basin (top panel) with reference river kilometers (red points and text), and the extent of parr sampling (samp; blue lines) and abundance surveys (abund; red lines) within the mainstem and tributaries (bottom panels). The total sampled or surveyed length within each stream is indicated in the right column. Black x's indicate that no juvenile Chinook were observed upstream of this location (i.e., upstream extent of Chinook Salmon parr within each stream).

Table 1: Parr sampling details for the mainstem Middle Fork John Day River and tributaries.

Stream	Rkm	July Q (m s ⁻¹)	June mean max. temp (°C)	July mean max. temp (°C)	Parr sampling				
					Dates sampled	Parr sampled	Mean fork length (mm)	Paired to female	%
Mainstem	-	596.0	15.7-23.0	21.8-26.2	7/13-8/9	1,592	67.3	595	37.4
Camp Cr.	79.8	12.2	20.2	21.3	7/9	28	60.0	13	46.4
Big Bldr Cr.	88.1	70.9	17.0	23.1	7/1-7/2	184	57.7	93	50.5
Beaver Cr.	92.8	21.3	16.8	19.6	7/14	63	62.2	12	19.0
Granite Bldr Cr.	95.1	124.3	12.7	17.5	6/30-7/22	292	68.3	80	27.4
Butte Cr.	96.4	4.1	15.8	18.8	7/14-7/20	248	61.1	70	28.2
Dead Cow Gulch	108.3	-	19.0	19.0	7/8-7/9	182	62.3	93	51.1
Vinegar Cr.	110.0	-	16.3	21.4	6/29-7/22	399	59.6	172	43.1
Davis Cr.	110.7	-	19.0	23.3	7/6-7/8	200	62.2	115	57.5
Clear Cr.	112.8	132.4	18.8	20.7	7/7-7/22	200	65.8	83	41.4
Total	-	-	-	-	6/29-8/9	3,389	64.7	1,326	39.1

Spring Chinook Salmon spawn throughout September with most spawning occurring between river kilometers (rkms) 80-115. Embryos incubate throughout the fall and winter, with emergence occurring between January and mid-May; however, emergence is spatially patterned with earlier emergence upstream (Kaylor et al. 2022). Parr are typically distributed throughout the mainstem spawning extent during summer and also utilize numerous tributaries, particularly during mid-summer when mainstem temperature peaks. The timing of juvenile immigration into tributaries is uncertain, but most leave tributaries in early fall as flows increase and temperatures decrease (L. Ciepiela, *unpublished data*). Parr in the MFJD exhibit two main life histories characterized by winter habitat use: 1) fall-migrants leave natal rearing reaches to overwinter in larger mainstem habitats downstream and 2) spring migrants overwinter in natal rearing reaches (i.e., overlapping with summer rearing habitats). Both fall and spring migrants spend approximately one year in the stream following emergence before smoltification and downstream migration to the estuary in spring. All sampling occurred prior to expression of the fall migrant life history. The John Day basin has no history of hatchery supplementation and hatchery adult strays into the basin are rare.

In addition to Chinook Salmon, the fish community is comprised of steelhead/rainbow trout (*O. mykiss*), mountain whitefish (*Prosopium williamsoni*), northern pikeminnow (*Ptychocheilus oregonensis*), reddsideshiner (*Richardsonius balteatus*), bridgelip sucker (*Catostomus columbianus*), largescale sucker (*Catostomus macrocheilus*), speckled dace (*Rhinichthys osculus*) and sculpin (*Cottus spp.*) (Torgersen et al. 2006).

Adult sampling

Adult Chinook Salmon were sampled in September 2020 during the peak of spawning activity (9/16/2020 – 9/23/2020). Surveyors walked the entire length of the spawning distribution on 9/16/2020 and 9/21/2020, collecting tissue samples from carcasses, recording redd locations, and noting locations of live adults. Between and after these dates, a smaller group of 2-4 surveyors conducted daily surveys, prioritizing areas of concentrated live adults. When a carcass was found, surveyors measured fork length, examined the body cavity to determine sex and pre-spawn mortality, and cut a small section of the fin that was in the best

condition for genetic analysis. Fin clips were placed within a sheet of Whatman paper and inserted in a paper envelope to air dry. The tail was cut off each carcass to indicate that it had been sampled. Some carcasses were too degraded to determine length or sex, but tissue samples were still taken. Live adults were occasionally sampled when it was clear that they had spawned and could be captured by hand. When live adults were sampled, a caudal fin clip was taken and the fish were released.

Parr sampling

We sampled parr throughout the mainstem MFJD as well as in nine tributaries (Figure 1; Table 1). Prior to parr sampling, we selected 30 mainstem sites: 10 sites were part of on-going research by Oregon Department of Fish and Wildlife (ODFW) and Confederated Tribes of the Warm Springs Reservation of Oregon (CTWSRO), and 20 additional sites that were semi-randomly selected from the Columbia Basin Master sample, a set of points generated through a Generalized Random Tessellation Stratified sampling design (Stevens and Olsen 2004). We limited potential sites to the core summer rearing range (rkms 79-118) and selected 20 sites (from 28 potential sites) that maximized spatial coverage. Given the potential importance of cool- and cold-water tributary use, we sampled parr from nine tributaries within the spawning extent of the mainstem, and snorkeled or electro-fished several others but juvenile Chinook Salmon were not present.

We sampled parr from the mainstem and tributaries between 6/29/2021 and 8/19/2021. We predicted median emergence timing to occur between April 6th and May 10th of 2021, with earlier emergence upstream (Figure S8; Kaylor et al. 2022), and thus we estimate that sampling occurred 3-4 months after emergence. Tributaries were generally sampled earlier than mainstem sites (Table 1), as unusually warm conditions in summer 2021 prohibited mainstem sampling for much of July. For all mainstem sites and larger tributaries (Big Boulder Creek, Granite Boulder Creek, Vinegar Creek, and Clear Creek), we captured parr using snorkel-herding, in which one or two snorkelers herded fish into a seine net. In smaller tributaries, we captured parr with a backpack electro-shocker. At each site, we navigated to a pre-determined GPS point and then progressed upstream. Parr from individual habitat units (e.g., a single pool) were kept in separate, labeled buckets and unit-specific GPS points were taken. We stopped sampling a unit once at least 25 parr were sampled to ensure that we sampled from multiple units at each site (mean parr/unit of 17.7 and 22.0 in mainstem and tributary units, respectively). When parr were scarce or when habitat units were short (< 10 m), such as in smaller tributaries, we sampled parr from approximately 50 m reaches encompassing multiple units.

Captured parr were anesthetized, measured (fork length, nearest mm) and weighed (nearest 0.1 g). Small caudal fin clips were taken for genetic analysis and pressed onto gridded Whatman paper with uniquely labeled cells. Whatman sheets were dried out of direct sunlight and stored in paper folders until processing. Most parr exceeding 55 mm fork length were tagged with passive integrated transponder tags: 9 mm tags for 55-64 mm and 12 mm tags for parr 65 mm and longer. We allowed parr to recover in aerated buckets and then released them to the unit they were sampled from.

Genotyping

Adult and parr tissue samples were sent to the Columbia River Inter-Tribal Fish Commission's Hagerman Genetics Laboratory to be genotyped. DNA was extracted from tissue samples using the Chelex 100 method and then was sequenced with the genotyping-in-thousands method (Campbell et al. 2015). The GTseq method entails one round of PCR to amplify targeted genetic loci and another to add barcodes to identify

individuals. Then each sample was normalized and pooled into a sequencing library. The library was then sequenced on an Illumina NextSeq 500 instrument. The adult carcass and offspring samples were genotyped at 354 SNPs, or genetic markers, which include a mix of putatively neutral and adaptive markers, and a sex marker. For quality control purposes, all samples and genetic markers with 10% or more missing SNPs were considered failed genotypes and were not retained for analyses.

Parentage assignments (i.e., parr-adult pairings) were performed using CKMRsim software (Anderson 2020), in which Monte Carlo methods (i.e., Close Kin Mark Recapture; Bravington et al. 2016) were used to estimate likelihoods between each adult and parr sample. We included pairwise relationships between parr and negative adult controls, i.e., adults originating outside the John Day River Basin, to assess the false positive and false negative rates expected for pairwise relationship inference in the adult-parr dataset and compared the log likelihood ratio (LLR) distributions of MFJD parentage assignments relative to negative control assignments to determine an LLR threshold. The LLR of negative control samples ranged from -29 to -1, whereas the distribution of MFJD parent assignments exhibited a bimodal pattern, intersecting at an LLR value of approximately 12 (Figure S2). We applied a conservative LLR threshold of 20 and excluded parr-adult assignments with LLR lower than this value.

We only evaluated dispersal using parr paired to female adults. Parentage Based Tagging (PBT) assigning parr to two parents revealed that male carcasses were often several kilometers away from females that they spawned with, and that male carcass locations were downstream biased relative to paired females (Figure S3; JT Lemanski, unpublished data). We are confident that female carcass locations are more representative of redd locations, and we prioritized single-parent assignments to females. We filtered all parr-adult assignments to only those with $LLR \geq 20$, and then if individual parr assignments included both a male and a female, they were assigned to the female, even if the LLR of the parr-male assignment was higher.

Abundance estimates

We evaluated juvenile Chinook abundance and distribution across the MFJD using snorkel and electrofishing surveys (Figure 1). We snorkeled 27 mainstem sites (total length = 7.58 km) as well as four larger tributaries: 1) Big Boulder Creek (0.47 km), Granite Boulder Creek (0.92 km), Vinegar Creek (2.91 km), and Clear Creek (1.78 km). We conducted equal-effort, single-pass, electrofishing surveys in tributaries that were too shallow to snorkel including Camp Creek, Beaver Creek, Butte Creek, Dead Cow Gulch, and Davis Creek – and tallied all parr captured within each habitat unit.

Snorkel surveys were conducted at the habitat unit-level, with one or two snorkelers (depending on habitat unit width) recording all Chinook Salmon parr observations. We began snorkel surveys at mainstem sites at a predetermined point and progressed upstream until survey length exceeded 15x bankfull width (mainstem survey lengths = 174-388 m). We visually delineated habitat units as pools, fast-non-turbulent (FNT; i.e., runs), fast-turbulent (FT; i.e., riffles), and other less common habitats such as alcoves. We sampled all pools and FNTs with the rare exception of skipping units when adult salmon were observed or known to be present. We sampled every other FT habitat unit due to logistical constraints and lower counts observed in these habitats (Kaylor et al. 2021). We measured unit-specific habitat attributes that can affect detection (Staton et al. 2022) including depth (at three equidistant points along each of three transects; $n = 9$ per unit), the number of large wood pieces within the wetted channel (pieces greater than 3.0 m in length and 0.15 m in diameter), and observer-estimated water clarity (value of 0 to 3). In snorkeled tributaries, we applied the same approach, but we sampled approximately every fourth FT habitat due to logistical constraints. In Granite Boulder Creek, Vinegar Creek, and Clear Creek, we progressed upstream until no

parr were observed in three consecutive pools. While low numbers of parr may be present farther upstream, their contribution to total tributary abundance would likely be negligible. In Big Boulder Creek, surveys were concluded at a private property boundary ~500 m upstream from the confluence. Parr were still observed at this point, but likely decreased shortly upstream due to a transition to steeper gradient.

We estimated detection for each snorkeled unit using measured habitat metrics and the model developed by (Staton et al. 2022), in which paired snorkel counts and mark-recapture estimates were used to model the effect of habitat attributes on observer detection. For each unit ($n = 432$) we generated 1000 detection estimates, which were used to extrapolate counts to 1000 abundance estimates per unit, providing a distribution of abundance estimates given uncertainty in modeled detection. We estimated abundance for skipped units using mean density ($\# \text{ m}^{-1}$) from sampled units of that site and unit type, which were converted back to abundance by multiplying by unit length. We then summed across all units for each iteration to obtain 1000 abundance and density estimates per site. We separated tributaries into smaller reaches to evaluate spatial patterns of density.

For electrofished tributaries, parr sampling surveys (1-3 reaches per tributary) were also used to estimate abundance. We extrapolated the number captured in each unit to abundance estimates using ODFW electrofishing capture efficiency estimates obtained from paired single-pass and mark-recapture surveys in Camp Creek ($n = 1$), Davis Creek ($n = 1$), and Vinegar Creek ($n = 3$) between 2019 and 2021. For each unit, we simulated 1000 abundance estimates by randomly drawing from the distribution of capture efficiencies (mean = 0.26; SD = 0.083), and we generated reach-scale estimates by summing across all units within each reach.

We generated stream- and basin-wide abundance estimates by predicting abundance at unsampled locations. We created prediction sites ~300 m in length between surveyed sites and predicted parr density for each unsampled site ($\# \text{ m}^{-1}$) using linear interpolation of sampled sites. We generated 1000 density predictions for each site, which were then multiplied by reach length to obtain abundance predictions. We assumed that mainstem abundance was zero downstream of rkm 83 and upstream of rkm 117, as surveys upstream or downstream, respectively, indicated few or no parr. Lastly, we summed abundance estimates across reaches for each iteration to obtain 1000 stream-specific and whole-basin abundance estimates.

Abundance correction

Parr dispersal patterns may be influenced by sampling bias if sampled parr do not represent a random sample of the population (Wacker et al. 2021). Ideally, the number of parr sampled at randomly selected sites would be proportional to parr abundance at that site (i.e., proportionally more parr sampled at sites with higher abundance and fewer sampled at sites with low abundance). This was logistically impractical as we did not have *a priori* abundance estimates and hot conditions in July prohibited mainstem sampling, resulting in increased tributary sampling. To account for sampling bias, we used network-scale abundance estimates to simulate abundance-proportional sample sizes at each site, i.e., the theoretical number of parr we should have sampled at each site given site abundance and our overall sample size. We then generated site-specific weights as the ratio of the simulated, abundance-proportional sample size to the actual number of parr sampled.

Weights were generated using a multi-step process. First, for each of the 1000 abundance estimates for each surveyed and predicted reach, we calculated the proportion of total parr within the mainstem and nine tributaries. Second, we multiplied this proportion by the total number of parr sampled ($n = 3,389$) to obtain

abundance-proportional sample size estimates for each stream. Third, we partitioned stream-specific sample size estimates ($\hat{N}_{\text{stream}[i]}$) among sampled reaches within that stream ($\hat{N}_{\text{stream}[i]\text{reach}[j]}$). We summed abundance estimates across all sampled reaches within each stream, calculated the proportion within each reach, and multiplied reach-specific proportions by $\hat{N}_{\text{stream}[i]}$. Lastly, these estimates of the theoretical number of parr we should have sampled at each stream were divided by actual site sample size to obtain weights.

Dispersal analyses

We calculated dispersal as the stream distance between each parr-female pair. Negative values indicate downstream dispersal (i.e., parr captured downstream of females) and positive values indicate upstream dispersal. Tributary distance was negative if the tributary confluence was downstream of the female location, and positive if upstream of females. Consequently, if an individual moved downstream in the mainstem and then upstream in a tributary, the entire distance moved is presented as negative. Whereas dispersal has directionality, we also evaluate total distance moved regardless of direction as a response variable.

We first evaluated dispersal patterns using the overall distribution of all dispersal estimates including the median, inter-quartile range (IQR), and 95% quantiles. We calculated metrics using weighted quantiles to better represent a random sample of the population, in which weights were an estimate of sampling bias at each reach and were applied to all parr captured within that reach (see above). We evaluated both dispersal and total distance, stratified by all parr, parr that were captured within the mainstem, and parr captured within tributaries.

We used general linear mixed-effects models to evaluate potential relationships between 1) dispersal patterns and parr origin (i.e., rkm of paired female), and 2) dispersal patterns and other factors that may influence dispersal, such as size. In all models, dispersal or distance moved was the response variable, and the unique identifier of each female was included as a random effect. We assessed model residuals for normality and any trends in the relationship between explanatory variables and residuals (e.g., heteroscedasticity). The relationship between model residuals and river kilometer indicated heteroscedasticity for some models, and in these cases, we modeled the variance relationship as a linear relationship between rkm and the response variable using the R package ‘glmmTMB’ (Brooks et al. 2017).

Evaluating relationships between parr size and dispersal required standardization prior to analysis as we discovered that size-at-capture was confounded by factors independent of dispersal. Emergence timing is progressively earlier upstream in the MFJD (Kaylor et al. 2022), and parr size was positively associated with the river kilometer of parr origin. In addition, sampling occurred earlier in tributaries than at mainstem sites (Table 1), potentially influencing size-at-capture. To account for these complicating factors, we first estimated relative size. We fitted a set of candidate models predicting individual parr fork length, with parr origin rkm, the stream of capture, and day of sampling (day of year) as fixed-effect explanatory variables and the unique identifier of each female as a random effect. We selected the model with lowest AICc, predicted parr fork length for each individual parr, and calculated relative length for each parr as the log-ratio of measured and predicted fork length (positive values indicate parr that were larger than predicted). Parr origin rkm was fitted as a 2nd order polynomial, as this was the relationship that best described the relationship between rkm and emergence (Kaylor et al. 2022).

Results

Genotyping

We identified 161 redds across the mainstem MFJD and a single redd in Clear Creek in September 2020 (Figure 2, Figure S4). We sampled tissue from 141 individual adults and 113 of these samples – 67 females, 46 males – were successfully genotyped (<10% of SNPs missing). The distribution of redds generally mirrored the distribution of successfully genotyped females across the study extent, except between rkms 90-100 (Figure S4) where genotyped females were relatively under-represented, and surveyors noted greater scavenging by otters and eagles.

Of the 3,389 sampled parr, 1,326 (39.1%) were paired to a female adult: 595 (37.4%) from mainstem sites and 731 (40.7%) from tributaries (Table 1). At least one parr was paired to 64 of the 67 genotyped females. The number of parr attributed to each female was not uniform, and we estimated that 7, 16, and 28 females accounted for 25%, 50%, and 75% of sampled parr, respectively (Figure S5).

Abundance estimates

The total parr abundance estimated across the MFJD was 67,753 (95% CI = 63,365 – 73,750), with the mainstem accounting for 72.6% (95% CI = 69.9% - 75.3%) of parr (Table 2). Among tributaries, total abundance estimates were greatest in Vinegar Creek ($\hat{N} = 4,642$; 6.9%), Granite Boulder Creek ($\hat{N} = 4,254$; 6.3%), and Clear Creek ($\hat{N} = 3,485$; 5.1%), accounting for 18.3% of total MFJD basin abundance, whereas the other six tributaries individually accounted for less than 2% of total parr abundance.

The highest estimated densities within the mainstem occurred between rkms 91-96 and 100-106 (Figure 2A,C). Few parr were observed or predicted downstream of rkm 90 or upstream of rkm 110 (6.4% of total abundance), despite these areas accounting for 35% of the redds observed in 2020 (Figure 2A). Consequently, the distribution of redds was not well associated with mainstem parr density ($p = 0.68$; Figure 2B). In contrast, mainstem parr density was inversely related to temperature (July mean daily max temperature; Figure 2C,D). While most parr were within the mainstem, the highest estimated densities (parr m^{-1}) were in Granite Boulder Creek, and mean density was greater in six of the nine tributaries compared to the mainstem (Table 2).

Sampling bias

Sampling-bias weights suggested that we under-sampled most mainstem sites and over-sampled most tributaries (Figure S6). The mean sampling-bias weight for parr sampled from the mainstem was 1.47, indicating approximately 50% more parr should have been sampled given our total sample size. In contrast, the mean sampling-bias weight for parr captured in tributaries was 0.51, indicating that we should have sampled around half as many parr. Among mainstem sites, weights ranged from 0.28 to 4.12, with a general trend of higher weights downstream and decreasing weights moving upstream (Figure S6A). Among tributaries, Camp Creek was estimated to be under-sampled (median weight = 1.89), Clear Creek (0.87), Granite Boulder Creek (0.73), and Vinegar Creek (0.58) were slightly over-sampled, and five tributaries were estimated to have been over-sampled by 3-5 times (i.e., 0.21-0.31).

Table 2: Abundance estimates for the mainstem Middle Fork John Day River and tributaries.

Stream	Rkm	Method	\hat{N}	\hat{N} 95% CI	% of total (95% CI)	Mean density (# m ⁻¹)	Max density (# m ⁻¹)
Mainstem	-	snorkel	49,096	45,149 - 54,937	72.6 (69.9 - 75.3)	1.27	4.85
Camp Cr.	79.8	shock	1,054	727 - 2,373	1.6 (1.1 - 3.4)	0.27	0.31
Big Bldr Cr.	88.1	snorkel	1,151	1,064 - 1,258	1.7 (1.5 - 1.9)	2.38	3.34
Beaver Cr.	92.8	shock	256	181 - 479	0.4 (0.3 - 0.7)	1.03	1.03
Granite Bldr Cr.	95.1	snorkel	4,254	3,323 - 5,813	6.3 (4.9 - 8.5)	5.16	11.37
Butte Cr.	96.4	shock	1,064	848 - 1,566	1.6 (1.3 - 2.3)	0.88	2.13
Dead Cow Gulch	108.3	shock	1,110	859 - 1,694	1.6 (1.3 - 2.4)	1.69	2.17
Vinegar Cr.	110.0	snorkel	4,643	4,400 - 4,981	6.9 (6.2 - 7.5)	1.53	2.96
Davis Cr.	110.7	shock	1,246	957 - 1,954	1.8 (1.4 - 2.9)	1.63	2.81
Clear Cr.	112.8	snorkel	3,485	3,052 - 4,025	5.1 (4.4 - 6.0)	2.11	4.12
Total	-	-	67,753*	63,365* - 73,750*	-	-	-

*Total abundance and confidence intervals differ from the sum of stream abundances, as total abundance was estimated from 1000 samples in which each sample was the sum of abundance estimates across all reaches.

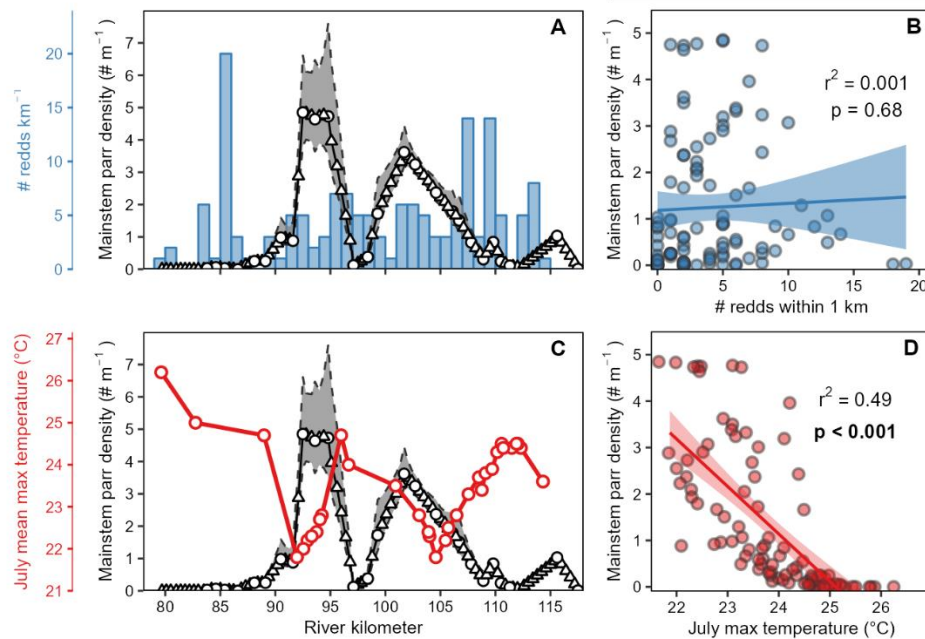


Figure 2: Mainstem spatial patterns of 2020 redds (A, blue bars), 2021 parr density (A,C; black/grey lines, points, and shading), July 2021 max temperature (C, red points and lines), and relationships between redds and density (B) and max temperature and density (D). For density estimates, points represent snorkeled sites, triangles indicated prediction reaches, and grey shading between dashed lines show 95% confidence intervals.

Overall dispersal

The distribution of all parr dispersal estimates was downstream-biased (median = -0.77 km) with 68% of all parr estimated to have dispersed downstream (Figure 3A). However, 25% of parr estimated to have dispersed more than 3.69 km downstream and 25% dispersed more than 0.92 km upstream. Dispersal patterns differed for parr captured within the mainstem vs tributaries (Figure 3B,C), with downstream-bias for mainstem-captured parr (median = -1.43 km; 78% dispersed downstream) but upstream bias for

tributary-captured parr (median = 0.67 km; 57% of dispersed upstream). The median estimated distance parr moved regardless of dispersal direction was 2.19 km with approximately 25% of parr that moved greater than 5.0 km (Figure 3D-F). Parr that dispersed downstream generally moved greater distances (Inter-quartile range (IQR): 0.73 – 6.03 km; max = 28.60 km) than parr that moved upstream 1.06 – 3.54 km; max = 10.61 km).

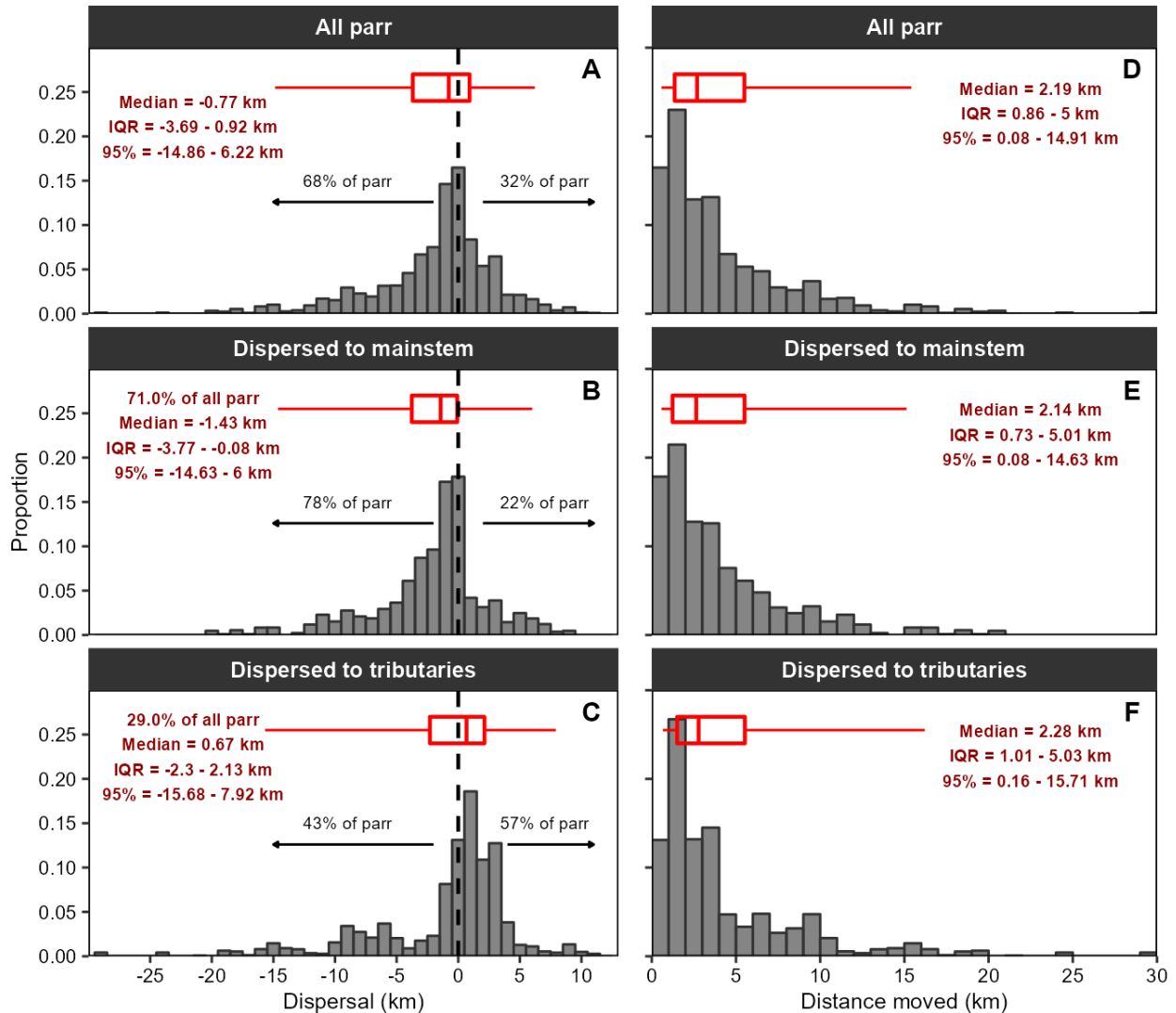


Figure 3: Overall distributions of sample-bias corrected dispersal (A-C) and total distance moved estimates (D-F). Box and whisker plots indicate median, inter-quartile range (IQR), and 95th percentiles.

Spatial patterns of dispersal

Parr dispersal patterns varied as a function of where they originated (i.e., redd rkm) and dispersed to (i.e., mainstem versus tributaries; Figure 4). For parr that dispersed to mainstem locations (Figure 4A), individuals originating low in the watershed exhibited upstream dispersal bias and relatively low variability in dispersal estimates. Dispersal progressively transitioned towards downstream bias higher in the watershed, which was accompanied by increasing variability in dispersal direction and distances. In

contrast, there was little apparent trend between parr origin and dispersal bias or distance for parr that dispersed to tributaries (Figure 4B).

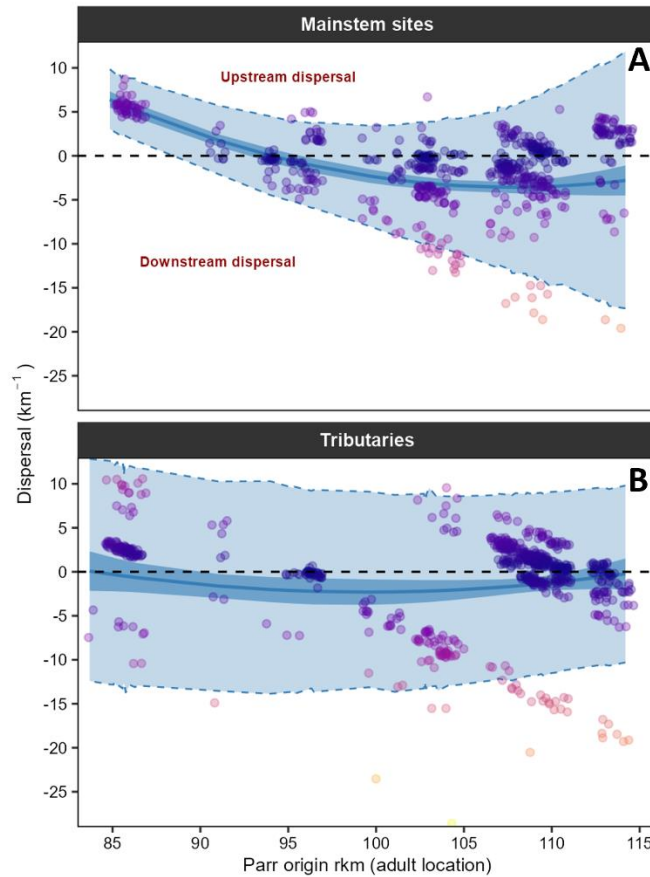


Figure 4: Parr origin (i.e., female adult location) versus dispersal separated by parr captured within mainstem sites (A) and parr captured within tributaries (B). The solid line indicates the fitted relationship between parr origin and dispersal; dark shading indicates the confidence interval of the fitted relationship; and the light shading and dashed lines indicate the prediction interval, encompassing 95% of dispersal estimates. The color of points reflects the range in dispersal estimates, with warmer colors indicating more downstream bias.

Dispersal patterns generally followed a trend of dispersal from warmer sections of the mainstem to slightly cooler sections or tributaries (Figure 5; Figure 6A). For example, parr originating from rkms 84-89, where July temperatures were among the highest, either dispersed upstream to mainstem habitats between rkms 91-97 or to one of four tributaries between rkms 79.8-96.4 (Figure 5) where the temperature at parr capture locations averaged ~ 3 °C cooler than their origin locations (Figure 6A). This is further exemplified by the negative relationship between individual parr origin temperature vs the difference between capture and origin temperature (Figure 6B) – there was little difference between capture and origin temperature for parr originating from areas < 23 °C, but the average parr capture temperature was nearly 2.5 °C lower than origin temperatures for parr originating from the warmest temperatures.

Insight on where juveniles originated from for a given section can also be gleaned from Figure 5. For example, the section between rkms 91-97 supported parr originating from nearly all parts of the watershed, whereas sections upstream only supported parr originating from nearby. Similarly, some tributaries such as

Granite Boulder (rkm 95.1) supported parr originating across a wide spatial extent, whereas upstream tributaries supported parr from within several kilometers.

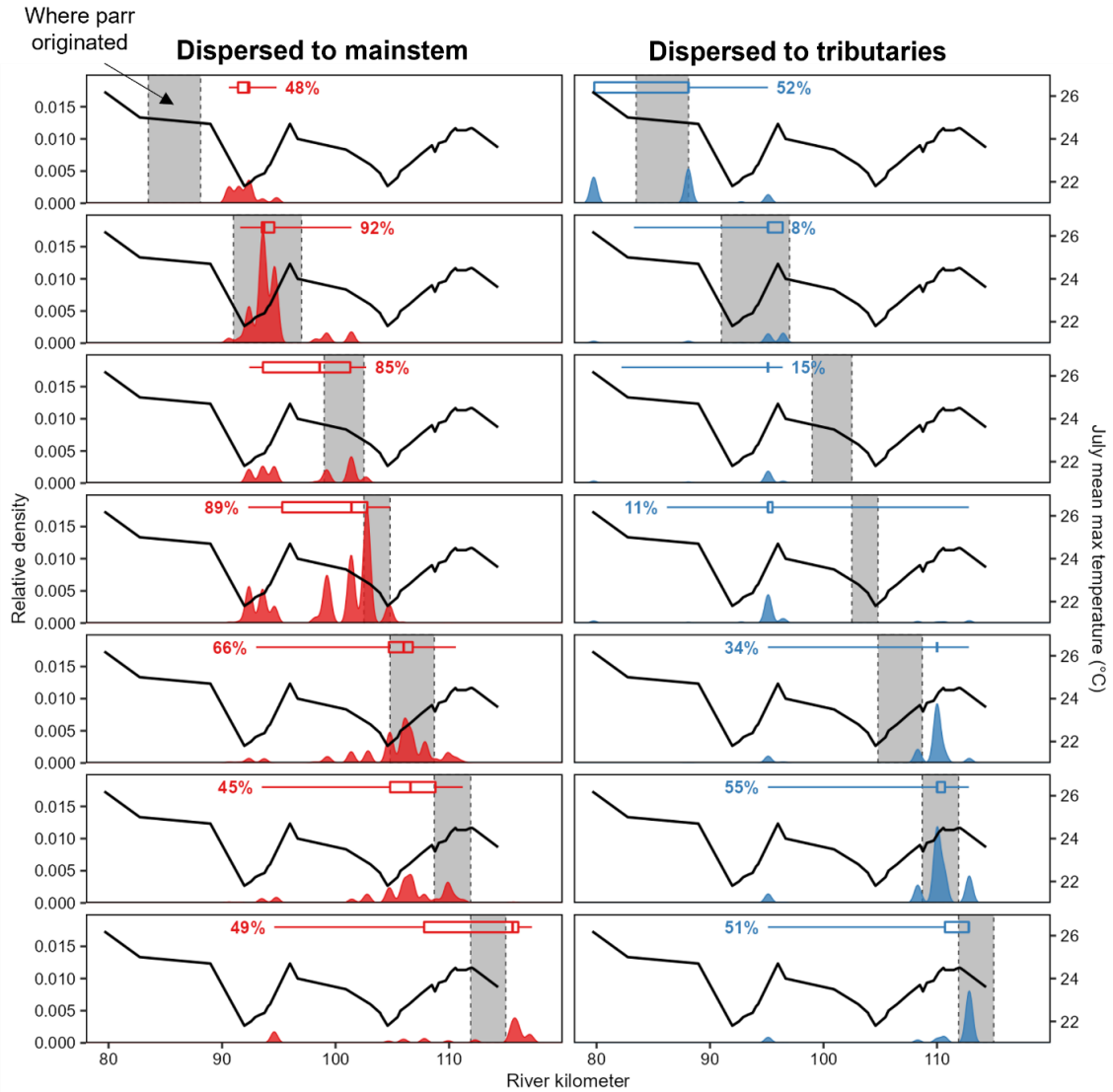


Figure 5: Parr dispersal patterns from different sections of origin (rows). Grey boxes indicate the section parr originated from; density distributions portray where parr from each section dispersed to across the mainstem (red distributions) and to tributaries (blue distributions); box and whisker plots indicate median, inter-quartile range (IQR), and 95th percentiles of parr distributions for each section; percentages indicate the estimated percent of parr from that section that dispersed to mainstem locations (red) or into tributaries (blue); and the solid black line shows mean July maximum August temperature.

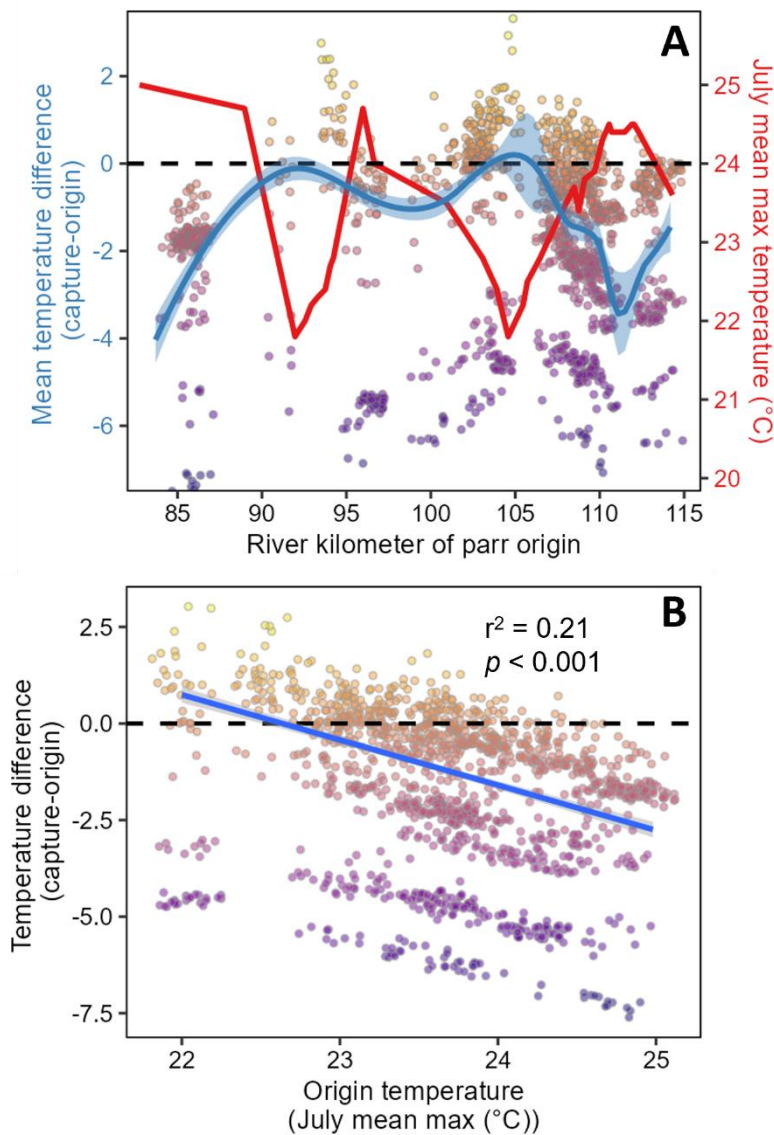


Figure 6: Spatial patterns of temperature across the mainstem MFJD in summer 2021 (red line; A), differences in estimated origin and capture temperatures (points; A), and the relationship between parr origin temperature and the difference in temperature between capture and origin locations (B). The blue line in the top panel indicates the fitted loess relationship between river kilometer and the mean difference in capture and origin temperatures. Point colors represent the gradient in temperature differences of individual parr from the maximum difference (i.e., movement to warmer rearing habitats) to the minimum difference.

Size vs. dispersal

Independent of dispersal, parr size was spatially structured and further depended on sampling date. The highest ranked model predicting parr length-at-capture across the MFJD included the river kilometer of parr origin (i.e., female location) as a second-order polynomial term, the stream of capture (e.g., mainstem or one of the nine tributaries), and day of the year sampling occurred. This model explained 31% of the variation in parr length and was used to calculate relative length as the difference between measured length and predicted length (positive numbers indicate a parr was larger than predicted) after accounting for these spatial and temporal effects on parr length. The highest ranked model predicting dispersal distance included

relative parr length, dispersal direction (upstream or downstream), and the interaction between relative parr length and dispersal direction, explaining approximately 26% of the variation in the distance parr moved. Dispersal distance was predicted to increase with greater parr relative length for parr that dispersed downstream but not for parr that dispersed upstream (Figure 7).

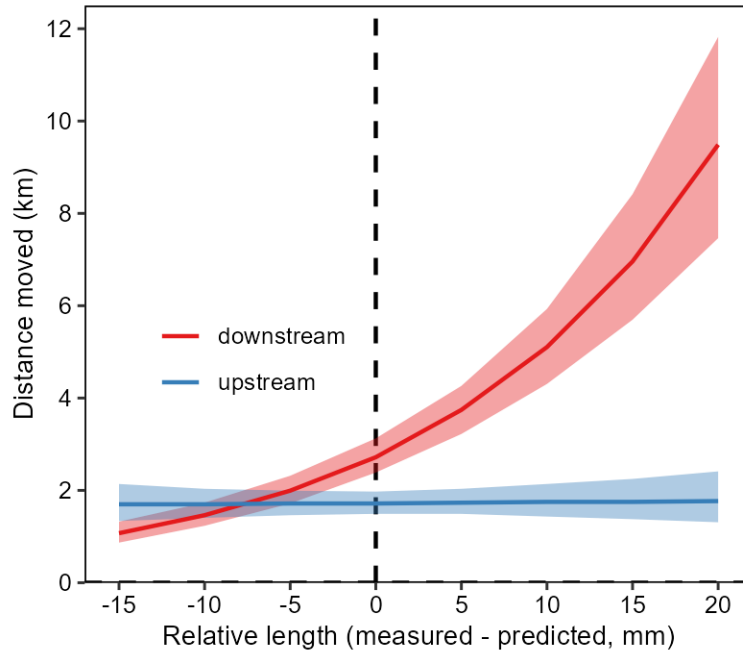


Figure 7: Fitted relationship between parr relative length (measured length – predicted length after accounting for parr origin river kilometer, stream of capture, and date of sampling) and the distance parr dispersed upstream (blue line) or downstream (red line).

Discussion

Overall findings

Understanding how fish habitat use and spatial distribution relates to the physical and biological variability in potential habitats is fundamental in guiding conservation and restoration approaches. Habitat use within river networks depends not just on habitat quality but also accessibility and ecological considerations including the proximity of habitats to spawning locations and dispersal patterns of juveniles from these locations. Through a riverscape evaluation of dispersal patterns in a wild population of Chinook Salmon, our results demonstrate widespread dispersal upstream, downstream, and into tributaries, suggesting dispersal may be more extensive in wild populations than previously thought (Rodriguez 2002; Eisenhauer et al. 2021). Further, our results demonstrate that dispersal patterns were not consistent throughout the basin, but rather dependent on spawning location and the subsequent environmental conditions juvenile salmon experience. In particular, we found that temperature was an important environmental condition driving dispersal patterns. This study provides an approach building upon previous efforts (Hudy et al. 2010; Anderson et al. 2013) to effectively evaluate riverscape patterns and drivers of dispersal from spawning locations to rearing habitats for naturally spawning populations.

Restricted Movement Paradigm

The Restricted Movement Paradigm (RMP) predicts that most individuals in a population are sedentary, remaining within short stream reaches near their point of origin (Gerking 1959). The RMP has been challenged and expanded upon, suggesting that many methodological approaches may incompletely sample individual movement (Gowan et al. 1994), that populations may be composed of mobile and stationary groups (“movers and stayers”) with distinct movement distributions (Rodriguez 2002), and that juvenile salmon dispersal may be more extensive and less downstream-biased than previously thought (Eisenhauer et al. 2021). By sampling across the spawning and rearing extent of a wild population of juvenile salmon, our results do not conform with the RMP or the concept of movers and stayers, but instead suggest greater overall mobility. In a review of published estimates of juvenile Atlantic salmon dispersal, Eisenhauer et al. (2021) found that dispersal estimates was generally downstream-biased and that nearly all individuals dispersed less than 500 m; however, the authors also presented original empirical estimates from 19 tributaries that demonstrated 1) a greater proportion of individuals that dispersed upstream (mean = 34%), 2) wider dispersal distributions, and 3) greater maximum dispersal distances, both downstream (mean = 2.13 km; max = 4.79 km) and upstream (mean = 1.23 km; max = 2.84 km). Our results similarly suggest that approximately a third of individuals dispersed upstream (32%), but that dispersal was even more widespread (95% range = 14.86 km downstream – 6.22 km upstream; max downstream = 28.1 km downstream; max upstream = 10.61 km). Further, there was no evidence of separate groups of mobile and stationary individuals – characterized by bimodal distribution (Rodriguez 2002). The expression of mobile and stationary groups may occur in other populations, species, and life-stages due to differences in genetic predisposition, environmental conditions, or biological factors such as variability in intra-specific competition. Alternatively, incomplete or biased sampling could generate bimodal distributions not characteristic of a random sample of the population.

Dispersal relative to studies of wild populations

Most studies evaluating juvenile salmon dispersal have utilized experimental approaches such as out-planting eggs or fry (reviewed by Eisenhauer et al. 2021), in which dispersal is evaluated under a narrow range of conditions. Reducing variability is often a necessity to test specific hypotheses and these studies have greatly informed our understanding of factors influencing dispersal (Einum and Nislow 2005; Einum et al. 2006; Brunsdon et al. 2017). However, few studies have evaluated dispersal in naturally spawning populations, including with respect to environmental and biological factors that may limit or promote it. Yet these studies suggest high variation in dispersal patterns characteristic of responses to variability in biophysical conditions and behavioral selection specific to localized watershed characteristics. For example, previous research has demonstrated extensive dispersal patterns of recently emerged salmon fry (10s or 100s of kilometers) associated with alternative early life-history strategies (Bradford and Taylor 1997; Daum and Flannery 2011; Schroeder et al. 2016; Scheu 2022). Understanding these patterns and the mechanisms that drive them have important implications for prioritizing the types and locations of management efforts to maximize habitat use and benefit to juvenile salmonids.

Greater variability upstream

Parr originating higher in the watershed exhibited greater variability in dispersal than parr originating lower in the basin, which may be attributed to several potential factors. First, emergence phenology within MFJD is spatially structured (Kaylor et al. 2022) with fry originating upstream estimated to have emerged approximately five weeks earlier than fry from the farthest downstream extent in 2021 (Figure S8),

potentially exposing them to higher flows. While higher flows experienced during and after emergence are likely to increase passive dispersal, flows were relatively consistent across the range of estimated emergence dates in the spring of 2021 and thus not likely to explain these patterns. Alternatively, emergence may have contributed to greater dispersal variability upstream through effects of ontogeny on swimming capacity. Larger individuals often disperse farther than smaller conspecifics (Anderson et al. 2013; Aparicio et al. 2018) and variability in dispersal may increase in later life stages (Yamamoto et al. 2021) as density-dependence exerts greater influence on dispersal (Einum et al. 2006). The longer duration since emergence and larger size of parr upstream suggests that differences in ontogeny could have been a contributing factor to greater dispersal variability. Lastly, these patterns could be attributed to the directional flow of river networks interacting with environmental conditions that contracted the juvenile rearing distribution. High summer temperatures were clearly a factor influencing dispersal and ultimately parr distribution. However, parr originating downstream where temperatures became unsuitable needed to move upstream against the current to find cooler habitats. In contrast, parr from upstream locations could move downstream with the flow to cooler sections of the mainstem or tributaries, thereby incurring lower energetic costs of movement.

Size vs dispersal

There was a positive relationship between parr size and downstream dispersal distance, but we did not find evidence for upstream dispersal associated with size. Numerous studies have found relationships between dispersal and body size, with the majority finding larger size associated with increasing dispersal distance (Close and Anderson 1992; Anderson et al. 2013; Aparicio et al. 2018), but some finding smaller individuals dispersing farther (Webb et al. 2001). These opposing patterns could stem from differences in swimming ability and in response to intra-specific competition. Positive relationships may arise if juveniles that disperse farther from nests to areas of lower density yield greater growth benefits (Brunsdon et al. 2017) or if larger individuals disperse farther due to greater swimming capacity. Negative relationships could result from smaller individuals passively dispersing farther downstream due to poorer swimming capacity and lack of ability to evade high flows (Saltveit et al. 1995), or if larger individuals establish competitive advantages due to prior residency or dominant feeding positions (O'Connor 2000; Einum and Fleming 2000; Harwood et al. 2003), forcing smaller individuals to disperse. A plausible explanation is that there was a competitive advantage for individuals dispersing downstream but not upstream due to earlier fry emergence with distance upstream (Kaylor et al. 2022). Earlier emergence is associated with larger size (Kaylor et al. 2021), and fry that disperse downstream – where conspecifics emerge later – may have a competitive size advantage or be better suited to establish dominant feeding positions through prior residency (Einum and Fleming 2000). In contrast, if fry disperse upstream, it would be more challenging to establish a competitive advantage given the lack of a size advantage and since habitat occupancy and feeding positions are more likely to be established. However, given the observational nature of this study, we cannot confidently attribute a causal mechanism to patterns observed in our system and it is likely that multiple factors interacted to shape the relationship between greater downstream dispersal distance and larger size.

Sampling bias corrections

Correcting for sampling bias had considerable effects on interpretation of dispersal patterns in the MFJD (Figure S7). For example, using uncorrected dispersal estimates there was little upstream or downstream directional bias in overall dispersal patterns (median = -0.03 km; 50% dispersed in each direction), but when sampling-bias corrections were applied, the overall dispersal distribution was clearly downstream-

biased (median = -0.77 km; 68% dispersed downstream). This highlights the importance of trying to obtain a random sample of the population (Wacker et al. 2021) when quantifying dispersal. While it is ideal to minimize sampling bias through careful study design, if possible, we believe our approach effectively reduced sampling bias and improved characterization of population-level dispersal. Our sampling design was not well suited to characterize fine-scale dispersal from single females as the spatial extent was large and necessitated large gaps between sampled sites, but the approach could be modified to smaller spatial scales with shorter and more frequent sampling locations. There were benefits to estimating parr distribution across the watershed beyond as a tool to apply sampling bias adjustments, but it also required a considerable amount of additional effort. However, if characterizing parr distribution is not a study objective, an alternative approach with less logistical challenges would be to conduct equal-effort sampling and genotyping of all (or a consistent proportion) juveniles captured at randomly selected habitats across the rearing extent.

Caveats

There are several caveats associated with our sampling design and the unusually warm conditions of summer 2021. First, we did not sample parr or conduct abundance surveys in the mainstem or tributaries downstream of Camp Creek, and consequently, our results may not reflect the full extent of dispersal and distribution present within this population. In an adjacent sub-basin of the John Day River (the Upper Mainstem), Chinook fry dispersed over 70 km downstream of spawning reaches including 10s of kilometers upstream into cooler tributaries (Scheu 2022), which is consistent with large-scale movements of downstream rearing parr life histories in other basins (Daum and Flannery 2011; Schroeder et al. 2016). However, far fewer age-0 juveniles from the MFJD are captured in a downstream screw trap, and downstream rearing is not thought to be a common life history (Ian Tattam; unpublished data and personal communication). Consequently, we assume that parr dispersal outside of our study area likely had minimal effects on overall dispersal patterns at the population-level. Second, it is important to note that our dispersal estimates only represent individuals that survived to summer and that our approach defines dispersal based on two points in time. Sampling earlier in the year may have revealed different patterns, such as greater downstream bias associated with passive dispersal of recently emerged fry exposed to high flows (Saltveit et al. 1995). It is likely that some individuals passively dispersed downstream and later actively dispersed upstream (Yamamoto et al. 2021), but our sampling approach would not detect these patterns. Lastly, the early summer of 2021 was characterized by abnormally high air and water temperatures and low discharge. These conditions are not representative of typical conditions within the MFJD, and the dispersal patterns we observed, especially the effect of summer temperature on dispersal and parr distribution, likely differ considerably in cooler years with greater summer baseflow. On the other hand, the conditions of 2021 do represent future conditions anticipated under climate change – high temperatures, earlier onset of baseflows, lower baseflows – and results from this study may provide important insight into habitat attributes and locations that may become increasingly common.

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Appendix B: Supplemental material

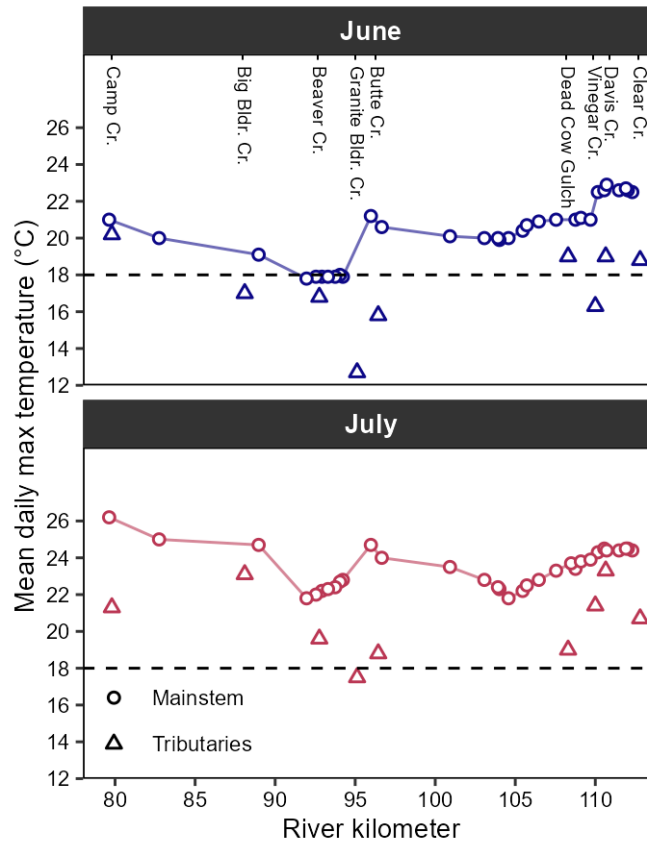


Figure S1: Mean daily max temperature for June (top) and July (bottom). Points indicate mainstem locations and triangles indicate sensors within tributaries.

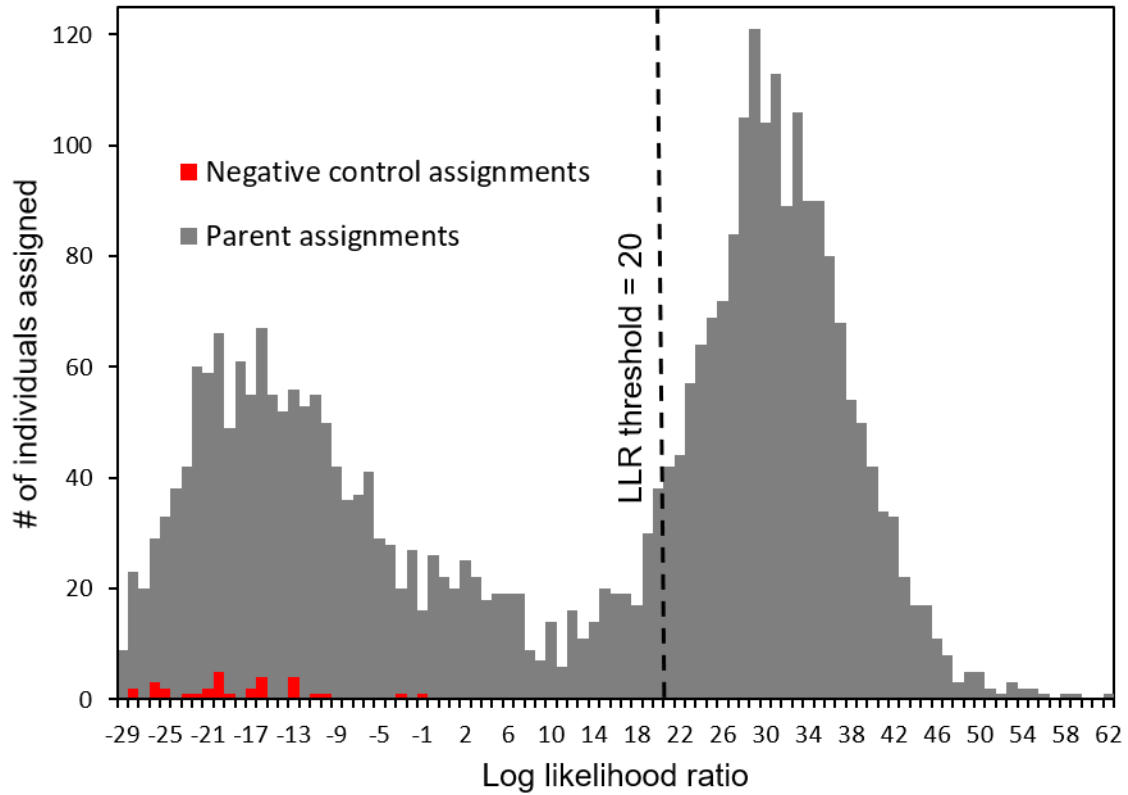


Figure S2: Distribution of log likelihood ratios for all MFJDR parr-adult assignments (grey bars) and for negative control assignments (red bars). The bimodal distribution indicates two groups of parr-adult pairings, with the right distribution representing parr-adult assignments in which the adult was sampled and correctly assigned.

Distance between female-male pairs

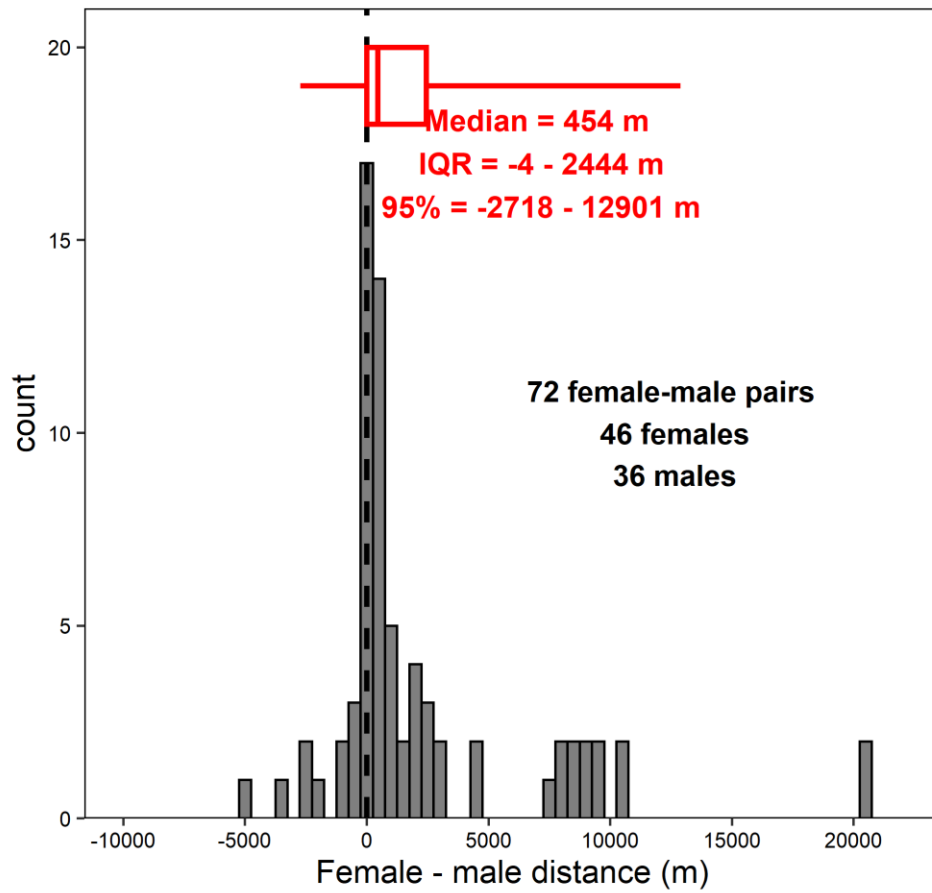


Figure S3: Distance between carcass locations of females and males that spawned with each other (72 unique female-male pairs). Positive values indicate that the female location was upstream of the male location in a male-female pair, whereas negative values indicate that the female location was downstream of the male. The distribution demonstrates 1) that males were often sampled far from females they spawned with and 2) downstream bias of males relative to females.

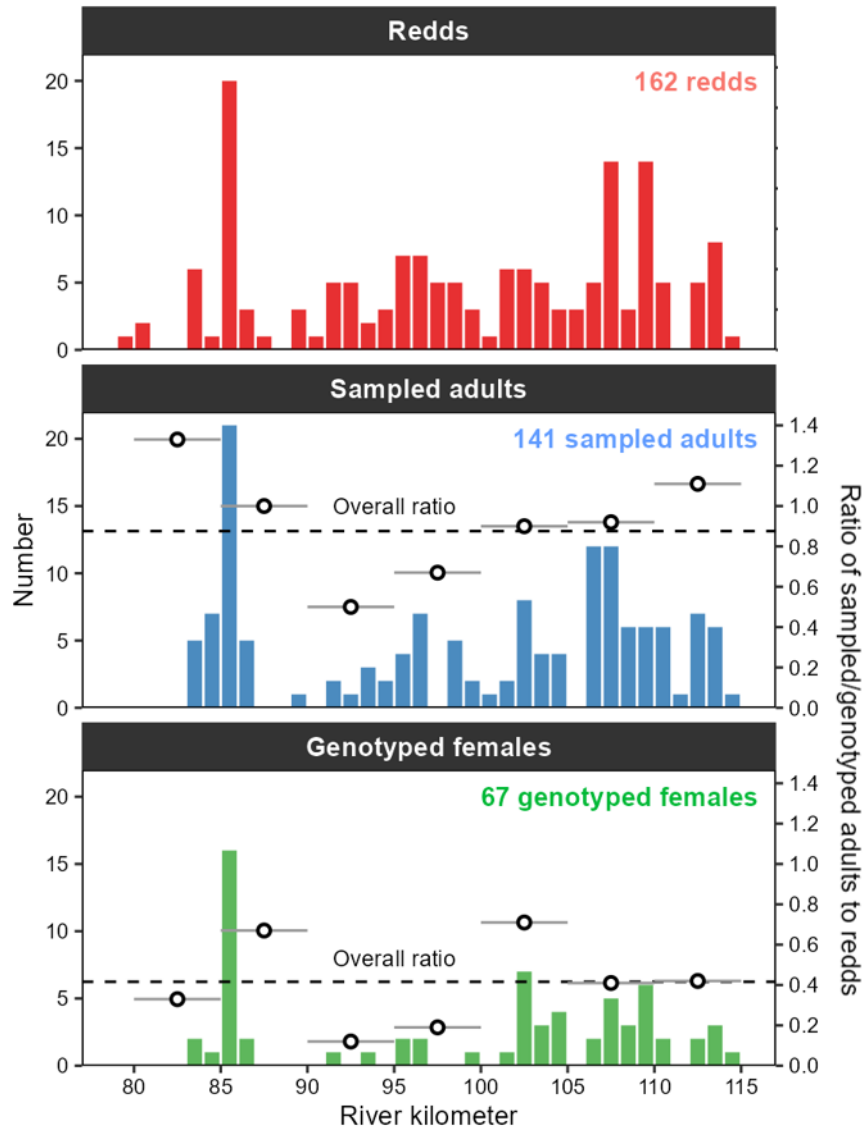


Figure S4: Number of redds (top; red bars), sampled adults (middle; blue bars), and successfully genotyped females (bottom; green bars) per kilometer in spawn year 2020. Points represent the ratio of sampled adults or genotyped females to redds for 5 km groupings (indicated by horizontal line associated with each point). Dashed lines indicate the overall ratio of sampled adults or genotyped females to redds. Ratios of genotyped females to redds (bottom panel) demonstrate sections where females were under-represented relative to redds (e.g., rkms 90-100).

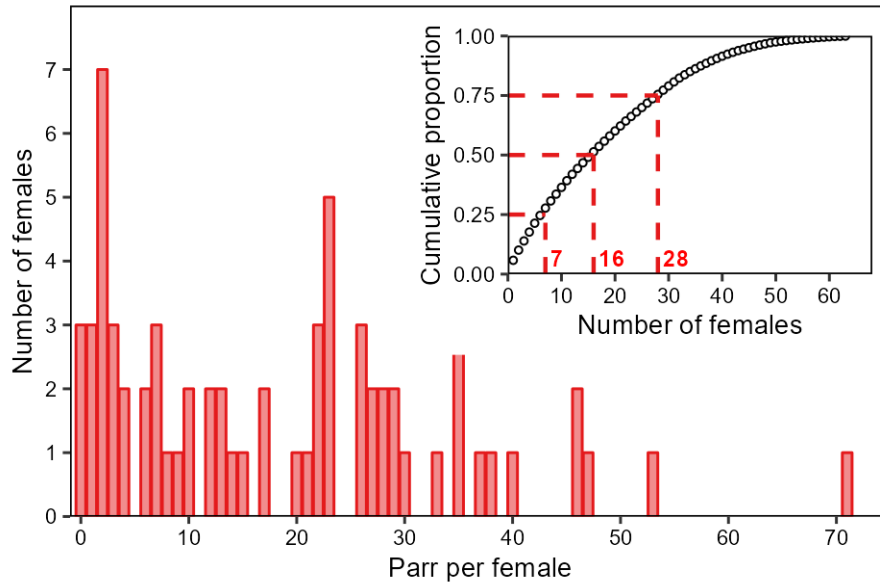


Figure S5: Distribution of number of parr paired to females after applying sampling bias correction. The inset shows the cumulative proportion of parr paired to females as the number of females – ranked by number of parr per female – increases. Out of the 67 females, 64 had ≥ 1 paired parr. The distribution shows non-uniform contributions of females to all sampled parr: the top 7, 16, and 28 females accounted for 25%, 50%, and 75% of all parr, respectively.

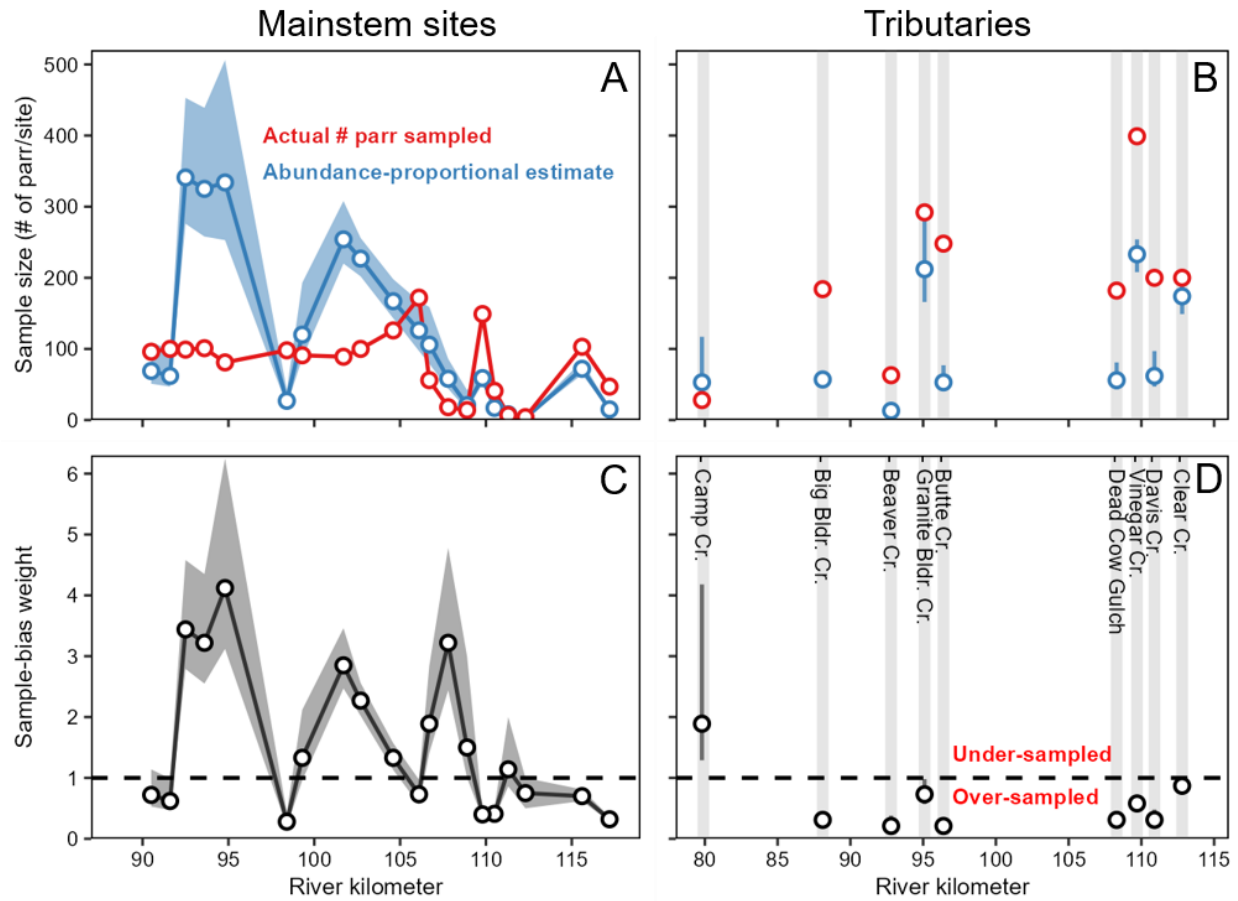


Figure S6: Number of parr sampled at each mainstem site or tributary (red points and lines; A,B), estimated abundance-proportional median number of parr that should have been sampled at each site (blue points, lines, and shading; A,B), and sample-bias weighting factors for each mainstem site or tributary (black lines, points, and shading; C,D). All shading and error bars represent 95% confidence intervals. Sample-bias weights greater than 1 indicate under-sampling (i.e., we should have sampled more parr), whereas values less than 1 indicate over-sampling. In general, most mainstem sites were under-sampled and most tributaries were over-sampled.

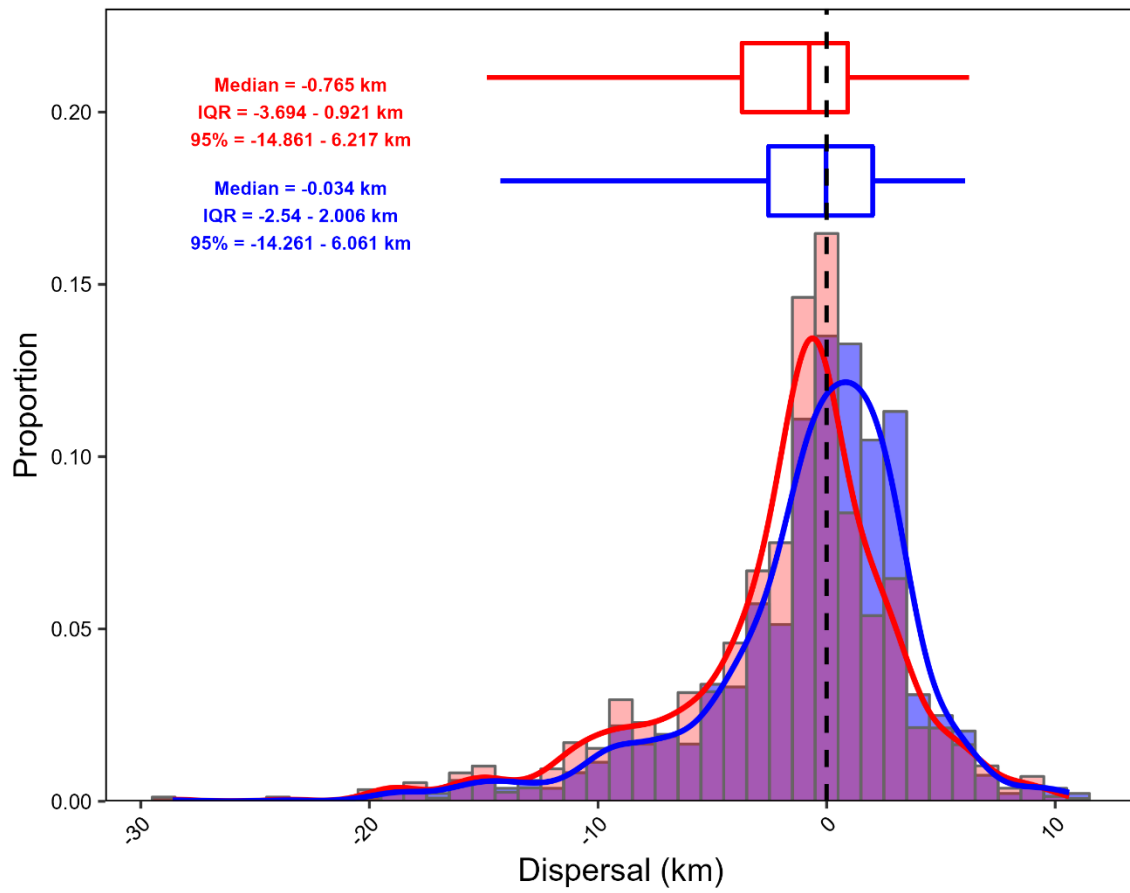


Figure S7: Distribution of dispersal estimates for all parr paired to females using raw values unadjusted for sampling-bias (blue bars and lines) and simulated, sampling-bias-adjusted estimates (red bars and lines). Box and whisker plots indicate median, inter-quartile range (IQR), and 95th percentiles.

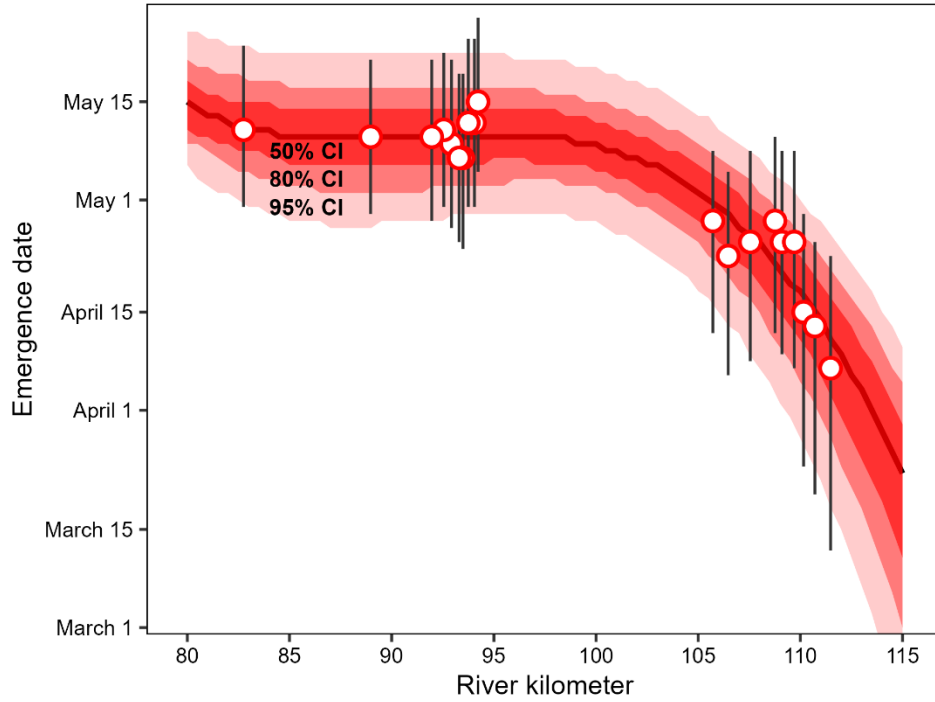


Figure S8: Emergence timing estimates across the MFJD for 2021. Emergence was first estimated at 19 locations with annual temperature data through simulating emergence given variation in spawn timing (see Kaylor et al. 2022). Points indicate median emergence timing and error indicates quantiles encompassing 95% of estimates. Emergence timing was then modeled as a function of river kilometer fit as a second-order polynomial. Red bands indicate 50%, 80%, and 95% ranges of predictions.

Appendix C: Grande Ronde State of the Science Adaptive Management Meeting Notes

November 18, 2022

Review of presentations/important takeaways:

- 1.) *Status and Trends* –
 - a. Lack of steelhead monitoring. It appears there are declines in steelhead productivity. Joe – not great that we are seeing declines, but it appears our population is doing better than other Lower Snake basins. Why is our population performing better?
 - b. Fred – potential for gathering lots more data with PIT tags and arrays. How are steelhead using tributaries? Where are they rearing vs. emerging?
 - i. Les – suggesting screw trap at lower end of Meadow Creek associated with CALR. Will be PIT tagging adult steelhead at weir to utilize new arrays to track adults rather than radio tags.
 - ii. 10,000 additional PIT tags funded by BOR for both steelhead and Chinook. See PIT Array Story Map <https://arcg.is/bzqP40>. Fred – it's going to take a few years of data collection to determine how we improve conditions for steelhead. Fred reviewed proposed locations. **Adapting monitoring strategy to include more PIT tags (specifically for steelhead).** Winston supports more focus on steelhead populations. Sean – program resources are tied to mitigation responsibilities, reaches where we can improve conditions for both steelhead/Chinook are great.
 - c. Has carrying capacity improved? How do we measure the change? When we first developed Atlas, increasing carrying capacity was a major objective for summer parr. Tom Cooney had identified this as an area of concern in the LCM. Jesse - Have we made progress over the last 6 years?
 - i. Sean – We have fish in, fish out. Phil – how much do you have to increase capacity before you start to see a response in the population? Typically, 25% increase in capacity before you start to see a response. Projects have increased habitat carrying capacity but maybe not to the extent where we would see a change in population carrying capacity.
 - ii. Sarah – other species utilize the new habitat; how does that affect carrying capacity of that habitat unit? Norm – complicated relationship dynamics between different species.
 - iii. Ian – seconding idea that we haven't reached threshold of enough work to see population response. Is growth a surrogate for habitat carrying capacity? Pre-spawn mortality will affect your smolts per returning adult (Polly clarified that it is smolts/spawner. Matt – spread spawners out to reduce density dependence; improve spawning habitat downstream of Vey. Joe/Matt – 50% of juveniles stay within a couple kilometers of where they emerged so spreading spawning out will result in spreading out rearing parr.
 - iv. Allen – we haven't really fully restored that much habitat upstream of Union, much of the work was focused on improving conditions in a

confined channel, probably not improving capacity much. Polly, we could look at survival of fish that reared specifically in a restored project.

- v. Joe – where and why are we losing fish?
 - 1. Sean – brought up the idea of a pilot project to implement and study the effects in the valley.

2.) *Adult Chinook Movement* –

- a. Winston - difficulty of ranking fish passage projects with other restoration projects. Atlas ranking doesn't work well for passage projects. Shift this study to upper Wallowa River next to help inform. Partial barrier ranking based on delay or failed attempts?
- b. **Importance of holding habitat near diversions.** This can be boulders or wood that create small pocket pools.
- c. Clear evidence of restoration improvement. Spawning in ditch entrances, may get dewatered. Ian – evidence of lack of side channel habitat?
- d. Natural barrier above Pole bridge. NPT no longer trucking fish around. End of distribution of some species (WF, Sculpin), we should not be modifying it – Kyle.
 - i. Casey – barrier at upstream extent of UGR, spawning habitat limited. Is anyone thinking about addressing? Matt - Are they truly natural, may have been more passable under historical conditions (flows). What is the habitat capacity upstream?
 - ii. Les - Indian Creek – is there an opp. lead for barriers? Three ESA species, should be a priority. Levi looked into barriers, not sure if he is pursuing any of them on private land. USFS working on culverts on public land with GRMW. There are Chinook returning and spawning but unsure of the population affect. Les - in the process of determining reproductive success, some limited genetic samples but have not be analyzed. Side note – looking for any remaining LKG genetics using endemic samples (Indian Creek is one of those possible sites.).

3.) *Stage 0* –

- a. Jesse – the technique of restoring to a stage 0 condition may be important in terms of Matt Kaylor/Joe Lemanski's work on emergence timing and fry/parr being able to access floodplain habitats. The sediment study provides important information as well. When thinking about maintaining an anastomosing channel condition we should be asking, is there sufficient sediment supply to maintain that condition?
- b. **Value of test pits. Only as valuable as the number of sites you test and whether you happen to hit the right locations. Important consideration in construction contract language.**
- c. **Several lessons learned – document these related to implementation.**
- d. Ground penetrating radar – used on BTS and Longley to look for confining layer, not defining particle size. Sean it's pretty standard geotechnical work to look at material type and historic channels.
- e. Les – use groundwater well sites to help with determining sediment in conjunction with test pits.

- f. **Leave pools, vegetated islands.** Sean - use Lidar and look at all returns above 1.5m and start looking at preserving these areas, fill channel in very targeted areas. Leave zones of high redd density.
- g. Bryan Endress – disseminate that data again. **Bryan noted a large percentage of vegetation growing above browse height is natural and not the planted stock.**
- h. Climate (eastern side of state) and grazing are big factors, reasons to preserve riparian veg. Fence maintenance takes an army.
- i. The sediment inputs from hillsides vs. filling the channel. Use the sediment study to inform were Stage 0 may be most appropriate and most successful.
- j. Keeping floodplains connected at lower flows so late emerging fry can utilize.

4.) *Water Quality Assessment/Valley Survival* –

- a. Fred – DO was below standards, would it be helpful to install sensors to monitor a stretch on Catherine Creek, starting at Ladd and going downstream? YES! DO should not be an issue in winter.
 - i. Allen – look at turbidity as well since it is linked to DO.
 - ii. Matt - Also, it may be worse at night due to primary productions.
Sensors are a \$1000, we need 24 hours sampling all year long.
- b. **We need to document possible causes of mortality (Favrot/Jonasson white paper) – Lower Valley.** Winston – can BOR help with sensors? Kira – is this a priority (Les). GRMW get a cost estimate from AP on adding sensors (DO, temp, turbidity)?
- c. Copper – spiking at rain events? More sampling to determine if it is high in sediments. Soil sampling.
 - i. Bio accumulation? Look at macroinverts? Other species? Ask Matt/Kara about bio accumulation, what species would be good to look at for copper?
 - ii. Winston - Copper at Hilgard – ODOT monitors Hilgard area, maybe Ladd Creek too? Allen said they monitor chain off areas.

5.) *Sediment Study* –

- a. **Active vs. passive** – If it’s going to take decades for sediment to accumulate, better off to actively add it rather than passively try to accumulate with LWD.
- b. **Good data to inform Atlas updates.**
- c. Sarah – so much of the USFS land management decisions are based on reducing sediment and keeping hillsides stable.
- d. **Importance of local sediment supply and even designing ways to produce sediment inputs. Erosion can be a good thing.**
- e. **Duplicate in Wallowa/Lostine, Catherine Creek (existing 1D model)?** Review sediment report and decide how to proceed from there.
- f. Would have been nice to have this study earlier. Third entry in Sheep Creek, and we are now adding the sediment.
- g. MUGR? – did not have time to discuss at this meeting.

6.) *LCM* –

- a. Casey gave a quick update since Ben was unable to present.

- i. getting close to running restoration scenarios. Should be really helpful with Atlas. Population level model, may not inform reach scale changes. UGR, CC, MR, LR included.
 - b. Schedule Ben to present to the IT
 - c. Winston – can the LCM inform improvement from past actions? Phil – my experience is that LCM’s may not provide that detail.
- 7.) *Floodplain Restoration Effectiveness*
 - a. **LiDAR** - coverage area and frequency. GRMW hopes to collect LiDAR every ~10 years.
 - b. Coordinate with other basins when trying to cover smaller areas.
 - c. We could collect more frequently on a project scale with a drone. Phil – if it is older than 3-4 years it may not be very helpful for project design. This would be the time to coordinate with other folks to make it worth getting a plane in the air.
 - d. Let’s make sure we get what we want next time (in regards to the LWD classification).
 - i. Did not get good bathymetric data in the valley presumably due to turbidity and depth.
 - ii. Current LiDAR classified LWD as unclassified.
 - iii. High wood density can interfere with bathymetric data.
 - iv. Make sure we have enough GCPs. Especially bathymetric points in deeper water - Lessons learned from Entiat. Don’t think we need a full profile, just individual points.
 - v. Request aerial imagery with LiDAR. CRITFC was wishing we would have had that done. Phil – the camera is on the plane, just need them to turn it on and they will charge maybe ~10% more.
 - vi. Casey – habitat classification using geomorphic unit tool (GUT), had company due a few small areas. Seth did a comparison, looked decent. Sean – but without the wood that is affecting units how can it be accurate. Casey – part of the reason we didn’t pursue it further.
 - vii. Need to follow-up on wood classification – Casey
 - e. Get as built designs, match design objectives with final design to check for successfulness.
 - i. Objectives need to be quantified or else you can’t determine success. Be more consistent. Consistent framework – online proposal? GRMW look at incorporating the metrics from Phil’s presentation into the online proposal objective builder. Have Phil review the objective section of the proposal.
- 8.) *Atlas update process* - Present on Atlas matrices, how it was developed. Ian present on Wallowa, Jesse on UGR/CC. Think about converting to Wallowa scoring matrix so the opportunities can be compared across the basin.
 - a. What will the Atlas update process look like?
 - b. Revisit key knowledge gaps – Joe
 - i. In both Wallowa and UGR/CC
 - c. Pull together new data to inform Atlas update.
 - i. USFS post doc first step will be to pull together a lot of the UGR data.
 - ii. M & E matrix

- 1.** Was the first Atlas in Catherine Creek? Or have others developed before and updated that we could learn from. The first was CC but other similar strategies have been updated
 - a.** Tucannon
- iii.** Fish Periodicity – changes, i.e. Indian Creek CHS.
- iv.** Fish Use
 - v.** Limiting Factors
- vi.** Restoration Actions – new actions, remove some that are no longer being used