MIDDLE FORK JOHN DAY RIVER

INTENSIVELY MONITORED WATERSHED

2024 Summary Report

PREPARED BY THE MIDDLE FORK IMW WORKING GROUP

Photo credit: CTWSRO

Oxbow Conservation Area below Granite Boulder in Phase 2 of the Mine Tailings Restoration Project.

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ODFW surveyor pointing out a water temperature logger in Camp Creek.

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The Middle Fork John Day River Intensively Monitored Watershed (MFIMW) Summary Report represents the dedicated effort of numerous individuals, agencies, organizations, community members, and landowners. We would like to thank all of the 'boots on the ground' who made the restoration, monitoring, and research possible. We would like to thank the authors of each individual chapter for their dedication, time, and thoughtful research that is advancing our knowledge of the MFIMW and beyond. The scope of the MFIMW work cannot happen without collaborative funding support. IMW-specific research was supported by Pacific States Marine Fisheries Commission funds - administered by Ken Fetcho with the Oregon Watershed Enhancement Board. Additional matching effort and funding was provided by USFS, BOR, ODFW, BPA, and CTWSRO. GIS map templates and creation of primary maps were created by Nadine Craft (ODFW). Graphic design and editing were provided by Linda Repplinger (OWEB). In addition, we would like to thank Dan Armichardy, Don Butcher, Mark Croghan, Jeremy Henderson, Casey Justice, Ryan Monzulla, Jeff Moss, Erica Porter, Jim Ruzycki, John Selker, Mary Lou Welby, and Alex Woolen for outside review and comments. Additional thanks to Chris Jordan (NOAA) for assistance in formulating next steps for the MFIMW. Finally, we would like to acknowledge the effort of the Report Compilation Core Team members: Kasey Bliesner, Melody Feden, Ken Fetcho, Linda Repplinger, and Lauren Osborne. Their commitment to work on the organization of the report and to complete numerous reviews and edits under a tight timeline helped this report come to completion.

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Credit: BMLT

Beaver dam on the upper MFJDR in Phipps Meadow provides refuge for many fish species.

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Please contact the author(s) of the specific research for additional data requests and prior to any documents being published with new analysis using the data in this report.

Report Scope

This report builds off the [10-Year Summary Report] completed in 2017 and represents additional years of work and voluntary reporting by numerous agencies and individuals, conducting restoration, research, and monitoring activities in the upper Middle Fork John Day River. On a voluntary basis, principal investigators and their co-authors wrote individual reports, describing their recent research and findings. The reports were compiled, along with pertinent background information, into the current Summary Report.

This report does not intend to summarize all the monitoring and research that has occurred over the life of the MFIMW. There are several monitoring efforts that have produced publications after 2018 that are not summarized in this document. To bring attention to these important documents we are providing links to some but not limited to the following reports. In addition, readers can access the MFIMW website or the PNAMP IMW website for additional MFIMW-related reports and information.

Additional Information:

- A variety of monitoring has occurred across multiple phases of restoration in the Middle Fork Oxbow Conservation Area, including the monitoring and assessment of critical thermal dynamics in the MFJDR performed by Oregon State University and the geomorphology and habitat monitoring performed by the University of Oregon which were summarized in Appendix H and D respectively, of the 10-year Summary Report.
- In addition, some data analyses and modeling were completed after the 10-Year Summary Report was finished are included in peer reviewed journal articles completed by Austin Hall and John Selker 2021 , Hall et al. 2020, and Nash et al. 2018.
- Additional sampling occurred in 2019 at 15 sites in the MFJDR and 10 sites in Camp and Lick creeks that were originally sampled in 2008/2009 and resampled in 2014 to track watershed scale stream habitat condition changes following the Pacfish/Infish Biological Opinion Effectiveness Monitoring Program (PIBO) sampling methods. For full detail on sampling results for this monitoring effort access the PIBO Final Report that was completed in 2021.
- Finally, Matthew Goslin completed his dissertation where he explored the effects of the riparian sedge, *Carex nudata* on geomorphic processes in the MFJDR as well as the environmental drivers of C. nudata's distribution.

The goals of this report are to:

1) Collectively reflect on the restoration and monitoring work that has occurred.
2) Use the Lessons Learned to guide future restoration and monitoring projects.
3) Strengthen the relationships between restoration practitioners, monitoring agencies, and data managers.

These goals allow for a continual increase in efficiency and efficacy of current restoration practices to ensure thriving populations of wild salmonids in the MFJDR. These lessons are meant to be applied to basins across the Pacific Northwest to assist in the recovery of wild salmonid populations.

Report Organization

The body of this Summary Report is organized such that projects are represented in the same order in each section of the report. An overview of MFIMW activities, key findings from provided reports, and recommendations can be found in the Executive Summary. For full details about a specific monitoring project including methods, analyses and results readers can refer to Chapters 1-8. Links, bookmarks, and navigation have been provided, where possible, to ease in viewing this document electronically.

Acronyms & Abbreviations

7DADM	Seven-day Average Daily Maximum
BACI	Before and After Control Impact
BI	Biotic Index
BMLT	Blue Mountain Land Trust
BOR	Bureau of Reclamation
BS	Bank Stabilization
CBMRCD	Columbia-Blue Mountain Resource Conservation & Development Area
CE	Capture Efficiency
Cfs	Cubic Feet per second
CHaMP	Columbia Habitat Monitoring Program
CR	Channel Reconfiguration
CREP	Conservation Reserve Enhancement Program
CRITFC	Columbia River Inter-Tribal Fish Commission
CTWSRO	Confederated Tribes of the Warm Springs Reservations of Oregon
DCA	Dunstan Conservation Area
DPS	Distinct Population Segment
DSL	Oregon Department of State Lands
DTS	Distributed Temperature Sensing
EDT	Ecosystem Diagnosis and Treatment model
ELJ	Engineered Log Jam
EPA	Environmental Protection Agency
EPT	Ephemeroptera (mayfly), Plecoptera (stonefly), and Trichoptera (caddisfly)
ESA	Endangered Species Act
ESU	Evolutionarily Significant Unit
FIP	OWEB funded Focused Investment Partnership
FLIR	Forward Looking Infrared
FNT	Fast Non-Turbulent
FR	Floodplain Reconnection
GL-GL	Greenline-to-Greenline
GRTS	Generalized Random Tessellation Survey
IHI	Instream Habitat Improvement
IMW	Intensively Monitored Watershed
IQR	Inter-quartile range
JDA	John Day Dam
JDBP	John Day Basin Partnership
JDR	John Day River
JDLM	John Day River Lower Mainstem
JDUM	John Day River Upper Mainstem
LLR	Log Likelihood Ratio
LWD	Large Woody Debris
MEPS	Medial Eye to Posterior Scale
MFFCA	Middle Fork Forrest Conservation Area
MFJDR	Middle Fork John Day River
MFIMW	Middle Fork (John Day River) Intensively Monitored Watershed
MPG	Major Population Group
MY	Migration Year
NFJDR	North Fork John Day River
NFJDWC	North Fork John Day Watershed Council
NHPA	National Historic Preservation Act
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NRCS	USDA Natural Resources Conservation Service
NWPPC	Northwest Power Planning Council
O/E	Observed/Expected
OCA	(Middle Fork) Oxbow Conservation Area
ODEQ	Oregon Department of Environmental Quality
ODFW	Oregon Department of Fish & Wildlife
OPRD	Oregon Parks and Recreation Department
OSU	Oregon State University
OWEB	Oregon Watershed Enhancement Board
PDO	Pacific Decadal Oscillation
pHOS	percent Hatchery Origin Spawners
PIBO	PacFish/InFish Biological Opinion
PIT	Passive Integrated Transponder
PNAMP	Pacific Northwest Aquatic Monitoring Partnership
PNW	Pacific Northwest
PSMFC	Pacific States Marine Fisheries Commission
RM	Riparian Management
Rm	River Mile
RST	Rotary Screw Trap
rkm	river kilometer
SNPs	Single Nucleotide Polymorphisms
SWCD	Soil and Water Conservation District
SWE	Snow Water Equivalence
SFJDR	South Fork John Day River
TDA	Total Dissolved Solids (TDS)
TIR	Thermal Infrared
TMDL	Total Maximum Daily Load
TNC	The Nature Conservancy
UMFWG	Upper Middle Fork (John Day) Working Group
UO	University of Oregon
USDA	Unites States Department of Agriculture
USFS	USDA Forest Service
USGS	Unites States Geological Survey
WTM	Water Temperature Monitoring

Credit: OWEB

MFIMW partners discussing the future restoration activities in the upper MFJDR Phipps Meadow.

Executive Summary

The Middle Fork John Day River (MFJDR) basin, located in northeast Oregon, has experienced nearly two centuries of land management practices that have contributed to the decline of federally threatened Mid-Columbia summer steelhead *Oncorhynchus mykiss* and depressed spring Chinook Salmon *O. tshawytscha*. Activities such as beaver trapping, road construction, clear-cut logging, fire suppression, channel rerouting, floodplain and wetland drainage, grazing, and mining have all had a lasting impact on the MFJDR. While the most damaging of these practices have been curtailed, their enduring adverse effects persist, resulting in and now recognized as key limiting factors to steelhead and salmon recovery in the MFJDR (CBMRCD 2005; Carmichael and Taylor 2010). Limiting factors include degraded floodplain function and connectivity, reduced habitat quantity, quality, and diversity, increased water temperature, and altered hydrology and sediment routing. A primary strategy to address the conditions that hinder salmonid recovery in Columbia Basin tributaries, including the MFJDR, is habitat restoration. However, investments in salmonid habitat restoration oftentimes do not include effectiveness monitoring (Roni et al. 2002; Roni P. ed. 2005, Bernhardt et al. 2005), leaving project planners to rely upon anecdotal evidence to infer benefits to fish populations.

To address this problem, an Intensively Monitored Watershed (IMW) program was created across the Pacific Northwest (PNW) to monitor fish population responses to restoration actions, evaluate restoration effectiveness, and better understand the relationships between fish and habitat. In 2008, the MFJDR joined the IMW network, with funding through the National Marine Fisheries Service (NMFS), in coordination with the Pacific States Marine Fisheries Commission (PSMFC), and the Oregon Watershed Enhancement Board (OWEB).

The primary goals of the Middle Fork Intensively Monitored Watershed (MFIMW) are to:

A. Evaluate the overall impact of restoration actions to summer steelhead and spring Chinook Salmon in the Upper MFJDR, and

B. Understand how specific restoration actions impact instream habitat, temperature, and salmonid metrics at the watershed, sub-watershed, and reach scales.

The MFIMW working group dedicated 2023 to evaluate and summarize:

1) Restoration actions and recommendations.

2) Findings from the MFIMW for ongoing research or new projects initiated since 2017, including eight individual research reports.

3) Adaptive management strategies implemented since the last MFIMW summary report was completed in 2017.

Additionally, we produced another set of lessons learned and recommendations from the members of the MFIMW working group to guide future restoration and monitoring approaches.

Restoration Actions

From 2008 to 2022, 149 restoration projects were implemented along the upper mainstem MFJDR and its tributaries, including 73 miles of instream habitat treated, removal of barriers that improved access to 135 miles of habitat, instream water leases that protect over 6 cfs of flow, and riparian planting and fencing along 39 stream miles. This habitat restoration work aimed to target key limiting factors, including decreased habitat complexity, degraded floodplain function and connectivity, and high water temperatures, (see the restoration inventory table; Table 1). Many of the restoration projects were multi-faceted and designed to simultaneously address multiple limiting factors, with the intent of maximizing ecosystem benefits from restoration investments. Restoration actions were applied throughout the basin (restoration map; Figure 10), with areas of concentrated restoration occurring on the Confederated Tribes of the Warm Springs Reservation of Oregon (CTWSRO)-owned Oxbow Conservation Area (OCA), Middle Fork Forrest Conservation Area (MFFCA), Dunstan Conservation Area (DCA) and the USFS-owned Camp Creek watershed. Restoration action types shifted slightly from the 2008-2016 to the 2017-2022 time periods, with more projects focusing on riparian improvement, floodplain connectivity, and instream restoration and fewer projects implementing instream flow restoration or fish barrier removals in the latter time period.

Key Findings

We found that approximately 86% of recommendations from the 2017 10-year Summary Report were addressed, have been partially addressed, or are part of ongoing efforts to address. These findings suggest that there was wide awareness of the recommendations and a deliberate intent to address them through improved planning, monitoring, and restoration efforts.

We are beginning to document signs of positive responses to restoration, despite the lack of a documented population-level increase in freshwater fish productivity. Areas of monitoring that are demonstrating positive responses include reach-scale fish response across multiple life stages, improvements in macroinvertebrate community assemblages, increases in riparian vegetation, increased stream channel and instream habitat complexity, and stabilization of water temperatures in some areas of the MFJDR where extensive restoration has occurred (Chapters 1, 2, 3, 5, 6, 7, 8).

Watershed scale Response of Salmonid Populations to Restoration Actions

We monitored watershed-scale summer steelhead and Chinook Salmon response to MFJDR restoration, including evaluating the underlying mechanisms driving observed responses. While abundance is an important metric for population assessments, productivity estimates (measured as the number of out-migrating offspring (juveniles) produced per adult) are key indicators of population responses to watershed restoration activities and can help us understand drivers of population dynamics, factors limiting population productivity, and relationships between freshwater habitat and capacity (Chapters 1, 2).

Detecting restoration responses in the MFIMW has been challenging given the size, diversity, and long-time span of restoration actions in the basin. The first 10 years of watershed-scale salmon and steelhead evaluation showed relatively little change in abundance or productivity when compared to reference watersheds (South Fork John Day (SFJD) population for steelhead, and John Day River Upper Mainstem portion (JDUM) for Chinook Salmon). Key findings for steelhead and Chinook Salmon are presented below:

Steelhead:

While steelhead productivity in the MFJDR has not increased since 2008, it also has not declined over the past 5 years, whereas the SFJD reference steelhead population has. The stable trend observed in the MFJDR potentially indicates either a positive response to improved habitat conditions or inherently more climate-resilient conditions for steelhead rearing in the MFJDR than in the SFJDR. Results at the watershed scale show that steelhead recovery is hindered by density dependence at the juvenile life stage likely due to limited rearing habitat and high stream temperatures. Density-dependent processes occur when population growth and survival are influenced/regulated by the density of the population, such that increasing adult production results in decreasing additional juvenile production (Chapter 1).

Chinook Salmon:

Chinook populations experienced very low adult and juvenile abundances in multiple years, likely due to environmental conditions including high water temperatures and low flow, which negatively affected freshwater productivity. The MFJDR population of adult Chinook Salmon experienced significant pre-spawn mortality in five of the last fifteen years because of low stream flows coupled with high temperatures in early summer. While Chinook abundance and productivity have not significantly increased since the inception of the MFIMW, freshwater productivity measured as smolts per redd show an increasing trend in the MFIMW when compared to the reference watershed population in the Upper John Day River mainstem from brood year 2016 to 2019. In addition, Chinook adults are redistributing spawning activity to restored reaches along the mainstem MFJDR. This result indicates that further restoration may create more desirable spawning locations for Chinook Salmon and may create population resilience by distributing spawning locations across the watershed. Improved spawning habitat coupled with targeted restoration to improve juvenile rearing habitat and reduce water temperatures will benefit Chinook abundance and productivity. Elevated stream temperature conditions must be improved before we expect to see any response to restoration actions targeted to reduce density dependence through increased habitat quality (Chapters 1, 2, 3, 4).

Mechanistic Understanding of Restoration Actions

- Long-term (20+ years) habitat trend monitoring showed that positive habitat responses including deeper residual pool depths, narrower channel widths, increased habitat complexity, and higher large wood densities, were greater in some reaches where both passive and active restoration approaches were applied compared to reaches where only passive restoration or adaptive grazing management was implemented (Chapter 5).

- Long-term monitoring is critical because changes to water temperature, habitat, ecosystem dynamics, vegetation, and geomorphology in response to restoration can be slow, often taking more than a decade to realize (All Chapters).
- The active placement of wood into streams "jump-started" an otherwise slow, long-term process (i.e., natural wood recruitment), yielding increases in large wood in active treatments that were not observed in reaches where only passive restoration treatments were applied (Chapter 5).
- Long-term ecological monitoring suggests that the active placement of in-stream wood and the passive-induced (i.e. reduced grazing) expansion of streamside vegetation, along with shifts in species composition (especially, *C. nudata*), were both contributing to enhanced habitat complexity (Chapter 5).
- In adaptive grazing lands (i.e. private ranches), some of the metrics assessed were also moving in a positive direction toward restoration goals, suggesting that adaptive grazing practices are indeed evolving over time, although responses were not as strong as shifts on lands where restoration was the key priority and our observation is based on a small sample size (Chapter 5).
- Though plantings have been intensively installed, the OCA riparian area remains sparsely vegetated by woody stems with little canopy cover present. The vegetation study conducted on the OCA in 2021 showed low survival of installed plants, with almost a fifth of the plants being lethally browsed by small rodents within the first-year post-installment (Chapter 6).
- Raising the groundwater elevation was a fundamental goal of the OCA restoration project to encourage groundwater recharge to the stream and to increase the duration of floodplain inundation. Results show increased water elevation levels at one well and more consistent ground water elevation levels throughout the summer months at another well (Chapter 6).

Restoration Impacts to Macroinvertebrate Related Findings

- We are seeing some positive responses across some metrics in areas of intensive restoration for example, benthic macroinvertebrate data analysis suggests that the MFJDR supports more diverse and species-rich assemblages of benthic communities that are less tolerant to fine sediment and thermal stress than those in the SFJDR (Chapter 7).
- Macroinvertebrate benthic and drift data indicated positive post-restoration changes in ecological conditions at one site located in the DCA. The DCA is a conservation property operated by the CTWSRO that has had several passive and active restoration actions implemented over a long period of time. In addition, the long-term monitoring of vegetation and habitat conditions in the DCA performed by McDowell et al. (2018) demonstrated improving conditions including narrowing channel widths, increasing wetland vegetation presence, LWD amounts, and stream habitat complexity (Chapters 5 and 7).
- Of the 14 drift sites evaluated, 8 showed significant increase in the number of mayflies and 4 showed significant increase in drift concentration, of which 3 of those 4 also showed a significant increase in drift biomass, collectively indicating improved food availability for juvenile salmonids.

These 3 sites are in the DCA and OCA and have had several large-scale restoration actions implemented over time (Chapter 7).

- No consistent relationship was detected between restoration intensity and macroinvertebrate community response. A general lack of consistent temporal trends or consistent pre/post-restoration changes in benthic and drift communities suggests that ecological conditions have remained largely unchanged in the MFIMW over the 2010-2022 monitoring period (Chapter 7).

Temperature Related Findings

- Focused parr-to-smolt survival monitoring across the riverscape identified a central zone within the MFJDR where high water temperatures negatively impact juvenile Chinook Salmon survival. Restoration effectiveness will be maximized when information on the impact of temperature, at the reach-scale, is incorporated into a restoration prioritization framework. (Chapter 2 and see Juvenile Chinook Limiting Factors map figure) (Figure E1).

- Chinook Salmon parr originating in sections with high water temperatures dispersed to cooler mainstem or tributary reaches, demonstrating that high water temperatures were a primary driver of Chinook Salmon parr dispersal in 2021 (Chapter 4).

- Mainstem MFJDR Chinook Salmon parr density was negatively associated with maximum July water temperatures. While the estimated total parr abundance within the mainstem accounted for 71% of all parr within MFJDR, the highest estimated parr densities were in Granite Boulder Creek, and mean density was greater in six of the nine tributaries compared to the mainstem (Chapter 4).

- Water temperatures over the life of the MFIMW have remained stable with no consistent warming or cooling trend during a period of increasing average air temperature and decreasing annual mean streamflow (Chapter 8). The stable water temperature trend as air temperatures were warming and streamflow was decreasing suggests that additional factors such as restoration actions and passive riparian regrowth may be stabilizing or otherwise mitigating for expected warming with increased air temperature and decreased flow (Chapter 8).

- Significant trend results for unadjusted water temperature metrics were dominated by tributary locations (over mainstem locations). Significant water trend results were relatively evenly split between increasing and decreasing trends. Fewer decreasing trends were significant for locations within restoration reaches compared to unrestored reaches (Chapter 8).

- Across all water temperature metrics there were 34 sites that demonstrated a decreasing trend in water temperature, 14 sites were located in restoration reaches and 8 were located in the OCA (Chapter 8).

Restoration Recommendations

We recommend implementing restoration aimed at reducing high temperature effects (Figure E1). Specific actions include:

- Convert long 'fast non-turbulent' (FNT) habitat units into a series of pool/riffle habitat units.
- Narrow the channel through island formations.
- Reconnect the stream to the floodplain to facilitate the re-establishment of a riparian corridor and promote floodplain-derived hyporheic exchange.
- Develop and execute a planting strategy that prioritizes planting species that will grow quickly and provide stream shade (i.e., alders), followed by those that will create a diverse and sustainable riparian community and will be able to contribute large wood to channels in future years (i.e., willows and cottonwoods etc.). In some areas (i.e, mainstem MFJDR in the OCA), rodent-proofing young plantings, even when inside an 8-foot elk exclusion fence, is important for maximizing early-life plant survival.
- Maintaining the riparian fence downstream of river kilometer (rkm) 95.0 and continuing to invest in a viable planting strategy.

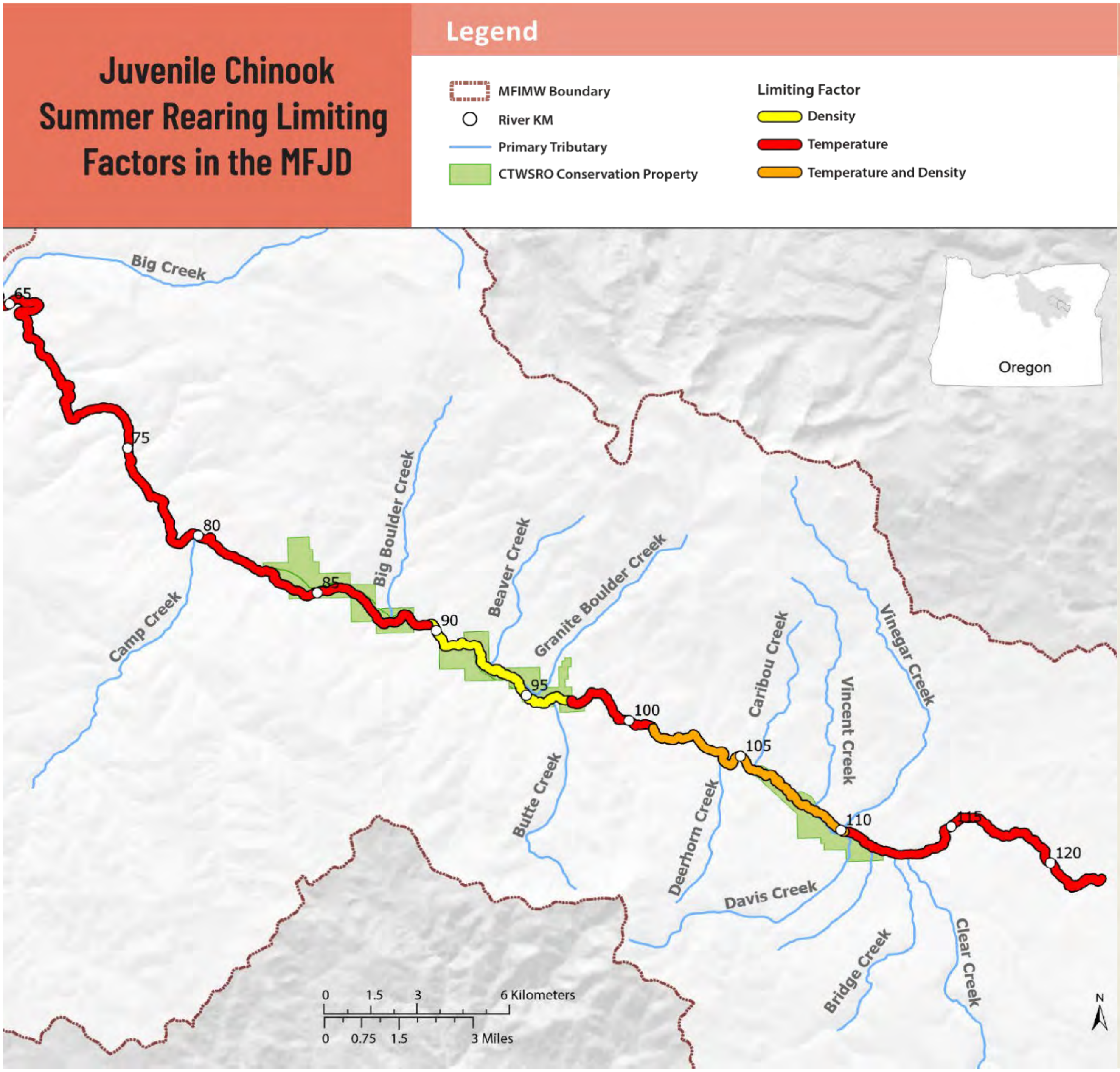
In addition, specific restoration actions should be implemented to protect and expand cool-water thermal refugia, thereby maximizing restoration effectiveness. Within the MFJDR, thermal refugia is found within and at the confluences of cool water tributaries (i.e., Granite Boulder, Vinegar, Davis, Dead Cow and Deerhorn creeks etc.) and several kilometers in the MFJDR downstream of RKM 95.0 (Granite Boulder Creek confluence) (Figure E1). Specific restoration actions to protect and expand cool-water thermal refugia include:

- Strategically place wood structures to deflect mainstem water and capture tributary water at confluences, with the goal of expanding the volume of cool-water plumes created at confluences.
- Maintain or improve connectivity to cool-water tributaries (i.e, Caribou Creek, Vinegar Creek and Bridge Creek).

Increasing the amount of high-quality physical habitat to reduce density dependence (alternatively viewed as increasing juvenile carrying capacity) among stream salmonids may be an effective strategy within the confines of suitable stream temperatures; especially considering these recommendations:

- Of the sites examined in chapter 2 (RKMs 94.5 to 114.2), restoration to alleviate density-dependent growth factors is hypothesized to be most effective near rkm 106.6 (~1.3 km upstream of Caribou Creek) and 107.7 (~0.5 km downstream of Dead Cow Gulch), and least effective around river kilometers 105.9 (0.6 km upstream of Caribou Creek) and 109.7 (0.3 km downstream of Vinegar Creek) (Figure E1).
- Restoration targeting Chinook fry carrying capacity should activate floodplain features within 1 km of high-density spawning areas. Activated floodplain features should be designed to maintain

connectivity with mainstem channels during low flows to avoid fry stranding and allow juveniles to access and benefit from floodplain food resources and favorable thermal conditions (Figure E1).



Restoration Recommendations

In the MFJDR, researchers have identified **temperature** and fish **density** as limiting factors for salmonids. Many sections of the MFJDR are too warm for salmonids, limiting growth and survival and creating high densities of fish in the thermally tolerable sections. Increased water temperature and high fish density limit the freshwater productivity of juvenile salmonids. **If water temperature conditions are improved throughout the MFJDR, this would allow juvenile salmonids to better disperse and utilize habitat throughout the basin, thus alleviating high density conditions.**

In areas where high water temperatures are the primary limiting factor (indicated by **red** and **orange**), we recommend restoration techniques that promote lower water temperatures:

- Converting long fast non-turbulent habitat units to a series of pool/riffle habitat units
- Narrowing the stream channel
- Reconnecting the floodplain
- Executing a riparian planting strategy that includes fast growing vegetation and protecting plantings from browse.

We also recommend strategies to protect and expand cool water refugia, found at the confluences of cool water tributaries (which are labeled on the map).

In areas where fish density is the primary limiting factor, restoration techniques that increase channel and habitat complexity and diversity (i.e. large woody debris, beaver dam(s) [analogs]) should be implemented to increase the growth potential and carrying capacity of the basin for juvenile salmonids. These restoration actions can be useful in areas of high water temperatures to provide complex habitats and flow refugia and allow for better growth of juvenile salmonids.

Figure E1: Map of restoration recommendations in the MFJDR.

Adaptive Management

Adaptive management is an important tool that should be used to guide restoration actions and be integrated within an IMW framework (Bouwes et al. 2016). Eighty-six lessons learned and recommendations emerged from the 2017 10-Year Summary Report. Using an adaptive management framework, the MFIMW Working Group evaluated how many recommendations were addressed by MFIMW partners from 2018-2023. We asked the researchers and restoration practitioners to reflect on the lessons learned and recommendations based on their involvement with the MFIMW, and recorded the number of recommendations that had been addresses, partially addressed, or not addressed overall and within the categories of planning, monitoring, and restoration. Results show approximately 86% of recommendations from the 10-year Summary Report (2017) were addressed, have been partially addressed, or are part of ongoing efforts to address, and suggest wide awareness and deliberate intent by MFIMW partners to incorporation recommendations through adjusting planning, monitoring, and restoration efforts.

To capture additional lessons learned and recommendations since 2018, we asked each contributing author and restoration practitioners to respond to a series of questions. These responses were summarized, and we grouped paired lessons learned and recommendations into three main topics: Planning, Monitoring, and Restoration. These lessons learned and recommendations provide valuable insights for ongoing planning, monitoring, and restoration efforts within the MFIMW and similar restoration efforts in the Mid-Columbia Basin. Full results of the new lessons learned and recommendations that were captured since 2018, and examples of how adaptive management was applied by MFIMW partners can be found in the Adaptive Management Section.

Next Steps

The MFIMW Working Group compiled a list of next steps to best utilize the list of lessons learned and recommendations and how to efficiently disseminate the information gained through this report. Efforts include:

1) Examining and implementing new lessons learned and recommendations
2) Evaluating consistency of lessons learned from the current Summary Report with the 10-Year Summary Report
3) Developing an outreach strategy to report MFIMW key findings to a variety of audiences, ranging from basin-wide partnership meetings to conference presentations
4) Updating the MFIMW public website with Summary Report findings and content
5) Initiating conversations with partnering agencies (ex. NMFS, PNAMP, additional IMWs) to reflect on the MFIMW findings across the broader IMW network and determine additional next-steps for encouraging implementation of lessons learned and recommendations to decision-makers and practitioners

Through these efforts, we hope to improve communication and implementation of lessons learned and recommendations for future evaluations.

Credit: ODFW

Coyote Bluff is a notable landmark for those that frequent the MFJDR.

Introduction

Salmon and steelhead populations are declining throughout the Pacific Northwest due to many complex factors, including hatcheries, harvest, hydropower, climate change, changes in ocean productivity, and degraded and altered freshwater habitat (Welch et al. 2020, Bilby 2022). Considerable resources have been allocated towards restoring salmon habitat. Since 2000, over a billion dollars have been spent on salmon habitat restoration in the Columbia Basin (Bernhardt et al 2005, Bilby 2022). Historically, restoration efforts rarely included effectiveness monitoring (Roni et al. 2002; Roni P. ed. 2005, Bernhardt et al. 2005, Wozniacka 2015), leaving project planners to rely upon anecdotal evidence or intuition to infer benefits to fish populations. To address this problem, restoration efforts with associated effectiveness monitoring programs were established, 13 are currently operating in the Pacific Northwest (PNW). Referred to as Intensively Monitored Watersheds (IMW), these programs seek to provide quantitative evidence of restoration impacts on salmonid populations and their habitat and are considered an effective approach for evaluating salmon and steelhead response to habitat restoration (Bilby et al. 2005; PNAMP 2005; Nelle et al. 2007, Bilby 2022).

The goal of an IMW is to improve our understanding of the relationship between anadromous fish and their freshwater habitat (Bilby et al. 2005; PNAMP 2005). IMW research can reveal causal mechanisms, allowing us to better predict restoration effects across river systems in a cost-effective manner. Through documenting and sharing the lessons learned from the network of IMWs, resource managers in the Pacific Northwest will be able to implement further restoration with greater confidence, and effectiveness monitoring efforts can be prioritized and directed for maximum value (Bennett et al. 2016).

Beginning in 2008 the National Marine Fisheries Service (NMFS), in coordination with the Pacific States Marine Fisheries Commission (PSMFC), began funding a network of IMWs across the Pacific Northwest. The Pacific Northwest Aquatic Monitoring Partnership (PNAMP) coordinates this IMW network. The Oregon Watershed Enhancement Board (OWEB), in coordination with PNAMP, and the National Oceanic and Atmospheric Administration (NOAA) funded an IMW in the upper Middle Fork John Day River (MFJDR) basin in Oregon. The goal of the Middle Fork John Day River IMW (MFIMW) is to understand the causal mechanisms between stream habitat restoration and changes in salmonid production at the watershed scale (UMFWG 2011).

MFIMW Development

The MFIMW is coordinated by a subset of stakeholders that originally participated in the Upper Middle Fork John Day Working Group (UMFWG). These participants— state and federal agencies, tribal entities, universities, and conservation groups—continue to coordinate MFIMW monitoring efforts and discuss where and how restoration should be implemented in the study area. Participation in the MFIMW fluctuates with funding, study plans, monitoring capacity, and research focus. Current active participants are included in Figure 1.

Figure 1. Participating agencies and organizations in the MFIMW as of report completion in 2024.

Participants of the UMFWG convened in April of 2007 and began to develop a MFIMW monitoring plan based on restoration planning efforts that were already underway. Restoration efforts moved forward independently of the monitoring efforts that formally began in 2008. Given that a minimum of 5-10 years is needed to detect a trend in steelhead or salmon populations, the study was anticipated to last at least a decade. The first few years of the MFIMW were used to determine the experimental design, monitoring methods, and metrics. The MFIMW's structure, focus, and study design was informed by the variety of pre-existing collaborative restoration and monitoring projects in the basin. These included monitoring by Oregon Department of Fish and Wildlife (ODFW) of spring Chinook Salmon *Oncorhynchus tschawytscha* and summer steelhead *Oncorhynchus mykiss*; PacFish/InFish Biological Opinion (PIBO) monitoring by USDA Forest Service (USFS); and conservation and monitoring efforts by The Nature Conservancy (TNC) and Confederated Tribes of the Warm Springs Reservations of Oregon (CTWSRO).

In 2010 the consulting firm, Eco Logical Research Inc., was contracted to complete a study design for monitoring planned restoration activities, improving our ability to detect changes in fish populations and determine whether the changes were caused by environmental factors, restoration, or a combination of these. The resulting Upper Middle Fork John Day River Intensively Monitored Watershed Draft Experimental Design and Implementation Plan was developed and is available online.

In 2017 the MFIMW working group completed a summary report detailing the first 10 years of monitoring and restoration work completed within the MFIMW area (Middle Fork IMW Working Group 2017). Given the success of the working group, collaboration, and the recognition that ongoing restoration required further monitoring, the MFIMW has been funded each year since 2017, although at a reduced level from initial funding.

In 2019, OWEB funded a [Focused Investment Partnership (FIP)] in the John Day Basin that included a focus in the MFIMW area to implement strategic restoration actions and measure ecological outcomes to improve native fish habitat. The MFIMW provides key monitoring in the MFJD FIP project area.

Recommendations from the 10-year report included:
1) Adaptations to monitoring to better understand mechanisms driving fish response to restoration.
2) Improved restoration techniques for increasing efficiency and effectiveness.
3) Strengthen collaborations, partnerships, and data management.

Authors of the 10-year report also strongly recommended continued reporting of results, data, and adaptive management strategies on a consistent 5-year basis, and 2022 marked the 5-year point. With those recommendations in mind, MFIMW coordinators asked for primary researchers with monitoring that was initiated or continued into the 2017-2022 timeframe, to collaborate and share their findings and summarize the restoration since 2017, which culminated in the compilation of this report.

Credit: CTWSRO.

Restoration implementation in progress by CTWSRO on the MFFCA spanning from Vinegar Creek downstream to Vincent Creek.

Credit: ODFW

Wildflowers blooming in the Camp Creek Watershed.

Study Area

The John Day basin lies in the Mid-Columbia Plateau Region in Northeastern Oregon. The basin consists of 5 main subwatersheds: the John Day Lower Mainstem (JDLM), the John Day Upper Mainstem (JDUM), the North Fork John Day (NFJDR), the South Fork John Day (SFJDR), and the Middle Fork John Day River (MFJDR). The MFJDR originates in the Blue Mountains of the Malheur National Forest, south of the NFJDR. The MFJDR flows westerly for 75 miles, and merges with the NFJDR about 18 miles north of the town of Monument. The MFJDR is a fourth-field watershed (USGS cataloging unit 17070203) that drains 806 miles^2 with a perimeter of 158 miles (Figure 2). Watershed elevations range from 2,200 feet near the mouth to over 8,200 feet in the headwater areas. The upper portion of the MFJDR, extending upstream and inclusive of the confluence with the MFJDR and Big Creek, was defined as the MFIMW study area (Figure 2). The upper portion of the MFJDR was chosen for the IMW because the majority of the restoration actions were occurring in this area, it provided a reasonable size to monitor changes, and the land is primarily publicly owned.

Land ownership in the MFIMW area is predominantly public (USFS owned and managed) with smaller portions that are private (Figure 2). In addition, several large parcels have been purchased and are managed by restoration-focused organizations such as the CTWSRO, the Blue Mountain Land Trust (BMLT) and the Oregon Parks and Recreation Department (OPRD). In 2021, (BMLT, with funding through OWEB), acquired a key parcel of private land in the headwaters of the MFJDR, called Phipps Meadow. These conservation-focused parcels combine to cover more than 14 miles^2 of habitat and have been important acquisitions for continued restoration progress.

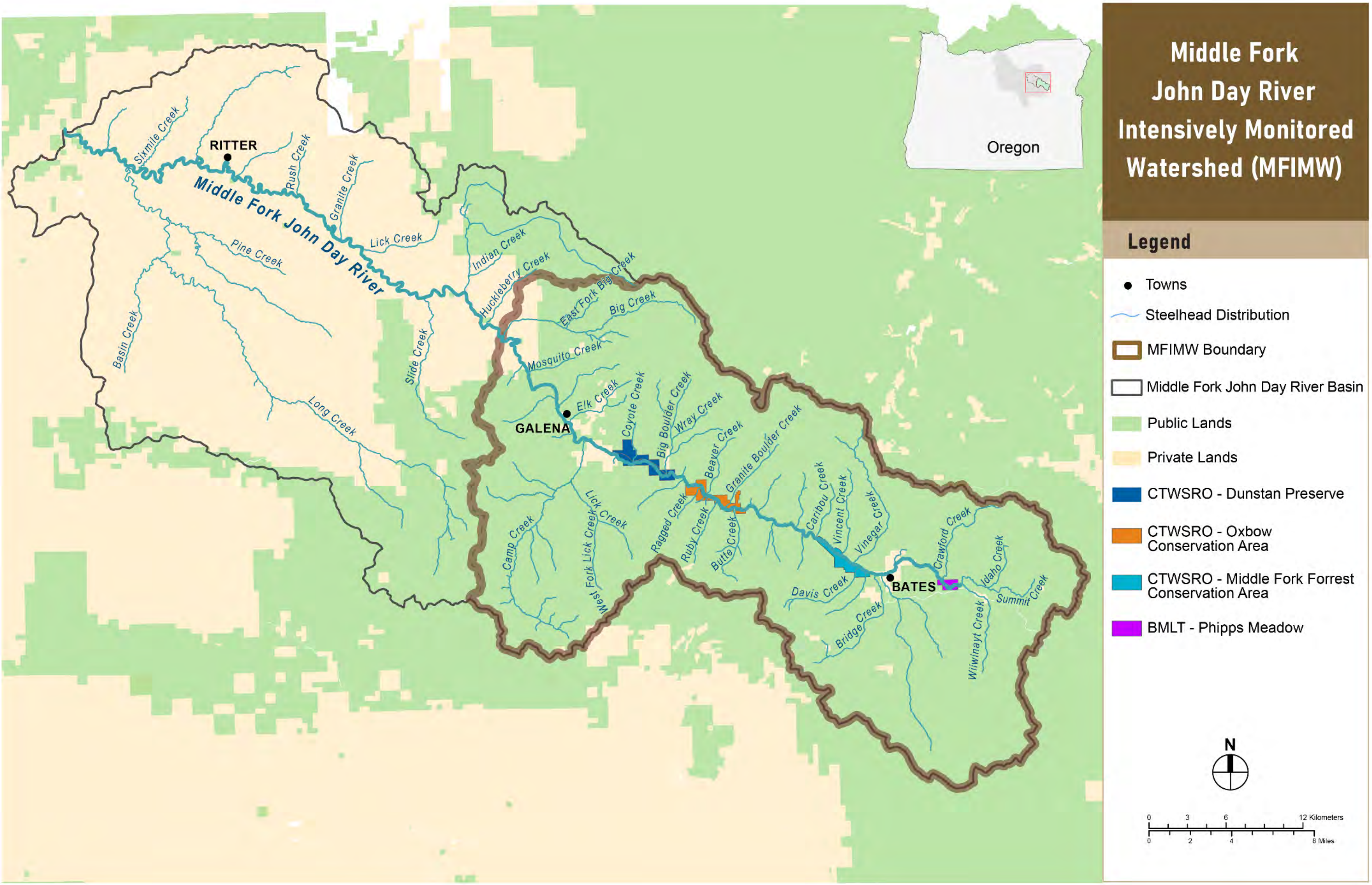


Figure 2. Map of the MFIMW and MFJDR basin showing public and private land ownership. Conservation Areas owned by CTWSRO and BMLT are highlighted. The majority of public lands in the MFIMW area are owned by the USFS.

Credit: CTWSRO

MFJDR Vinegar Creek confluence at high flows, post 2022 Vinegar to Vincent restoration implementation.

Geomorphology

The geomorphology of river channels and their associated floodplains and valleys strongly influence the process for creating and maintaining salmon and steelhead habitat. The upper MFJDR follows a common geomorphic pattern, characterized by laterally unconfined valleys interspersed with narrower, semi-confined reaches. Most of the land use and visible impacts to streams and floodplains occurs in the upper watershed, which is the focus of the MFIMW.

A [geomorphic framework](#) to inform stream restoration planning for the MFJDR (O'Brien 2017) examined river diversity, evaluated geomorphic condition, and determined the potential for geomorphic recovery. About two-thirds of the watershed was found in good geomorphic condition with the remaining one-third in moderate to poor condition. Most reaches in moderate condition were determined to have a high potential for recovery.

Credit: ODFW

Early spring sampling conditions for Chinook Salmon fry.

Credit: ODFW

MFJDR in the fall, picturing Dixie Mountain.

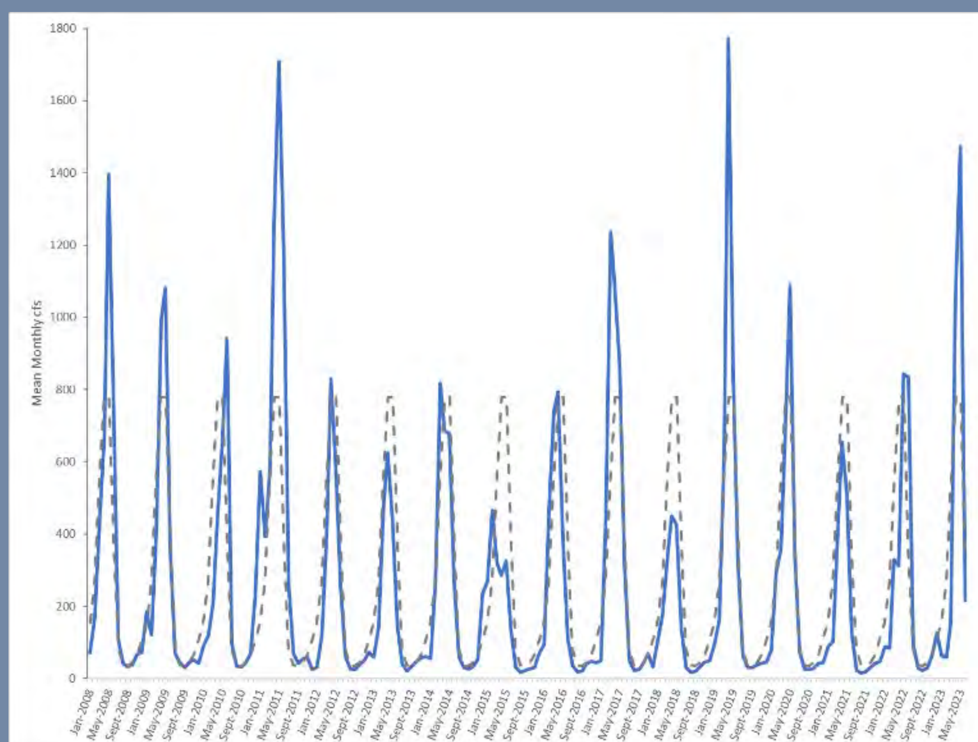
Climate

The John Day Subbasin has a continental climate characterized by low winter and high summer temperatures, low average annual precipitation, and dry summers. Most precipitation falls between November and March as snow. Less than 10% of the annual precipitation falls as rain during July and August, usually from sporadic thunderstorms that can ignite wildfires. The upper elevations receive up to 50 inches of precipitation annually mostly in the form of snow; lower elevations receive 12 inches or less of precipitation. Most water in the John Day Subbasin is derived from the upper watershed, primarily in the form of melting snow resulting in highly variable discharge from peak to low flows (CBMRCD 2005). The hydrologic curve has shifted from historic times, with peak flows greater and earlier than in the past and late season flows more diminished. It is suspected that these effects are due to greatly reduced rates of soil infiltration, reduced capacity for ground water/riparian storage, and diminished in-channel storage in beaver ponds due to loss of beaver populations and channel complexity (NWPPC 2001). It is further believed that the hydrologic regime changes are due to increasing air temperatures and its impact to snowfall and snowmelt. Decreased snowpack and increasing spring temperatures, which hasten the onset of snowmelt during spring, have shifted the timing and magnitude of discharge in the MFJDR. Summer base flows in the Blue Mountains have declined 21-28% between 1949 and 2010, possibly due to these changing climate conditions (Safeeq et al. 2013). The years of high and low snow water equivalent (SWE) coincide with the highest and lowest monthly discharge recorded over the last 15 years in the MFJDR (Figure 3; Figure 4; Figure 5). These fluctuations in precipitation directly influence streamflow conditions in the MFIMW (Figure 3).

A changing climate has led to an increase in the intensity, frequency, and size of wildfires in the PNW (Dennison et al 2014, Halofsky et al 2020). The twenty largest wildfires ever recorded in Oregon have occurred since 2002 (Price and Rein, 2021). Large fires are associated with warm, dry conditions which are predicted to increase as climate change intensifies (Halofsky et al 2020). Decades of fire suppression has increased fuel loads available for catastrophic fire. In the MFJDR basin, wildfire activity has been minimal for the last 20 years (CalTopo, n.d.) despite warm, dry conditions and dense tree stands. The largest fire to occur since the beginning of the MFIMW, was the Crockett Knob fire in 2022 effecting 4,287 acres (Inciweb, n.d) around the headwaters of Big Boulder Creek, located on the north side of the basin (CalTopo, n.d). Consecutive or a large catastrophic fire(s) in the MFJDR basin may lead to decreased species diversity within nearby streams and may be devastating for fish species that depend on cold-water refugia (Whitney et al. 2015; Isaak et al. 2010). In the last several years, the Forest Service has released several strategies to prevent the severity of wildfires with prescription burns and fuel reduction strategies (USFS). Specifically, in the MFJDR basin, treatments have mainly included pre-commercial thinning (pers. communication with Becky Long and Cody Lund, USFS). Funding for fuel-reduction treatments have increased to treat more area in the coming years (pers. communication with Becky Long and Cody Lund, USFS), per the overall goals of the USFS (USDA, 2022).

To assist in understanding climate impacts on MFJDR discharge regimes (Figure 3), monthly discharge from the gauge at Ritter is presented. In addition, weather information was compiled from the Tipton SnoTel site located in the headwaters of the upper MFJDR at an elevation of 5,150 feet. Tipton is a part of the USDA Natural Resources Conservation Service (NRCS) SnoTel program. Data from the Tipton site includes hourly soil moisture/temperature data, and 24-hour precipitation and snow water equivalence.

Figure 3. Monthly discharge, MFJDR at Ritter. Average discharge from the 2008-2023 is denoted with the dashed grey line.



From 2008–2023 the highest precipitation was recorded in water year 2011, which was followed by 5 consecutive years of precipitation which fell below the long-term 1981-2022 average precipitation (Figure 4). Since the inception of the MFIMW in 2008, 10 of the 15 years have recorded precipitation below the long-term average (Figure 4).

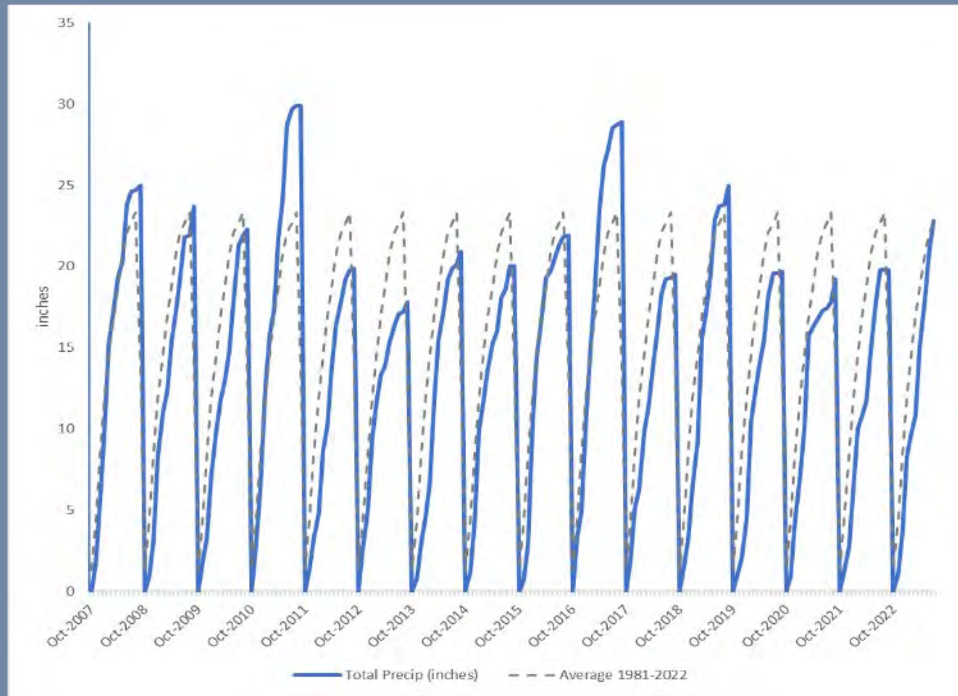


Figure 4. Total monthly precipitation at the Tipton SnoTel site for water years 2008-2022 in blue. Average monthly precipitation is displayed with the grey dashed line. Monthly precipitation falls below-average precipitation in most years from 2007-2022.

Another important weather attribute influencing streamflow is snow water equivalent (SWE), defined as the amount of water available in the snow. The March 9th SWE was evaluated for water years 2008-2022 (Figure 5), and compared to the 30-year average SWE on March 9th (11.0 inches). March 9th was selected for analysis because it is the 30 year median high SWE date (https://nwcc-apps.sc.egov.usda.gov/awdb/site-plots/POR/WTEQ/OR/Tipton.html). The lowest SWE was observed in 2015 while the highest SWE was observed in 2017. Since the inception of the MFIMW in 2008, 9 of the 15 years of SWE have fallen below the long-term high SWE.

Figure 5. Snow water equivalent (SWE) on March 9 (date of median long-term high SWE) at Tipton SnoTel Site for water years 2008 to 2022. The 30-year mean SWE on March 9th is indicated by the grey dashed line.

Credit: CTWSRO

Dredge mining operation on the MFJFDR (circa 1939). One dredge still remains on display in Sumpter Valley.

Historic and Current Land Use

Over the past two centuries, the MFJDR incurred significant post-Euro American settlement impact from beaver trapping, road building, clear-cut logging, fire suppression, channel re-routing, floodplain/wetland drainage, grazing, and mining. One of the most dramatic changes was dredge mining of a large portion of the MFJDR in the 1930s, near what was then referred to as the Oxbow Ranch, now the Middle Fork Oxbow Conservation Area (OCA), resulting in the destruction of floodplain vegetation and soils, and the creation of a straight, trench-like channel. Fortunately, the most damaging of these practices have since been curtailed and the watershed has good recovery potential. Grazing still occurs across the basin on public and private property, in lower densities than historic use and many sensitive areas have been fenced to exclude cattle.

Credit: ODFW

Log weirs spanning Camp Creek that have since been removed.

Over the last 35 years, both public and private MFJDR landowners have been working to improve watershed conditions. However, some measures have proven to be deleterious and not beneficial to the watershed. Perceived impacts from stream erosion and incision in the MFJDR spurred USFS to install many log weirs in streams on public forest land, especially in Camp Creek, a major MFJDR tributary. While these structures successfully prevented bed erosion and incision, they had negative effects to salmonids such as limiting fish passage, lateral erosion, widening channels, trapping sediment, and inhibiting natural pool formation. Subsequent scientific studies on log weirs revealed that these structures did not mitigate any limiting factors for salmonids, nor did they increase salmonid abundance, as had been hypothesized (Reeve et al. 2006; Beschta et al. 1991; Wissmar et al. 1994). Since log weirs potentially impeded juvenile salmon passage in Camp Creek and limited habitat complexity, their removal has been a major focus of current USFS restoration the Camp Creek watershed.

Credit: ODFW

Davis Creek bridge and rock barbs prior to restoration.

Credit: ODFW

Example of incised channel and trampled vegetation in the MFJDR basin.

Credit: ODW

Bates Pond (left), dam (middle) and fish ladder (right) on Bridge Creek.

Alterations to the stream (often for logging and mining) would employ the use of riprap and rock barbs to stabilize stream banks, reduce stream bank erosion, and prevent lateral channel migration. These structures were installed on several miles of the MFJDR. While successful in preventing lateral channel migration, these measures unintentionally negatively impacted salmonid habitat in streams, which is actually supported by channel migration. Bank stabilization measures inhibit meander development, preventing the formation of large meander bend pools, and disrupt the natural processes essential for riparian vegetation on stream banks (Beschta et al. 1991).

Land ownership changes starting in the 1980s led to changes in resource management across the MFIMW study area, including the purchase of key properties by the TNC, and later transferred to CTWSRO. CTWSRO implemented passive restoration actions through installation of riparian fencing to exclude cattle grazing. In addition, CTWSRO enrolled 254 acres across two of their properties in a 15-year Conservation Reserve Enhancement Program (CREP) agreement, removing grazing and irrigation from these areas until 2020, and providing new riparian plantings and 180-ft riparian buffers. Current restoration projects in the Middle Fork Forrest Conservation Area (MFFCA) have included the removal of log weirs, riprap, and rock barbs to reestablish natural processes and with them natural bank conditions, channel sinuosity, and pools.

Formerly a major timber-producing watershed, the upper MFJDR was once home to the company mill town of Bates. The area around the former townsite and mill of Bates has also experienced change. Bates is located at Bridge Creek, a crucial cold-water tributary with the MFJDR. Bates Pond, the mill pond built by the timber company in the early 1900s, elevates Bridge Creek temperature by increasing the surface area exposed to sunlight and slowing the flow of water before it reaches the MFJDR, as Bridge Creek flows into and out of the pond. Elevated temperatures significantly reduce the ability of Bridge Creek to serve as a source of cold water to the MFJDR, and thermally blocks migrating salmonids’ access to upstream spawning and rearing habitats (Middle Fork Working Group 2017).

Aquatic Species Presence and Distribution

The John Day River supports several species of fish and freshwater mussels including spring and fall Chinook Salmon *Oncorhynchus tshawytscha*, summer steelhead *O. mykiss*, bull trout *Salvelinus confluentus*, Pacific lamprey *Lampetra tridentata*, westslope cutthroat trout *O. clarkii lewisi,* mountain whitefish *Prosopium williamsoni,* western pearl shell mussels *M. falcata,* floater mussels *A. oregonensis*, and western ridge mussels *G. angulata*. Spring Chinook Salmon and summer steelhead are the predominate salmonids inhabiting the MFJDR watershed although bull trout are also found in tributaries with limited seasonal use of mainstem habitats by fluvial adults. Steelhead are the most widely distributed salmonid species occupying most tributaries and mainstem habitats. Chinook Salmon distribution is more confined to mainstem habitats and larger tributaries compared to steelhead although juvenile Chinook Salmon often migrate into cool-water tributaries during warm summer periods. Bull trout distribution is limited by their temperature tolerance to only the upper reaches of tributaries, especially Granite Boulder, Clear, and Big Creeks (Curry et al. 2011). Mussel population assessments are ongoing in the MFJDR, with CTWSRO partnering with Xerces and CTUIR to collect presence-absence data of these species on CTWSRO properties. Though populations of these mussels have been confirmed on the MFJDR, distributions appear patchy with population numbers largely on the decline.

Credit: NFJDWC

Beaver swimming in Butte Creek.

Credit: ODFW

Beaver dam on Camp Creek.

Beaver

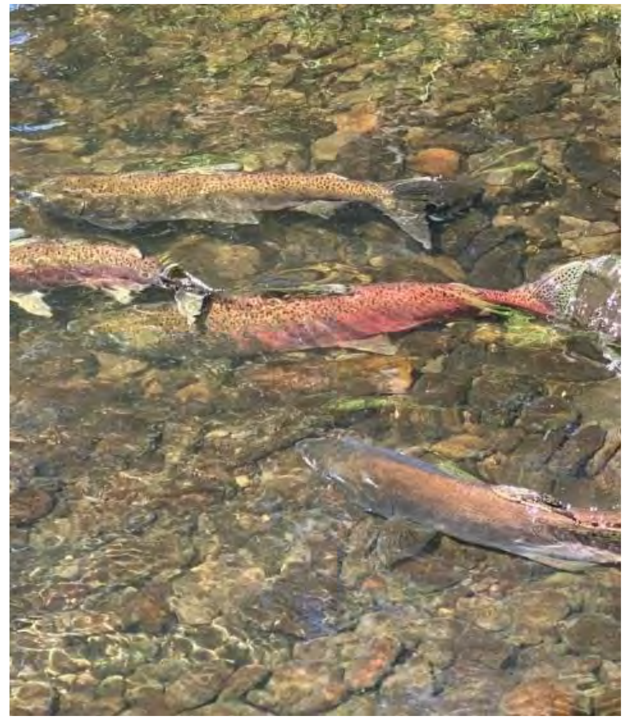
The American Beaver (*Castor canadensis*) are an important keystone species for the fish species and the landscape in the JDR basin. Floodplain connectivity, wetlands, and slow-moving water created by beaver dams provide key habitat for the depressed salmonid species and other important wildlife in the JDR basin (ODFW 2023). Protection and restoration of floodplain and riparian communities and beaver-modified habitat have been identified as key restoration actions in Oregon salmonid conservation plans (ODFW 2023). Numerous beaver dams have been identified in the MFJDR and restoration practitioners and biologists recognize the importance of beaver on the landscape and work to protect and enhance beaver habitat where possible.

Focal Species

Spring Chinook Salmon and **summer steelhead** are the focal species of the MFIMW. Mid-Columbia summer steelhead are listed as a federally threatened species (U.S. Office of the Federal Register 1999, 2006), and while Chinook Salmon are not currently listed, ODFW has conducted a Population Viability Analysis (Falcy 2022) to assess the depressed population. These focal species were selected because of strong interest in anadromous fish recovery throughout the Pacific Northwest region. Below, we summarize life-history information presented in the John Day Subbasin Plan (CBMRCD 2005) and elsewhere.

Spring Chinook Salmon

The spring run of Chinook Salmon in the John Day is included with the Mid-Columbia River Evolutionarily Significant Unit (ESU). Similar to steelhead, Chinook Salmon in the JDR have not been subjected to direct hatchery introductions. Chinook Salmon in the MFJDR compose one of the three populations in the JDR (others include the NFJDR and Upper John Day). Adult spring Chinook Salmon enter fresh water in April and migrate upstream into the MFIMW during May and June. The adults then hold and reach maturity in freshwater until they spawn in late August through late September. The MFIMW encompasses spawning, rearing, and migration corridor habitats for spring Chinook Salmon. Chinook Salmon spawning distribution is confined to mainstem habitats and larger tributaries, although juveniles often migrate into and rear in cool-water tributaries during warm summer periods. Chinook Salmon emerge from the gravel during early spring and rear in freshwater for one year before smolting the following spring and migrating to the ocean as age-1 juveniles. They spend 1-3 years in the ocean and return as age 3-5 adults.



Credit: Lauren Osborne CTWSRO

Adult spring Chinook Salmon spawning.

Summer Steelhead

The John Day subbasin contains one of the few remaining summer steelhead Major Population Groups (MPGs) in the interior Columbia Basin that have not been influenced by direct hatchery introductions. Within the John Day steelhead MPG the Interior Columbia Technical Recovery Team defined five Distinct Population Segments (DPS), of which, the MFJDR steelhead population is one. Spawning, rearing, and migration corridor habitats for summer steelhead are all found in the MFIMW. Summer steelhead are the most widely distributed salmonid species in the watershed, occupying most tributaries and mainstem habitats.

Credit: ODFW

Adult summer steelhead.

Adult summer steelhead typically enter fresh water during the summer before the year they spawn. After returning, most adults spend the first winter downstream of the MFIMW. They begin entering their spawning grounds as the ice and snow melts and typically initiate spawning activity during March (lower reaches) and April (upper reaches). Fry emerge from the gravel early in the summer and rear within the MFIMW as juvenile parr for 1-3 years before smolting during the spring and migrating downstream. They spend 1-2 years in the ocean before returning to freshwater on their spawning migration. Unlike other anadromous salmonids, steelhead can return to the ocean after spawning although only a small minority survive to spawn for a second time.

Credit: ODFW

Mainstem MFJDR with little to no riparian vegetation. High water temperature, caused by limited riparian vegetation and widened channels, is the primary limiting factor in the MFJDR.

Limiting Factors

Limiting factors are the conditions that inhibit populations of organisms or ecological processes and functions relative to their restoration and protection potential (CBMRCD 2005). Identifying primary limiting factors can greatly improve the effectiveness of restoration actions and reassessment of limiting factors over time will ensure that restoration actions are focused on the most important attributes limiting fish recovery (Bilby 2022). Limiting factors for the focal fish species were evaluated prior to implementation of the MFIMW in 2008 to guide restoration actions and experimental design processes (Middle Fork IMW Working Group 2017). Through the first ten years of the MFIMW, restoration projects targeted multiple limiting factors, including habitat diversity, fish passage barriers, large-woody debris (LWD), and channel reconfiguration. Research during the first 10-years indicated that water temperature, and related flow, and limited juvenile salmonid habitat are the primary limiting factors for salmon and steelhead (McHugh 2018, Bilby 2022, Middle Fork IMW Working Group 2017). While current restoration practices strive to target limiting factors, researchers hypothesize that until that restoration is effective at improving or eliminating current limiting factors, population level changes for anadromous salmonids will not be realized.

Below, we summarize specific limiting factors for each species, then describe some of the most pressing limiting factors in the MFIMW in greater detail. For additional information related to limiting factors in the MFJDR see CBMRCD 2005, Carmichael and Taylor 2010, NOAA 2017, Middle Fork IMW Working Group 2017, CTWSRO 2018, Falcy 2019, Bilby 2022.

Summer Steelhead

The limiting factors affecting steelhead in MFJDR include habitat diversity, degraded floodplain, channel structure, altered sediment routing, altered hydrology, and water temperature (CBMRCD 2005; Carmichael and Taylor 2010). The Ecosystem Diagnosis and Treatment (EDT) model identified habitat diversity, sediment load, temperature and key habitat quantity as limiting factors for summer steelhead early in the MFIMW process (CBMRCD 2005). In addition, steelhead abundances in the MFJDR basin are limited, in part, by freshwater rearing habitat. The annual number of smolts produced per spawner is regulated by juvenile density, where relatively fewer smolts per adult are produced at greater adult spawner escapement levels within the MFJDR watershed (Chapter 1). This observation of density-regulated abundance within the MFJDR suggests that 1) juvenile habitat quality and quantity are limiting salmonid production and 2) adult fish production should increase in response to juvenile habitat restoration.

Spring Chinook Salmon

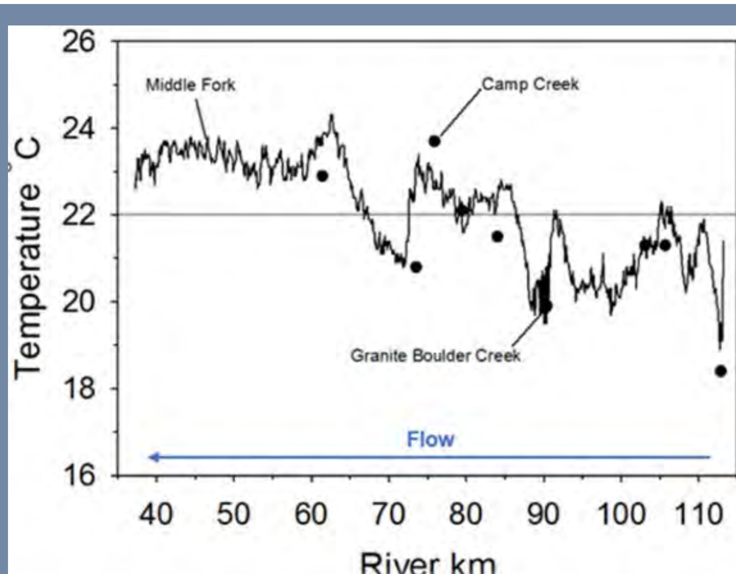
For Chinook Salmon the EDT model identified habitat diversity, sediment load, habitat quantity, temperature, and discharge as significant factors limiting productivity (CBMRCD 2005). The Upper Middle Fork Subbasin Plan identified a need for increased habitat complexity, such as areas with large woody debris (LWD). Results of monitoring and modeling exercises show that high water temperatures limit abundance and productivity of both Chinook Salmon and steelhead (Middle Fork IMW Working Group 2017), and higher than normal water temperatures in current years led to cohort failures for Chinook Salmon (Chapter 1). Population viability modeling by the Oregon Department of Fish and Wildlife (ODFW) shows that flow and temperature primarily limit adult Chinook Salmon abundance in the MFIMW (Falcy 2019).

Temperature

Salmonids are sensitive to stream temperatures above 18°C, resulting in depressed growth and survival, while sustained temperatures above 24°C have direct lethal effects (Bell 1991). The JDR basin Total Maximum Daily Load (TMDL) was approved in December 2010 to develop pollution control targets and improvement plans for impaired waters within the area (ODEQ 2010). TMDL targets of 18°C have been established for instream temperature in the MFJDR subbasins, and the UMFWG identified temperature as the most important stream attribute requiring restoration in the MFJDR. In addition to the lethal affects, high water temperatures limit fish distribution, and therefore habitat quantity, leading to density dependance (Crozier et al 2010).

Forward-looking infrared (FLIR) and fish distribution surveys conducted during 2006 on the MFJDR indicated a two-order magnitude difference in Chinook Salmon parr density between the warm mainstem (19.5°C) and cooler tributary (15°C) habitats, suggesting that parr were using cold tributaries as thermal refugia to escape stressful or lethal temperatures in the mainstem. Surface water temperatures during 2003 FLIR flights on the mainstem MFJDR exceeded 22°C throughout much of the range occupied by salmonids (Figure 6) (Middle Fork IMW Working Group 2017).

Figure 5. Longitudinal profile of surface water temperatures from thermal infrared surveys conducted during August 2003 by Watershed Sciences LLC. The horizontal line indicates the temperature where models have shown a significant decline in Chinook Salmon parr survival. Cool water tributaries are noted by black dots, note the influence of these tributaries on the MFJDR.



Surveys for Chinook Salmon in August and September 2007 revealed high pre-spawning mortality in the MFJDR subbasin due to warm stream temperatures (Ruzycki et al. 2008). In 2021, an exceptionally hot year, direct predation by river otters on sluggish adults affected by high water temperatures was witnessed by ODFW and CTWSRO crew members. In addition, monitoring and modeling conducted by ODFW in 2015-2016 show that water temperatures above 18°C significantly limit juvenile Chinook Salmon and steelhead distribution (Figure 7) (Middle Fork IMW Working Group 2017). These temperature and biological observations support evidence that temperature is a highly significant, if not the primary, limiting factor for salmonid production in the MFJDR.

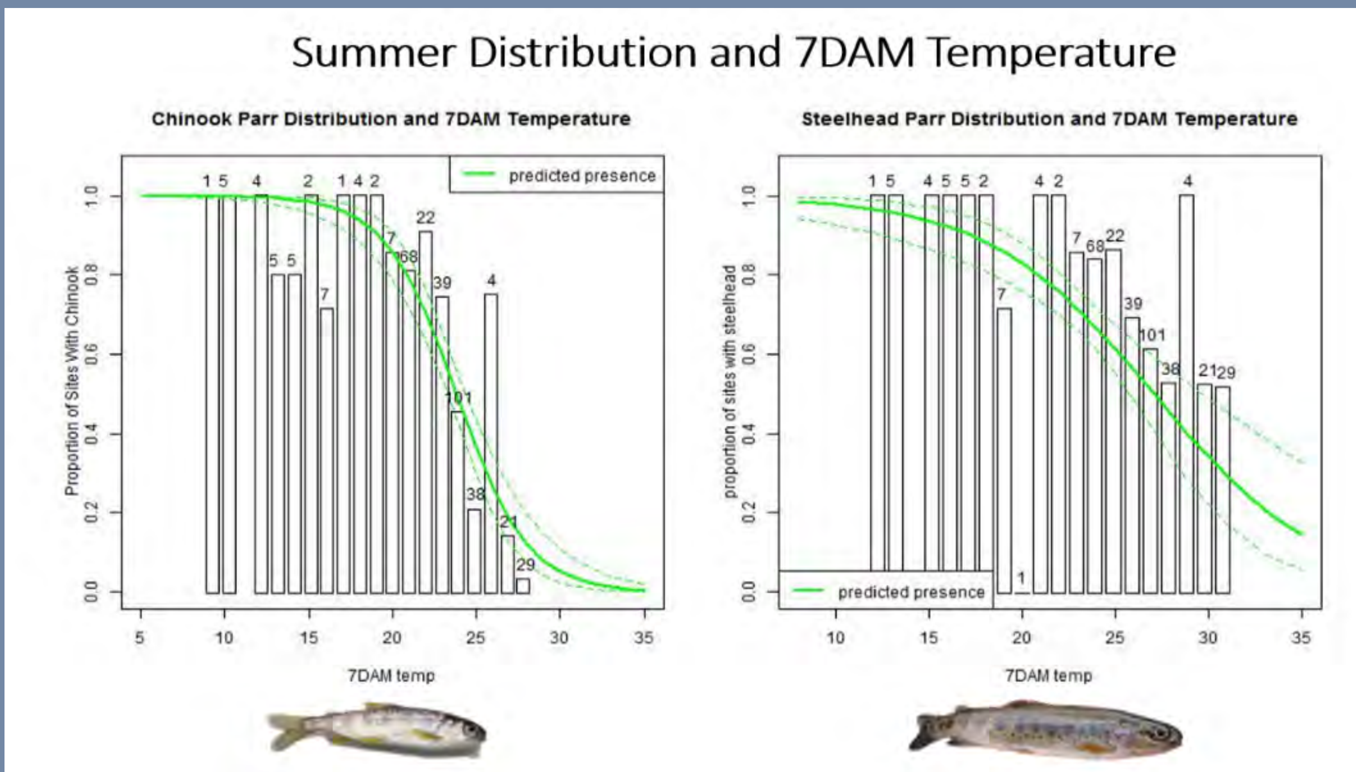


Figure 6. Summer distribution of Chinook Salmon and summer steelhead parr in the MFIMW, as a response to water temperatures

Habitat diversity and quantity

Habitat diversity refers to an array of complex habitat types supporting salmonid freshwater life stages. The distribution, dimensions, and quality of stream channel habitat units greatly affect the health of fish populations (Bjornn and Reiser 1991). Fish use pools, riffles, pocket water, off-channel backwaters, and other habitat types depending on species, life-stage, activity-level, and stream conditions. Degraded channel structure and complexity was identified as a limiting factor for salmonids in the John Day Watershed Restoration Strategy (2014); therefore, restoring habitat to encourage floodplain connection and large wood additions to the stream would improve habitat diversity. Key habitat quantity refers to the available physical area of suitable habitat required for each life stage for each species, accumulated across all life stages. Channelization of streams and rivers can affect almost all suitable habitat over the range of life stages. A major loss of just a few habitat types for some of the life stages would produce a limiting factor; for example, the loss of floodplain features would severely limit Chinook Salmon fry habitat (Chapter 3), causing density dependent issues effecting growth and survival.

Sediment load refers to increases in delivery of sediment to the stream channel. Sediment loads from erosion can increase due to land use practices, or from isolation of the channel from the floodplain, eliminating important off-channel sediment storage areas and increasing the sediment load beyond the transport capacity of the stream. Historically, clear cutting, mining, and grazing has contributed to increased delivery of sediments, embedding critical spawning gravels. Currently, actions such as logging or road construction destabilizes the landscape in high slope areas, increasing the frequency and severity of sediment loading. Increases in the frequency and magnitude of floods, wildfires, and/or loss of floodplain vegetation, will also increase erosion. Increased sediment delivery to a channel increases the proportion of fine sediments in the bed, which can reduce the survival of incubating eggs in the gravel, disrupt benthic invertebrate production, and alter hyporheic flow processes. In some areas of the tributaries to the MFJDR, sediment transport is limited and in some reaches the stream bed has been scoured to the bedrock, severely limiting fish habitats.

Altered Hydrology

Reduced summer base flow discharge contributes to elevated water temperatures and reduces available suitable habitat in the MFJDR. Both increased temperature and alterations in hydrology impact fish movement, survival, and growth. Juveniles migrating from unfavorably high stream temperatures in mainstem reaches to cooler tributary habitat are blocked during times of natural low flows or low flows due to high irrigation demands. Limited suitable habitat increases local densities in areas with cooler water inputs (Chapter 4) also limiting growth and survival (Chapter 2). Stream surveys of the distribution of salmonids in the MFJDR revealed that when mean daily stream temperatures exceed 22° C in the mainstem, juvenile Chinook Salmon either die or escape to cooler tributaries.

Credit: NFJDWC

Planting at Camp Creek after restoration to establish streamside vegetation to promote stream shading.

Restoration Efforts

Implicit in stream restoration is the notion that there is a range of reference “pre-Euro American settlement” ecosystem conditions, and that one can evaluate the degree of departure from this range in order to quantify ecosystem degradation or improvement. However, defining a specific, pristine “reference” condition for a watershed is untenable because natural disturbance processes have continually shaped river systems over time (Mann 2011). Metrics of restoration success should not be based on an imaginary static condition that once existed but instead be focused on re-establishing dynamic natural ecosystem structure and function. These functions include riparian biodiversity and natural plant community regeneration, nutrient cycling between the floodplain and channel, maintenance of natural channel morphology through hydraulic processes, and resilience to natural disturbance processes such as floods and fires (Kauffman et al., 1997; Palmer et al., 2005; Williams and Reeves, 2006). Re-establishing and maintaining these natural processes are especially important to ecosystem resilience as the Pacific Northwest faces impacts from a changing climate.

We can list limiting factors in a qualitative hierarchy to highlight that changing habitat diversity will have limited effectiveness if summer water temperatures remain too high and to guide actions into to addressing temperature (the most important limiting factor) first, and then address others once temperature is controlled.

We recognize that the MFJDR system is in a much different state than historic conditions, and massive structural changes or treatments to restore ecological processes may be needed in some reaches to effectively deal with temperature and there is interplay between factors (indicated by the curved arrows) (Figure 8).

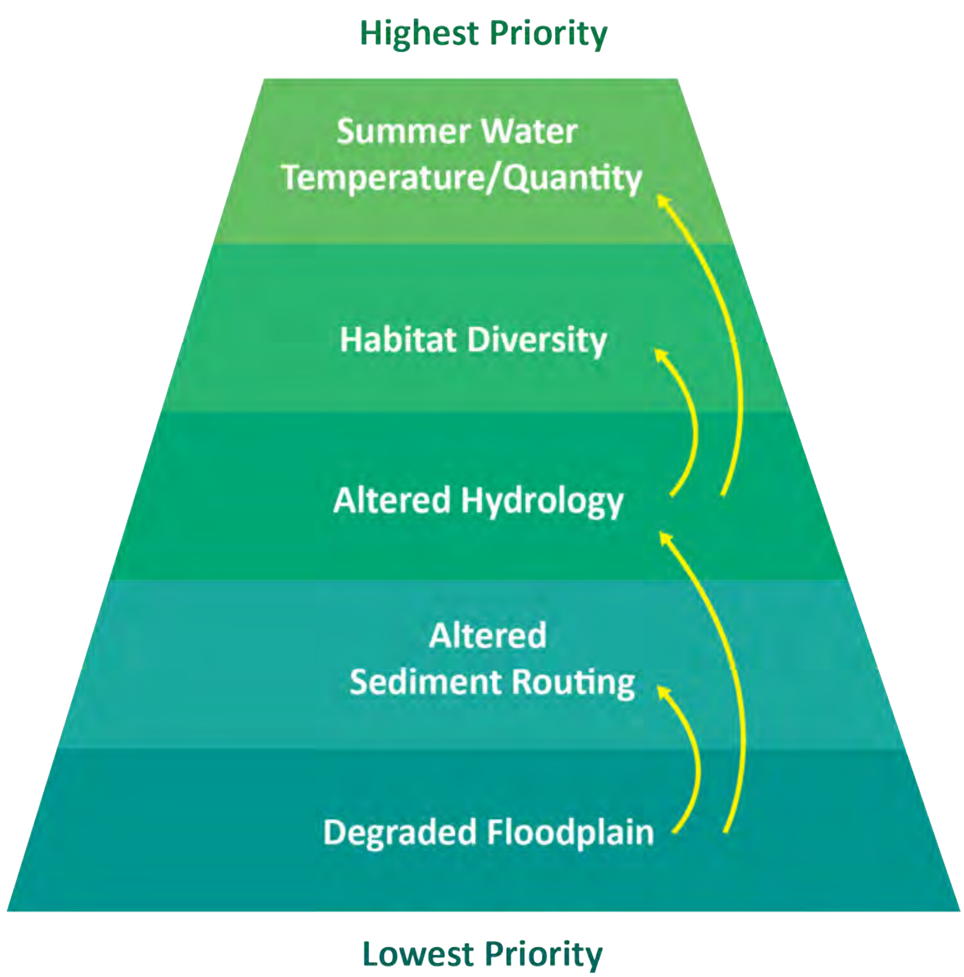


Figure 8. Hierarchy of Limiting Factors. The yellow arrows indicate the complex relationships between these factors.

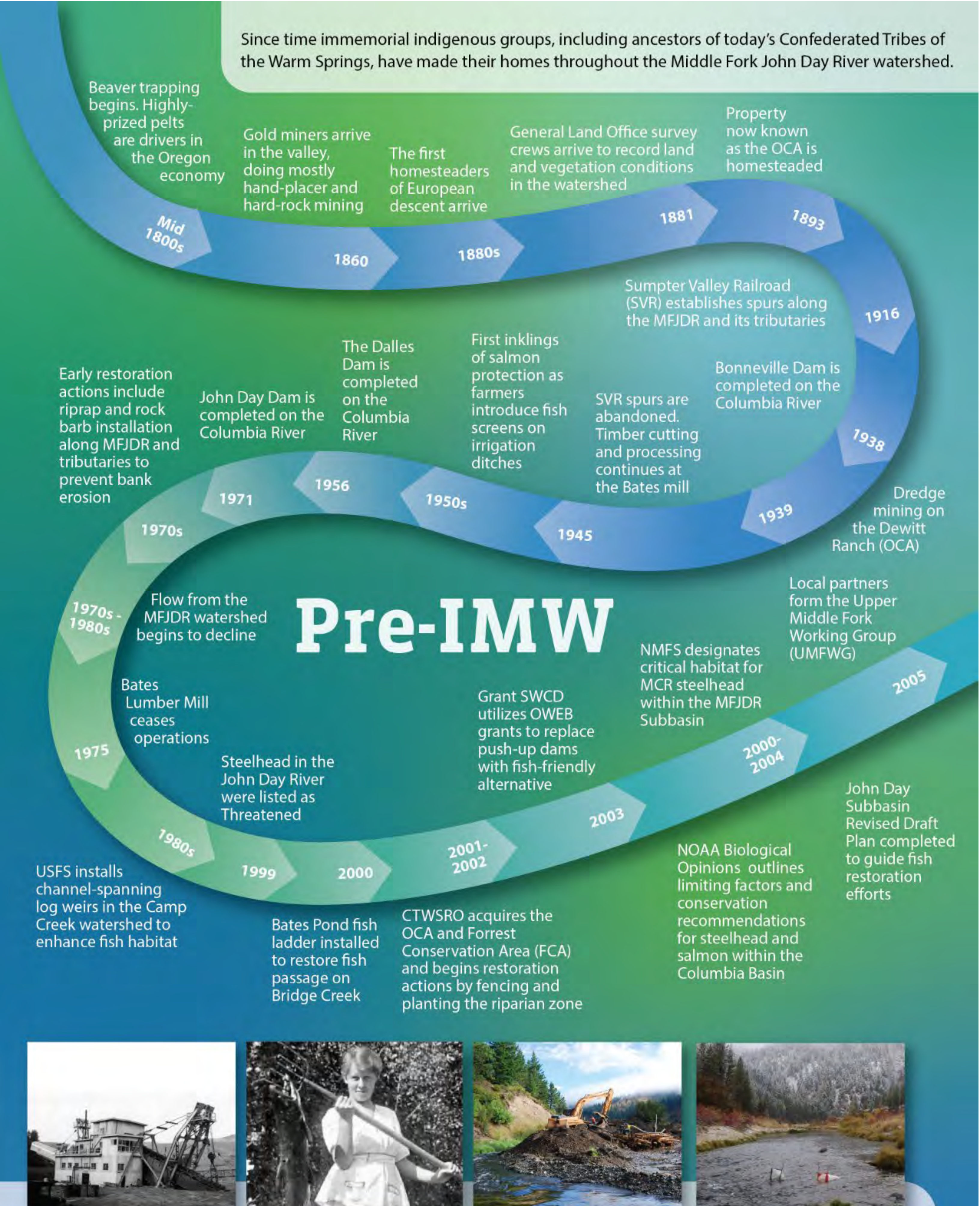


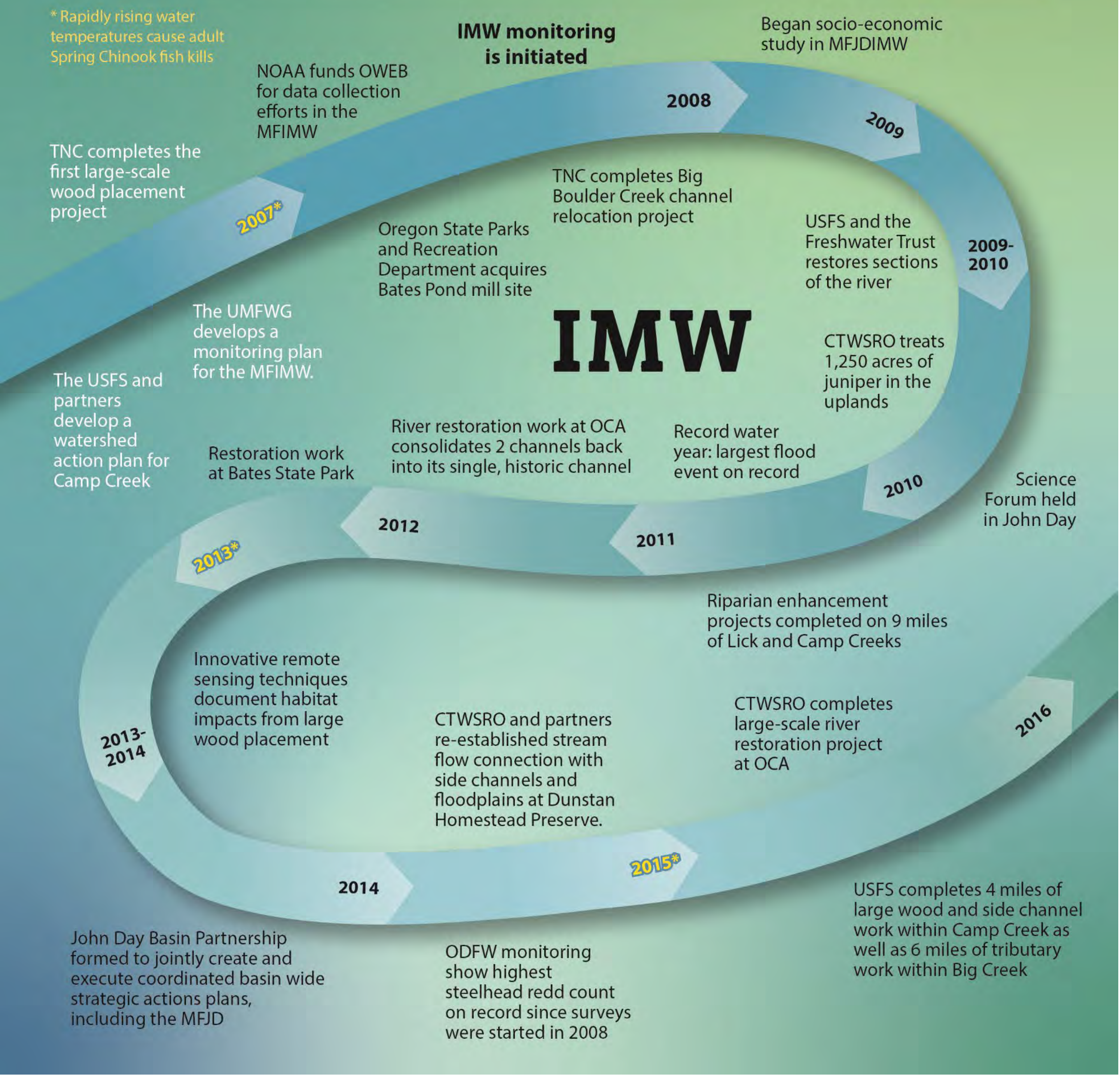
Figure 9. Timeline describing important events in the history of the MFIMW.

Middle Fork John Day River – Intensively Monitored Watershed

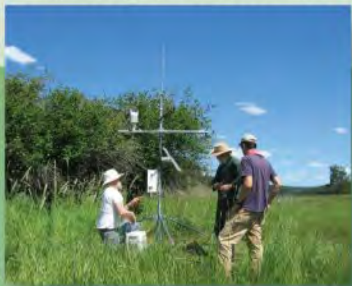
Restoration Timeline

Since the 1990s, the Middle Fork has been the focus of enormous and complex restoration efforts to repair the damage done by previous logging, gold dredging, and cattle grazing. Restoring habitat for Chinook salmon and steelhead in the Middle Fork is key to their population recovery throughout the entire Northwest region.

Steelhead in the John Day River were listed as Threatened in 1999, and ODFW survey information is available as far back as the 1960's. Fish habitat restoration is a complex process, with climate, ocean, and natural variability potentially influencing local responses. This timeline highlights important scientific and restoration milestones before and after the MFIMW was initiated in 2008.



Monitoring and restoration continues in the Middle Fork John Day River.




Butte, Ruby, Beaver Creek restoration project completed

Phipps Meadow acquired by BMLT

Vinegar to Vincent restoration completed

Initiation of juvenile Chinook genetics and dispersal project led by CRITFC

2022

2021

2023

Only 25 Chinook redds observed due to juvenile cohort "failure", restoration projects paused in mainstem MFJDR to protect adult Chinook during spawning season

Extensive parr distribution surveys conducted

Water temperature forecast model released

Heat wave restricts monitoring, Chinook prespawn mortality observed

IMW

Project at FCA initiates removal of the riprap and rock barbs installed in the 1970s and replacement by log structures and beaver dam analogs

2020

Camp Valley Restoration initiated to reconnect floodplain habitat with Camp Creek, add wood structures and plant riparian vegetation

Riparian Planting coordinated plan completed

Vincent to Caribou restoration project completed

2019-2022

2019

ODFW & CTWSRO install 7 PIT antennas in Mainstem MJFDR to track juvenile salmonid movement

Final 10-year report features accomplishments and recommendations for future restoration and monitoring in the MFIMW.

CTWSRO acquired the TNC Dunstan Preserve property

John Day Basin Partnership receives OWEB Focused Investment Partnership (FIP) funding

2017

Vincent to Caribou restoration initiated to remove riprap and install large wood structures to create pools

First John Day Basin Partnership and MFIMW joint meeting

2018

ODFW begins fry emergence monitoring to compare to data collected 40 years previously

Wiwaanaytt Creek restoration completed – including removal of log weirs to make 2.6 stream miles accessible

Report features accomplishments and recommendations for future restoration and monitoring of MFIMW.


Credit: NFJDWC
Extensive restoration on Camp Creek, including LWD installments and exclosures used to protect plants from browse.

Restoration

With new perspectives on river ecosystems have come new paradigms in restoration approaches. This new approach is characterized by reestablishing natural processes that in turn do most of the restorative work in rivers and streams (Palmer et. al. 2014). However, this involves un-doing much of the river engineering manipulation that was performed in past decades, where the primary goal was to prevent channel processes from proceeding naturally.

Entities involved in restoration designed their restoration efforts to address the limiting factors identified through multiple plans and strategies produced by partner agencies and organizations, including actions identified in the ESA Status Review and Recovery Planning, the CTWSRO John Day River Watershed Restoration Strategy and Subbasin processes, ODFW Native Fish Conservation Planning, and USFS priority watershed planning. In addition, the John Day Basin Partnership (JDBP) was formed in 2017 to pursue OWEB Focused Investment Partnership (FIP) funding to further increase coordination and funding of restoration within the John Day Basin. FIP funding was secured by the JDBP in 2018 and the MFIMW was identified as a priority watershed for restoration and monitoring funding. The JDBP also produced a strategy document to assist in identifying and vetting restoration projects for funding. Coordination between the JDBP and MFIMW exists through an annual joint meeting where monitoring and restoration project results are shared and discussed. Many members involved with the MFIMW also attend quarterly JDBP meetings.

Many passive and active restoration projects of varying size and scope have been implemented by various organizations in the MFIMW over the 15-year period since the initiation of the MFIMW. The NOAA Restoration Center defines active restoration as "on-the-ground" or "dirt-moving" activities, and passive restoration as actions that change management practices and use of landscapes. Examples of active restoration in the MFIMW include channel re-configuration, riparian plantings, and installation of log structures. Examples of passive restoration include changes in grazing management and riparian fencing. Chapter 5 describes results from long term ecological monitoring of passive and active restoration projects.

Over the last 15 years of MFIMW implementation, the scope of restoration in the MFIMW has been substantial. Restoration in the MFIMW from 2017-2022 has primarily been led, managed, and constructed by a few key partners including, the USFS, NFJDWC, CTWSRO, and the Bureau of Reclamation (BOR), with funding provided by OWEB, BPA, USFS, BOR, and PCSRF. From 2008-2016 an inventory of restoration projects in the MFIMW identified 100 projects implemented in the MFIMW during the study period, with 30 of these projects on the mainstem MFJDR and 70 in the tributaries. From 2017-2022, an additional 49 projects were completed, including 14 in the mainstem MFJDR and 35 in the tributaries (Figure 10; Appendix B). This inventory is a conservative estimate of restoration as several projects completed during 2008-2016 were not included in the inventory because we did not have complete information about the restoration actions. In addition, these restoration inventories primarily categorized and measured instream and riparian projects but did not include the hundreds of upland actions implemented by the USFS and their partners.

The restoration inventory (Appendix A) lists each restoration project completed in the past 15 years, including the lead entity, restoration activities, the construction year and total cost, if available. Based on the limiting factors described above, restoration projects were divided into 6 categories: fish passage, channel reconfiguration, instream habitat improvement, flow increase, upland management, and riparian fencing and planting. Each restoration project may address multiple limiting factors. Figure 10 and Table 1 summarize the number of projects, outcomes, project examples, and limiting factors addressed by each category of restoration activity for time periods 2008-2016 and 2017-2022. It is important to note that most restoration projects were multifaceted and consisted of several restoration strategies implemented within a given reach.

As restoration practitioners continued to learn and adapt methods to better address limiting factors, restoration strategies have shifted in the last five years. From 2008-2016 restoration actions primarily included channel reconfiguration and fish passage improvements. In the more recent five years (2017-2022) restoration projects have focused on in-stream habitat improvements (such as large-wood placement and floodplain reconnection) and riparian fencing and planting (Table 1).

Current efforts in the MFIMW uplands focus on improving the health of low-elevation dry forests, reducing fire hazard, restoring functional fish passages, improving habitat for several wildlife species, and improving stream conditions. We wish to acknowledge these upland projects as important steps towards natural watershed process reestablishment. The national Watershed Condition Framework (Potyondy and Geier 2011) addresses a long legacy of fire exclusion and timber practices that have created densely stocked stands on public forest land. The Framework aims to strategically reduce fuel loads in USFS forests by thinning forests and prescribed burning. These efforts also seek to limit insect outbreaks, reduce wildfire severity, and encourage prescribed fire use (Rainville et al. 2008; USFS 2013). Sensitive species whitebark pine (*Pinus albicaulis)* and quaking aspen (*Populus tremuloides)* are focal species of protection efforts. Stewardship contracting, which allows timber receipts to stay within the forest to fund restoration efforts, is a popular strategy on the Malheur National Forest (Rainville et al. 2008).

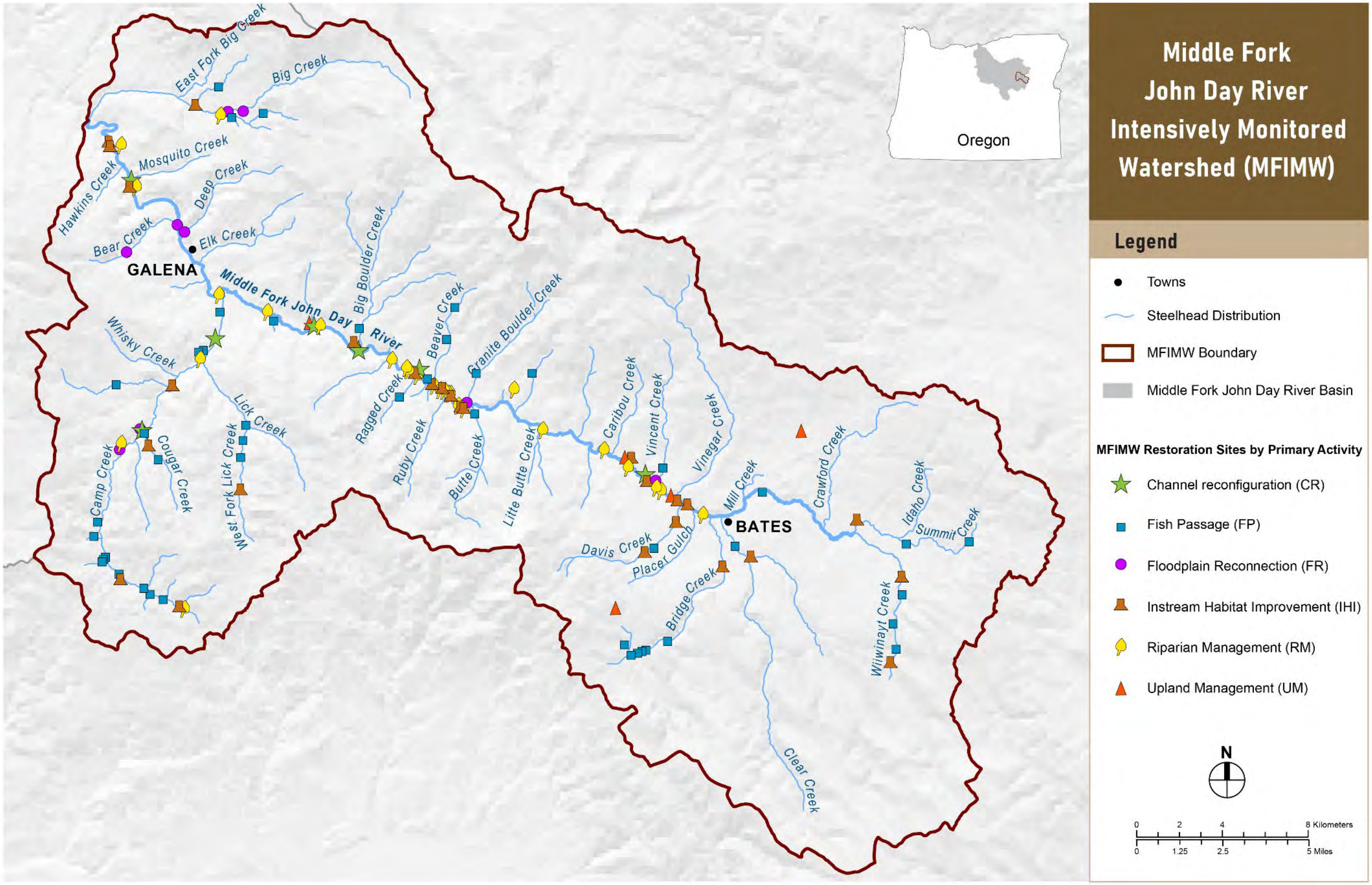


Figure 10. Map showing the location of restoration actions implemented from 2008-2022. Symbols and colors represent the primary restoration action employed at each location. While restoration projects often encompass large areas or stream reaches, for ease of viewing, projects are represented by their downstream point on this map.

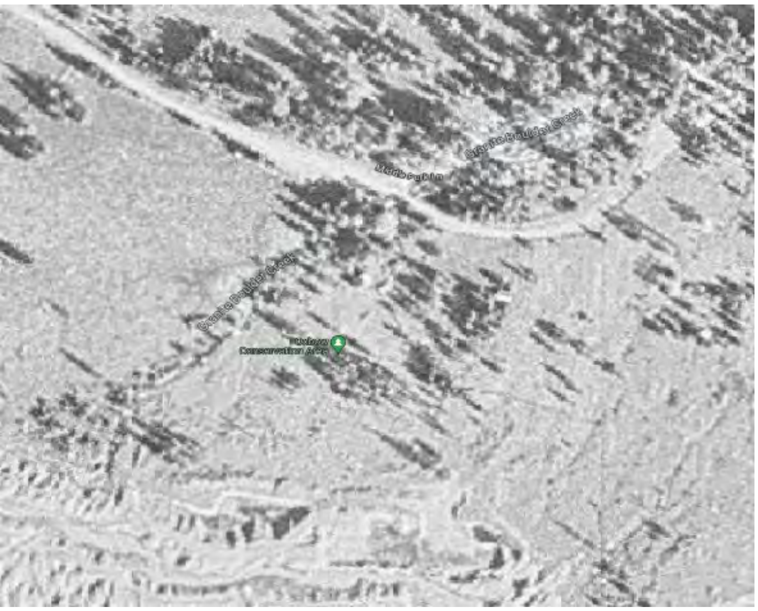
2008-2016 Total Projects*	2008-2016 Outcomes*	2017-2022 Total Projects	2017-2022 Outcomes	Restoration Action Examples	Limiting Factors Addressed
FISH PASSAGE					
44	112 mi of habitat opened or improved, including 93 mi within tributaries to the MFJDR	17	23.4 mi of habitat opened or improved, including removal of 132 barriers	Culvert removal or upgrade, bridge installation	Temperature, habitat quantity and diversity
CHANNEL RECONFIGURATION					
15	Improved over 35 mi of previously channelized stream	3	Improved over 0.6 mi of previously channelized stream	Reconnecting existing channels to old meanders	Degraded floodplain function & connectivity, habitat diversity
INSTREAM HABITAT IMPROVEMENT					
29	Installed hundreds of complex wood structures; when coupled with other actions, worked together to enhance over 35 mi of habitat	37	Installed over 3,000 complex wood structures; when coupled with other actions, worked together to enhance over 39 mi of habitat	LWD, ELJ, off channel habitat and pool development	Temperature, habitat quantity and diversity, floodplain function & connectivity
FLOW INCREASE					
16	Instream leases on 6 tributaries to the MFJDR provide over 6 cfs of water	0		Lease or sell water rights to keep water instream during critical low-flow periods	Temperature, altered hydrology
UPLAND MANAGEMENT					
4	Quantified 1,621 acres treated; there are many more upland projects implemented that we were unable to quantify	5	Quantified 12.6 acres treated; there are many more upland projects implemented that we were unable to quantify	Juniper removal, road removal or stabilization, aspen enclosures, plantation thinning	Habitat quantity & diversity, altered hydrology
RIPARIAN FENCING AND PLANTING					
25	Planted native trees and shrubs along 15 stream miles and fenced over 21 mi of riparian habitat in the MFIMW study area	22	Planted over 200,000 native trees and shrubs along 17.6 mi and fenced over 8 mi of riparian habitat in the MFIMW study area	Riparian fencing, cattle exclosures, native vegetation planting	Temperature, habitat quantity and diversity

Table 1. Summary of restoration projects implemented in the MFIMW during the 15-year study period.
*Total Projects and Outcomes reported are likely an underestimation. Outcomes were reported at varying levels of detail and some projects from 2008-2016 did not report outcomes.

Key Areas of Restoration

Several areas of the MFIMW are the focus of extensive restoration efforts, including:

- Oxbow Conservation Area and tributaries
- Dunstan Conservation Area (DCA)
- Middle Fork Forrest Conservation Area
- Camp Creek Watershed
- Phipps Meadow



Credit: Working Lands for Wildlife, Landscape Explorer.
Oxbow Conservation Area, circa 1950s.

Credit: Working Lands for Wildlife, Landscape Explorer.
Oxbow Conservation Area, circa 2020s.

Middle Fork Oxbow Conservation Area (OCA)

The OCA is a 1,022-acre property owned by the CTWSRO. The multiphase Oxbow Tailings Project was initiated in 2011 and the latest phase was completed in 2018, with another phase planned for 2025. The restoration project aimed to remediate the effects of 1940s gold dredging, in which a house-size dredge reworked all floodplain sediments and pushed the channel to one side of the valley, creating a straightened ditch flowing through a barren landscape of rock piles. To address this, the project included extensive habitat enhancement work: re-sorting dredge tailings and using them to create new, more natural instream habitat; removal/fill of existing channels that lacked fish habitat; extensive transplanting of existing vegetation; installing riparian plants; seeding; fencing; bio-engineering; and installing large wood structures instream.

Phases 1 and 2, completed in 2011 and 2012, focused on merging Granite Boulder Creek (an important cold-water tributary) directly with the MFJDR, while filling in the artificial 'North Channel' that had been created during dredging. Previously, the creek had emptied into the North Channel, which was a separate unshaded channel which caused its temperature to increase substantially before reaching the mainstem MFJDR. Phases 3, 4, and 5 were completed in 2014, 2015, and 2016, respectively. These phases focused mainly on re-meandering straightened portions of the river downstream of Granite Boulder Creek, while simultaneously enhancing and re-naturalizing the floodplain by adding large woody debris and native plants. Phase 6 implementation is currently in designs and will build upon previous work completed between Ruby and Beaver Creek. This work will potentially include the enhancement of side channels, removal of historic berms, and/or addition of willow trenches/woven structures.

The restoration actions at the OCA included construction of approximately 1.3 miles of mainstem channel and creation of more than 2,200 feet of alcoves and side channels to provide important habitat for juvenile salmonids and reconnect the river to the floodplain. The Oxbow Tailings Project included placement of 190 complex LWD structures including approximately 2,500 whole trees to enhance instream fish habitat. Dredge-mining had removed most of the topsoil, lowered the water table, and therefore made it difficult for riparian plants to recruit naturally; as part of this project, considerable efforts are being made to improve the riparian vegetation along the stream including the instillation of 8-foot-tall exclusion fencing around the project area and over 20,000 plants and 8,000 willow cuttings throughout the project area. An additional 5,740 plants are projected to be planted for Phase 6. In addition to these phase-specific plantings, the OCA has been planted by a contracting service and the CTWSRO native plant nursery team in off-project years. Click here to view a high-resolution orthomosaic of the OCA that was taken in June 2022.

This report only presents limited results for the Oxbow Tailings Project for riparian plantings and groundwater levels that CTWSRO has performed over time. Survival of plantings has been studied infrequently for the project; however, a survival analysis comparing plantings survival near Granite Boulder compared to more downstream reaches can be found in the Planting Survival section, Chapter 6.

Credit: CTWSRO

OCA before restoration, 2014.

Credit: CTWSRO

OCA after restoration, 2018.

Credit: CTWSRO.

Dunstan Conservation Area, before 1898.

Credit: CTWSRO.

Dunstan Conservation Area, circa 2010.

Dunstan Conservation Area (DCA)

The Dunstan Conservation Area is a 1,199-acre property that was acquired by TNC in 1994 and was transferred to the CTWSRO for long-term ownership and management in 2019. This property is situated approximately 3 miles downstream of the OCA and occupies 4.5 miles of the MFJDR and its tributaries. Since 2008, numerous restoration actions have been implemented to restore natural river function and processes and to enhance fish habitat in the MFJDR and its tributaries. In addition, cattle were excluded from the riparian area of the DCA since its acquisition by TNC in 1990.

The collective restoration actions at the DCA comprised treatment of more than 3,600 feet of the MFJDR, creation of two alcoves, and reconnection of two side channels that had been abandoned due to historic land management practices. Instream fish habitat complexity was improved by installing approximately 60 large wood and boulder structures to provide pool habitats and capture spawning-sized gravels moving downstream. In addition, fish passage was improved by removing one push-up dam and three concrete culverts to provide access to 0.3 miles of fish habitat.

In 2021, the historic railroad grade was breached to increase floodplain function and connectivity along the MFJDR. The historic Sumpter Valley rail line extends through the DCA, intercepting flow from the river and isolating the river from the adjacent floodplain. The railroad was breached in 18 different locations selected using drone imagery and LiDAR. There are additional phases of work on the DCA being planned currently with BOR and CTWSRO. Click here to view a high-resolution orthomosaic of the DCA that was taken in June 2022.


Credit: CTWSRO

Middle Fork Forrest Conservation Area Restoration – Vincent to Vinegar reach – pre-restoration of Vincent Creek Confluence in 2022 (left) and post-restoration of Vinegar Creek Confluence in 2023 (right).

Middle Fork Forrest Conservation Area (MFFCA)

The Middle Fork Forrest Conservation area is 787 acres (300 floodplain, riparian, wetland, and 487 timbered uplands) owned and maintained by CTWSRO and is located one mile west from the junction of Highway 7 and County Road 20 along the MFJDR. The MFFCA encompasses confluences of 5 cold water tributaries (Davis Creek, Vinegar Creek, Vincent Creek, Dead Cow Creek, and Caribou Creek). Restoration on this property started in 2017 and is ongoing today.

Starting in 2017, Phase I of the Vincent to Caribou project was completed on 1.9 miles of the MFJDR. This project removed over 3,000 cubic yards of riprap, added 400 cubic yards of seed gravel, constructed 38 log structures, added three beaver dam analogs, and created or enhanced 17 pools. Structures consisted of apex jams, forced pool log structures, pivot log structures, pocket log structures, and sweeper log jams. Post-project, approximately 2,200 rooted riparian shrubs species were planted along the mainstem in the Vincent to Caribou Phase I zone in November of 2017.

In 2020, Phase II of the Vincent to Caribou project was completed. Phase II of the project consisted of installing over 80 large wood instream habitat structures, removing 8,000 cubic yards of a historic railroad grade that bisected the floodplain, removing 1,200 cubic yards of riprap and rock barb, placing 700 cubic yards of instream gravel, and excavating 22 pilot channels. Native riparian grass seed was placed in all disturbed areas. As part of this project, in 2021 three exclosures were installed encompassing the project, protecting 77 acres of land that was planted with 29,200 plants in the fall of 2021.

Phase I of the Vincent to Vinegar Project was also implemented along 0.9 miles of the MFJDR in the fall of 2020. Phase I work was primarily restricted to the south side of the historic Sumpter railroad grade. A series of primary, secondary, and tertiary channels were excavated, and large wood was placed throughout the channels. 3,800 linear feet of Willow trenches were implemented throughout the floodplain to create floodplain roughness, helping to slow and retain overland flows which will help recharge the groundwater. A total of 11,233 feet of main channel and 5,111 feet of secondary and tertiary channels were excavated as part of this project. Approximately 60 pools were created, 180 cubic yards of spawning gravel were added to the channel in the form of constructed riffles, 378 anchored individual logs were placed, and over 35 logjams were placed within the channels. An 8-foot-tall wildlife-excluding fence was installed surrounding the first phase of the project, which will protect the over 23,000 new plantings that were installed within the project area, allowing them to become established and over time grow above browse height.

In 2021, Vinegar to Vincent phase 1.5 was completed. This project included floodplain complexity work that was completed as part of phase I in 2020. For phase 1.5, 48 floodplain complexity features (e.g., flood fences) were installed throughout the floodplain of the project area. These structures consisted of willow bundle posts and wood posts. These structures function to help slow and retain floodplain flows, helping to recharge groundwater. These structures were woven in 2022.

Additional work completed in 2021 included a small instream project on Dead Cow Creek, a tributary to the MFJDR. The purpose of this work was to provide a low-tech and minimally invasive approach to improving habitat across approximately 0.3 miles of stream using readily available materials (i.e., small juniper trees/limbs removed from CTWSRO properties not exceeding a 6-inch DBH). Wood was not to be buried or anchored, but rather surface placed to provide shade and habitat complexity. An estimated 30 small trees/limbs were placed within this project area. Fencing that was currently in place was removed and this project was enclosed in fencing that was completed on this section of the MFJDR and Dead Cow Gulch in 2021.

In 2022, Phase II of the Vinegar to Vincent Project was completed along 0.9 miles of the MFJDR. The Vinegar to Vincent Fish Habitat Improvement project removed the railroad grade along the MFJDR, rerouted the channel, introduced large woody debris, rebuilt and lengthen a cattle exclusion fence, and reactivated historic side channels and floodplains. By restoring this reach to a more natural condition, habitat was improved for not only anadromous salmonids and life stages, but for lamprey, resident rainbow trout, bull trout, other native fish species, and freshwater mussels. Project components included the excavation and reactivation of new and historic main and side channels. 6,706 feet of side channel was added, 333,854 feet of new main channel was added, and 3,432 feet of old main channel was filled. The floodplain was enhanced through the removal of 4,300 feet of the railroad grade. The removal is expected to provide inundation during spring flows at 2+ year events across 54 acres of floodplain. 4,520 feet of willow trenches were implemented throughout the floodplain to create floodplain roughness, helping to slow and retain overland flows which will help recharge the groundwater. Approximately 105 pools were created, 433 logjams were installed, 8,500 square yards of sod was salvaged and transplanted into the project and, 170 y^2 of spawning gravel was installed in the form of constructed riffles. The 444,224 feet of temporary fencing along the railroad grade was removed and the remainder of the 555,016 feet of 8-foot exclusionary fencing was completed in the fall of 2022 encompassing the entire project. This project was planted in the fall of 2022 with 26,000 plants installed inside the exclosure on the Vinegar to Vincent Project. Seeding and mulch were applied to all disturbed areas to prevent erosion and limit spread of invasive species. Click here to view an orthomosaic of the Vinegar to Vincent Project that was taken in June 2023.

Credit: NFJDWC

Buck and pole exclosure fencing on Camp Creek.

Credit: NFJDWC

Aerial view of the Camp Creek Valley restoration area.

Camp Creek Watershed

Camp Creek is a major tributary sub-watershed (40,294 acres) within the MFIMW that hosts steelhead as well as juvenile Chinook Salmon rearing habitat and is predominantly within USFS boundaries. The USFS and many partners identified critical limiting factors affecting steelhead from 2004-2008 and developed the Camp Creek Watershed Action Plan (USFS 2008). This action plan identified biological and hydrologic function degradation from past management activities including logging, road placement, beaver trapping, and past overgrazing. Some of the first restoration actions included replacement of stream culverts that had previously impeded fish passage and cut off access to habitat. Another action was removal of 151 legacy log weirs that were installed in the early 1980s along 11 stream miles. Removal of these legacy log weir structures in Lick and Camp Creeks accelerated restoration of channel structure and complexity to improve spawning and rearing habitat. Along with other identified priority fish passage, road, and channel/riparian improvement projects, these actions were key in improving sustainable fish population viability and overall watershed health in Camp Creek.

From 2020-2022 an extensive planting and fencing project managed by the NFJDWC and USFS focused on bolstering the cold water refugia potential of Camp Creek by increasing streambank shading and plant density in thermally sensitive areas. Concurrent with the planting and fencing project, the NFJDWC and USFS implemented a multi-year research study of planting exclosures and plant success. This monitoring effort is still underway, and findings will be presented at a later date.

Beginning in 2024 the NFJDW, with USFS, and funding through the John Day FIP and USFS will implement the Camp Creek Reach 1 Restoration. The project proposes to fill in the existing channel and reconnect all the multiple threaded channels across the valley. Large wood and small wood as well as slash will be placed throughout the channels and embedded within floodplains as well as any cut or fill areas. Beaver dam analogs (BDAs) and post-assisted log structures (PALs) will be utilized to lock wood in place and facilitate aggradation, water storage, increase habitat complexity and cover for juvenile fish and beaver.


Credit: Blue Mountain Land Trust.


Credit: Blue Mountain Land Trust.

Phipps Meadow, acquired by BMLT in 2022, supports a diverse range of species. Though restoration work focuses on benefits to fish, riparian revegetation efforts will not only decrease thermal loading on the stream but will also increase habitat and food availability for terrestrial species including a resident beaver population.

Phipps Meadow

Phipps Meadow is an ecologically diverse 278-acre property surrounded by U.S. National Forest, composed of wetland meadow, pine forest, and 1.58 stream miles in the headwaters of the MFJDR. It includes the confluences of Crawford Creek, Summit Creek, and Little Phipps Creek. The meadow contains a network of freshwater springs that feed into the MFJDR providing warm water in the winter and cold water in the summer, with temperature spikes buffered by these cool water sources (CTWSRO unpublished data). The JDBP identified that restoration of the Phipps Meadow property would greatly benefit salmon by improving spawning and rearing habitat. Historically, cattle were heavily grazed on the meadow accessing the river, streams, and springs for water. Phipps Meadow was purchased by BMLT in 2021 through funding administered by OWEB. Cattle grazing no longer occurs on the property; however, the property is home to elk and deer, with a large herd of elk utilizing the property in the winter months. Restoration designs are currently being drafted to restore stream channel form and revitalize the riparian vegetation. Buck and pole exclosures were recently installed around portions of the stream to protect already established plants. Several members of the JDBP meet regularly to plan and eventually execute restoration actions and monitoring.

Monitoring Objectives and Actions

The objectives of the MFIMW are to evaluate the impact of the combined restoration actions on anadromous salmonid populations and to understand how specific types of actions impact habitat and fish metrics at the watershed, sub-watershed, reach, and restoration project scale. Within the MFIMW, several types of restoration and monitoring actions have been implemented over a range of time frames. Given this complexity, a hierarchical design framework has been used to evaluate the study objectives through multiple research projects. The hierarchal framework included a watershed scale evaluation of restoration actions and nested experiments within the larger framework that targeted specific restoration actions. Table 2 and Figure 11 describe the scale of inference for each type of monitoring and/or modeling conducted from 2004-2023 and, if applicable, what type of restoration actions the monitoring focused on. This summary report focuses on monitoring and research that has occurred in the 2017-2022 time frame, and where research was voluntarily analyzed and written into chapters. Monitoring projects marked with an asterisk in Table 2 are discussed in research chapters. Monitoring and research results pre-2017 are comprehensively described in the 2017 10-year Summary Report.

Table 2. Description of the scope of inference for monitoring or modeling activities completed as part of the MFIMW and, if applicable, what type of restoration actions the monitoring targeted. Monitoring projects marked with an asterisk in Table 2 are discussed in research chapters.

MONITORING	RESTORATION ACTIONS MONITORED	SCOPE OF INFERENCE	DATE RANGE
Remote Sensing			
Drone Imagery	Multiple actions throughout watershed	Watershed, Reach, Project	2020-2023
LiDAR	Multiple actions throughout watershed	Watershed	2006, 2009, 2021
FLIR	Multiple actions throughout watershed	Watershed	2003, 2020
Fish			
Adult Spring Chinook Salmon*	Multiple actions throughout watershed	Watershed	1958-2023
Adult Steelhead*	Multiple actions throughout watershed	Watershed	2004-2023
Juvenile salmonid outmigrant monitoring (rotary screw trap) *	Multiple actions throughout watershed	Watershed	2004-2023
Chinook Salmon and Steelhead productivity*	Multiple actions throughout watershed	Watershed	2004-2023
Juvenile Chinook Salmon Floodplain Use + Parentage*	Floodplain reconnection, Channel reconfiguration	Reach, Project	2021, 2023

MONITORING	RESTORATION ACTIONS MONITORED	SCOPE OF INFERENCE	DATE RANGE
Juvenile steelhead abundance & survival (Recent data for Camp, Vinegar, Lick, Davis, older data for MFJD, Granite Boulder)	Log weir removals	Sub-watershed	2008-2023
Chinook Salmon Fry Emergence*	Multiple actions throughout watershed	Watershed	1980s, 2019, 2021-2022
USFS Aquatics Monitoring	Upland, riparian management	Project, Reach	2004-2023
Mainstem PIT arrays (Galena + Ritter)	Multiple actions throughout watershed	Watershed	2009-2023
Bates Pond Fish Passage	Channel reconfiguration	Project	2014
Summit Creek juvenile steelhead movement	BDA, Floodplain reconnection	Reach, Project	2018-2019
Spring Chinook Salmon parr distribution (upper MFJD + some tribs)	Instream habitat improvement, riparian fencing, and planting, channel reconfiguration	Sub-watershed	2008-2020
Spring Chinook Salmon parr survival	Instream habitat improvement, riparian fencing, and planting, Channel reconfiguration	Sub-watershed	2011-2015
Juvenile salmonid movement + PIT Arrays	Floodplain reconnection, channel reconfiguration	Watershed	2018-2021
Steelhead Life cycle model	Simulation of riparian fencing and plantings, and instream habitat improvement	Watershed	2015
Habitat			
CTWSRO Habitat Surveys	Riparian plantings, instream habitat improvement	Reach, Project	2023
PIBO	Instream habitat improvement, riparian fencing, and planting, channel reconfiguration	Watershed, Sub-watershed (Camp Creek)	2009, 2014, 2019,2024
Geomorphology (Pat McDowell, et al.)	Instream habitat improvement, riparian fencing, and planting	Watershed	2009-2016
Physical Habitat*	Active/passive restoration	Reach	1980, 2019
Greenline Habitat*	Active/passive restoration; Riparian plantings, floodplain reconnection	Reach	1980, 2019, 2023

MONITORING	RESTORATION ACTIONS MONITORED	SCOPE OF INFERENCE	DATE RANGE
CHaMP Habitat	Instream habitat improvement, channel reconfiguration	Reach, Watershed	2013-2019
Suneye (Watershed)	Riparian plantings	Watershed	2018
Riverstyles	Multiple actions throughout watershed	Watershed	2013
EMAP Aquatic Inventories	Multiple actions throughout watershed	Watershed	2004-2007
Macroinvertebrates			
Macro benthic*	Multiple actions throughout watershed	Watershed, Reach, Project	2010-2023
Macro drift*	Multiple actions throughout watershed	Watershed, Reach, Project	2010-2023
Macroinvertebrate abundance/diversity	Active/passive restoration	Reach	1980, 2019
	Socio-economic		
Socio-Economic	Specially developed set of metrics to reflect outcomes and community economic health	County-wide impacts of watershed-wide restoration in MFJDR	2007-2017
Vegetation			
Exclosure Fencing – Vegetation monitoring	Riparian plantings	Project	2022-2023
Vegetation Monitoring*	Riparian plantings, floodplain reconnection	Project	2012-2023
Wild ungulate browse impact on plant growth and survival	Riparian fencing and plantings on CTWSRO conservation areas	Reach	2009-2010
Vegetation state-and-transition models	Simulation of riparian fencing and plantings, and instream habitat improvement	Reach	2016
Water Quality			
Discharge	Flow increases	Watershed	2014-2023
Water Temperature*	Multiple actions throughout watershed	Watershed	2002-2023

MONITORING	RESTORATION ACTIONS MONITORED	SCOPE OF INFERENCE	DATE RANGE
Camp Creek Gauge	Flow increases	Watershed	2006-2023
Groundwater*	Channel reconfiguration, instream habitat improvement, and floodplain reconnection	Reach, Project	2008-2023
Meteorological	Baseline monitoring	Watershed	2012-2018
Bates Pond Water Quality	Channel reconfiguration	Project	2017-2021
Water Temperature	Distributed Temperature Sensing (DTS)	Project and sub-reach	2008-2015
Water Temperature	Water temperature modeling (HeatSource)	Simulation of riparian planting	2016
Hydro-thermal stream monitoring	Multiple actions throughout watershed	Reach	2008-2016

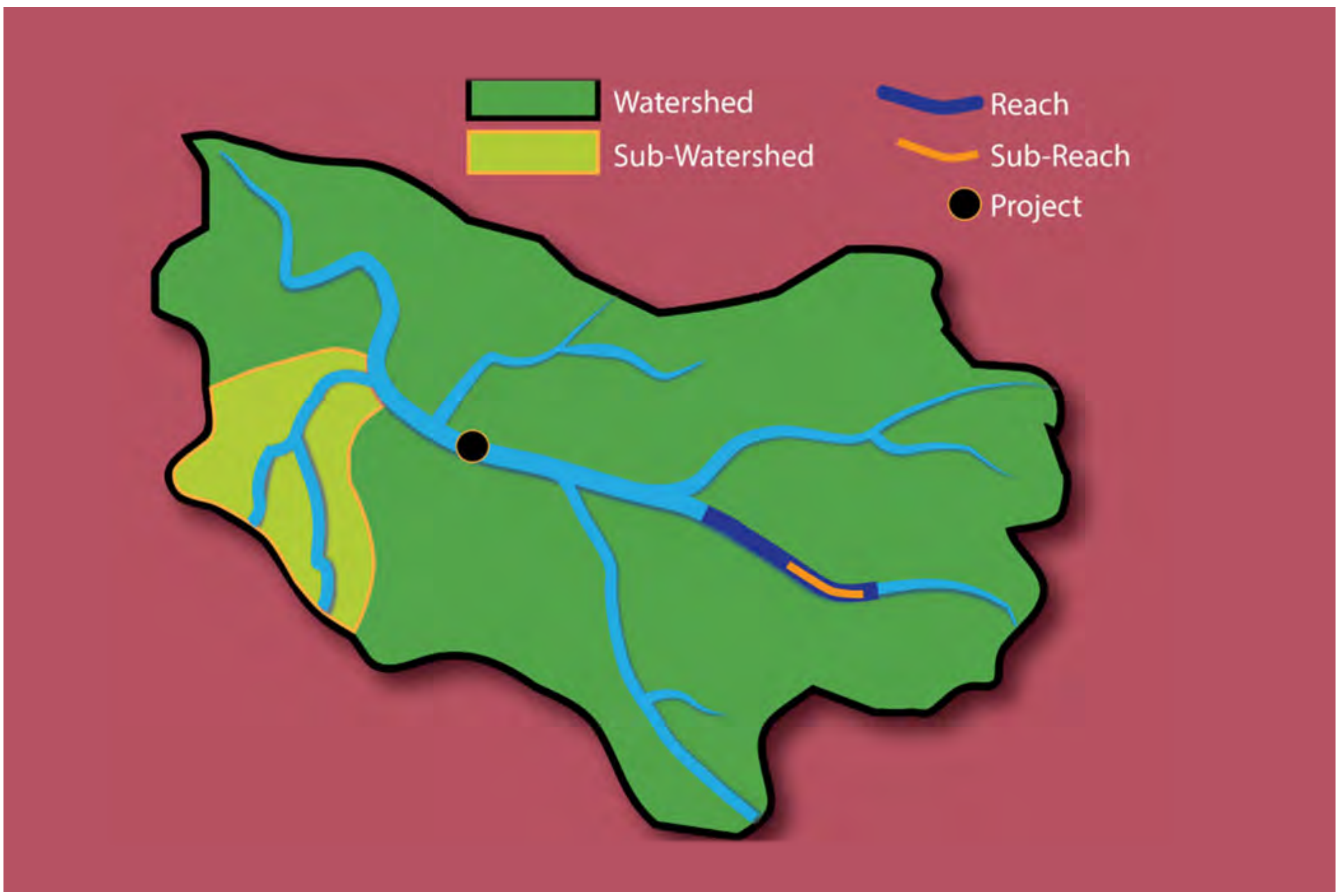


Figure 7. Scales of inference for monitoring projects in the MFIMW.

Credit: BMLT

Contractors compile plant species inventory in Phipps Meadow, upper MFJDR upon acquisition by BMLT.

Chapter Summaries

Partners within the MFIMW have undertaken many monitoring and research projects since 2008. Research results for projects focused in years 2008-2016 can be found in the 10-Year Summary Report. For years 2017-2022, eight research project reports were completed. For a brief overview of the research reports abstracts are listed below. For in-depth reading of complete reports, please see (Chapters 1-8). While each report can be a stand-alone report, read together with the entire report, they give a picture of the more recently studied ecological and biological processes in the MFIMW.

Credit: ODFW

Spawned-out Chinook Salmon carcass sampled during spawning ground surveys on the MFJDR.

CHAPTER 1 Abstract

Watershed-scale Chinook Salmon and Steelhead Abundance and Productivity

We monitored the watershed (population) scale response of steelhead *Oncorhynchus mykiss* and spring Chinook Salmon *O. tschawytsha* to restoration actions in the MFJDR. Monitoring included measures of abundance and productivity for both juveniles and adults. Results for steelhead at the watershed scale show density dependance at the juvenile life-stage, likely due to limited juvenile habitat rearing habitat, is negatively affecting steelhead recovery. Chinook Salmon populations experienced very low adult and juvenile abundances in multiple years, likely due to environmental conditions including high water temperatures and low flow, which negatively affected productivity. However, densities of Chinook Salmon redds increased in restored areas, despite no significant change in overall redd counts across the MFJDR indicating a distributional response to restoration.

››› (read the full chapter)

Credit: ODFW

PIT-tag array and solar panels on Granite Boulder Creek.

CHAPTER 2 Abstract

Quantifying riverscape productivity to inform limiting factor analysis and guide reach-based restoration goals

Effective species conservation requires addressing threats associated with limiting factors operating across spatial and temporal scales. For Columbia Basin salmonids, limiting factors operating at the population scale are well understood, but information on how limiting factors impact a population at smaller spatial scales and across a riverscape is lacking. To identify spatially varying population limiting factors we quantified the spatial variation in Chinook Salmon and steelhead productivity metrics and habitat variables across mainstem rearing habitats. We then evaluated the relationship between productivity metrics and habitat variables. We found density-dependent and density-independent processes were limiting the population but parr-to-smolt survival, the most valuable indicator of progress towards recovery, was most strongly correlated with density-independent processes, mainly temperature ($R = -0.58$). Monitoring across the riverscape revealed the impact of temperature on productivity was not spatially uniform and a central zone of temperature impact existed. Our results suggest restoration effectiveness will be maximized when efforts are successful at alleviating temperature limitation in the central zone of temperature impact.

››› (read the full chapter)

Credit: ODFW

Sampling for Chinook fry in an inundated floodplain on the MFJDR.

Credit: ODFW

Chinook Salmon constructing a redd on the MFJDR.

CHAPTER 3 Abstract

Emergence and Dispersal Patterns of Spring Chinook Salmon Fry in the Middle Fork John Day River

There is strong selection on salmon emergence timing to maximize survival through alignment with long-term patterns of optimal rearing conditions, which could be disrupted by climate change induced shifts in temperature, flow, and precipitation patterns. Timing of emergence relative to high flows, available floodplain habitat, and distance dispersed from redds could have profound effects on patterns of growth and survival of juvenile salmon. We sampled emerging Spring Chinook Salmon (Oncorhynchus tshawytscha) fry from sites distributed across the mainstem of the MFJDR from 2019 to 2022 to determine emergence windows and to characterize longitudinal and lateral dispersal from redds using genetic parentage assignments to link individuals to their maternal parents spawning location. Sampling occurred within areas affected by in-stream and floodplain restoration that are in various stages of recovery and implementation. We found that the fry emergence window occurred from mid-March to mid-May, which has not changed from a similar study completed over 40 years ago. Our results indicate a slight declining trend in annual cumulative thermal units during Chinook Salmon egg incubation period, although long-term temperature records are limited. We also found that fry only dispersed in the downstream direction, with the median recorded dispersal at 0.8 km (95% range: 0.05 - 12.6 km), although many fry dispersed much further and up to 20 km. We also found that restoration stage and variable flow patterns, can affect the type of habitat that fry utilize. Most fry dispersed less than a kilometer from where they were born and most heavily utilized floodplain habitat that was adjacent to the main channel. These findings highlight the importance of considering the arrangement of spawning and rearing location for restoration planning and implementation, suggesting that efforts targeting the fry life stage may be most effective just downstream of concentrated spawning. »» (read the full chapter)

Credit: ODW

Juvenile Chinook Salmon being measured and assessed for PIT-tagging.

Credit: ODFW

Large woody debris installment on the MFJDR.

CHAPTER 4 Abstract

Juvenile Chinook Salmon Dispersal Patterns Across the Middle Fork John Day Watershed

We assessed juvenile Chinook Salmon dispersal patterns across the MFJDR, in which parr sampled from rearing habitats in summer 2021 were traced back to potential adults sampled from spawning locations in 2020 using genetic parentage assignments. We estimated that 68% of all parr dispersed downstream (n = 1,326) and that 25% dispersed at least 3.7 km downstream, whereas 25% were estimated to have dispersed more than 0.9 km upstream. Dispersal patterns varied across the watershed, with parr originating farther upstream generally exhibiting more downstream bias, greater variability in dispersal, and farther dispersal distances compared to parr originating lower in the watershed. Parr originating in areas with higher maximum summer stream temperatures generally dispersed to cooler sections of the mainstem or tributaries, suggesting temperature was a primary driver of dispersal in 2021. Overall, these findings suggest farther dispersal at early life-stages than prior published estimates, and that longitudinal temperature patterns and the configuration of accessible cold-water tributaries along river profiles contribute to variability in dispersal patterns within and among watersheds.

››› (read the full chapter)

CHAPTER 5 Abstract

Long-Term Effects of Passive and Active Restoration in the Middle Fork John Day

The objectives of this project (OWEB #218-6041) were to quantify long term changes in 1) floodplain and greenline (streamside) vegetation and 2) in-stream geomorphology and habitat relative to different restoration strategies. We quantified change by re-measuring sites in 2018-19 that were first monitored in 1996-7 under a different project, and by comparing aerial imagery between 1989 and 2017. We compared change across five management classes: Class 1 represented ongoing livestock grazing and Classes 2-5 represented various combinations of active and passive (grazing reduction or cessation) restoration implemented after 1996. Floodplain vegetation types did not show significant changes in area over time or by management class, but an increase in riparian woodlands and a reduction in gravel bar area was suggested. In contrast, greenline vegetation showed clear changes across classes:

Credit: ODFW

Torrent sedge (C. nudata), an important riparian plant, acting as a vegetative boulder.

communities transitioned from mesic grasses toward deep-rooted sedges and other hydric species. Consistent with the establishment of more wet-adapted species closer to the water's edge, greenline-to-greenline channel widths narrowed across classes, and the full passive and passive + active classes showed greatest narrowing. Channel complexity as measured by number of habitat units per km increased in most full passive and passive + active reaches. Large wood loading increased in the two passive + active reaches sampled, due to placement of large wood in active restoration projects. Other geomorphic metrics – residual pool depth, percent channel length in pools, substrate size – did not show consistent patterns of change. These results indicate that both passive and active restoration can show positive effects on aquatic riparian habitat, but effects may take decades to be evident, and changes proceed at different rates for different processes. Furthermore, while active restoration projects may jump-start certain processes such as large wood accumulation, passive restoration can drive systemic changes such as greenline vegetation change and channel narrowing. »» (read the full chapter)

Credit: CTWSRO

CTWSRO technician conducting planting efficacy studies on plants buried by invasive birdsfoot trefoil.

CHAPTER 6 Abstract

Planting Efficacy and Ground Water Monitoring on the Middle Fork Oxbow Conservation Area

The Confederated Tribes of the Warm Springs Reservation of Oregon have made significant investments in restoring riparian conditions and monitoring groundwater in the MFJDR, with many of these efforts focused on the Tribes' Oxbow Conservation Area. To assess planting success, two separate planting efficacy studies were conducted on this property. The original 2012 study enumerated all woody stems in established cross-sections along the riparian, which included recently installed plantings and existing woody stems. A subsequent 2021 study used real-time kinematic positioning equipment to electronically tag 330 installed plantings along the riparian to track survival. Groundwater elevation assessments used data from six wells in proximity to planting locations to evaluate changes in patterns and trends in groundwater levels pre-and post-implementation of restoration actions. The 2012 planting efficacy study showed variation in survival and additional recruitment within monitoring plots, whereas the 2021 study showed little survival of installed plants, with almost a fifth of the plants being lethally browsed by small rodents within the first-year post-installment. Groundwater

Credit: CTWSRO.

Willow trenches on the floodplain of the MFJDR Vincent to Vinegar reach.

elevation analyses showed mixed results, with only some well locations showing improved water elevation post-restoration. Two lessons learned from these monitoring efforts that are potentially easiest and most impactful to address are 1) protection of established plants may result in quicker revegetation of the stream than installing new plants and 2) fine-meshed rodent exclusionary fencing may be a necessary addition to protect newly installed plants from small-animal browse, especially when plants are sparse and immature. The groundwater elevation analyses highlighted the importance for continuous datasets to monitor water elevation over time as it relates to restoration monitoring. Restoration practitioners are urged to consider well locations during future project installations. ››› (read the full chapter)

Credit: NFJDWC

Javan Bailey (NFJDWC) collecting benthic macroinvertebrates samples on the SFJDR.

CHAPTER 7 Abstract

Middle Fork John Day IMW Macroinvertebrate Community Analysis

These analyses focused on detecting long-term trends in drift and benthic macroinvertebrate data, followed by a “before-after” restoration analysis at each site. Little agreement in trends or changes occurred between drift and benthic results at co-located sites in the MFJDR, and no consistent relationship was seen between restoration intensity and macroinvertebrate community response. A general lack of consistent temporal trends or consistent pre/post-restoration changes in benthic and drift communities suggests that ecological conditions have remained largely unchanged in the MFIMW over the 2010-2022 monitoring period. The drift data exhibit some trends and pre- versus post-restoration changes, but the limited utility of drift data is discussed. Benthic data indicate positive post-restoration changes in ecological conditions at only two of 10 sites (MF-2, MF-3); one of these (MF-2) is co-located with a drift site that also showed relatively consistent evidence of improved conditions (D 003). Continued monitoring of the benthic community at both MF-2 and MF-3 should reveal whether these apparent ecological changes will persist as a result of restoration efforts or if they are related to other drivers and will continue to vary. We recommend discontinuing drift sampling and adding physical habitat assessment and continuous temperature monitoring to the benthic sampling to produce a more robust data set to facilitate detection of potential drivers of observed ecological change over time. ››› (read the full chapter)

Credit: NFJDWC

Water temperature logger secured to submerged brick attached rebar on Tin Cup Creek.

CHAPTER 8 Abstract

Freshwater Temperature Trend in the Intensively Monitored Watershed of the Middle Fork John Day River

Stream restoration is a rapidly maturing field and effectiveness monitoring is critical for informing restoration design, evaluating restoration success, and identifying adaptive management opportunities. Stream temperature is a driver of many ecological processes in aquatic environments and has been identified as a common limiting factor for juvenile salmonids in the Pacific Northwest. This study investigated temperature trends at 86 in-stream temperature monitoring locations in the Middle Fork John Day River, Oregon – a watershed which has been the subject of intense restoration and monitoring efforts over the last 15 years. I performed trend analysis for the months of July, August, and September for data available within and between 2005 – 2021 using response metrics of total degree hours and degree hours above the temperature threshold causing stress to juvenile salmonids. These two metrics were examined using both unadjusted values and by adjusting values for annual variation in streamflow and air temperature. Many sites did not exhibit significant trends during the period of record. Significant trend results for unadjusted temperature metrics were dominated by tributary locations, had a relatively even distribution between increasing and decreasing trends, and fewer decreasing trends located in restoration reaches compared to unrestored reaches. Significant trends for flow and air temperature adjusted metrics were more evenly distributed between mainstem and tributary locations, were mostly decreasing, and a greater proportion of trends were located in restoration reaches. The relatively small number of significant trends observed, compared to the number of tests performed, indicates that the system has minimal increasing or decreasing trends over the period of record. Tributary systems may be more sensitive to external influences (e.g. restoration, natural disturbance events) and annual variation in climate. Lastly, the observation that temperature-mitigating effects of restoration tend to emerge after accounting for stream flow and air temperature, suggesting that restoration efforts currently have less influence over stream temperature than annual climate fluctuations. Benefits of recent, ongoing, and planned restoration, may take additional time to be realized and continued monitoring will be necessary to capture long-term effects. Historic trends in stream flow and air temperature, as well as projections of future climate conditions, suggest that restoration effectiveness will need to increase to outpace the influence of background climate effects.

››› (read the full chapter)

Monitoring and Research Project Chapters

CHAPTER 1 Watershed-scale Chinook Salmon and Steelhead Abundance and Productivity

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Ian Tattam, *Oregon Dept. of Fish and Wildlife*, La Grande, OR
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ABSTRACT

We monitored the watershed (population) scale response of steelhead *Oncorhynchus mykiss* and spring Chinook Salmon *O. tschawytsha* to restoration actions in the Middle Fork John Day River. Monitoring included measures of abundance and productivity for both juveniles and adults. Results for steelhead at the watershed scale show density dependance at the juvenile life-stage, likely due to limited juvenile rearing habitat, is negatively affecting steelhead recovery. Chinook Salmon populations experienced very low adult and juvenile abundances in multiple years, likely due to environmental conditions including high water temperatures and low flow, which negatively affected productivity. However, densities of Chinook Salmon redds increased in restored areas, despite no significant change in overall redd counts across the MFJDR indicating a distributional response to restoration.

INTRODUCTION

Background

Salmon and steelhead populations within the John Day River (JDR) basin have been in decline for decades, with the summer steelhead populations listed as threatened under the Endangered Species Act (ESA) in 1999. While the spring Chinook Salmon population has not been ESA listed, the population remains depressed compared to historic levels.

Numerous actions have been implemented in the JDR basin in an effort to restore these depressed populations including protective and habitat restoration actions. The MFIMW was established as an IMW in 2008 to assess restoration effectiveness on these focal fish species. The monitoring efforts described in this report are intended to provide watershed scale (status and trend) information to evaluate programmatic restoration effectiveness.

The watershed scale comparisons were designed as Before-After-Control-Impact (BACI) experiments, in which the MFIMW was compared to either the South Fork John Day summer steelhead population (SFJD) or the John Day Upper Mainstem reference area (JDUM) for Chinook Salmon. A BACI design was employed to provide spatial and temporal contrast and account for out-of-basin effects. For comparisons, we chose reference watersheds inhabited by nearby fish populations where significant background information was already being collected. While some restoration was occurring in the reference watersheds, we assumed that the amount of restoration implemented for the MFIMW would be more extensive and hence the primary variable tested by this comparison.

Region wide trends in adult escapement are often driven by out of basin, ocean, and climatic conditions. For this reason, adult escapement estimates alone are not reliable indicators to determine the success or effectiveness of restoration actions in freshwater. While abundance is an important metric for population assessments, productivity estimates are key indicators of population responses to restoration activities and can help us understand population dynamics and relationships to habitat and capacity.

Salmonid populations frequently exhibit density-dependence during freshwater rearing (Achord et al. 2003). With density dependance, productivity decreases with increasing brood year redd or spawner abundance. Thus, we expect lower productivity values at higher levels of brood year redd abundance and higher productivity values at lower levels of brood year redd abundance. We estimated freshwater productivity for the MFIMW and reference populations for spring Chinook Salmon and summer steelhead. In addition, we plotted stock-recruitment curves for Chinook Salmon to estimate and visualize density dependence and differences between the MFIMW and reference populations of Chinook Salmon. We also present stock-recruit residual analysis for Chinook Salmon to assess trends in freshwater productivity without the confounding effects of adult abundance (Peterman et al. 1998, Mueter et al 2007).

GOALS AND OBJECTIVES

To evaluate the effectiveness of restoration actions in the MFIMW at the watershed (population) scale on recovering depressed anadromous steelhead and Chinook Salmon populations we employed three objectives:

1) Estimate spawner abundance of steelhead and Chinook Salmon populations in the upper MFJDR.
2) Estimate freshwater productivity of steelhead (out-migrant juveniles per spawner) and Chinook Salmon (smolts per redd).
3) Use a Before-After-Control-Impact (BACI) design to compare the MFIMW abundance and productivity indicators for years before and after MFIMW implementation and to reference areas within the JDR basin.

SITE SELECTION

We selected sample sites in order to make abundance and productivity estimates at the watershed (population) scale to address the primary objective goal of the MFIMW to understand how restoration actions implemented across a wide temporal and spatial scale influenced fish performance metrics. Chinook Salmon and steelhead monitoring in the JDR has been ongoing for >50 yrs. Index surveys of adult steelhead and Chinook Salmon spawning activity and redds throughout the JDR basin was initiated during the 1960’s. In 2004, steelhead redd monitoring in the JDR basin employed a randomized and spatially balanced approach (Generalized Random Tessellation Survey design; GRTS) while smolt monitoring, initiated in 2002, employed rotary screw traps (RST).

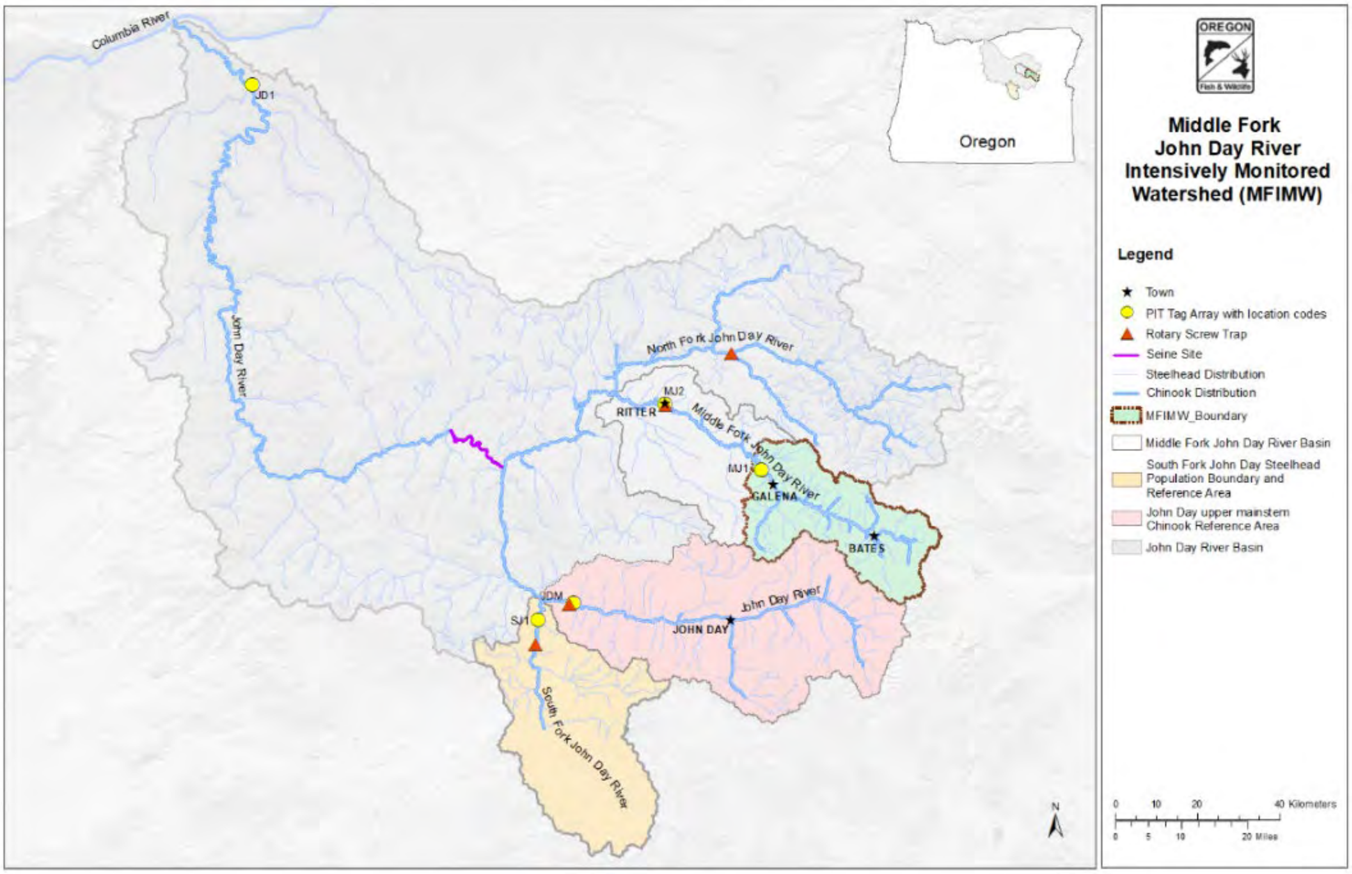


Figure 8. Map of the MFIMW steelhead and Chinook Salmon boundary, the SFJD steelhead population boundary, and the JDUM Chinook Salmon reference area. For the MFIMW watershed scale comparisons - the SFJD population is used as a reference for steelhead and the JDU Mainstem is used as a reference for Chinook Salmon.

METHODS

Objective 1. Estimate spawner escapement (abundance) of steelhead and Chinook Salmon populations to the MFJDR.

Spawner escapement for steelhead and Chinook Salmon in both the MFIMW and the reference watersheds, was measured using redd surveys on spawning grounds.

Spring Chinook Salmon spawning surveys were conducted during August and September to encompass the temporal distribution of Chinook Salmon spawning in the John Day River basin. The intention of Chinook Salmon spawning ground surveys is to complete a total redd count of all available spawning habitat in the MFIMW (Bare, Tattam and Ruzycki 2021). Surveys were conducted by walking upstream through identified sampling reaches and counting observed redds, live fish, and sampling of carcasses. We estimated spawner escapement by using the following equation.

$$\widehat{N}_p = r_p \cdot \hat{f}$$

where:

$\widehat{N}_p$ = Estimated number of spawners in the population

r_p = Number of redds observed in the population

$\hat{f}$ = Estimated fish per redd above Catherine Creek weir located in the adjacent Grande Ronde River basin (ODFW unpublished data)

See McCormick et al. (2010) and Bare, Tattam and Ruzycki (2021) for a complete description of Chinook Salmon redd survey methods.

Steelhead redd surveys, based on standard ODFW methods utilizing a Generalize Random Tessellation Survey (GRTS) design (Susac and Jacobs 1999; Jacobs et al. 2000; Jacobs et al. 2001), were conducted annually during the spring (April to June) coinciding with steelhead spawn timing in the MFJDR. Survey sites were selected using a GRTS design which randomly selects sites based on the spatial structure of the stream network of interest. Site sample points were then assigned to one of three different panels: sites visited every year (Annual sites), sites visited every other year beginning with year-1 (Two-1), or sites visited every other year beginning in year-2 (Two-2). Thirty sites were selected to be surveyed each year and were equally distributed between annual (n=15) and two-year sites (n=15 for each panel). Additional sites were selected within each panel as replacement sites in the event that a site had to be removed due to access restrictions, unidentified in-stream barriers, or unsuitable habitat conditions.

We used a 1:100,000 EPA river reach file of summer steelhead distribution in the MFJDR subbasin for site selection. This spatial dataset is based on best professional knowledge provided by ODFW managers as well as other local agency biologists. The actual dataset utilized for site selection was modified to meet the objectives of this project. Specifically, stream segments downstream of Ritter (rkm24) were excluded since this area was outside of the MFIMW area.

Beginning in 2019 due to funding and staff limitations, steelhead spawning ground survey sites were reduced to 25 and further reduced in later years to remove rotating panel sites and only include most annual sites. Site-reduction impact analysis was conducted to determine the effect of reducing the number of sites on the abundance estimate. Results of the site reduction analysis determined that adequate results with error were achieved using only annual sites.

Sites were surveyed on multiple occasions, to quantify the number of unique redds constructed at each site, and at approximately two-week intervals to account for the temporal variation in spawning activity. Survey reaches were approximately 2 km in length and encompassed the GRTS sample site sample point. Surveyors walked upstream from the downstream end of each reach and counted all redds, live fish, and carcasses observed. New redds were flagged and the location marked with a GPS. During each visit, surveyors recorded the number of previously flagged redds and new unflagged redds.

Overall redd density (R_D) was estimated by:

$$\mathrm{RD} = \sum_{i=1}^{n} ri/\mathrm{d}i \qquad (1)$$

where r_i is the number of unique redds observed at site i, d_i is the distance surveyed (km) at site i, and i is the individual sites surveyed. The total number of redds (R_T) occurring throughout the subbasin was estimated by:

$$R_T = R_D \cdot d_u \qquad (2)$$

where d_u is the total kilometers available to steelhead for spawning. Steelhead escapement (E_S) was then estimated by:

$$E_S = C \cdot R_T \qquad (3)$$

where C is an annual fish per redd constant (e.g. 2.1 fish/redd for 2022) developed from repeat spawner surveys in the Grande Ronde River basin (Flesher et al. 2005; Jim Ruzycki, ODFW, unpublished data).

A locally weighted neighborhood variance estimator (Stevens 2004), which incorporates the pair-wise dependency of all points and the spatially constrained nature of the design, was utilized to estimate 95% confidence intervals of the escapement estimate using R statistical software (R Development Core Team 2008).

The percent hatchery origin spawners (pHOS) is calculated as a five-year running average (the yearly pHOS value is averaged with the two years prior and post the spawning year). The 5-year running average is used to provide a balance in the event that a particular year didn't provide enough information on number of hatchery spawners. From 1992-2001, the yearly pHOS value was estimated by dividing the total number of hatchery origin steelhead (live and dead) observed in basin-wide index surveys by the total number of live and dead fish observed in basin-wide index surveys. Since 2002, pHOS is calculated jointly for the upper basin populations and is computed separately from the John Day Lower Mainstem population. The pHOS for the upper basin is derived from counts of spawners encountered during spawning ground surveys, and opportunistic observations during juvenile trapping and seining activities (Bare et al. 2021).

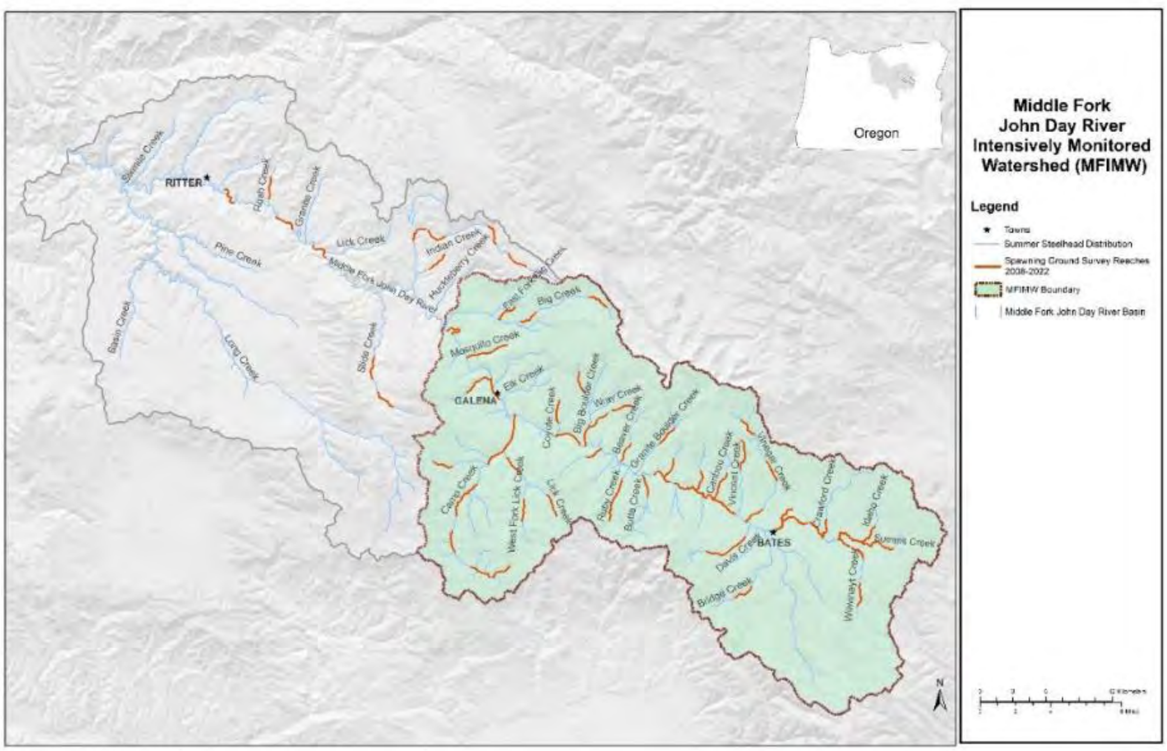


Figure 9. Summer steelhead spawning ground survey reaches in the MFIMW monitored between 2008 and 2023.

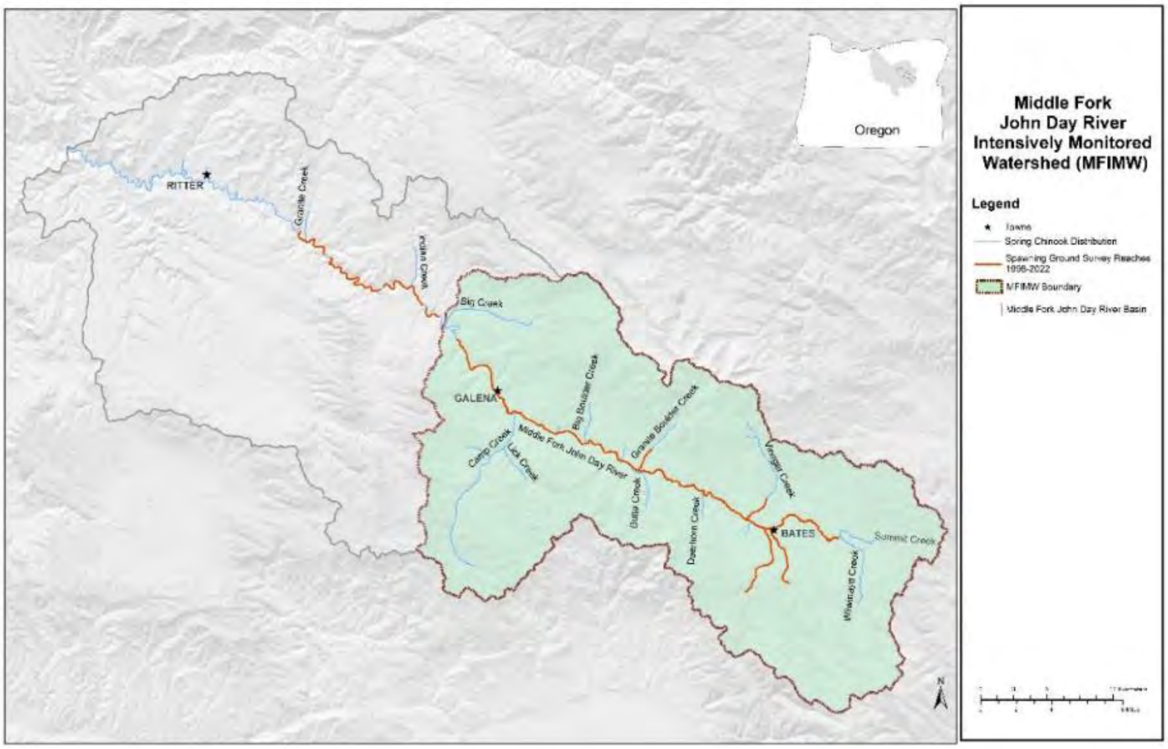


Figure 10. Spring Chinook Salmon spawning ground survey reaches in the MFIMW monitored between 1998 and 2022.

OBJECTIVE 2. Estimate freshwater productivity of Chinook Salmon and steelhead populations.

For the measurement of recovery of listed fish species, NOAA is primarily interested in estimates of fish production or survival which relate directly to their recovery. Using a juvenile recruits-per-spawner analysis is the most direct approach we currently have to estimate freshwater production (juvenile out-migrants per spawner (for steelhead) or smolts per redd (for Chinook Salmon).

Out-migrating juvenile spring Chinook Salmon and steelhead from the MFIMW were captured using a 1.52 m RST operated on the MFJDR near the town of Ritter. Complementary RSTs were operated on the SFJDR, and JDUM (Figure 1). Trap operation typically began each year during early October and continued into June of the following year to encompass a migration year. Traps were either removed or stopped during times of ice formation, high discharge, and during warm summer months after fish ceased migrating.

All RSTs were typically fished four days/week by lowering cones on Mondays and raising cones on Fridays and checked daily during the weekly fishing periods. We assumed that all fish captured were migrants. Non-target fish species were identified, enumerated, and returned to the stream. Captured juvenile spring Chinook Salmon and steelhead out-migrants were anesthetized, interrogated for passive integrated transponder tags (PIT tags) or pan jet paint marks, enumerated, weighed to the nearest 0.1 g, and measured (fork length, mm). A subsample of fish was released above the trap to estimate out-migrant abundance at the trap using mark-recapture techniques. We used linear extrapolation to account for un-sampled nights. Abundance at the RST of out-migrating juveniles was estimated using counts at the RST and a mark-recapture model with a GAM
p-spline to fill in for missed days. Then survival to John Day Dam (JDA) was estimated using the Cormack-

Jolly Seber model GUI found at https://www.cbr.washington.edu/dart. Further details of our RST operation are available in Bare et al. (2021).

Out-migrant abundance estimates at the traps coupled with survival estimates to JDA were then used to estimate smolt equivalent abundance for the JDUM and MFIMW populations of Chinook Salmon. To estimate smolt equivalents passing the trap, we account for the overwinter mortality that the early (fall) migrant fish experience, after they have already passed the trap and have been counted and PIT-tagged. We make the simplifying assumption that the relative proportional difference in survival between the early migrants (fall) and the late (spring) migrants (S_{early}/S_{late}) estimates the overwinter survival rate experienced by both groups. For the late (spring) migrants, the number of smolt equivalents passing the trap is the same as the estimated abundance passing the trap since they have already experienced overwinter mortality before being counted, tagged and passing the trap.

For the early (fall) migrant group, to get the number of estimated smolt equivalents passing the trap in the spring we take the number of fish estimated to have passed the trap in the fall and adjust it by the estimated overwinter survival. We chose to use out-migrant estimates of steelhead rather than smolt equivalent estimates due to inconsistent survival estimates. For Chinook Salmon we chose to use redd counts as the “spawner” metric in the recruits-per-spawner calculation because we count all redds in the MFIMW with no expansions or extrapolations and this total redd count is the closest to a true population count with no sampling bias.

To evaluate Chinook Salmon productivity trends and evaluate differences between the MFIMW and JDUM populations, we followed methods described in Bare, Tattam and Ruzycki 2021 and modeled productivity metrics for each population of Chinook Salmon with Ricker stock-recruitment curves fit to the total redd abundance dataset from 2000 to present. Then the natural log of smolt recruits per brood year redd was regressed against brood year redds to parameterize a Ricker stock-recruitment curve for each population. We then measured the difference between the predicted and estimated smolt recruits using a “residual” analysis approach as demonstrated in Warkentin et al. 2022. The residuals from this regression measure the deviation between observed recruitment and the recruitment rates predicted after adjusting for density-dependence. A positive residual indicates higher than expected productivity, whereas a negative residual indicates lower than anticipated productivity. We then plotted the residuals against brood year to evaluate temporal trends in productivity (Bare, Tattam and Ruzycki 2021).

Steelhead productivity was calculated as out-migrating juvenile estimates at the trap per total spawners. While it is possible to calculate smolt equivalents for steelhead, recent downstream conditions have made PIT-tag detections difficult, and survival estimates to JDA for steelhead are unreliable.

OBJECTIVE 3. Use a Before-After-Control-Impact (BACI) design to compare the MFIMW abundance and productivity indicators for years before and after MFIMW implementation and to reference areas within the JDR basin.

To evaluate abundance and productivity before and after implementation of the MFIMW, we plotted the MFIMW and reference populations and calculated an “index” of the difference between the MFIMW and the reference population. An index of the influence of the MFIMW restoration actions is shown as the difference of productivity between the MFIMW (treatment) population and the JDUM (reference) population. Resulting plots of the index metrics will show trends in the differences between the MFIMW and the reference population. An increasing trend or values above 0 indicate a positive MFIMW population response in comparison to the reference population. BACI analysis assumes that reference and control populations experience similar climatic and out-of-basin conditions, such that changes in the difference between the two populations (i.e. index metric) show the results of the treatment (i.e. restoration actions). A true BACI experiment utilizes a control, in which no treatment is applied to the "control" sample. Because of ongoing (although limited) restoration (our treatment variable) in the JDUM and SFJD, we do not have a true control watershed, and thus use "reference populations" instead.

RESULTS

Adult Escapement and Freshwater Productivity

Steelhead

Average steelhead redd densities varied across years and within sites. The highest redd density was in Camp Creek, while in many sites, zero redds were detected (Figure 5). Overall average redd density decreased from 2018-2022, with the lowest average density observed in 2022 for both the MFIMW and SFJD (Figure 4).

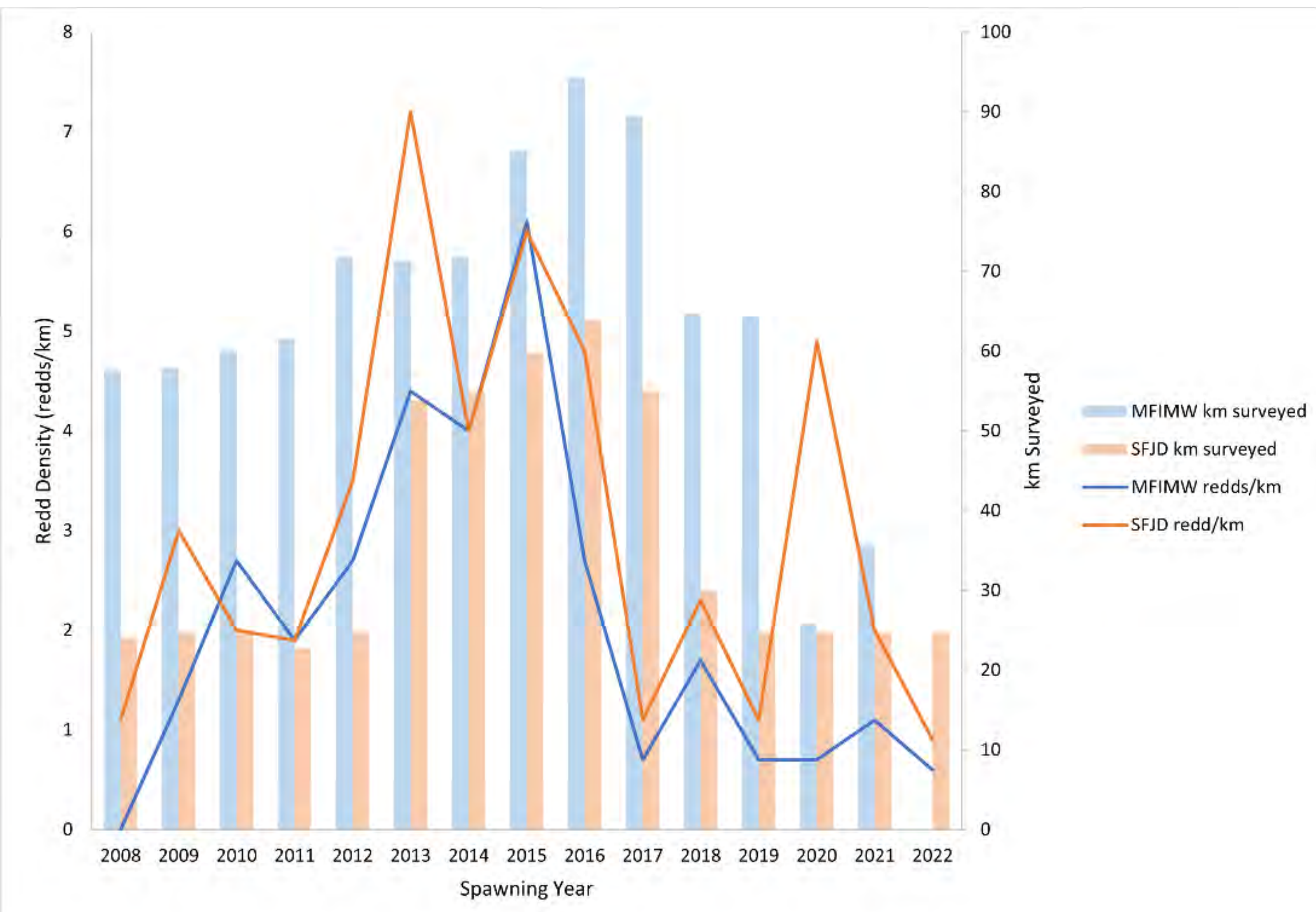


Figure 11. Average yearly summer steelhead redd density (redd/km) for sites surveyed in the MFIMW and SFJD from 2008-2022. Observed redd density was low in 2008, the first year of GRTS surveys, in the MFIMW due to implementation of new survey methods. Redd densities were higher than expected in the SFJD in 2020 due to completion of surveys occurring before high flows

and when redds were visible, while MFIMW surveys were completed after high spring flows when redds were difficult to observe.

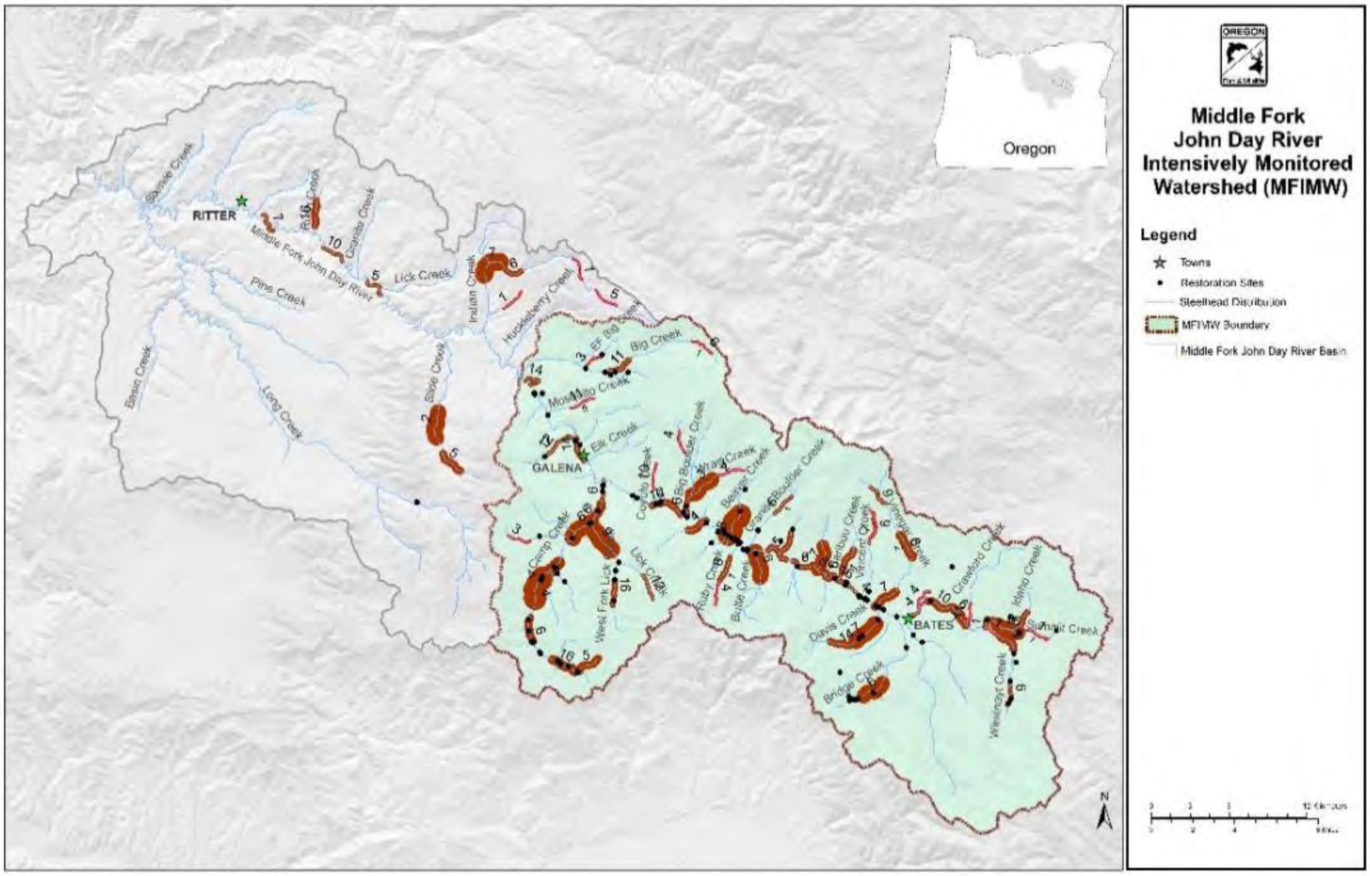
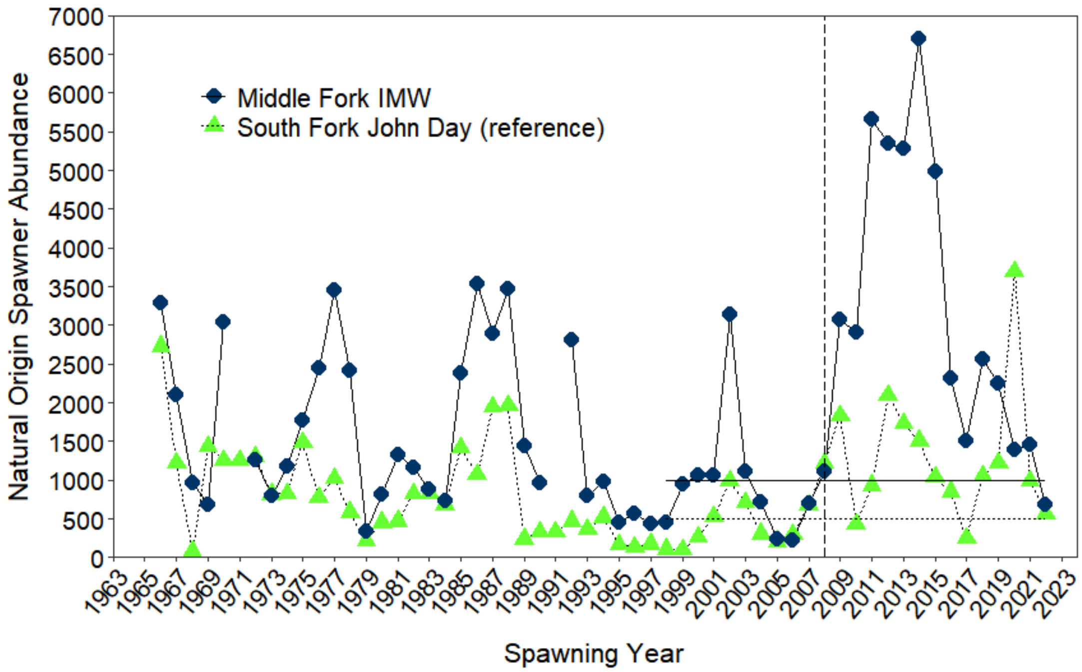


Figure 12. Average redd densities (redds/km) for summer steelhead in the MFIMW for years 2008-2022. Thicker lines denote reaches with higher densities, while reaches with zero observed redds are shown in red. Numbers beside each reach depict the number of years the site was surveyed. Steelhead spawning ground survey sites were chosen using a Generalized-Tessellation Randomized Survey (GRTS) spatially balanced design.

Adult steelhead escapement in the MFIMW followed a similar trend to the adult escapement in the South Fork John Day prior to 2008 when the MFIMW was established (Pearson Product Moment Correlation = 0.66, P < 0.01; Figure 6). For most years the difference between the two populations is above zero, indicating that the MFIMW population is performing better than the reference, SFJD. (Figure 6).



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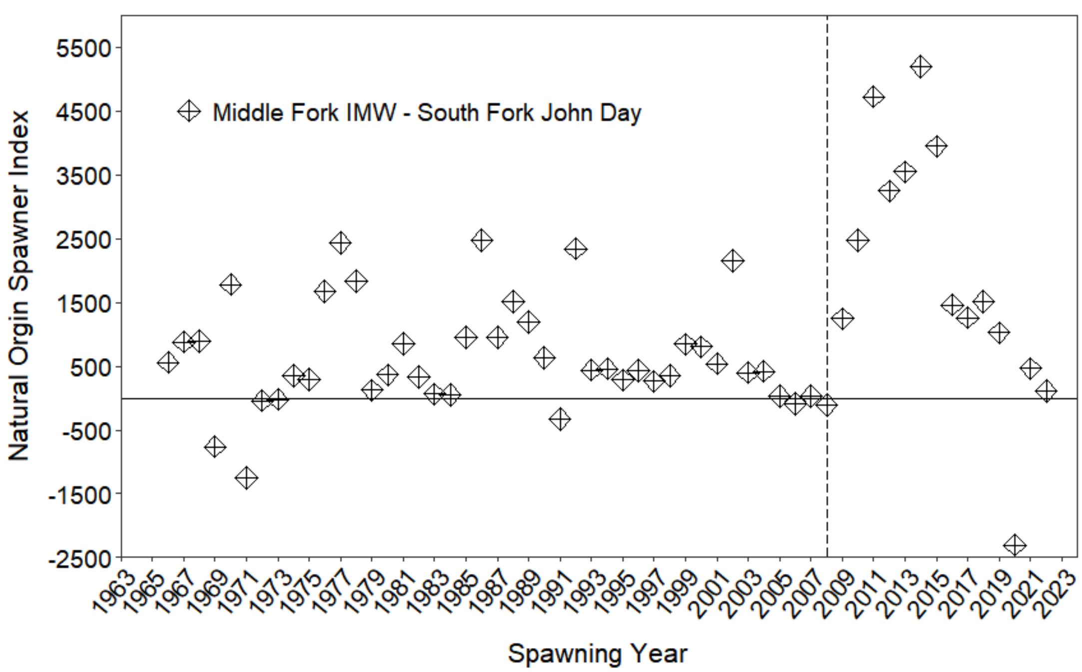
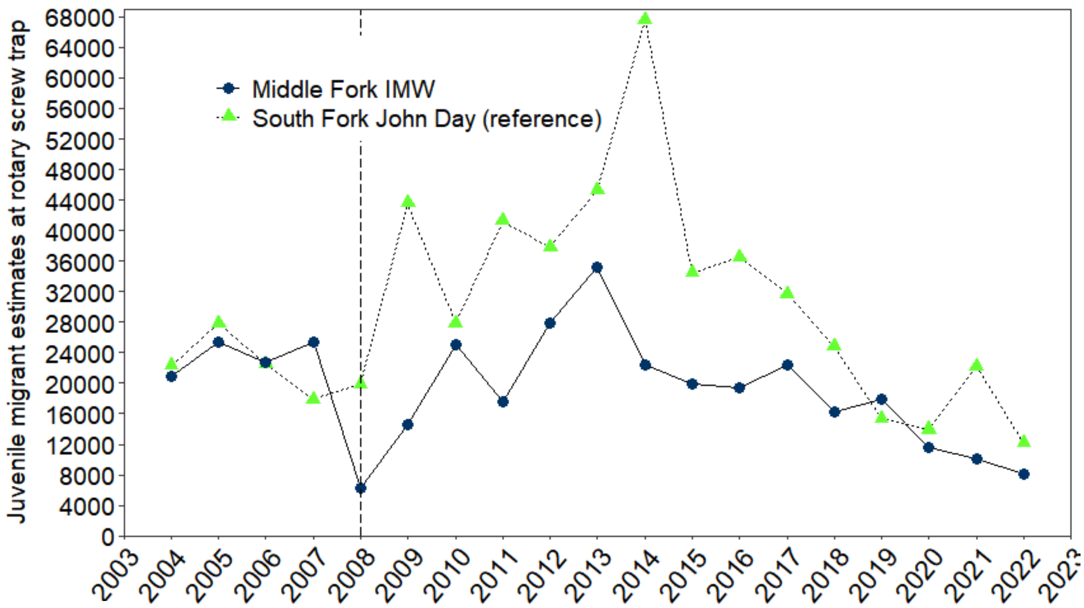


B

Figure 13. A. Long-term trends in adult spawner abundance for the MFIMW and SFJD steelhead populations. Horizontal solid line represents the MFIMW natural origin spawner Minimum Abundance Threshold (1000); horizontal dotted line represents the SFJD natural origin spawner Minimum Abundance Threshold (500). **B**. Index of adult steelhead spawner abundance before and after initiation of the MFIMW. The index shown is the difference of spawner abundance of the MFIMW (treatment) population and the SFJD (reference) population. Vertical dashed line indicates initiation of the IMW experimental period (2008). Values above zero indicate the reference population is performing better than the control population.

Estimates of juvenile steelhead out-migrants passing the RSTs for both populations have been declining since Migration Year (MY) 2014, while the difference between the MFIMW and SFJD shows a slight increasing trend since 2014 (Figure 7).



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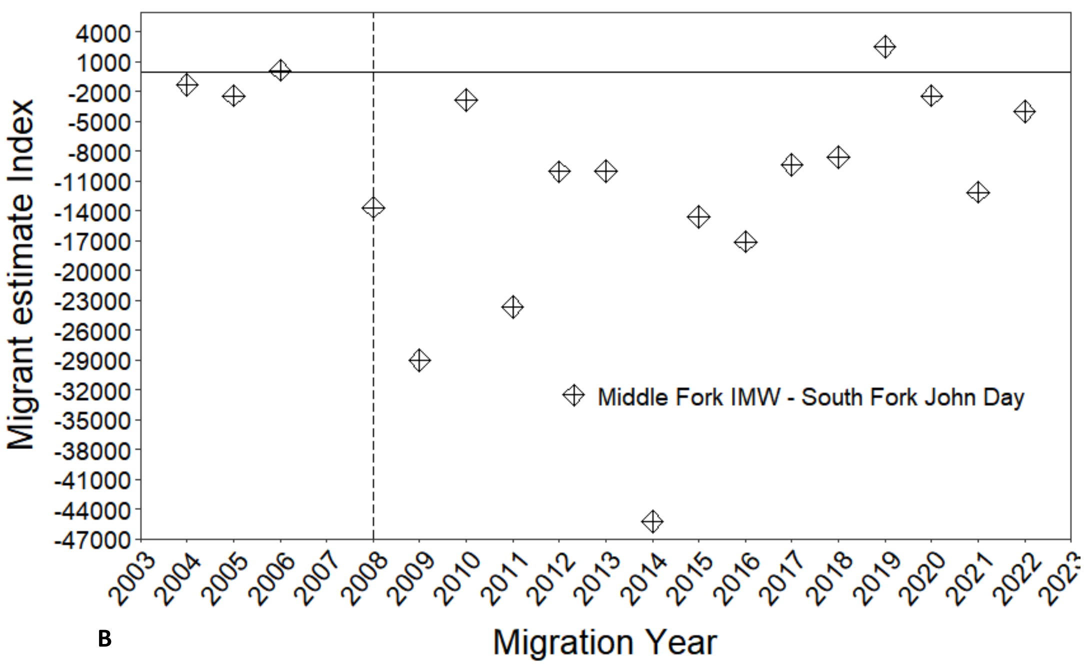
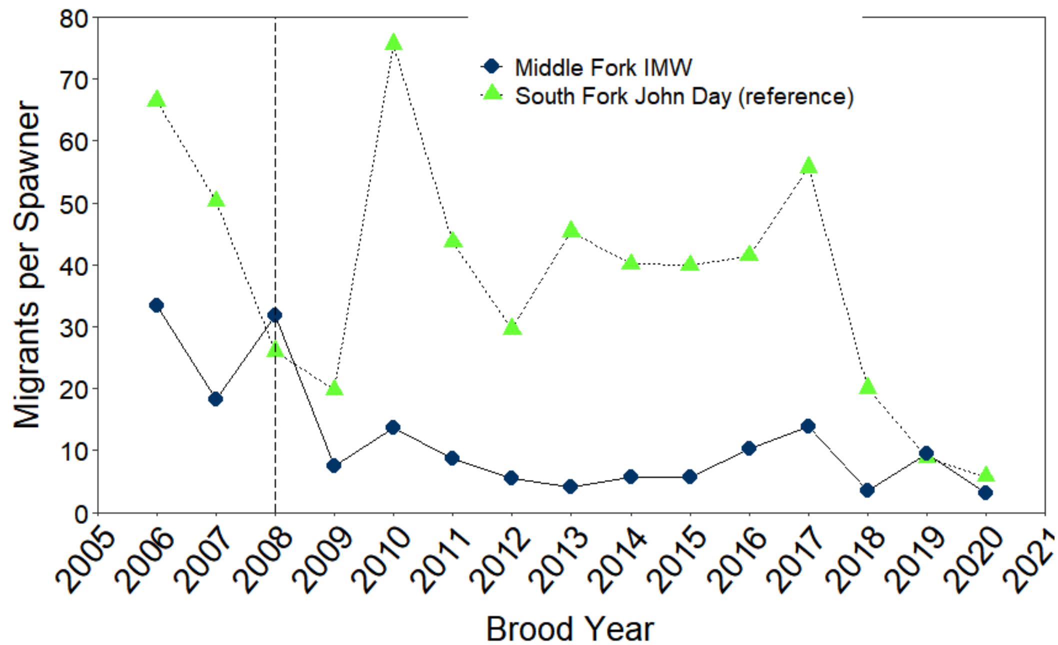


B

Figure 14. A. Long-term trends in juvenile steelhead out-migrant estimates at rotary screw traps. Vertical dashed line represents the initiation of the IMW experimental period. *SFJD screw trap is located within the spawning and rearing distribution, whereas the MFIMW screw trap is located downstream of most spawning and rearing. **B**. Index of juvenile out-migrant abundance before and after initiation of the MFIMW. The index shown is the difference of out-migrant abundance of the MFIMW (treatment) population and the SFJD (reference) population. Vertical dashed line indicates initiation of the IMW experimental period (2008).

Freshwater productivity measured as out-migrants per spawner has remained relatively consistent in the MFIMW since 2009, while freshwater productivity has trended downward in the SFJD since 2017 (Figure 8). While the index (MFIMW-SFJD) has been well below zero for most the time series, in recent years, the index is closer to zero, indicating that the MFIMW population is performing better than the SFJD.



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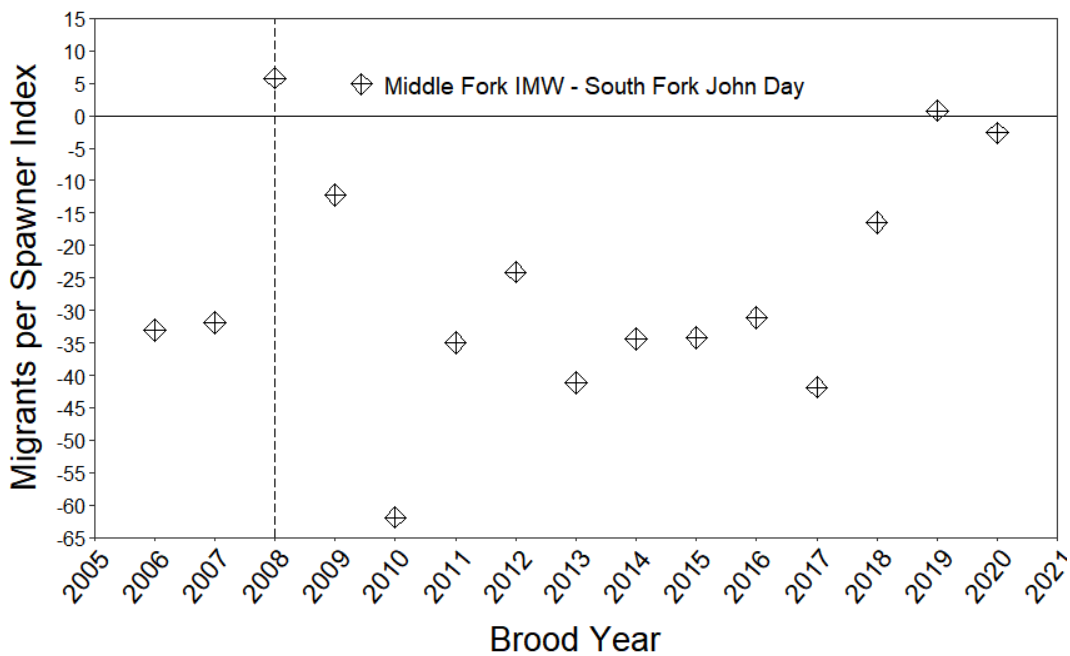


B

Figure 15. A. Long-term trends in juvenile freshwater productivity (out-migrating juveniles per spawner) for the MFIMW and SFJD summer steelhead populations. **B**. An index of the influence of the MFIMW restoration actions is shown as the difference in productivity of the MFIMW (treatment) population and the SFJD (reference) population. Vertical dashed line indicates initiation of the MFIMW experimental period (2008).

Total steelhead out-migrants produced from the MFIMW remains consistent, with most years producing around 18,000 out-migrants regardless of the number of spawners; indicating density dependance of juveniles in freshwater habitats, whereas the SFJD has had greater variation in out-migrant abundance, suggesting density-independent factors, such as climate, may have greater influence on SFJD productivity than in the MFIMW (Figure 9).

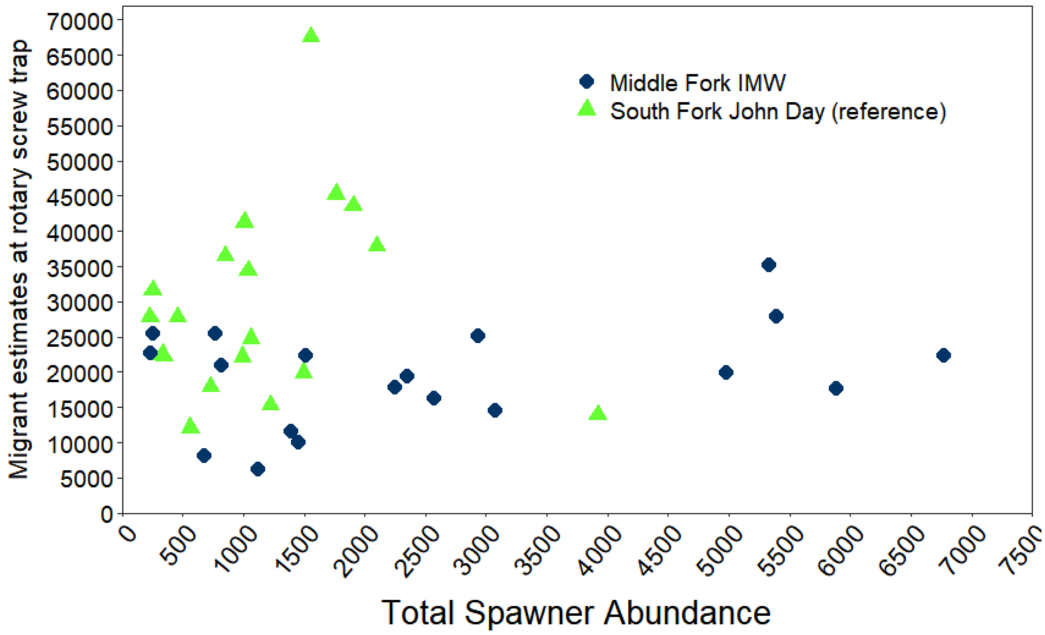


Figure 16. Steelhead out-migrants produced per total spawner for brood years 2006-2020.

Similar patterns were observed for juvenile productivity as a response to spawner abundance. Steelhead out-migrants per spawner for the MFIMW remains consistent with most years producing around 6 out-migrants per spawner regardless of the number of spawners (Figure 10). This models density dependence for the MFIMW by predicting lower recruitment rates at higher brood year spawner abundances.

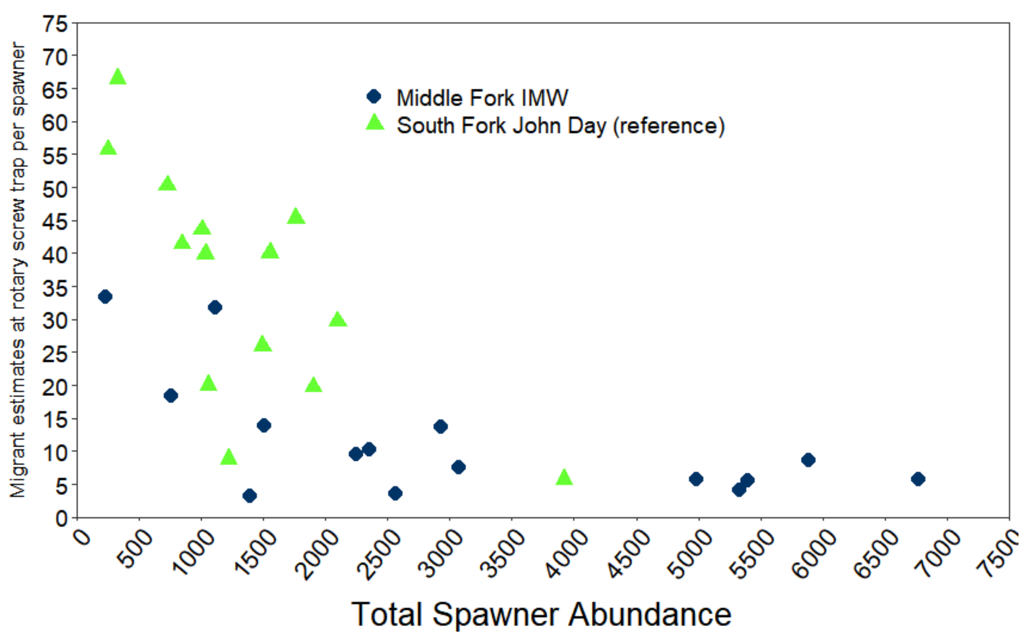
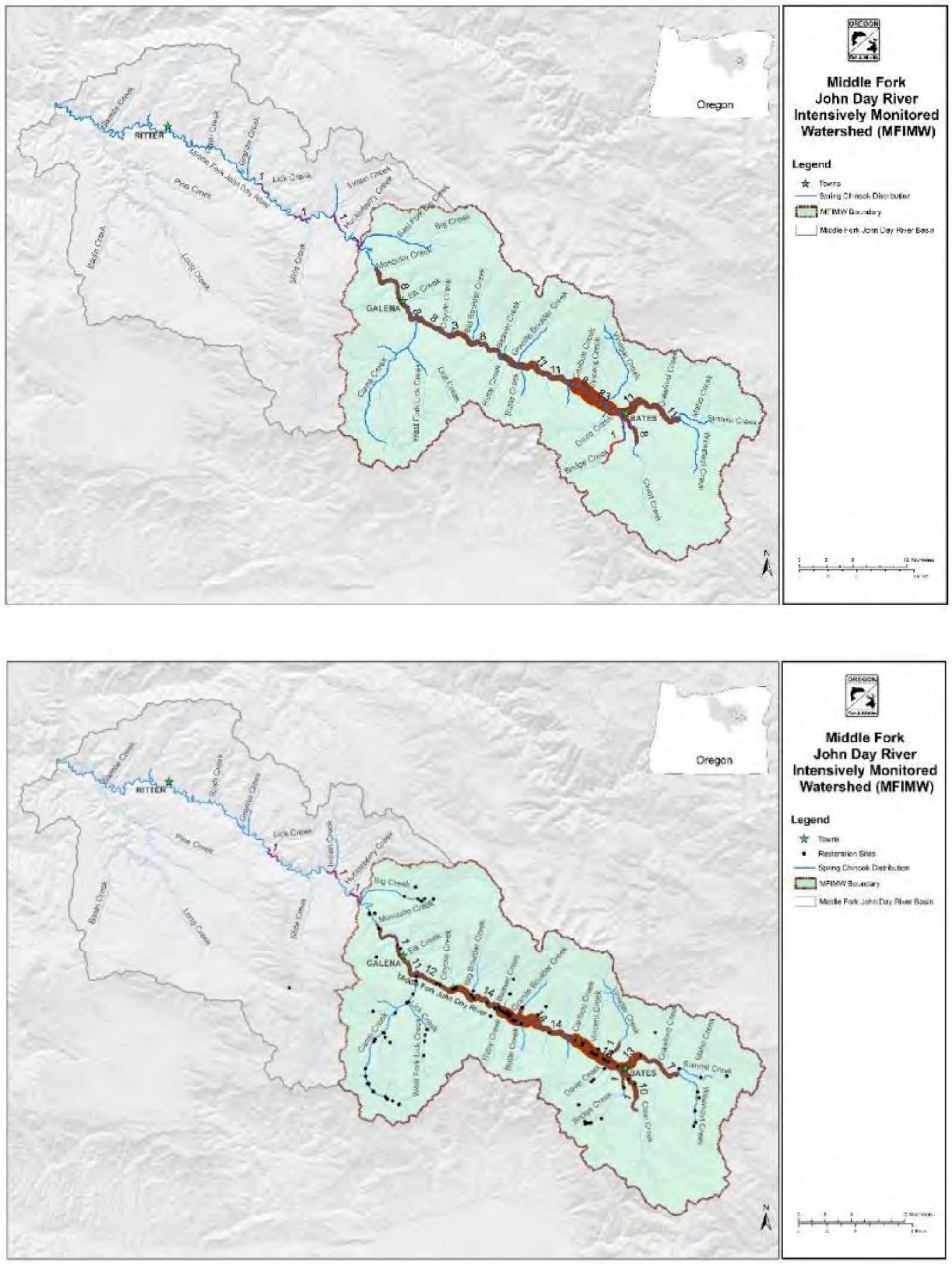


Figure 17. Steelhead out-migrants per spawner per total spawner for MFIMW and SFJD summer steelhead populations for brood years 2006-2020

Chinook Salmon

Average redd densities or redd counts did not increase during the time period, but redd spatial distribution appears to have shifted downstream into areas of more intensive restoration in the mainstem MFJDR ([Figure 11](#); [Figure 12](#)).



A

B

Figure 18. A. Average redd density by survey reach for Chinook Salmon in the MFIMW from 1998-2008. Average densities ranged from 0-19 redds/km. **B**. Average redd density by survey reach for Chinook Salmon in the MFIMW from 2009-2022. Average densities ranged from 0-36 redds/km. Thicker dark red lines denote higher average redd densities. Bright red lines are reaches where no redds have been observed. Numbers next to reaches indicate the number of years that the reach had been surveyed.

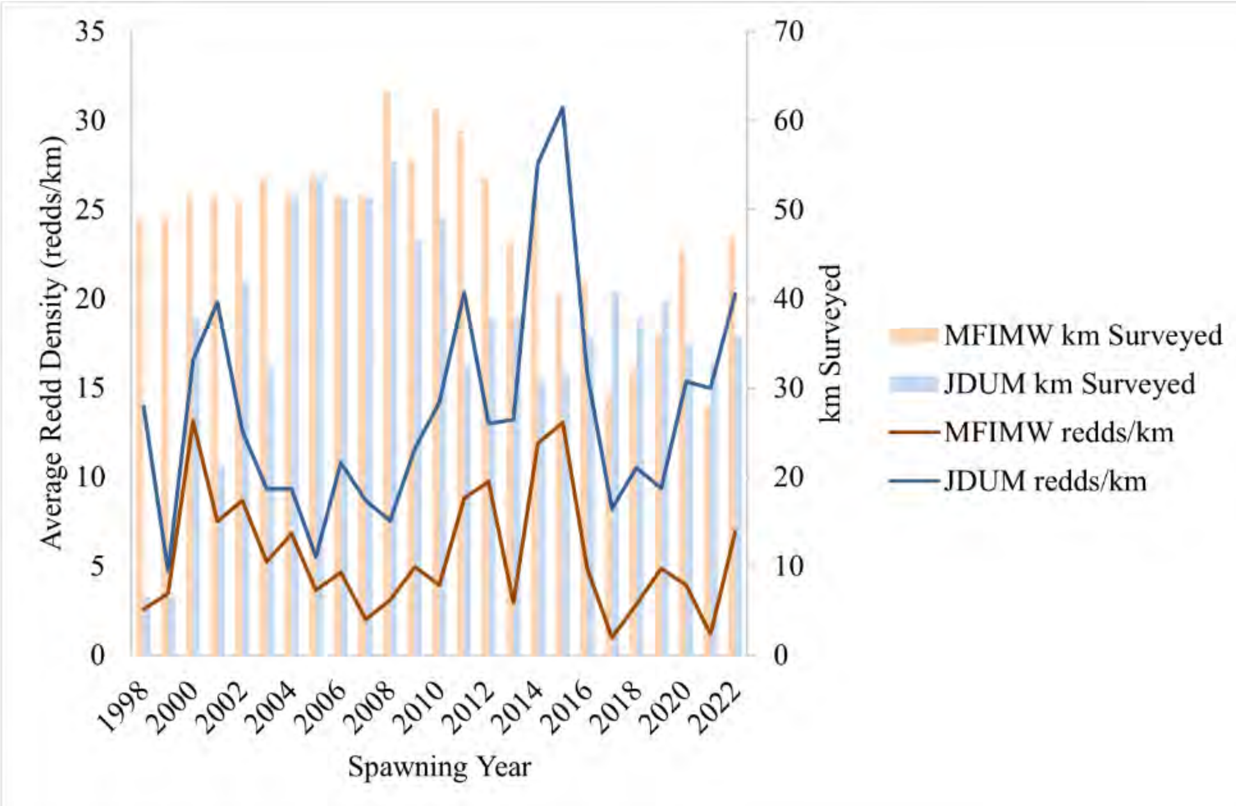
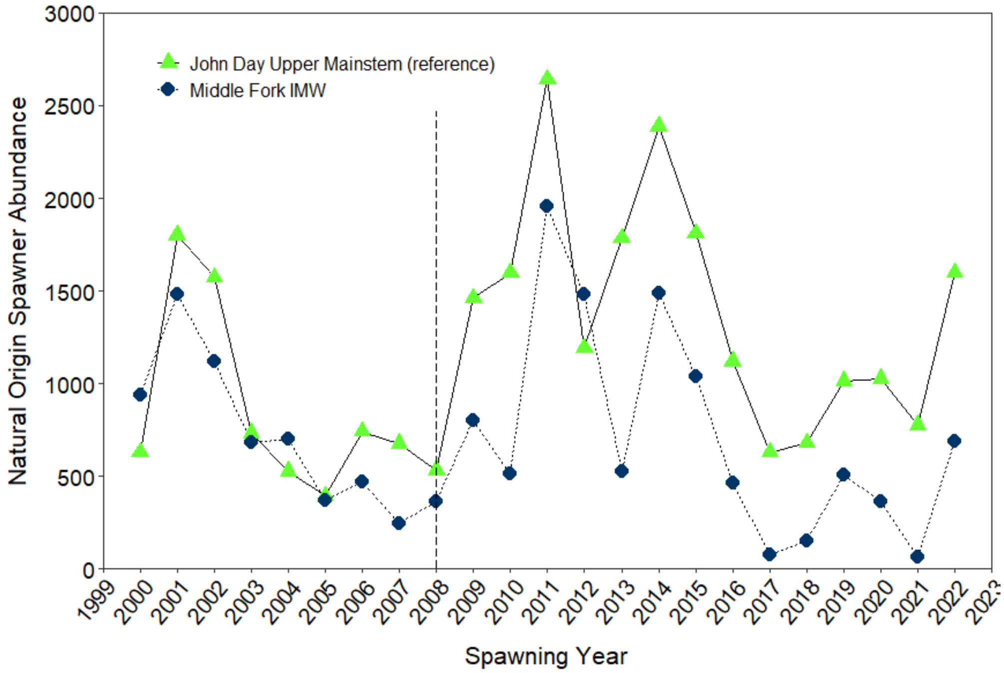


Figure 19. Yearly average redd densities and km surveyed for MFIMW and JDUM Chinook Salmon.

Adult Chinook Salmon escapement in the MFIMW followed a similar trend to the adult escapement in the JDUM since the IMW was established (Pearson Product Moment Correlation = 0.61, P < 0.01; Figure 13. Chinook Salmon spawner abundance in the MFIMW has not increased relative to the JDUM (Figure 13).



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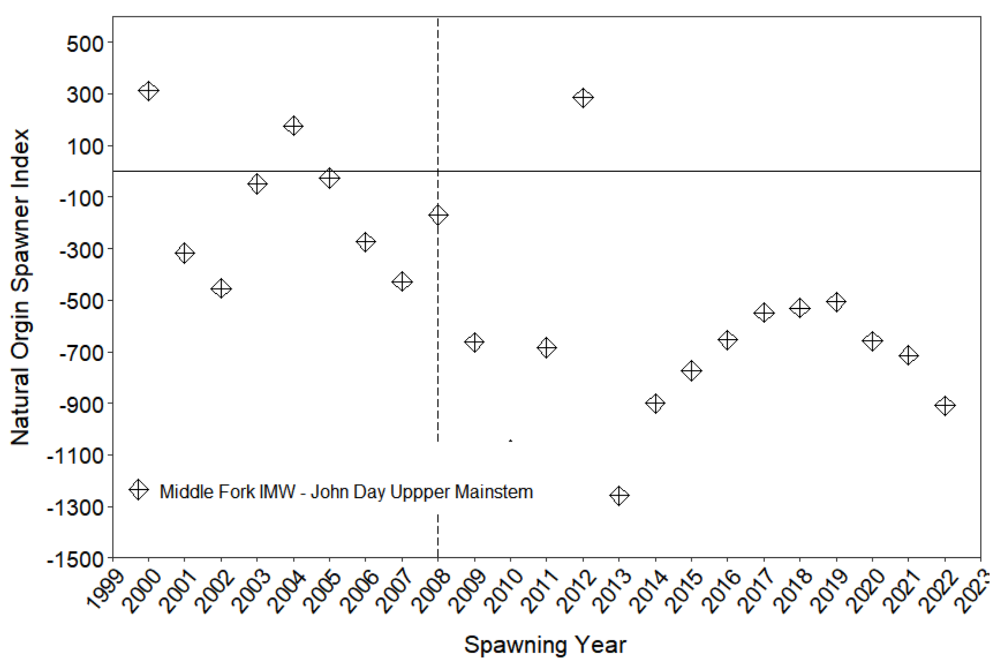
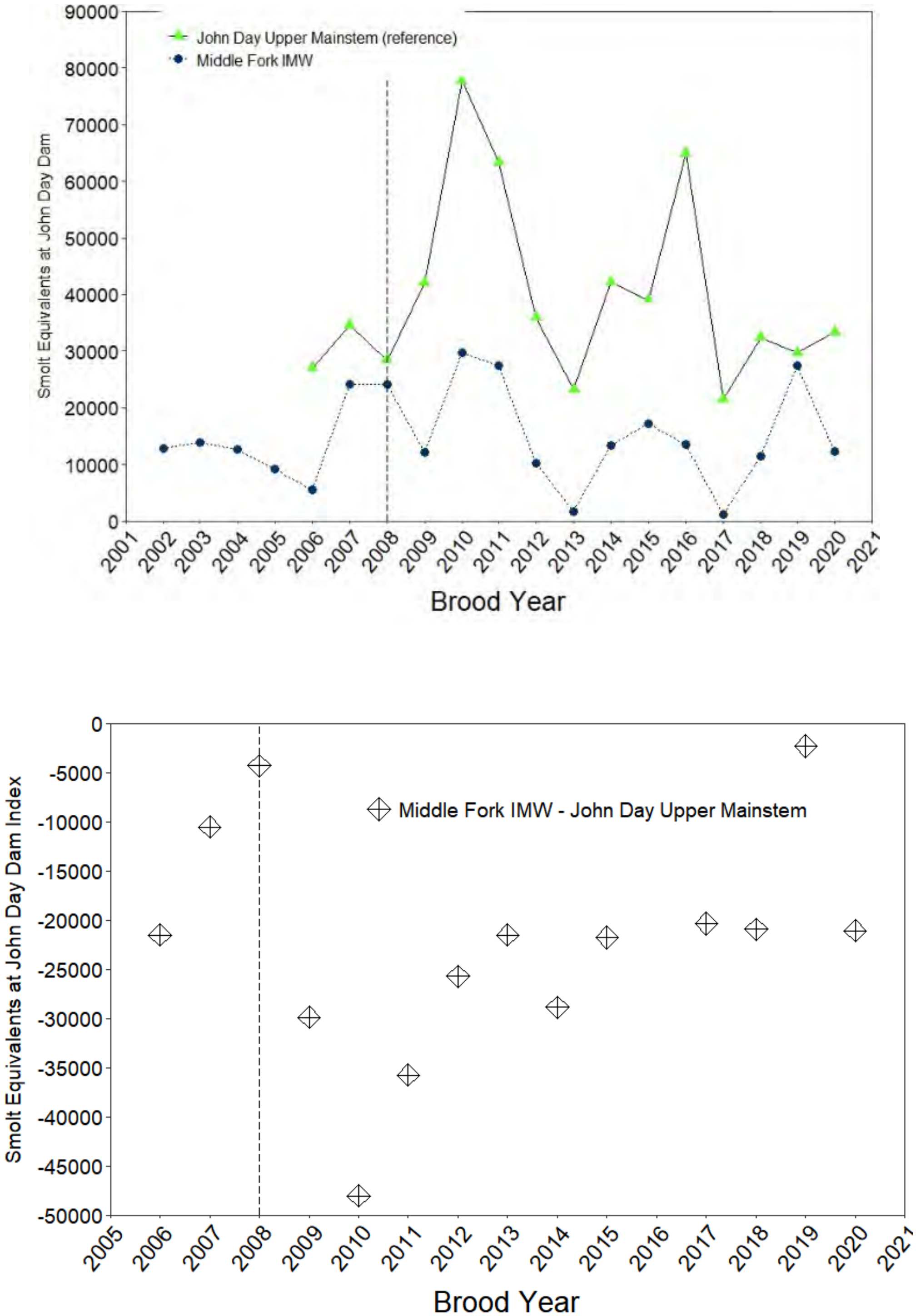


B

Figure 20. A. Long-term trends in adult spawner abundance for the MFIMW and JDUM spring Chinook Salmon populations. **B**. The index shown is the difference of spawner abundance of the Middle Fork (treatment) population and the JDUM (reference population). Values below 0 in the index indicate poorer performance by MFIMW Chinook Salmon than in the JDUM. Vertical dashed line indicates initiation of the IMW experimental period (2008).

Trends in smolt abundance in the MFJDR basin relative to the JDUM population were relatively stable [(Figure 14).](#) Smolt equivalent estimates for the MFIMW were very low in BYs 2012-2017, with near cohort failure in 2017 [(Figure 14).](#)

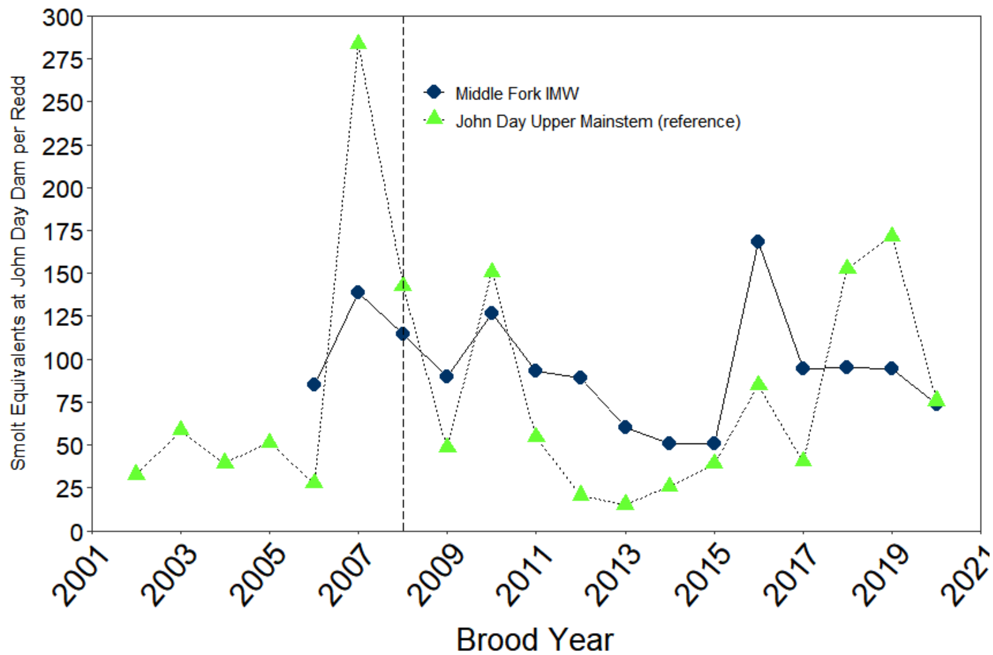


A

B

Figure 21. A. Trends in Chinook Salmon juvenile out-migrant abundance for the MFIMW and John Day Upper Mainstem reference. **B.** An index of the influence of the MFIMW restoration actions is shown as the difference of out-migrant abundance of the MFIMW (treatment) population and the Mainstem (control) population. Vertical dashed line indicates initiation of the MFIMW experimental period (2008).

Freshwater productivity measured as smolts per redd show an increasing trend in the MFIMW when compared to the reference watershed population from BY 2016 to 2019 (Figure 15).



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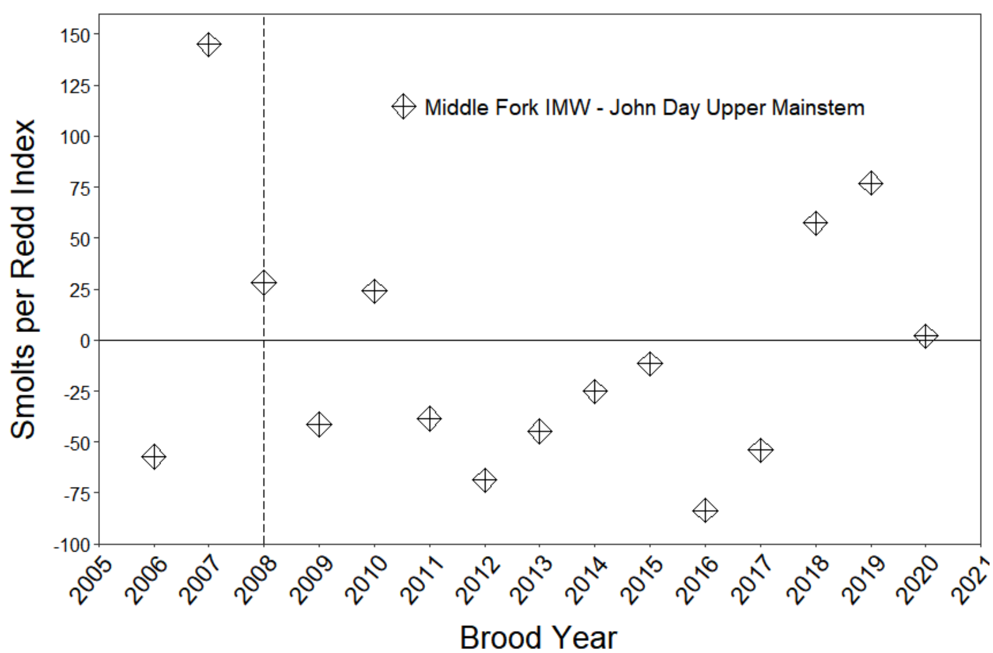
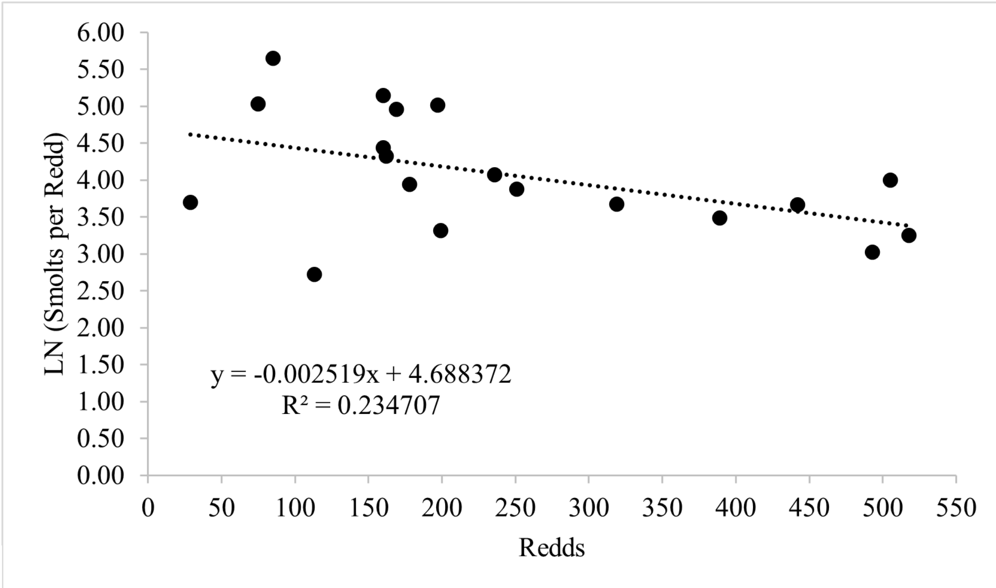


B

Figure 22. A. Trends in juvenile freshwater productivity for the MFIMW and JDUM spring Chinook Salmon populations. **B.** An index of the influence of the MFIMW restoration actions is shown as the difference of productivity of the Middle Fork (treatment) population and the JDUM (reference) population. Vertical dashed line indicates initiation of the MFIMW experimental period (2008).

We used the relationship between smolts per redd plotted against brood year redds to parameterize a Ricker stock recruit curve (Figure 16). This describes the strength of the relationship between progeny and parents, which allows us to identify the degree to which production is density-dependent (influenced more by parental abundance and potentially modified by stream habitat quantity and quality) versus density-independent (influenced more by external factors such as drought or flood).



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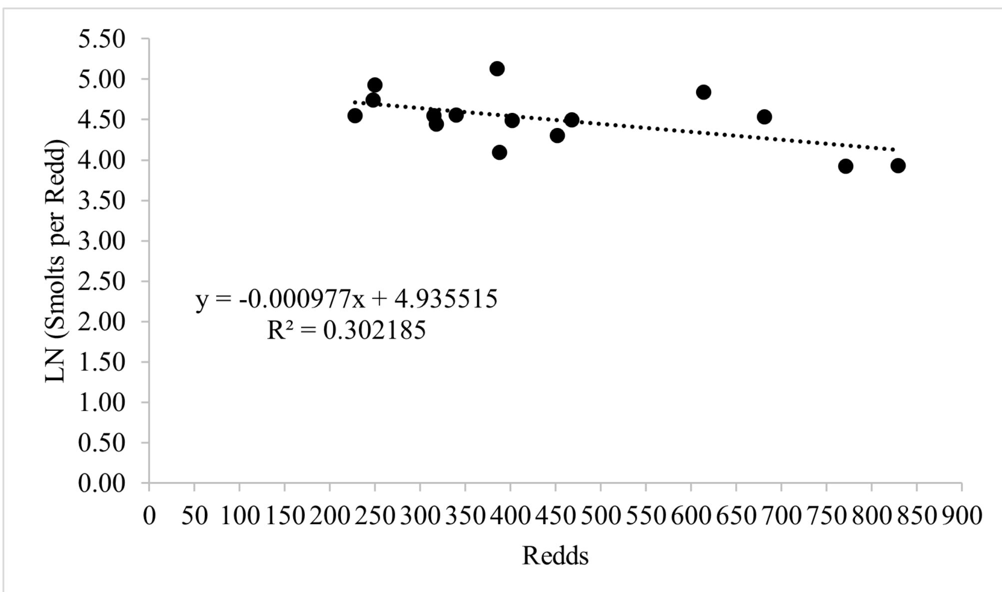
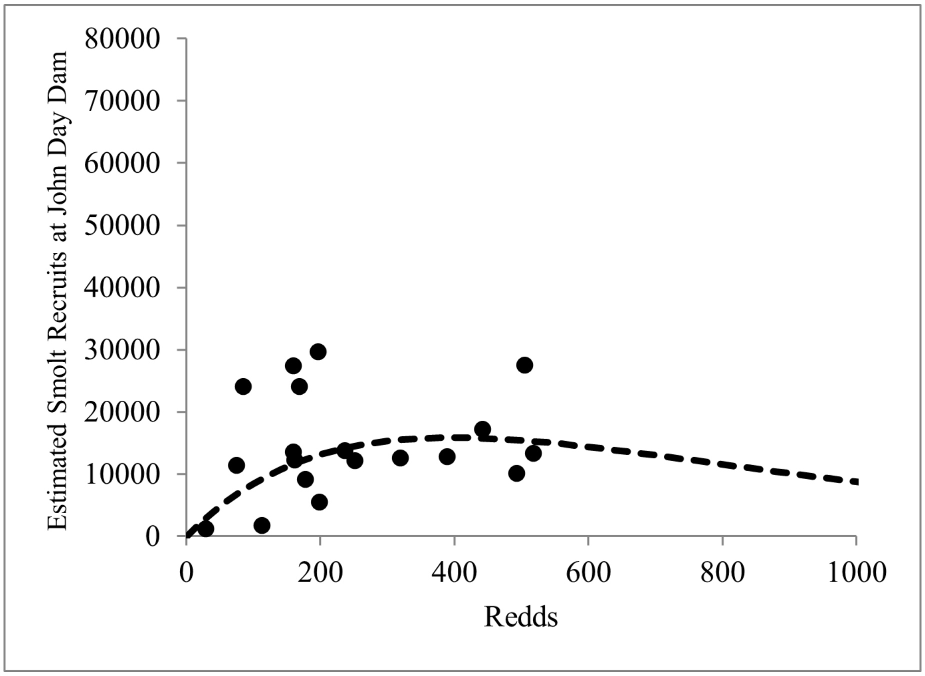


B

Figure 23. Relationship between spring Chinook Salmon redd abundance and ln smolt equivalents per redd in the MFIMW (**A**) and JDUM (**B**) populations.



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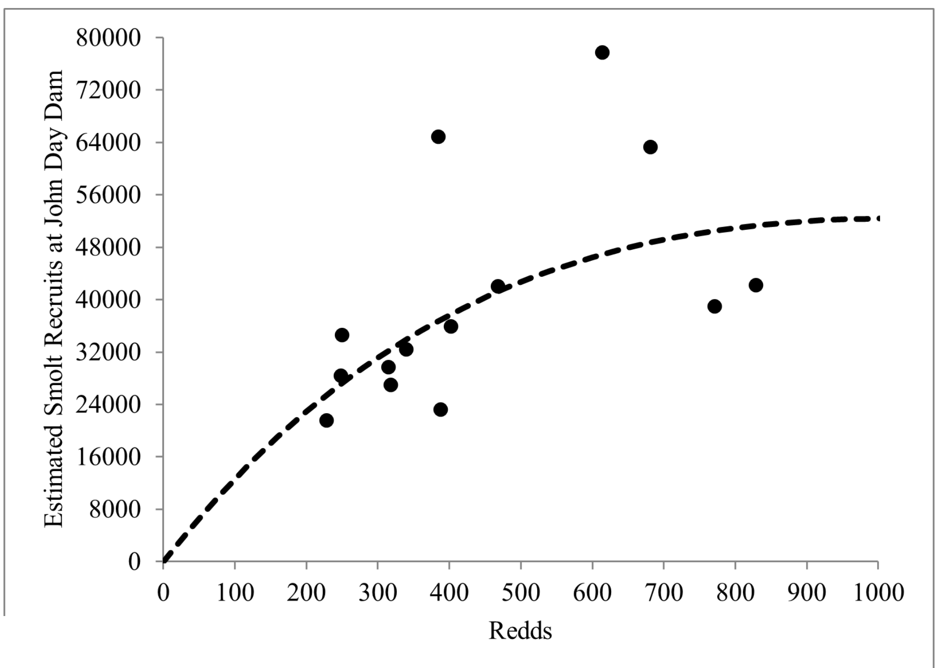
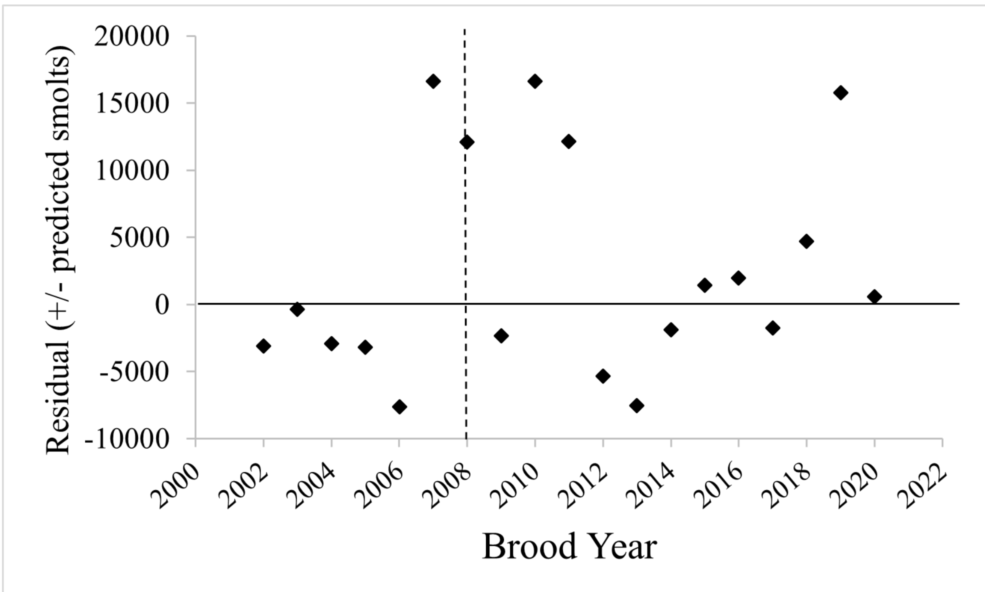


B

Figure 24. Estimated smolt recruits plotted against brood year redds for the MFIMW (**A**) and JDUM (**B**) spring Chinook Salmon populations. Black dashed lines are the fitted Ricker stock-recruit curve.

There is marginal evidence for density-dependence in the MFIMW Chinook Salmon population, as evidenced by the low coefficient of determination in Figure 17, panel A. The JDUM shows less evidence of density dependance than the MFIWM (Figure 17, panel B). When we plot the residual values (predicted-estimated) we see an increasing trend across brood years for the MFIMW since 2008, with a higher proportion of broods since 2008 producing more progeny than predicted after scaling for parental density (I.e., a residual that is above the 0 line). A less distinct positive trend is observed for JDUM Chinook Salmon (Figure 18).



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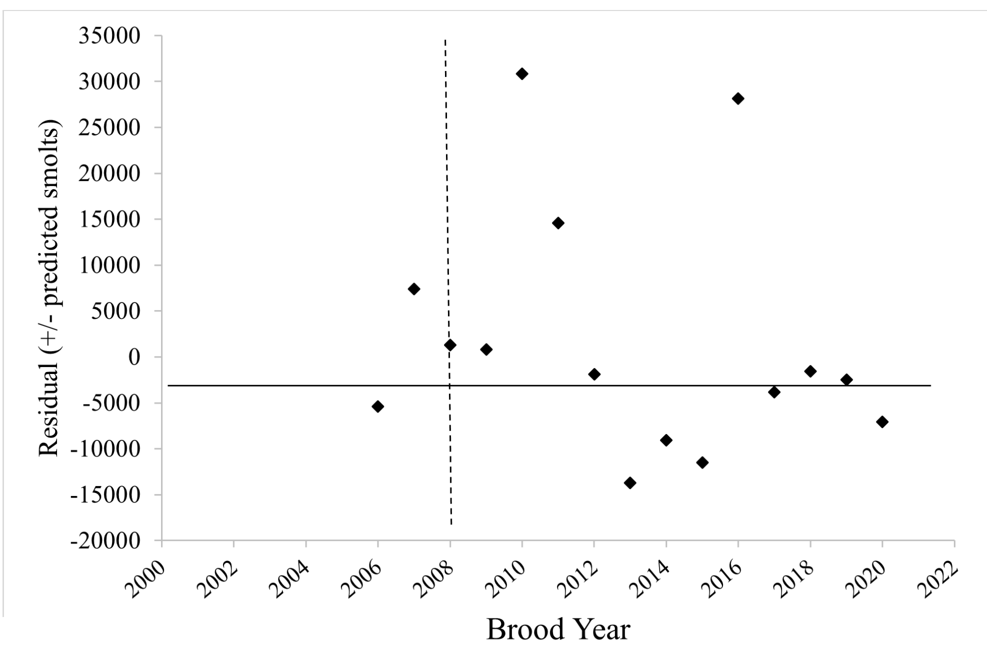

B

Figure 25. MFIMW Spring Chinook Salmon smolt residual values (**A**) and JDUM spring Chinook Salmon smolt residual values (**B**). The residual shows the difference between predicted smolt estimates from the Ricker stock-recruitment curves and measured smolt abundance estimates.

DISCUSSION

The approach IMW's employ can be an effective experimental design for evaluating watershed-scale salmon and steelhead responses to habitat restoration, especially for IMW's with single or defined restoration actions. Detecting a response in the MFIMW has been challenging given the size, diversity, and long-time span of restoration actions in the basin. The first 10 years of evaluation of salmon and steelhead at the watershed scale showed relatively little change in abundance or productivity when compared to the reference watersheds (Middle Fork IMW Working Group 2017). The latest five years of data potentially show slight positive responses in productivity. Productivity for steelhead in the SFJD has fluctuated more and has generally trended higher than the MFIMW until 2020, the latest brood year for which we have data. While MFIMW steelhead productivity has not increased since the inception of the IMW in 2008, it also has not declined like the SFJD, potentially indicating either a positive response to habitat conditions or inherently more climate-resilient conditions for steelhead rearing in the MFJD than steelhead rearing in the SFJD.

Out-of-basin influences, and climate impact steelhead and Chinook Salmon abundance and productivity. Previous work by Bare, Tattam and Ruzycki 2021 show strong correlations in Chinook Salmon spawner abundance and the Pacific Decadal Oscillation (PDO) and prespawn mortality. Adult Chinook Salmon escapement is measured on the spawning grounds after these fish have spent the summer holding in locations near their spawning grounds. The MFIMW population of adult Chinook Salmon experienced significant pre-spawn mortality in three of the last ten years as a result of seasonally low stream flows and high temperatures early in the summer (Middle Fork IMW Working Group 2017). These events occurred in 2007, 2013, 2015, 2017, and 2021, reducing the number of adults building redds on spawning grounds during those years. High prolonged air temperature conditions in 2021 and a cohort failure of juveniles in brood year 2017 resulted in a near quasi-extinction event for spring Chinook Salmon in the MFIMW (unpublished data ODFW 2021; unpublished imagery data CTWSRO 2021).

While Chinook Salmon abundance and productivity have not significantly increased since the inception of the MFIMW, Chinook Salmon spawners are responding to restoration activities and redistributing spawning to restored reaches (Figure 11; Figure 12; Lemanski ODFW, unpublished data). This result indicates that further restoration may create more desirable spawning locations for Chinook Salmon. Improved spawning habitat coupled with targeted restoration to increase juvenile habitat capacity and reduce density dependence will benefit Chinook Salmon abundance and productivity (Chapter 2).

While we continue to see density dependence in the MFIMW Chinook Salmon population, the relationship is not as strong as expected. This is largely due to surprising near-recruitment failures at low redd abundance, when we expect to see high values of recruits per spawner. Our working hypothesis is that high stream temperatures during summer rearing for these broods created extremely poor conditions across the riverscape. Hence these 'global' density-independent occurrences weaken the signal of a density dependent relationship, which potentially exists and is modifiable through habitat restoration actions. These broods when density-independent conditions (mainly summer stream temperature) created lower than expected juvenile production (i.e., a residual value < 0) impede population recovery and detection of a signal in terms of response to habitat restoration. Monitoring and modeling efforts by

ODFW indicate that stream flow and water temperature are limiting freshwater production of both steelhead and Chinook Salmon in the MFIMW (Middle Fork IMW Working Group 2017, Handley ODFW, unpublished data 2017). The limited summer distribution of juveniles in the mainstem MFJDR, and the movement of juveniles into cool-water tributaries (Ciepiela 2023 – Chapter 2; Kaylor 2023 – Chapter 4) also demonstrates that water temperature is limiting the availability of productive habitat for rearing Chinook Salmon and steelhead. Further, juvenile Chinook Salmon are more vulnerable to temperature limitation in the MFIMW due to their lower association with water temperatures > 20°C (Handley ODFW unpublished data 2017). The basic quantity of habitat availability in terms of survivable stream temperatures across the riverscape segments used by juvenile Chinook Salmon must be met first before we expect to see any response to habitat restoration actions targeted to reduce density dependence through increased habitat quality.

LESSONS LEARNED AND RECOMMENDATIONS

Future Restoration

Lesson Learned – Restoration to improve high water temperatures is a slow process and it takes generations to affect changes in salmon and steelhead populations.

Distribution of juvenile salmonids, especially Chinook Salmon continues to be limited by summer stream temperatures. Future work should continue to focus on improving thermal conditions throughout the watershed to increase salmonid distribution downstream.

Continue with existing monitoring to determine if restoration affects salmonid populations as riparian plantings mature and planting techniques improve success of plantings.

Consider prioritizing restoration to reduce water temperature in areas of low abundance at the upper threshold of temperature limits. (presentation by Ian Tattam and Steph Charette to the John Day Basin Partnership in Spring 2023).

Monitoring

Lessons Learned –Watershed scale analysis helps us understand the overall picture of population status but may be too broadscale to pinpoint exact variables to affect change.

Look into habitat or geographic characteristics of Chinook Salmon redd locations in restored reaches to better understand what might be affecting productivity. Use these results to improve habitat conditions in marginal areas or in reaches where productivity could be increased.

Investigate if higher densities of steelhead redds in Camp Creek (compared to other MFIMW reaches) are related to restoration?

Investigate other methods for estimating steelhead spawner abundance, including estimates using PIT tags and/or side-scanning sonar (DIDSON). These methods will require dedicated (i.e., fully funded staff in addition to current staff).

Investigate environmental variables that may be affecting watershed scale Chinook Salmon productivity using the residual analysis methods described by Warkentin et al. (2022). Environmental variables to consider include flow, air temperature and water temperature metrics.

Planning

Lessons Learned – Continued involvement and participation in PNW IMW networks and partnerships provide valuable results about fish response to restoration actions in other basins that may be applicable to the MFIMW.

Many research projects were referenced in the report as "unpublished". Prioritize publication of results to document research, demonstrate effects of climate, habitat conditions, and restoration actions and reach wider audiences.

Investment in stable long-term funding and staffing for monitoring of all life-stages is crucial to the success of the MFIMW.

A combination of Covid pandemic effects on hiring and difficulty in housing in John Day has resulted in difficult circumstances for stable staffing.

Continue coordination with restoration practitioners to conduct pre-restoration monitoring in important locations of restoration activity like Camp Creek, Summit Creek and Phipps Meadow. Follow up as restoration is implemented to conduct post-restoration monitoring and analysis.

ACKNOWLEDGEMENTS

We thank the 2008-2023 Oregon Department of Fish and Wildlife John Day Fish Research team for consistent and quality data collection. We thank Jim Ruzycki and Kirk Handley who established the monitoring and analysis framework this research is built on, Melody Feden for leading steelhead adult monitoring, Chris Bare and Alex Woolen for leading adult Chinook Salmon monitoring, and Jeremy Henderson for leading juvenile monitoring at the RSTs. This research was supported by Pacific States Marine Fisheries Commission funds - administered by Ken Fetcho/Oregon Watershed Enhancement Board, and BPA funds - administered to project 1998-016-00 by Josh Ashline.

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CHAPTER 2: Quantifying Riverscape Productivity to Inform Limiting Factor Analysis and Guide Reach-based Restoration Goals

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ABSTRACT

Effective species conservation requires addressing threats associated with limiting factors operating across spatial and temporal scales. For Columbia Basin salmonids, limiting factors operating at the population scale are well understood, but information on how limiting factors impact a population at smaller spatial scales and across a riverscape is lacking. To identify spatially varying population limiting factors we quantified the spatial variation in Chinook Salmon and steelhead productivity metrics and habitat variables across mainstem rearing habitats. We then evaluated the relationship between productivity metrics and habitat variables. We found density-dependent and density-independent processes were limiting the population but parr-to-smolt survival, the most valuable indicator of progress towards recovery, was most strongly correlated with density-independent processes, mainly water temperature (R=-0.58). Monitoring across the riverscape revealed the impact of temperature on productivity was not spatially uniform and a central zone of temperature impact existed. Our results suggest restoration effectiveness will be maximized when efforts are successful at alleviating temperature limitation in the central zone of temperature impact.

INTRODUCTION

Over the last four decades, pursuit of native salmonid recovery in the Pacific Northwest has led to the extensive application of stream restoration aimed at addressing threats associated with population limiting factors occurring during freshwater life-stages. Identified threats include reduced habitat quality and quantity, degraded water quality, predation, reduced habitat complexity, reduced access to floodplain habitat and altered hydrology. For Chinook Salmon *Oncorhynchus tshawytscha* and steelhead *Oncorhynchus mykiss* returning to the Middle Fork John Day River (MFJDR) in-basin threats include degraded water quality (i.e., water temperature), altered hydrology, and reduced habitat quality and complexity. Of these, elevated stream temperature remains the most significant threat (Middle Fork IMW Working Group 2017). Our current understanding of salmonid population limiting factors is largely guided by research conducted at the population, watershed, or sub-watershed scale (e.g. Oregon Steelhead Recovery Plan 2009; Haring 2002; Andonaegui 2001).

While an understanding of limiting factors at the watershed scale can guide overarching goals and strategies for recovery, delving deeper into the specific challenges faced by salmonids, at a scale relevant to on-the-ground restoration actions, may enhance the precision and thus effectiveness of on-the-ground restoration efforts. The complex and dynamic nature of riverscapes can lead to significant variation in the environmental conditions, habitat quality and other factors that affect a population along a riverscape. As a result, individual factors limiting productivity may vary by reach and through time. For example, Torgersen et al. (1999) found holding adult Chinook Salmon were nonuniformly distributed but that their holding distribution was tightly correlated to water temperature. They concluded that the heterogeneity in the thermal spatial structure of the MFJDR may be responsible for the persistence of Chinook Salmon within the MFJDR despite water temperatures frequently exceeding Chinook Salmon upper tolerance levels (25 °C).

The impact of limiting factors can be more pronounced in certain areas compared to others based on local, within-watershed characteristics. Identifying reach-scale limiting factors and the magnitude of their individual and combined impacts on the population across space and time requires intimate knowledge of the life-history of the species of interest, survival bottlenecks within the species life-cycle, the location of habitats occupied during each life-stage, the spatial relationship of habitats occupied during sequential life stages, and finally, the density-independent and density-dependent effects of limiting factors associated with life-stages limiting population productivity. Recognizing, identifying, and incorporating, the non-uniform impact of population limiting factors is essential for developing successful restoration and management strategies that can address the specific challenges faced by salmonids across space and time.

Goals and objectives

To inform restoration efforts and a reach-scale limiting factor analysis our study objectives were to 1) quantify spatial variation in juvenile salmonid productivity, measured through abundance, growth, and survival, across mainstem salmonid rearing habitats, and 2) evaluate the relationship between juvenile salmonid productivity metrics and habitat variables.

Juvenile salmonid monitoring site selection

In 2020 and 2021 we sampled nine and 11 sites, respectively, in the MFJDR between river kilometers (RKMs) 94.5 to 114.2 (Figure 1). Monitoring sites were selected to maximize spatial coverage of Passive Integrated Transponder (PIT) tagged salmonids while also providing a sampling structure that would allow for effectiveness monitoring of two large-scale restoration efforts: Caribou to Vincent (Treatment 1) and Vincent to Davis (Treatment 2). Three sites were located in Treatment 1, three sites were located in Treatment 2, one site was located in a downstream control reach within the Oxbow Restoration Area (Control 1), and the remaining four sites were located in two connected control reaches (Control 2 and Control 3) located upstream of Treatment 2. The three control reaches exhibit a gradient of habitat conditions. Control 3, located upstream of Bridge Creek, has not undergone restoration, is actively heavily grazed, and lacks a riparian fence. Control 2, located between Davis Creek and Bridge Creek, also has not undergone restoration, lacks a riparian fence, but is no longer actively grazed. Control 1, located between Granite Boulder Creek and Beaver Creek has received extensive restoration (restored 2011 – 2015) and a wildlife riparian fence is maintained.

Figure 1. Location of the 2020 and 2021 parr monitoring sites located within each control (blue spectrum) and treatment reach (brown spectrum) in the MFJDR.

METHODS

Habitat monitoring

In 2020 we followed the channel unit classification and measurement methods outlined in the 2014 CHaMP Habitat Monitoring Protocol (CHaMP 2014) to conduct a habitat census. We classified and recorded the location of every channel unit within each control and treatment reach. We then used a sliding window analysis to count the number of pools, riffles, fast non-turbulent (FNT) and the total number of habitat units per 300 rolling meters. We selected a sliding window length of 300 meters to encompass habitat features contained in our largest juvenile salmonid monitoring sites (250 meters long).

Temperature monitoring

To quantify the spatial variation in water temperature throughout the study area we used water temperature data from *21 water* temperature loggers deployed and maintained by the Middle Fork John Day Water Temperature Monitoring (WTM) Subgroup (Feden and Bliesner 2022). Water temperature data was collected hourly and met minimum data standards set by the WTM Subgroup and outlined in Feden and Bliesner (2022). All temperature data were passed through a standardized QA/QC process and visually inspected for errors. Once QA/QC processes were completed we calculated the rolling 7-day average maximum stream temperature (7DADM; Sturdevant 2008) for each water temperature monitoring site.

Juvenile salmonid monitoring

To monitor juvenile salmonid density and deploy PIT tags for juvenile salmonid movement and growth monitoring, we conducted either single pass, or mark-recapture snorkel-herding at parr monitoring sites from 2019 - 2021. Sample timing varied among years due to crew availability, sampling conditions (i.e., flow and water temperature), and landowner access. Pilot study sampling was conducted in 2019 and consisted of sampling five locations, one in each of the treatment sections. In 2020 we sampled five of the ten sites in June, July, and August, four sites in July and August, one site in June and August and one site in July. In 2021 we sampled the 11 parr monitoring sites in June and August. During sampling we held captured fish in aerated buckets, we then anesthetized fish in 20 ppm Aqui-S (clove oil) and interrogated them for existing PIT tags. We weighed (g), measured fork length (mm), and, if not previously PIT tagged, we inserted a PIT tag in the fish's peritoneal cavity. In 2019 and 2020 we PIT tagged steelhead ≥ 70 mm and Chinook Salmon ≥ 65 mm and in 2021 we PIT tagged steelhead ≥ 65mm and Chinook Salmon ≥ 55 mm. We allowed all anesthetized fish to recover in aerated 5-gallon buckets until they regained equilibrium (approximately 5-10 minutes). Once recovered, we returned salmonids to the stream.

Juvenile salmonid abundance

We corrected raw catches of juvenile Chinook Salmon and steelhead from tagging reaches using the mean 2020 mark-recap capture efficiency (CE; Eq. 1, 2). We did not conduct mark-recapture sampling in 2021 due to the elevated stream temperature, and instead used the mean 2020 CE to estimate 2021 abundances. We then estimated fish density (Eq. 3) for each tagging site.

Estimation of abundance based on mean capture efficiency:

$$(1)\quad CE = \frac{1}{n} \times \sum_{i}^{n} \frac{R_i}{M_i},$$

$$(2)\quad N = \frac{C_1}{CE}$$

where,
i = mark-recapture event
C_1 = total number of fish caught during first pass
CE = mean capture efficiency
N = population estimate
R = number of recaptures in the second pass.
M = number of fish caught, marked and released in first pass

Site-level fish density:

$$(3)\ D_i = \frac{N_i}{d_i}$$

where,
D_i = fish density (fish/linear m) at site *i*
N_i = estimated abundance based on mean capture (or snorkeler efficiency) at site *i*
d_i = linear stream length (m) sampled at site *i*

Using the 2020 site-level fish densities we used linear regression to explore the relationship among survivorship and three habitat variables (water temperature, scour pool density, and riffle density) and two population productivity metrics (growth and density).

Juvenile Salmonid Growth

To calculate over-summer Chinook Salmon and steelhead growth rate (mm/day) we tracked changes in fork length of individual fish between June, July and August sampling events from 2019-2021. We identified individual fish using their PIT tag identification number. We summarized overall growth rate for the two species across growth periods and years. We then summarized Chinook Salmon July to August growth rate by sampling location and used linear regression to explore the relationship among growth rate and three habitat variables (water temperature, scour pool density, and riffle density) and two population productivity metrics (abundance and survival). We selected July to August growth rate by sampling locations for summarization because it was the growth interval with the largest sample size. We were unable to summarize steelhead growth rate by sampling location, due to a small sample size. For all growth analyses we only included individuals that were captured at the same sampling location during each sampling event used in the growth interval.

Survival analysis

We fit state-space Cormack Jolly-Seber models using Bayesian methods implemented with Markov chain Monte Carlo (MCMC) methods using program JAGS (Plummer 2003) invoked through program R (R Core Team 2021) to estimate 2019-2021 mainstem steelhead and Chinook Salmon and 2020 site-specific Chinook Salmon parr-to-smolt survival probability, here-in referred to as parr-to-smolt survival or survival (i.e. survival from a sampling event to the Galena PIT array).

The model incorporated capture data from juveniles PIT-tagged during July and August juvenile monitoring surveys, as well as stationary antenna detections recorded at the Galena, John Day River, Columbia River dams as well as detections recorded during the estuary trawl surveys. Across years and between species we observed variable detection efficiencies at each of the PIT arrays and therefore ran independent models for each species and year combination. We estimated survival of all Chinook Salmon tagged with a 12 mm PIT Tag. The site-specific survival estimates used linear regression to explore the relationship among survivorship and three habitat variables (water temperature, scour pool density, and riffle density) and two population productivity metrics (abundance and growth). We were unable to summarize steelhead survivorship by sampling location, due to a small sample size.

RESULTS

Juvenile salmonid monitoring

Juvenile salmonid abundance

In August 2020 and 2021 we conducted single pass or mark-recapture snorkel herding at 9 and 11 abundance monitoring sites, respectively. Mean (± 1 SD) 2020 mark-recapture capture efficiency was 0.30 ± .12, and 0.42 ± .18 for steelhead ≥ 80 mm, and juvenile Chinook Salmon*,* respectively.

We observed relatively low (≤ 1 fish/meter) steelhead ≥ 80 mm densities across all sampling sites in 2020 and 2021 (Figure 2) as compared to juvenile *O. tshawytcha* (Figure 3). At one site, located at RKM 109.7, we observed substantially higher densities (0.70 more fish/meter) in 2021 compared to 2020. At three sites, located at RKMs 105.9, 106.6, 108.8, we observed slighter higher densities (0.20 – 0.23 more fish/meter) in 2021 compared to 2020. At one site, located at RKM 114.1, we observed similar (0.06 more fish/meter) in 2021 compared to 2020. At the remaining four sites we observed reduced densities (0.25 – 0.51 less fish/meter) in 2021 compared to 2020.

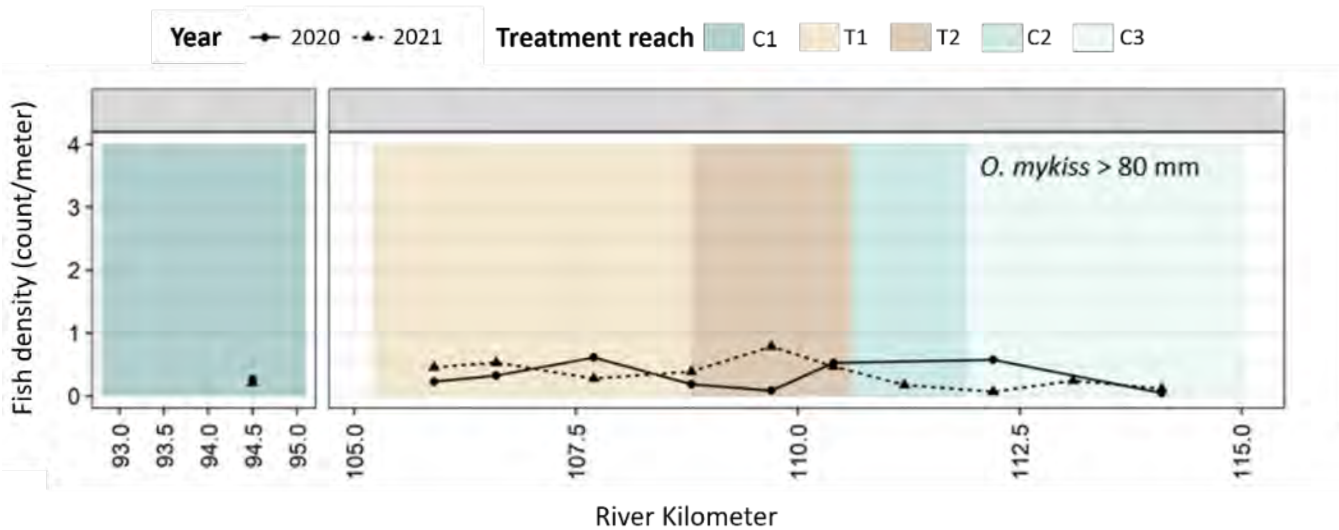

Figure 2. August density (fish/m) of steelhead ≥ 80 mm observed at each parr monitoring site in the MFJDR in 2020 (solid line) and 2021 (dotted line).

In 2020 we observed variable Chinook Salmon densities across sampling sites with fish densities ranging from 0.21 to 3.51 fish/meter (Figure 3). In 2021 Chinook Salmon densities were less variable, ranging from 0-1.5 fish/meter. At six sites, located at RKMs 94.5, 106.6, 108.8, 112.2, 110.4 and 111.2, we observed substantially lower densities (0.4 – 3.0 less fish/meter) in 2021 compared to 2020. At one site, located at RKM 114.1, we observed slight lower densities (0.2 less fish/meter) in 2021 compared to 2020. At the remaining two sites we observed higher densities (0.06 – 0.55 more fish/meter) in 2021 compared to 2022.

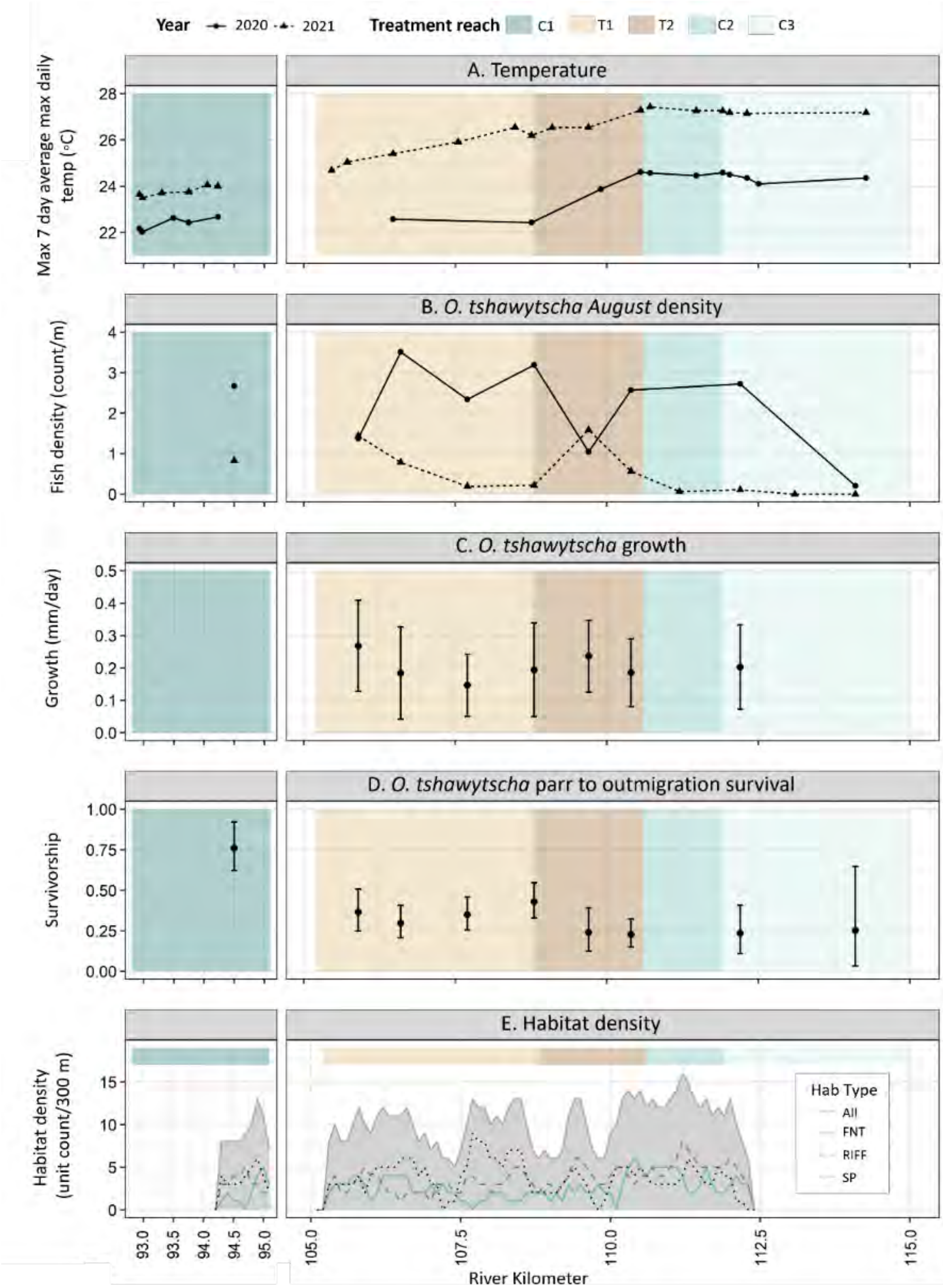

Figure 3. ***A)*** 2020 (solid line) and 2021 (dotted line) maximum seven day rolling average maximum daily water temperature calculated for each MFJDR temperature monitoring site *B)* August density (fish/m) of *Chinook* Salmon observed at each parr monitoring site in 2020 (solid line) and 2021 (dotted line). *C)* Over-summer (July to August) *Chinook* Salmon growth rates (mm/day ± SD) observed at each of the 2020 parr monitoring sites. of *D)* Parr-to-smolt survival rate (i.e., August tagging event to the Galena PIT array; estimate ± CI) recorded for each 2020 parr monitoring site. *E)* Density of fast non-turbulent (FNT), riffle (Riff), scour pool (SP) and total count (All) habitat unit types (unit/ 300 sliding meters) observed during the MFJDR habitat census survey. Control and treatment reaches are indicated by blue and brown spectrum-colored rectangles within each plot, respectively.

Juvenile salmonid growth

Across all sites, average mainstem Chinook Salmon and steelhead growth rates were consistent across years, and growth intervals (Figure 4). Growth rates were also comparable between species. Average mainstem Chinook Salmon growth rates ranged from 0.18-0.21 mm/day and steelhead growth rates ranged from 0.15-0.21 mm/day from 2019-2021. During the 2020 July to August growth interval (the year with the largest growth interval sample size) we observed a wide range of individual Chinook Salmon and steelhead growth rates. Chinook Salmon growth rates ranged from 0.06-0.44 mm/day and steelhead growth rates ranged from 0.0-0.36 mm/day. To explore if the observed variation in individual growth rates varied by site within mainstem rearing habitats we summarized Chinook Salmon growth rate by sampling site (Figure 3). We were able to summarize Chinook Salmon site-specific growth rates for seven out of the nine juvenile monitoring sites sampled in 2020. Chinook Salmon reach-specific growth rate sample sizes ranged from 7-36 individual fish. We were unable to summarize steelhead growth rate by sampling reach due to a small sample size at the majority of sampling sites (sample size range = 1-10 individual fish).

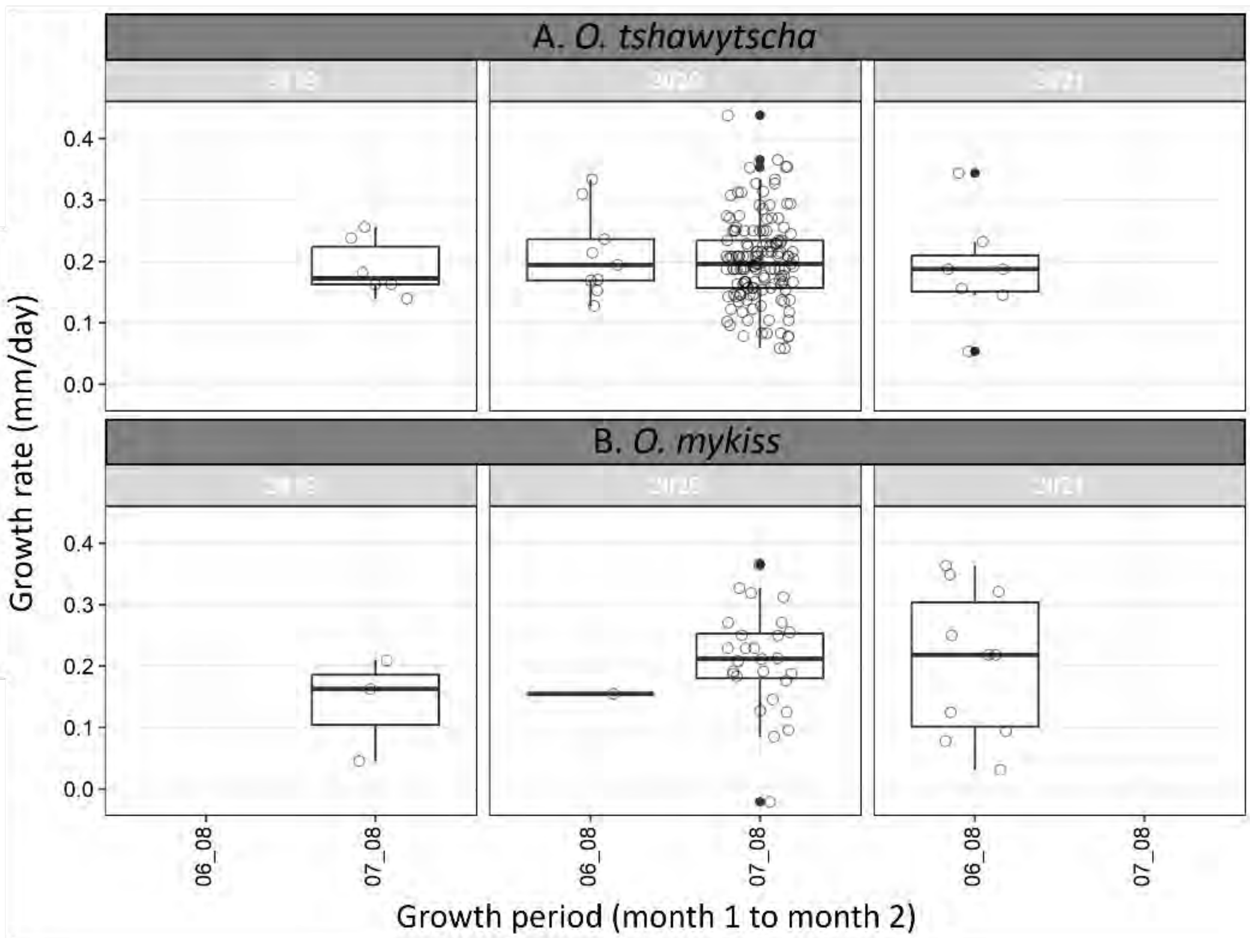

Figure 4. Chinook Salmon and steelhead over-summer growth rates observed across two growth periods, June to August and July to August, during three years of parr sampling in the MFJDR.

We observed variable Chinook Salmon average growth rates across sampling sites with site specific average growth rates ranging from 0.17- 0.27 mm/day. We observed the two highest growth rates (mean ± SD) at sites located at RKMs 105.9 (0.27 ± 0.07) and 109.7 (0.24 ± 0.06) and the two lowest growth rates at sites located at RKMs 106.6 (0.17 ± 0.07) and 107.7 (0.17 ± 0.05).

Survival analysis

We observed variability average parr-to-smolt survival of Chinook Salmon and steelhead tagged throughout the MFJDR in August in 2019 to 2021 (Figure 5). Parr-to-smolt survival (mean ± SD) of Chinook Salmon tagged in August ranged from 0.08 (± 0.04) to 0.42 (± 0.14). Survival of steelhead tagged in August was similar to Chinook Salmon survival proportions and ranged from 0.13 (± 0.13) to 0.45 (± 0.23). Our sampling schedule and sample sizes of fish tagged with 12 mm PIT tags precluded us from estimating survival of fish tagged in July across all years. We estimated the survival of Chinook Salmon tagged in July in 2019 and 2020, but not 2021 and estimated the survival of steelhead tagged in July in 2020, but not in 2019 and 2021. In 2019 the survival of July tagged Chinook Salmon was comparable to the survival of August tagged Chinook Salmon. In 2020 the survival of July tagged Chinook Salmon was significantly (as indicated by non-overlapping credible intervals) lower than the survival of August tagged Chinook Salmon, indicating high over-summer mortality. In 2020 the survival of July tagged steelhead was similar to the survival of August tagged steelhead. It is important to note however, that the small sample size of PIT-tagged steelhead made detecting a survival difference, if present, very difficult (as indicated by the large confidence intervals).

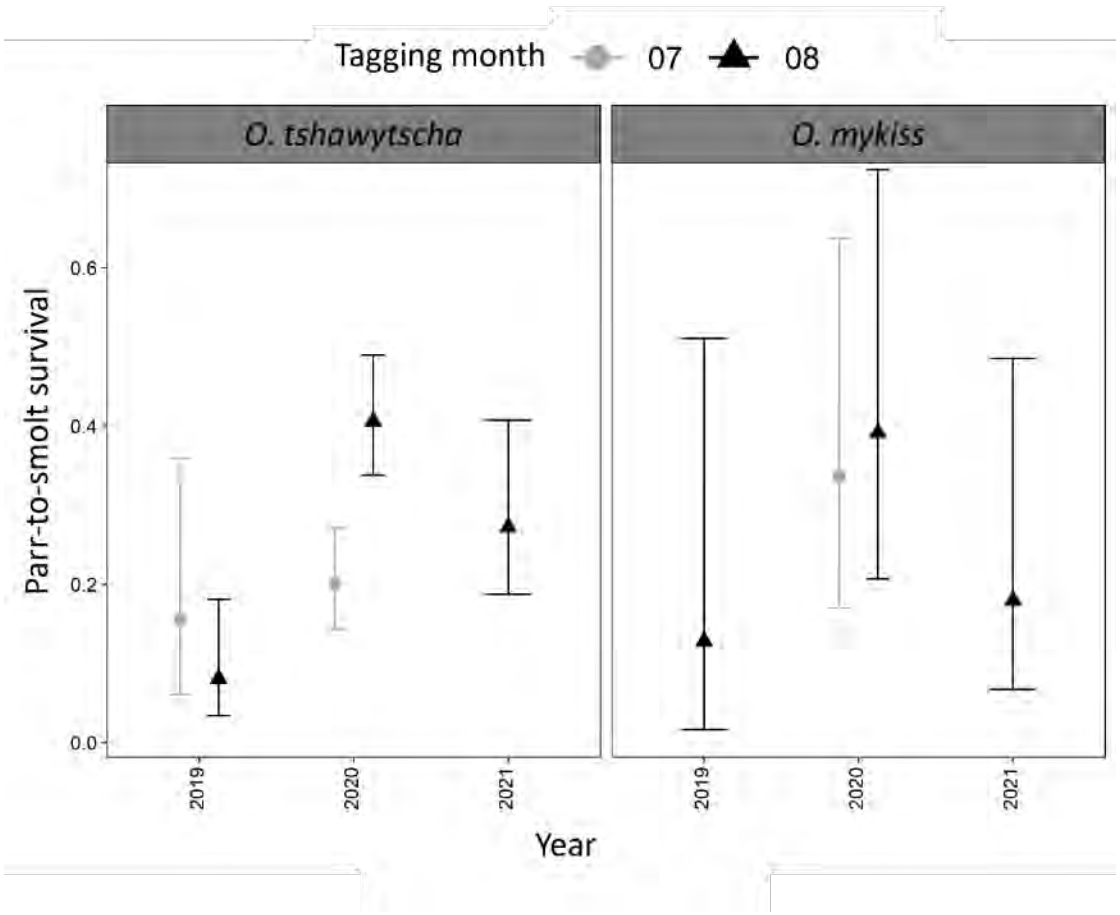

Figure 5. 2019 – 2021 Chinook Salmon and steelhead parr-to-smolt survival rates estimated for parr PIT tagged during July (grey circles) and August (black triangles) tagging events in the MFJDR.

To investigate heterogeneity in survival proportions across mainstem rearing habitats we calculated site specific survival proportions of Chinook Salmon tagged in August at all nine of the juvenile monitoring sites sampled in 2020 (Fig. 3). We were unable to calculate steelhead site-specific survival estimates due to a small sample size of PIT tagged and re-detected steelhead.

Parr-to-smolt survival (mean proportion ± SD) of Chinook Salmon PIT tagged in August at sites located upstream of RKM 105 were moderately variable and ranged from 0.21 ± 0.04 to 0.43 ± 0.05. Of the sites located upstream of RKM 105 the fish tagged at the site located at RKM 110.4 had the lowest survival and fish tagged at the site located at RKM 108.8 had the highest survival. Parr-to-smolt survival of Chinook Salmon PIT tagged in August at the farthest downstream site located at RKM 94.5 was significantly higher than any of the other eight tagging sites (RKM 94.5 survival probability = 0.76 ± 0.07). This site is located within the Oxbow Conservation Area, a region that underwent extensive restoration over the course of 5 years from 2011 to 2015.

Habitat and productivity metric comparisons

We examined the linear relationships among three habitat variables (i.e., water temperature, pool density, and riffle density) and three 2020 population productivity metrics (survivorship, fish growth rate, and fish density) (Figure 6).

Stream temperature, measured as 7DADM, had a moderate negative linear relationship with parr-to-smolt survival (R = -0.58), a weak negative linear relationship with fish density (R = -0.41), and a negligible linear relationship with fish growth (R = 0.08). Pool density, measured as the number of pools per 300 meters, had a moderate negative linear relationship with fish growth (R= -0.50), and a weak linear relationship with parr-to-smolt survival (R= 0.20) and fish density (R = 0.28). Riffle density, measured as the number of riffles per 300 meters, had a strong positive linear relationship with parr-to-smolt survival (R = 0.64), a weak, positive linear relationship with fish growth (R = 0.45) and a weak linear relationship with fish density (R = -0.28). Fast non-turbulent (FNT) density, measured as the number of FNT's per 300 meters, had a very strong negative linear relationship with parr-to-smolt survival (R= -0.81) and a negligible linear relationship with fish growth (R= -0.07) and fish density (R = -0.02).

To investigate how population productivity metrics were related to each other we examined the pair-wise linear relationships between the three population productivity metrics, parr-to-smolt survival, fish growth and fish density (Figure 7). Parr-to-smolt survival had a negligible linear relationship with both fish density (R=0.18) and fish growth (R=-0.035). Fish density had a moderate, negative linear relationship with fish growth (R = -0.74).

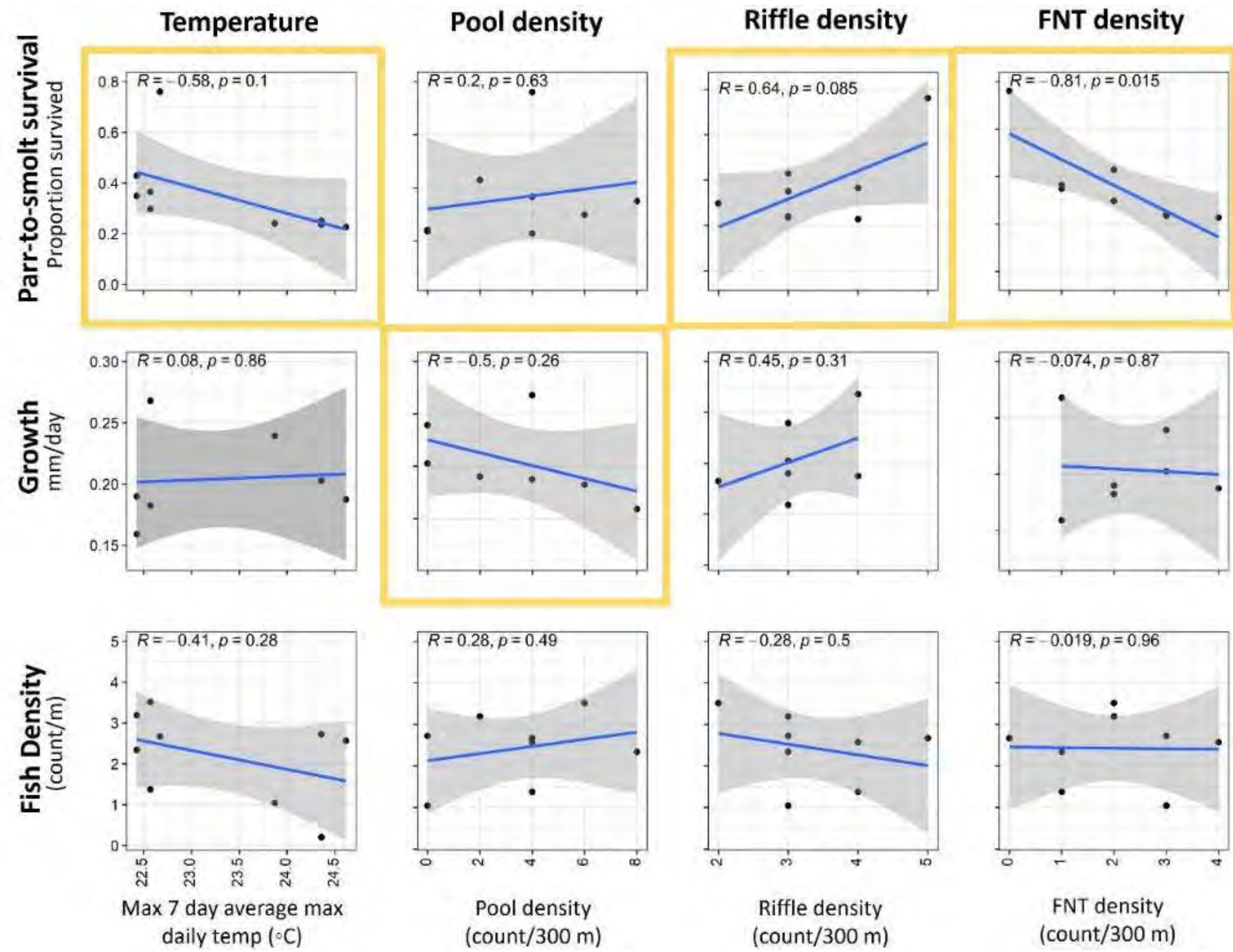

Figure 6. Linear relationship between three habitat metrics (water temperature, pool density and riffle density) and three Chinook Salmon population productivity metrics (parr-to-smolt survival, growth and fish density). Shaded bands reflect 95% linear regression confidence intervals. Linear relationships with a correlation coefficient ≥ ± 0.50 are indicated by a yellow box.

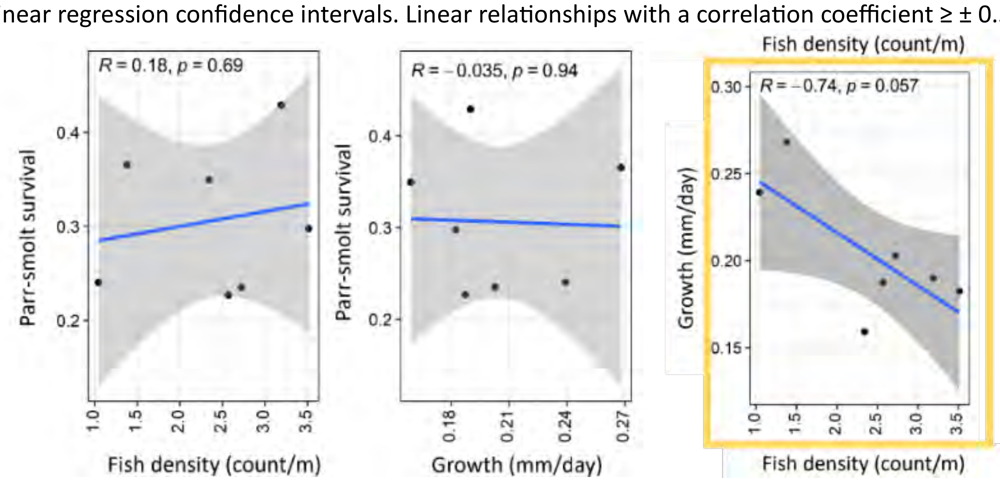

Figure 7. Pairwise linear regression comparisons of three *Chinook Salmon* population productivity metrics (fish growth, parr-to-smolt survival, and fish density). Shaded bands reflect 95% linear regression confidence intervals. Linear relationships with a correlation coefficient ≥ ± 0.50 are indicated by a yellow box.

DISCUSSION

Riverscape monitoring of Chinook Salmon population productivity and habitat metrics provide evidence that both density-dependent and density-independent forces are impacting Chinook Salmon in the MFJDR.

Density-dependent processes

The negative correlation between growth and fish density and the negligible correlation between measured habitat metrics, including water temperature, and fish density, provides evidence that density-dependent processes are present within the MFJDR. Our findings are similar to Teichert et al. (2011) who concluded density-dependence was manifesting through growth as evidenced by a negative relationship between fish density and fish growth and a negligible relationship between other measured habitat variables. Observing density-dependent growth is not uncommon in salmonid populations. Grossman and Simon (2019) reviewed 199 published datasets to assess the relative frequency of various density-dependent processes in salmonids and found 71% of the datasets showed density-dependence in growth. For stream-dwelling salmonids density-dependent growth can occur across a wide range of fish densities including at very low (< 1 fish/m^2) fish densities, and is primarily attributed to intraspecific competition for habitat, or competition for available food resources (see Grant and Imre 2005; Myrvold and Kennedy 2015). Density-dependence can also exhibit spatial heterogeneity because the mechanisms (e.g., water temperature, habitat quality, anthropogenic disturbances, nutrient input etc.) leading to density dependence are spatially variable. Stream restoration targeted at alleviating the mechanism leading to density-dependence, such as pool-riffle creation (increasing complexity), island formation (increasing complexity, and large woody debris additions (increasing cover and complexity) will thus be most effective in reaches where density dependence is strong and less effective in reaches where density-dependence is less prevalent. Of the sites examined, stream restoration targeted at alleviating density dependent growth factors is hypothesized to be most effective near river kilometers 106.6 (~1.3 km upstream of Caribou Creek) and 107.7 (~0.5 km downstream of Dead Cow Gulch) and least effective at river kilometers 105.9 (0.6 km upstream of Caribou Creek) and 109.7 (0.3 km downstream of Vinegar Creek).

Our most valuable indicator of progress towards recovery through restorative actions throughout the MFJDR is improvements in parr-to-smolt survival. Despite useful observations made regarding density-dependent processes such as growth, we did not find convincing evidence of density-dependent parr-to-smolt survival. These results were surprising given a large body of literature linking density-dependent growth to density-dependent mortality (see Rose et al. 2001). These findings were also in direct contrast to Connor and Tiffan (2012) who found a proportional relationship between reduced growth during freshwater rearing and parr-to-smolt survival. To explain our results, we offer three hypotheses. First, we hypothesize observed density dependent growth did not result in a size advantage at smolt age. That is, it is possible the observed density-dependent growth was only seasonally present, and that fish in high density reaches, during the summer, were able to match or exceed the growth rates of fish located in low-density summer-rearing reaches at some point between the end of summer and outmigration. Future research should work to understand if size at outmigration is linked to survival and how growth rates observed during summer rearing might influence this. Secondly, we hypothesize density dependent growth did result in a size advantage and thus presumably a survival advantage (Connar and Tiffan 2012), but that the survival advantage was realized farther downstream in the outmigration route and was therefore not captured in our survival analysis. Finally, we hypothesize that density dependent growth rates did result in a size advantage and thus presumably a survival advantage, but that the density dependent influence on survival was muted by a stronger density independent influence on survival. This hypothesis is supported by other work within the basin. Bliesner (2023) found evidence for, but weaker than expected, density-dependence signals in the stock recruit curve for Middle Fork John Day Chinook Salmon, noting that near-recruitment failures have occurred at low redd abundances when increased recruits per spawner were expected if density-dependent factors alone were driving population dynamics. Bliesner (2023) went on to hypothesize that high stream-temperatures during summer rearing created extremely poor conditions across the riverscape resulting in poor recruitment, independent of density.

Density independent processes

Water temperature, riffle density and FNT density, had the strongest correlations with parr-to-smolt survival. Temperature and FNT density were negatively correlated with parr-to-smolt survival, while riffle density was positively correlated. Interestingly, riffle and FNT density had a weak (R = -0.36) negative linear correlation. Within the MFJDR, FNT habitat units are characterized by wide, uniform channels of moderate depth with laminar flowing water and are highly susceptible to solar heating. The negative relationships between FNT density and parr-to-smolt survival is likely an indirect measure of the effect of water temperature on parr-to-smolt survival. The positive correlation between riffle density and parr-to-smolt survival may be correlated to increased survival in reaches with increased habitat complexity, and reduced water temperatures. Within the MFJDR riffle habitat units represent units with increased habitat complexity and cover and often contain many small pockets of pool-like habitat located behind larger substrate and along stream margins. The downstream end of riffle units (regardless of the unit the riffle flows into) also create small plunge pools where fish are able to feed on a conveyer belt of food brought by the faster moving riffle water.

The negative correlation between water temperature and parr-to-smolt survival provides evidence that water temperature, an environmental density-independent factor, is influencing population productivity through decreased parr-to-smolt survival. This finding, coupled with the absence of supporting evidence for density-dependent survival aligns with the hypothesis proposed earlier and articulated by Bliesner (2023) that density-independent factors have an overwhelming impact on population productivity attenuating the signal of a density-dependent relationship.

Within the MFJDR Bliesner (2023) (Chapter 1) detected an inconsistent temporal influence of water temperature, indicating that high summer water temperatures created poor conditions across the riverscape during some brood years. Our findings build on this and demonstrate the effects of elevated stream temperatures also vary spatially, likely influencing parr-to-smolt survival in all years. In 2020, riverscape monitoring of parr-to-smolt survival revealed that elevated stream temperature had the largest impact on productivity in reaches located upstream of RKM 109 (Vincent Creek), as indicated by reduced parr-to-smolt survival in those reaches. The observed trends likely represent a ‘central zone’ of water temperature influence, which we anticipate to expand and contract from year to year. In years characterized by exceptionally high temperatures and low water quantities, such as conditions observed in 2021, the spatial extent of negative thermal impact (i.e., central zone of impact) will expand, while in milder temperature years, and adequate summer flows the central zone of impact is anticipated to diminish.

Implication for restoration

Across monitoring scales, we detect a strong density-independent and a weak density-dependent impact on population productivity. Bliesner (2023) (Chapter 1) hypothesized water temperature was the density-independent parameter reducing the number of smolts per spawners in low escapement years and our results provide further evidence supporting this hypothesis. The overwhelming impact of water temperature on productivity suggests that detectable benefits of restoration aimed at increasing population productivity will only be realized after limitations of stream temperature are addressed. Therefore, addressing stream temperature, through passive and active restoration actions, continues to be a vital strategy for recovery of salmonids in the MFJDR. Given the spatial heterogeneity in the impact of water temperature across the riverscape we recommend two strategies to address water temperature limitations. First, we suggest protecting and expanding current cool-water inputs and thermal refugia's. Within the MFJDR thermal refugia is found within and at the confluences of cool water tributaries (i.e., Granite Boulder, Vinegar, Davis, Dead Cow and Deerhorn creeks etc.) and in the MFJDR downstream of RKM 95.0 (Granite Boulder Creek). Specific restoration actions to protect and expand cool-water thermal refugia should include:

- i) Maintain or improve connectivity to cool-water tributaries (i.e, Caribou Creek).
- ii) Protect and expand tributary riparian corridors via riparian fencing and plantings near the stream margins.
- iii) Protect and expand MFJDR riparian corridors at, and downstream of, tributary confluences via riparian fencing and plantings near stream margins.
- iv) Strategically place wood structures to deflect mainstem water and capture tributary water at confluences, with the goal of expanding the volume of cool-water plumes created at confluences.
- v) Increase the habitat complexity (to increase the carrying-capacity) in tributaries and in the MFJDR near tributary confluences.
- vi) Downstream of RKM 95.0 maintain the riparian fence and continue to invest in a viable planting strategy, which may include bringing in topsoil to improve the success of plantings.

Second, we recommend implementing restoration aimed at alleviating high water temperature within or upstream of the central zone of temperature influence. Observed reduced parr-to-smolt survival upstream of RKM 109 (Vincent Creek) indicates this area is a central zone of temperature influence. Specific restoration actions aimed at shrinking the central zone of temperature influence should include:

- i) Convert long FNT habitat units into a series of pool/riffle habitat units.
- ii) Narrow the channel through island formations.
- iii) Reconnect the stream to the floodplain to facilitate the reestablishment of a riparian corridor.
- iv) Develop and execute a planting strategy that prioritizes planting species that will grow quickly and provide stream shade (i.e., alders), followed by those that will create a diverse and sustainable riparian community (i.e., willows and cottonwoods etc.).

Addressing water temperature limitations through restoration will also indirectly reduce density-dependence. Kaylor (2023) observed parr dispersing from inhospitable warm environments to hospitable cool environments, which we hypothesize is leading to crowding and density-dependence. Ultimately, shrinking central zones of temperature influence and expanding the area of, and carrying capacity within, cool-water refugia's is hypothesized to increase suitable rearing habitat and subsequently indirectly reduce density-dependence in current rearing habitat, compounding to improve overall population productivity.

Chinook Salmon as an indicator species

While the emphasis of the presented data is primarily on Chinook Salmon, we propose extending its application to guide the restoration and management efforts of steelhead as well. The use of Chinook Salmon as an indicator species for steelhead is a pragmatic approach to overcome challenges in obtaining fine-scale information on steelhead due to limited sample sizes. This strategy involves leveraging the knowledge and characteristics of Chinook Salmon populations to gain insights into the environmental conditions that may also affect steelhead.

Chinook Salmon and steelhead share similar habitat and environmental requirements over the course of their life cycles. By studying Chinook Salmon, which are more abundant, easier to monitor, and more sensitive to environmental influences such as changes in temperature, we can extrapolate information about limiting factors that are crucial for both species. This can provide valuable information to guide effective restoration efforts for both species.

The success of this approach relies on the assumption that the environmental conditions influencing Chinook Salmon are also indicative of those affecting steelhead. By continuing to monitor both steelhead and Chinook Salmon at a coarser scale we can verify if trends in both populations are proportionally responding to environmental changes.

Restoration effectiveness monitoring

Our research represents a comprehensive, adaptive monitoring strategy that simultaneously builds foundational knowledge of the mechanisms limiting salmonid populations, at a scale relevant to restoration, it also provides the necessary data to apply a Before-After-Control-Impact sampling design to assess the effectiveness of ongoing restoration activities on population productivity. Presented findings represent conditions prior to restoration on some reaches of the MFJDR. Continued monitoring will be required to track the effectiveness of recently implemented restoration activities at alleviating density-independent and density-dependent forces impacting population productivity across the riverscape.

Current restoration effectiveness monitoring focuses on measuring population-level productivity responses. Population productivity is an in-direct response of environmental change (i.e., working through temperature). Future monitoring should consider incorporating dedicated thermal mapping and monitoring including spatially continuous water temperature monitoring as well as monitoring temperature throughout the water column (to detect thermal refugia created via thermal stratification). Thermal mapping and monitoring would complement productivity monitoring and provide a direct method to quantify restoration effectiveness.

Study limitations

The presented results focus on monitoring density-dependent and density-independent impacts on a single life-stage – over-summer rearing of Chinook Salmon parr. Our provided recommendations are therefore specific to alleviating population productivity limitations occurring during over-summer parr-rearing. Maximizing restoration effectiveness will require identifying and addressing limiting factors during the most vulnerable life stage and those that persist across multiple life stages. Additional research quantifying riverscape population productivity limitations occurring during other freshwater life-stages is a critical knowledge gap and needs further investigation.

Lessons Learned and Recommendations

1. Tracking the spatial heterogeneity in density-independent and density-dependent factors impacting population productivity complemented and provided finer-scale restoration recommendations than population monitoring alone.
    a. Following Fausch et al. (2002) recommendations we recommend expanding the spatial extent of monitoring to better incorporate the spatial heterogeneity across the riverscape. Working towards a continuous view of how processes are interacting in the riverscape and across life-stages allows us to identify unique and rare features that are disproportionally impacting populations.
2. Using Chinook Salmon as an indicator species for steelhead is a pragmatic approach to overcome challenges in obtaining fine-scale information on steelhead due to limited sample sizes.
3. Restoration effectiveness will be maximized when it is implemented to 1) protect and expand thermal refugia or 2) address water temperature limitations in central zones of temperature influence.
    a. Incorporating thermal mapping and monitoring of thermal refugia and central zones of temperature influence into ongoing monitoring will further refine the location of where restoration will be most effective.
    b. Restoration that alleviates water temperature limitations will also reduce density dependence through indirect mechanisms.
4. Continued riverscape monitoring will be required to track the effectiveness of recently implemented restoration activities at alleviating density-independent and density-dependent forces.
5. The data presented in this report represents a small fraction of the available data on the relationships between juvenile salmonids and their rearing environments. Analyzing and publishing additional available data is solely dependent upon dedicated funding and personnel time.

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CHAPTER 3: Patterns of Spring Chinook Salmon Fry Emergence & Dispersal Across the Middle Fork John Day River Basin

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ABSTRACT

There is strong selection on salmon emergence timing to maximize survival through alignment with long-term patterns of optimal rearing conditions, which could be disrupted by climate change induced shifts in temperature, flow, and precipitation patterns. Timing of emergence relative to high flows, available floodplain habitat, and distance dispersed from redds could have profound effects on patterns of growth and survival of juvenile salmon. We sampled emerging spring Chinook Salmon (Oncorhynchus tshawytscha) fry from sites distributed across the mainstem of the Middle Fork John Day River from 2019 to 2022 to determine emergence windows and to characterize longitudinal and lateral dispersal from redds using genetic parentage assignments to link individuals to their maternal parents spawning location. Sampling occurred within areas affected by in-stream and floodplain restoration that are in various stages of recovery and implementation. We found that the fry emergence window occurred from mid-March to mid-May, which has not changed from a similar study completed over 40 years ago. Our results indicate a slight declining trend in annual cumulative thermal units during Chinook Salmon egg incubation period, although long-term temperature records are limited. We also found that fry only dispersed in the downstream direction, with the median recorded dispersal at 0.8 km (95% range: 0.05 - 12.6 km), although many fry dispersed much further and up to 20 km. We also found that restoration stage and variable flow patterns, can affect the type of habitat that fry utilize. Most fry dispersed less than a kilometer from where they emerged and most heavily utilized floodplain habitat that was adjacent to the main channel. These findings highlight the importance of considering the arrangement of spawning and rearing locations for restoration planning and implementation, suggesting that efforts targeting the fry life stage may be most effective just downstream of concentrated spawning.

INTRODUCTION

Juvenile salmonids experience high rates of mortality during their early life stages, including the fry stage (Einum and Fleming, 2000; Skoglund et al., 2011), when they have recently emerged from nesting sites called ‘redds’. Understanding patterns of emergence and dispersal during early life stages of salmonids is critical for population recovery, and additionally, can improve and inform restoration practices. Early life stage survival depends on numerous factors, including the availability and access to resources (e.g. prey, cover, flow refugia) (Kaylor et al., 2021; Skoglund et al., 2011; Jenson and Johnson, 1999). Access to resources can depend on population density (Bujold et al., 2004) and available territory (Skoglund et al., 2011), which is determined by when and where a fry emerges in a heterogeneous riverscape (Kaylor et al., 2021). Spring Chinook Salmon (*O. tshawytscha*) fry emerge from their redds throughout the spring when snow melt, precipitation and temperature can be highly variable throughout and among years (Jenson and Johnson, 1999). The variability of flow can determine when inundation of floodplain habitat occurs (Kaylor et al., 2021) allowing an increased amount of desirable habitat to be accessible to fry (Beechie et al., 2005).

The timing of floodplain inundation relative to emergence timing could be critical for fry growth and survival (Durant et al., 2007).

Emergence timing is determined by when redds were placed the previous fall and the thermal regime experienced at that locality throughout the winter (Jones et al., 2015, Sparks et al., 2019), both of which have the potential to shift due to climate change. Although when studied, it appears that spawn timing within a population remains consistent (Lisi et al., 2013, Quinn et al., 2016) indicating that it is a fixed trait (Sparks et al., 2019). Yet the timing of egg development is highly dependent on temperature (Kaylor et al., 2020, Steel et al., 2012, Whitney et al., 2012, Sparks et al., 2017) and therefore subject to change with changing climate conditions. The timing of adult salmon spawning evolved to optimize competitive and environmental conditions for their offspring (Murray et al., 1990; Web and McLay, 1996). A changing climate has highlighted the possibility that emergence timing could be decoupled from the most favorable environmental conditions for survival (Jones et al., 2015, Sparks et al., 2017).

After emergence from redds, fry disperse longitudinally and laterally across the freshwater landscape (Monnet et al., 2022). We refer to longitudinal dispersal as 'with the flow' in the upstream or downstream direction and lateral dispersal as 'perpendicular to the flow' to margins or floodplain habitat. Fry disperse from redds in order to find shallow, low-velocity, areas with specific cover-types (Beechie et al., 2005). Additionally, they disperse in order to decrease localized density and avoid competition (Skogland et al., 2011). Salmonid adults are highly fecund and will commonly place redds in overlapping locations (Beechie et al., 2008), resulting in a high density of fry at those locations. Density dependence is one of the primary mechanisms regulating growth for salmonids (Spalding et al., 1995; Anderson et al., 2008), therefore dispersing offers the opportunity to reduce competition and increase growth (Bujold et al., 2004, Chapter 4). Dispersal can offer the possibility of more favorable habitat and growth conditions (Armstrong et al., 2010), and escape from sub-optimal conditions (Hahlback et al., 2022). However, dispersal may be limited by environmental (e.g. barriers, flow conditions) and biological (e.g. size, predators) constraints. Understanding dispersal patterns, drivers, and limitations at every life stage can inform restoration practices in order to increase the potential of population recovery.

Decades of land use change has caused many river systems to become channelized and disconnected from their floodplain (Pollock et al., 2007), including the Middle Fork John Day River (MFJDR). This has decreased available slow-water habitat optimal for the fry life stage (Beechie et al., 2008). Restoration practices that aim to reconnect the floodplain in order to increase water retention and riparian vegetation could also impact the dispersal and density conditions for salmonid fry by increasing habitat and limiting the need for dispersal. In the MFJDR, several sections exist in varying degrees of restoration including none, passive, active, and recovery. Understanding how restoration impacts fish populations on micro and macro scales for a variety of life-stages is essential in order to implement adaptive management strategies required as an Intensely Monitored Watershed (IMW).

Emergence timing was studied in the John Day River basin in the late 1970s and early 80s (Lindsay et al., 1985). For this study, we aim to determine if emergence timing has shifted since it was last studied over forty years ago. This was done by sampling areas of high redd density several weeks prior to estimated emergence to capture the earliest estimated date of emergence (Kaylor et al., 2021). We sampled in several sections of the MFJDR until it was determined that emergence was over (≤5% fry captured were less than 37mm in FL). The channel habitat and cover type where fry were captured was recorded to later be compared to flow conditions and proximity to the main channel. We also recreated the temperature graphs from Lindsay et al., 1985 and completed an updated temperature analysis from more recent data to determine if there has been a temperature shift during the spring Chinook Salmon egg incubation period (Sept – May). Then, using genetics-based parentage assignments, we also characterize dispersal patterns from redd locations and habitat utilization by flow patterns. Insight into the complexities of emergence and dispersal can be applied to salmonid populations across their range to improve ongoing restoration strategies and meet management goals in order to restore wild salmonid populations.

METHODS

Site Selection

Spring Chinook Salmon spawn in the headwaters of the MFJDR. Most recently, the areas with the highest densities occur approximately from river kilometer 80 to 120 (Bare et al., 2021). Within this range, we sampled several river sections that have undergone different levels of restoration practices (Figure 1):

1) Upstream of the Middle Fork Forrest Conservation Area (MFFCA)[1] (rkm >111, upstream of Placer Gulch): No restoration, private land
2) MFFCA (rkm 106-111, ~Caribou Creek to Placer Gulch): LWD placement, riparian planting, channel reconstruction, floodplain reconnection and increased roughness from 2017 to 2022. Large scale restoration in V2V section in summer 2022 after fry sampling had occurred.
3) Between MFFCA and Oxbow Conservation Area (OCA) (rkm 100-106, ~Windlass Creek to Caribou Creek)[1]: no restoration, USFS constricted canyon section
4) OCA (rkm 93-99, ~Beaver Creek to Windlass Creek): major active restoration from 2011-2016 comparable to MFFCA
5) Dunstan Conservation Area (DCA) (rkm 83-88, Horse Creek to Big Boulder Creek): mainly passive restoration, including fencing, with some off channel floodplain reconnection
6) Camp Creek area (rkm 80-81): No restoration at time of sampling.

Within these areas, fry sampling sites (Figure 1) were chosen based on the previous year's redd density. In order to maximize our chances of catching fry, we aimed for areas with the highest redd density in sections of several hundred meters. In 2020 and 2021, genetics were collected from adult Chinook Salmon carcasses, with the goal of matching fry to parent samples. An effort was also made to sample in areas where genetic sampling from adult carcasses was successful, although this generally coincided with areas of high redd density. The fry samples from 2022 have not yet been analyzed for genetic pairings.

[1] Upstream of FCA and Between FCA and Oxbow sections were only sampled for adult carcasses, all other sites were sampled for carcasses and fry.

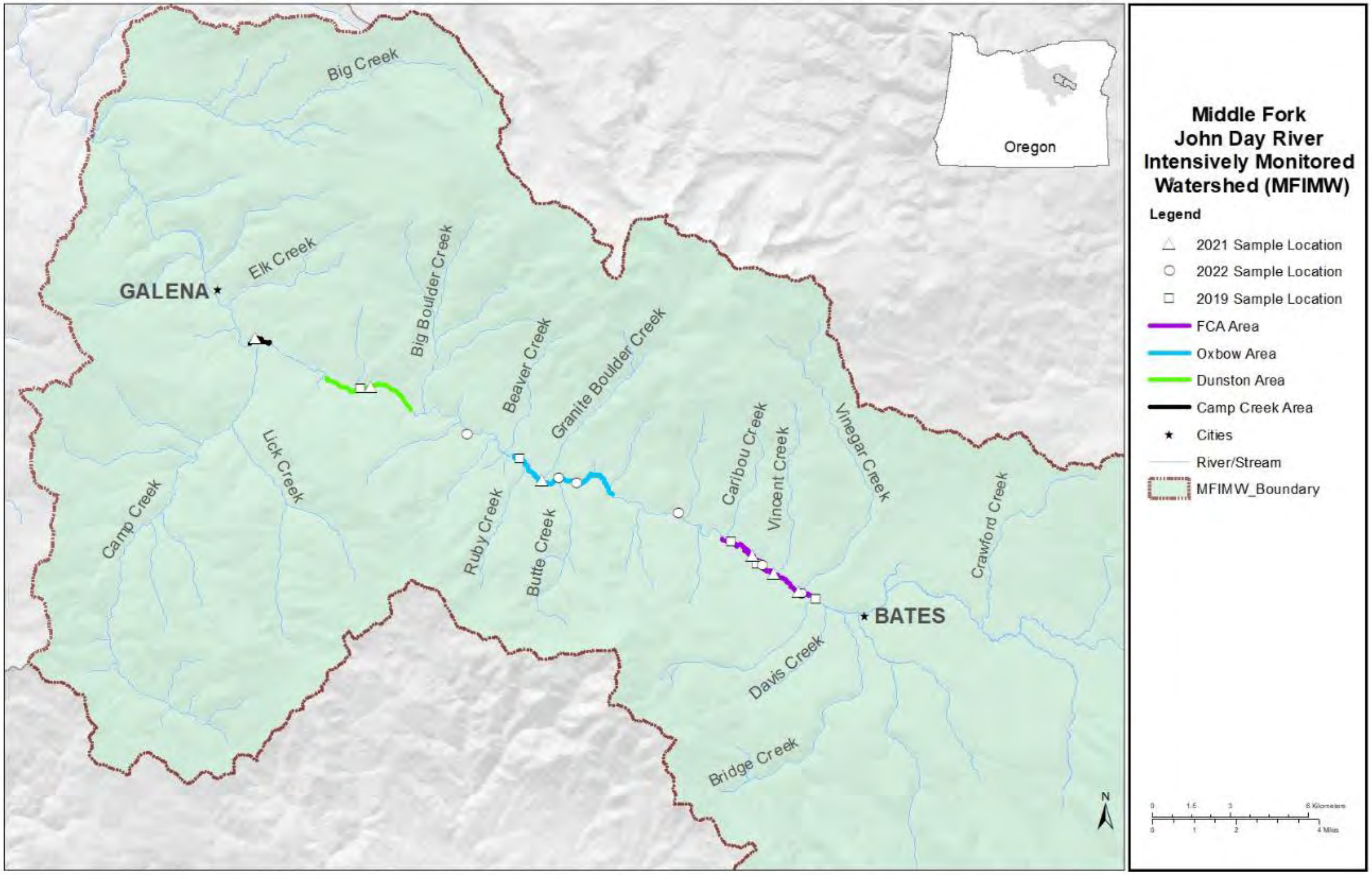

Figure 1. Map of the Middle Fork John Day basin showing fry sampling locations by year. The highlighted sections of the river show the areas that differ due to intensity and timing of restoration. The points represent general areas of sampling, not exact locations.

FIELD METHODS

Fry Collection and Handling

Collecting fry proved challenging and elusive, but through the several years of sampling, we adapted to what methods worked best. In 2019, snorkeling and seining was attempted but the water was generally too turbid for snorkeling. Seining through slow, vegetated water worked for some sites on the MFFCA, but no other areas had successful fry detection, despite attempts across the basin. Sampling in 2020 did not occur due to employee turnover and global events. We resumed in 2021 and had a very successful year. Fry were captured by sweeping dip nets through vegetated areas at and near areas with the highest redd density. Fry were successfully captured in the MFFCA, OCA, DCA, and Camp Creek areas (Figure 1). In 2022, dipnet sampling occurred in the MFFCA, OCA, and DCA determined by redd distribution the previous Fall. Only 28 redds were counted in the MFJDR basin during the fall of 2021, due to a combination of a lower-than-average cohort from 2017 and a heat dome weather event that impacted most of Oregon. It was difficult to detect increased mortalities due to heat, but ODFW and CTSWRO samplers did note that adults looked very sluggish and direct predation from otters was witnessed very early in the holding season.

Every year, sampling occurred in the river channel and floodplain (when available). The habitat feature fry were captured in was recorded (margin, side channel, alcove, etc), along with cover type and a GPS point. Captured fry were kept in buckets with aerators before being processed. In 2021, it was determined that anesthetizing the fry led to a higher accuracy fork length measurement (in 2019, only total length was collected). Fish were anesthetized with Aqui-S (clove oil), measured to the nearest millimeter (mm), and a fin clip genetic sample was collected. Processed fry were then placed back into an aerated bucket and allowed to recover for 5-10 minutes. Recovered fry were released in slow, vegetated areas as close as possible to where they were found.

Adult sample collection

Spring Chinook Salmon spawn annually in September in the MFJDR, where ODFW and several partner agencies survey for redds, live adults, and carcasses. During these surveys, carcasses encountered are sampled for sex, length (MEPS, medial eye to posterior scale), and scales are collected. During surveys in 2020 and 2021, genetic material was also collected that we could later match to fry and parr offspring. Samplers were instructed on how to evaluate the freshness of each carcass they encountered, then collected a tissue sample from the operculum (relatively fresh carcass) or heart (less fresh carcass). An operculum sample was deposited onto Whatman paper and allowed to dry, while a heart tissue sample was preserved in a vial of ethanol.

GENETIC SAMPLING AND ANALYSIS

Fry and adult genetic materials were sent to the CRITFC's Hagerman Genetics Laboratory to be genotyped. DNA was extracted and sequenced, then one round of PCR was done to amplify and label genetic material. The adult and fry samples were genotyped at 354 genetic markers, including a sex marker (Kaylor et al., 2023).

Parentage assignments were performed using Close Kin Mark Recapture (CKMR Sim software) (Bravington et al., 2016, Kaylor et al., Chapter 4). To assess the false positive and false negative rates, Monte Carlo methods were used to estimate the likelihood between each adult and fry sample (Chapter 4). A conservative log-likelihood ratio (LLR) threshold was set based off distribution of MFJD parentage assignments relative to negative control (adults originating outside of the John Day River basin) assignments. We eliminated fry-adult pairings that were below this threshold (Chapter 4).

Dispersal was only evaluated using fry paired to female adults, because when using two-parent assignments, it was determined that the male Chinook Salmon carcasses were often recovered several kilometers away from females they spawned with (JT Lemanski and E Collins, unpublished data using Parentage-Based Tagging). This is likely due to the redd guarding behavior of the females, versus males that will continue to search for additional mating partners (Murdock et al 2009). Therefore, we were more confident that female carcass locations were representative of the fry origin locations. For a more detailed description of the genetic sample and data analysis, see Chapter 4.

DATA ANALYSIS

All data analysis and manipulation was completed using R Statistical Software (v4.2.3; R Core Team, 2023).

Temperature comparison and analysis

In Lindsay et al. (1985) thermographs were presented during the years emergence was sampled in order to track the thermal regime of different basins (to explain differences in emergence timing between basins). We repeated that graph, just for the Middle Fork John Day, in order to detect possible shifts in thermal units during the Chinook Salmon egg incubation period since the late 1970s and 1980s. Water temperature data was provided by the NFJDWC, from an ODFW logger placed in the MFJDR near the confluence with Vincent Creek (similar location as Lindsay et al., 1985). To calculate cumulative thermal units, the mean daily temperature (°F) was calculated from temperature data taken hourly. Then 32 was subtracted because temperatures below 32°F do not contribute to thermal units (when done in °C, this step does not need to occur). Finally, the daily means were summed from September 17th to May 26th for both 2021 and 2022, as this is the peak spawning date to the end of emergence. Data shown in Lindsay et al., 1985 is ambiguous and the integrity of the equipment used is unknown. Because of these uncertainties we also calculated cumulative thermal units from 2012 to 2022, over the egg incubation period by year (using the method described above), to identify changes over time. The logger placed near the Vincent Creek confluence had one of the longest records that included fall, winter, and spring temperatures of the MFJDR. Until recently, measuring water temperature only in the summer was the most common protocol (J. Bailey, NFJDWC, personal communication).

Fry emergence timing

To compare recent and historic emergence timing we used the same criteria used from historic data (Lindsay et al. 1985). The beginning of emergence was the first day Chinook Salmon fry were caught in the MFJDR basin. The last day of emergence was when less than 5% of the fry captured had a fork length greater than 37 mm. We documented these dates and compared them to the dates stated in Lindsay et al. (1985) listed for the MFJDR basin. There were some limitations in the historic data because they did not always record a beginning of emergence date for the MFJDR basin.

Habitat use and flow variability

When fry were captured, the type of habitat they were found in (e.g. margin, alcove, side channel) was recorded and we designated if it was mainstem or floodplain. We used the percentage of fry caught that year found in each habitat to compare the different years. The flow data was retrieved from the USGS Camp Creek gauge near Camp Creek on the MFJD, using the dataRetrieval R package (v4.2.3; R Core Team, 2023). For each sampling year, we plotted the volume (cfs, cubic feet per second) by date to compare timing and variability of flow conditions.

Fry dispersal

For fry that were matched to an adult, their origin was determined by first using ArcMap to snap each adult carcass location to the nearest river meter in the MFJDR. When a fry was matched to an adult carcass, the carcass location was assigned as that fry's origin. To calculate how far each genetically matched fry traveled, the fry capture location was snapped to the nearest point in the mainstem MFJDR and that stream meter was assigned. The difference between these meter assignments equals the longitudinal distance that fry dispersed.

For all of the fry captured in 2021 (not just fry paired to adults), each assigned a habitat group: mainstem/margins, floodplain adjacent (to mainstem), and floodplain far (in relation to the mainstem). These were assigned by first using ArcMap to generate the distance that each fry was captured to the center flowline of the mainstem MFJDR. Fry within 0-5m from the centerline were considered 'mainstem/margin', fry within 5-15m were considered 'floodplain adjacent', and fry greater than 15m from the centerline were considered 'floodplain far'. Each location assignment was confirmed using aerial imagery.

RESULTS

Temperature

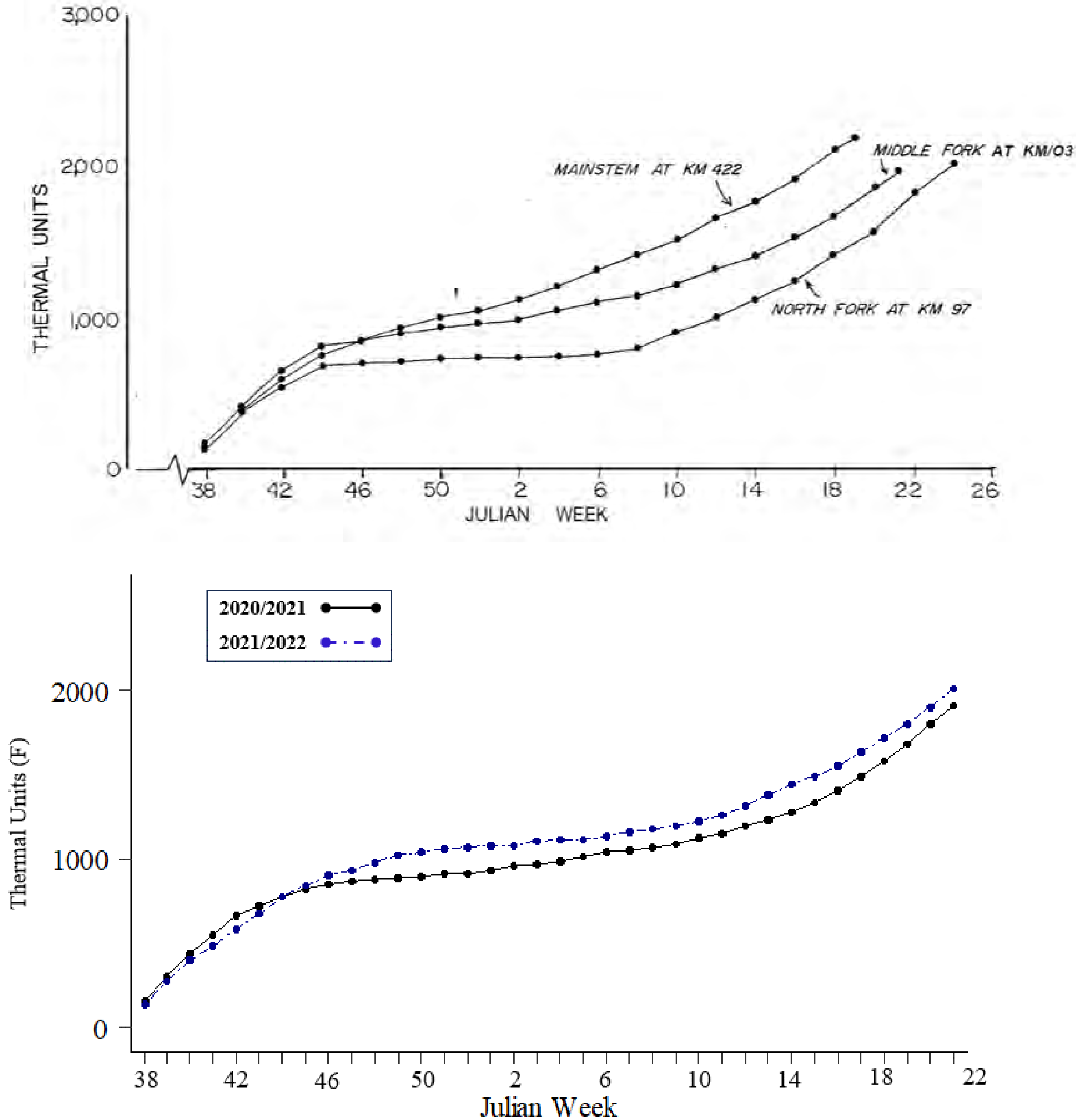

Figure 2. Thermographs from mid-Sept through late May (Julian week 38-21) showing cumulative thermal units by Julian week from Lindsay et al., 1985 (top graph) shows temperature from one year but from multiple places around the John Day basin. The recreated graph using recent data (lower) only shows the temperature data from the Middle Fork but for two different incubation years. This cursory analysis indicates the thermal units over the egg incubation period has not shifted since the 1980s.

We analyzed temperature during Chinook Salmon salmon egg incubation period two different ways 1) recreated the temperature analysis from the Lindsay et al. (1985) report (Figure 3) and 2) calculated the cumulative sum of mean daily temperature from 2012 to 2022 (Figure 4). The first comparative analysis does not give us enough evidence to say temperature has or has not shifted during egg incubation period since the 1980s. For the second trend analysis, we found a moderate negative correlation between cumulative thermal units and year (r = -0.46, p = 0.18) providing preliminary evidence for a decreasing temperature trend during temperature development from 2012 to 2021. This was the longest available dataset from the MFJDR that captured year-round data.

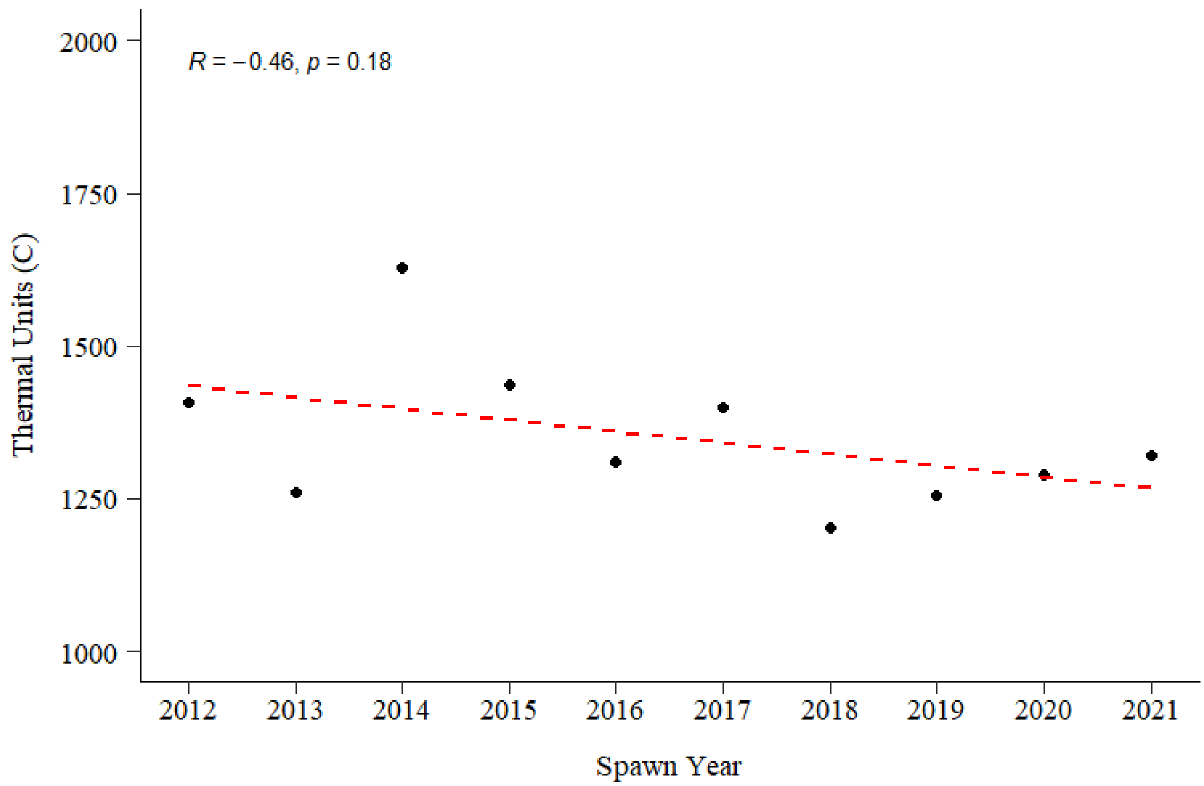

Figure 3.

Cumulative thermal units throughout the spring Chinook Salmon egg incubation period for each spawn year since 2012. For each year, mean daily temperature is added from Sept 17th to May 26th. A moderate decreasing trend was found for cumulative thermal units by year.

FRY EMERGENCE TIMING

Lindsay et al. (1985) reports that emergence occurred in the MFJD from mid-March to late-May (Table 1, Lindsay et al. 1985). The emergence window we identified in 2021 and 2022 is approximately the same (Table 1 and Figure 4). Therefore, we conclude that fry emergence timing has not shifted from historic sampling, beyond normal variation among years. Our confidence about the end of emergence is higher because historic emergence sampling documented three dates, compared to just one for the start of emergence. The historic sampling period determined an end of emergence for three years ranging from May 7th to May 25th, where we documented the end of emergence as May 18th and 19th for 2022 and 2021, respectively. The end of emergence was defined as when ≤5% of captured fry are smaller than 37mm in fork length. This protocol was used in 2021 and 2022, and sampling did not continue after this criterion was met. In 2019 this protocol was not followed, and fish were measured in total length, therefore an end-of-emergence date was not determined. Figure 4 shows peak emergence occurring from mid-April to mid-May, with downstream areas peaking later than upstream areas. This occurs because temperatures stay relatively moderate through the winter due to springs that feed the headwaters of the Middle Fork (Kaylor et al., 2020). Unfortunately, peak emergence was not determined by historical sampling as the number of fry captured was not recorded. The location of historic sampling was also not recorded in Lindsay et al. (1985) except to say that temperature was recorded around Vincent Creek confluence. Since this is a high-redd-density area, we are assuming at least some historic emergence sampling occurred in this area.

Table 1. Summary showing a comparison of results from historic and recent fry emergence sampling and redd surveys. Since the historic sampling results reporting was limited, we could only conclude that the end of emergence has not changed since that time.

Year	First sample day	Start of emergence	End of Emergence	Number of fry catured	#Redds[1]
Historic					
1979	--	--	May 7th	--	281
1980	--	March 19th	May 15th	--	235
1981	--	--	May 25th	--	155
Recent					
2019	March 5th	April 4th[2]	--	565	75
2021	January 27th	March 10th	May 19th	1672	162
2022	February 16th	March 30th	May 18th	511	28

1. This column refers to the number of spring Chinook Salmon redds counted the previous Fall to fry sampling season. Historic number of redds were reported in Lindsay et al. (1985). Redd counts from 2018 are from Bare et al. (2021). Redd counts from 2020 and 2021 redd counts are from personal communication with Alex Woolen (ODFW).

2. Methods for capturing fry in 2019 was exploratory. Even though this was the first date fry were captured, we cannot say with confidence this was the start of emergence for this year.

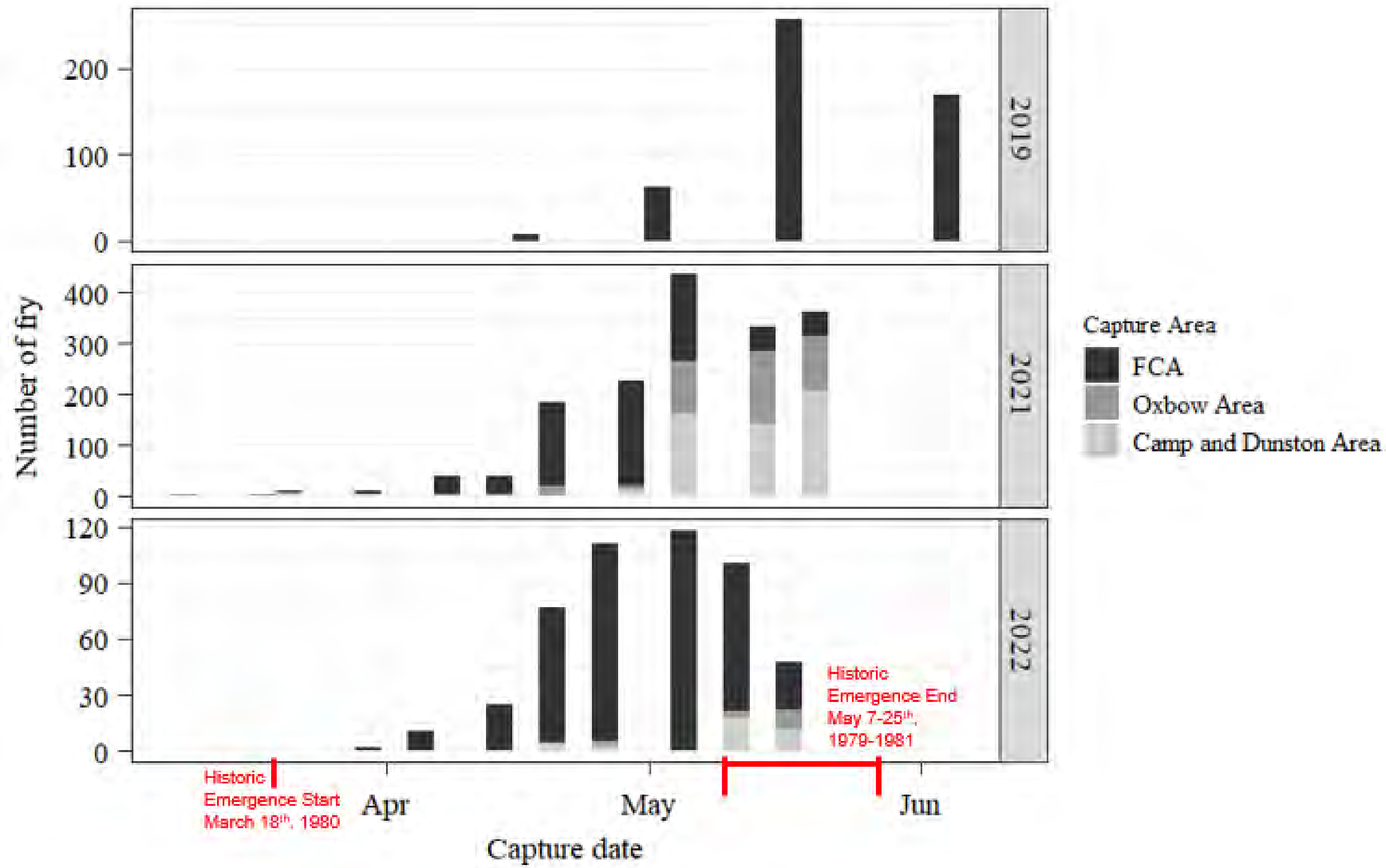

Figure 4. The number of fry caught by date and capture area, displayed by sampling year. The success of capturing fry in 2019 was sporadic and sampling continued past the established criteria for end of emergence. In 2021 and 2022, peak emergence occurs from mid-April to mid-May.

HABITAT USE AND ANNUAL FLOW VARIABILITY

Fry were distributed laterally across the river landscape occupying areas in the margins of the mainstem and floodplain. The most common place to find fry were in the margins of the main channel but we also found fry inhabiting features such as alcoves, side channels, and irrigation ditches (Table 2). Fry were generally not found in open or swift water. Their main source of cover was flooded grass on margins and floodplain habitat. In most cases, that was the only option for cover.

The years we sampled had different flow regimes due to the timing of snow melt and precipitation over the spring season (Figure 5). This variety also determined the type of habitat that was available for fry (Table 2), in relation to emergence timing. For example, in 2019 and 2021, there were high flows early enough in the year that fry were able to more heavily utilize floodplain habitat. In 2022, there was no inundation of the floodplain until much later in the year, once emergence was almost complete. That year, the majority of fry were captured in the margins of the mainstem.

Table 2. For each year of fry emergence sampling, this shows what type of habitat the fry were found in. All habitat except margins was considered floodplain habitat.

		%fry by habitat				
Year	#fry	Margins	Alcove	Side Channel	Irrigation Ditch	Isolated Pool
2019	565	57%	17%	15%	6%	5%
2021	1672	46%	42%	8%	3%	--
2022	511	88%	8%	4%	--	--

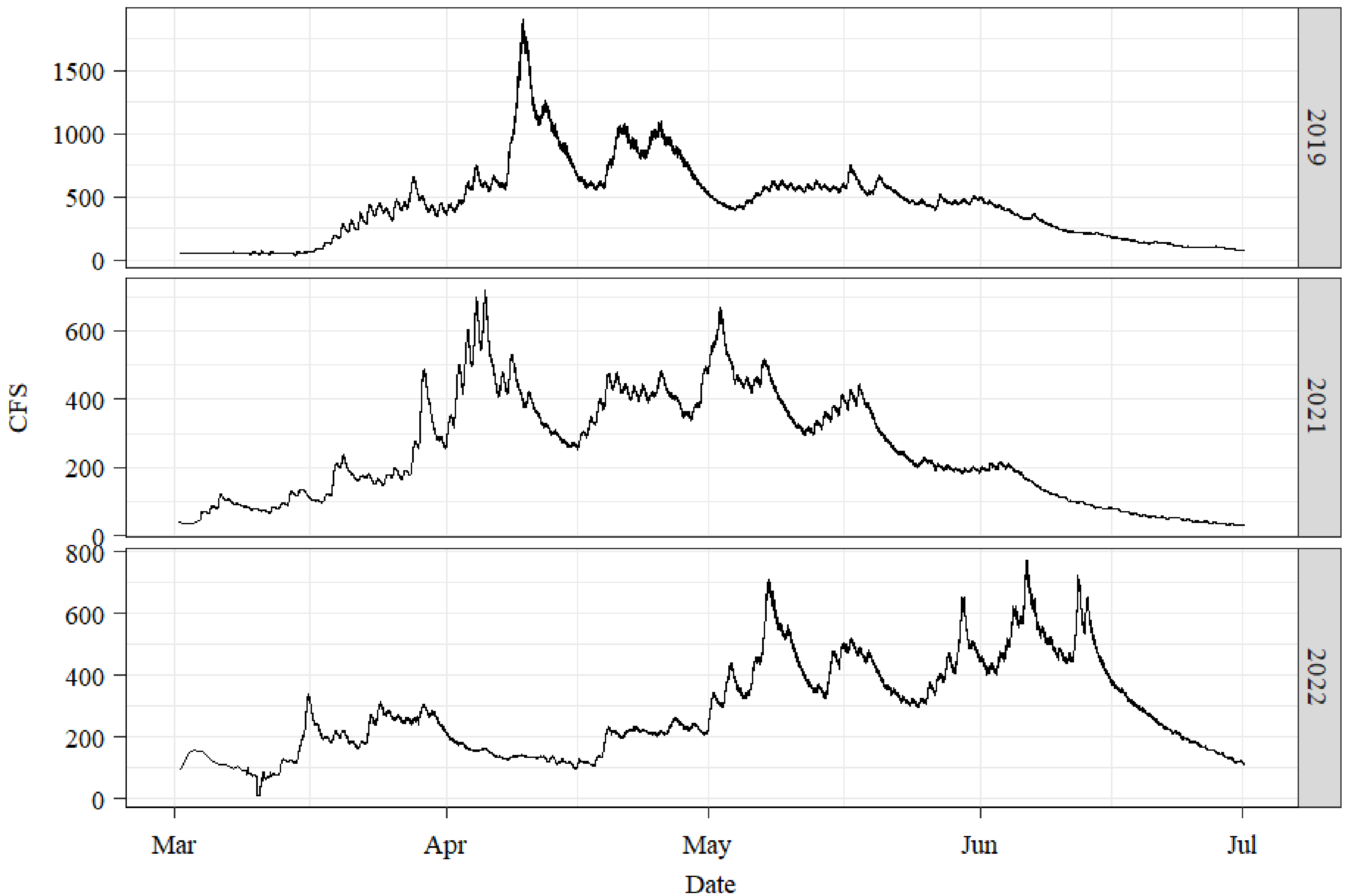

Figure 5. Flow data recorded at the USGS gauge station on the MFJDR above Camp Creek near the town of Galena, OR. In 2019 and 2021, flows in April inundated the floodplain during the peak fry emergence sampling period.

GENETIC ANALYSIS AND PAIRING

Genetic material was sampled from 142 adult carcasses and out of those 113 contained material that could be successfully genotyped. Sixty-seven of those were females (we only used females to determine origin as described in the Methods section) and we matched 397 fry to fifty-seven females. Table 3 shows the number of females versus redds per origin section. They were not equally distributed among sections, and this could have an effect on the dispersal results. The number of carcasses sampled per redd by section was fairly evenly distributed across the basin, but the number of females genotyped and matched to fry was not.

Table 3. Summary of genetic results by river section. The number of carcasses sampled per redds and section was fairly evenly distributed but the number of females genotyped was not.

Section (US to DS)	#Redds	#Adults Sampled:	#Females Genotyped	#Females matched to fry	# of fry matched to a female	Ratio Matched Fry to Redds
Upstream of FCA	15	15	6	6	36	2.4
FCA	44	42	18	17	141	3.2
Between FCA and Oxbow	21	18	15	9	14	0.7
Oxbow	32	25	6	5	66	2.1
Between Oxbow and Dunston	15	3	1	0	NA	--
Dunston	30	38	21	20	140	4.7
Camp	3	0	NA	NA	NA	--
Below Camp	2	0	NA	NA	NA	--
Totals:	**162**	**141**	**67**	**57**	**397**	--

FRY DISPERSAL

Longitudinal dispersal

For each fry that was captured and matched to an adult, we documented longitudinal dispersal from origin to capture location. We documented dispersal in the downstream direction only with the majority of fry dispersing less than a kilometer or not at all from where they were born (Figure 6). The median dispersal distance calculated as 0.8 km (95% = 0.052 – 12.59). There were also numerous fry that dispersed much further, including some that traveled approximately 20 kilometers (Figure 6). A common dispersal distance was 5 km, we attribute this to our sampling design where several sites were approximately 5 km apart.

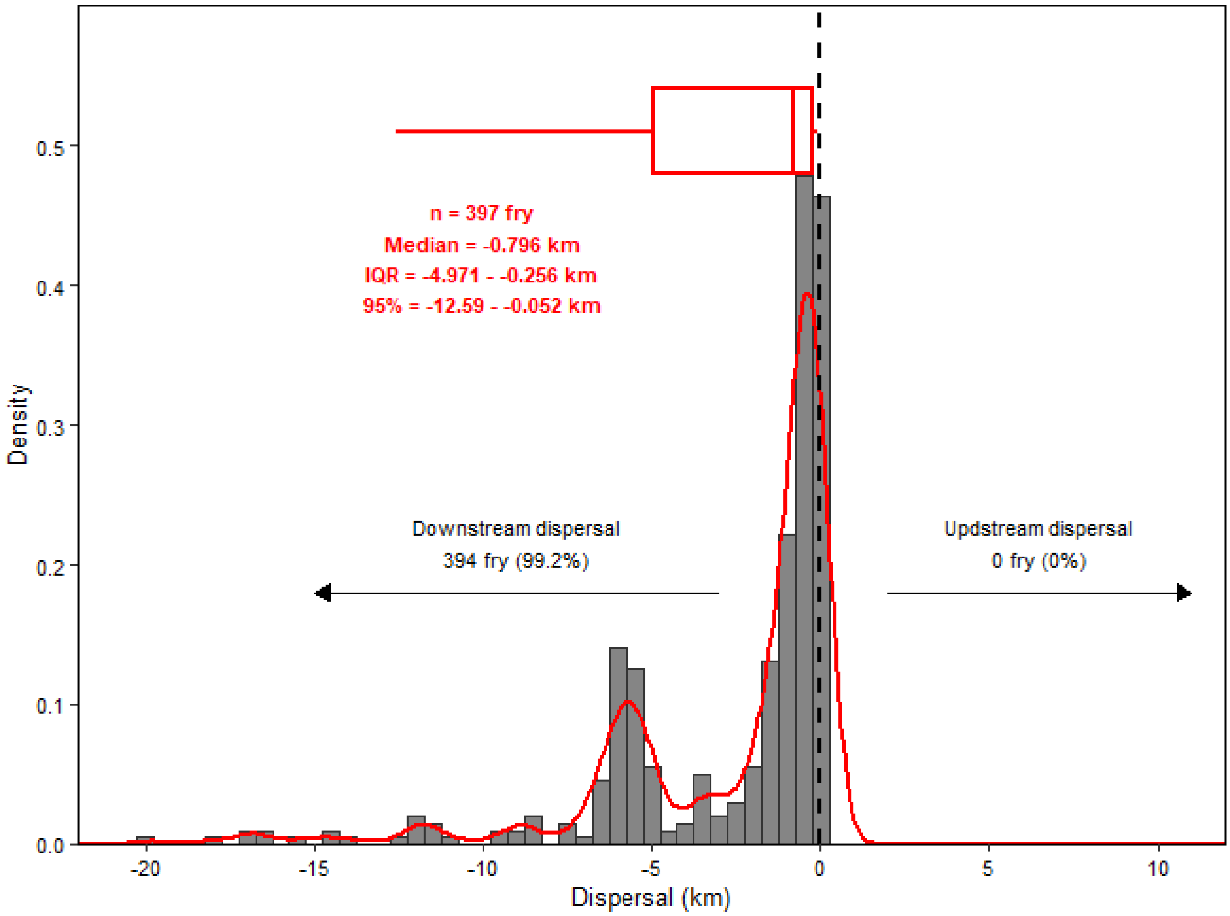

Figure 6. Overall dispersal of fry from their origin location to where they were captured. No fry dispersed in the upstream direction and the longest dispersal distance was 20 km. But the majority of fry moved less than 1 km. Figure by Matt Kaylor.

We also looked at longitudinal dispersal within different river sections in the MFJD (described in the Methods) by dividing fry into capture and origin area categories (Figure 7). This data, like the data above, is only fry captured in 2021 that we were able to genetically match to an adult. Each capture area shows a slightly different pattern. For the MFFCA, drifters (fry with a different origin area from capture) and residents (fry where origin and capture area are the same) are both present, but we also see drifters from these areas down in the OCA. The OCA draws in drifters but also retains residents, as we don't see many fry from the OCA in downstream sections. The DCA only had resident fry, but several fry from the DCA were also found near Camp Creek (rkm 80). Camp Creek area attracted drifters from the OCA and DCA, with some of the largest dispersals recorded coming from the OCA to Camp Creek. The figure does not show any residents from Camp Creek because no carcasses were sampled in that area.

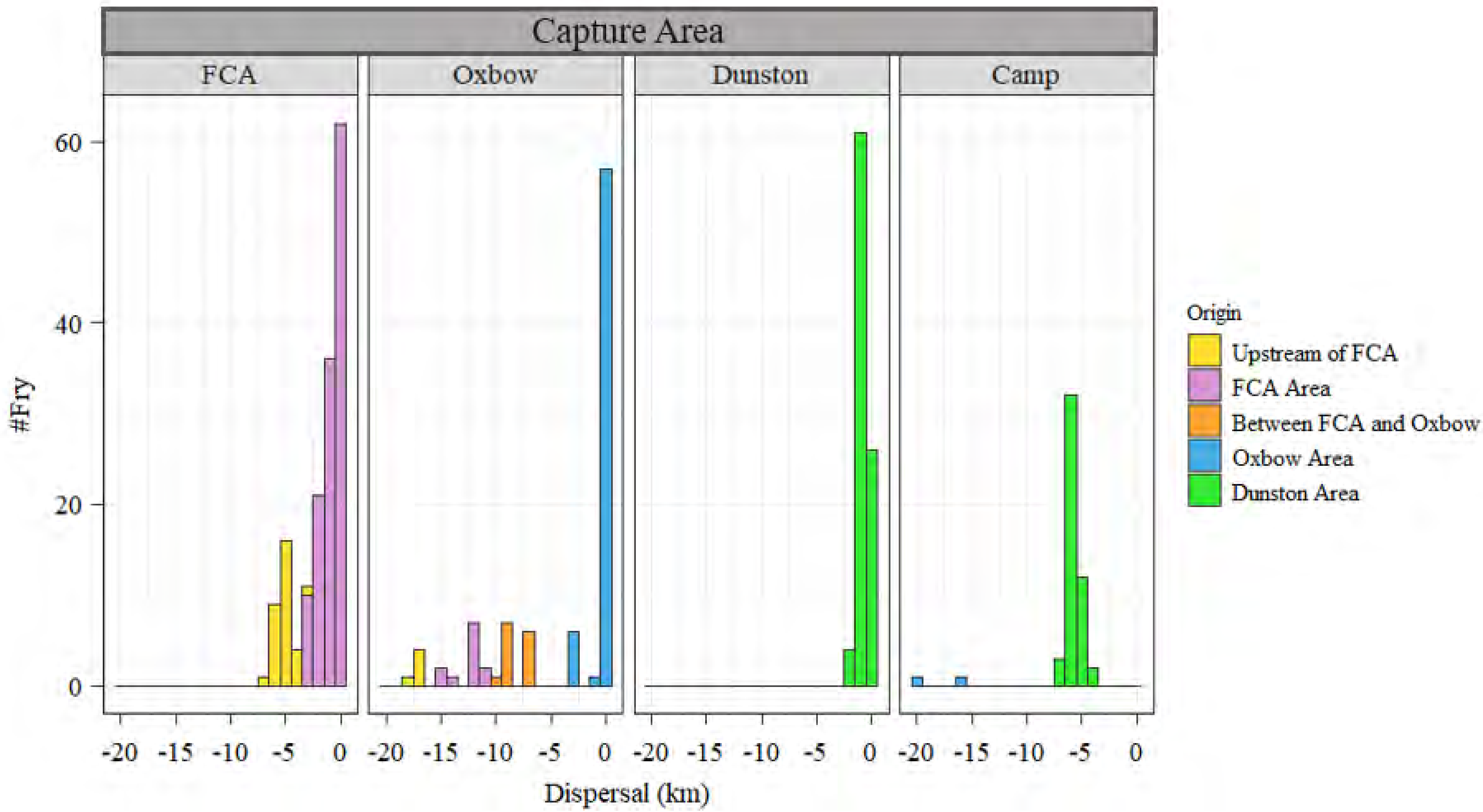

Figure 7. Dispersal distance by capture area and origin. The OCA attracted fry from great distances upstream and some of the longest dispersals were fry from the OCA to the mouth of Camp Creek and from Upstream of MFFCA to the OCA.

Lateral dispersal

Fry not only dispersed longitudinally downstream, but also laterally from deposited redds in the mainstem to wetted areas in floodplain habitats (Figure 8). We found that 30 percent (n = 467) of fry remained in the margins of the mainstem channel (MS: 0-5m to river center), 44 percent (n = 676) were found in floodplain adjacent (FPA: 5-15m to river center), and 26 percent (n = 395) were found in floodplain far habitat (FPF: >15m to river center). A small minority of fry were found greater than 100m from the river center. Figure 9 shows drone footage of two sections of the MFJDR, including examples of the types of areas where fry were found by the dispersal groups they were assigned to.

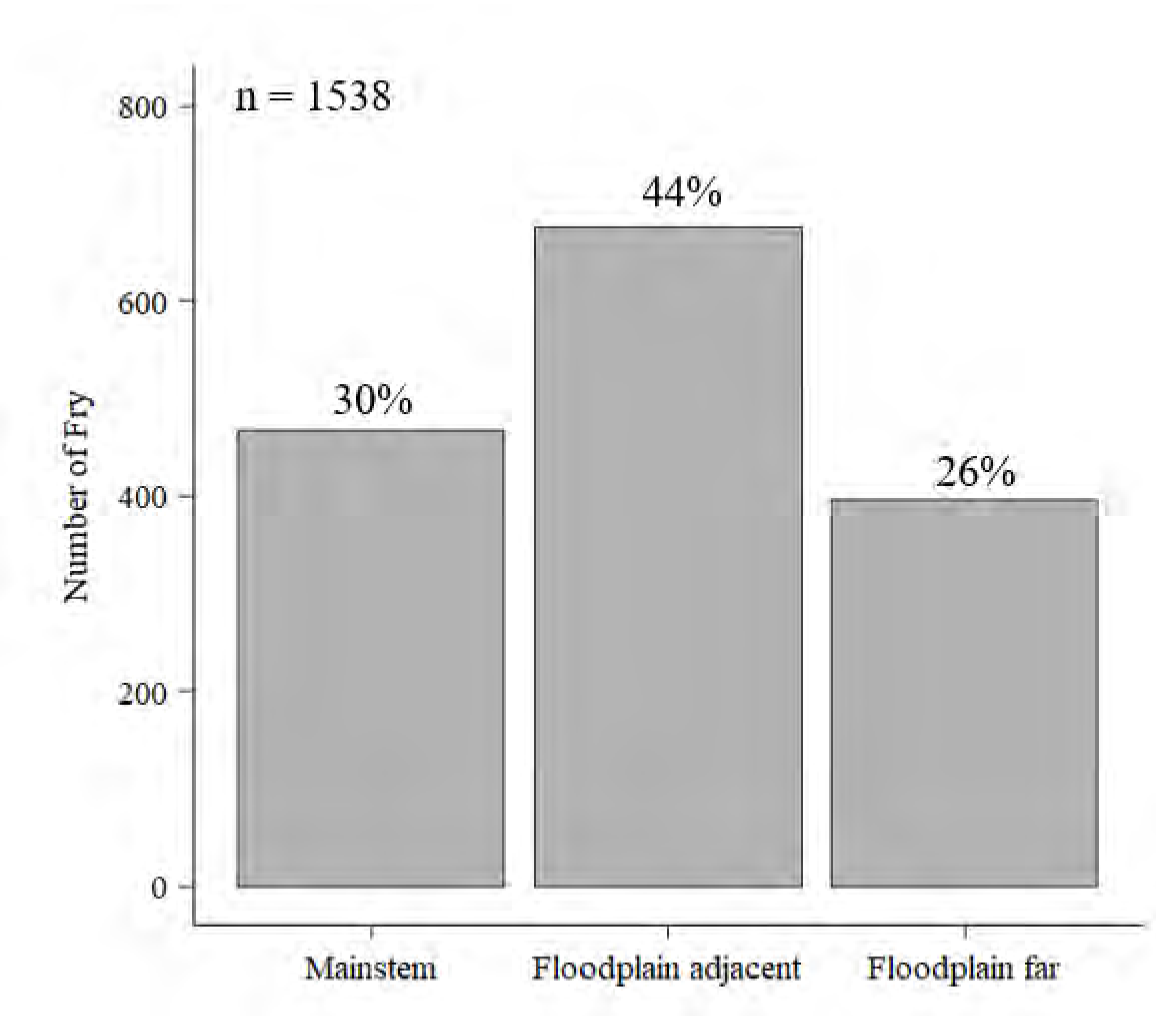

Figure 8. Number of fry by lateral dispersal location for all sampling locations. Most fry were found in floodplain habitat adjacent and well connected to the mainstem, followed by mainstem, then floodplain far.

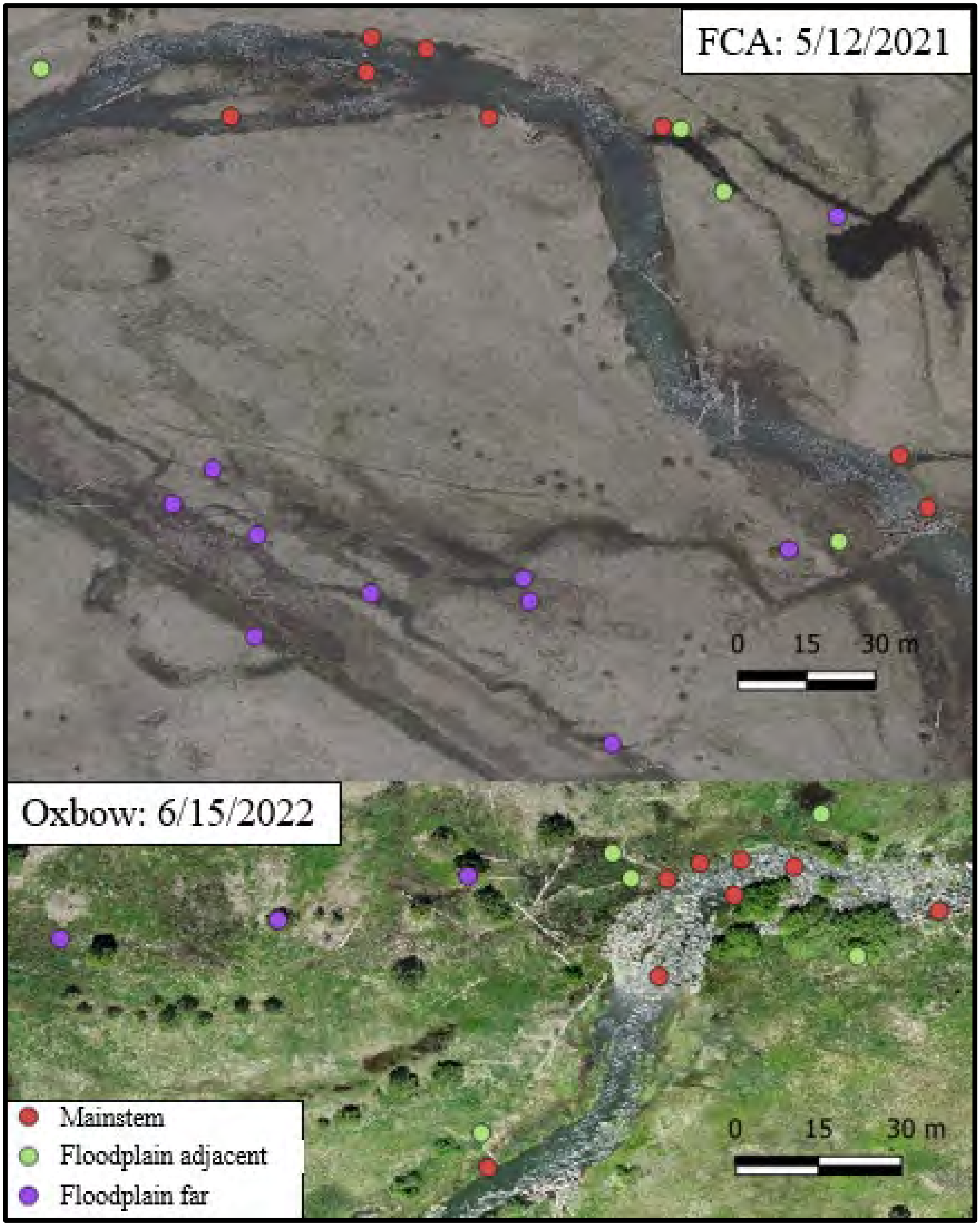

Figure 9. Drone imagery and fry capture waypoints from the MFFCA and OCA sections of the MFJDR, showing designations of mainstem, floodplain adjacent, and floodplain for. Both images are taken while the floodplain was inundated from high flows, the date of each flight is labeled on the photo. Drone imagery provided by CTWSRO. The OCA photo was taken in 2022, but is indicative of floodplain inundation when fry were sampled in 2021.

Lateral dispersal by capture area

Fry distributed themselves across the landscape where they emerged (residents) and when they dispersed (drifters), utilizing available wetted habitat (mainstem, floodplain adjacent, and floodplain far described above). We found that in different restoration impacted areas (defined in the Methods section), fry utilized available habitat differently depending on if they were residents or drifters (Figure 10). In the MFFCA, residents were more likely to utilize floodplain habitat that was far away from the mainstem, but drifters onto the MFFCA utilized the three habitat types in approximately equal proportions. Conversely, in the OCA, resident fish were more likely to utilize mainstem habitat, but drifters utilized all habitat types. In the DCA and Camp Creek area, drifters were more likely to utilize floodplain habitat versus mainstem habitat. There were no fry found in the 'floodplain far' category in the DCA and Camp Creek areas. We are currently quantifying the available habitat in each section using drone imagery at various flow conditions to better understand if habitat utilization is a function of available habitat.

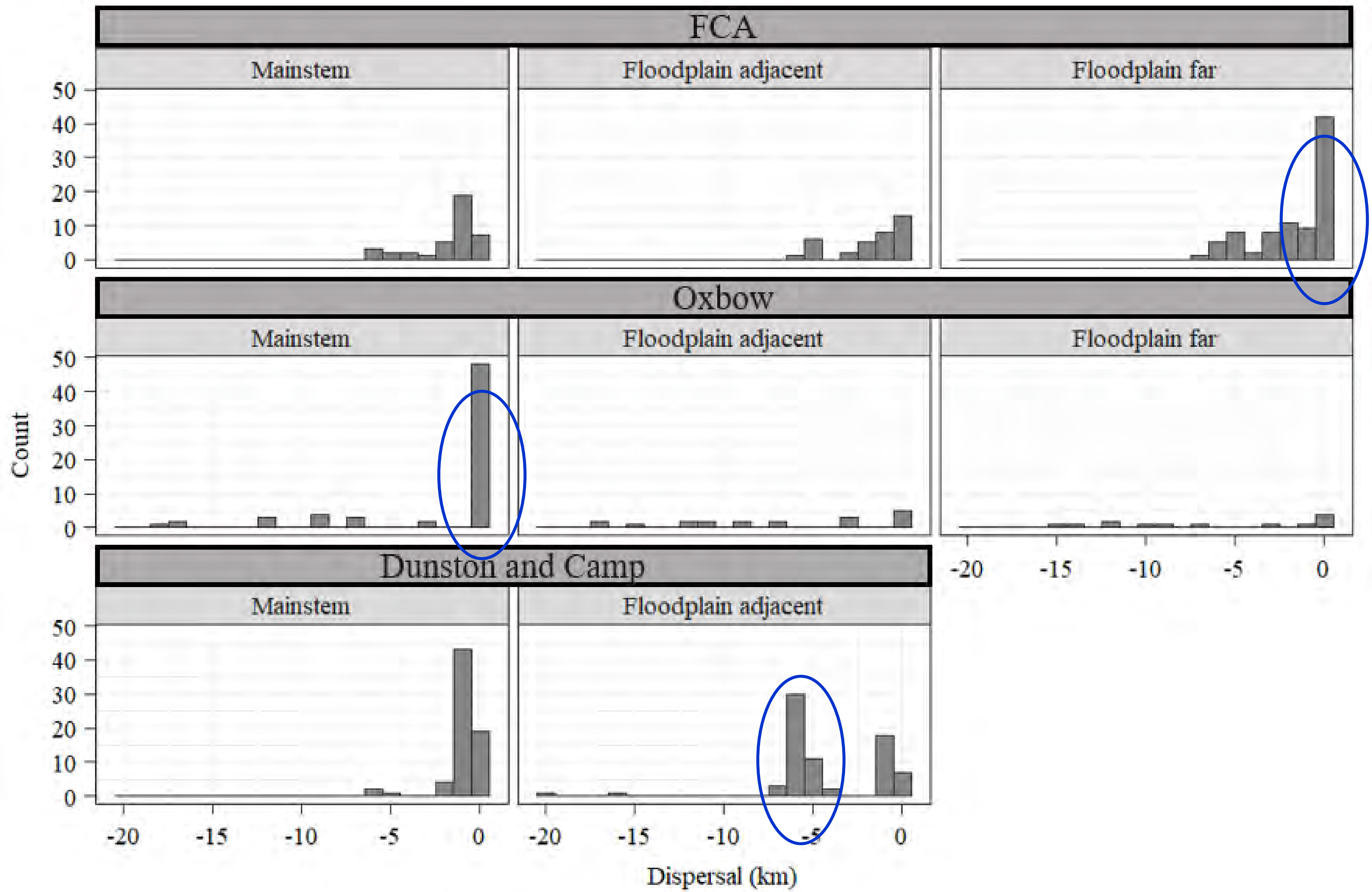

Figure 10. Number of fry by dispersal distance grouped into their capture area. In the MFFCA, it is the resident fish that utilized the 'floodplain far' habitat. Where in the OCA, resident fish stayed in the mainstem. In the Camp Creek and DCA, drifters utilized the floodplain adjacent habitat. On these graphs, residents have a small dispersal distance (<2 km) and drifters dispersed further (>2km). The blue circles highlight the trends discussed in the results text.

DISCUSSION

Temperature and Emergence timing

Emergence from redds should ideally align with favorable environmental conditions (e.g. floodplain inundation, moderate flow, food resources) in order to maximize growth and survival for salmonid fry (Murray et al. 1990, Web and McLay 1996). Emergence timing is determined by when adult salmon spawn and the thermal conditions experienced by the eggs during incubation (Kaylor et al. 2020). Spring Chinook Salmon eggs incubate over the fall and winter, and emergence occurs early to late spring. Climate-driven temperature variation during these seasons is studied comparatively less versus the summer, when thermal maximum occurs, yet variation occurring throughout the incubation period could deleteriously disrupt emergence phenology or other life-cycle events for advanced life stages. In one year-round temperature analysis (Isaak et al. 2012), warming trends were documented (reported in degrees Celsius per decade) for the fall and winter, but cooling trends in the spring in streams and rivers across the Pacific Northwest from 1980 to 2009.

In the midst of a changing climate, we expected shifts in the thermal regime during egg incubation, and therefore a shift in emergence timing compared to when it was last studied in the late 1970s and early 1980s in the MFJDR (Lindsay et al. 1985). While we did not see a shift in the emergence timing window (Table 1 and Figure 4), we did detect a moderate declining trend in cumulative thermal units during the egg incubation period (Figure 3) from 2012 to 2022. If this trend continues or intensifies, this could shift the emergence timing window to later in the year. Although, according to Sparks et al. (2019) calculating accurate emergence timing requires more complex temperature modeling that could be applied in future studies.

We could only compare an emergence window, not peak, due to the type of historic data reported. We also do not know exactly where historic fry emergence occurred and emergence timing varies by several weeks depending on location (Kaylor et al., 2020). It was recorded that the temperature data reported in Lindsay et al., 1985 was collected from river km 103 near Vincent Creek. Coincidentally, one of the longest records of year-round temperature was from an ODFW logger placed near the mouth of Vincent Creek. Until recently, it was not the protocol for most agencies in the basin to leave loggers in year-round, as the focus has been on the most concerning temperature increases during the summer, when conditions can become inhospitable for salmonids. However, monitoring water temperatures throughout the year should be prioritized, as all seasons are important to salmonid growth and development (Armstrong et al., 2021).

The magnitude of climate change is predicted to increase over the coming decades (Isaak et al., 2012), so regular emergence sampling paired with long-term fall, winter, and spring temperature collection and a more robust analysis, should continue in order to identify and document patterns in emergence timing that could negatively affect survival at the fry life stage.

Habitat Use and Floodplain Inundation

Floodplain habitat provides a mosaic of thermal and trophic conditions that can be beneficial for fry survival and growth (Baldeck et al., 2016, Sommer et al., 2001). Floodplain inundation is controlled by stream flow, and therefore, is not always available to juvenile salmonids (Baldeck et al., 2016). Restoration practices that encourage floodplain reconnection (e.g. large woody debris, beaver dam analogs, berm and diversion removal, etc.), has only just begun to repair the channelization and incision caused by historic land use practices (Pollock at al. 2007). In the MFJDR, these practices have been effectively implemented in the OCA and MFFCA sections, successfully inundating large sections of floodplain during spring flows.

During the winter and early spring, rivers in eastern Oregon are near baseflows, with snowbanks and ice dominating the margins of the river. The majority of floodplain habitats are unavailable for fishes until the first large pulse of snowmelt or precipitation increases flows. We found in years where the floodplain was inundated before or during peak emergence (2019 and 2021, Table 1), a higher proportion of fry were found in floodplain features like alcoves and side channels. In 2022, flows high enough to inundate the floodplain did not occur until mid-May, past peak emergence throughout the spawning distribution (Kaylor et al., 2022). Throughout emergence in 2022, floodplain features (e.g. alcoves, side channels, etc) were not available due to low, consistent flows throughout early emergence (Figure 5), severely limiting the use of off-channel habitat (Table 2).

Studies have documented that low flow years are strongly associated with lower survival in a variety of life-stages for Chinook Salmon (Warkentin et al., 2022, Michel et al., 2015), partially due to density dependent factors. It has been well documented that density dependance is a primary mechanism regulating growth in salmonids (Spalding et al., 1995; Jenkins et al., 1999; Keeley 2000; Imre et al., 2005), this mechanism is of vital importance considering size has often been correlated with survival rates at a variety of life stages (Quinn and Peterson 1996; Biro et al., 2004; Ebersole et al., 2006). Considering the benefits of decreased density via increased available habitat during the fry life stage, restoration practitioners should continue to prioritize floodplain connection in the MFJDR, especially near areas of high redd density.

Dispersal and Genotype Pairing

Fry dispersal from redds is poorly understood, as this fragile life stage does not allow for the utilization of traditional techniques, like PIT tags, which allows researchers to track the same individual over time to detect movement throughout development. To overcome this challenge, we used genetic single-parentage assignments to assign a point of origin for fry that matched with samples collected from spawned adult carcasses, to look at dispersal from redds to point of capture. When adult carcasses were sampled, they were sampled opportunistically across the spawning range and in general, one carcass was sampled per redd in each section (Table 3, except for the section between the OCA and DCA). However, some samples were not able to be genotyped due to deterioration and some samples did not have any offspring matches (possibly indicating an unsuccessful redd). We further eliminated adult male genotyped adults, as their carcass location does not necessarily reflect offspring origin because males do not exhibit the redd guarding behavior as the females do (JT Lemanski and E Collins, unpublished data using Parentage-Based Tagging). This uneven sampling distribution could have implications on the dispersal results of this study.

From the 397 fry that were matched to adult samples, our results show that during the fry life stage, longitudinal dispersal occurs only in the downstream direction and laterally onto floodplain features. For most fry, the extent of movement was limited to less than a kilometer in the downstream direction from the redd (Figure 6). Yet some fry (n = 99, 25%) traveled great distances (5 to 20 km) downstream from their point of origin. To our knowledge, fry dispersal of this magnitude has not been recorded in literature.

Yet several things are still unclear, including if dispersal is passive or active, and the causal mechanisms of downstream dispersal. Our results suggest that restoration practices influence dispersal patterns. For example, the OCA section of the MFJDR, draws in fry from several sections upstream but also fry seem less likely to disperse from the OCA (Figure 7). The OCA underwent extensive active restoration from 2011-2016, which reconstructed the channel and reconnected the floodplain, additionally rejoined Granite Boulder Creek, a crucial cold-water tributary, to the MFJDR, and eliminated the "North Channel" (a relic of dredging) concentrating more flow into the main channel. In the years since the OCA restoration, that section has also been documented as the highest Chinook Salmon redd density in the MFJDR (L. Ciepiela, unpublished data) and an area of high juvenile density (Chapter 4). This culmination of results strongly suggests that the restoration was effective for several life stages of spring Chinook Salmon.

We also documented lateral migration of fry from redds deposited in the mainstem, to floodplain habitats. For this analysis, we were able to use the entire data set of captured fry, not just fry that were matched to adults. Most fry (74%, n = 1143) were captured in the margins or floodplain features adjacent to the mainstem (Figure 8). While only 26% (n = 395) were found in floodplain features greater than 15 m from the mainstem (Figure 8). Some of those were found as far as 50-100m away from the main channel. We hypothesize this is caused either by fry getting swept to places further in the floodplain by high flows or the fry are seeking out areas of lower fish. In these areas far away and sometimes disconnected from the main channel, the possibility of stranding in outlying floodplain areas is greater than habitat close to the main channel. The benefits and drawbacks of creating floodplain features like this should be considered when planning and implementing restoration actions.

Sections of the MFJDR exhibited different lateral dispersal patterns, with respect to where residents versus drifters distributed themselves across the landscape (Figure 10). In the MFFCA, residents were more likely to use the floodplain habitat greater than 15m from the main channel, compared to the OCA where residents were more likely to stay in the main channel. We hypothesize this implies there is more quality fry habitat (slow, deep, covered) in the main channel, or the restoration techniques utilized minimized floodplain habitat far from the main channel. It is also possible the uneven distribution of genotyped adults yielded these results. A future analysis will include using drone imagery to measure available flooded habitat in restoration sections for a variety of different flow conditions.

This new understanding of dispersal generates implications for informing restoration practices and implementation. Redds deposited in the furthest extent upstream of spawning areas have the most possible access to resources, as fry can keep going downstream for territory, where fry born at the downstream extent of spawning distribution have very little opportunities for accessing desirable habitat, until they are large enough (in tandem with decreasing flows) for upstream dispersal. Restoration targeting the fry life stage should focus on high density spawning areas in order to have the greatest benefit, as most fry do not disperse greater than 1 km from redds. Activated floodplain features should be planned to maintain a connection with the mainstem in order to avoid stranding. Although during high flow events, even in a pristine riverscape, stranding most likely occurs to some extent. Climate change predictions indicate more extreme conditions could occur, like high flow events, increasing the need for more research on what is driving fry dispersal and the types of restoration conducive for survival.

Caveats and future research goals

There are several caveats for this research due to the unknowns of capturing and handling fry, which has limited established methods in literature. The first year of this study was a pilot year, and later years determined that dip nets were the best way to capture fry and it was important to anesthetize fish in order to accurately measure fork length. In hindsight, we determined that sampling should occur in areas of low redd density as well as high (implement random sampling), as it is possible we would capture further dispersal distances and different lateral dispersal patterns. We only took waypoints where fry were captured, but recording areas fry were not found, could also be informative for restoration implementation and general knowledge. As mentioned throughout the text, due to the methods of genotyping and origin assignment, the number of fry matched to adults was not evenly distributed across the landscape. Future studies should try to achieve a more balanced design and include random sampling for fry capture not just in areas of high redd density.

This study should be repeated periodically in order to detect emergence timing shifts paired with an analysis of water temperature during the egg incubation period. Kaylor et al., 2022 states that emergence timing is on a gradient, occurring earlier upstream versus downstream in the MFJDR. For this study, our results were not robust enough to detect that pattern, but future studies need to clarify sample locations in order to correctly determine if emergence timing is shifting due to climate change. We only sampled during the day, and it is possible that fry occupy different areas during that time. Developing a method of marking fry to recapture them, could answer questions like growth and survival in different habitat types.

LESSONS LEARNED

- Have a very clear research question in mind before collecting data
- Choose sites randomly to prevent sampling biases that can affect results. In our effort to collect data on many fish, we forget to collect data where there are few fish
- Determine a viable method of sampling when planning new projects
- Staff turnover makes data analysis and management very difficult; have a process lined out where there is communication between outgoing and incoming project leaders
- Think about biased sampling when designing. This will be important for determining habitat preferences
- Consider all life stages of salmonids when planning and executing restoration projects
- When considering the fry life stage for restoration projects, only consider downstream of spawning. Try to avoid creating off-channel habitat that becomes disconnected as water recedes where fry can get stranded
- Redd placement can potentially have a huge impact on parr survival, because redd placement can limit access to resources

FUTURE ANALYSIS

- Determine what type of habitat use will increase survival at the fry life stage

- When looking at temperature trends and deploying temperature loggers, include winter temperature metrics and data collection. Water temperature changes overtime could affect development of eggs and emergence timing
- Future work should also sample tributaries; we find parr in tributaries, but we do not know when they start entering tributaries. It is possible downstream migration is the only option in the mainstem Middle Fork until they are larger
- Quantify available flooded habitat for fry to occupy at different flow conditions and try to relate back to survival by year

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CHAPTER 4: Juvenile Chinook Salmon Dispersal Patterns Across the Middle Fork John Day Watershed

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ABSTRACT

We assessed juvenile Chinook Salmon dispersal patterns across the Middle Fork John Day River (MFJDR), in which parr sampled from rearing habitats in summer 2021 were traced back to potential adults sampled from spawning locations in 2020 using genetic parentage assignments. We estimated that 68% of all parr dispersed downstream (*n* = 1,326) and that 25% dispersed at least 3.7 km downstream, whereas 25% were estimated to have dispersed more than 0.9 km upstream. Dispersal patterns varied across the watershed, with parr originating farther upstream generally exhibiting more downstream bias, greater variability in dispersal, and farther dispersal distances compared to parr originating lower in the watershed. Parr originating in areas with higher maximum summer stream temperatures generally dispersed to cooler sections of the mainstem or tributaries, suggesting temperature was a primary driver of dispersal in 2021. Overall, these findings suggest farther dispersal at early life-stages than prior published estimates, and that longitudinal temperature patterns and the configuration of accessible cold-water tributaries along river profiles contribute to variability in dispersal patterns within and among watersheds.

INTRODUCTION

Background

Riverine fishes experience a heterogenous 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 subsequent survival (Einum and Nislow 2005; Brunsdon 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 biased downstream (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 only a very few studies have evaluated dispersal in naturally spawning populations (see Anderson et al. 2013). Wild populations exposed to greater variability in inter- and intra-specific competitor densities, 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.

Goals and objectives

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.

Our specific ***objectives*** included:

1) Evaluate population-level dispersal patterns.
2) Assess how dispersal patterns vary throughout the watershed as a function of origin (i.e., redd location).
3) Evaluate potential mechanisms driving dispersal patterns.

And we tested the following ***predictions***:

1) Population-level dispersal patterns will reflect those observed in other studies, with downstream bias and the majority of individuals remaining with 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.
3) The size of fish in summer months will be positively associated with dispersal distance, which could reflect greater swimming capacity of larger individuals or greater growth of individuals after they dispersed to more favorable habitats (e.g., lower competition, greater prey availability).

METHODS

Adult sampling

Adult Chinook Salmon were sampled in September 2020 during the peak of spawning activity (9/16/2020 – 9/23/2020). Surveyors (1-2 individuals per survey reach) 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 using standard spawning ground survey techniques (Bare et al. 2021). Standard surveys were supplemented with intensive daily surveys conducted by a smaller group of 2-4 surveyors, prioritizing areas of concentrated live adults. During both standard and intensive surveys, surveyors located carcasses and measured length (medial eye to posterior scale), examined the body cavity to determine sex, measured the volume of retained eggs, collected tissue from the operculum using a hole punch for subsequent genetic analysis, and removed the tail to indicate the carcass had been sampled. Occasionally, a fin punch or small section of heart tissue were collected when carcasses were scavenged or degraded. Operculum and fin tissue samples were placed within a sheet of Whatman paper and inserted in a paper envelope to air dry, and heart tissue samples were preserved in 2 milliliter vials filled with 95% ethanol.

Parr sampling

We sampled parr throughout their known distribution in the mainstem MFJDR as well as in nine tributaries (Figure 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 sites were selected from 28 potential Generalized Random Tessellation Stratified (GRTS) sampling design (Stevens and Olsen 2004) sites located in the core summer rearing range (rkms 79-118). Given the potential importance of cool- and cold-water tributary use, we also sampled parr from nine tributaries within the spawning extent of the mainstem.

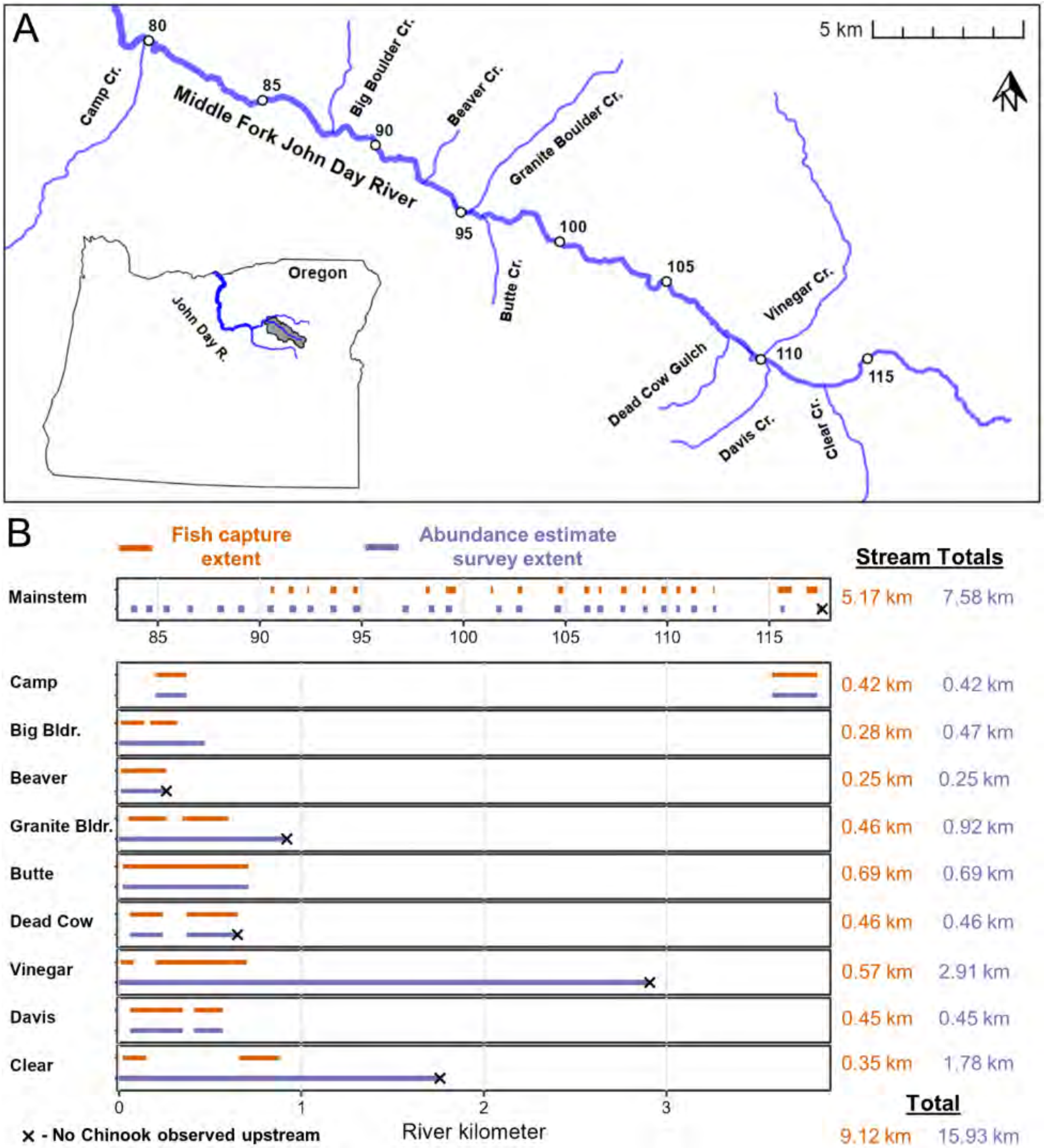

Figure 26. Study extent of the mainstem MFJDR and the nine tributaries sampled for parr in summer 2021 (A) and the extent of parr capture sampling (orange lines) and abundance estimate surveys (purple lines) within the mainstem and tributaries (B). Open points in Panel A indicate river kilometers from the MFJDR mouth (5 km intervals) for reference in Panel B and subsequent figures. The total length of capture and abundance surveys within each stream is indicated in the right column of Panel B. Black x's indicate that no juvenile Chinook Salmon were observed upstream of this location for at least three successive pools (i.e., upstream extent of Chinook Salmon parr within each stream). No x indicates that sampling did not occur upstream due to property access or other sampling constraints.

We sampled parr between 6/29/2021 and 8/19/2021. Based on predicted median emergence timing between 4/6/2021 and 5/10/2021(Figure S1; Kaylor et al. 2022), parr sampling occurred approximately 3-4 months post-emergence. Tributaries were generally sampled earlier than mainstem sites (Table 1), as 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.

			Parr sampling			
Stream	**July Q ($m^3 s^{-1}$)**	**July MDMT (°C)**	**Dates sampled**	**Parr sampled (*n*)**	**Parr paired to female (*n*)**	**Mean fork length (mm)**
Mainstem	596.0	21.8-26.2	7/13-8/9	1,592	595	67.3
Camp Cr.	12.2	21.3	7/9	28	13	60.0
Big Bldr Cr.	70.9	23.1	7/1-7/2	184	93	57.7
Beaver Cr.	21.3	19.6	7/14	63	12	62.2
Granite Bldr Cr.	124.3	17.5	6/30-7/22	292	80	68.3
Butte Cr.	4.1	18.8	7/14-7/20	248	70	61.1
Dead Cow Gl.	-	19.0	7/8-7/9	182	93	62.3
Vinegar Cr.	-	21.4	6/29-7/22	399	172	59.6
Davis Cr.	-	23.3	7/6-7/8	200	115	62.2
Clear Cr.	132.4	20.7	7/7-7/22	200	83	65.8
Total	-	-	6/29-8/9	3,389	1,326	64.7

Table 1: Parr sampling details for the mainstem Middle Fork John Day River and tributaries. MDMT = mean daily maximum temperature.

Genotyping

Tissue samples from adults and parr were genotyped to enable parentage analyses of parr to their parents. DNA was extracted from tissue samples using the Chelex 100 method, and then DNA libraries of barcoded individuals were prepared and sequenced following 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 quantified and then sequenced on an Illumina NextSeq 2000 instrument. The GTseq panel included 354 single nucleotide polymorphisms (SNPs), with a subset of 254 putatively neutral markers intended for parentage analyses along with a genetic sex marker to verify males vs. females (Hess et al. 2023). 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. Because some sampled carcasses were too degraded or scavenged to accurately determine sex, we relied on genetic sex assignments for all retained adults.

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 (adults originating outside the John Day River (JDR) 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 MFJDR 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 MFJDR 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 (false positive rate < 0.01). We filtered all parr-adult assignments to only those with LLR ≥ 20 and only evaluated dispersal using parr-female pairs, as male carcass locations were not expected to provide reliable proxies of redds due to movement after spawning and spawning with multiple females (Murdoch et al. 2009). Further, male carcasses sampled in this study were downstream-biased and often several kilometers away from females that they spawned with.

Abundance estimates

We evaluated juvenile Chinook Salmon abundance and distribution across the MFJDR 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 ranged from 174 to 388 m). We visually delineated habitat units as pools, fast-non-turbulent (FNT; i.e., runs, glides), fast-turbulent (FT; i.e., riffles), and 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 alternating FT habitat units 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 visibility (categorical value ranging from 0 to 3). In snorkeled tributaries, we applied the same approach, but we sampled approximately every fourth FT habitat unit. 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 due a transition to steeper gradient.

We predicted detection probability 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 estimate effects of habitat attributes on observer detection probability. For each unit (n = 432) we sampled 1000 detection probability values from the posterior predictive distributions given by Staton et al.'s model, which were used to expand the partial snorkel counts to 1000 abundance estimates per unit, providing a distribution of abundance estimates that accounted for uncertainty in modeled detection probability. We estimated abundance for skipped units using mean density (abundance 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 expanded 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 reach-, 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 (# m^{-1}) using linear interpolation of sampled sites. We generated 1000 density predictions for each site, which were then multiplied by reach length to a distribution of 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 a distribution of stream-specific and whole-basin abundance estimates.

Sampling bias adjustments

Interpretation of population-level dispersal patterns (i.e., the overall distribution) may be influenced by sampling bias if sampled parr do not represent a random sample of the population (Wacker et al. 2021). For example, over-sampling of sites (i.e., sampling a higher proportion of the fish present compared to other locations) would inflate the influence of these fish and the dispersal patterns they exhibit on inference of overall dispersal relative to fish captured at sites that were under-sampled. Ideally, the number of parr sampled at randomly selected sites would be proportional to parr abundance at each site (i.e., 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 because abnormally warm temperatures in July prohibited mainstem sampling. To reduce sampling bias effects on dispersal inference, we calculated and assigned sampling weights to individual fish based on capture reach.

Sampling weights were equal to the predicted proportion of the population located at each sampling reach (P_{pop}) divided by the respective reach sampling proportion (P_{samp}). Where,P_{pop} is equal to the mean predicted abundance at each reach divided by the predicted mean basin abundance (67,753) and P_{samp} is equal to the reach sample size divided by the total sample size (n = 3,389).

Dispersal analyses

We calculated dispersal as the stream distance between each parr-female pair such that negative values indicated downstream dispersal (i.e., parr captured downstream of females) and positive values indicated 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. While dispersal indicates directionality of movement, we also evaluated 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 for all parr and stratified by parr that were captured within the mainstem versus tributaries.

We used general linear mixed-effects models to evaluate potential relationships between dispersal patterns and parr origin (i.e., rkm of paired female), parr size, and temperature. 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 because size-at-capture was confounded by factors independent of dispersal, such as emergence timing and day of capture, as well as time elapsed between sampling events across all locations. Emergence timing is progressively earlier upstream in the MFJDR (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 most mainstem sites (Table 1) potentially influencing size-at-capture. To account for these complicating factors, we calculated relative size - an individual's measured fork length relative to predicted fork length. We fitted a set of candidate models predicting 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. 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). We then selected the model with lowest AICc, predicted parr fork length for each individual parr, and calculated relative length for each parr as the natural logarithm of the ratio of measured to predicted fork length (positive values indicate parr that were larger than predicted).

RESULTS

Genotyping

We identified 161 redds across the mainstem MFJDR and a single redd in Clear Creek in September 2020 (Figure S3). 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 S3) 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 of the 1,326 parr (44.9%) were captured from mainstem sites and 731 (55.1%) from tributaries (Table 1). Of the 67 females retained after genotyping, 64 had one or more parr that was assigned to them. 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 S4).

Abundance estimates

The estimated total parr abundance across the MFJDR was 67,753, with the mainstem accounting for nearly three quarters of all parr (Table 2). Among tributaries, total abundance estimates were greatest in Vinegar Creek, Granite Boulder Creek, and Clear Creek, accounting for 18.3% of total MFJDR 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 S5). 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 S5-A). Consequently, the distribution of redds was not well associated with mainstem parr density (p = 0.68; (Figure S5-B). In contrast, mainstem parr density was inversely related to July mean daily maximum temperature ($MDMT_{July}$; Figure S5 C-D). While estimated parr abundance was highest 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 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. 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). Adjusting for sampling bias shifted the overall distribution of dispersal estimates (Figure S7) from neutral directional bias (median = -0.03 km; IQR = -2.45 – 2.01 km) to slight downstream bias (median = -0.77; IQR = -3.69 – 0.92 km).

Stream	Method	N̂	N̂ 95% CI	% of total	Mean density (# m^{-1})	Max density (# m^{-1})
Mainstem	snorkel	49,096	45,149 - 54,937	72.6	1.27	4.85
Camp Cr.	shock	1,054	727 - 2,373	1.6	0.27	0.31
Big Bldr Cr.	snorkel	1,151	1,064 - 1,258	1.7	2.38	3.34
Beaver Cr.	shock	256	181 - 479	0.4	1.03	1.03
Granite Bldr Cr.	snorkel	4,254	3,323 - 5,813	6.3	5.16	11.37
Butte Cr.	shock	1,064	848 - 1,566	1.6	0.88	2.13
Dead Cow Gl.	shock	1,110	859 - 1,694	1.6	1.69	2.17
Vinegar Cr.	snorkel	4,643	4,400 - 4,981	6.9	1.53	2.96
Davis Cr.	shock	1,246	957 - 1,954	1.8	1.63	2.81
Clear Cr.	snorkel	3,485	3,052 - 4,025	5.1	2.11	4.12
Total		67,753*	63,365* - 73,750*	-	-	

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

Overall dispersal patterns

Parr dispersal was downstream-biased (median = -0.77 km) with 68% of all parr estimated to have dispersed downstream (Figure 2A). However, 25% of parr were 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 2B,C), with more 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 moving greater than 5.0 km (Figure 2D-F). Parr that dispersed downstream generally moved greater distances IQR: 0.73 – 6.03 km; max = 28.60 km) than parr that moved upstream 1.06 – 3.54 km; max = 10.61 km).

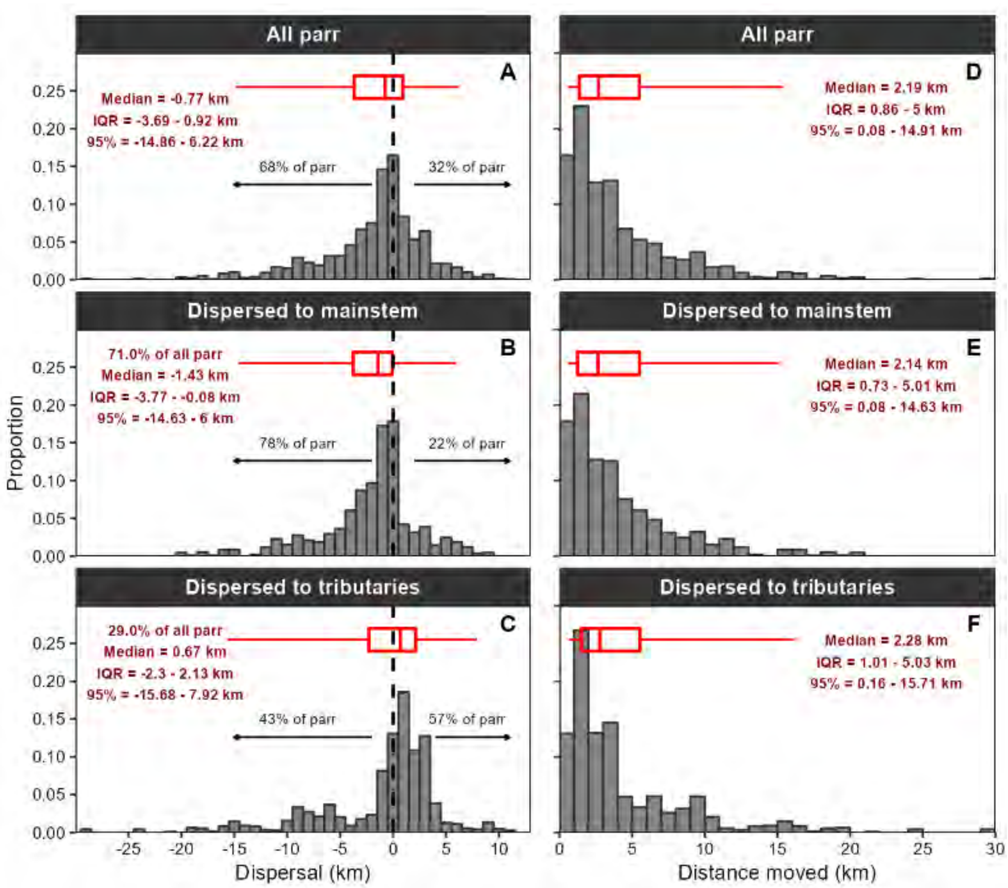

Figure 27: Overall distributions of sample-bias corrected dispersal (A-C) and total distance moved estimates (D-F). Box and whisker plots indicate median, IQR, and 95th percentiles.

Spatial patterns of dispersal

Distance and directionality exhibited in parr dispersal patterns varied as a function of where they originated (i.e., redd rkm) and dispersed to (i.e., mainstem versus tributaries; Figure 3). For parr that dispersed to mainstem locations (Figure 3A), 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 3B).

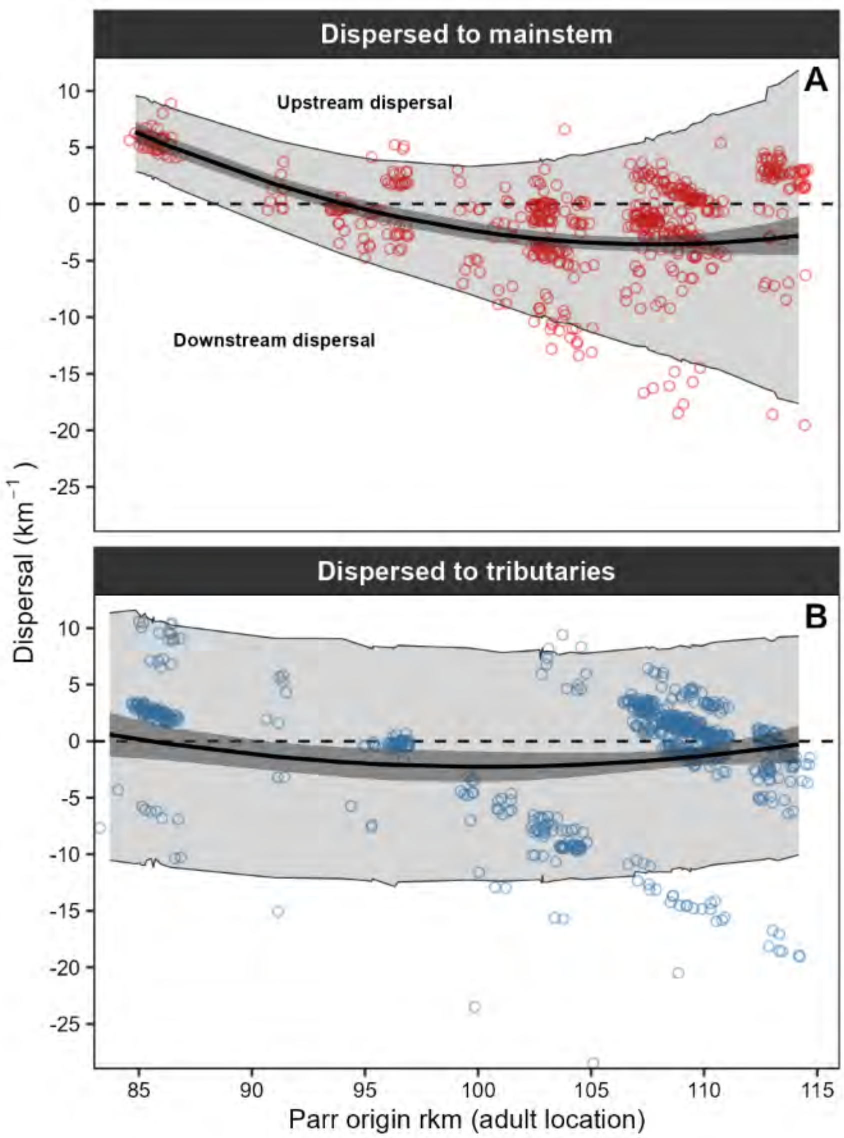

Figure 28: Parr origin (i.e., female adult location) versus dispersal for parr captured within the mainstem (A; red points) and parr captured within tributaries (B; blue points). The solid line indicates the fitted relationship between parr origin and dispersal; dark shading indicates the confidence interval of the fitted relationship; and the light shading and thin outer lines indicate the prediction interval, encompassing 95% of dispersal estimates.

Dispersal patterns generally followed a trend of movement from warmer mainstem sections to cooler mainstem sections or tributaries (Figure 4, Figure 5). 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 where $MDMT_{July}$ averaged ~2.3 °C cooler (Figure 5A) or to one of four tributaries between rkms 79.8-96.4 where $MDMT_{July}$ was 2-7 °C cooler. This is further exemplified by the negative relationship between individual parr origin $MDMT_{July}$ vs the difference between capture and origin $MDMT_{July}$ (Figure 5B) – there was little difference between capture and origin temperature for parr originating from areas where $MDMT_{July}$ was 23 °C or lower, but for parr originating from sections exceeding 23 °C, $MDMT_{July}$ averaged 2.5 °C lower at capture locations.

Insight on where juveniles originated from for a given section can also be gleaned from Figure 4. 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.

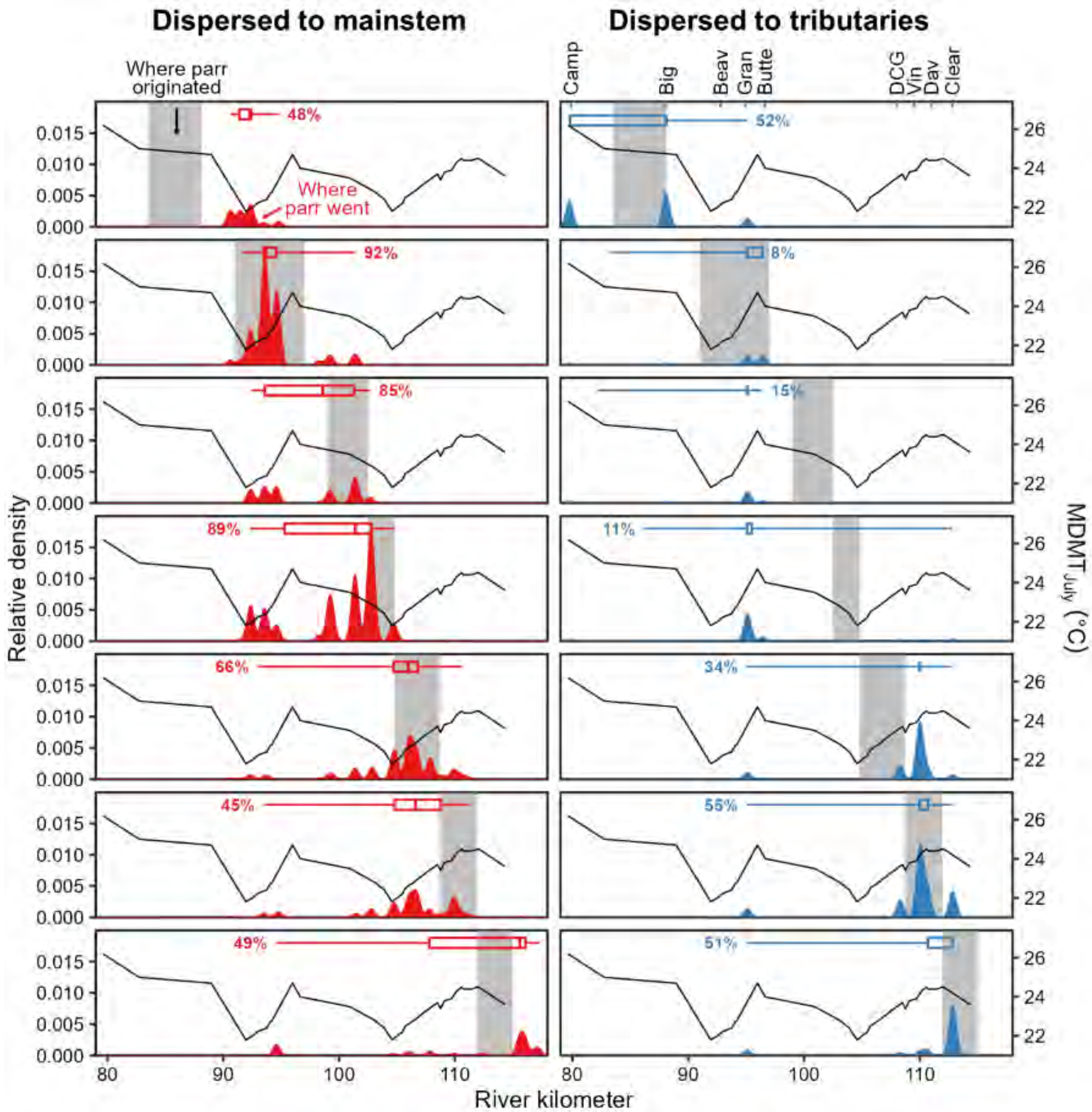

Figure 29: Parr dispersal patterns from different sections of origin (rows). Grey boxes indicate the section parr originated from; density distributions portray where parr from each section dispersed to across the mainstem (red distributions) and to tributaries (blue distributions); box and whisker plots indicate median, 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 longitudinal patterns of $MDMT_{July}$.

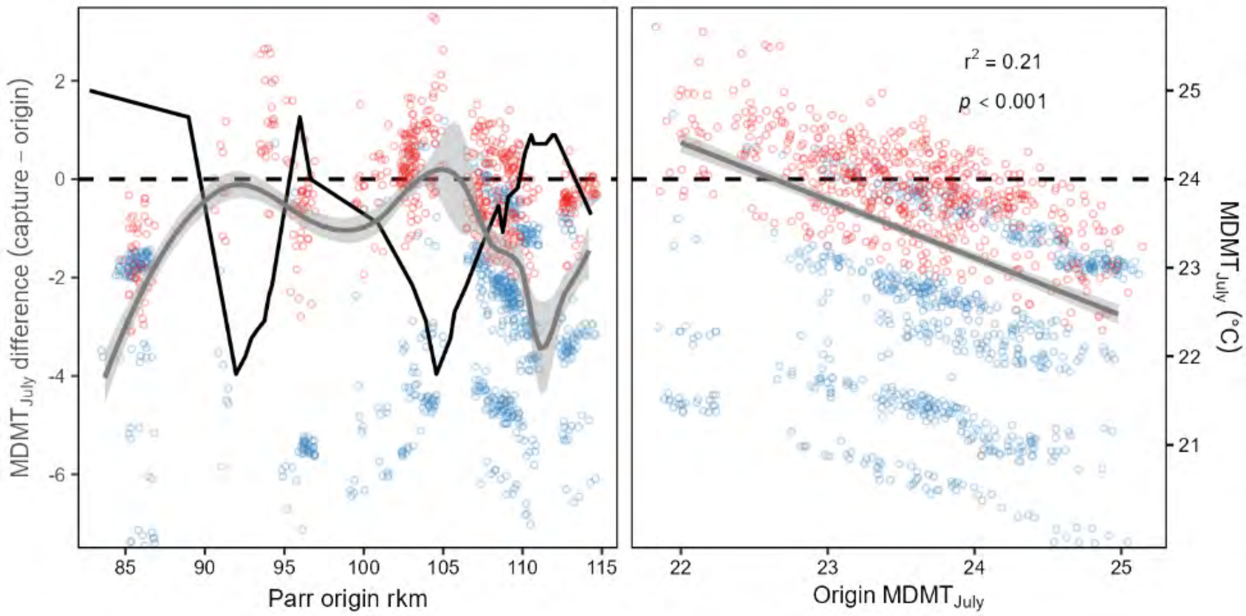

Figure 30: Stream temperature (MDMT July; solid black line) and estimated differences in temperature between capture and origin locations for individual parr (points) across the MFJDR (A) and the relationship between parr origin temperature and the difference in temperature between capture and origin locations (B). The color of points indicates whether each individual parr was captured within the mainstem (red) or tributaries (blue). The solid grey line in panel A shows a Loess fit. The figure highlights that parr originating in warmer areas tended to move to cooler areas.

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 MFJDR 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 6).

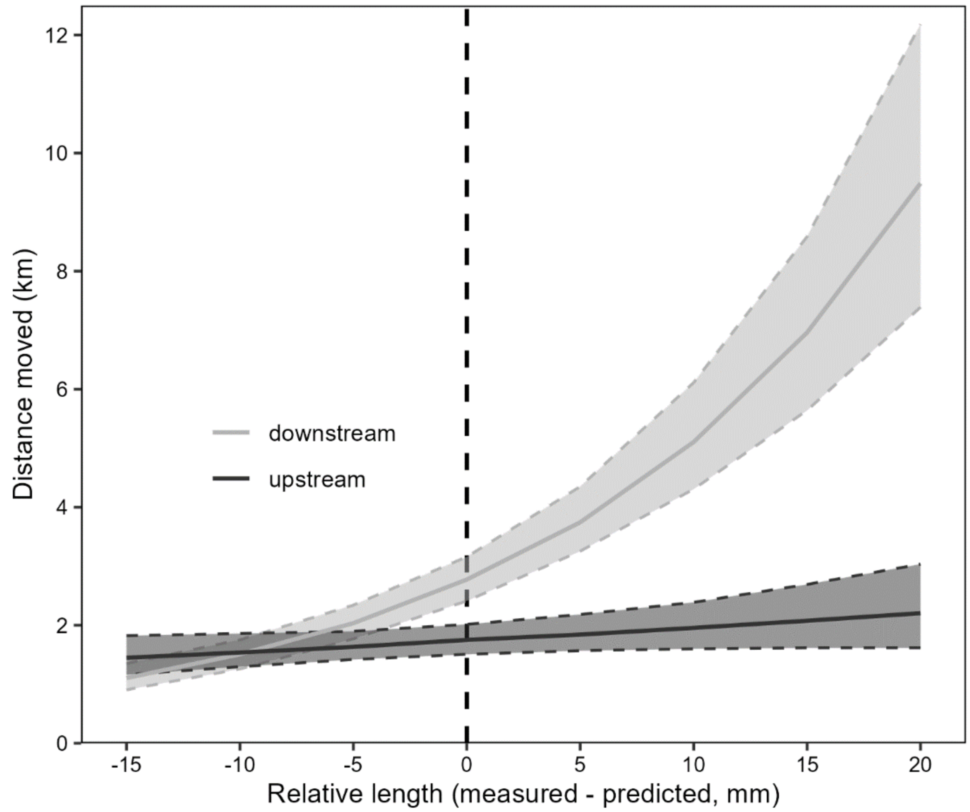

Figure 31: Fitted relationship between parr relative length (measured length – predicted length after accounting for parr origin river kilometer, stream of capture, and date of sampling) and the distance parr dispersed upstream (dark grey line and shading) or downstream (light grey line and shading).

DISCUSSION

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 than previously thought (Rodriguez 2002; Eisenhauer et al. 2021), and may vary as a function of population origin (e.g., wild vs. hatchery), geographic setting (e.g., high-elevation arid vs. coastal landscapes), and heterogeneity in habitat conditions within watersheds (e.g., thermal regimes, presence and configuration of cold-water tributaries). 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.

The Restricted Movement Paradigm (RMP) predicts that most stream dwelling fish (or freshwater life stages of anadromous fish) 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 binary concept of movers and stayers, but instead suggest a continuum of overall mobility. In a review of published estimates of juvenile Atlantic salmon dispersal, Eisenhauer et al. (2021) found that dispersal 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 substantial proportion (over one-third) of individuals dispersed upstream, 2) wider dispersal distributions, and 3) greater maximum dispersal distances, both downstream (nearly 5 km) and upstream (nearly 3 km). Our results similarly suggest that approximately a third of individuals dispersed upstream, but that dispersal distance was even greater (over 28 km downstream; nearly 11 km upstream). Further, there was no evidence of distinct 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 in previous studies could artificially generate bimodal distributions not characteristic of a random sample of the population.

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, earlier emergence of fry originating higher in the watershed (Kaylor et al. 2022) may have exposed them to greater flows, and by extension passive dispersal, compared to fry emerging up to five weeks later downstream. Indeed, estimated flows across the emergence period were greater (rkms 100-110) or more variable (rkms 110-115) in some upstream sections compared to downstream sections (Figure S1), potentially contributing to the greater variability in dispersal observed. Additionally, emergence timing may have contributed to greater dispersal variability upstream through ontogenetic effects 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. These patterns could also 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. Lastly, we recognize that the direction and distance of dispersal is constrained by the upstream and downstream limits of suitable rearing habitats (e.g., fish dispersing from headwater redds have less upstream habitat available to them, and fish dispersing from lower river redds are constrained by high water temperature in downstream habitats). This phenomenon of fish being "hemmed in" by upstream and downstream boundaries is not accounted for in our analysis of spatial patterns of dispersal.

There was a significant relationship between downstream dispersal distance and parr length in mid-summer after accounting for parr origin and capture location (i.e., relative size). Mid-summer size integrates emergence timing and growth prior to, during, and after dispersal and, consequently, the mechanism driving this association could stem from larger individuals dispersing farther, from greater post-dispersal growth rates of individuals that dispersed farther, or both. Indeed, numerous studies have found relationships between dispersal and body size at capture, 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 redds 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 prior to their arrival at a location. 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.

Correcting for sampling bias had considerable effects on interpretation of dispersal patterns in the MFJDR (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 our sampling extent 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 fewer 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 JDR (the Upper Mainstem), Chinook Salmon 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 MFJDR 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 MFJDR (but becoming increasingly more common) 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.

Lessons Learned

- Collecting samples from a high proportion of spawning adults, and in particular female adults, is critical.
  - Carcasses degraded rapidly which substantially reduced genotyping success. Further scavengers removed many carcasses or live fish from the river.
  - Consequently, considerable effort is required during active spawning. A crew of 4-6 persons solely dedicated to surveying for carcasses on a daily basis is recommended during peak spawning.
- For juvenile sampling, it is important that the sample design try to achieve a random sample of the population. In other words, the number of parr sampled at each randomly selected site should have been proportional to abundance at that site.
  - We attempted to correct for sampling bias using watershed-scale abundance estimates, which we believe was appropriate.
  - Future studies could consider sampling for a fixed amount of time and processing all individuals, although bias may still arise due to differences in capture efficiency and methodology.
- We may have underestimated the downstream extent of parr and parr in unsampled tributaries. Snorkeling efforts in summer 2023 and prior ODFW surveys have revealed parr utilizing numerous small tributaries that we were not able to sample in 2021 (e.g., Big Creek, Bridge Creek, Deerhorn Creek).

- If possible, juvenile sampling should be conducted in the shortest time window possible, as sampling across a wide temporal extent adds challenges to evaluate size vs dispersal relationships. This was not possible in 2021 given the extreme heat wave which prohibited mainstem sampling.

Future Directions

- The dispersal patterns in this study may not represent those in other years with different environmental and biological conditions.
- Evaluating dispersal patterns across a broader range of conditions may reveal different dominant drivers of dispersal (e.g., habitat quality vs temperature), which may be an important link between spawning and juvenile rearing distributions among years.
- Evaluating dispersal patterns in populations with a mix of natural- and hatchery-origin spawners, providing information on potential ecological impacts of hatchery parr on wild parr in rearing areas.

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Implementing this study was not possible without the contributions of numerous agencies and individuals who aided in field efforts, planning, providing data, and interpreting results. Spawning surveys were conducted by ODFW, CTWSRO, the North Fork and South Fork John Day Watershed Councils, the U.S. Forest Service, and volunteers from numerous other agencies. We especially thank C. Bare for efforts to coordinate spawning surveys and modify protocols to accommodate this study. We thank the numerous individuals who assisted in juvenile fish sampling including, but not limited to, J. Bailey, L. Blackburn, E. Booher, M. Cottingham, Z. Cunningham, H. Latzo, T. Sparrow, L. Osborne, C. Sheen, and A. Woolen, who are all rockstars. Additional logistical and planning support was provided by the MFIMW, particularly E. Booher, K. Bliesner and I. Tattam. Genotyping was supported in the laboratory by E. Collins, L. Maxwell, and M. Moore and genotyping analyses were supported by E. Collins. J. Hetfield, L. Ulrich, K. Hammett, R. Trujillo, and J. Newsted provided motivation and creative inspiration that was instrumental for report preparation. This research was supported by Bonneville Power Administration (BPA) funds as part of the Columbia Basin Fish Accords Agreement (project numbers 2009-004-00 & 2007-397-00), BPA funds administered to ODFW (1998-016-00), Pacific States Marine Fisheries Commission funds, State of Oregon General funds, and the Bureau of Reclamation (adult genotyping costs).

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APPENDICES

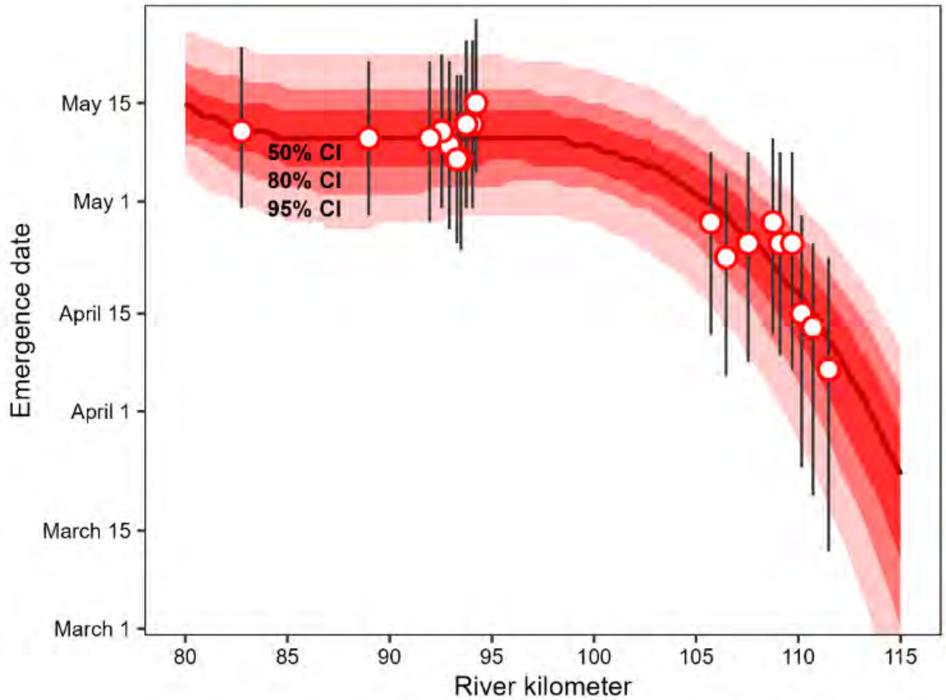

Figure S1: Emergence timing estimates across the MFJDR for 2021. Emergence was first estimated at 19 locations with annual temperature data through simulating emergence given variation in spawn timing (see Kaylor et al. 2022). Points indicate median emergence timing and error indicates quantiles encompassing 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.

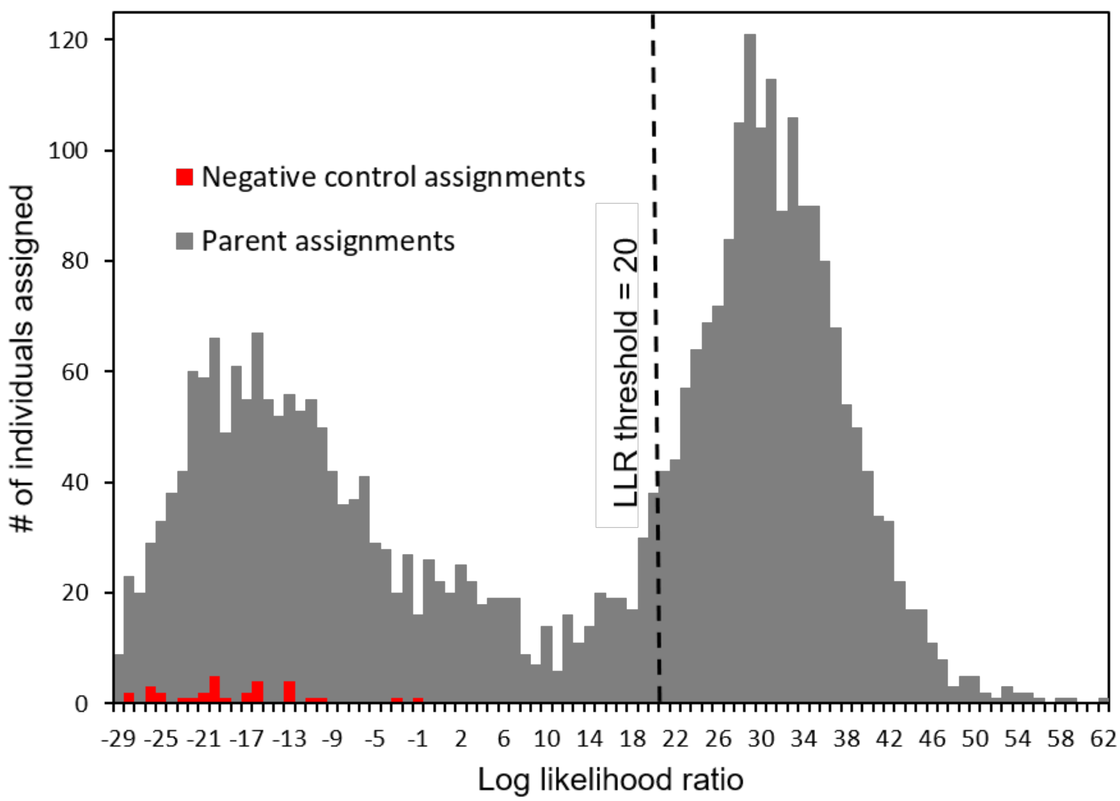

Figure S2: Distribution of log likelihood ratios for all MFJDRR parr-adult assignments (grey bars) and for negative control assignments (red bars). The bimodal distribution indicates two groups of parr-adult pairings, with the right distribution representing parr-adult assignments in which the adult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

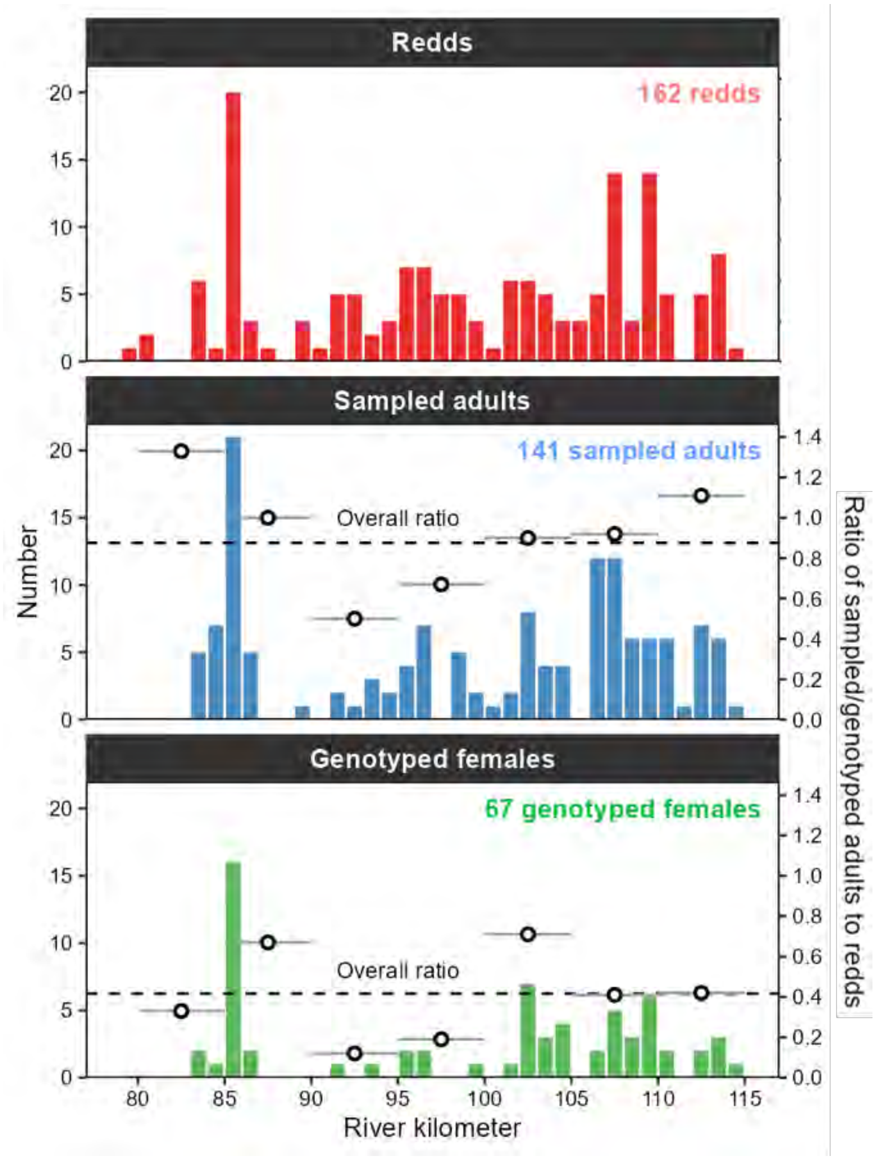

Figure S3: Number of redds (top; red bars), sampled adults (middle; blue bars), and successfully genotyped females (bottom; green bars) per kilometer in spawn year 2020. Points represent the ratio of sampled adults or genotyped females to redds for 5 km groupings (indicated by horizontal line associated 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).

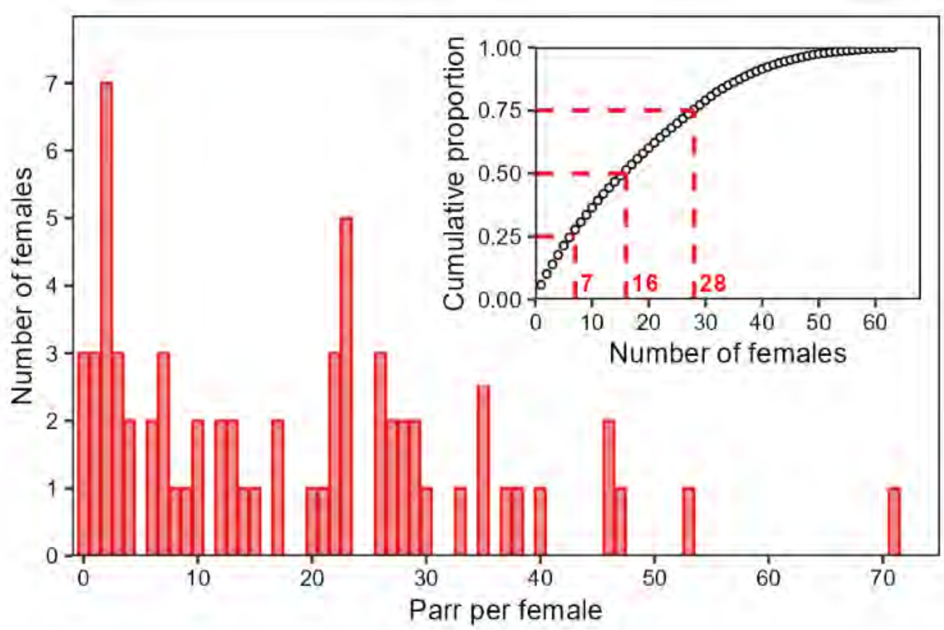

Figure S4: Distribution of number of parr paired to females after applying sampling bias correction. The inset shows the cumulative proportion of parr paired to females as the number of females – ranked by number of parr per female – increases. Out of the 67 females, 64 had ≥ 1 paired parr. The distribution 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.

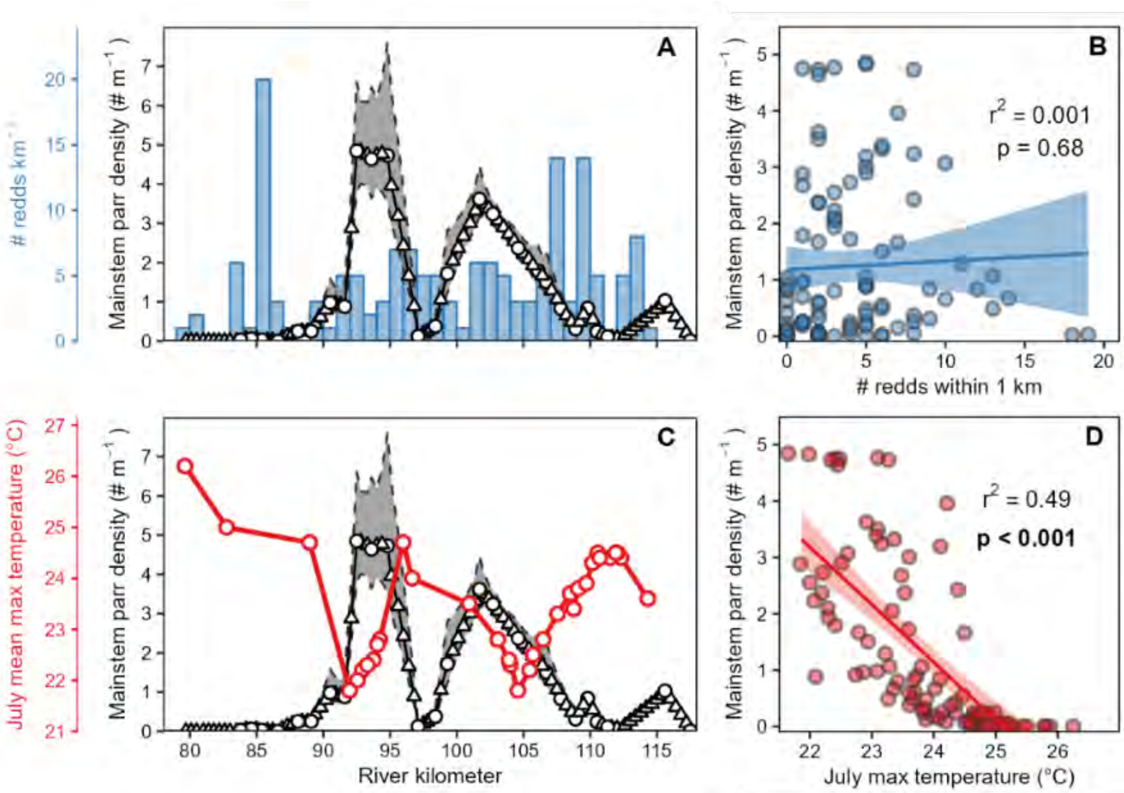

Figure S5: Mainstem spatial patterns of 2020 redds (A, blue bars), 2021 parr density (A,C; black/grey lines, points, and shading), July 2021 MDMT (C, red points and lines), and relationships between redds and density (B) and July MDMT and density (D). For density estimates, points represent snorkeled sites, triangles indicated prediction reaches, and grey shading between dashed lines show 95% confidence intervals.

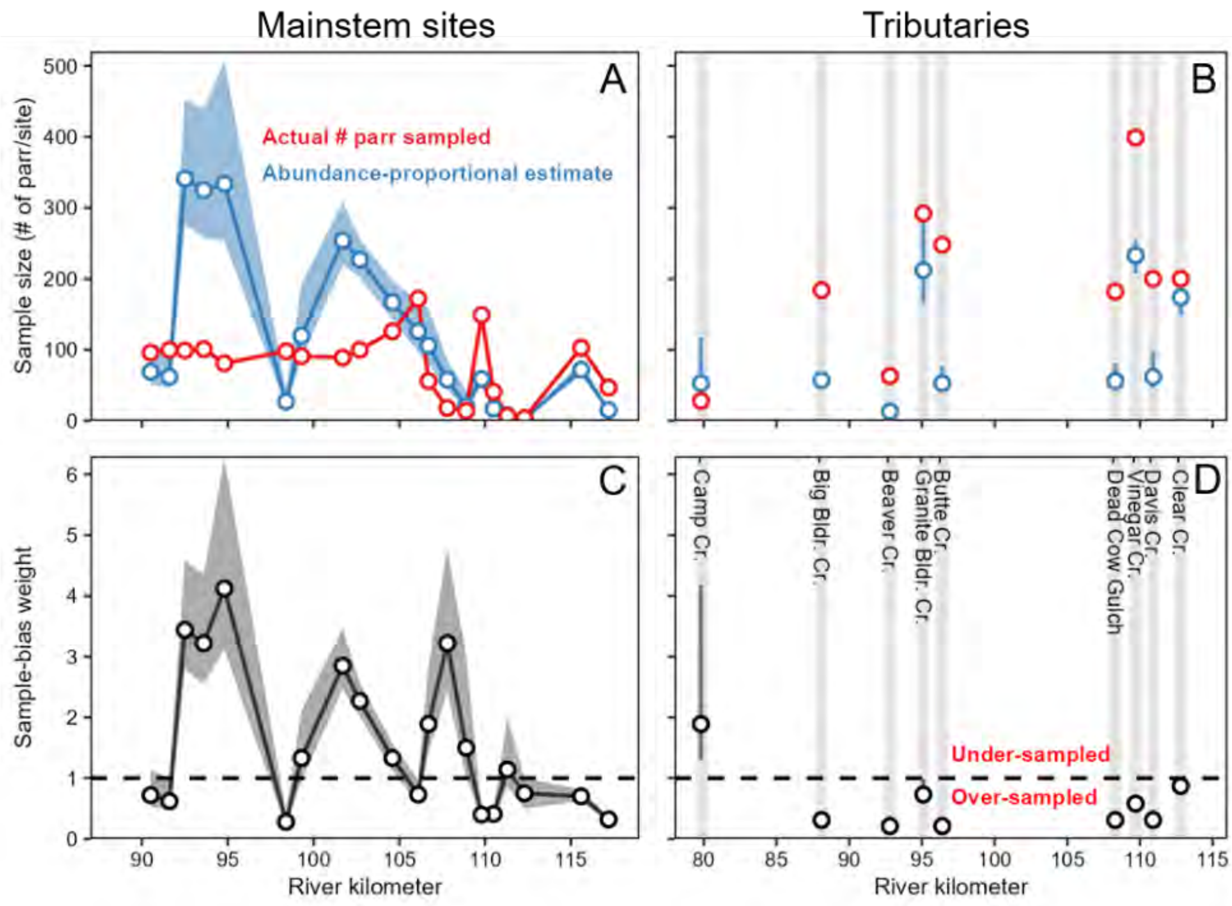

Figure S6: Number of parr sampled at each mainstem site or tributary (red points and lines; A,B), estimated abundance-proportional median number of parr that should have been sampled at each site (blue points, lines, and shading; A,B), and sample-bias weighting factors for each mainstem site or tributary (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.

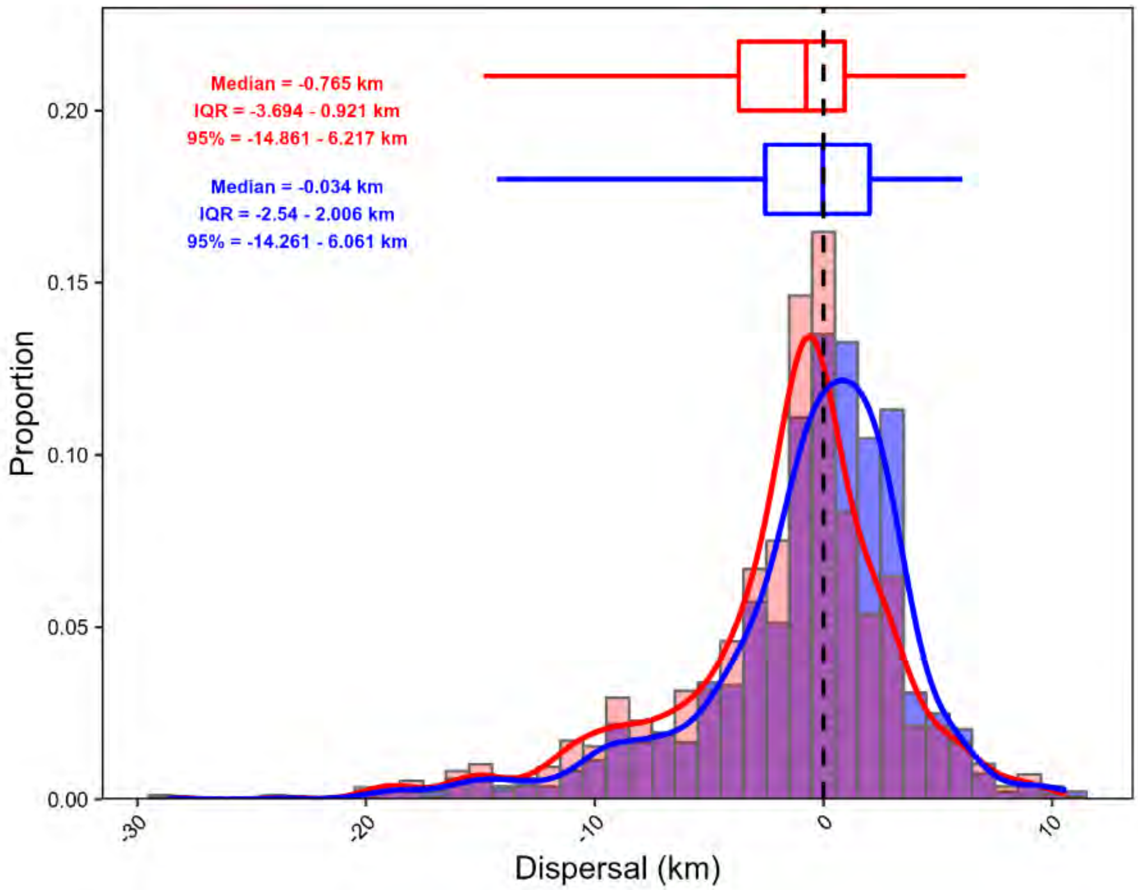

Figure S7: Distribution of dispersal estimates for all parr paired to females using raw values unadjusted for sampling-bias (blue bars and lines) and simulated, sampling-bias-adjusted estimates (red bars and lines). Box and whisker plots indicate median, IQR, and 95th percentiles.

CHAPTER 5: Long Term Effects of Passive & Active Restoration in the Middle Fork John Day River

Project leaders: Patricia McDowell (University of Oregon), Lisa Ellsworth (Oregon State University), and Matthew Goslin (University of Oregon and Oregon State University)

ABSTRACT

The objectives of this project (OWEB #218-6041) were to quantify long term changes in 1) floodplain and greenline (streamside) vegetation and 2) in-stream geomorphology and habitat relative to different restoration strategies. We quantified change by re-measuring sites in 2018-19 that were first monitored in 1996-7 under a different project, and by comparing aerial imagery between 1989 and 2017. We compared change across five management classes: Class 1 represented ongoing livestock grazing and Classes 2-5 represented various combinations of active and passive (grazing reduction or cessation) restoration implemented after 1996. Floodplain vegetation types did not show significant changes in area over time or by management class, but an increase in riparian woodlands and a reduction in gravel bar area was suggested. In contrast, greenline vegetation showed clear changes across classes: communities transitioned from mesic grasses toward deep-rooted sedges and other hydric species. Consistent with the establishment of more wet-adapted species closer to the water's edge, greenline-to-greenline channel widths narrowed across classes, and the full passive and passive + active classes showed greatest narrowing. Channel complexity as measured by number of habitat units per km increased in most full passive and passive + active reaches. Large wood loading increased in the two passive + active reaches sampled, due to placement of large wood in active restoration projects. Other geomorphic metrics – residual pool depth, percent channel length in pools, substrate size – did not show consistent patterns of change. These results indicate that both passive and active restoration can show positive effects on aquatic riparian habitat, but effects may take decades to be evident, and changes proceed at different rates for different processes. Furthermore, while active restoration projects may jump-start certain processes such as large wood accumulation, passive restoration can drive systemic changes such as greenline vegetation change and channel narrowing.

INTRODUCTION

Background

Physical habitat conditions in the channel and riparian zones of the Middle Fork John Day River (MFJDR) historically incurred significant degradation due to anthropogenic land use activities. Since about 2000, landowners and resource managers have initiated activities to restore this area, including both active and passive restoration (primarily reduction or removal of livestock grazing). Active restoration involves direct human actions to improve habitat conditions (i.e. riparian plantings, channel re-configuration, instream structures, etc.), while the approach of passive restoration is to remove stressors negatively impacting habitat (i.e., reduction or removal of livestock grazing, cessation of water diversion, etc.) and let habitat conditions evolve over time under natural processes (Kauffman et al., 1995)

In many cases, active restoration projects are implemented following passive restoration, and the two strategies work in concert to facilitate ecosystem recovery. However, the effects of passive restoration are typically not monitored separately from active restoration effects. In general, while active restoration projects are often monitored for at least a few years after implementation, management for passive restoration is rarely monitored due to the longer time span expected for response, and the fact that monitoring often is not required for passive restoration as it is for active restoration projects. Furthermore, it is rare to have available baseline data acquired prior to large-scale management changes that allow for assessment of long-term change across a watershed.

In the case of the MFJDR, a set of baseline data had been collected in 1994-6, immediately prior to significant changes in management, as part of a project named “Hydrologic, Geomorphic and Ecological Connectivity in Columbia River Watersheds: Implications for Endangered Salmonids,” funded by the U.S. Environmental Protection Agency (EPA) and conducted by six investigators from Oregon State University and University of Oregon (Li et al., 2000). The goal of this earlier project was to understand what factors and processes influenced summer stream temperatures in the upper MFJDR, particularly factors responsible for cool water sites that were important to native salmonids, rather than to evaluate or plan for restoration. Nevertheless, data were collected on channel morphology, bed material, floodplain and riparian vegetation, large wood in the channel, and other characteristics that allowed for an evaluation of physical habitat changes since the mid-1990s. These data were collected on the upper mainstem channel and floodplain upstream of Big Creek (Fig.1). The baseline data provided by the multi-disciplinary 1994-96 study (Li et al., 2000) was augmented by two other historic data sets: 1996 ODFW habitat surveys and 1989 imagery.

Subsequent to the 1994-6 period (mainly after 2000), a variety of active and passive restoration actions were implemented on some of the properties in the upper MFJDR. This allowed for division of the MFJDR mainstem into reaches of five management types (Table 1). Active restoration approaches implemented in the upper MFJDR that are relevant to this project include riparian and floodplain tree and shrub plantings, instream habitat improvement (particularly addition of large wood and pool development), channel reconfiguration, and floodplain reconnection (such as opening side channels). Passive restoration consisted mainly of reduction or cessation of livestock grazing. Adaptive grazing management refers to reaches that were managed by ranchers with adjustments in stocking levels and seasons reflecting their goals in managing specific properties, but without any explicit passive or active restoration. Management type 5 was added to reflect the impacts of recent restoration involving earth moving on vegetation cover with limited time for recovery. This management type was included as a distinct class only for floodplain vegetation assessment and not for greenline vegetation, given that none of the greenline vegetation sites occurred in areas with earth-moving impacts.

Table 1. Management types used in this report

Management type	Explanation
Adaptive grazing management (1)	Primary land use is domestic livestock grazing. Individual landowners manage animal numbers, timing, and type of grazing based on annual productivity and expert knowledge of the land. Neither passive nor active restoration are explicit goals.
Partial passive restoration (2)	Reduced domestic livestock grazing in terms of stocking rate, season of use, or duration of grazing
Full passive restoration (3)	Complete removal of domestic livestock grazing
Full passive restoration with active restoration (passive + active; 4)	Full passive restoration in combination with active restoration projects
Full passive restoration with active earth moving (used in floodplain vegetation analyses only) (5)	As in full passive with active, but also with recent earth-moving activities that impacted floodplain vegetation

Objectives and Questions

Field work for this project was undertaken in 2018-19 and other analyses continued into 2020. The overall goal of the project was to quantify changes in physical aquatic and riparian habitat since the 1990s, to determine whether there were differences in the trajectories of the five management types listed above, and to determine which of the five management approaches was most effective in improving habitat.

We hypothesized that full passive restoration with active restoration would show the most positive changes, and that adaptive grazing management would show the least positive changes. Under each objective, we identified specific questions to be answered related to the indicator variables we were able to measure. For these indicators, we identified what direction would be associated with positive ecological change [listed in brackets below]. For each objective, question or indicator we also indicate the data sources that were used (in parentheses).

Objective 1. Quantify long-term changes in vegetation

1a. Floodplain vegetation (mapping from aerial photos with field validation)

- How has the area of woody-dominated vegetation communities changed? [increase]
- How has the relative area of early, mid, and late-seral communities changed? [increase in late-seral]

1b. Greenline vegetation (greenline transects)

- How have species assemblages changed? Which key species have driven changes in the plant communities by diminishing or increasing? [not specified]
- How has the relative proportion of different plant forms (grasses, forbs, sedges and rushes, shrubs and trees) and adaptive types (mesic vs. hydric) changed? [increase in hydric types, sedges and rushes, shrubs and trees]
- How have indicators of wetland species, substrate stabilization capacity of vegetation, and diversity changed?
    - Wetland species index (mesic to hydric gradient) [increase]
    - Winward stability index (root strength/depth/coverage, longevity) [increase]
    - Plant diversity and richness [not specified]

- How has percent cover of woody species and canopy cover changed?
  - Percent cover of woody species [increase]
  - Canopy cover [increase]

Objective 2. Quantify long-term changes in instream geomorphology and habitat

- How has channel morphology changed?
  - Percent channel length in pool habitat (aquatic habitat survey) [increase]
  - Pool residual depth (aquatic habitat survey) [increase]
  - Channel width (greenline-to-greenline width from aerial imagery) [decrease]
- How has channel complexity changed?
  - Number of habitat units per unit length of channel (aquatic habitat survey) [increase]
  - Abundance of large wood in the channel (aquatic habitat survey) [increase]
  - Channel depth variability (longitudinal profile surveys) [increase]
- How has channel bed sediment changed? (gravel count)
  - Median grain size [not specified]
  - D95 grain size [not specified]

Site Selection

With the exception of channel width (greenline-to-greenline width assessed with aerial imagery) and habitat/geomorphology characteristics derived from the aquatic habitat surveys, site selection was limited by the locations and extents of individual sampling sites available in the 1990s studies. Sites and longitudinal extents are shown in Figure 1. Individual sampling sites are listed in appendix.

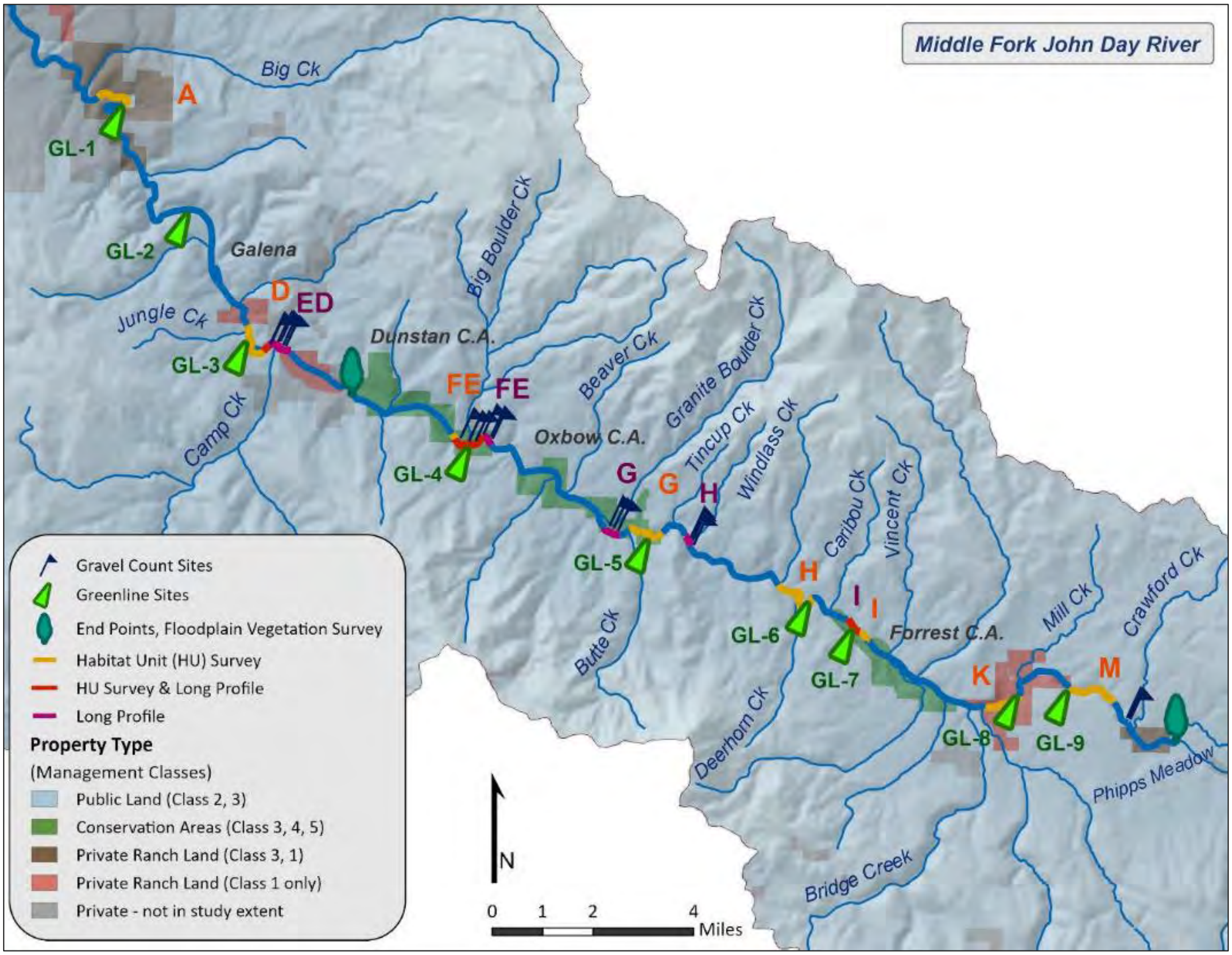

Figure 1. Study sites for vegetation, geomorphology and physical habitat. Sites are labelled as in the original 1990s study. Letters above the river indicate reach names. Habitat unit (HU) and longitudinal profile surveys (orange and purple labels, respectively) are shown as segments within those reaches. Greenline (GL, green labels) and gravel count sites are shown as points. As with the HU and long surveys, gravel count sites are identified by reach (see Appendices). Points show the upper and lower ends of the floodplain vegetation survey, an areal extent that varied in width. Property types are shown as colored areas. Management classes (1-5) found within these types (within our study extent, the mainstem MFJDR) are shown in parentheses. Management classes are defined in Table 1.

METHODS

Based on data available from the 1990s data sets, we identified a set of indicators that could be replicated, and we followed the 1990s methods as closely as possible.

Objective 1. Quantify long-term changes in vegetation

1a. Floodplain vegetation

The upper 39km (24 miles) of the MFJDR floodplain vegetation was mapped in 1996 and 2018 by drawing vegetation community (Crowe and Klausnitzer 1997) boundaries on a base map of aerial imagery, in the field. These data were converted to shapefiles (minimum polygon size $4m^2$), and polygon areas were calculated in GIS. Vegetation patches (polygons) were classified into 9 general riparian communities: cottonwood, ponderosa, mixed riparian woodland, alder, black hawthorn, willow, wet meadows (sedge-dominated), dry meadows (grass-dominated), and gravel bars.

1b. Greenline vegetation

The greenline is defined as “the first perennial vegetation that forms a lineal grouping of community types on or near the water’s edge” (Winward 2000). The greenline survey consisted of 3 components measured streamwise along the channel margins: 1) a greenline point transect that sampled species of all vegetation types, 2) a belt transect sampling percent woody cover (shrub and tree), and 3) a point transect measuring canopy cover. Greenline point transects consisted of two 50 m transects, one on each riverbank, following the greenline as defined by protocol in Winward (2000) and Burton et al. (2011). At 1m intervals along each point transect, we recorded the stem rooted nearest to the point, yielding a total of 100 species occurrence records at each site. For the belt transects, we recorded percent foliar cover of all shrubs and trees within a 2m-wide belt extending from the greenline onto the floodplain. For canopy cover, we used a densiometer (gridded mirror) to estimate percent cover at 10 sample points (5 m intervals) along each greenline point transect which were then averaged by site. In assigning management classes to greenline sites, we defined “active restoration” only on the basis of whether activities had occurred that could affect species composition within the transects, i.e. greenline and nearby floodplain plantings.

We described overall changes in species assemblages and the key species driving change using the multivariate ordination method, NMDS, from the vegan package of R statistics (Oksanen et al. 2019). We assessed changes within and between management classes for the following variables: a) percent of plant type (mesic vs. hydric, plant form), b) wetland species index, c) Winward greenline stability index, d) species richness and diversity, e) shrub percent cover, and f) canopy cover. We calculated wetland and stability indices as a weighted average of the ratings for all recorded species using those provided by the Multiple Indicator Monitoring protocol (Burton et al. 2011) which are derived from Winward (2000) for greenline stability and the National Wetland Plant List (Lichvar et al. 2016). The Winward stability index assigns plant species ratings that represent their potential to stabilize substrate based upon root depth, form and coverage as well as plant longevity.

Objective 2. Quantify long-term changes in instream geomorphology and habitat

We assessed changes in instream geomorphology using field methods in 1996 and 2018-9, including habitat surveys, longitudinal profiles, and gravel counts. In addition, we used 1989 and 2017 aerial imagery to measure greenline-to-greenline channel widths. In assigning management classes to the sites, we defined active restoration on the basis of activities expected to influence instream geomorphology and habitat, such as instream log structures and streambank planting, but excluding active vegetation planting on the floodplain that did not impinge on the channel.

Aquatic habitat surveys

An aquatic habitat survey was conducted on the entire 35 miles of study area in 1996. In 2018-19, we selected 8 reaches distributed across different management types, each about 0.75-1.3 miles in length, to repeat the habitat survey. The 1996 habitat surveys were conducted by Oregon Department of Fish and Wildlife following the methods of Moore and others (1999); data for these surveys was downloaded from Oregon Dept. of Fish and Wildlife (2014). We followed the same protocol in 2018-19, except that the dimensions of each habitat unit were measured directly using a tape, rather than sampled and estimated. From the habitat survey data we extracted five indicator variables: pools per km, percent of channel length in pools, residual pool depth, habitat units per km, and large woody debris. We minimized measurement inconsistency among teams through careful training and explicit written instructions.

Because of the small sample sizes across multiple treatments (8 habitat survey reaches, 5 longitudinal profiles, and 22 gravel counts), statistical tests were not feasible, so visualization and qualitative comparisons were used to analyze these channel geomorphology variables.

Longitudinal profiles

Longitudinal topographic profiles of the thalweg, about 0.5 to 1 km long, were surveyed in 1996 by McDowell and her team, and the surveys were repeated in 2019 using the same method (Li and others, 2000; McDowell, 2001). A total station instrument was used in 1996 and an RTK GPS in 2018-19. The goal was to represent the bed morphology realistically so, points were surveyed at irregular spacing to capture each riffle and pool. Average spacing between points was 6 to 10 m for each profile. One reach (reach I) was surveyed in 1996-7 by R. Beschta of Oregon State University (Li and others 2000), using a different protocol, and that protocol was repeated in 2019. Bed elevation was standardized by removing channel slope, and then the standard deviation in bed elevation was calculated for each reach. This standardized deviation was used as an indicator of depth variability, reflecting one aspect of channel habitat complexity.

Aerial imagery: Greenline-to-greenline widths

We assessed changes in greenline-to-greenline (GL-GL) widths using aerial imagery from 1989 (0.56m resolution) and 2017 (0.3m resolution). GL-GL cross section lines were measured in GIS at sample points spaced at regular intervals (100 m or 200 m) along the river centerline, yielding 50-60 sample points for each management class. Within each management class, sample points were distributed evenly across different management units across the watershed. Both unconstrained reaches with wide floodplains, and constrained canyon reaches are found within the MFJDR, but we measured only unconstrained reaches because Class 1 and 4 sites are found almost entirely within unconstrained, wide floodplains. We assessed within-class changes between years using absolute width measurements. However, in order to compare change among classes, we compared change as a percent of the 1989 width, a necessary step to normalize the data given that widths vary from upstream to downstream and management classes were not distributed evenly upstream to downstream. Within-class changes were assessed with non-parametric paired Wilcoxon tests, and between-class differences in change were assessed with the non-parametric Kruskal-Wallis.

Gravel counts

Bed material was measured in 1996 by McDowell and her team through surface gravel counts on riffles following the methods of Harrelson and others (1994). In 1996-7, 100 gravels were measured at each site, and in 2018-19, the same riffles were located, and 300 gravels were measured on each riffle. Based on these data, the mean size and the 95th percentile (95% of gravels are smaller than this diameter) were calculated for each data set, following the methods of Bunte and Abt (2001). Fine sediments were not recorded in 1996, so it was not possible to compare the percent of fines between 1996 and 2018-19.

RESULTS

Objective 1a: Floodplain Vegetation

The most common floodplain vegetation cover types were wet meadow (42.4% in 2018) and dry meadow (40.9% in 2018), followed by mixed riparian woodland, ponderosa pine, willow, gravel bar, cottonwood, alder and hawthorn. Generally, there was no statistically significant difference in vegetation cover type by time or time*treatment, but riparian woodland tended to increase in all classes except adaptive grazing management (Fig. 2-4).

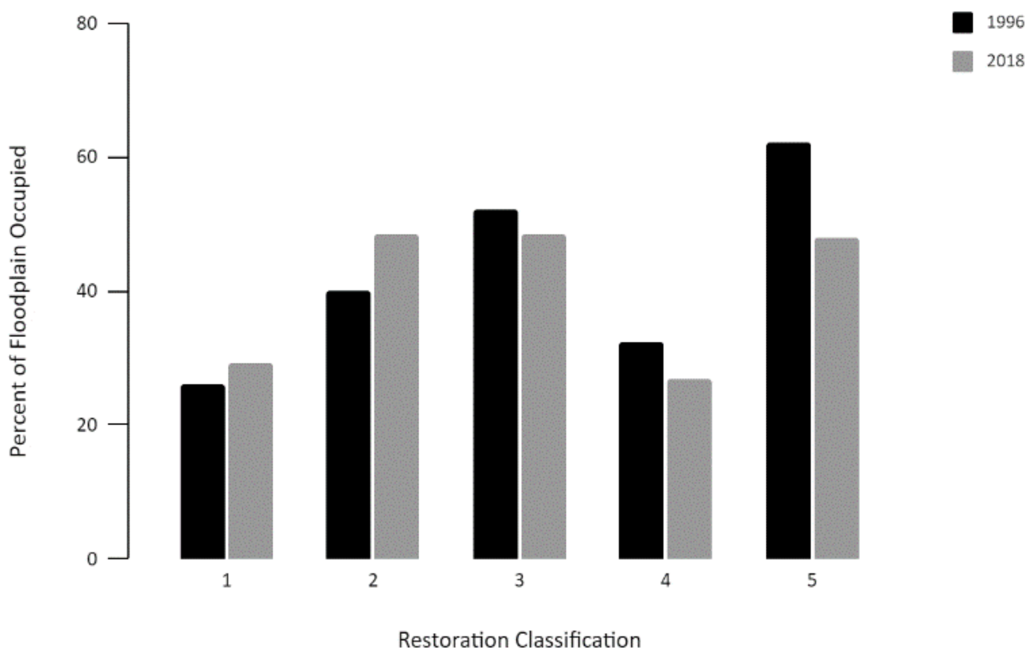

Figure 2. Change in area of dry meadow. Restoration Class 1: Adaptive grazing management; 2: Partial passive restoration; 3: Full passive restoration; 4: Full passive + active; 5: Recent earth-moving disturbance on floodplain during active (+ passive)

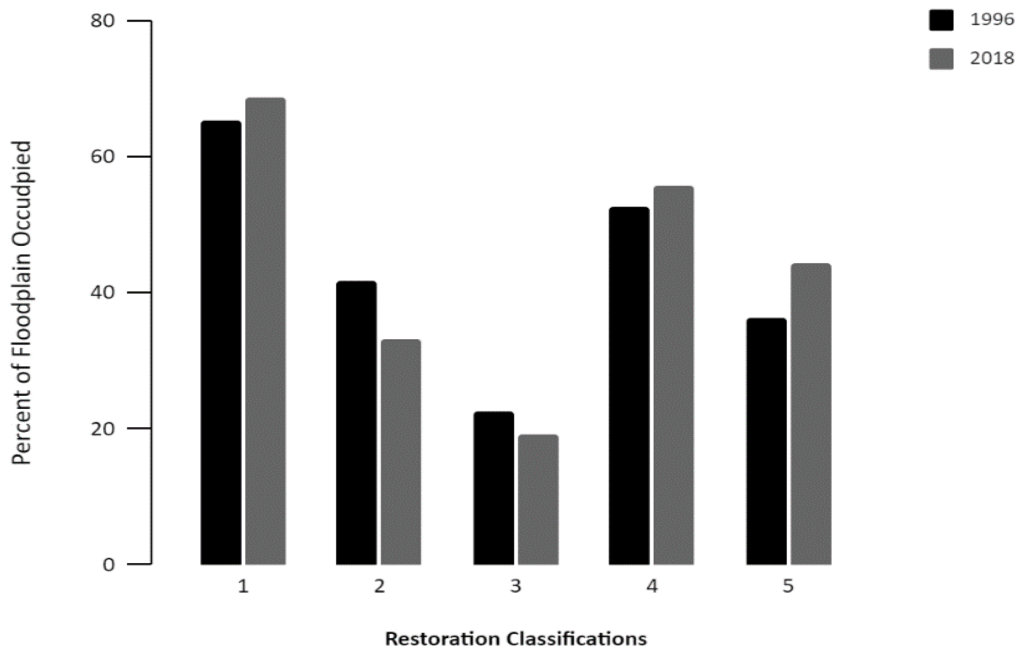

Figure 3. Change in area of wet meadow.

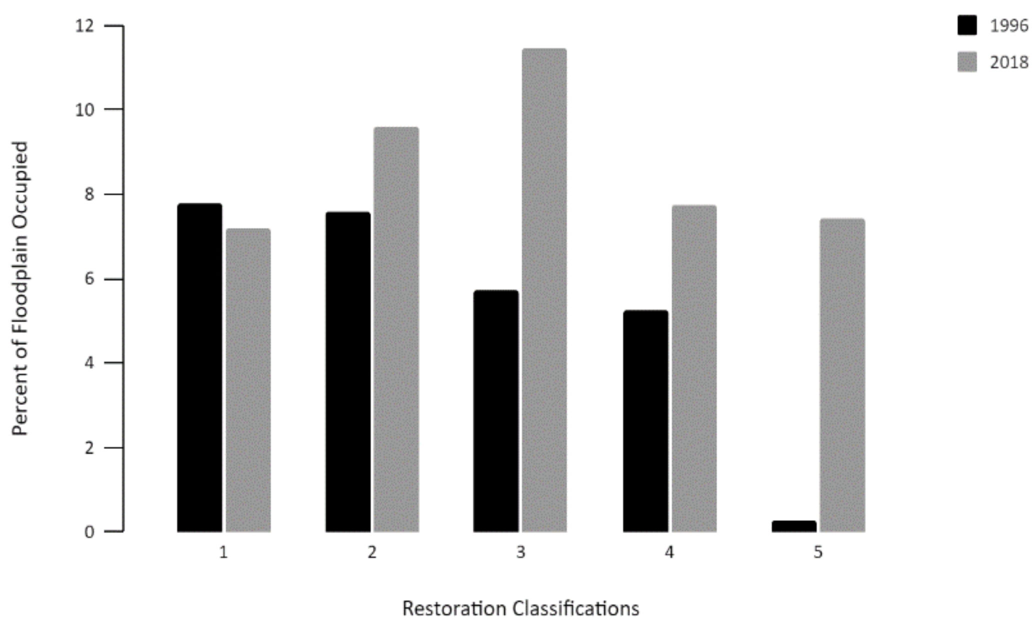

Figure 4. Change in area of mixed riparian woodland.

The number of polygons within a vegetation type (reflecting floodplain vegetation heterogeneity) also generally did not vary by time or time*treatment. The exception was a decrease in the number of gravel bar polygons over time in all management classes, suggesting revegetation of gravel bars as grazing pressure decreased.

Objective 1b: Greenline Vegetation

The greenline, "the first perennial vegetation that forms a lineal grouping of community types on or near the water's edge" (Winward 2000) is a critical point for monitoring. Greenline position and species composition reflects the balance between a) the ability of particular plants to colonize and maintain themselves at the river's edge, b) the scouring energy of the river and c) other vegetation-disturbing agents such as grazing by cattle or wild ungulates. The greenline may run along the top of cut banks or within the active channel between banks, for instance as the leading edge of vegetation colonizing a gravel bar or as vegetation established at the base of a cut bank.

From 1996 to 2018, greenline plant communities across all classes shifted dramatically in species composition. No difference in direction or degree of change was evident among management classes. The NMDS analysis identifies this overall change as being driven by a shift from mesic grasses such as *Poa pratensis* (Kentucky bluegrass) and *Agrostis stolonifera* (redtop) in 1996 toward sedges such as *Carex nudata* (torrent sedge), *Carex nebrascensis* (Nebraska sedge) and *Eleocharis palustris* (common spike-rush) as well as other associated species such as *Mentha arvensis* (wild mint) and the hydric grass, *Phalaris arundinacea* (Reed canary grass) in 2018 (Fig. 5).

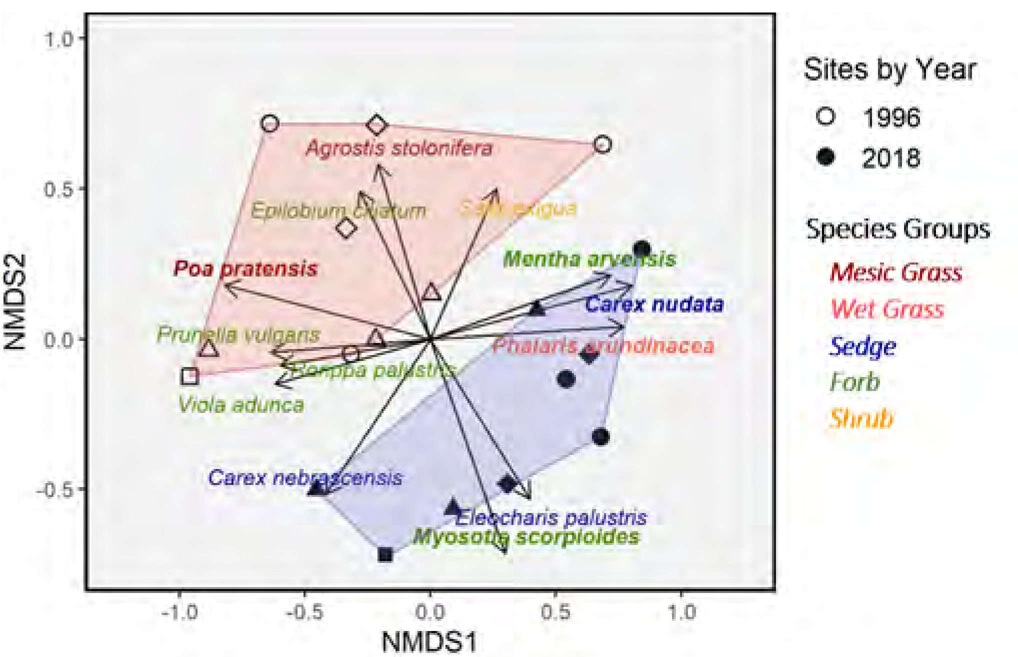

Figure 5. NMDS ordination plot of greenline sites by year (point symbols) and the species vectors that contribute significantly (p<0.025). Vectors labelled by species name indicate which species are contributing most to the differences among sites and years. Species in bold type are significant at $p < 0.001$. Polygons outline the species-defined space occupied by the sites in 1996 (light red) and 2018 (light blue). Point symbol shapes represent management class by site: Square = 1. Adaptive grazing; Triangle = 2. Partial passive; Circle = 3. Full passive; Diamond = 4. Full passive + active.

All of the key species defining the 2018 space -- *C. nudata, C. nebrascensis, Mysotis scorpiodes*, *P. arundinaceae,* and *M. arvensis* -- are wetland obligate or facultative wetland species that would be expected close to the water table and the river's low flow edge. In contrast, the species defining the 1996 space are a mix of facultative wetland and facultative or obligate upland species more typically associated with the floodplain or elevated areas of gravel bars. *C. nudata* is a native species that was suppressed by livestock grazing in 1996. It occurs in the middle and downstream parts of the study area but has an upstream limit. Within its extent of occurrence, *C. nudata* is now the dominant species whose recovery (as a native species) and expansion is driving the 1996-2018 shift. Above *C. nudata*'s upstream limit (GL06), *C. nebrascensis* becomes the dominant greenline species.

The NMDS results are corroborated by other metrics. The proportion of mesic grasses and forbs decreased from 1996 to 2018, while the proportion of sedges and rushes (predominantly *C. nudata*), wet grasses and wet forbs increased (Fig. 6)

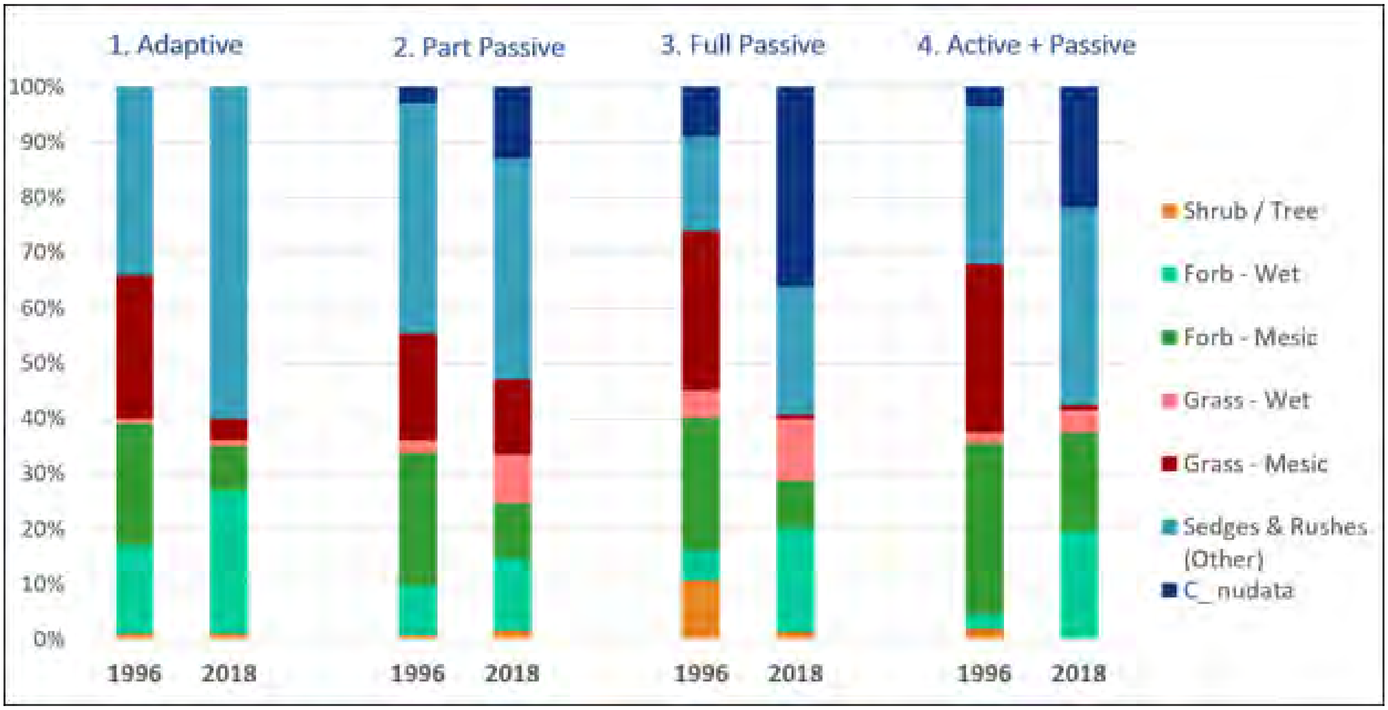

Figure 6. Proportions of species groups by management class and year. Note: Class 1 (adaptive grazing) only includes one site and this site is above the upstream limit of *C. nudata* extent. Sedges and rushes were not split into mesic and wet groups given that wet species made up >95% of their occurrence across years.

Across all classes, the wetland species indices increased from 1996-2018 (Fig. 7). In addition, the greenline stability index increased from 1996-2018 in all classes (Fig. 8). The increase in greenline stability reflects the decrease in relatively shallow-rooted grasses and an increase in long-lived sedges and rushes that are either rhizomatous (*C. nebrascensis*, *C. utriculata*, *Myosotis scorpiodes*) or non-rhizomatous but characterized by exceptionally strong, extensive root systems (*C. nudata*). The data suggest that these increases were greater for Class 3 and 4 than Class 2 but differences among classes were not significant. For species richness and diversity, there were no significant changes from 1996-2018 and no differences among management classes.

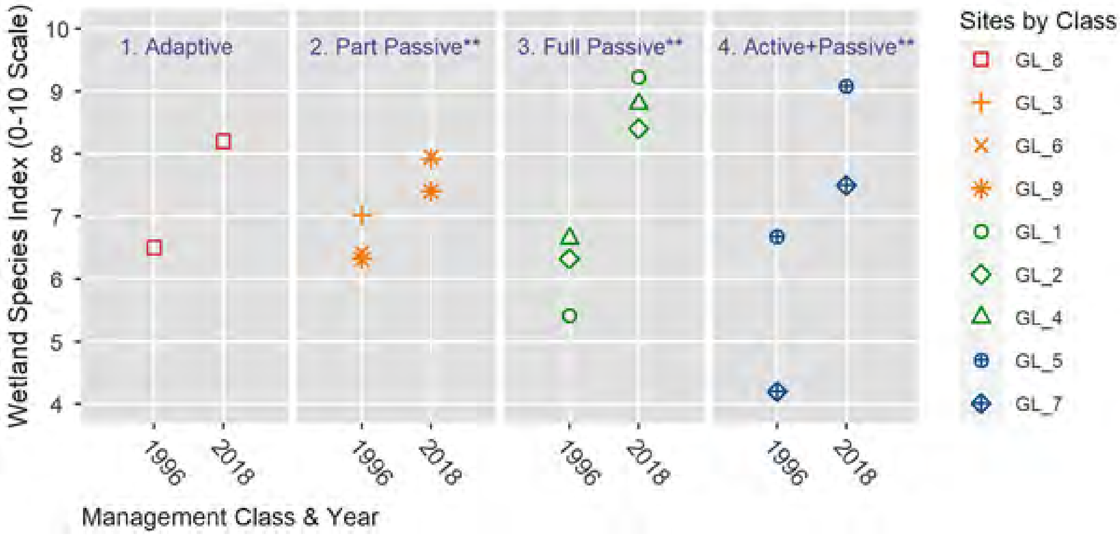

Figure 7. Wetland species index by management class and year. **$p < 0.05$ for within-class differences 1996-2018.

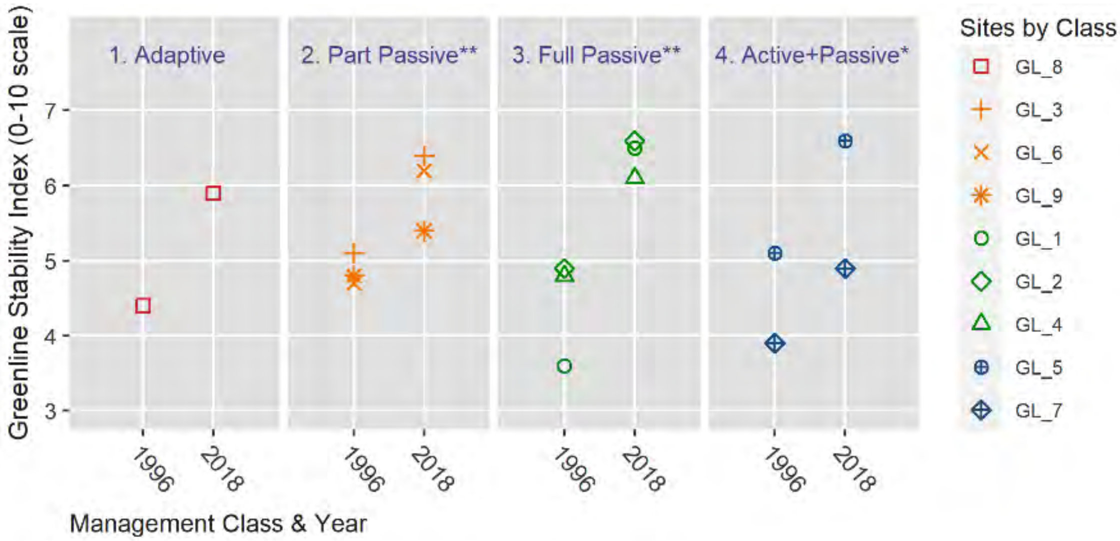

Figure 8. Winward greenline stability index by management class and year. **$p < 0.05$ for within-class differences 1996-2018.

Shrub cover in the belt transects declined in Class 2 and 3 and for the one site in Class 4 that had substantial shrub cover in 1996 (Fig. 8). Canopy cover did not show change between 1996-2018 for any classes. The decline in shrub cover is a counter-intuitive result that is largely a product of sampling protocol and actually consistent with other changes in greenline vegetation and channel narrowing (see Interpretation of Findings).

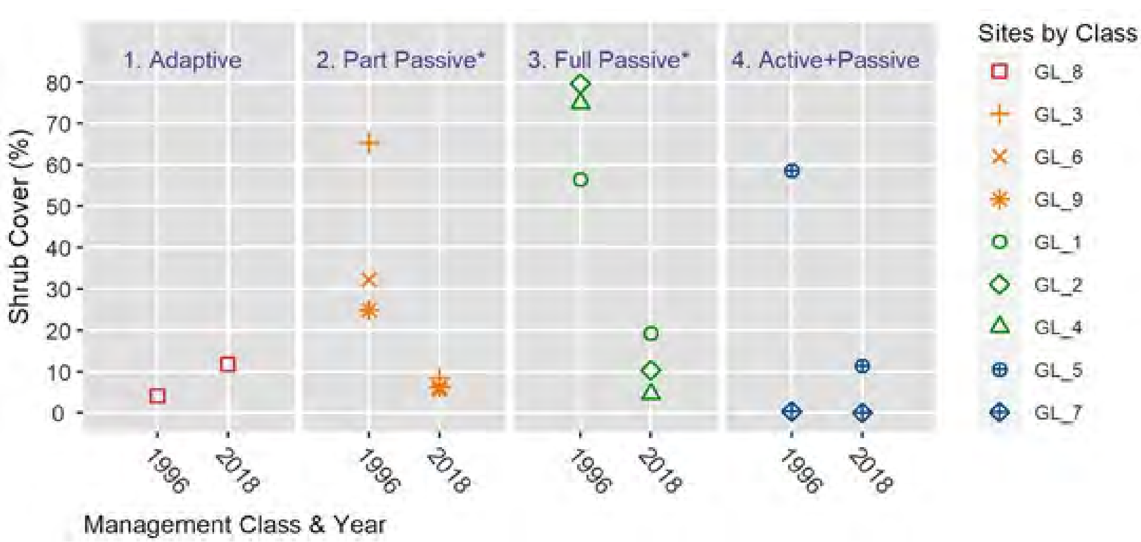

Figure 9. Shrub cover by management class and year. **$p < 0.05$, *$p < 0.1$ for within-class differences, 1996-2018.

Objective 2a: Pool quality

Percent of reach length in pools (Fig. 10) remained relatively constant or decreased in most reaches. Modest increases were observed in two reaches, one in passive restoration and the other in passive + active. There is no clear effect of management. Percent of length in pools varies widely across the eight reaches, reflecting differences in geomorphic potential to develop pools and other local site conditions, with unconfined low-gradient reaches having more pool length than confined or steeper reaches. In contrast to pools, almost all reaches showed a large increase in glides. This suggests that either units called glides in 1996 were called pools in 2018-19, or pools were converted to glides between 1996 and 2018-19. Pools can be converted to glides by filling with sediment, in this case gravel and cobbles. Possibly both factors, differences in identification and filling of pools, contributed to the changes observed over time.

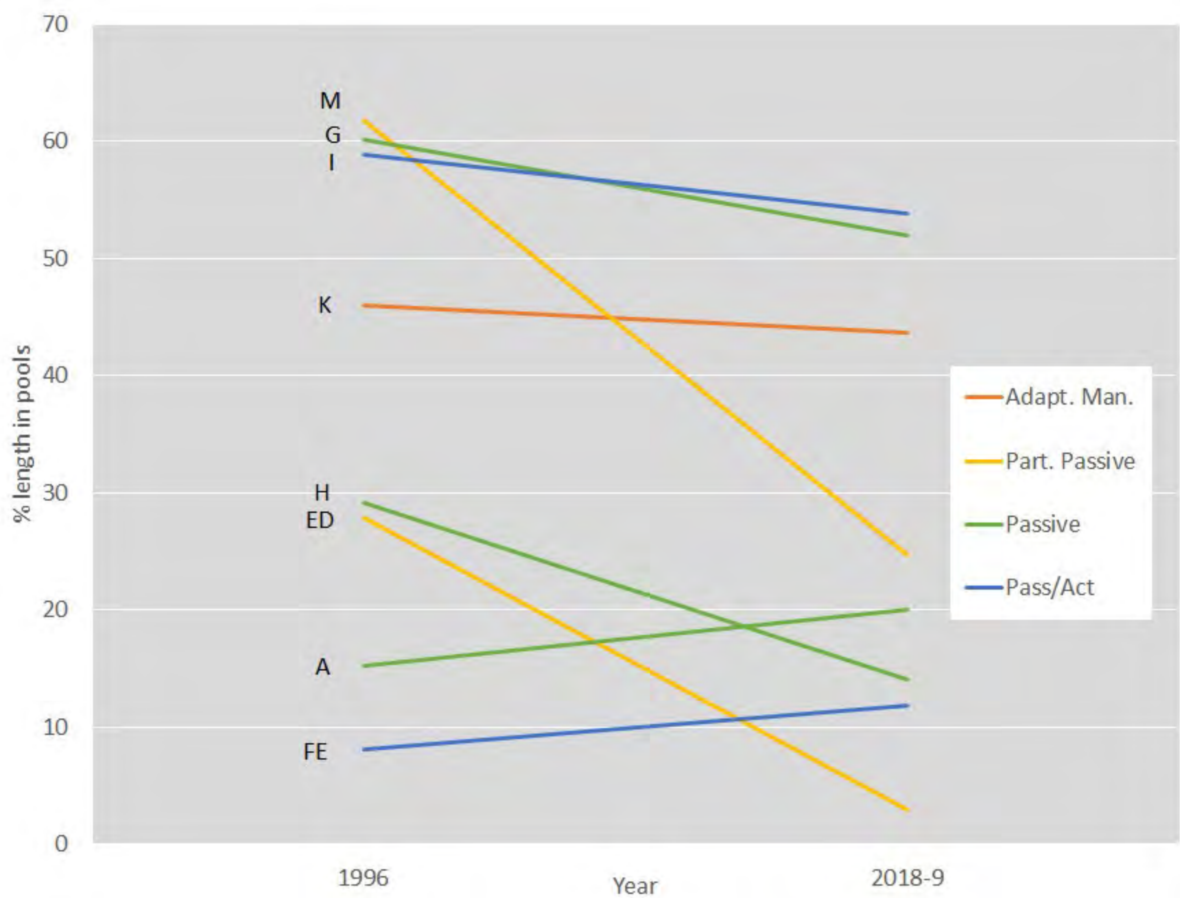

Figure 10. Changes in percent of channel length in pools.

We hypothesized that passive restoration reaches would have an increase in pool depth (Fig. 11), as removal of livestock disturbance and vegetation response would stabilize and narrow channels and suppress sediment deposition. However, both passive reaches had decreases in residual pool depth. We hypothesized that passive + active restoration sites would also increase in residual pool depth, because in these projects many pools were mechanically dug deeper and LWD structures were installed. Responses were inconsistent among the three passive + active reaches measured. Reach K (adaptive grazing) had a large increase in pool depth, possibly due to the very low pool depth in 1996 which apparently was ameliorated over time under adaptive grazing. In general, reaches which had deep pools in 1996 tended to decrease, and reaches with shallow pools in 1996 tended to increase, but it's not clear what the mechanism for this would be.

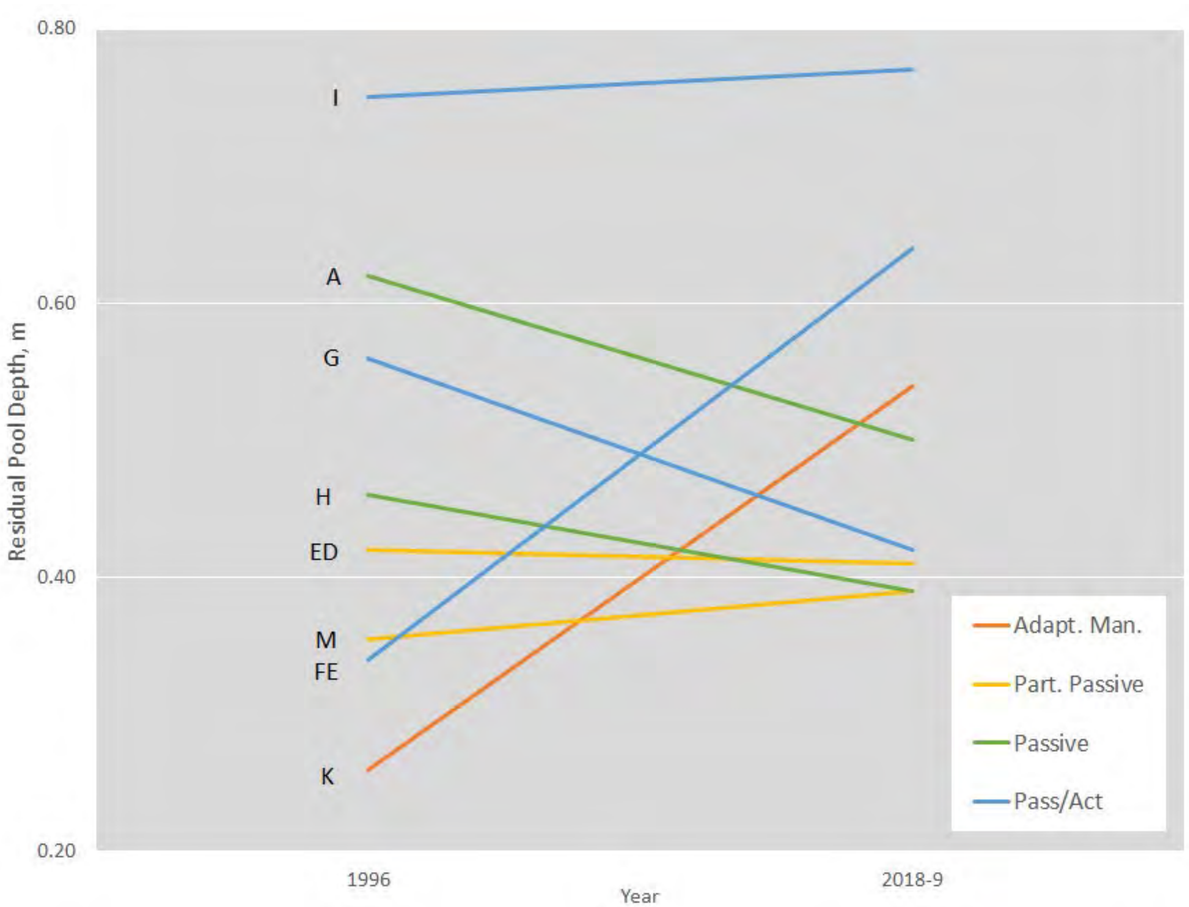

Figure 11. Changes in residual pool depth

Objective 2b: Channel width

Greenline-to-greenline (GL-GL) widths in absolute terms decreased from 1989 to 2017 across all management classes (Fig. 12). Comparing normalized-by-percent differences, the percent change in GL-GL widths also differed significantly among classes (Fig. 13). GL-GL widths narrowed more in Class 3 and 4 (-36% and -38%, respectively) than Class 2 (-24%), and Class 2 narrowed more than Class 1 (-9%). There was no significant difference between Class 3 and 4.

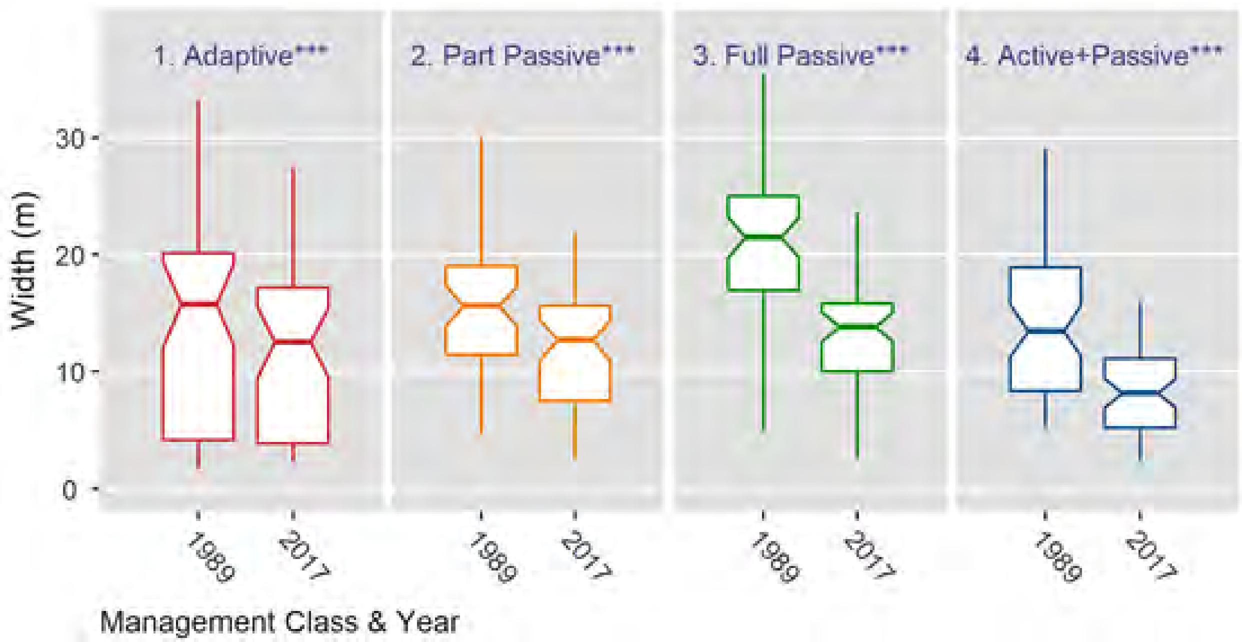

Figure 12. Absolute change in greenline widths 1989-2018 by management class. *** indicates (p<0.001)

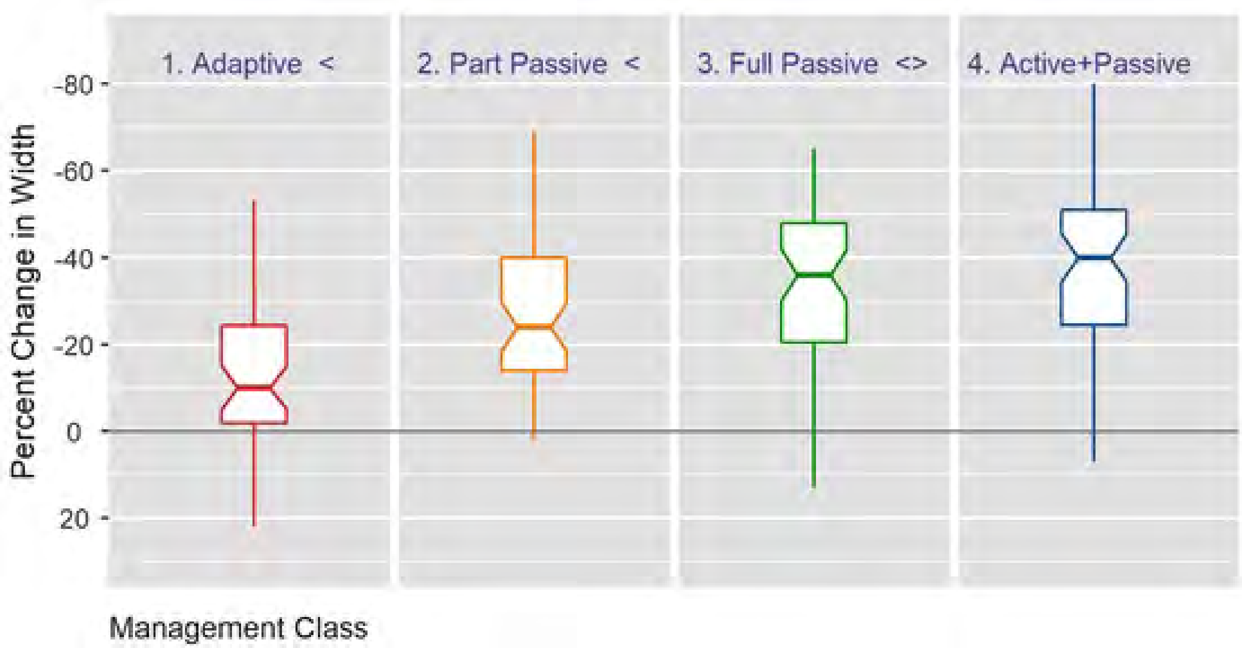

Figure 13. Percent change in greenline widths 1989-2018 by management class. "<" indicates significant difference (p<0.001) less than adjacent class and "<>" indicates no significant difference (p>0.1)

Objective 2c: Channel complexity

A higher number of units/km indicates more complex and therefore better aquatic habitat. Five reaches showed an increase in habitat units/km from 1996 to 2018-19, one (M) showed a decrease, and two (K, H) showed little change (Fig. 14). The reaches showing an increase in complexity are in passive and passive + active management, with one (ED) in partial passive restoration. Therefore, habitat units/km appears to have a positive response to passive and passive + active restoration management.

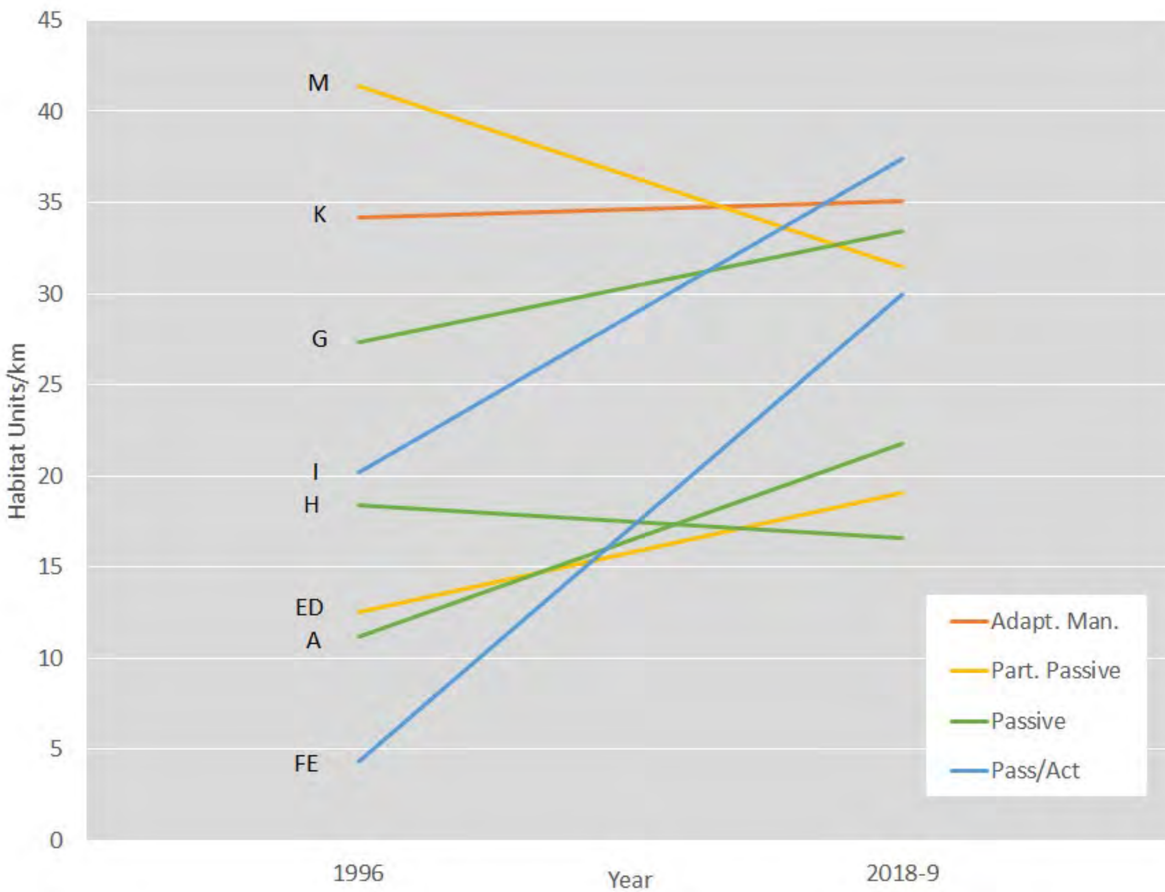

Figure 14. Changes in habitat units per km.

Another indicator of channel complexity is large wood loading in the channel. The main mechanism for increasing LWD input to the channel is increasing tree cover along the channel. Any planting or change in floodplain management to favor trees would require decades for the trees to grow to maturity and begin to contribute LWD to the channel. Passive restoration and active planting of trees on the MFJD floodplains since the 1990s has not increased trees because of heavy browsing of young woody plants by elk and deer, and planted trees are not mature enough or close enough to the channel to contribute LWD. Most reaches showed a decrease in LWD from 1996 to 2018-19 (Fig. 15). Reach M, omitted from the figure because it could not be displayed at the scale of the graph, shows a dramatic decline from 158.8 pieces in 1996 to 32 pieces in 2018-19. The two passive + active restoration reaches, I and FE, showed a clear increase in LWD. The LWD results indicate that passive restoration has not been effective in increasing LWD at the timescale monitored in this project. The more recent active restoration projects, in which woody plantings were protected with deer-elk exclosure fencing, are more likely to produce an increase in LWD, over several decades. The only increases in LWD are due to direct addition of LWD to the channel in active restoration projects.

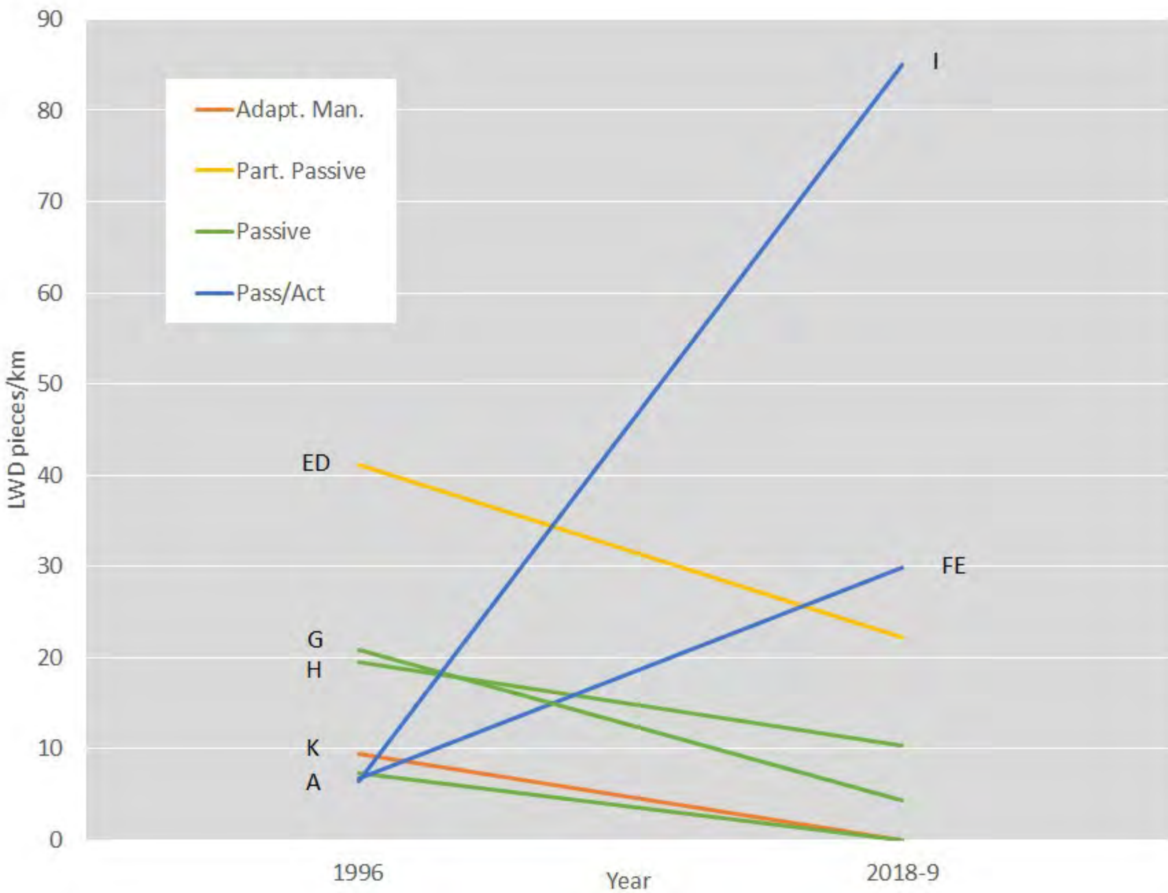

Figure 15. Changes in large woody debris. Reach M is not included.

Another indicator of channel complexity is depth variability (standardized deviation of normalized bed elevation). Five reaches were measured for depth. Three reaches (passive + active, passive and partial passive) increased in depth variability, and two reaches (passive + active) decreased slightly. There is no evident effect of management on depth variability, but the small sample size and limited measurements across the five management classes limit the interpretability of these results.

Objective 2d: Bed material

Gravel counts were completed at 22 sites. The mean gravel size decreased at all sites, indicating that the overall gravel population is getting somewhat smaller. The 95th percentile decreased at most sites. It is not possible to draw a direct inference about changes in the percentage of fines from this result. We also compared the maximum size found at each site, usually a boulder. The maximum size increased at about half the sites and decreased at about half the sites. Most of these boulders probably are not transported by water flows but may be transported downstream in the channel by sediment-rich, high-density flow, such as a debris flow. An increase in these very large boulders could be due to either transport in or a new boulder being transported in, or a new boulder introduced through mass wasting from adjacent hillslopes. A decrease in the maximum boulder size could be due to a large boulder being transported out of the reach, or to a stable boulder being reduced in size by mechanical erosion during high flow events. Overall, these results indicate that substantial gravel transport occurred between 1996 and 2018-19. There is no apparent effect of management.

Summary of Analyses

For vegetation, clear change was observed in the greenline vegetation along the channel's edge, but floodplain vegetation was relatively stable. From 1996 to 2018, greenline community assemblages shifted significantly across all management classes from mesic grasses and a mix of wetland-upland species toward deep-rooted sedges and rushes, hydric grasses and forbs, wetland facultative or obligate species. Wetland species indices and greenline stability indices increased. There were no differences in the direction or degree of change among the management classes. Shrub cover measured in belt transects generally decreased, significantly in class 2-3, due to the shifting of the transect location as the greenline expanded toward the water edge and away from mature shrubs at higher bank positions. Vegetation mapping of the whole floodplain showed no significant changes over time or by management class, but results suggested an increase in the area of mixed riparian woodlands across all classes except the adaptive grazing management class. The number of gravel bar polygons decreased in all classes, suggesting revegetation of gravel bars as grazing pressure decreased.

For channel geomorphology, the clearest changes were 1) channel narrowing (GL-GL width) and 2) some improvement in complexity. For channel narrowing, passive and passive + active reaches narrowed more than partial passive which narrowed more than adaptive grazing reaches; For complexity, habitat units/km increased in most passive and passive + active reaches, with mixed results in other management classes. Also, LW loading increased in passive + active restoration reaches where LW was added. Depth variability, another indicator of complexity, showed no consistent change over time by management class. No improvement was evident in pool quality indicators (% length in pools, residual pool depth).

Interpretation of Findings

Overview

The overarching goal of restoration is to restore natural processes within a system. A key principle in integrating and understanding our results is that change in natural systems occurs at different rates and changes in one component of the system may be dependent on changes in another part of the system. Therefore, we would expect to see clear responses in certain system components, but other system components may lag given slower rates of response or their dependence on other changes occurring first.

Another key principle in interpreting our results is that change in certain components may be occurring but may not have yet reached a level of detection to be deemed significant. Furthermore, our ability to make inferences was also limited by constraints imposed by the small sample size and sampling design used in the 1990s surveys. Finally, another principle, particular to interpreting the differences between passive versus passive + active restoration, is that active restoration practices evolved as restoration practitioners learned from initial actions. As a result, there may be a range of action effectiveness, and initial active restoration projects may not be as effective as later actions that were informed by initial shortcomings. For instance, floodplain plantings in the 2000s were largely unsuccessful given that the impact of browsing pressure from wild ungulate populations was underestimated. Subsequently, exclosures have been added to existing and later floodplain plantings, thus creating a lag in the effectiveness of these actions. All of these factors add "noise" to the data and may reduce our ability to detect statistically significant responses over time or to management.

Greenline vegetation and Greenline-to-Greenline Channel Widths

Greenline vegetation is the system component that would be expected to be affected most directly by cattle grazing, the altering agent that is the focus of passive restoration. Greenline vegetation also consists of species that can expand quickly leading to a relatively rapid response. Our results indeed showed a strong directional change in greenline species composition away from mesic grasses and forbs toward hydric grasses, forbs and sedges/rushes better adapted to the river edge environment, in particular, *Carex nudata* (torrent sedge). Similar directional changes in greenline species composition were seen across classes, but our adaptive grazing management class consisted of only one site limiting our ability to make inferences about this class.

Changes in greenline-to-greenline (GL-GL) width were consistent with the changes seen in greenline species composition. Following passive restoration, species that are susceptible to grazing disturbance but are well-adapted to fluvial disturbance with stabilizing root systems are able to colonize gravel bars and bank bases, expand towards the water's edge, and stabilize these edges, narrowing GL-GL width. *Carex nudata*, in particular, may accelerate this process. *C. nudata* establishes by water-carried seeds deposited along the edge of the low-flow summer channel, thus "leap-frogging" many other species in stabilizing the leading edge of any open areas and facilitating further colonization and infilling (Fig. 16).

Figure 16. Imagery from 1989 (false color, red indicates vegetation) and 2017 including greenline site, GL_05 (star). The images illustrate the advancing of the greenline across gravel bars as well as at the base of banks (e.g. the north bank just above the star where *C. nudata* has established at the bank base). The bank north of the star also illustrates how the mature shrubs established on the floodplain are at a greater distance from the advancing greenline in 2017, but shrub coverage has increased across the floodplain at this site. The large gravel bar at the left of the 1989 photo also illustrates how younger shrubs have established at elevations above and behind the advancing greenline by 2017.

The reported decline in shrub cover is explained by the greenline-to-greenline channel narrowing catalyzed by this colonization of gravel bars and bank bases, moving the greenline away from higher surfaces and closer to the water's edge. As a result, mature shrubs measured in the 1996 survey are now farther away from the advancing greenline and thus cover less of the 2018 belt transect which has moved with the greenline (Fig. 16). The canopy cover results are consistent with this explanation: the densiometer mirror method captures foliar coverage at greater distances from the greenline (beyond the belt transect), and canopy cover did not decline across any classes. As surfaces closer to the water's edge are revegetated, succession should progress with the establishment of new willows and other shrubs (Fig 16).

While all classes showed changes in both greenline species composition and GL-GL widths, the GL-GL width analysis showed differences in response by management class. Classes 3 and 4 (full passive with or without active) narrowed the most, class 2 (partial passive) narrowed the next most, and class 1 (adaptive) narrowed the least. In contrast, there was no difference among classes in GL species composition change. By using historic aerial imagery, the GL-GL analysis was not constrained by the 1990s field sampling design. Given its much larger sample size and balanced distribution of samples, the GL-GL width analysis is likely more representative of the relative scale of changes across classes. Taken together, the GL species composition and GL-GL width analyses point to changes in streamside vegetation moving in a similar direction across the landscape - suggesting that private ranchers have also adopted some measures that have facilitated changes consistent with restoration goals -- but that these changes are greater with implementation of partial and full passive restoration.

Returning to our conceptual framework of differing rates of response and dependencies among system components, greenline species composition may be the component most directly impacted by grazing and also able to respond rapidly to its modification. Changes in species composition can lead to relatively rapid changes in GL-GL widths as stabilizing species colonize river margins lacking vegetation. The mixed results among other geomorphological metrics -- while not always conforming to our initial hypotheses -- is understandable given that in the context of passive restoration, in-stream geomorphological change may proceed at a slower rate and may follow behind, dependent upon these initial system-wide changes in greenline vegetation and narrowing.

Floodplain vegetation

Collectively, the floodplain vegetation results most strongly show that from 1996 to 2018 woody vegetation tended to increase, and gravel bars became vegetated, across floodplain management and ownership types, possibly as a result of collaborative management influences across management classes. While we had few statistically significant results to separate out the effects of different active and/or passive restoration treatments, we caution that 1) sample size for individual treatments may be too small to detect significant change, and that the patterns in the data, while statistically insignificant, may be meaningful and worth longer-term monitoring. 2) With that, we believe that we may still be in a lag period with respect to vegetation recovery, whereby individual seedlings are beginning to revegetate the floodplain but they are not large enough, or at high enough densities yet to take up a lot of 'space' in the floodplain map. As these seedlings mature, we expect increases in canopy cover, infilling, and larger representation on the floodplain.

Geomorphology and aquatic habitat

The overall results for Objective 2 on channel morphology, complexity and bed material show a clear effect by management class for residual pool depth, GL-GL channel width, habitat units per km, and LWD loading. Reaches in passive + active management showed the most improvement, but passive only reaches also show improvement in GL-GL channel width and habitat units per km. These responses probably reflect the positive effects of increased streamside herbaceous vegetation due to passive restoration, plus the direct addition of LWD and digging of pools in active restoration projects.

For the other metrics of channel morphology, complexity and bed materials, there is no clear effect by management class, for several possible reasons. We expected that % pools, pool depth, and depth variability would increase in passive restoration sites through the effects of increased vegetation on the banks and floodplain which would change hydraulics of flows, resulting in scour, pool formation, deepening, etc. These geomorphic effects were not observed. We also expected that passive or passive + active management would lead to increased LWD input to the channel, but due to suppression of woody vegetation by heavy wild ungulate impacts, no response is evident yet. In addition, some of these geomorphic and habitat responses may require a longer recovery time following the management change. In addition, for most characteristics there was a wide, non-systematic range in starting conditions (1996) across the reaches. This indicates that there are local factors, such as valley constraint or long term land use history, that are affecting the conditions and trajectory of change of each reach.

While passive + active management shows the greatest increase in habitat units/km, the increase seen across most passive restoration surveys is consistent with the greenline species composition changes and may also be an example of the importance of local factors. Complementary research by Goslin and McDowell in the MFJDR has found that *C. nudata* is enhancing geomorphic complexity in the system. The most apparent *C. nudata* effect is the development of *C. nudata* islands which result in multi-threaded channel segments, a process that could lead to new habitat units. All passive restoration surveys showed increases in habitat units/km except the two surveys that are at (H) or above (M) the upper limit of *C. nudata* occurrence.

Integrating vegetation and geomorphology: implications for restoration

In the case of the MFJDR, greenline vegetation responded strongly to passive restoration with no added benefit apparent from passive + active restoration, suggesting that greenline plantings were largely ineffective, whereas passive restoration, in and of itself, yielded system-wide changes in greenline species and channel narrowing. In other cases, passive + active restoration may have positive effects that passive restoration does not or may outperform passive restoration alone. In the MFJDR, the active placement of wood into streams “jump-started” an otherwise slow, long-term process, yielding increases in LWD in active treatments that were not seen in passive only. For habitat complexity, passive + active outperformed passive alone, but both increased, suggesting that the active placement of in-stream wood and the passive-induced expansion of greenline vegetation along with shifts in species composition (especially, *C. nudata*), were both contributing to enhanced complexity.

DISCUSSION

Long-term monitoring is critical because changes to ecosystem dynamics, vegetation, and geomorphology in response to restoration can be slow, often taking more than a decade. Rivers are disturbance-driven systems, and their condition varies naturally over time as a function both of droughts, large floods, wildfires, etc., and the vegetation that recolonizes following disturbance. In particular, *C. nudata* (torrent sedge) may play a key functional role, and has expanded dramatically in the MFJDR. A disturbance-adapted species, *C. nudata*'s dense root system allows it to colonize the edge of formerly bare channel bed and bars, facilitating the expansion of other important and previously absent riparian species behind it, and altering patterns of erosion and deposition across the river's bed and banks, enhancing channel complexity (the spatial variation in depth and velocity that produce diverse microhabitats).

On this project we saw early positive benefits of restoration on greenline vegetation and some geomorphic characteristics. These observed changes in vegetation have important implications for channel morphology and physical in-stream fish habitat. Vegetation on the bed and banks redirects water flow in more complex patterns, causing scour in places and deposition in others. Conversion of largely bare channel edges to a continuous greenline of vegetation enhances movement of species, organic matter and nutrients between the terrestrial and aquatic environments. Fish habitat is also improved by increased overhanging vegetation and channel complexity. Although resource managers have been able to observe, and in some cases measure, these changes sporadically over the years, a comprehensive and quantitative assessment helps complete the picture of restoration's long-term ecological effects. While this study describes the impacts of the first two decades of riparian restoration, we recommend continued monitoring to assess longer-term patterns.

Our findings imply that in future restoration projects, the role of passive restoration should be explicitly identified and monitored. Restoration strategies should consider which riparian vegetation species might respond, and which might not, as well as the implications of that response. The response of vegetation through passive restoration should be used as a restoration tool. Active restoration of riparian vegetation (planting) also can be important, but it should be planned in concert with response to passive restoration. In addition, our results showed that passive + active restoration (including instream habitat restoration) has positive effects, sometimes outperforming passive restoration alone. We had no sites with active restoration only, so we could not directly compare the effectiveness of active and passive restoration.

Another implication of our findings is that response of riparian and aquatic systems to restoration takes time. While some parts of the system respond quickly (greenline vegetation), other parts respond slowly. For example, growing trees large enough to supply LWD to the stream may take decades. Therefore, our 23-year span of monitoring is an important and valuable step, but monitoring should continue on the MFJDR. In particular, the valuable systematic monitoring under the MFJD IMW program, which started in 2008, should be continued.

Another intriguing finding is that, for some of the metrics assessed, adaptive grazing lands (i.e. private ranches) were also showing movement in a positive direction toward restoration goals even if that movement was not as strong as shifts on lands where restoration was the key priority. This finding suggests that adaptive grazing practices are indeed evolving over time, although our observation is based on a small sample size. Some managers may be making decisions that improve habitat and participating in the conversations and learning process within the watershed. Our findings suggest that whole watershed strategies should not discount the potential contributions private landowners can make and that it is critical to include them in the conversations around restoration. We suggest that this reflects “collaborative management” -- because restoration is visible at some sites, other land managers in the neighborhood change practices, perhaps in subtle ways, that lead to ecological improvements on land without explicit restoration projects. This is a generally unrecognized benefit of passive and active restoration.

The information derived from this project will help restoration practitioners, planners and funders better understand their suite of options, and choose restoration strategies that are most effective for their desired outcomes over the long term, and yield the most benefit relative to cost. The results will not only be useful for more effective restoration planning and implementation within the MFJDR watershed, where additional restoration projects are planned for the future, but will also have relevance and application for management strategies across the Interior Columbia Basin, the Intermountain West, and beyond.

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APPENDICES

Note: The management classifications of different site types in the same area may differ because very short sites (greenline and gravel count) may not contain some of the restoration actions that apply to longer sites (aquatic habitat survey and longitudinal profile reaches).

Table 1. Greenline Sites.

Site ID	Name/Location	Class	Ownership	Notes
GL_01	Big Ck.	3	private ranch, retired	
GL_02	Galena Tailings (rm 22)	3	USFS – no pasture allotment	
GL_03	Jungle Ck.	2	USFS	
GL_04	Big Boulder Ck.	3	Dunstan C.A. (TNC -> CTWSRO)	FP plantings distant
GL_05	Coyote Bluff (Butte-Tincup Ck.)	4	Oxbow C.A. (CTWSRO)	FP plantings <5m; possibly GL plantings
GL_06	Scum Pool (Deerhorn Ck.)	2	USFS	
GL_07	580 Pool (Forrest C.A.)	4	Forrest C.A. (CTWSRO)	FP plantings <5m; possibly GL plantings
GL_08	Mill Ck.	1	private ranch, active	
GL_09	Hwy. 7 (Crawford Ck.)	2	USFS	

Notes: C.A. = Conservation Area. CTWSRO = Confederated Tribes of the Warm Springs. TNC = The Nature Conservancy. USFS = U.S. Forest Service

Table 2. Aquatic habitat survey Reaches

Site ID	Reach name	Class	Ownership	Reach length, km (2018-19)
A	Big Ck.	3	private ranch, retired	2.1
D	Jungle Ck.	2	USFS	1.6
FE	Dunstan C.A.	4	TNC, CTWSRO	1.8
G	Oxbow C.A. (Beaver-Butte Ck.)	4	CTWSRO	1.4
H	Deerhorn Ck.	3	USFS	1.7
I	Forrest C.A. (Caribou-Vincent Ck.)	4	CTWSRO	1.2
K	Mill Ck.	1	private ranch, active	1.5
M	Canyon (Crawford Ck.)	2	USFS	1.8

Table 3. Longitudinal Profile Reaches

Site ID	Reach name	Class	Ownership	Reach length, km (2018-19)
ED	Jungle-Camp Ck.	2	USFS	1.06
FE	Dunstan C.A.	4	TNC, CTWSRO	1.56
G	Oxbow C.A. (Butte-Tincup Ck.)	3	CTWSRO	.57
H	Windlass Ck.	3	USFS	.36
I	Forrest C.A. (Caribou-Vincent Ck.)	4	CTWSRO	.58

Table 4. Gravel Counts

Site ID	Reach name	Class	Ownership
ED09	Jungle-Camp Cr.	2	USFS
ED15	Jungle-Camp Cr.	2	USFS
ED22	Jungle-Camp Cr.	2	USFS
ED26	Jungle-Camp Cr.	2	USFS
ED36	Jungle-Camp Cr.	2	USFS
ED40	Jungle to Camp Cr.	2	USFS
FE02	Dunstan CA	4	TNC, CTWSRO
FE05	Dunstan CA	4	TNC, CTWSRO
FE16	Dunstan CA	4	TNC, CTWSRO
FE25	Dunstan CA	4	TNC, CTWSRO
FE30	Dunstan CA	4	TNC, CTWSRO
FE36	Dunstan CA	4	TNC, CTWSRO
FE47	Dunstan CA	4	TNC, CTWSRO
G06	Oxbow CA (Beaver-Butte Cr)	4	CTWSRO
G21	Oxbow CA (Beaver-Butte Ck)	4	CTWSRO
G28	Oxbow CA (Beaver-Butte Ck)	4	CTWSRO
H02	Windlass Cr.	3	USFS
H04	Windlass Cr.	3	USFS
H08	Windlass Cr.	3	USFS
H10	Windlass Cr.	3	USFS
NM62	Canyon	2	USFS
NM69	Canyon	2	USFS
NM77	Canyon	2	USFS

CHAPTER 6: Planting Efficacy and Groundwater Monitoring on the Middle Fork Oxbow Conservation Area

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ABSTRACT

The Confederated Tribes of the Warm Springs Reservation of Oregon have made significant investments in restoring riparian conditions and monitoring groundwater in the Middle Fork John Day, with many of these efforts focused on the Tribes' Oxbow Conservation Area. To assess planting success, two separate planting efficacy studies were conducted on this property. The original 2012 study enumerated all woody stems in established cross-sections along the riparian, which included recently installed plantings and existing woody stems. A subsequent 2021 study used real-time kinematic positioning equipment to electronically tag 330 installed plantings along the riparian to track survival. Groundwater elevation assessments used data from six wells in proximity to planting locations to evaluate changes in patterns and trends in groundwater levels pre- and post-implementation of restoration actions. The 2012 planting efficacy study showed variation in survival and additional recruitment within monitoring plots, whereas the 2021 study showed little survival of installed plants, with almost a fifth of the plants being lethally browsed by small rodents within the first-year post-installment.

Groundwater elevation analyses showed mixed results, with only some well locations showing improved water elevation post-restoration. Two lessons learned from these monitoring efforts that are potentially easiest and most impactful to address are 1) protection of established plants may result in quicker revegetation of the stream than installing new plants and 2) fine-meshed rodent exclusionary fencing may be a necessary addition to protect newly installed plants from small-animal browse, especially when plants are sparse and immature.

The groundwater elevation analyses highlighted the importance for continuous datasets to monitor water elevation over time as it relates to restoration monitoring. Restoration practitioners are urged to consider well locations during future project installations.

INTRODUCTION

Background

Riparian plantings and maintenance have become a key component of stream restoration projects; however, plantings have reduced success rates when planted within riparian zones with poor soils largely composed of mine tailings. Studies have shown that shade resulting from dense riparian vegetation can greatly reduce stream temperatures (Middle Fork IMW Working Group 2017; D'Souza et al. 2011). As a result of these findings, the Confederated Tribes of Warm Springs, Reservation of the Oregon (CTWSRO) have incorporated plantings as components of stream restoration projects within the Middle Fork John Day River (MFJDR). Though the riparian zones have been planted extensively, with over 20,000 plants being planted within the Oxbow Conservation Area (OCA) phased project alone, monitoring of these plantings has been infrequent.

The OCA Dredge Tailings Restoration Project was aimed at mitigating the impacts of historic dredge mining within the property. There have been five phases executed since 2011, with an additional two phases proposed for the upcoming years. Portions of the OCA have been fenced using 8-foot ungulate exclusion fencing and within these zones saplings and new recruits from established plants have been observed. Extensive resources have been directed towards supplemental plantings by the CTWSRO Native Plant Nursery and subcontractors. Additional attempts have been made to irrigate these plantings in hopes of increasing success. Though plantings have been intensively installed, the OCA riparian area remains sparsely vegetated by woody stems with little canopy cover present.

Raising the groundwater elevation was a fundamental goal of the OCA restoration project to encourage groundwater recharge to the stream and to increase the duration of floodplain inundation. Groundwater recharge can result in decreased stream temperatures (Kaandorp et al. 2019). An increase in the duration of groundwater connection to the floodplain and therefore the riparian area could result in increased productivity of riparian plants where water accessibility is a limiting factor for growth. Pairing the two riparian planting efficacy studies with groundwater elevation data collected pre- and post-restoration at the OCA, we hope to address the following questions:

Planting Efficacy Questions

1) Are riparian plantings successful?
2) Are installed plants recruiting new saplings into the area?

Groundwater Monitoring Questions

3) Was restoration successful at raising the groundwater elevation?

Goals and objectives

The goals of the planting efficacy studies were to 1) monitor plant survival between and across years, and 2) understand which species survive in these areas to better inform future plantings. The 2021 planting efficacy study was also designed to assess 1) survival of small and medium cottonwoods installed, 2) on the effectiveness of mycorrhiza inoculation on planting success, and 3) planting success as a function of distance from stream. However, due to high rates of browse and low survival, the analyses were limited to general survival assessments. The goals of the groundwater monitoring efforts were to gather a robust pre- and post-restoration data set that could be used to detect changes in groundwater elevations and also compare groundwater elevations across MFJDR monitoring locations. In order to achieve these goals, permanent monitoring cross-sections were established for the initial planting efficacy study in order to monitor survival of plants in established plots over time. The subsequent study used Real-Time Kinematic positioning (RTK) to establish the location of each planting installed with 3cm accuracy to assess survival rates of specific plants one- and two-years post-installment. Groundwater elevation data has historically been collected near these planting locations. These data provide a general understanding of how restoration efforts impact groundwater elevations. Future work intends to translate groundwater elevations to water accessibility for plants.

Site Selection

Planting Efficacy

For the initial planting efficacy study, plants were installed in the Granite Boulder Creek and Ruby Creek areas (Figure 1). Phase 2 of the OCA Project included the Granite Boulder Creek planting area which concentrated plantings around the mouth of Granite Boulder, and Phase 3 included the Ruby Creek planting area which spans from above Beaver Creek to upstream of the mouth of Ruby Creek. Hereafter these plantings will be referred to as Phase 2 and Phase 3 respectively when discussing the planting efficacy study at these locations. Phase 2 was planted in 2012, and the Phase 3 riparian area has been planted heavily since 2014, with little success. Because of low planting success in the Phase 3 location, this area was ideal for the subsequent 2021 efficacy study to potentially identify areas where planting efforts should be focused and plant types that survive out-planting in the degraded soils.

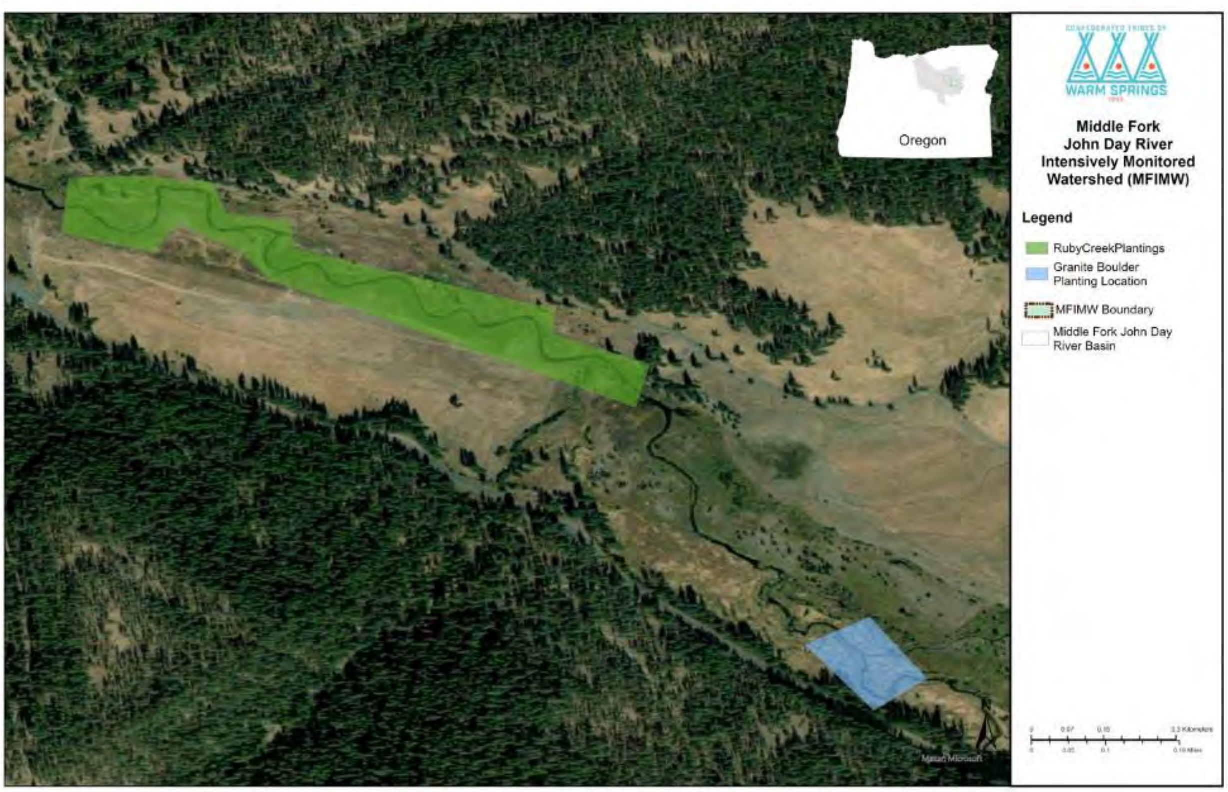

Figure 32. Planting locations for the Ruby Creek area (green) and the Granite Boulder Creek area (blue).

Groundwater Monitoring

The CTWSRO actively monitors and maintains 44 groundwater monitoring wells across three tribally-owned conservation properties on the MFJDR. Fourteen of these wells are situated on the OCA property (Figure 2). Historically, additional wells were located within the OCA property (Figure 3); however, during the implementation of the tailings restoration some of these wells were destroyed. For these analyses, a subset of groundwater elevation data were used from loggers with sufficient data sets to compare pre- and post-restoration groundwater elevation near planting locations. The loggers used and data available for this analysis are listed in Table 1.

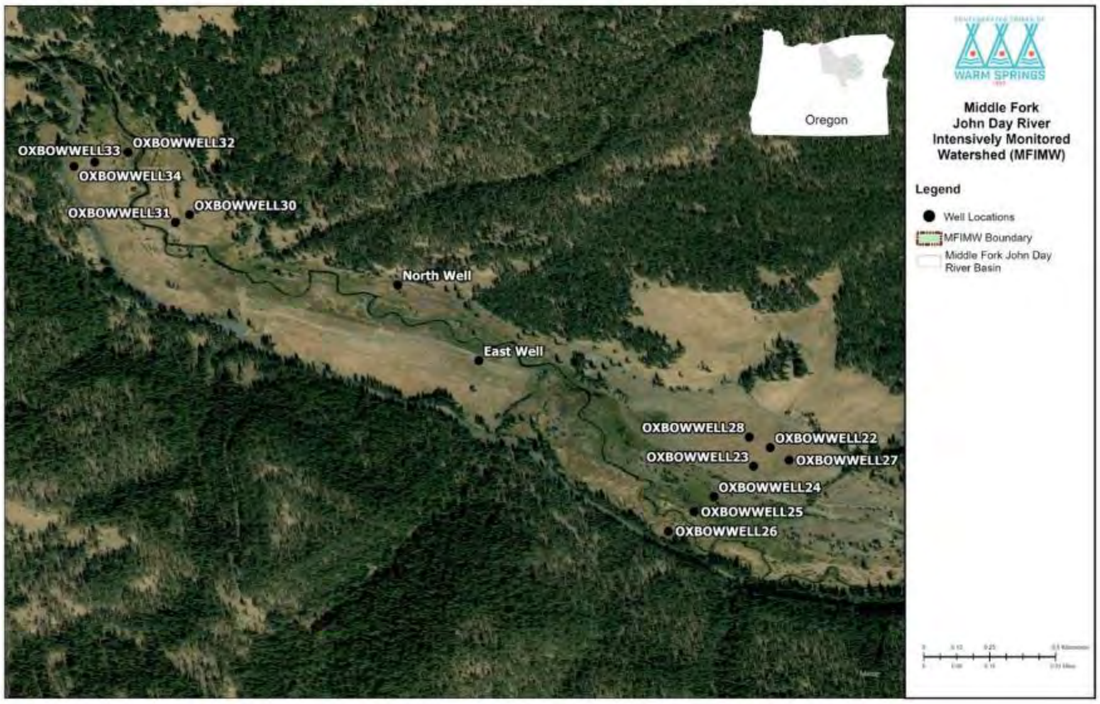

Figure 33. Active groundwater wells on the OCA and the current configuration of the MFJD post-phase-5 restoration.

Figure 34. All historic OCA groundwater well locations and MFJD location pre-restoration implementation.

METHODS

Planting Efficacy

Two planting efficacy studies have been conducted on the OCA. The 2012 planting efficacy study was established for Phases 2 and 3 of the OCA tailings restoration, with Phase 2 located near Granite Boulder and Phase 3 located near Ruby Creek (Figure 1). Planting occurred in July of 2012 at Phase 2, with plants being revisited to assess survival in the fall of 2012, the fall of 2013, and again in the fall of 2016. Only initial baseline counts of plants within each cross-section were recorded, including both planted and previously established plants in these locations. Phase 3 was planted in 2014 and revisited in the fall of 2015 and 2016 to assess plant survival at established cross sections. More data was collected for the Phase 3 cross-sections, including number of plants planted, and whether these plants were cuttings or rooted plants. At each phase, plants were planted in five 30' radius cross-sections along the stream within the riparian area. A cross section was established by placing a piece of rebar in the ground on the bank at bankfull, mounting a measuring tape to the piece of rebar, and then measuring 30' out from this point parallel with the stream. Once 30' was measured, the surveyor walked in an arc away from the bank until they reached the bank again, creating a semi-circle. Cross-sections were numbered one through five (CS-1 through CS-5), upstream to downstream. Because Phase 2 was planted in July of 2012, September 2012 survival represents mortalities over the hot summer months.

A subsequent planting efficacy study was initiated in 2021 for plants installed near Ruby Creek. An auger was used to create holes for planting due to difficulty digging holes by hand in the rock. Plants were installed in three bands parallel to the stream within four plots along the riparian area. Each band contained thirty plants for a total of ninety plants per plot. The four plots consisted of one mixed species control plot, one mycorrhiza treated mixed species plot, one medium-sized cottonwood plot (cottonwoods greater than one-foot in height from the base), and one small-sized cottonwood plot (cottonwoods under one-foot in height from the base). Soil was trucked into the site to assist with establishment due to poor soil condition at the planting location. Each hole was first filled with a layer of soil before the plant was placed in the hole. The remaining space around each plant was filled with the new soil and tightly packed in. Each plant was watered after installation. Previously, physical tags were used to identify plants for survival analysis; however, for this study RTK-GPS equipment was used to establish points at the location of 330 installed plants. This created a digital location tag, which allowed for survival to be attributed to specific plant types along the stream. Because plants can be located with 3cm accuracy using this equipment, each planting was successfully revisited.

For the initial planting efficacy study, baseline rooted plant counts, cuttings, and total plant counts were recorded for each cross section. The survival was assessed by percent of plants surviving between and across years. The 2021 study assessed first year survival of different plant types due to low survival resulting in reduced statistical power for more robust analyses. Plants in this study were also recorded as browsed or non-browsed for this location, as it appeared to be an issue. Only survival to year-one post-installment is discussed as the assessment of year-two survival is ongoing.

Groundwater Monitoring

CTWSRO began collecting well water elevation data in 2008 across several areas of the MFJDR associated with the tribal properties. Loggers were deployed at nine locations within the OCA between 2008 and 2010. The OCA Dredge Tailings Restoration Project was completed in multiple phases from 2011-2016, and six well loggers were maintained both pre- and post-restoration (Table 1). This provides an opportunity to explore potential changes in floodplain water elevation in response to restoration. Well loggers (HOBO U20-001-01-Ti and Solinst 3001 Levellogger) recorded pressure measurements (mbar) at 15-minute to 1 hour intervals. Water pressure predictably changes with water depth after correcting for temperature, and measurements were converted water surface elevation (m) estimates using well water depth and well elevation.

Table 3. Date ranges for well loggers within the Oxbow Conservation Area that were maintained pre- and post-restoration.

Logger_ID	Date of earliest record	Date of latest record	Year of restoration	# days of data
Oxbow-East	2010-06-16	2016-08-15	2014	1,122
Oxbow-North	2010-06-17	2017-06-01	2014	1,862
Oxbow-23	2008-08-29	2017-06-01	2012	2,149
Oxbow-24	2008-05-24	2017-06-01	2012	2,246
Oxbow-25	2008-08-29	2015-10-01	2012	1,484
Oxbow-26	2008-08-29	2017-06-02	2012	1,937

We first summarized water elevation by year and month for each logger to visually explore overall patterns and trends over time. We focused on May-August as this is a critical period for plant establishment and growth. However, water elevation is strongly influenced by discharge in addition to hydrogeomorphic processes, the latter of which are expected to change with restoration. To account for inter-annual differences in discharge when evaluating restoration response, ideally, we would compare pre- and post-restoration water elevation measurements within the Oxbow against a control reach lacking significant changes in hydrogeomorphology (e.g. before-after control-impact design). However, we were not able to identify suitable control loggers that met the criteria of being relatively close in proximity, did not experience restoration or major changes, and covered the same approximate time period. Consequently, we conducted an exploratory analysis that attempts to account for inter-annual variation in discharge. As such, we evaluated pre- and post-restoration relationships between discharge and well water elevation for each logger. With this approach, we hypothesize that if restoration increased water table elevation (i.e., reduced depth to water table), the fitted relationship between discharge and water elevation will have a higher y-value (i.e., water elevation) post- relative to pre-restoration for a given discharge value.

We used daily discharge estimates obtained from a USGS stream gauging station on the mainstem MFJD near Ritter (gauge # 14044000). The gauging station near Camp Creek – closer in proximity to the OCA – only has discharge estimates since 2011 and thus were not sufficient to evaluate pre-restoration relationships. Note that discharge patterns may differ between the OCA and Ritter gauge, especially in summer, due to inputs (e.g., tributaries) and withdraws (e.g., irrigation) between these locations.

Visual examination of scatterplots between discharge and well logger elevation revealed non-linear patterns. We first fitted models predicting water elevation using the natural log of discharge, which fit the data reasonably well. However, stream channel morphology is heterogeneous and increases in discharge can result in relatively large increases in water elevation when flows are contained within the active channel. Proportionally we noted far smaller increases in water elevation after flows exceed bankfull height and spread out on the floodplain. Further complexity in discharge-water elevation relationships may arise as additional channels become active in addition to changes in terrain features. We therefore fitted loess models predicting well water elevation for each logger as a function of discharge and treatment period (i.e., pre/post restoration).

This analysis should be viewed as preliminary and exploratory as time and data constraints prohibited us from fully assessing assumptions and apply appropriate measures to account for potential violations. In particular, random effects to account for variation stemming from annual and temporal autocorrelation are likely needed. Consequently, we did not utilize traditional statistical significance (i.e., *p*-values) to assess potential restoration impacts on water elevation.

RESULTS

Summary of Analyses

Planting Efficacy Study

A total of 361 plants were counted within the established cross-sections ranging from 30 woody stems in CS-1 to 124 wood stems counted in CS-5 within Phase 2 of the OCA restoration area. The majority of plants survived the summer, with the CS-1 having the lowest survival at 73% and CS-5 having 100% survival (Figure 4a). However, the percentage of plants that were still alive during the August 2013 assessment was drastically lower. Between the July 2012 and August 2013 visits, only 20% of the plants were alive at CS-1, with CS-5 having the highest percentage of surviving plants at 56%. There were no obvious trends in survival for plants with regards to their position along the river (Figure 4a). The number of plants counted decreased from the initial planting in 2012 to the August 2013 visit; however, some cross-sections showed increased numbers of woody stems during the 2016 survey (Figure 4b). Baseline number of species declined at all cross-sections, excluding CS-5, which gained three additional species during the August 2013 survey that were not previously identified in the September 2012 survey. This number dropped drastically during the 2016 survey, with only four species identified in this cross-section. On average, there were 6 unique species per cross-section at the time of the first survey, reduced to 4 unique species per cross-section for the 2016 survey (Figure 5).

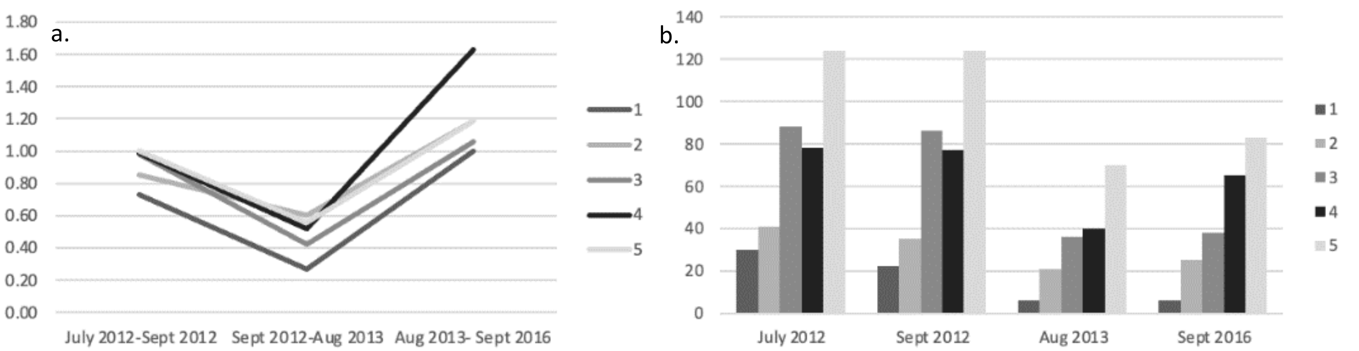

Figure 35. Comparisons of survival between monitoring visits at each cross-section with 1.00 indicating 100% survival from the count of the previous visit (a.) verses actual plant counts (b.) with baseline counts at the time of installation occurring in July 2012 for Phase 2 of the OCA restoration project.

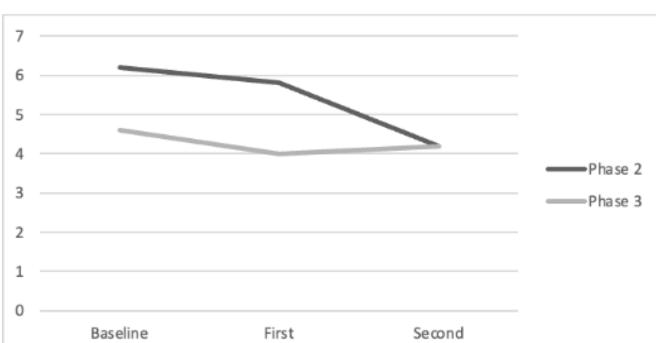

Figure 36. Average plant diversity at the initial baseline assessments and subsequent visits for both phases of planting for the initial efficacy study. Y-axis displays the number of unique plant species counted per cross-section.

Within Phase 3 of the OCA restoration project, rooted plants and cuttings were installed within the five cross sections for a total of 284 rooted plants and 583 cuttings. Baseline total plant counts included naturally established plants. Including the cuttings and rooted plants, 888 plants were counted within the study area during the baseline visit. Significantly more cuttings were installed than rooted plants in all of the cross-sections excluding CS-5 which had 7 more cuttings than root plants installed (rooted: *mean = 57±39*, cuttings: *mean = 117±37*; *p= 0.043)*. No rooted plants were installed in CS-3; however; 103 cuttings were installed (Figure 6). Baseline number of woody stems counted within the cross-sections ranged from 109 at CS-3 to 227 at CS-2 (Figure 6a). There was no apparent difference in survival of rooted plants (Table 2) when compared to cuttings (Table 3) across years. Rooted plants did see recruitment in two out of the five cross-sections when comparing 2013 and 2016 results (Figure 6). For example, in CS-5, rooted plants had low survival from 2014-2015 with only 23% (n = 20) of those plants surviving; however, established plants were able to survive and recruit woody stems that were then counted in 2016 for a survival of 125% (n = 25). A higher percentage of rooted plants survived compared to cuttings from initial installment through the final survey, although not significantly (rooted: *mean = 0.33±0.21*, cuttings: *mean = 0.13±0.06*; *p= 0.062).* The baseline number of species ranged from three to eight unique species per cross-section (Figure 5). Plant diversity decreased in two cross-sections, increased in two cross-sections, and remained consistent in one cross-section between the 2014 and 2016 surveys. CS-1 was the only cross-section with drastic changes in diversity, dropping from 8 to 4 species over time, with all other cross-sections only varying by one species from the baseline survey.

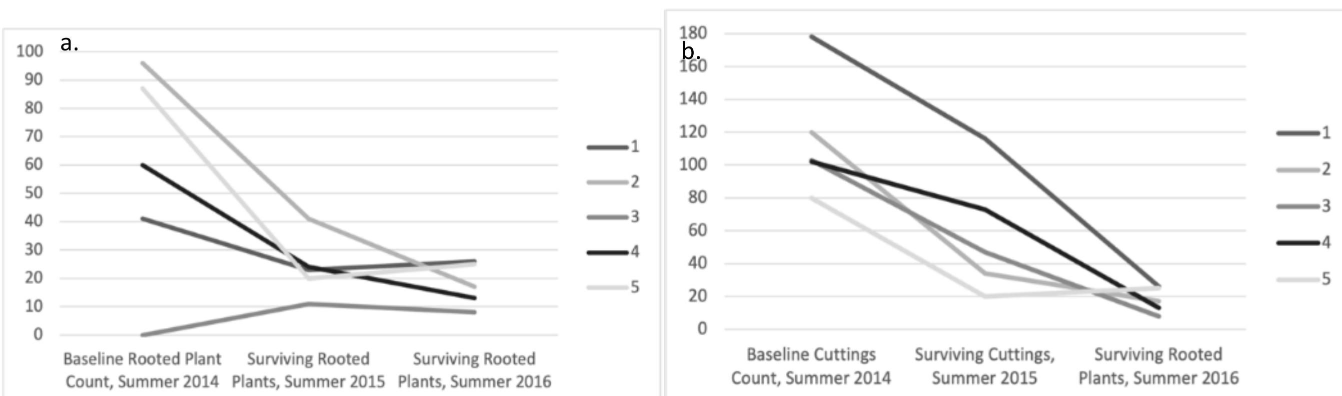

Figure 37. Number of living rooted plants (a) and plant cuttings (b) counted in Phase 3 of the OCA restoration project at each cross-section across years.

Table 4. Percent survival of rooted plants between surveying years located in Phase 3 of the OCA restoration project. No rooted plants were planted in CS-3 resulting in NA's for the cross-section.

Cross-section number	*Summer 2014 - Summer 2015*	*Summer 2014 - Summer 2016*	*Summer 2015 - Summer 2016*
1	56	63	113
2	43	18	41
3	NA	NA	NA
4	40	22	54
5	23	29	125

Table 5. Percent survival of cuttings between surveying years located in Phase 3 of the OCA restoration project.

Cross-section number	*Summer 2014 - Summer 2015*	*Summer 2014 - Summer 2016*	*Summer 2015 - Summer 2016*
1	65	19	29
2	28	12	41
3	46	22	49
4	72	17	23
5	25	6	25

All digitally tagged plants were successfully located and marked as alive or dead at the time of the visit. Preliminary results show that plant survival is low within the planted area. Out of 330 plants installed, only 23 plants (7%) were marked as alive at the one-year post-installation assessment. Nearly 1/5 of the plants installed had been browsed by what was attributed to small rodents. No plant that had been browsed also had living buds, resulting in these plants classified as dead. Chokecherries survived in greater numbers than alders and cottonwoods, making up 83% of the plants that survived (Table 4). Because none of the installed plants were of a mature age, no recruitment was attributed to these plants within the plot, nor was recruitment of newly installed plants observed. Survival will be re-evaluated at the two-year post-installment assessment to assess plant survival and recruitment.

Table 6. Type of plants installed within the monitoring reach with associated planting numbers and survival through the first-year post-installment visit for the 2021 RTK planting efficacy study.

Plant Type	*Total Planted*	*Percent*
Alder	90	2 (n = 2)
Cottonwood	180	1 (n = 2)
Chokecherry	60	32 (n = 19)

Groundwater Monitoring

Figure 1 shows water elevation by month and year for each logger. While it is useful to visualize these water elevation data, it is difficult to interpret potential changes in water elevation pre- and post-restoration given inherit variability in discharge between years. Evaluating differences in the relationship between discharge and water elevation (Figure 2) provides an approach to evaluate potential changes associated with restoration by limiting the influence of inter-annual variability in restoration. Visual examination of these plots reveals mixed responses among loggers. For example, fitted water elevation is greater post-restoration across the range of observed flows for Oxbow-North, Oxbow-25, and Oxbow-26, but in contrast, fitted water elevation is lower post-restoration for any given flow for Oxbow-23 and Oxbow-24.

There is also considerable variability among loggers in the magnitude of differences in water elevation pre- and post-restoration. For example, differences in the fitted water elevation relationship pre- and post-restoration are small and not likely biologically relevant for Oxbow-26, but comparably larger for Oxbow-North, where fitted water elevation ranges from 0.3-0.5 m greater post-restoration compared to pre-restoration across the range of flows. In contrast fitted water elevation is often 0.1-0.25 m lower post-restoration for Oxbow-25.

The relationship between discharge and water elevation for Oxbow-East post-restoration is relatively uniform and does not conform with our other fitted relationships. This anomaly warrants further exploration into potential logger errors or unique site attributes post-restoration, particularly factors that could influence greater water elevation post-restoration such as the establishment of beaver dams.

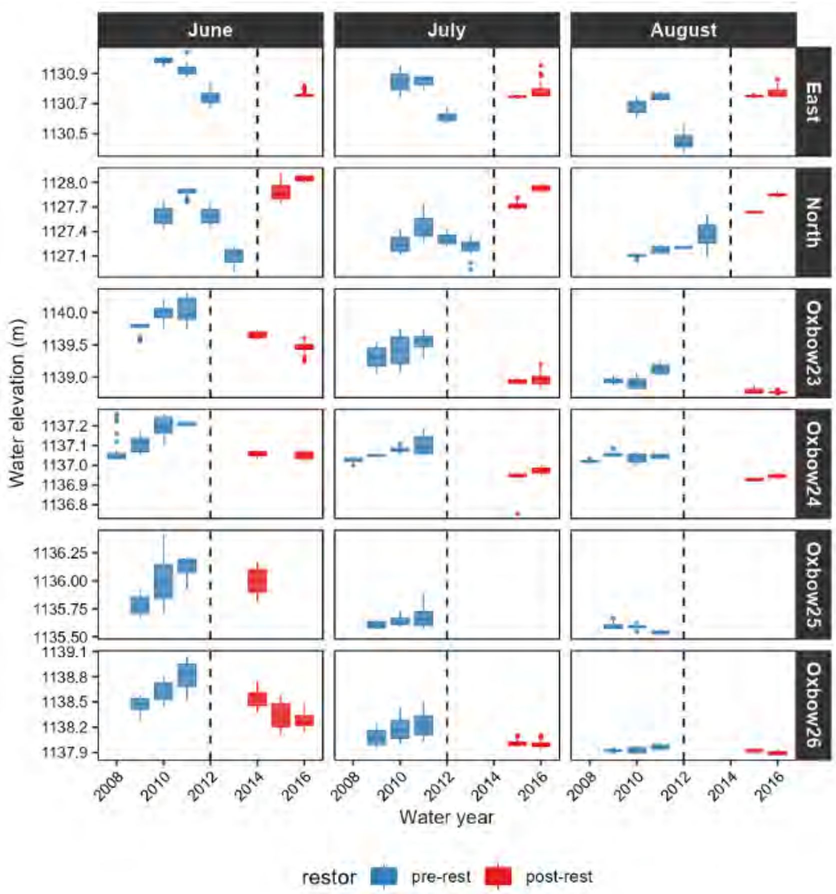

Figure 38. Boxplots of water elevation by month (columns) and year (x-axis) for each logger (rows). Blue fill indicates year-month combinations prior to restoration, whereas red fill indicates post-restoration.

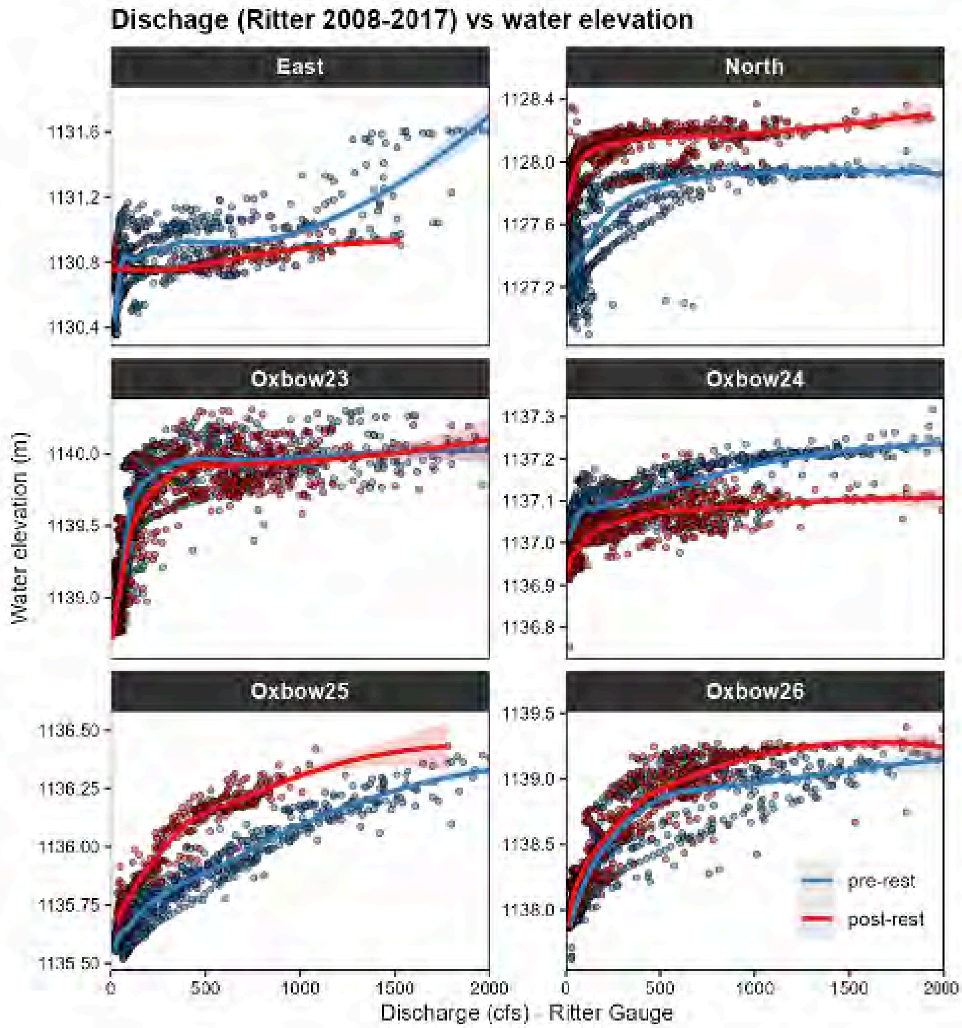

Figure 39. Fitted relationships between discharge at the Ritter gauging station and the water elevation of each logger before (blue) and after (red) restoration. Relationships were fitted using smoothed Loess curves to allow for non-linear and unique patterns arising from differences in geomorphology and channel structure.

DISCUSSION

Results from the 2012 planting efficacy study showed mixed outcomes for plants installed within the OCA. These planting and monitoring efforts paired with the 2021 planting efficacy study highlight that the surviving plants from the 2012 study were likely not installed plants, rather existing plants that were counted in conjunction with the plantings. The 2021 study revealed low establishment success of plants installed within the reach, with only 8% of installed plants surviving at the one-year post-installment revisit. By using RTK equipment to visit each individually installed plant (Figure 9a), we now know that the installed plants are heavily browsed by rodents that can pass through the elk exclusion fencing. Plant heights were recorded two-weeks post-installment. It was noted that over one third of the plants had already experienced browsing. No height was recorded for these plants as they were cut near the base of the stem. Muskrats appear to be responsible for the browse, favoring large cottonwoods to other plants. Over seventy-five percent of the large cottonwoods installed were browsed within the two-week period compared to only 28% of small cottonwoods browsed, and the mixed species plots of alder and chokecherries experiencing the least amount of browsing pressure. The pressure of rodent browsing on installed plants was previously unknown before this assessment and provides a clear direction moving forward. It is evident that large-meshed 8-foot exclusion fencing may not be adequate to protect newly installed plants, especially where riparian woody stems are sparse. It appears they become the target for browsing rodents and pests with larger, more visible plants more easily located. Because the installed plants are single stemmed, browsing frequently resulted in complete take of the stem resulting in plant death.

Figure 40. RTK surveying for planting efficacy is depicted in both images, highlighting the thick matting of Birdsfoot trefoil (a. and b.) and the browsed vegetation (b.) within the monitoring area.

In order to protect installed plants, smaller meshed exclusion fencing must be placed around plants that are installed in zones away from established growth. Plantings near other woody stems were not browsed as often as plantings away from established vegetation. Though these plants were not browsed as often, a number of them were still dead and sometimes overtaken when planted adjacent to thriving willows as the branches spread from the base. This illuminates the caution that must be taken when installing immature plants next to established vegetation that spreads quickly. Three of the surviving chokecherries (*Prunus virginiana*) were found buried by birdsfoot trefoil (*Lotus corniculatus*) and exhibited minimal growth. Birdsfoot trefoil is a non-native, invasive species to the United States (Center for Invasive Species and Ecosystem Health 2023). Birdsfoot trefoil can be pasture grown for cattle as an alternative to alfalfa; however, once it is established in disturbed areas it creates a mat-like layer, which prevents growth of other plants (Figure 9b) (Minnesota Department of Natural Resources 2023). The matting of birdsfoot trefoil may provide a make-shift drip system to the underlying surviving plants when dew collects in the mornings leading to increased survival through the summer. The likelihood of the chokecherries growing underneath emerging through the mat of birdsfoot trefoil without consistent removal is unlikely.

It is recommended that continued planting efforts within mine tailings utilize RTK equipment to track survival of plantings as tags hung on plants in the 2012 efficacy study were difficult to relocate, which caused confusion differentiating survival of installed plants versus previously established plants. The increase in cuttings counted in the final survival assessment for Phase 3 plantings is attributed to plants either being misidentified as dead during first survival assessment or naturally recruited plants being included in the counts. When RTK equipment is not available, general woody stem density counts may be the best method for tracking revegetation. It is important to note that plants counted in subsequent assessments may not be surviving installed plants and are likely naturally recruited plants.

Additionally, it appeared that removing birdsfoot trefoil would be required to have success in plantings within the OCA where the ground is heavily matted. After further discussion with CTWSRO nursery staff, installing plants with the stems that protrude above ground cover may be adequate to promote increased plant survival. This would allow for capitalizing on the benefits of water retention by birdsfoot trefoil without the setbacks of plants struggling to penetrate through the matting. However, as mentioned previously, taller plants have the potential to be browsed almost immediately post-installment; therefore, further protection of these plants would be required.

Initially it appeared that restoration had potentially negative effects on groundwater elevation at Oxbow wells 24 and 25. Upon further investigation into historic channel configuration, lower groundwater elevations at wells 23 and 24 are consistent with the filling of the north channel in Phase 2 which historically ran alongside these wells. With flows now diverted into the south channel, this may have contributed to the increased groundwater elevation at the Oxbow 25 and Oxbow 26 wells. Pre-restoration, more wells were maintained as shown in Figure 1; however, due to restoration actions, a number of these wells were decommissioned when they were compromised during construction. Data gaps within the groundwater elevation dataset created difficulties when making comparisons of groundwater elevation levels pre- and post- restoration. With the destruction of the South and Center wells during construction, it is unclear how restoration affected groundwater elevation in the areas where wells were destroyed. Increased water elevation levels at the North well and more consistent ground water elevation levels throughout the summer months at the East well prove promising for increased groundwater elevations along the OCA stream reach. Investigation into re-establishing the decommissioned wells on the OCA found that establishing new wells is financially burdensome with current efforts to recommission well sites being hindered by costs. We recommend restoration practitioners carefully consider well locations in designs so that monitoring of groundwater levels can continue. This would allow for comparisons in groundwater elevation pre- and post-implementation.

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CHAPTER 7: Middle Fork John Day IMW Macroinvertebrate Community Analysis Phase 2 Report

Prepared for the North Fork John Day Watershed Council
Attention: Javan Bailey, Restoration Project Manager
Date: 1 December 2023

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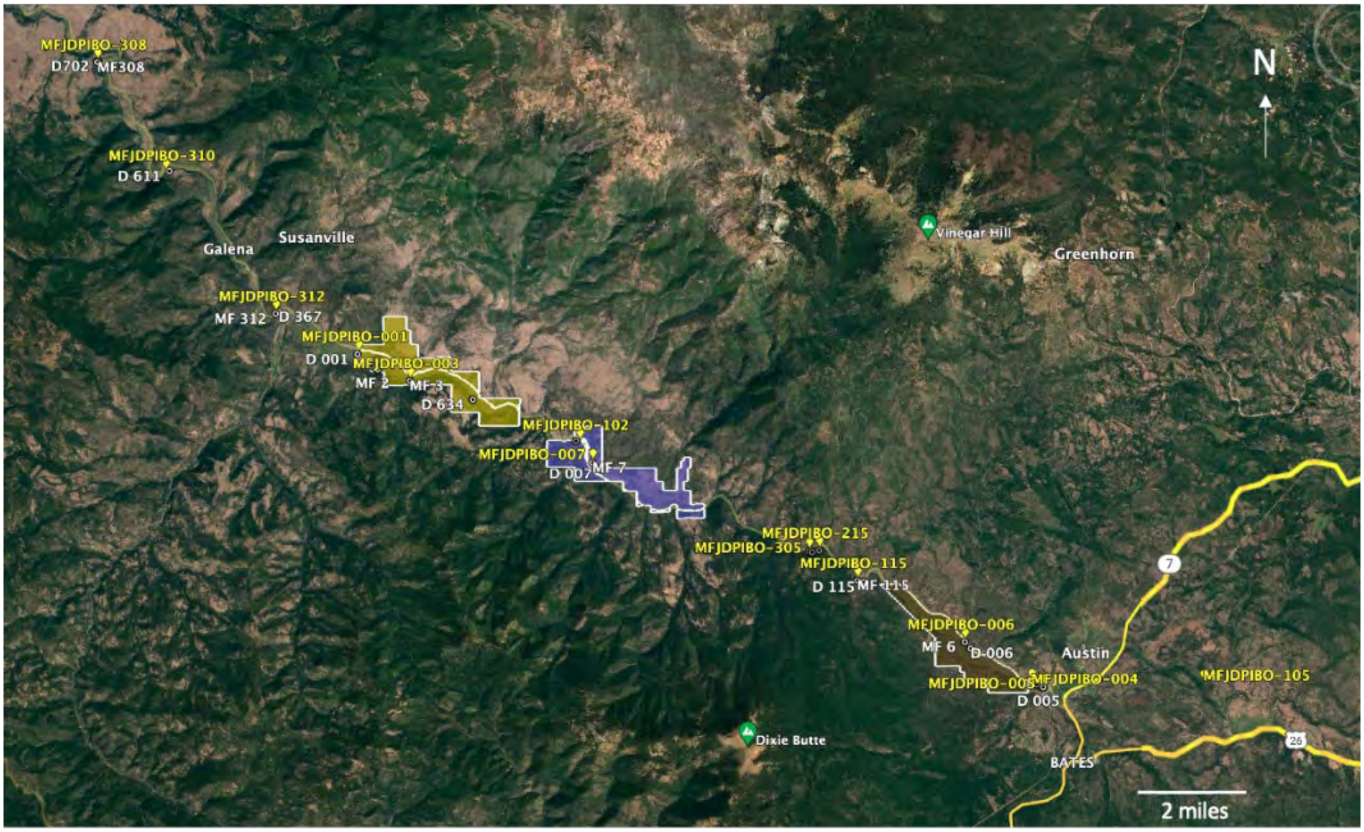
ABSTRACT

These analyses focused on detecting long-term trends in drift and benthic macroinvertebrate data, followed by a “before-after” restoration analysis at each site. Little agreement in trends or changes occurred between drift and benthic results at co-located sites in the MFJDR, and no consistent relationship was seen between restoration intensity and macroinvertebrate community response. A general lack of consistent temporal trends or consistent pre/post-restoration changes in benthic and drift communities suggests that ecological conditions have remained largely unchanged in the MFIMW over the 2010-2022 monitoring period. The drift data exhibit some trends and pre- versus post-restoration changes, but the limited utility of drift data is discussed. Benthic data indicate positive post-restoration changes in ecological conditions at only two of 10 sites (MF-2, MF-3); one of these (MF-2) is co-located with a drift site that also showed relatively consistent evidence of improved conditions (D 003). Continued monitoring of the benthic community at both MF-2 and MF-3 should reveal whether these apparent ecological changes will persist as a result of restoration efforts or if they are related to other drivers and will continue to vary. We recommend discontinuing drift sampling and adding physical habitat assessment and continuous temperature monitoring to the benthic sampling to produce a more robust data set to facilitate detection of potential drivers of observed ecological change over time.

BACKGROUND

Restoration projects in the Middle Fork John Day River Intensively Monitored Watershed (MFIMW) have been implemented since the mid-1990s. The overarching goal is to improve degraded instream and riparian conditions and enhance ecological functions to benefit native fish and improve the ecological integrity of the watershed. Habitat changes resulting from restoration are also expected to improve conditions for macroinvertebrate assemblages in the Middle Fork John Day River (MFJDR). For example, reductions in sediment load and substrate embeddedness may be accompanied by decreased abundance of burrowing organisms and increases in clingers; improved riparian conditions can enhance populations of organisms that feed as shredders; increases in canopy cover and stream flow can support communities with more temperature-sensitive taxa; and increased habitat stability may support organisms that are more sensitive to disturbance and/or have a longer egg-to-adult development time.

Because macroinvertebrate communities are potentially useful indicators of the effects of watershed restoration activities on ecological conditions, both benthic and drift macroinvertebrate sampling has occurred in the MFIMW since 2010 (Figure 1). This report summarizes the results of analyzing the MFIMW benthic and drift macroinvertebrate data with an emphasis on the Phase II analyses. The Phase I analyses are presented in detail in Searles Mazzacano and Cole (2023; Appendix B).


MFJDPIBO-308
D702
MF308
MFJDPIBO-310
D 611
Galena
Susanville
MFJDPIBO-312
MF 312
D 367
MFJDPIBO-001
D 001
MFJDPIBO-003
MF 2
MF 3
D 634
MFJDPIBO-102
MFJDPIBO-007
D 007
MF 7
Vinegar Hill
Greenhorn
MFJDPIBO-215
MFJDPIBO-305
MFJDPIBO-115
D 115
MF 115
MFJDPIBO-006
MF 6
D-006
Austin
MFJDPIBO-005
MFJDPIBO-004
D 005
MFJDPIBO-105
Dixie Butte
BATES
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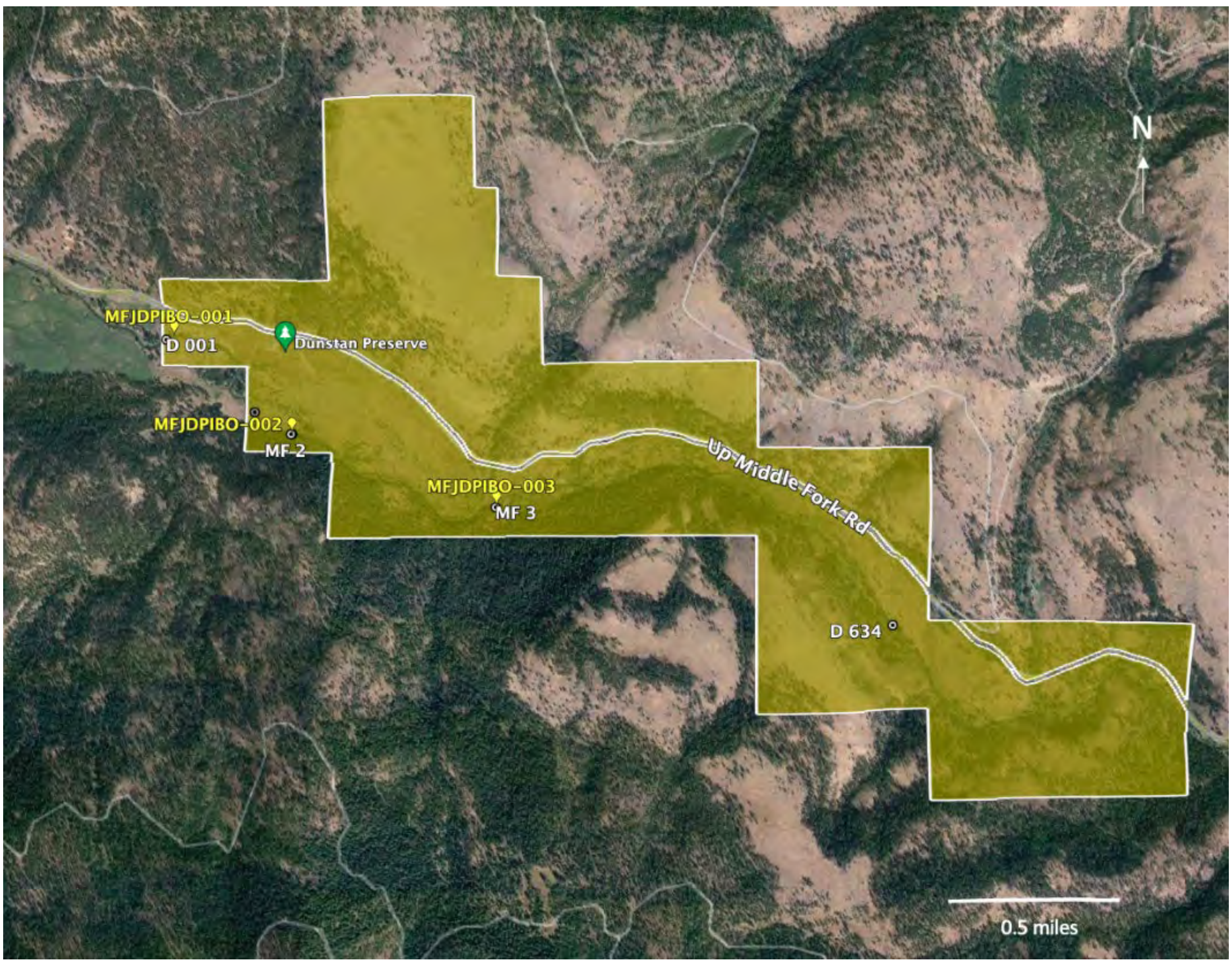
MFJDPIBO-001
D 001
Dunstan Preserve
MFJDPIBO-002
MF 2
MFJDPIBO-003
MF 3
Up Middle Fork Rd
D 634
N
0.5 miles

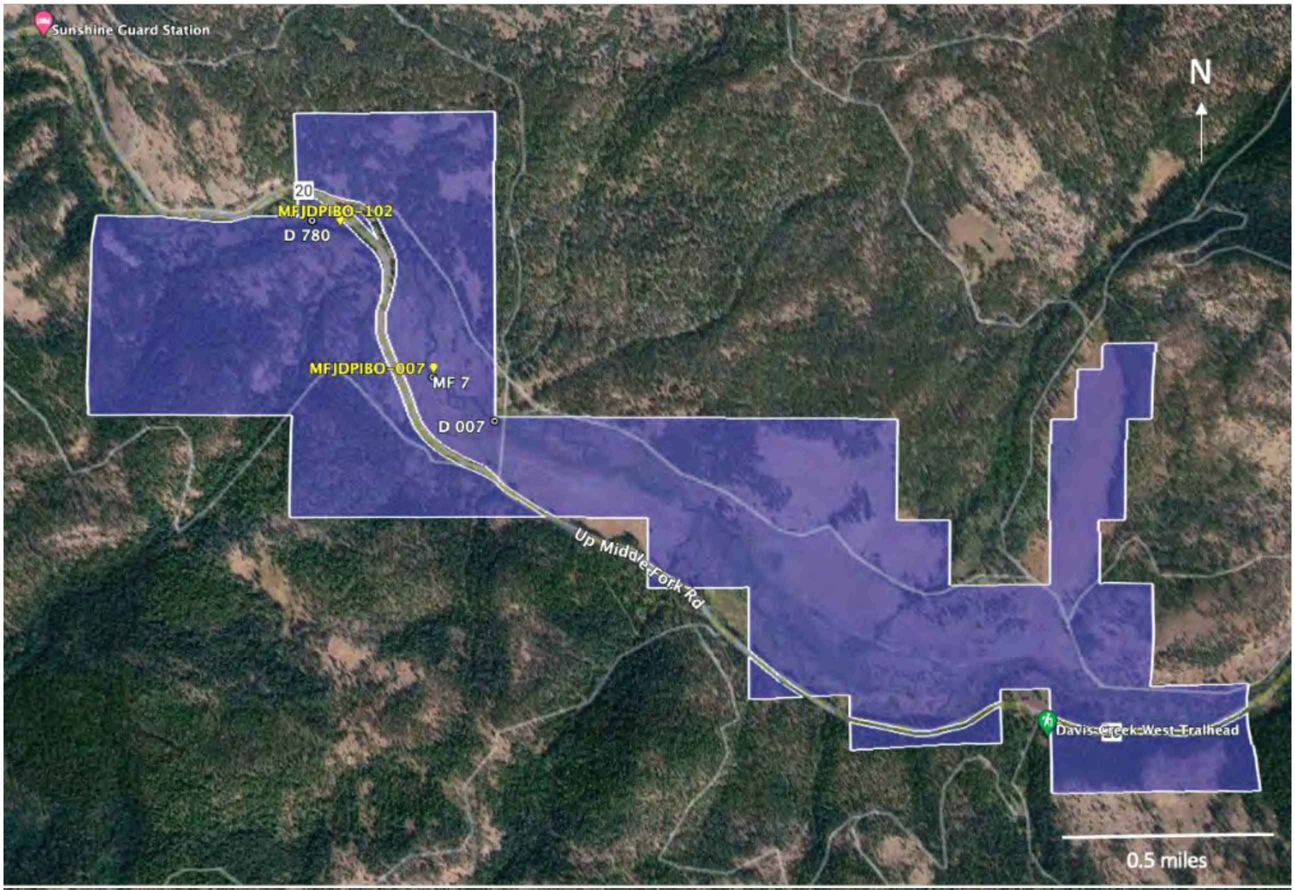

Figure 1. Location of drift (D series), benthic (MF series), and PIBO sampling sites. Site overview (top); Dunstan property (second); Oxbow property (third); Forrest property (bottom). All are owned by CTWSRO.

Analyzing long-term ecological monitoring data involves separating restoration effects from other sources of variation, such as the normal annual fluctuations in macroinvertebrate communities and longer-term stressors such as climate change. The most robust monitoring designs include before-after/control-impact (BACI) designs, which effectively isolate restoration effects in space and time from these other sources of variation. However, restoration programs do not always allow for such designs to be implemented, owing to insufficient opportunity to collect pre-restoration monitoring data or a lack of available suitable control sites. Typically, restoration that occurs at the stream or basin level is more effective in improving overall habitat conditions, compared to smaller scale reach-level restoration in a basin still experiencing multiple stressors. However, larger-scale restoration, such as that in the MFIMW, can present challenges to implementing BACI designs and cleanly separating "before" versus "after" effects because projects are implemented at different times across the watershed. With these challenges in mind, we compared Oregon Department of Environmental Quality (ODEQ) PREDATOR model O/E (observed vs. expected) scores, community composition, and multiple ecological metrics between restored MFJDR and control South Fork John Day River (SFJDR) sites over time, as well as long-term changes at sampling sites, to answer the initial questions posed in Phase 1 of the project.

Overall, Phase 1 analysis found more significant trends in calculated community measures in drift data compared to benthic data. Statistically significant unidirectional trends for increasing biomass and concentration over the 2010-2022 period were seen at three drift sites (D 003, D 367, D 780). However, few overall trends in macroinvertebrate community composition were found, and between-year community differences at both benthic and drift sites appeared to be more closely related to overall sampling period (2010-2015 vs. later years) than to restoration activity. Furthermore, there were no consistent trends between the intensity of restoration done at a site, community change, and number of significant unidirectional trends among sites. The greatest number of significant drift metric trends (eight community measures) was seen at a site that experienced passive restoration in a single year (D 367), while a site that underwent extensive restoration in multiple years (D 007) showed a significant unidirectional trend in only one community measure. Additionally, no longitudinal (upriver to downriver) trends in drift community measures were seen.

Analysis of benthic data found that macroinvertebrate communities were generally in better condition in the MFJDR compared to the SFJDR, with conditions potentially declining at several SFJDR sites but remaining stable across all MFJDR benthic sites over the monitoring period. Few trends in community measures were seen among MFJDR sites, suggesting that benthic community conditions at most sites remained unchanged. Some community variation did appear to occur along the length of the MFJDR, with scores for community measures improving from the lower to the middle sites and then declining between the middle and upper MFJDR sites.

Overall, more statistically significant unidirectional trends in community measures were observed for drift than for benthic samples. Most trends in drift community metrics were suggestive of improving habitat conditions, while trends in benthic measures indicated mixed results or declining habitat conditions. There was no concordance between the type and direction of trends seen at the eight co-located benthic and drift sites.

This Phase 2 analysis examines individual sites and associated restoration activities more closely, with predictions of the likely directions of community change resulting from restoration year, type, activity, and extent. Site-specific trends in taxonomic and ecological metrics were calculated and assessed in the additional context of any changes in habitat conditions gleaned from co-located PacFish/InFish Biological Opinion (PIBO) monitoring program data and other habitat assessments conducted during the sampling period. Co-located drift and benthic sampling sites were examined to determine whether there was agreement in any community trends.

For macroinvertebrate communities collected by drift sampling, Phase 2 questions are:

- How do shifts in macroinvertebrate community structure and biomass compare among sites with differing levels of habitat improvements, i.e., passive, active, passive + active, none?
- For each site where a shift in macroinvertebrate community structure occurred, what does the shift in community structure look like, i.e., is it a shift in functional feeding groups, sensitive taxa, etc.?

For macroinvertebrate communities collected by benthic sampling, Phase 2 questions are:

- What are the mechanisms driving the stability or shift in macroinvertebrate communities through time or space, i.e., is the shift related to restoration?
- For each site where a shift in macroinvertebrate community structure occurred, what does the shift in community structure look like, i.e., is it a shift in functional feeding groups, sensitive taxa, etc.?
- How do shifts in macroinvertebrate community structure compare among sites with differing levels of habitat improvements, i.e., passive, active, passive + active, none?

METHODS

Drift Data Calculations

In Phase 1, all years of drift data (2010-2022) were compiled into a single database and the taxonomy was brought into agreement with the most recent standard taxonomic effort established by the Pacific Northwest Aquatic Monitoring Partnership (PNAMP). Because duration of sampling and mean water velocity differed widely among drift samples, the concentration and biomass of organisms in each drift sample was standardized by calculating organismal concentration as # individuals per m^3 of water that passed through the net during the sampling interval; and by calculating biomass as mg dry weight in the sample per m^3 of water that passed through the net during the sampling interval. The volume of water (m^3) that passed through the net during the sampling interval was calculated as [mean flow (m/s) x duration of sampling event (s) x net area (0.09 m^2)]. Total number of organisms per sample was calculated by multiplying the number of individuals picked from the sample by the inverse of the percentage of the entire sample that was sub-sampled. This was necessary because macroinvertebrate samples are routinely sub-sampled to an organismal count of 500 individuals, and the percent of the total sample picked among all drift samples across time to obtain this count ranged from 13-100%.

ODEQ models applied to the benthic data cannot be applied to drift taxa because the models were developed specifically for benthic macroinvertebrates collected in riffle habitats. However, some of the metrics relating to taxonomic richness, diversity, and tolerance that were calculated for benthic samples were also calculated for drift samples to facilitate detection and comparison of trends. Calculated community metrics used in the Phase 2 analysis included:

- Concentration (# individuals/m^3)
- Biomass (mg/m^3)
- total richness (# taxa in sample)
- Shannon Diversity Index H (measure of species diversity; lower values reflect less diversity)
- % terrestrial (relative abundance of terrestrial invertebrates)
- Ephemeroptera (mayfly), Plecoptera (stonefly), and Trichoptera (caddisfly) richness (individually and as total #EPT taxa)
- % diversity EPT (proportion of total richness comprised of EPT taxa)
- Community BI (biotic index; weighted average of individual taxa tolerance scores; note that these scores are not assigned to all taxa in a sample, as they do not apply to terrestrial taxa and are not known for all aquatic or aquatic/terrestrial taxa)
- % dominance of the top taxon (relative abundance of the most numerically abundant taxon in a sample; higher values reflect a more disturbed or constrained community)
- % small (0-6 mm), % medium (6-12 mm), and % large (12-100 mm) (relative abundance of organisms in different size classes in a sample)

The duration of the drift sampling period varied greatly among years, ranging from as few as two hours to more than 18 hours. The time of day during which drift sampling was performed also varied, occurring in the late morning, afternoon, or evening/night in different years. These variations can alter sample composition, as many insects that exhibit diel drift patterns are more likely to enter the drift at night, while some Trichoptera (caddisfly) and Acari (mite) taxa drift more during the day (Waters 1972; Brittain and Eikeland 1988). Water velocity can also affect the number and type of organisms that become entrained in the drift. Therefore, correlation analyses were done between sampling duration or mean water velocity and total richness, EPT richness, and Shannon diversity index, at the individual site level and among all samples, to determine whether observed trends were more likely due to variation in sampling conditions and methodology as opposed to changes in habitat.

Statistical analyses were done using PAST 4.0 (Hammer et al. 2001) and PRIMER-e v7 (Clarke et al. 2014) software. Comparisons of community composition were run on Bray Curtis similarity indices of square root-transformed taxonomic data. Because the first few years of drift samples were identified only to family level, all drift sample data were collapsed to this level to standardize these community comparisons. Best professional judgment was used to link restoration intensity and timing to individual site data, given the complete restoration inventory. Each site had its own individual mix of restoration types, years, and distances from sampling start sites; we consulted the restoration inventory, which was spatial in nature.

Statistically significant changes in pre- and post-restoration means of each calculated community measure at each site were assessed using t-tests with results reported at alpha = 0.01 and alpha = 0.05 to facilitate assessment at different levels of stringency. Statistically significant changes in community composition were assessed using one-way ANOVA on pre- and post-restoration communities at each site with results reported at alpha = 0.01 and alpha = 0.05. ANOSIM was used to identify taxa that contributed the most to any significant community differences.

Benthic Data Calculations

In Phase 1, all years of benthic data (2010-2022) were compiled into a single database and the taxonomy was brought into agreement with the most recent standard taxonomic effort established by the Pacific Northwest Aquatic Monitoring Partnership (PNAMP). The following community measures were selected for inclusion in these analyses:

- PREDATOR WCCP (Western Cordillera + Columbia Plateau) O/E scores
- ODEQ temperature stress scores
- ODEQ fine sediment stress scores
- Total taxa richness
- Ephemeroptera (mayfly), Plecoptera (stonefly), and Trichoptera (caddisfly) taxa richness
- Percent shredders (percent of individuals in sample belonging to the shredder functional feeding group)
- Shannon Diversity Index H (measure of species diversity; lower values reflect less diversity)
- ODEQ temperature stressor model
- ODEQ fine sediment stressor model

For Phase II, comparisons of community composition were run on Bray Curtis similarity indices of square root-transformed taxonomic data. Best professional judgment was used to link restoration intensity and timing to individual site data, given the complete restoration inventory. Each site had its own individual mix of restoration types, years, and distances from sampling sites, and was determined from consulting the restoration inventory, which was spatial in nature.

Statistically significant changes in pre- and post-restoration means of each calculated community measure at each site were assessed using t-tests with results reported at alpha = 0.01 and alpha = 0.05 to facilitate assessment at different levels of stringency. Statistically significant changes in community composition were assessed using one-way ANOVA on pre- and post-restoration communities at each site with results reported at alpha = 0.01 and alpha = 0.05. ANOSIM was used to identify taxa that contributed the most to any significant community differences.

Graphic analysis was used to examine trends among co-located drift and benthic sites to determine whether similar community changes were seen in different types of samples taken in the same or closely contiguous reaches. Co-located drift and benthic sampling sites are shown in Table 1.

Table 1. Co-located drift (D series) and benthic (MF series) macroinvertebrate sampling sites and PIBO habitat assessment sites on the Middle Fork John Day. Sites are ordered from downstream to upstream.

Drift site	Benthic site	PIBO site
D 702	MF-308	MFJDPIBO-308
D 611	---	MFJDPIBO-310
D 367	MF-312	MFJDPIBO-312
D 001	---	MFJDPIBO-001
D 002, D 003	MF-2	MFJDPIBO-002
---	MF-3	MFJDPIBO-003
D 634	---	—
D 780	---	MFJDPIBO-102
D 007	MF-7	MFJDPIBO-007
D 215	MF-305, MF-215	MFJDPIBO-305, MFJDPIBO-215
D 115	MF-115	MFJDPIBO-115
D 006	MF-6	MFJDPIBO-006
D 004	---	MFJDPIBO-004
D 005	MF-1	MFJDPIBO-005

Habitat Data Calculations

Stream discharge and temperature regimes were investigated using data recorded at the USGS Middle Fork John Day River at Camp Creek gauge (USGS 14043840). Temperature data from this gauge were only available from 2017-2022; flow data was available from 2012. Seven-day average maximum temperatures and the number of days in which a temperature maximum of 23ºC was exceeded were calculated. For discharge, daily average flows were calculated, and the minimum daily average flow for the calendar year was determined, to provide insight into the relative severity of low flows among sampling years.

Potential changes in habitat were assessed by investigating habitat data collected in 2009, 2014, and 2019 at PIBO monitoring sites co-located with macroinvertebrate sampling sites. PIBO monitoring sites co-occurred with 13 of 14 drift sites and at all 10 benthic sites (Table 1). PIBO habitat assessment was done on multiple metrics relating to:

- water chemistry: total dissolved solids (TDS)
- channel dimensions: average bankfull width (Bf), reach length (RchLen), stream reach gradient (Gradient), stream reach sinuosity (Sin), residual pool depth (PoolDep), number of pools per km (PoolFreq), % pools in reach (PoolPct), bankfull width-to-depth ratio (WDTrans), wetted width-to-depth ratio (WDwetTrans)
- substrate: diameter of the 16th, 50th, and 84th percentile streambed particles (D16, D50, D84), % pool tail fines <2 mm and <6 mm (Fines2, Fines6)
- streambanks: % stable banks (Stab), % vegetatively stable banks (VegStab), bank angle (BankAngl), % of bank angles < 90^o (UnCutPct)
- wood: large wood frequency (LWFreq), large wood volume (LWVol).

Pearson's Product-Moment Correlation analyses were run on individual PIBO habitat metrics to detect unidirectional trends at individual sites. PIBO habitat measures at co-located macroinvertebrate sampling sites were examined pre- and post-restoration in conjunction with macroinvertebrate community measures in an attempt to identify habitat trends and potential drivers of macroinvertebrate community changes. Importantly, the PIBO sampling design is not intended to assess changes or trends at individual sites, but rather at the watershed-wide scale. As such, we recognize the limited utility of the PIBO habitat data for these correlation analyses performed at the site scale.

RESULTS

Annual Temperature and Stream Discharge Patterns

The maximum values of seven-day average maximum temperatures measured at the Camp Creek gauge ranged from 25.7-27.4^{o}C between 2017 and 2022. The highest of the seven-day average maximum temperatures (Figure 2) were seen in 2018 (27.2^{o}C) and 2021 (27.5^{o}C); values in other years were lower and similar to each other. In addition, 2021 and 2022 had the greatest number of days in which the temperature exceeded 23^{o}C (54 and 53 days respectively; Figure 3), suggesting that the greatest temperature stress occurred in the most recent sampling years. In contrast, the lowest minimum daily discharges occurred in 2017 and 2019 (4.8 cfs and 3.6 cfs, respectively; Figure 4), suggesting that the most extreme high temperature and low-flow conditions did not occur in the same years.

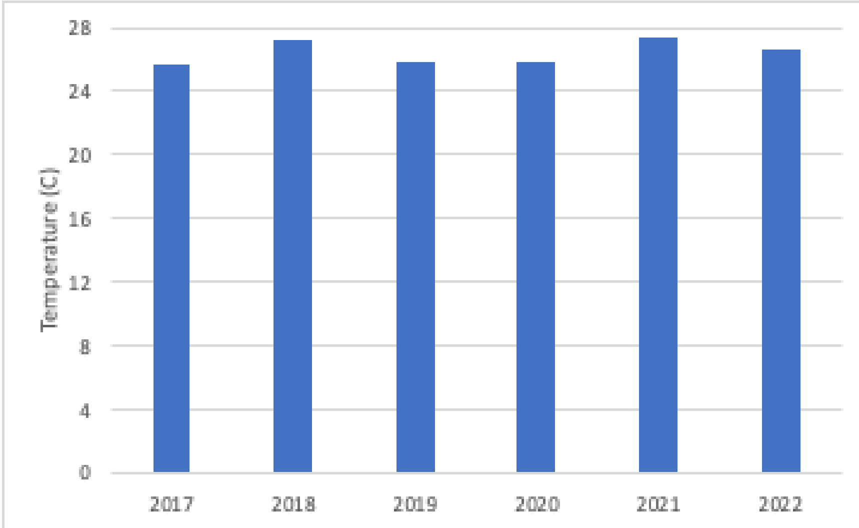

Figure 2. Maximum values of 7-day average maximum temperatures measured by the USGS Camp Creek gauge, 2017-2022.

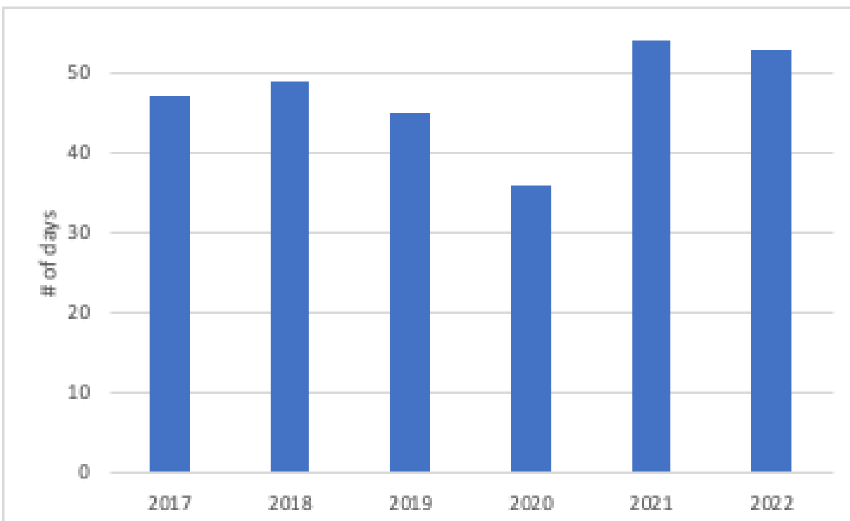

Figure 3. Total number of days in which stream temperature exceeded 23°C at the USGS Camp Creek gauge, 2017-2022.

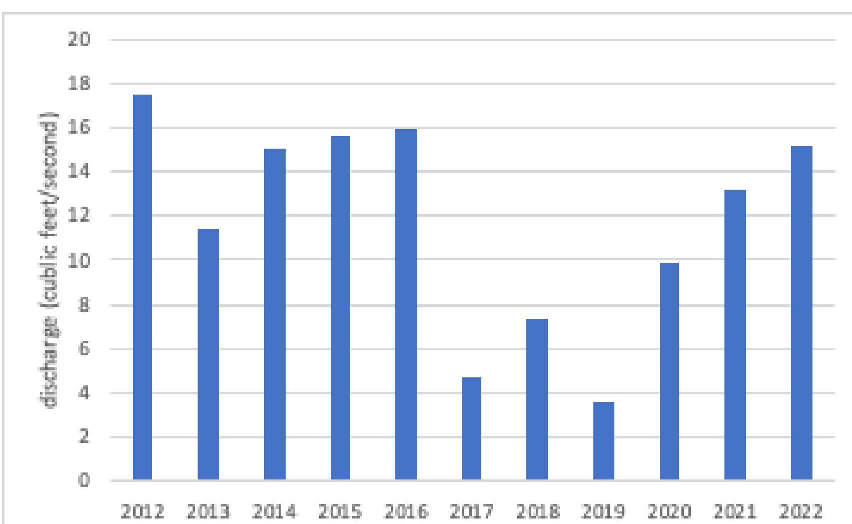

Figure 4. Minimum annual values of average daily discharge (cubic feet/second, cfs) measured at the Camp Creek gauge.

Trends in PIBO Metrics

Trend analysis was conducted on 19 habitat metrics from the PIBO monitoring sites. We conducted the analysis on a site-specific basis; PIBO data are intended to assess watershed-wide changes, but it was the only data available to us for investigating changes in habitat at individual sites to relate to macroinvertebrate community changes. Of a possible 266 individual correlations (19 metrics x 14 PIBO sites), only six (2%) statistically significant unidirectional trends were found (Table 2). Nine PIBO sites had no significant trends in any metrics and of the remaining five PIBO sites, no more than two metrics trended significantly within the monitoring period. These results suggest generally stable habitat in sample reaches through the period spanned by the PIBO monitoring. However, the PIBO data are not collected for the purpose of assessing reach-scale changes. As such, the power of these correlation analyses is low and unlikely to detect change at this scale unless changes are particularly large.

Benthic macroinvertebrates were collected at PIBO monitoring sites in 2009, 2014, and 2019. The RIVPACS predictive model (observed taxa/expected taxa), which returns overall site condition scores between zero and one, was used to determine a site quality score, with scores >0.78 indicating good quality habitat and scores below this threshold indicating poorer habitat. From 2009 to 2019, RIVPACS scores increased at 11 PIBO sites; decreased at two sites; and remained about the same at one site. Mean RIVPACS scores were significantly different across all PIBO sites in 2019 compared to both 2009 and 2014 (Figure 5). However, there was no year or site in which the RIVPACS score was 0.78 or greater; the highest score (0.76) was seen in 2019 at PIBO-001 (co-located with the D 001 macroinvertebrate site) and PIBO-006 (co-located with the D 006 and MF-6 macroinvertebrate sites). However, all mean scores were well below the 0.78 threshold for good condition. Observed differences between years were small and can't be related to specific restoration activities, as model scores experience some degree of annual fluctuation based on flow events and climatic conditions that are unrelated to restoration-related ecological uplift.

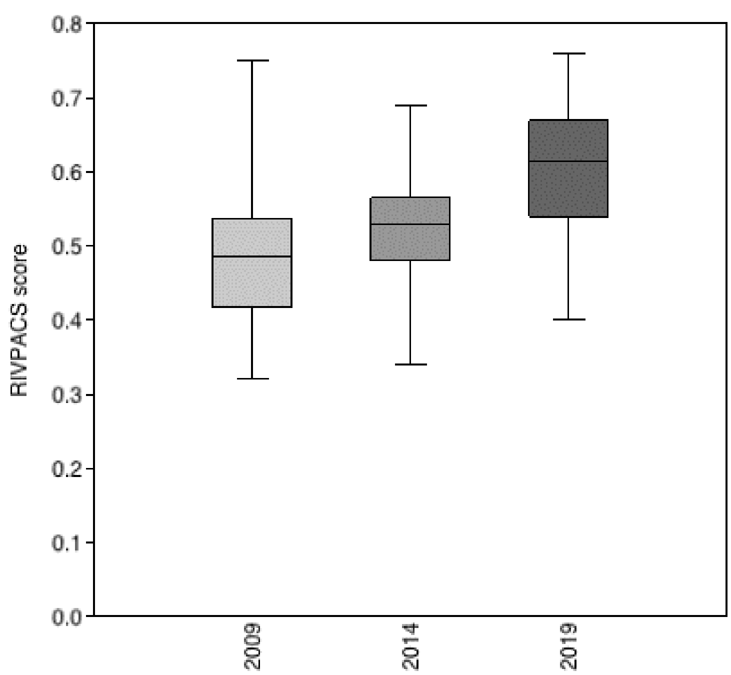

Figure 5. Box and whisker plot of macroinvertebrate RIVPACS scores at PIBO monitoring sites, 2009-2019.

Table 2. Summary of significant unidirectional trends at alpha = 0.01 (**) and alpha = 0.05 (*) at each individual PIBO monitoring site co-located with a macroinvertebrate site for 19 PIBO metrics assessed in 2009, 2014, and 2019. Sites are ordered from downstream to upstream. I = increasing, D = decreasing, N = no statistically significant trend.

	PIBO-308	PIBO-310	PIBO-312	PIBO-001	PIBO-002	PIBO-003	PIBO-102	PIBO-007	PIBO-305	PIBO-215	PIBO-115	PIBO-006	PIBO-004	PIBO-005
Bf	N	N	N	N	N	N	N	N	N	N	N	N	N	N
RchLen	I**	N	N	N	N	N	N	N	N	D*	N	N	N	N
Gradient	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Sin	N	N	N	N	N	N	N	N	N	N	N	N	N	N
PoolDep	N	N	N	N	N	N	N	N	N	N	N	N	N	N
PoolFreq	N	N	D*	N	N	I*	N	N	N	N	N	N	N	N
PoolPct	N	N	N	N	N	N	N	N	N	N	N	N	N	N
WDTrans	N	N	N	N	N	N	N	N	N	N	N	N	N	N
WDwetTrans	N	N	N	N	N	N	N	N	N	N	N	N	N	N
D50	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Fines2	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Fines6	N	N	N	N	N	N	N	N	N	N	N	N	N	N
Stab	N	N	N	N	N	N	I*	N	N	N	N	N	N	N
VegStab	N	N	N	N	N	N	N	N	N	N	N	N	N	N
BankAngl	N	N	N	N	N	N	N	N	N	N	N	N	N	N
UnCutPct	N	N	N	N	N	N	N	N	N	N	N	N	N	N
LWFreq	N	N	N	N	N	N	N	N	N	N	N	N	N	N
LWVol	N	N	N	N	N	N	N	N	N	N	N	N	N	N
RIVPACS score	N	N	N	N	N	N	I**	N	N	N	N	N	N	N

Impact of Sampling Duration and Mean Water Velocity on Drift Sample Metrics

No correlation was observed for the complete drift sample set between sampling duration or mean water velocity and sample richness, EPT richness, or Shannon diversity index (R^2 values from graphic analysis ranged from 0.0007 to 0.0332). Some weak correlations were seen at the individual site level, but R^2 values did not exceed 0.3653 at any site. Thus, any between-site or between-year differences or trends observed in these community measures are not likely to be due to variations in sampling duration or stream flow.

Potential Outcomes of Restoration Activity Types

The impact of stream restoration on aquatic macroinvertebrate communities depends on the type, spatial scale, and setting of the restoration activity. Even a project that is considered "successful" may produce too small an effect to have a measurable impact on macroinvertebrate assemblages, especially when communities have a high intrinsic annual variation (Rubin et al., 2017). Greater changes in the aquatic macroinvertebrate community are expected when restoration activities result in measurable changes in habitat, but such community changes often require years of monitoring to detect. In addition, the effects of restoration projects that lead to increased habitat heterogeneity (i.e., greater diversity of habitat units or more substrate heterogeneity) on benthic macroinvertebrates may not be adequately detected if only riffle habitats are sampled. Further, models and measures established to evaluate water quality may not be suitable for the type of restoration done; for example, increasing the number of pool units is likely to support taxa that are more tolerant of slower flows, warmer temperatures, or higher sediment levels, which is often reflected in water quality models as a decrease in habitat condition. Disturbance generated by the restoration action itself can alter community measures for a period of years as well.

Restoration projects in the MFJDR were implemented at different times, had different spatial extents, and varied in their proximity to macroinvertebrate sampling reaches. They are thus likely to accrue different physical, hydrologic, and geomorphic changes; as a result, the size and nature of biological responses would be expected to vary among individual MFJDR macroinvertebrate sampling sites. Restoration activities implemented in the MFJDR along with their potential outcomes and hypothesized impacts on macroinvertebrate community composition and ecological traits are summarized in Table 3. These were considered when analyzing changes in site-specific metrics and community composition in drift and benthic samples.

Table 3. Potential responses of macroinvertebrate communities to restoration activities.

Restoration action	Potential habitat impacts	Potential macroinvertebrate responses
logjam	build alluvial streambed, govern channel migration; increase pool units	more xylophilic taxa; increased shredder, collector-gatherer, and/or predator organisms
riparian planting	increased riparian vegetation, improved water quality	increased shredder organisms; more terrestrial taxa
livestock exclosure	increased riparian vegetation, improved water quality, increased bank stability	increased total and EPT richness; increased shredder and/or scraper organisms, fewer collector-filterer; more sensitive and/or sediment-sensitive organisms

Restoration action	Potential habitat impacts	Potential macroinvertebrate responses
fencing	increased riparian vegetation, improved water quality, increased bank stability	increased total and EPT richness; increased shredder and/or scraper organisms, fewer collector-filterer; more sensitive and/or sediment-sensitive organisms
dam removal	flow restoration, increased mobilization of fine sediment	increased total and EPT richness; more sensitive and/or sediment-sensitive organisms
channel reconfiguration	increased habitat and flow heterogeneity, decreased sedimentation	increased total and EPT richness; more sensitive and/or sediment-sensitive organisms
floodplain reconnection	increased lateral connectivity, slower flows, decreased channel incision	increased richness, fewer sediment-tolerant organisms
riparian management	increased riparian vegetation, improved water quality	increased total and EPT richness; increased shredder and/or scraper organisms, fewer collector-filterers; more sensitive and/or sediment-sensitive organisms
bank stabilization	decreased sedimentation, incision	more sensitive and/or sediment-sensitive organisms
side channel creation	increased habitat and flow heterogeneity	increased richness
instream habitat improvement	increased habitat heterogeneity, improved water quality	increased total and EPT richness; more sensitive and/or sediment-sensitive organisms

Changes in Community Measures Following Restoration

Community measures at 11 drift sampling sites were examined for significant differences between pre- and post-restoration means. Three of the 14 total drift sites were omitted: D 611, which did not undergo any restoration actions; D 007, which experienced ongoing restoration across so many different years that pre- and post-restoration dates could not be determined; and D 001, where no pre-restoration data were available as all restoration actions were done prior to 2009.

Pre- and post-restoration means did not differ significantly for any community metrics at five of the 11 sites assessed (Table 4). The remaining six sites had significantly different pre- and post-restoration means for 1-4 of the 15 different community measures, most of which related to richness. The direction of change post-restoration suggested improving conditions for all but a single metric, community BI (a measure of tolerance to organic inputs), which increased at two sites following restoration.

Table 4. Significant differences between pre- and post-restoration means of community measures at drift sites, with significance at alpha = 0.01 (**) and alpha = 0.05 (*). Three sites are not shown: Site D 611 was not restored; D 001 was restored pre-2009; D 007 was restored in multiple years such that pre-and post-restoration means could not be calculated. Sites are arranged from downstream to upstream. Letters in parentheses after site names indicate whether restoration was passive (P), active (A), or active+passive (A+P). Green = results improving conditions post-restoration, orange = results declining conditions post-restoration. I = increasing, D = decreasing, N = no statistically significant trend.

	D 702 (A+P)	D 367 (P)	D 002 (A+P)	D 003 (A+P)	D 634 (A+P)	D 780 (A)	D 215 (P)	D 115 (P)	D 006 (A+P)	D 004 (A+P)	D 005 (A+P)
concentration	N	N	N	N	N	I*	N	N	N	N	N
biomass	N	N	N	N	N	I*	N	N	N	N	N
% terrestrial	N	N	N	N	N	N	N	N	N	N	N
total richness	N	N	I*	I**	I*	N	N	N	N	N	N
Shannon H	N	N	N	N	N	N	N	N	N	N	N
#Eph	N	N	I*	I**	I*	I*	N	N	N	N	N
#Ple	N	N	N	N	N	I	N	N	N	N	N
#Tri	N	N	I**	I*	I*	N	N	N	N	N	N
EPT richness	N	N	I*	I*	I*	N	N	N	I*	N	N
rel. div. EPT	N	N	N	N	N	N	N	N	N	N	N
Community BI	N	I*	N	N	N	N	N	N	I*	N	N
% top taxon	N	N	N	N	N	N	N	N	D*	N	N
%small	N	N	N	N	N	N	N	N	N	N	N
%medium	N	N	N	N	N	N	N	N	N	N	N
%large	N	N	N	N	N	N	N	N	N	N	N

There were significant differences in pre- and post-restoration means at nine of the 10 benthic sampling sites for anywhere from 1-7 of the 23 different community measures. In contrast to the drift sites, a greater variety of benthic community measures differed significantly post-restoration (i.e., not just those relating to richness) and changes at many more of the benthic sites suggested declining conditions (Table 5).

Table 5. Significant differences between pre- and post-restoration means of community measures at benthic sampling sites, with significance at alpha = 0.01 (**) and alpha = 0.05 (*). Sites are arranged from downstream to upstream. MF-7 was omitted, as it was restored across so many years such that pre-and post-restoration means could not be calculated. Green = improving conditions post-restoration, orange = declining conditions post-restoration. I = increasing, D = decreasing, N = no significant difference.

	MF-308	MF-312	MF-2	MF-3	MF-305	MF-215	MF-115	MF-6	MF-1
Temperature stressor score	N	I*	N	I**	N	I*	N	N	N
Sediment stressor score	N	N	N	N	N	N	N	N	N
O/E score	D**	N	I*	N	N	N	N	N	N
Richness	D*	N	N	I**	N	N	N	N	N
# EPH	N	N	N	N	N	N	N	N	N
# PLE	N	D*	N	N	N	N	N	N	N
# TRI	N	N	N	I**	N	N	D*	N	N
#EPT	N	N	N	I**	N	N	N	N	N
# sed tol taxa	D*	N	N	N	N	I*	N	N	N
% sed tol org	N	N	N	N	D**	N	N	N	N
# sed sens taxa	N	N	N	N	N	N	N	N	N
% sed sens org	N	N	N	N	N	N	N	N	N
# tol taxa	N	N	N	I*	N	N	N	N	N
% tol org	N	I*	N	N	N	N	N	N	N
% CF	N	N	I*	N	I*	N	D*	N	N
% CG	N	D**	N	N	D**	D*	N	N	N
% PR	N	N	N	N	N	N	D*	N	N
% SC	N	I*	I**	I**	I**	N	I*	N	N
%SH	N	N	N	N	D*	D*	N	N	N
Shannon H	N	N	N	I*	N	N	N	N	N
Evenness EH	N	N	N	N	N	N	N	N	I*
Simpson D	N	N	N	N	N	N	N	N	N
Evenness ED	N	N	N	N	D*	N	D*	N	N

Individual Reach Assessments

To facilitate detection of similar trends between benthic and drift sampling sites, and to attempt to correlate those trends with any habitat changes suggested by the PIBO assessments, sites are discussed as spatially co-located clusters and presented in order from downstream to upstream. A summary of significant trends can be found in Table 6.

D 702/MF-308/PIBO-308

The macroinvertebrate sampling sites located farthest downstream in the system are D 702 and MF-308; they are co-located with the MFJD PIBO-308 monitoring reach on the RPB property. D 702 experienced restoration prior to 2009 (engineered logjam and planting within 500-1000 m of the sampling start site) and in 2020 (planting and exclosures within 500-1000 m of the sampling start site); the same restoration activities were done in the same years at MF-308. PIBO data indicate that little habitat change occurred between 2009 and 2019. The only habitat metric that showed a significant unidirectional trend was reach length, which increased; however, this is a somewhat unreliable metric, as reach length can often vary to some extent across years due to differences in measurement methods and surveyors.

There was no significant difference between the means of any drift community measures pre- and post-restoration, and macroinvertebrate community composition did not differ significantly pre- and post-restoration. However, across the entire sampling period there were statistically significant unidirectional trends in mayfly taxa, caddisfly taxa, EPT taxa, relative diversity of EPT taxa, relative abundance of medium-bodied taxa (all increasing) and community biotic index (decreasing), all of which suggest overall improving conditions from the drift data.

Three benthic community metrics differed significantly pre- and post-restoration, with two indicating declining conditions (lower PREDATOR O/E score, lower total richness) and one indicating improving conditions (fewer sediment-tolerant taxa). MF-308 did not have significant unidirectional trends for any community measures, and the pre- and post-restoration macroinvertebrate communities did not differ significantly. This lack of consistent trends or pre- versus post-restoration differences in benthic and drift community conditions, considered along with a lack of significant PIBO habitat trends, suggests that physical and ecological conditions have largely remained unchanged at this site over the 2010-2022 monitoring period.

D 611/PIBO-310

D 611 is located two miles downstream of the town of Galena and one mile downstream from Bear Creek. This site did not undergo any active or passive restoration. PIBO data indicate that little habitat change occurred here between 2009 and 2019, as there were no significant unidirectional trends in any habitat measures assessed. However, there was a significant unidirectional trend for a single drift community metric, relative abundance of the dominant taxon (decreasing), which corresponds with improving habitat conditions and better habitat stability. There is no co-located benthic sampling site in this reach.

D 367/MF-312/PIBO-312

The benthic MF-312 and PIBO monitoring sites are in the same reach, while the drift D 367 sampling start point is about 100 m upstream. Both sites experienced restoration in the form of riparian livestock fencing installed on Camp Creek within 150-500 m of both the drift and benthic sampling start sites in 2012. PIBO data indicate that little habitat change occurred from 2009-2019; the only metric that showed a significant unidirectional trend was pool frequency (decreased).

A single drift community measure differed significantly at D 367 pre- and post-restoration, with a higher mean community BI post-restoration, which suggests declining habitat conditions. Pre- and post-restoration macroinvertebrate community composition were not significantly different. However, despite an apparent lack of restoration-related changes, multiple community measures showed significant unidirectional trends over time, with increased organismal concentration and biomass; total, mayfly, stonefly, caddisfly, and EPT richness; and increased community BI. Except for community BI, all of these suggest improving conditions in the drift assemblages.

In contrast, five community measures differed significantly pre- and post-restoration in the benthic MF-312 samples, with some reflecting declining habitat conditions (higher temperature stress score, fewer stoneflies, more tolerant organisms) and some reflecting improving conditions (fewer collector-gatherers, more scrapers). Temperature stress scores also showed a significant unidirectional increasing trend over time at this site, suggesting increased heat stress. However, the pre- and post-restoration benthic macroinvertebrate communities were not significantly different. This general lack of consistent trends or pre- versus post-restoration differences in benthic and drift community conditions, considered along with the lack of significant PIBO habitat trends, suggests that physical and ecological conditions largely remained unchanged at this site over the 2010-2022 monitoring period.

D 001/PIBO-001

D 001 is located on the lower end of the Dunstan conservation property owned by the Confederated Tribe of the Warm Springs (CTWSRO). All restoration at D 001 was done prior to 2009, with placement of riparian livestock fencing within 50 m of the sampling start site and removal of a push-up dam located within 50-150 m of the sampling start site. PIBO data indicate little habitat change at the site from 2009-2019, as there were no significant unidirectional trends in any habitat measures assessed. Pre- and post-restoration comparisons could not be made because all sampling occurred after restoration. However, drift data at this site showed a significant increasing unidirectional trend for the number of mayfly taxa over time. While this single measure is suggestive of potentially improving ecological conditions, the lack of similar evidence among all other community measures suggests that drift assemblage conditions have largely remained unchanged at D 001. There is no co-located benthic sampling site in this reach.

D 002/D 003/MF-2/PIBO-002

D 003 and MF-2 are in the same sampling reach as PIBO-002, and D 002 is about 200 m downstream; all are located on the Dunstan property. Restoration activities at D 002 and D 003 were the same, with riparian livestock fencing installed prior to 2009 within 50 m of the sampling start point, and multiple activities implemented in 2016 within 500-1000 m of the sampling start points (channel reconfiguration, floodplain reconnection, instream habitat improvement, bank stabilization). Restoration actions at MF-2 were the same as at the co-located drift sites, except that conifer planting was also done within 50 m of the sampling start site prior to 2009.

Despite extensive restoration, there were no significant unidirectional trends for any PIBO metrics from 2009-2019. However, this span includes just a single assessment after the 2016 projects. Values for most habitat metrics were similar in 2019 compared to the two prior (and pre-restoration) assessment dates, although D50 (diameter of the 50th percentile streambed particle) and WDTrans (wetted width-to-depth ratio at transects) were higher.

For purposes of comparison, the period from 2016-2022 was considered to be post-restoration. Macroinvertebrate community composition in the drift was significantly different post-restoration at both D 002 ($p = 0.0048$) and D 003 ($p = 0.0009$), due primarily to greater mean abundance post-restoration of terrestrial Hemiptera (true bugs) and Chironomidae (non-biting midges) at both sites. The same four community measures were significantly different post-restoration at the two drift sites, with more total, mayfly, caddisfly, and EPT taxa, all of which suggest improved habitat conditions. These sites also had significant unidirectional trends over time in several community metrics including increasing mayfly and caddisfly taxa (D 002), and increasing organismal concentration, biomass, and total and mayfly richness (D 003), all of which suggest improving conditions.

Three benthic community measures were significantly different at MF-2 post-restoration, with a higher O/E score and more scrapers and collector-filterers, suggesting improved habitat conditions. MF-2 also had significant unidirectional trends over time for three community measures, with increasing EPT taxa (suggesting improving conditions) as well as increasing temperature stressor scores and fewer shredders (suggesting declining conditions). Macroinvertebrate community composition was significantly different post-restoration at MF-2, due primarily to greater mean abundance post-restoration of *Cheumatopsyche* (tolerant net-spinning caddisfly that is a warm thermal indicator) and lower mean abundance of *Lepidostoma* caddisflies and Tanytarsini (non-biting midges).

MF-3/PIBO-003

These sites are located on the Dunstan property. Different types of restoration occurred at MF-3 in multiple years. Prior to 2009, riparian livestock fencing and conifer plantings were done within 50 m of the sampling start point. In 2016, channel reconfiguration, floodplain reconnection, instream habitat improvement, and bank stabilization were done within 50 m of the sampling start point. Two sets of restoration activities were done within 50-150 m of the sampling start site; in 2017, upland management, riparian planting, and exclosures were implemented; and in 2021, berm removal and floodplain enhancement were done. To be able to assess pre- and post-restoration community changes, and because much more extensive restoration was done from 2016 to 2021, community measures were examined with 2010-2015 considered pre-restoration, and 2016-2022 as post restoration.

Despite extensive restoration, the only PIBO habitat metric that showed a significant unidirectional trend from 2009-2017 was pool frequency (increasing). Seven benthic community measures were significantly different pre- and post-restoration at MF-3, with a higher temperature stressor score and Shannon diversity H; more total caddisfly, EPT, and tolerant taxa; and more scrapers. All but two of these measures (temperature stress and tolerant taxa) suggest improved community conditions.

These results are generally similar to the significant unidirectional trends seen over all sampling years at this site, with trends in some measures suggesting improved habitat (increasing Shannon Diversity H and total and EPT richness) and others suggesting declining habitat conditions (increasing temperature stress, fewer shredders). These mixed results at this site preclude the ability to infer that ecological conditions have improved over the course of the monitoring period or post-restoration.

Macroinvertebrate community composition was significantly different pre- and post-restoration (p = 0.0005), due primarily to greater mean abundance post-restoration of *Epeorus* mayflies and lower mean abundance of *Lepidostoma* caddisflies and Tanytarsini non-biting midges. There is no co-located drift sampling site in this reach.

D 634

D 634 is located on the Dunstan property, approximately 0.5 miles below Big Boulder Creek. Much of the restoration at D 634 occurred prior to 2009 at multiple distances from the sampling start point, including riparian livestock fencing and conifer planting within 50 m of start, and channel reconfiguration, instream habitat improvements, and engineered logjams within 150-500 m from the sampling start. Multiple restoration activities were implemented in 2015 within 50 m of the sampling start, including channel reconfiguration, floodplain reconnection, instream habitat improvements, and bank stabilization. There is no co-located benthic sampling or PIBO monitoring site in this reach, so changes in habitat metrics could not be assessed.

The means of four community measures were significantly different pre- (2010-2014) versus post-restoration (2015-2022), with greater total, mayfly, caddisfly, and EPT richness. These changes all suggest improved habitat conditions and to a greater extent than the single significant unidirectional trend that was found for the entire sampling period (increasing caddisfly richness). Macroinvertebrate community composition was also significantly different pre- and post-restoration (p = 0.0298), due primarily to greater mean abundance of terrestrial Hemiptera (true bugs) and Baetidae (small minnow mayflies) post-restoration.

D 780/PIBO-102

D 780 is located at the lowest end of the Oxbow property, owned by CTWSRO. Restoration occurred at D 780 in 2009, with channel reconfiguration, floodplain reconnection, and instream habitat improvement within 500-1000 m of the sampling start site; and in 2017, with riparian management done within 50 m of the start site. PIBO measures from 2009-2017 showed a significant unidirectional trend in two metrics that suggested improved habitat conditions (increasing bank stability, increasing RIVPACS O/E score).

Pre-and post-restoration periods are difficult to determine for this site, especially as restoration done in 2009 was active while passive restoration was done in 2017, but for the purposes of analysis 2010-2016 was treated as pre-restoration and 2017-2022 as post-restoration. Mean organismal concentration and biomass as well as mayfly and stonefly richness were all significantly higher post-restoration, suggesting improved conditions. These results are similar to the significant unidirectional trends seen over time for this site, which included increasing concentration and biomass, and total, mayfly, and stonefly richness.

However, there was no significant difference in community composition between pre- and post-restoration periods. There is no co-located benthic monitoring site in this reach.

D 007/MF-7/PIBO-007

MF-7 and the PIBO monitoring station are in the same reach; the D 007 drift site is approximately 300 m upstream. These sites are located on the Oxbow property below and above Beaver Creek, respectively. D 007 experienced extensive restoration in multiple years (prior to 2009, 2012, 2014-2017, 2021) such that pre- and post-restoration periods could not be determined. A similar situation was seen at MF-7, where active and passive restoration was done prior to 2009 and in 2014-2017 and 2021.

However, despite extensive restoration in several continuous years, the only drift community measure at D 007 to show a significant unidirectional trend over the entire sampling period was organismal concentration (increasing), which suggests improved drift assemblage conditions. In contrast, the two benthic community measures at MF-7 that showed a significant unidirectional trend suggested declining benthic conditions, with increasing fine sediment stressor scores and decreasing relative abundance of shredders. There were no significant unidirectional trends seen in any PIBO habitat metrics measured in 2009-2017. This general lack of corresponding unidirectional trends among habitat, drift, and benthic data collectively suggests that habitat and ecological conditions at this site did not change across the entire monitoring period.

D 215/MF-305/MF-215/PIBO-305/PIBO-215

These sites are all within the same 300 m span, below Deerhorn Creek in a more confined reach of the MFJDR. MF-305 and PIBO-305 are co-located at the downstream end of the reach, MF-215 and PIBO-215 are in the same location at the upstream end of the reach, and D 215 is approximately 100 m upstream of MF-305/PIBO-305. All restoration done at D 215, MF-305, and MF-215 was passive, with upland livestock fencing and riparian management in 2009-2011 within 50 m of the sampling start point at all three macroinvertebrate sampling sites. PIBO data reflect no change in habitat from 2009-2017; there were no significant unidirectional trends for any habitat measures at PIBO-305 and only a single significant trend at PIBO-215 (increasing reach length).

Restoration occurred in each year from 2009-2011, and all subsequent years were considered post-restoration. There was no significant difference in post-restoration means for any drift community measures at D 215, and drift community composition was not significantly different between the two periods. Mean values of four benthic metrics at MF-215 were significantly different post-restoration, including higher temperature stressor score, more sediment-tolerant taxa, fewer collector-gatherers, and fewer shredders post-restoration; all except relative abundance of collector-gatherers are suggestive of declining conditions.

Benthic results were also mixed at MF-305, with significantly greater mean relative abundance of collector-filterers and scrapers post-restoration, and fewer sediment-tolerant organisms, collector-gatherers, and shredders. There was no significant difference in benthic macroinvertebrate community composition post-restoration at MF-305 but the communities were significantly different at MF-215 ($p = 0.0395$), due primarily to greater mean abundance post-restoration of *Cinygmula* (mayfly, cold thermal indicator) and lower mean abundance of *Paraleptophlebia* (mayfly, cool-cold thermal indicator) and Tanytarsini non-biting midges.

D 115/MF-115/PIBO-115

The benthic and PIBO sampling sites are in the same reach; the drift site is approximately 200 m upstream. All are above Deerhorn Creek, in a more confined reach of the MFJDR. Restoration at both macroinvertebrate sampling sites was passive, with upland livestock fencing and riparian management in 2009-2011 within 50 m of the sampling start point. There were no significant site-specific unidirectional trends from 2009-2017 in any of the PIBO habitat measures, suggesting little change in physical habitat conditions in that span.

Restoration occurred in each year from 2009-2011, and all subsequent years were considered post-restoration. There was no significant difference for the means of any drift community measures at D 115 post-restoration, and drift assemblage composition was not significantly different pre- versus post-restoration. The only community measure that showed a significant unidirectional trend was abundance of large-bodied insects, which increased over time, suggesting potentially more stable habitat conditions.

Means of five benthic community metrics were significantly different post-restoration at MF-115. Most suggested declining conditions, with lower Shannon diversity H, fewer caddisfly taxa, and fewer predators post-restoration. However, there were also significantly fewer collector-filters and more scrapers post-restoration, which suggests improved sediment conditions. There were no significant unidirectional trends in any community measures over time at this site, but benthic macroinvertebrate community composition was significantly different post-restoration ($p = 0.0167$), due primarily to greater mean abundance post-restoration of *Optioservus* (tolerant riffle beetle, cool_warm thermal indicator) and lower mean abundance of Oligochaeta (tolerant and sediment-tolerant segmented worm) and Tanytarsini non-biting midges. This general lack of trends or consistent post-restoration differences in benthic and drift community conditions, suggests that ecological conditions remained largely unchanged at this site over the 2010-2022 monitoring period.

D 006/MF-6/PIBO-006

The benthic and PIBO sampling sites are in the same reach; the drift site is approximately 250 m upstream. All are in the Vincent to Vinegar project area in the Forrest Conservation Area owned by CTWSRO. Much of the restoration at both D 006 and MF-6 was done before 2009, with planting and grazing management implemented within 50 m of the sampling start site, and instream habitat improvement consisting of riparian management, side channel creation, planting, and large wood placement done within 500-1000 m of the sampling start. Planting and grazing management were done in 2012. In 2020, there was additional planting and fencing; in 2022, channel reconfiguration, floodplain reconnection, instream habitat improvement, and additional planting was done. PIBO data showed no significant unidirectional trends for any habitat metrics from 2009-2017, suggesting no significant physical changes in habitat in that span.

For the purposes of comparison, the pre-restoration period was treated as 2010-2011. Means of three drift assemblage metrics were significantly different post-restoration at D 006, with more EPT taxa and a lower relative abundance of the top taxon (suggesting improved conditions), and a higher community BI (suggesting declining conditions). However, pre- and post-restoration community composition was not significantly different. A similar pattern of mixed results was seen when trends in drift assemblage measures across the entire sampling period were examined, with increasing community BI (suggesting declining conditions) and decreasing top taxon abundance and increasing total richness (suggesting improved conditions).

There was no significant difference between pre- and post-restoration means for any benthic community metrics at MF-6, and pre- and post-restoration macroinvertebrate community composition was not significantly different. The only community metric that showed a significant unidirectional trend across the entire sampling period suggested declining conditions (decreasing total richness). This general lack of consistent trends or consistent pre-versus-post restoration differences in benthic and drift community conditions, along with a lack of significant PIBO habitat trends, suggests that physical and ecological conditions have largely remained unchanged at this site over the 2010-2022 monitoring period.

D 004/PIBO-004

D 004 is located approximately 150 m downstream of the PIBO-004 monitoring site adjacent to Bates Pond State Park. Restoration at D 004 included riparian management, instream habitat improvement, fencing, and planting within 50 m of the sampling start site in 2013. There were no statistically significant unidirectional trends in any PIBO habitat metrics from 2009-2017, even though two of those monitoring years occurred post-restoration. The post-restoration community was not quite significantly different at D 004 ($p = 0.0502$); the differences were due primarily to greater mean abundance of terrestrial Hemiptera (true bugs) and fewer aphids post-restoration. There is no co-located benthic sampling site in this reach.

D 005/MF-1/MFJDPIBO-005

The three sampling stations are located in the same reach, adjacent to Bates Pond State Park. Restoration at D 005 and MF-1 was done in the same year (2013) and included riparian management, instream habitat improvement, fencing, and planting within 50 m of the sampling start site. There were no statistically significant unidirectional trends in any PIBO habitat metrics from 2009-2017, although two of those monitoring years occurred post-restoration.

Means of drift community measures did not differ significantly pre- and post-restoration. However, pre- and post-restoration communities did differ significantly ($p = 0.0102$), due primarily to greater mean abundance of terrestrial Hemiptera (true bugs) and fewer aphids post-restoration.

Means of benthic community measures did not differ significantly pre- and post-restoration except for greater evenness ED post-restoration, which suggests a more balanced or stable community. However, macroinvertebrate community composition at MF-1 was not significantly different pre- and post-restoration, and the only benthic community measure that showed a significant unidirectional trend across the entire sampling period was relative abundance of shredders (decreasing). Both habitat and benthic macroinvertebrate measures suggest that conditions have largely remained unchanged at this site over the monitoring period.

Table 6. Summary of pre- and post-restoration changes in community measures and macroinvertebrate community composition at drift and benthic sampling sites (alpha = 0.05). Significant site-specific unidirectional trends in PIBO measures (2009-2019) are included to provide context for degree of habitat change. Pre- and post-restoration periods could not be determined for D 007 due to ongoing restoration throughout the sampling period. Inc = increasing, dec = decreasing, none = no significant difference.

			Pre- and post-restoration changes				
Co-located monitoring sites			Drift		Benthic		PIBO
Drift	Benthic	PIBO	Community measures	Community composition	Community measures	Community composition	Habitat trends
D 702	MF-308	MFJDPIBO-308	none	none	dec O/E, RICH, # SedTol	none	inc RchLen, WDTrans
D 611	—	MFJDPIBO-310	no rest. done	no rest. done	—	—	none
D 367	MF-312	MFJD PIBO-312	inc CommBI	none	inc TempStress, %Tol, %SC; dec PLE, %CG	none	dec PoolFreq
D 001	—	MFJD PIBO-001	all rest. pre-2009	all rest. pre-2009	—	—	none
D 002, D 003	MF-2	MFJD PIBO-002	inc RICH, EPH, TRI, EPT for both sites	more Hemiptera, Chironomidae post-rest.	inc O/E, %CF, %SC	more *Cheumatopsyche*, fewer *Lepidostoma*, Tanytarsini post-rest	none
—	MF-3	MFJD PIBO-003	—	—	inc TempStress, RICH, TRI, EPT, %Tol, %SC, Shannon H	more *Epeorus*, fewer *Lepidostoma*, Tanytarsini post-rest	inc PoolFreq
D 634	—	—	inc RICH, EPH, TRI, EPT	more Hemiptera, Baetidae post-rest	—	–	—
D 780	—	MFJD PIBO-102	inc conc., biomass	none	—		inc Stab, RIVPACS score
D 007	MF-7	MFJD PIBO-007	no pre-/post-rest yrs	rest. in too many yrs to determine pre- & post periods	rest. in too many yrs to determine pre- & post periods		none
D 215	MF-305, MF-215	MFJD PIBO-305, MFJD PIBO-215	none	none	MF215: inc TempStress, #SedTol; dec %CG, %SH; MF305: dec %SedTol, %SH,	MF-305 none; MF-215 more *Cinygmula*, fewer *Paraleptophlebia*, Tanytarsini	PIBO-305 none; PIBO-215 dec RchLen

			Pre- and post-restoration changes				
Co-located monitoring sites			Drift		Benthic		PIBO
					Evenness ED; inc %CF, %SC		
D 115	MF-115	MFJD PIBO-115	none	none	dec TRI, %CF, %PR, evenness ED; inc %SC	more Optioservus, fewer Tanytarsini, Oligochaeta	none
D 006	MF-6	MFJD PIBO-006	increasing EPT, CommBI, dec % top taxon	none	none	none	none
D 004	—	MFJD PIBO-004	none	D004 none; D005 more Hemiptera, fewer Aphididae	—	—	none
D005	MF-1	MFJD PIBO-005	—	—	inc evenness EH	none	none

DISCUSSION

The following Phase 2 questions were posed for the drift data:

- How do shifts in drift macroinvertebrate community structure and biomass compare among sites with differing levels of habitat improvements, i.e., passive, active, passive + active, none?

Of the 11 drift sites at which restoration occurred and within a time frame that pre- and post-restoration communities could be assessed, one site experienced only active restoration, three experienced only passive restoration, and seven experienced a combination of active+passive restoration. There was no significant difference between the pre- and post-restoration community composition at seven of the drift sampling sites that experienced restoration. Pre- and post-restoration communities differed significantly at four of the 11 restored drift sites that could be analyzed for change (D 002, D 003, D 005, and D 634), all of which experienced both passive and active restoration. Three of these sites are contiguous along the same reach (D 002, D 003, and D 634), while the fourth (D 005) is further upstream.

A greater number of drift sites showed significant differences in pre- and post-restoration means of community measures. Six sites showed significant changes in anywhere from one to five measures post-restoration; of these, one site experienced passive restoration (D 367, one significant trend); one experienced active restoration (D 780, four significant trends); and four experienced active and passive restoration (D 006, three significant trends; D 002 and D 634, four significant trends; D 003, five significant trends).

Overall, drift sites that experienced both passive and active restoration showed greater changes in community structure, even though PIBO data did not indicate much change in physical habitat. However, this could also simply reflect the fact that the majority of restored drift sites experienced active plus passive restoration. Some changes did occur over time even in the absence of restoration; D 611 was not restored but the drift community showed a significant unidirectional trend over time for decreasing relative abundance of the dominant taxon, which suggests more stable and/or improved habitat. Given the quantity of restoration that occurred across multiple years and sites, and the location of D 611 (which is downstream of all but one drift sampling reach), there may have been overall habitat uplift within the system. However, this was not seen consistently among the drift sites that experienced only passive restoration; two (D 115, D 215) showed no significant difference in any community metrics pre- and post-restoration, while one (D 367) had significantly greater mean community BI post-restoration, which suggests declining habitat conditions.

- For each site where a shift in drift macroinvertebrate community structure occurred, what does the shift in community structure look like, i.e., is it a shift in functional feeding groups, sensitive taxa, etc.?

When considering pre- and post-restoration differences in both macroinvertebrate community composition and the mean values of community measures, two drift site groups (D 634 and D 002/D 003) showed the greatest changes in community structure and relatively consistent evidence of improving conditions. D 002 and D 003 experienced extensive restoration, with riparian livestock fencing installed prior to 2009, and channel reconfiguration, floodplain reconnection, instream habitat improvement, and bank stabilization done in 2016. The main contributors to the taxonomic changes seen included more terrestrial Hemiptera (true bugs) and adult Chironomidae (non-biting midges) post-restoration. Chironomidae are a ubiquitous group that often form a large part of the diversity and/or biomass of aquatic samples. Members of this family have a range of tolerances and feeding modalities, but because identification of Chironomidae in drift samples was left at family level there is no way of knowing if specific ecological traits within the family were favored over time. Increased numbers of terrestrial Hemiptera could reflect the impacts of floodplain reconnection or channel reconfiguration, if riparian habitat was expanded or improved. Post-restoration changes in community metrics at both D 002 and D 003 included greater invertebrate biomass and concentration and more total, Ephemeroptera, Trichoptera, and EPT taxa. This increase in invertebrate abundance and in diversity of sensitive groups suggest improved habitat conditions. PIBO data from this reach included only a single assessment after the 2016 projects, and values for most habitat metrics were similar in 2019 compared to the pre-restoration years, although D50 (diameter of the 50th percentile streambed particle) and WDTrans (wetted width-to-depth ratio at transects) were higher. A higher D50 may indicate improved substrate conditions, as it can reflect decreased fine sediment deposition which in turn would improve conditions for macroinvertebrates that are sensitive to fine sediment being deposited on gill surfaces and compromising respiration, such as EPT.

Site D 634 experienced similar extensive active+passive restoration prior to 2009 and in 2015, with much of the same activities as at D002/D 003 (fencing, planting, logjam installation, channel reconfiguration, floodplain reconnection, instream habitat, and bank stabilization). The significantly greater mean numbers of total, Ephemeroptera, Trichoptera, and EPT taxa seen post-restoration also suggest improved habitat. The main contributors to the taxonomic changes seen post-restoration included more terrestrial Hemiptera (true bugs), which could reflect improved riparian conditions; and more Baetidae (small minnow mayflies), many of which are fairly cosmopolitan and have a wider range of temperature tolerances. However, there was no co-located PIBO monitoring site in this reach, so changes in habitat metrics could not be assessed and no inferences can be made regarding potential drivers of observed changes.

The following Phase 2 questions were posed for the benthic macroinvertebrate data:

- What are the mechanisms driving the stability or shift in benthic macroinvertebrate communities through time or space, i.e., is the shift related to restoration)?

While the 10-year analysis of the macroinvertebrate data exclusively utilized a single numeric measure derived from a predictive model (see Appendix J in Middle Fork IMW Working Group 2017), this 15-year analysis assessed the benthic macroinvertebrate data using multiple measures because concordant results among measures would provide stronger evidence for real changes or trends. The 15-year analyses found that consistent lines of evidence for improved ecological conditions are generally weak in the benthic data.

Significant differences in pre- and post-restoration community composition were seen at seven of the restored benthic sites: MF-2, MF-3, MF-115, MF-215, MF-305, MF-308 and MF-312. Significant changes that consistently suggested improved benthic conditions pre- versus post-restoration occurred at only MF-2 and MF-3; one of these (MF-2) is co-located with drift site group D 002/D 003, which also showed more significant community changes and relatively consistent evidence of improving conditions. D 002/D 003 and MF-2/MF-3 are all located within the Dunstan conservation area. Results at the other five benthic sites were mixed with respect to improving or declining ecological conditions. Co-located PIBO habitat data show few trends from 2009-2019 to suggest meaningful habitat change at MF-2 or MF-3 that could explain or suggest a mechanism of ecological change implied by the benthic data. PIBO data from MFJD-002 (co-located with MF-2) suggest that substrate conditions have potentially improved from 2009 to 2019 with an increase in median substrate particle size (D50).

The apparent improvements in ecological condition at MF-2, which include an increased O/E score and an increase in scraper relative abundance, may be resulting from improved stream substrate conditions, which in turn could be the result of decreased sediment loading. Decreased fine sediment deposition (implied by the higher D50) increases the interstitial spaces between larger substrates and improves substrate quality for macroinvertebrates with a scraper feeding modality. Decreased fine substrate can also allow better survival of macroinvertebrates that are sensitive to fine sediment being deposited on gill surfaces and compromising respiration. While the substrate and benthic data at MF-2 suggest that ecological improvement through such mechanisms could be occurring, continued monitoring of both habitat and the benthic community at this site will reveal whether such apparent relationships indeed exist.

Significant pre- to post-restoration changes in the benthic community at MF-3 most consistently suggested improved ecological conditions. These apparent positive changes included increased total taxa richness, increased Trichoptera richness, increased EPT richness, and increased scraper relative abundance, all of which suggest that ecological conditions have improved at MF-3. However, increasing temperature stress scores at MF-3 suggest increased community tolerance to thermal stress. PIBO habitat data that would allow any inference of mechanisms driving improvement at MF-3 are scant. The only PIBO habitat measure to show any change from 2009 to 2019 at this site was an increase in pool frequency, but at the same time, percent pool length decreased in the reach. This suggests that overall habitat heterogeneity, improvements in which may lead to increased biodiversity and ecological integrity, has not remarkably changed between 2009 and 2019. As such, the amount of habitat data currently available precludes any ability to infer what might be driving the potentially improved ecological conditions at MF-3.

- For each site where a shift in benthic macroinvertebrate community structure occurred, what does the shift in community structure look like, i.e., is it a shift in functional feeding groups, sensitive taxa, etc.?

Between the two benthic sites where significant changes most consistently suggested improved benthic conditions (M-2 and MF-3), both sites showed an increase in the relative abundance of scrapers. An increase in scrapers can suggest improved substrate conditions with respect to deceased fine sediment loading and deposition. However, other mechanisms can also drive such changes in the relative abundance of functional feeding guilds. For example, variation in streamflow, sunlight, and nutrient loading can produce conditions that increase algal growth on stream substrates and result in similar increases in macroinvertebrates with a scraper feeding modality.

Otherwise, shifts in the macroinvertebrate communities were not consistent between these two sites, despite their relatively close proximity to each other. As discussed earlier, changes at MF-2 also included an increased O/E score (a measure of overall ecological condition), while changes at MF-3 included a higher total taxa richness and higher caddisfly richness. Among all benthic sites, the results at MF-3 provide the most compelling evidence of changes in the macroinvertebrate community that would be consistent with or expected to result from restoration activities aimed at improving habitat and water quality. Continued monitoring of the benthic community at both MF-2 and MF-3 should reveal whether these apparent ecological changes will persist as a result of restoration efforts or if they are related to other drivers and will continue to vary.

- How do shifts in benthic macroinvertebrate community structure compare among sites with differing levels of habitat improvements, i.e., passive, active, passive + active, none?

The significantly higher temperature stressor scores observed at several sites post-restoration could be influenced by the fact that 2021 and 2022 were the most severe high-temperature years based on the Camp Creek gauge. Changes in the relative abundance of shredders seen at several sites were likely due to an unusually high abundance of this feeding group in 2011-2013, although the habitat driver for that period of elevated abundance is not known.

Overall, there was no consistent relationship between the intensity of restoration actions and the macroinvertebrate community response. However, some sites experienced more extensive restoration prior to 2009 while restoration done in later years was more passive and/or not done instream, so it is possible that the ecological signal from those later restoration years may be weaker and more difficult to detect.

For the purposes of analysis in this project, drift and benthic assemblages were examined separately to detect and compare trends. However, drift and benthic fauna are not discrete, separate communities (Waters 1972); rather, the benthic fauna participate in the drift to differing extents based on changing biotic and abiotic conditions. Drift assemblages are more often used to determining prey availability for insectivorous fish or to investigate the life history of selected invertebrate taxa (Danehy et al. 2017; Rader 1997; Haskell and Tiffan 2011; Woo et al. 2018; Cheney et al. 2019), but fewer studies have evaluated the temporal and spatial impacts of steam restoration, catchment land use, or habitat changes on drift community composition (Culp et al. 1986; Svendson et al. 2004; Nagel et al. 2022).

Composition of the drift assemblage is influenced by a large array of factors such as the habitat, season, invertebrate body size and life stage, and stream discharge. Entry into the drift may be passive (i..e, accidental) when invertebrates are dislodged due to changes in the physical condition of the stream; active (i.e., intentional), to avoid predation, seek food, or encounter better habitat; or catastrophic, when organisms are dislodged due to a strong stressor or disturbance, such as flooding or water abstraction (Müller 1954; Waters 1965; Wiley and Kohler 1980; Brittain and Eikeland 1988; Wooster and Sih 1995; Wooster et al. 2016; González et al. 2018; Gomi et al. 2020). In addition, some taxa may be more frequent intentional drifters, exhibiting a diel periodicity; for example, small minnow mayflies (Baetidae) and chironomid midges entrain in the drift more actively at night, while water mites and some caddisflies drift more actively by day (Waters 1972; Danehy et al. 2017). In temperate habitats, drift assemblages are generally dominated by mayflies, black flies, stoneflies, and caddisflies (Danehy et al. 2017; Nagel et al. 2022). Drift samples facilitate examination of the contribution of terrestrial insects to the stream food base but may also exhibit greater taxonomic variability compared to other aquatic invertebrate sampling methods (Gibbs et al. 2023). Because drift composition is constrained by multiple factors and the aquatic component of the drift is a small subset of the benthic fauna, the drift is a less sensitive tool to measure restoration response compared to the benthos.

These analyses found very little agreement between drift and benthic results at co-located sites. However, the drift data seemed to show more trends than did the benthic data. While on the surface this may seem a compelling reason to include drift sampling in the MFIMW monitoring program, we argue that the apparent trends in the drift data do not result from improved habitat or water-quality conditions, or even inter-annual variation in flows, thermal regimes, or water quality. Rather, the drift data, having been collected using less standardized methods than the benthic data, and being subject to much greater intrinsic variation in taxonomic composition for the reasons described above, are considerably less useful for detecting ecological change resulting from restoration activities. Furthermore, the original MFIMW implementation plan (Bennett and Bouwes 2011) suggests that the drift data were being collected to assess the abundance and availability of food for juvenile salmon for the purpose of modeling juvenile salmonid growth rates and not for assessing ecological integrity of the macroinvertebrate community. Accordingly, we recommend cessation of drift sampling in the MFIMW monitoring program unless the originally intended use of the data is still desired and additional years of data are warranted. For purposes of assessing ecological integrity, the drift data are redundant with the benthic data at best and very likely less reliable or meaningful.

While the drift sampling component of the MFIMW macroinvertebrate monitoring program is unnecessary for assessing improvements in ecological conditions, the benthic sampling should continue, despite a lack of compelling results to date at most sites. The 10-year analysis of the benthic data took an approach of a BACI design and ANOVA analysis to assess the benthic data for ecological changes in response to restoration. In contrast, the present analysis focused on trend detection (Phase 1), followed by a closer “before-after'' restoration analysis at each site. Generally, these data are not well suited to a BACI design because the nature of watershed-wide restoration implementation renders identifying treatment groups (restoration versus controls and pre- versus post-restoration) and ensuring relatively well-balanced sample sizes difficult to achieve. In the case of the MFIMW, restoration activities occurred as early as 2009 at many sites and additional restoration followed, effectively resulting in no pre-restoration data at those sites. Furthermore, reaches receiving less or later restoration work may still accrue benefits of work occurring in upstream reaches, confounding the ability to identify “control” sites where no restoration effects are expected to occur.

As such, trend detection will be the most effective use of the current benthic data set, as originally proposed by Cole and Saltman (2010). Overall, the longer-term benthic data from the MFIMW do not consistently indicate trends in ecological conditions at any sites across the 12-year monitoring period. Rather, conditions in the IMW largely remained stable. In contrast, the longer-term data from the South Fork suggest potentially declining ecological conditions at several sites (SF-5, SF-7, SF-a3, and SF-a4; see Searles Mazzacano and Cole 2023). These apparent declining trends in the South Fork are largely the result of the past few years of data. Data collected in the next several years should indicate whether these apparent trends will persist.

Importantly, longer-term data that potentially span decades could be used to determine whether conditions in the IMW improve, remain stable, or even decline. The South Fork data could continue to serve as a “control” to determine whether condition trajectories continue to diverge between the two watersheds, as appears to be potentially occurring currently at some sites. Both watersheds are subject to the ongoing and likely increasing stressors of elevated temperatures and variable stream flows. However, the MFIMW may have higher resilience from these increasing stressors due to the restoration that has been done and continues to occur. Accordingly, we recommend that benthic monitoring continue in both the South Fork and the MFJDR basin. The number of monitoring sites could be pared down to five or six in each watershed, particularly if this would allow additional effort to be put into accompanying habitat and water quality monitoring that would better enable identification of potential drivers of any observed macroinvertebrate community shifts.

This analysis was tasked with seeking to answer questions about what may be driving observed trends or changes in the macroinvertebrate communities in the MFIMW. Only habitat data collected under the PIBO program were available for such analyses. Importantly, the PIBO habitat data are not collected with reach-scale assessment in mind. The PIBO habitat data are collected at five-year intervals, resulting in only three data points per site over a 10-year period, which are unlikely to provide a meaningful assessment of trends or changes at the reach scale. Ecological monitoring programs that seek to understand why observed ecological changes occur are best served to collect attendant physical and even chemical data. In doing so, linkages between changes in the ambient environment and ecological changes can explicitly be made. In the case of ecological changes induced by stream restoration, those changes will most likely be shaped by changes in the stream habitat quality or heterogeneity, by changes in the thermal regime, or by changes in water quality during peak-stress periods.

Monitoring programs aimed at assessing restoration benefits should seek to capture and detect changes in these ambient environmental conditions and then analyze the biological data for improvement in response to these ambient changes. Only then can inferences be made about what factors are likely responsible for driving measured ecological change. Without such ambient environmental data, one cannot know whether a lack of measured ecological change was the result of a lack of environmental change (insufficient net effect of the restoration) or if the ecological community has not (yet) responded to any measured environmental change.

Lessons Learned/Recommendations

Comments on Lessons Learned presented in the 10-year monitoring report

Recommendations made in the 10-year report focused primarily on sample design and analysis but provided little concrete guidance for specific changes to be implemented. Reviewers noted that the initial sampling design was not balanced, which minimized the ability to conduct meaningful statistical analyses, but recommendations under this heading were vague, and included "*Ensure sufficient sample size and power to answer research questions*" and "*Carefully consider all attributes of the predictive model used to guide stream restoration*". While these are valid points, they provided little on-the-ground guidance for future work. We address aspects of these shortcomings in our recommendations below.

Recommendations made under Monitoring included "*Explore if functional group analysis and spatial models would support the hypothesis that management actions are affecting the biotic integrity of the MFJDR*" and "*...increase the number of macroinvertebrate collection sites within control reaches to better explore biotic integrity changes with stream restoration*". Functional group analysis of the macroinvertebrate community was used in the current analysis but yielded inconsistent results in the context of restoration impacts. The challenges of identifying true "control" sites where no restoration effects are expected to occur in the MFJDR were addressed in the current report, along with our rationale to sample fewer sites over a longer period with an accompanying intensification of physical and chemical habitat monitoring to provide better context for potential drivers of community changes.

Recommendations made under Analysis sought to mitigate the impacts of inconsistent streamflow and temperature data collection on statistical analysis. Reviewers encouraged "*a consistent data collection effort across data types, years, and sites to limit noise and variability and increase power of the analysis*", but again provided no specific guidance to implement the recommendation. We provide our own Lessons Learned and Recommendations below, many of which also relate to issues found in the 10-year report.

Lessons Learned and Recommendations for the current report

Analysis of benthic and drift samples:

Lesson Learned: The macroinvertebrate sampling program was not established with a BACI design in mind, yet attempts have been made to use the data with this design. Too few pre-restoration data and control sites obviate the ability to effectively use this design.

Recommendation: Future analyses should continue to focus on trend monitoring and not assessing before-after/control-treatment effects.

Lesson Learned: Drift sampling has been performed without a clear understanding of how the data are to be used in ecological assessments, and drift data are less useful for measuring ecological condition than are benthic data.

Recommendation: Omit drift sampling from the MFIMW monitoring unless there is an explicit intended purpose for using them for assessing juvenile salmonid food abundance.

Lesson Learned: Physical habitat and water quality data for the Middle Fork are scant and are not collected under the MFIMW monitoring program. PIBO habitat data are not collected at the same scale or frequency as the MFIMW macroinvertebrate data, and therefore are of limited utility for assessing habitats changes at individual MFIMW monitoring sites.

Recommendation: Consider adding physical habitat assessment to occur concurrently with benthic macroinvertebrate sampling. Do not attempt to use PIBO habitat data to assess changes at the reach scale. Consider adding continuous temperature monitoring (i.e., HOBO loggers) at each macroinvertebrate monitoring location to better characterize the thermal regime. Because habitat assessment increases the effort and cost of monitoring, assessment could focus on select habitat attributes that are most likely to change more rapidly in response to restoration and that are more likely to elicit a response from the macroinvertebrate community. Such attributes could include selected substrate and riparian metrics such as particle size/embeddedness, percent composition of habitat types in the sampled reach (riffles, pools, glides, etc), % stream shading (measured with a densiometer), % cover of native and non-native plants in riparian zone at different levels (groundcover, understory, canopy). This list is not prescriptive, but only intended to assist with developing a more specific list of habitat metrics, should the addition of habitat assessment be considered.

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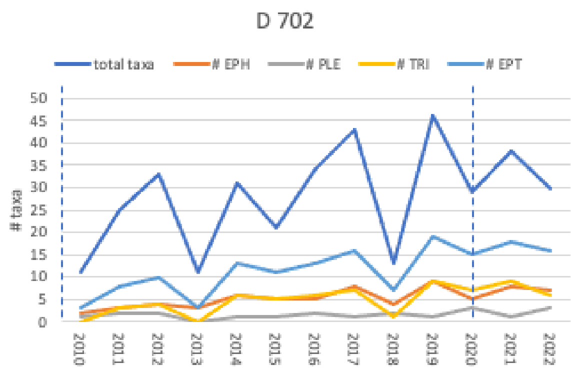
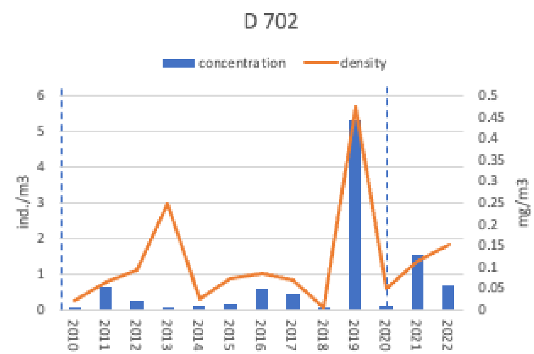
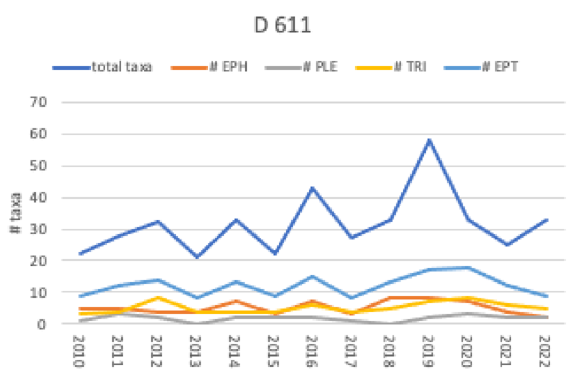
APPENDICES

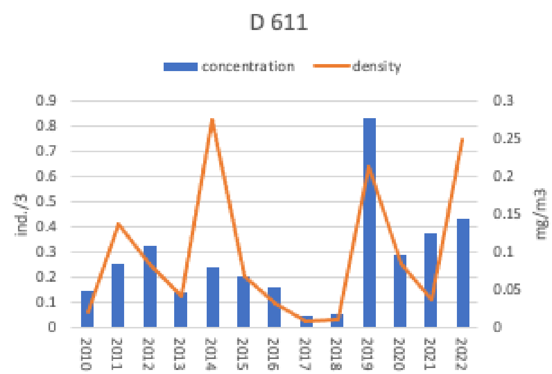
A. KEY COMMUNITY MEASURES AT MACROINVERTEBRATE SAMPLING SITES IN RELATION TO RESTORATION YEAR(S)

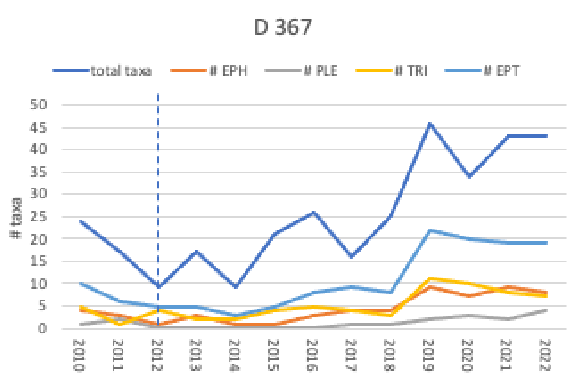
A. Drift sites, presented in order from downstream to upstream. Dashed line(s) indicates restoration years. Note that no restoration was done at D 611, and that D 001 was restored prior to 2009.

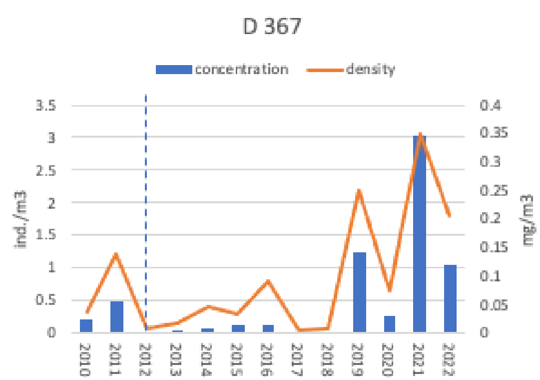


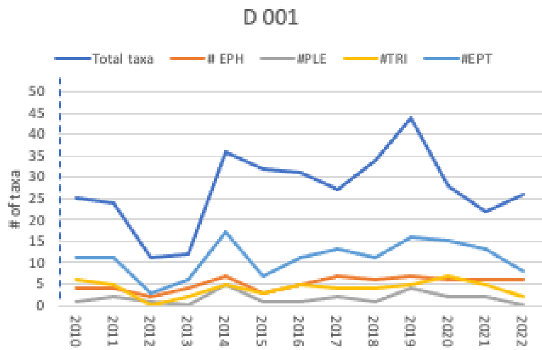








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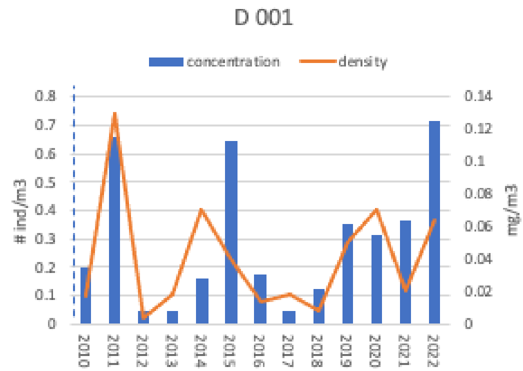
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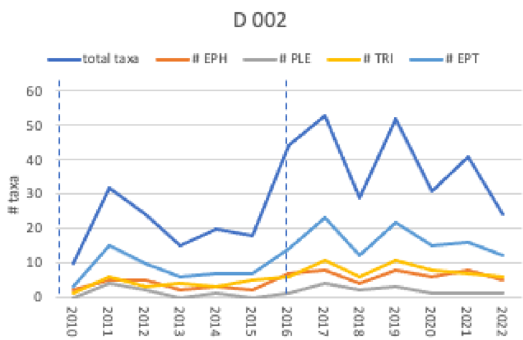
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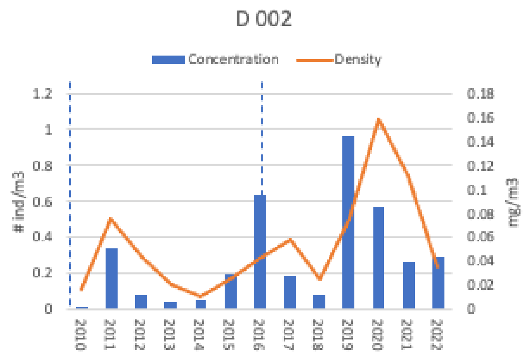
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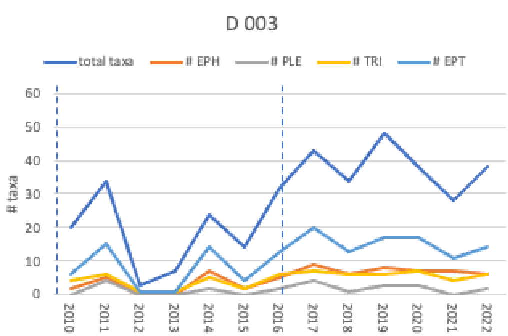
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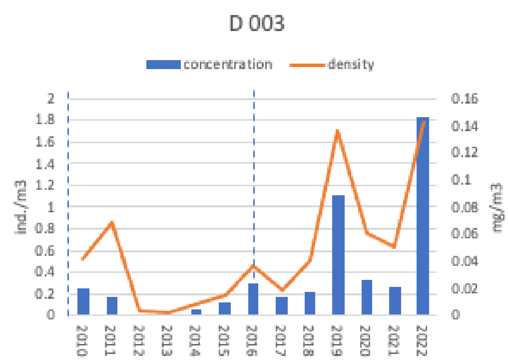
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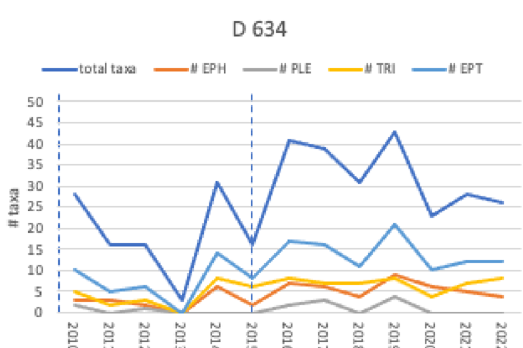
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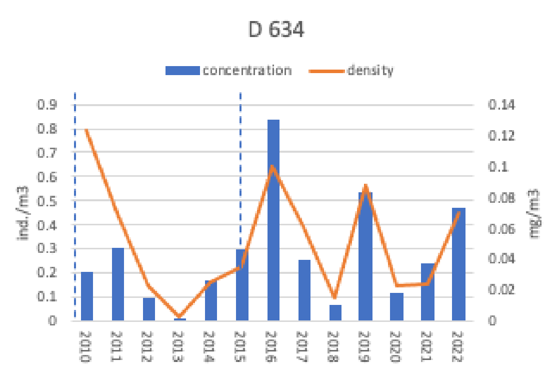
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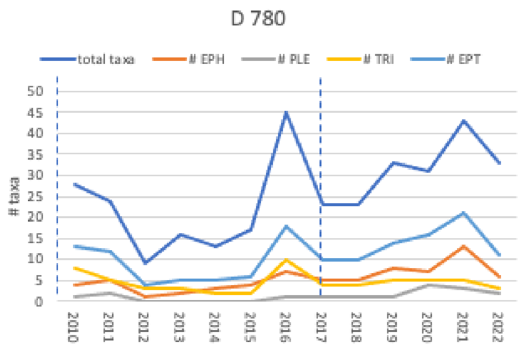
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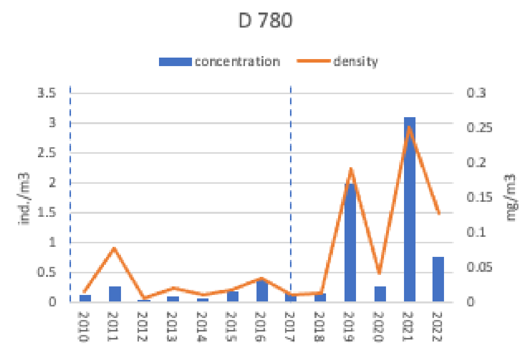
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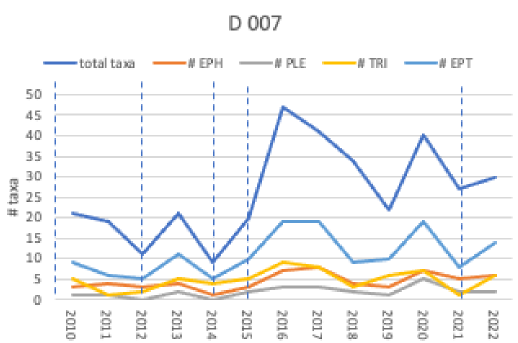
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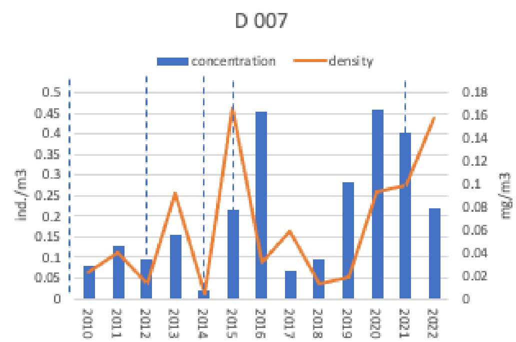
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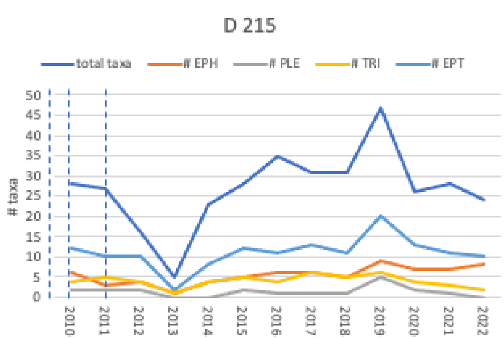
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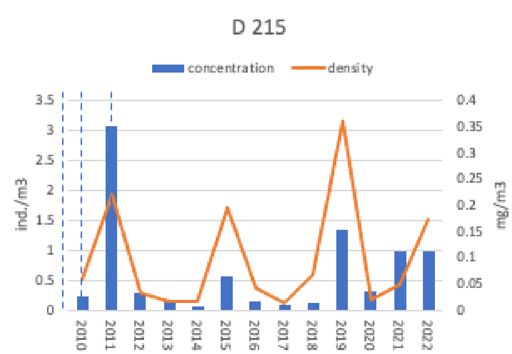
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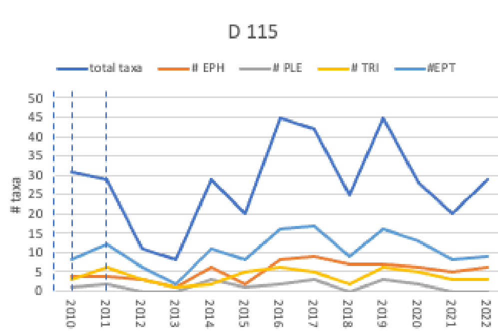
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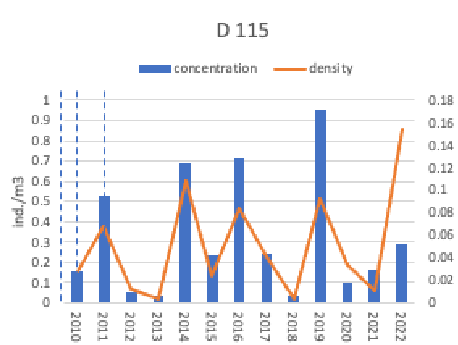
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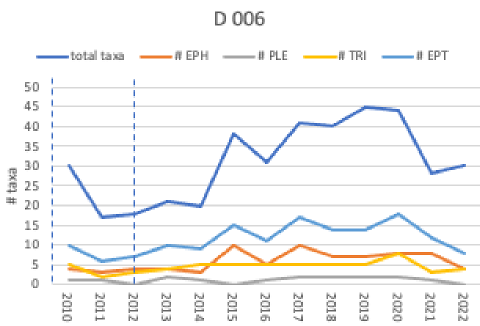
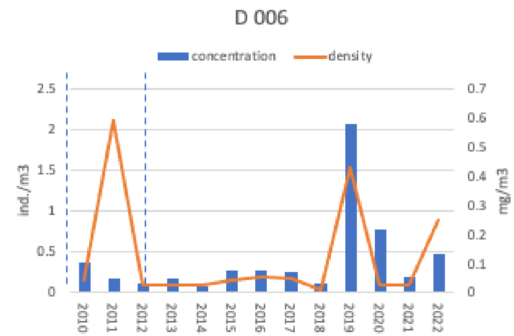
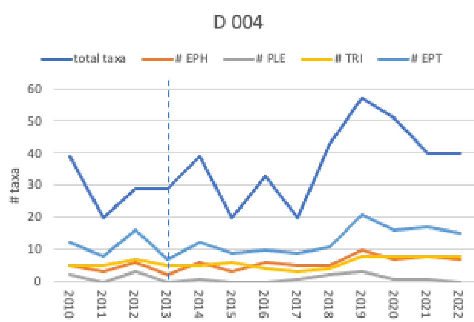
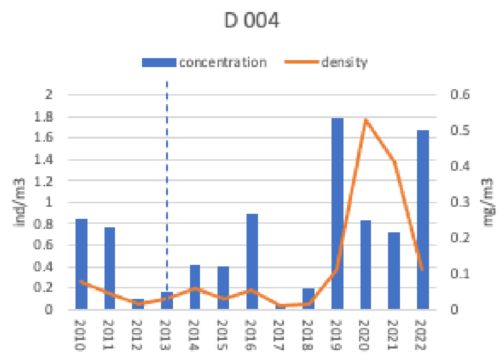
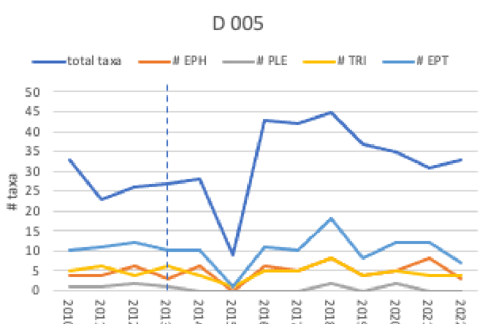
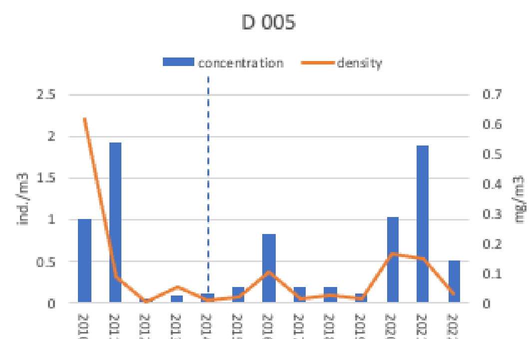




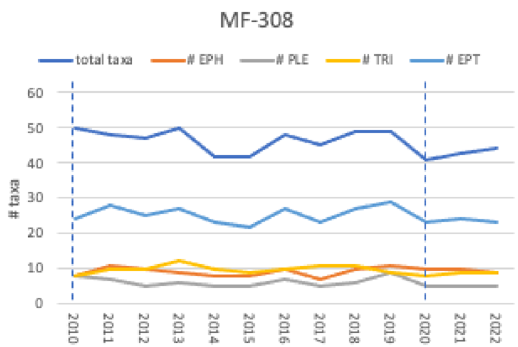








B. Benthic sites, presented in order from downstream to upstream. Dashed line(s) indicates restoration years.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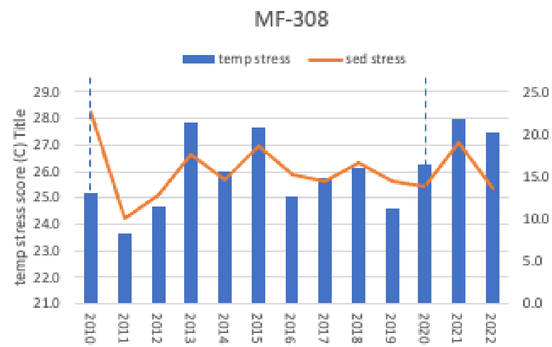
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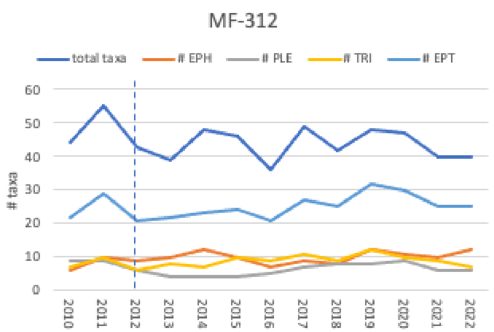
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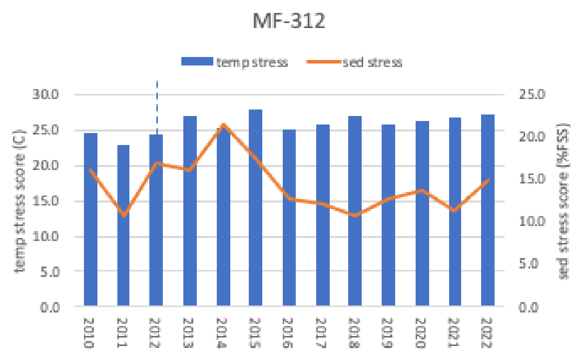
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sed stress
temp stress score (C)
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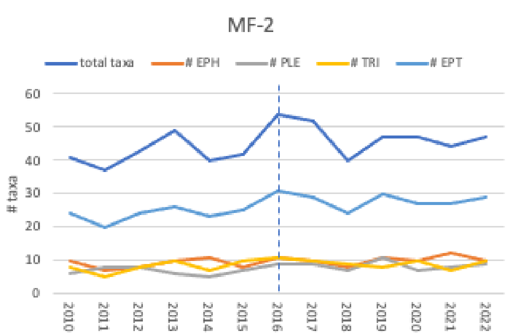
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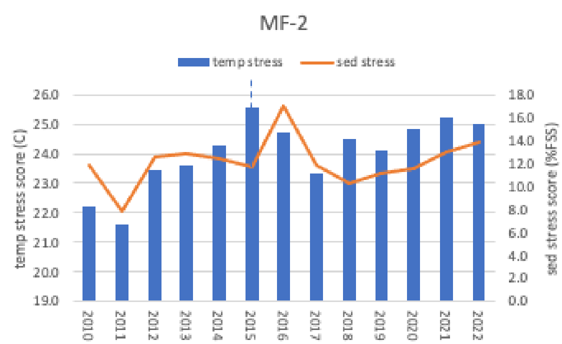
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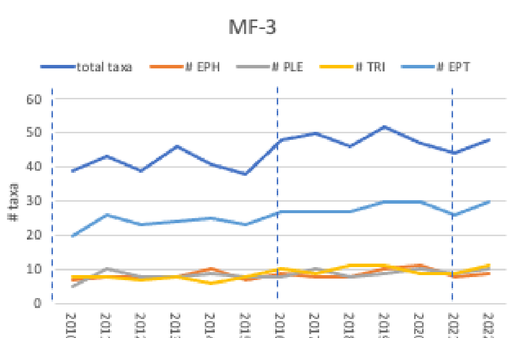
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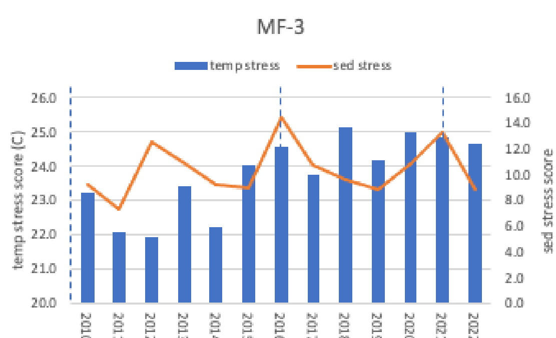
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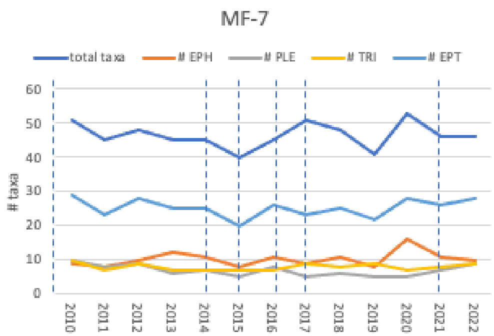
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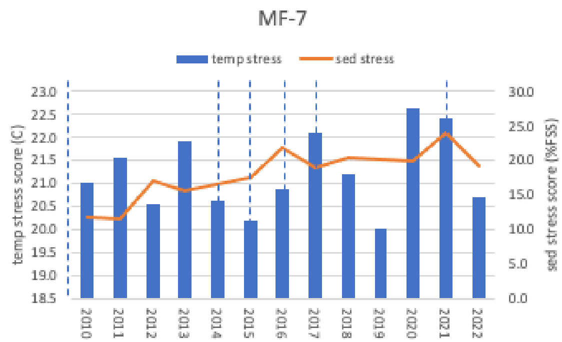
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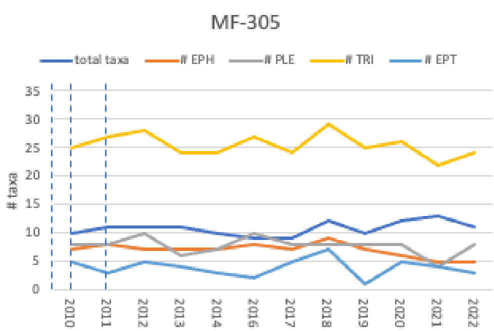
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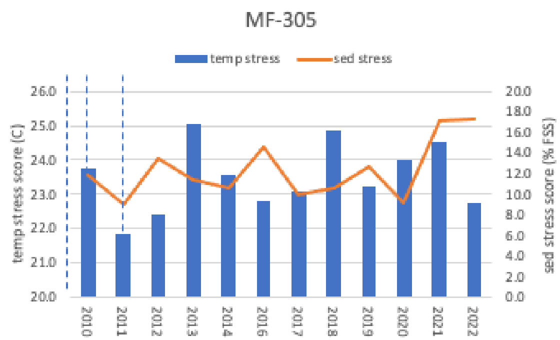
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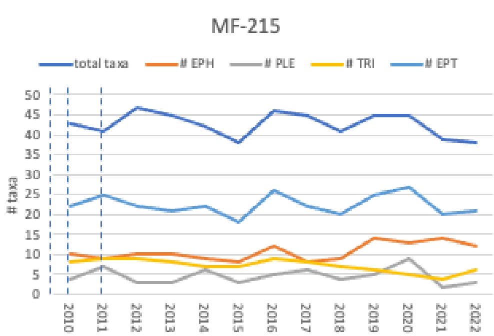
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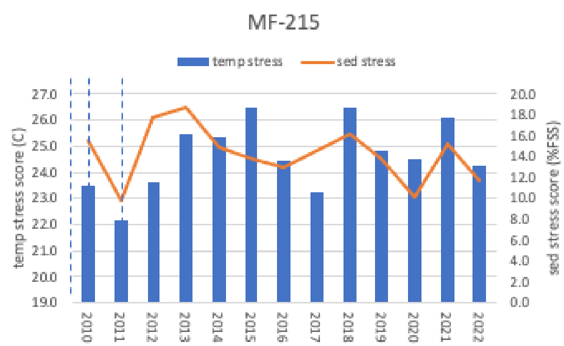
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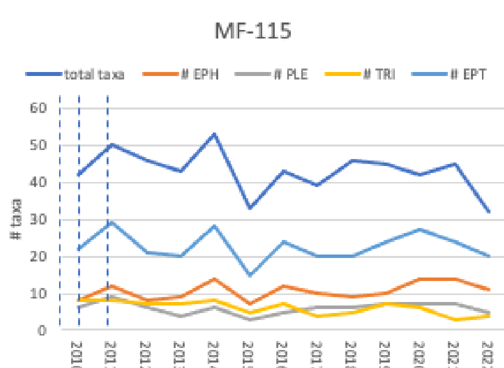
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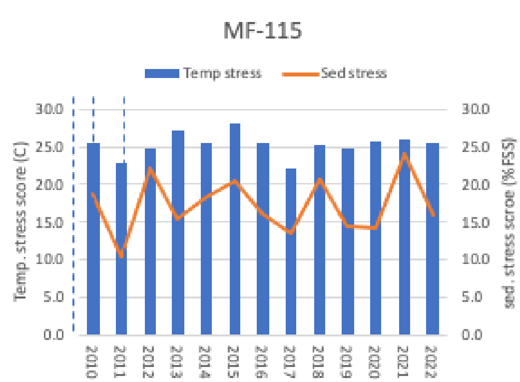
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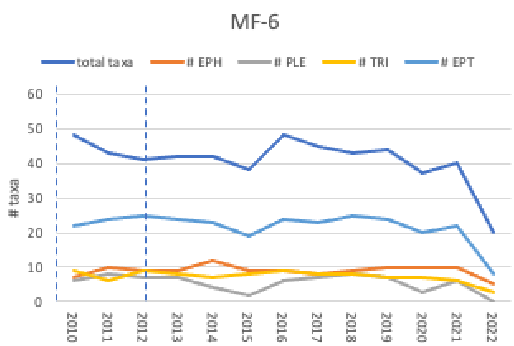

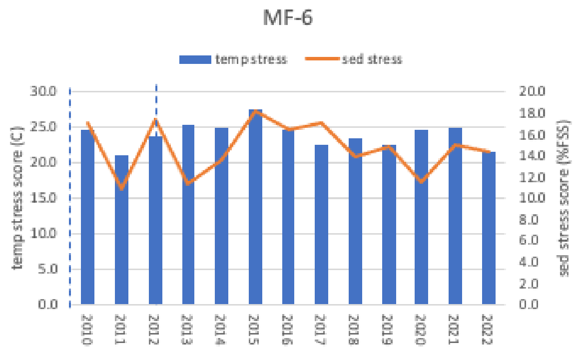

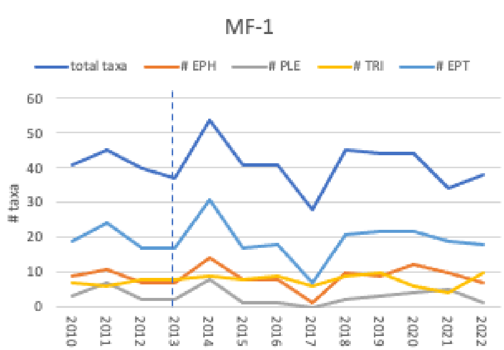

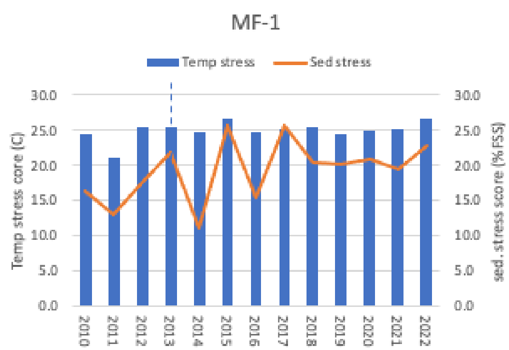

B. PHASE I MACROINVERTEBRATE REPORT (See Appendix C)

CHAPTER 8: Freshwater Temperature Trend in the Intensively Monitored Watershed of the Middle Fork John Day River, Oregon

Stefan Kelly, *CTWSRO,* John Day Oregon

ABSTRACT

Stream restoration is a rapidly maturing field and effectiveness monitoring is critical for informing restoration design, evaluating restoration success, and identifying adaptive management opportunities. Stream temperature is a driver of many ecological processes in aquatic environments and has been identified as a common limiting factor for juvenile salmonids in the Pacific Northwest. This study investigated temperature trends at 86 in-stream temperature monitoring locations in the Middle Fork John Day River, Oregon – a watershed which has been the subject of intense restoration and monitoring efforts over the last 15 years. I performed trend analysis for the months of July, August, and September for data available within and between 2005 – 2021 using response metrics of total degree hours and degree hours above the temperature threshold causing stress to juvenile salmonids. These two metrics were examined using both unadjusted values and by adjusting values for annual variation in streamflow and air temperature. Many sites did not exhibit significant trends during the period of record. Significant trend results for unadjusted temperature metrics were dominated by tributary locations, had a relatively even distribution between increasing and decreasing trends, and fewer decreasing trends located in restoration reaches compared to unrestored reaches. Significant trends for flow and air temperature adjusted metrics were more evenly distributed between mainstem and tributary locations, were mostly decreasing, and a greater proportion of trends were located in restoration reaches. The relatively small number of significant trends observed, compared to the number of tests performed, indicates that the system has minimal increasing or decreasing trends over the period of record. Tributary systems may be more sensitive to external influences (e.g. restoration, natural disturbance events) and annual variation in climate. Lastly, the observation that temperature-mitigating effects of restoration tend to emerge after accounting for stream flow and air temperature, suggesting that restoration efforts currently have less influence over stream temperature than annual climate fluctuations. Benefits of recent, ongoing, and planned restoration may take additional time to be realized and continued monitoring will be necessary to capture long-term effects. Historic trends in stream flow and air temperature, as well as projections of future climate conditions, suggest that restoration effectiveness will need to increase to outpace the influence of background climate effects.

BACKGROUND

Stream temperature is an important characteristic of lotic systems that governs physiological conditions of aquatic organisms including fish (Isaak et al. 2017; Sullivan et al. 2001). In particular, temperature conditions – and especially non-ideal conditions – are reported as affecting the growth, behavior, and survival, of juvenile salmonids (Myrick and Cech 2005; Richter and Kolmes 2005) and can elicit physiological indicators of stress above certain thresholds (Feldhaus et al. 2010). Degraded habitat conditions resulting from anthropogenic activities, including timber harvest, irrigation diversions, channel manipulation, historic mining activity, livestock grazing, and alteration of the historic wildfire regime, are generally understood to contribute to in-stream temperature increases in the Middle Fork John Day watershed (Middle Fork IMW Working Group 2017).

Stream temperature is controlled by multiple environmental factors including discharge (Sinokrot et al. 2000), air temperature (Mayer 2012, Webb 2003), groundwater exchange (Constantz 1998), physical characteristics such as substrate (Johnson 2004), channel morphology (O'Briain et al. 2017), and nearby vegetation (Lyons 2000). Ultimately, these environmental factors interact with heat sources and transfer mechanisms (e.g. solar radiation, evaporation, conduction, etc.) to regulate a given stream's thermal profile (Beschta 1997). Atmospheric warming is expected to alter the behavior of some of these controls under projected climate scenarios, including potential departures from historic flow and air temperature dynamics (Mantua et al. 2010).

The Middle Fork John Day River (MFJDR) has received varying levels of restoration treatment over the last 50 years. At least 184 unique restoration projects were implemented between 2008 and 2022 (Middle Fork IMW Restoration Inventory), and additional restoration activities are documented as far back as the 1970s (Oregon Department of Fish and Wildlife, personal communication).

These restoration actions span a gradient of physical extent and intensity and include project types such as fish passage barrier removals, large wood placement, riparian and upland plant installation, cattle exclusion fencing, deer and elk exclusion fencing, reactivation of historic channels, flow restoration, and floodplain restoration, among others. An individual project can include one or many of these restoration treatments. The Middle Fork IMW (MFIMW) maintains an up-to-date restoration inventory which places restoration actions into activity type categories. There are 14 unique restoration classifications in this inventory, five of which are relevant to this analysis: channel reconfiguration (CR), riparian management (RM), floodplain reconnection (FR), instream habitat improvement (IHI), and bank stabilization (BS) (Middle Fork IMW Restoration Inventory).

While some aspects of temperature variation have been evaluated (Middle Fork IMW Working Group 2017), a rigorous investigation of temperature trend has not been performed. Identifying temperature monitoring sites which have displayed significant trends over the period of record is a critical and informative part of the overall management framework, and may allow for identification of successful restoration actions, prioritization of areas which are increasingly contributing to stream warming, or determining when and where adaptive management is necessary to ensure project success. And importantly, whether restoration activities, and which type, are leading to the desired objective of aquatic species recovery and protection.

METHODS

Data Collection

The Middle Fork IMW temperature logger network is a collective of monitoring stations established as either restoration-specific effectiveness monitoring locations or as reference monitoring locations. Stream temperature data are collected using Hobo Pro-V2 data loggers (Onset Computer Corporation, Bourne, MA) and recorded at 1-hour intervals during logger deployment. Loggers are maintained in place using steel cable attached to a soil anchor, or in some instances by being fixed to a stationary object such as a tree or boulder. A PVC or metal housing is used as a solar shield to prevent measurement errors caused by solar insolation. Loggers are either deployed year-round or are deployed each year in the early spring and retrieved in late fall. The greater logger network is maintained by multiple organizations including the Confederated Tribes of Warm Springs, Oregon Department of Fish and Wildlife, North Fork John Day Watershed Council, and US Forest Service, amongst others, and subtle differences exist protocols used by each agency such as logger intent, deployment timing, and logger QA/QC. While the timing of logger deployment may vary by agency or between years, the duration of deployment for all loggers includes the three hottest months of the year (July, August, September) which are the focus of this study. Most loggers are validated for temperature accuracy using Onset's recommended "ice-bath" QA/QC process (Onset 2022) or by corroborating in-stream measurements with a calibrated thermometer. Logger placement is not specific to habitat unit type and, while typically not documented, the habitat units captured within the logger network are expected to cover the broad habitat unit classifications of pools, riffles, and fast-non-turbulent waters. Additionally, some loggers within the network have been placed in non-conventional habitat units including constructed alcoves to capture restoration-specific effects. Logger locations are distributed between the mainstem MFJDR and its tributaries, and three of the loggers included in this analysis are deployed in a unique habitat unit created through the impoundment of a steelhead-bearing tributary for historic use as a mill pond.

Site Selection for Data Analysis

Data from a total of 252 unique sites are housed in the MFJDR IMW data repository. These individual datasets are of varying annual completeness, have lengths of record ranging from one to fourteen years, are composed of legacy (inactive) and current (active) locations, and undergo Quality Assurance/Quality Control measures which may not be consistent between managing agencies. The variable data quality standards of individual monitoring stations necessitated the development of selection criteria to identify individual datasets suitable for trend analysis. Initial selection criteria were: (1) at least five years of data collection through the 2021 monitoring year, (2) data were collected during the 2021 field season, and (3) the logger had an "active" designation for the 2022 field season. These criteria were developed with the objectives of identifying monitoring locations with lengths of record sufficient for trend analysis; identifying monitoring locations with data from the most recent field season; and identifying monitoring locations where data will continue to be collected, with the intention of selecting sites which will continue to contribute to trend detection through future monitoring years.

Filtering for these criteria produced 86 candidate sites (Figure 3). These sites were reasonably balanced between tributary and mainstem locations with 47 and 39 stations, respectively. There was a higher concentration of candidate sites in the upper half of the MFJDR subbasin. This is a reflection of the distribution of the entire temperature logger network, and also corresponds to the concentration of restoration activities that have occurred in the more-upstream portion of the system.

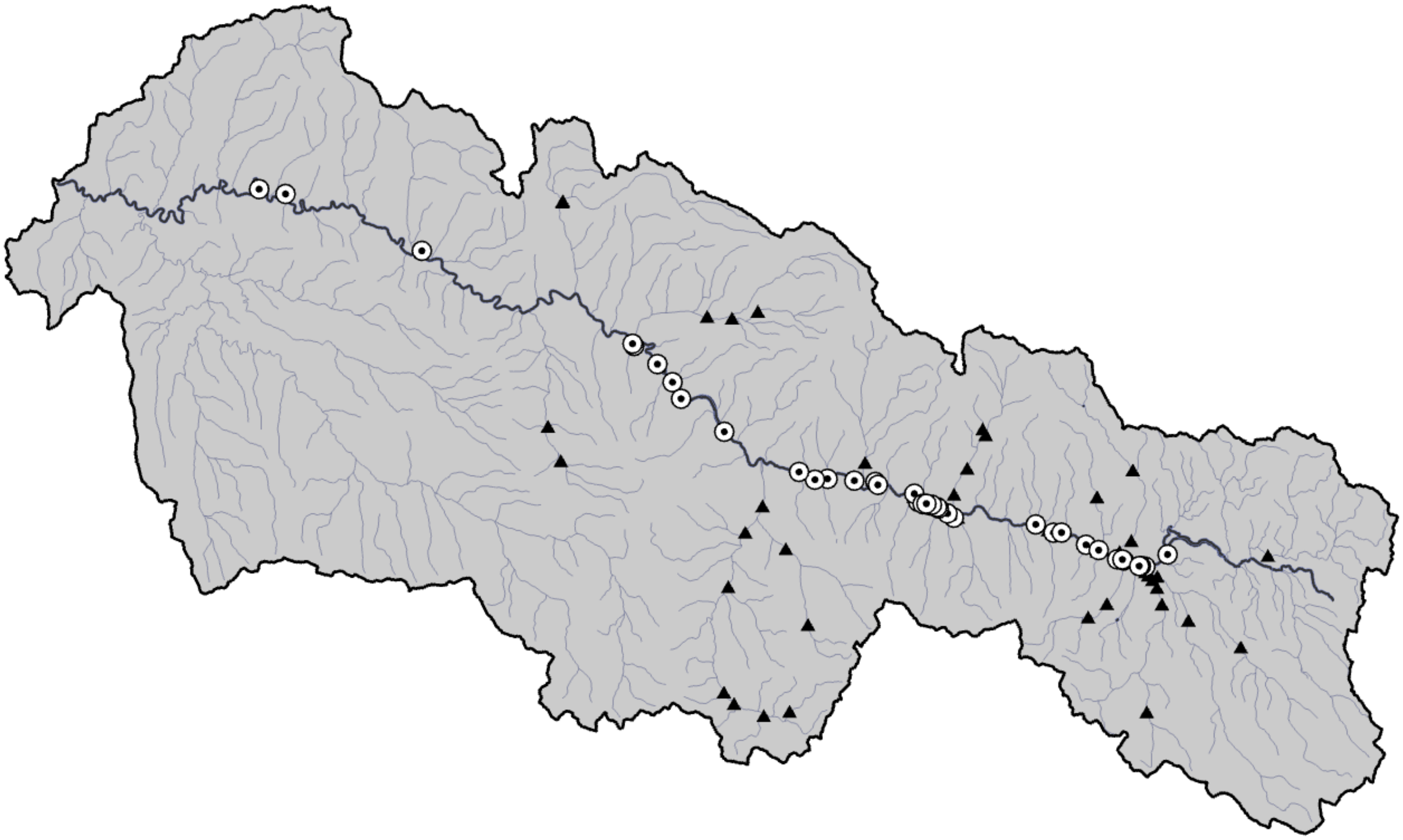

Figure 41. Locations of the 86 candidate sites. Triangles represent tributary sites and circles represent mainstem sites.

Data QA/QC

A standardized quality assurance/quality control (QA/QC) protocol was applied to all temperature data to identify erroneous data, incomplete data, or data behavior indicative of dry channel conditions. These criteria include: total daily temperature change greater than or equal to 12 °C (indicative of dry, exposed conditions, i.e. the logger is recording air temperature instead of water temperature); hourly temperature change greater than or equal to 3 °C (indicative of dry conditions); hourly temperature reading greater than or equal to 28 °C (typically observed in dry conditions, or logger is malfunctioning); hourly temperature reading less than 0 °C (logger is frozen and/or malfunctioning). Any data meeting these criteria were flagged and removed. Daily datasets were required to have a full 24 hourly datapoints to be included in the monthly dataset, and monthly datasets were required to have a full complement of daily datasets to be included in the trend analysis. A minimum of 5 full yearly datasets, either continuous or discontinuous, for a given month and site was required to be included in the trend analysis.

All QA/QC and data analysis was performed using R software (R Core Team 2018). Many candidate sites failed to meet these criteria and were subsequently excluded from the trend analysis. A total of 41 sites met QA/QC criteria for July, 49 for August, and 60 for September.

Response Variables and Analysis Technique

Previous findings published by the MF IMW determined over-summer temperature to be the primary limiting factor for salmonids in the MFJDR (Middle Fork IMW Working Group 2017). Thus, the three hottest months of the year were selected as the focal period for this study. These months are July, August, and September – a period of about 12 weeks in which low discharge and seasonally high ambient temperatures are expected to synergistically contribute to stressful and potentially lethal conditions for juvenile salmonids (Middle Fork IMW Working Group 2017).

Several response variables were considered for trend detection analysis. All of these describe temperature characteristics in unique ways: daily minimums and maximums, magnitude of temperature change within or between days, number of days in which a specific temperature threshold is exceeded, or 7-day average daily maximum (7DADM). The latter metric is commonly used to describe temperature regime in a regulatory context (Sturdevant 2008). While these metrics can be useful for exploring temperature characteristics, they have distinct shortcomings when performing a trend analysis in a salmonid-habitat context. For example, a daily maximum temperature is an index of the most extreme conditions observed at a site but does not describe how long the site experienced extreme conditions, the latter of which being an important regulator of salmonid stress response. The number of days a site experiences above a certain threshold is not always useful; some sites will exceed even relatively high thresholds for the full duration of a given timeframe, while others will not exceed relatively low thresholds. Further, the “days of exceedance” metric will fail to capture subtle trends if they only occur above or below the defined threshold. Lastly, the 7DADM is calculated as a trailing moving-average of daily maximum values which, by definition, requires each 7DADM value to share five datapoints with the 7DADM from the previous day. This type of data-smoothing may obscure trends or introduce unacceptable bias.

I ultimately selected two metrics for analysis: total degree hours (TDH), and total degree hours in exceedance (DHE). A degree hour is defined as the temporal departure, in °C, of the hourly temperature from a given threshold (degrees C/hour). For this analysis, the TDH threshold is 0 °C and the DHE threshold is 18 °C. The DHE threshold is specific to the definition of stressful conditions for juvenile salmonids as given by the Oregon Department of Environmental Quality (Dadoly and Michie 2010). Using these metrics is justified for multiple reasons. For degree hour (DH) metrics, the resulting summation of degree hours reflects both a temperature magnitude and the duration for which those temperatures persist. It also allows for the inclusion of all datapoints within a given day, and all days within a given month, to contribute to the overall summary statistic. This is not the case for metrics that relay on extreme values; for example, maximum daily temperature excludes 23 hourly datapoints from each day and must be averaged to generate a monthly temperature statistic. These metrics either exclude data, as is the case for the former, or smooth data over a period of time, which is true for the latter, and do not adequately characterize the temperature dynamics experienced by fish over a given period of time.

Trend Analysis

Various analysis frameworks have been used to examine the effects of habitat restoration, including Before-After Control-Impact, Control-Impact, and Extensive Post Treatment study designs (Roni 2018, Stewart-Oaten et al. 1986). While these designs can be informative when appropriate, several factors contributed to my selection of a trend-analysis framework for this investigation rather than one of the aforementioned methods.

Many of the sites used in this analysis have received multi-phased or sequential restoration efforts which make it difficult to parse the periods within a dataset into distinct categories for a treatment/control framework. For example, some restoration actions include an initial in-stream component followed by several years of riparian planting and/or maintenance. Other sites are restored through multiple years, and at least one restoration location included in this analysis received in-stream treatment during five consecutive years. This makes it extremely challenging and, in some instances, impossible to identify a reasonable amount of pre-implementation and post-implementation data points to compare.

There are also challenges in grouping monitoring locations in treatment/control sites due to the effect that watershed placement has on local temperature regime. Previous reports (Middle Fork IMW Working Group 2017) have demonstrated that temperature tends to decrease in a downstream to upstream fashion. Sites located in tributary systems and in the more-upstream portions of the mainstem Middle Fork tend to be cooler overall than those farther downstream. This makes it difficult, and sometimes inappropriate, to compare treatment groups.

Lastly, trend can be an informative tool for restoration practitioners, and IMW partners have expressed desire to visualize data in this way (Middle Fork IMW Working Group, personal communication). An important part of the restoration and management framework is to adaptively manage restoration sites and apply additional treatment if necessary to achieve desired goals. Being able to identify if, and at what rate, site conditions are trending towards desired conditions allows restoration practitioners to prioritize adaptive management actions when the trajectory of site conditions does not align with the intended response timeline. Alternatively, knowing if a given site is responding as intended provides valuable information for informing the design of future management actions.

I summarized several descriptive temperature metrics for each site to provide site characterization and context: across all sites and months minimum daily temperature ranged from 2.7 to 16.9 °C; maximum daily temperature ranged from 12 to 26.5 °C; average daily temperature ranged from 7.9 to 21.1 °C; and the average difference between daily maximum and minimum temperature ranged from 0.3 to 7.8 °C.

Linear Regression

I initially used linear regression to examine warm-season water temperatures through time. Each response variable – Total Degree Hours and Degree Hours of Exceedance – was summed within a given month and year. These degree hour totals were used as response variables with observation year being the regressor. A simple linear regression of water temperature does not account for annual variation in climate characteristics, such as air temperature and discharge volume (a function of precipitation) as described below, but is important as a quantitative description of conditions experienced by fish within the period of record at these monitoring stations. Results from simple linear regression analysis are subsequently referred to as “unadjusted”, as they do not consider the climate characteristics listed above in degerming trend. Management practitioners are concerned with how temperature regimes respond in context of climate change and if restoration actions can produce detectable trends towards desired conditions regardless of climate behavior.

Residuals Regression

Annual variation in climate can have significant effects on in-stream temperature. In particular, ambient air temperature and annual flow regime can mask the influence of restoration activities on stream temperature. To address this, the methods described in “Techniques of Trend Analysis for Monthly Water

Quality Data" (Hirsch 1982) were adapted to adjust stream temperatures for annual variation in streamflow and air temperature. Hirsch (1982) describes a methodology for examining trend in situations where a water quality constituent load varies strongly with discharge. In that scenario, annual variation in discharge may obscure changes in underlying processes that affect how a given constituent enters the stream system. Hirsch's approach uses linear regression to find the best-fit relationship between discharge and constituent level, calculates the difference between expected and observed levels, and then applies a Mann-Kendall trend test to the resulting set of residuals.

Building on Hirsch's methods, I performed a multiple linear regression using average monthly air temperature (NOAA 2022) and average monthly stream discharge (USGS 2022) as predictor variables and TDH and DHE as response variables. The residuals from this analysis were used as response variables for a linear regression using time (i.e., observation year) as the predictor variable. Thus, my methodology departs from Hirsch's in two ways: (1) I use multiple regression to control for multiple environmental conditions, and (2) I use linear regression instead of a Mann-Kendall test to examine trend in residuals. Controlling for annual environmental conditions in this way allows for the identification of significant changes in local stream temperature conditions that could result from some external influence other than annual variation in climate and flow conditions (i.e., restoration).

Significance Levels

I used a significance level of 0.1 to identify and organize potentially significant results from regression analyses. This significance level is more liberal than the typical convention and may result in more Type 1 errors than would result from a conservative value such as 0.05 and 0.01. The use of alpha=0.1 is justified because the overarching objectives of this analysis are largely exploratory and the relatively small amount of annual datapoints available for trend analysis, as well as the long amount of time over which restoration effects may occur (Klein 2007), implies that trends may be weak. It is my intention to identify locations where a trend may be occurring, to investigate whether these trends tend to cluster within restoration reaches or differ between tributary and mainstem locations, and to provide insight for guiding future monitoring and analysis efforts. P-values of significant trends are provided in accompanying tables.

RESULTS

Trend Detection – Unadjusted TDH

Unadjusted total degree hours from sites were regressed against observation year within the months of July (n=41 sites), August (n=49), and September (n=60). Six sites were observed to have significant trends in July and August, and four sites had significant trends in September (Figure 2). July had four sites with decreasing trends and two with increasing trends, while August had one decreasing site and five increasing sites. All significant trends for July and August were located in tributaries, with two July trends occurring in restoration reaches and no August trends occurring in restoration reaches. September had two increasing sites and two decreasing sites, of which three were located on tributaries and one on the mainstem. This mainstem monitoring location (MFJD_lwrForrestCABoundary) is located at the downstream end of a heavily restored reach, while the other three sites are not located within, or downstream from, restoration activity.

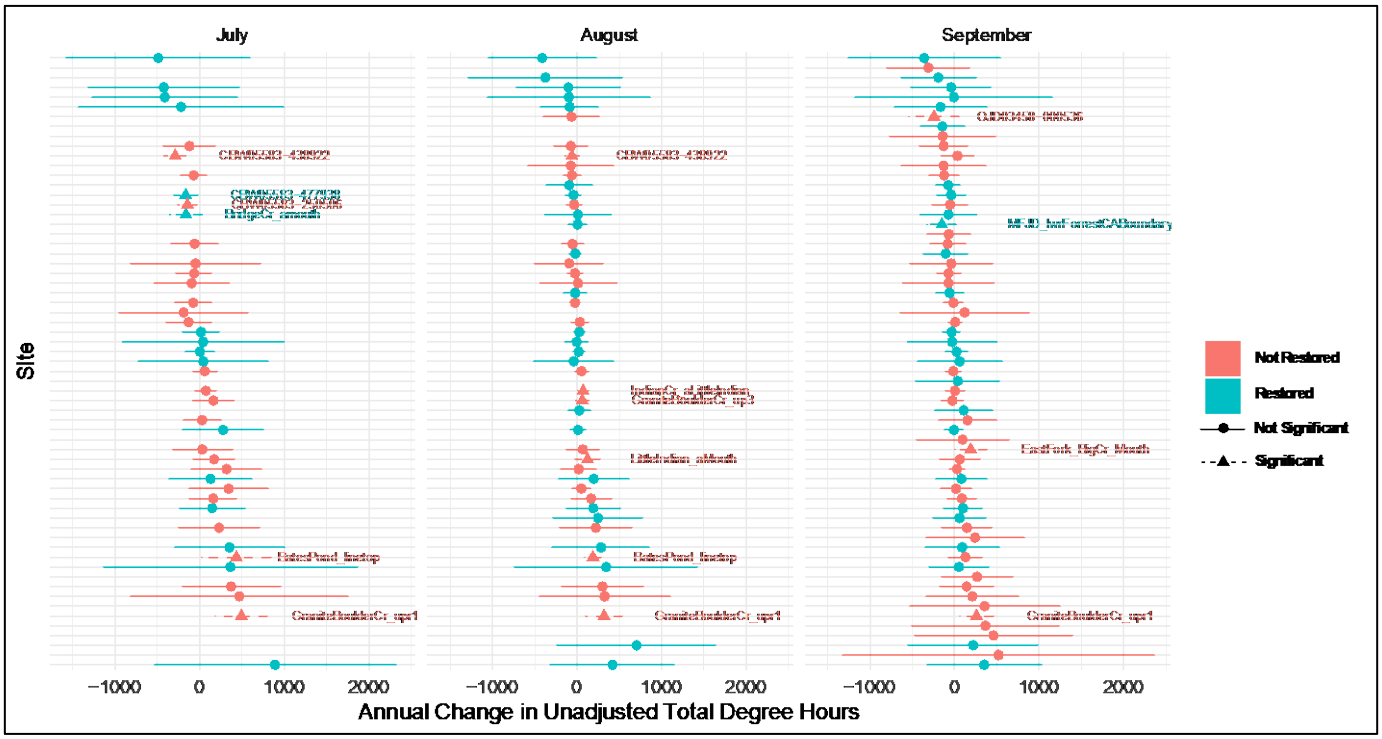

Figure 42. Annual change and confidence intervals in unadjusted TDH for July, August, and September for the period of record. Sites with significant trends are labeled.

Trend Detection – Unadjusted DHE

Significant trends in degree hours in exceedance of 18 °C were found for four sites in July, eight sites in August, and one site in September (Figure 3). July had two increasing and two decreasing sites, all of which were located in tributary stations. One decreasing trend (BridgeCr_amouth) was located downstream from a restoration reach.

Of the eight sites with significant trends identified in August, three showed a decreasing trend and five showed an increasing trend. Two of these sites were located on the mainstem MFJDR (MFJD_TNC_WBoundary, MFJD_ARelocationFS) with the remainder located on tributaries. Two of the eight significant trends were observed within or below restoration reaches. The MFJD_TNC_WBoundary location, which had an increasing DHE trend, was located just downstream of less-intense restoration reach.

One location was observed to have a significant trend in September. This site is located on the mainstem MFJDR and was not within a restoration reach. Many sites were observed to not accumulate any DHE in September: a total of fifteen tributary and two mainstem sites did not exceed the 18 °C exceedance threshold at any point during the period of observation within the month of September.

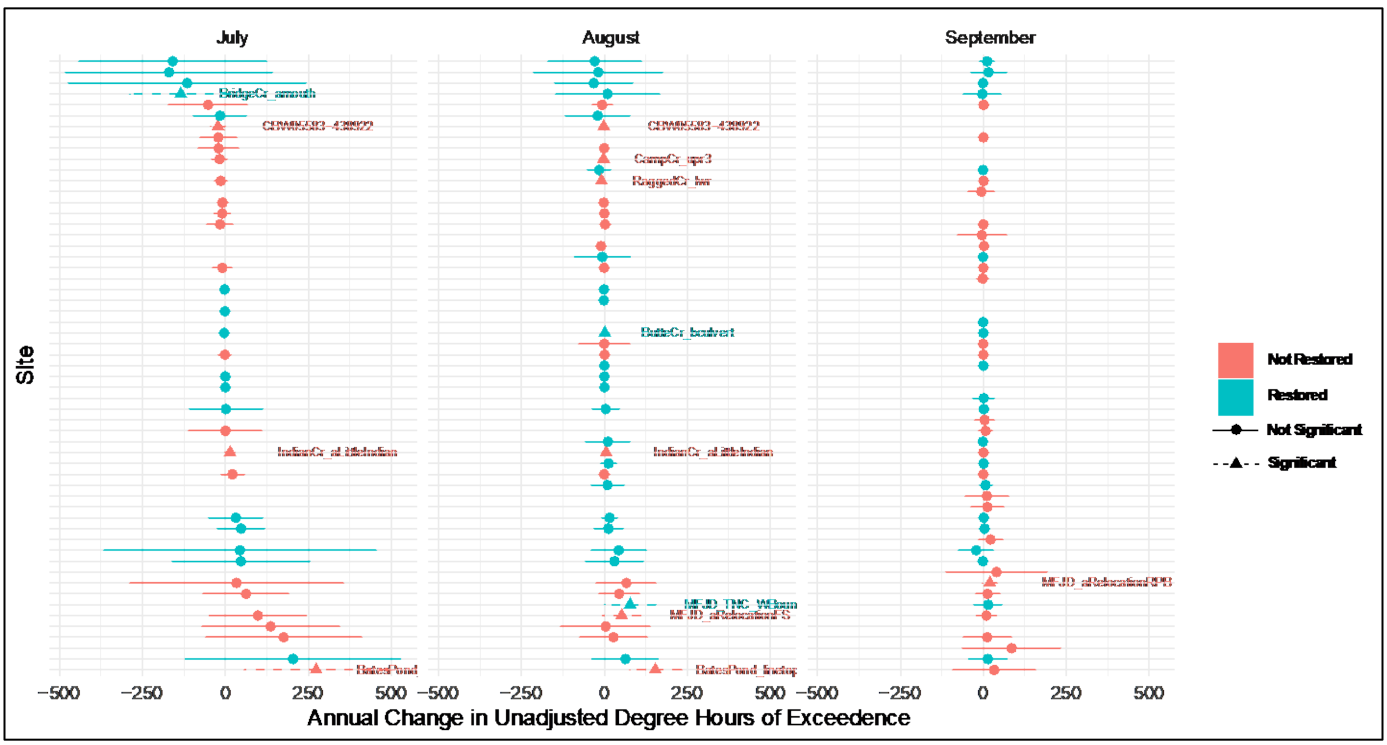

Figure 43. Annual change and confidence intervals in unadjusted DHE for July, August, and September for the period of record. Sites with significant trends are labeled.

Trend Detection – Adjusted TDH

I examined variation in total degree hours after accounting for air temperature and streamflow. The amount of TDH variability explained by flow and air temperature ranged from 22.7% to 99.6% for specific sites and months, and for most sites the regression model relating TDH to air temperature and streamflow produced an R^2 value greater than 0.50. Eleven locations were found to have significant trends for the month of July (Figure 4) all of which were decreasing. That is, stream temperature was declining over time after accounting for effects of air temperature and streamflow. These sites were split between tributary and mainstem monitoring stations. Six of the eleven sites were located within reaches of dense restoration activity, and the remaining five locations were situated outside of any significant restoration activity.

August showed four significant trends of which three were decreasing and one increasing. All of these trends were detected at tributary monitoring sites. One significant trend was located within a restoration reach and was decreasing. September also had four sites with significant trends (Figure 4). Three sites showed increases in adjusted TDH and one showed a decrease in adjusted TDH. Three sites were located on the mainstem MFJDR and one was located in a tributary system. Both of the MFJD_inAlcove4 and MFJD_inAlcove2 locations are within a heavily restored reach and showed a decreasing and increasing trend, respectively.

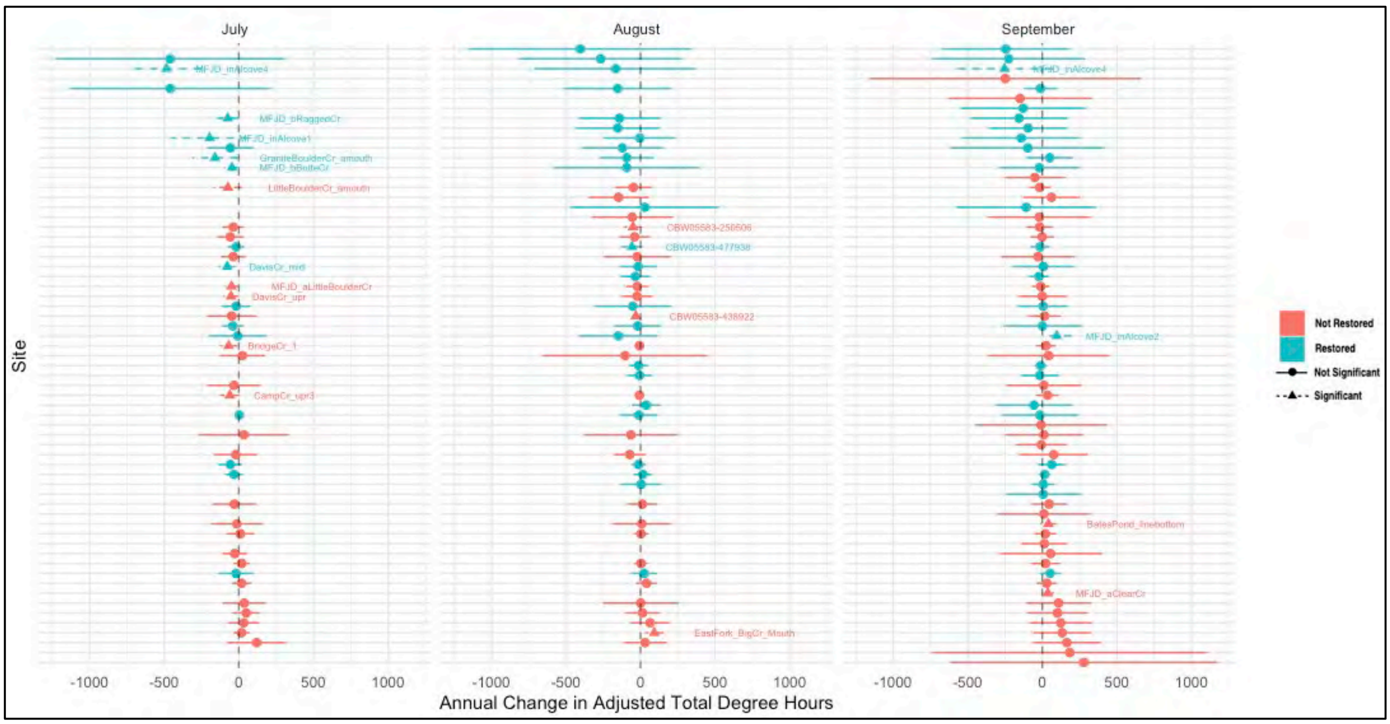

Figure 44. Annual change and confidence intervals in adjusted TDH for July, August, and September for the period of record. Sites with significant trends are labeled.

Trend Detection – Adjusted DHE

Accounting for streamflow and temperature explained less variability in DHE than for TDH, ranging from 5.4% to 85.2%. A significant trend for adjusted DHE was identified in six sites for the month of July with five decreasing sites and one increasing site (Figure 5). Three of these sites were located in the mainstem MFJDR and three are within tributaries. The MFJD_InAlcove4, MFJD_InAlcove1, GraniteBoulderCR_amouth, and CampCr_upr3 locations are near or within restored reaches and showed decreasing trends.

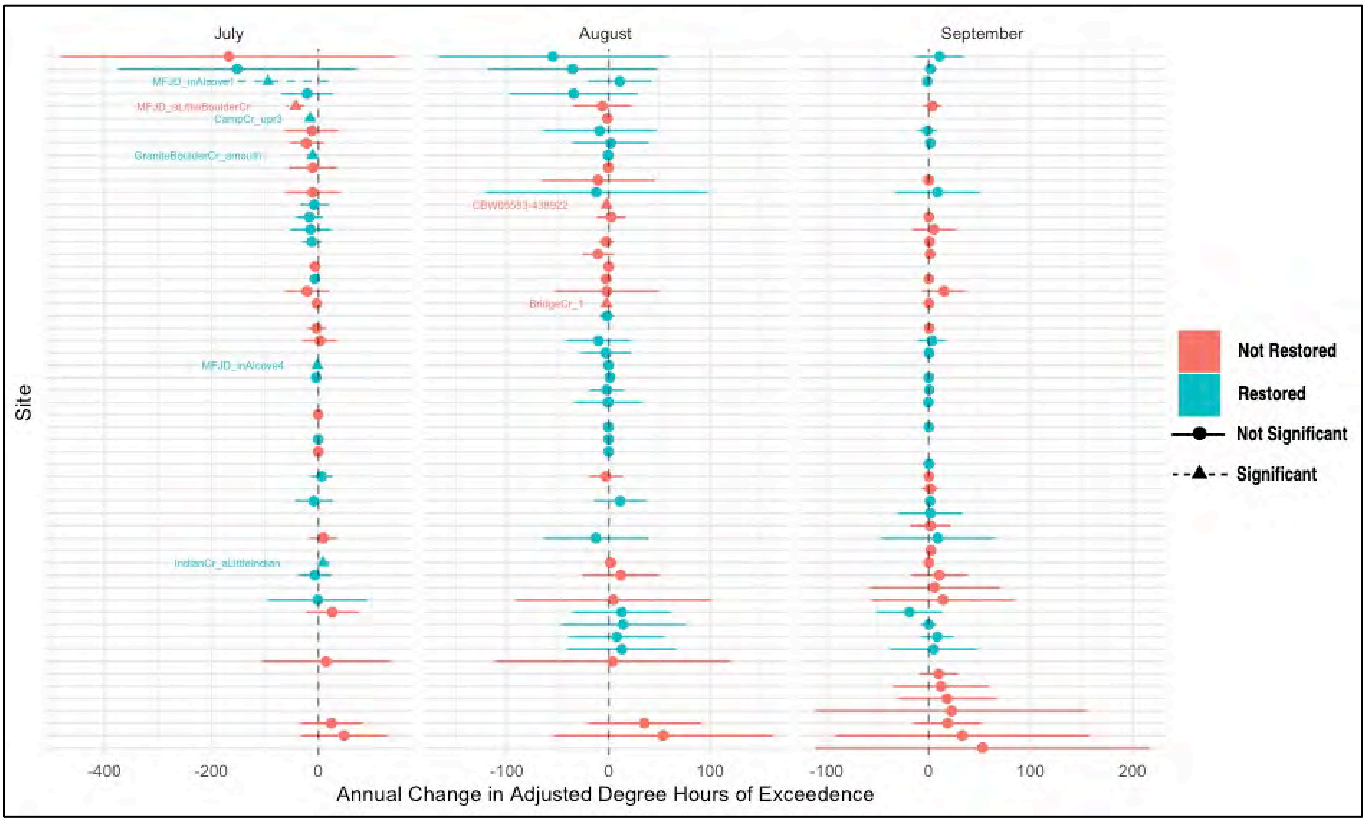

Figure 45. Annual change and confidence intervals in adjusted DHE for July, August, and September for the period of record. Sites with significant trends are labeled.

A significant trend for adjusted DHE was identified in two sites for the month of August. These monitoring locations are in tributaries which have not received restoration treatment. Lastly, no significant trends were observed during the month of September, although it should be noted that the same number of sites were observed to have accumulated no DHE (i.e. no time with water temperature above 18 °C) as were identified in the unadjusted DHE trend analysis.

Historical Air Temperature and Flow

External environmental factors are the primary drivers of in-stream temperature, with air temperature and discharge demonstrated to account for up to 94% of annual in-stream temperature variance (Mayer 2012; Poole and Berman 2001). Therefore, it is important to place any sort of in-stream temperature trend analysis in context of recent and historical flow and air temperature trends. Annual average air temperatures for Grant County, in which the MFJDR watershed is located, has displayed a significant increasing trend of 0.11 °C per decade ($p < 0.01$) since 1930 (NOAA 2022). The period of record (2008-2021) associated with the MFIMW temperature database shows an increase of 0.78 °C per decade ($p < 0.1$). Thus, water temperature data representing the monitoring years 2008-2021 is pursuant to an air temperature regime that is generally warming and may be increasing at a faster rate than the historical average (Figure 6).

Mean annual discharge shows a weak increasing trend during the historical period of record from 1930-2021 (USGS 2022). However, the average annual discharge from 2008-2021 is decreasing – albeit with a relatively high amount of variability – at a rate of 101.8 CFS per decade. Neither of the trends observed for discharge over the historic period of record or the IMW stream temperature period of record are significant, yet the latter suggests conditions which may exacerbate high in-stream temperature conditions.

Overall, both air temperature and discharge between 2008 and 2021 display trends which would tend to contribute to warming in-stream conditions. Any trends observed in unadjusted TDH and TDE during this time should be considered in this context; increasing trends may be the expectation given these external influences, while decreasing or non-existent trends suggest additional factors which may be stabilizing or otherwise mitigating for increased air temperature and decreased flow.

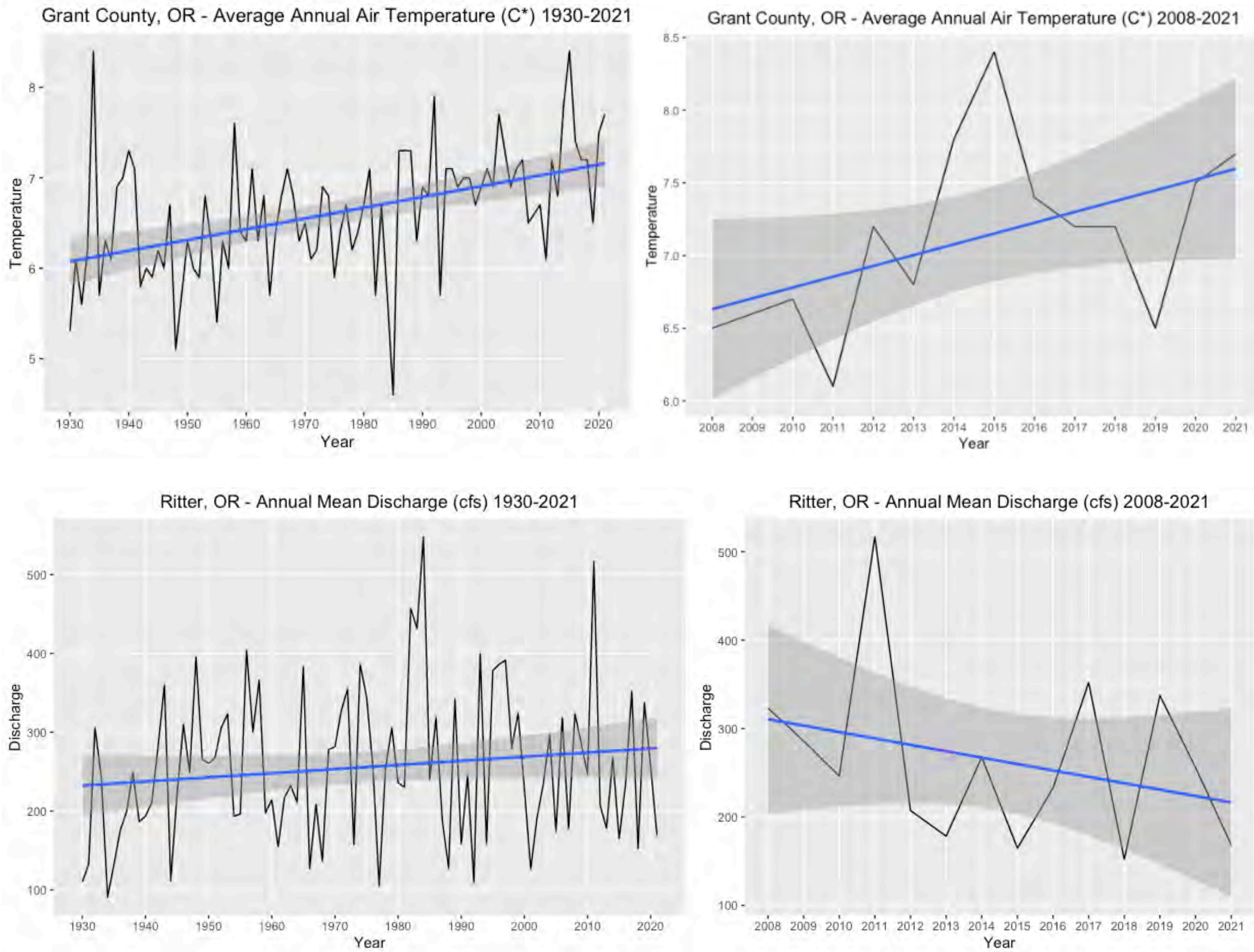

Figure 46. Clockwise from top left: average annual air temperature in Grant County since 1930; average annual air temperature during stream temperature monitoring period of record; average annual discharge at the USGS gauge station in Ritter, OR during stream temperature monitoring period of record; average annual discharge at the USGS gauge station in Ritter, OR since 1930.

Tributary vs Mainstem

Tributary sites dominated the unadjusted temperature portion of the analysis with 15 of 16 significant TDH trends and 10 of 13 significant DHE trends occurring within tributaries. Of the 15 significant unadjusted TDH trends found in tributaries, 6 were found to be decreasing with the remaining 9 increasing. The 10 significant tributary trends in DHE were split evenly between increasing (5) and decreasing (5). Given that all sites are subject to approximately the same external climate influences, which would promote increased water temperatures from 2008-2021, it is logical to presume that some non-climate influence is driving decreasing temperature trends at certain sites. Yet, very few of the these decreasing tributary trends are within or downstream-adjacent to restoration activity. In a meta-analysis, Arismendi et al. (2012) found that many stream temperature monitoring locations with relatively short and recent periods of record exhibited cooling trends, at odds with predictions based on climate trends over the same time period. Passive, natural recruitment and maturation of riparian vegetation is one possible explanation for these observations in non-restored areas, but additional site-specific data would be necessary fully determine the cause of any decreasing trends.

Adjusted TDH trend analysis produced a more even balance of trends detected between mainstem and tributary sites, with 11 of 19 significant trends observed in tributaries. Adjusted DHE showed a similar distribution between site types with 5 of 8 significant trends occurring in tributaries.

Tributaries have previously been identified as providing crucial cold-water input to the MFJDR and as being sources of thermal refugia for salmonids during the hottest portions of the year (Middle Fork IMW Working Group 2017). Examining the distribution of significant temperature trends between tributary and mainstem sites was one objective of this analysis. It seems that unadjusted TDH and DHE trends occur in tributary locations at a much higher rate than in mainstem locations, but adjusted temperature response is relatively even between the two groups. This might suggest that tributaries are more sensitive to external factors affecting temperature than their mainstem counterparts – however, the direction of unadjusted tributary trends is neither overwhelmingly increasing or decreasing. This suggests that climate variation – which has trended towards conditions which would increase stream temperature during the study period – is not wholly responsible for the observed trends in unadjusted TDH/DHE. Additional site-specific data will be necessary to determine why certain tributary sites are increasing or decreasing, especially given that very few of them are located proximate to inventoried restoration.

Restoration vs Control

Examining the proximity of monitoring stations with significant trends to local or upstream restoration was a primary objective of this analysis. Of the 16 significant trends in unadjusted TDH, three were within or downstream-adjacent to restoration activity. One site (BridgeCr_amouth) was downstream of an instream habitat improvement and riparian management project. The other site (MFJD_lwrForrestCABoundary) was at the downstream end of a Tribally-owned conservation property which has seen significant upstream restoration it the form of large wood placement, and riparian protection and enhancement. Both of these sites exhibited decreasing trends for single months (July and September, respectively).

Two restoration sites were observed to have decreasing trends in unadjusted DHE (BridgeCr_amouth and ButteCr_bculvert) and were both located in tributaries. One other site (MFJD_TNC_WBoundary) had an increasing trend in unadjusted DHE for August and is located downstream of an instream habitat improvement project.

Adjusted TDH for July revealed the greatest number of decreasing trends within restoration reaches, with 5 locations in the Oxbow Conservation Area and one location in Upper Camp Creek. The OCA received an extensive, multi-phased restoration effort between 2012-2017 which included channel realignment, large wood supplementation, floodplain reconnection, and riparian protection and enhancement. For the purpose of this analysis, the Oxbow Tailings Project was classified as channel reconnection, although it included aspects of several other restoration types. It is noteworthy that a cluster of significant decreasing adjusted TDH trends occurred in this reach, as the OCA restoration project is by far the most intensive and extensive restoration action included in the referenced restoration inventory. Yet, these trends manifest only after correcting for climate variation, and with the exception of MFJD_inAlcove4, are only observed during July. While this may be encouraging for restoration practitioners, it highlights the importance of stream-temperature drivers that are outside the purview of watershed-specific management. Restoration often targets site or watershed specific limiting factors, yet some of the most important influences of instream temperature occur at the regional (or even global) scale – as is the case with climate change. Additional actions may be necessary to improve mitigation of climate-driven stream temperature influences, but it is unclear how this might be achieved.

Relatively few significant trends in adjusted DHE were observed, but three of these were located within the OCA restoration reach. These three sites – MFJD_inAlcove4, MFJD_inAlcove1, and GraniteBoulderCr_amouth – were found to be decreasing in adjusted DHE for July while also having decreasing trends in adjusted July TDH.

The restoration inventory used for this analysis is current through 2022 and was used to examine results in a restoration context (Table 1). However, future analyses would benefit from incorporating land-use changes or environmental disturbances when attempting to understand the causes of observed trends. For example, changes in USFS cattle grazing rotations would not be captured in the IMW restoration inventory, yet the subsequent effect on riparian condition may have important implications for understanding local temperature trend. Similarly, natural disturbance events such as forest fires may alter landscape features and result in altered stream temperature trends. While these factors may not be directly related to restoration effectiveness monitoring, they may provide important information for prioritizing future restoration reaches or evaluating other management actions affecting stream temperature.

Table 1. Distribution of decreasing trends for several restoration categories found in the MFIMW restoration inventory.

Month and Metric	Floodplain Reconnection	Instream Habitat Improvement	Channel Reconnection	Multiple	Total Decreasing	% in Restoration Reaches
July TDH (Unadjusted)	1	1	-	-	4	50%
August TDH (Unadjusted)	-	-	-	-	1	0%
September TDH (Unadjusted)	-	-	-	1	2	50%
July DHE (Unadjusted)	1	-	-	-	2	50%
August DHE (Unadjusted)	-	-	-	-	3	0%
September DHE (Unadjusted)	-	-	-	-	0	-
July TDH (Adjusted)	-	1	5	-	11	55%
August TDH (Adjusted)	-	1	-	-	3	33%
September TDH (Adjusted)	-	1	-	-	1	100%
July DHE (Adjusted)	-	2	-	2	5	80%
August DHE (Adjusted)	**-**	**-**	**-**	**-**	**2**	0%
September DHE (Adjusted)	**-**	**-**	**-**	**-**	**0**	0%
Total	**2**	**6**	**4**	**2**	**34**	**52%**

Lastly, the type and intensity of restoration employed at a given location may dictate the magnitude of temperature response. For example, the placement of in-channel wood structures may elicit a smaller temperature decrease (Nichols and Ketcheson 2013) than the re-establishment of a riparian canopy (Sugimoto et al. 1997). A restoration inventory that includes metrics which quantify specific aspects of a restoration action (e.g., number of plants installed, number of wood structures installed) would provide critical insight regarding the effectiveness of specific strategies in mitigating in-stream temperature.

CONCLUSION

The results of this analysis are typified by a temperature monitoring network which is, for the most part, not displaying any consistent warming or cooling trend. The vast majority of sites are seemingly stable; of those that presented significant trends, most were only present for a single month. The general stability of temperature is not necessarily a bad thing, and could be indicative of mitigation that is occurring to offset external factors which are trending towards conditions which would be expected to increase temperature. Since 2008, average air temperature has increased and MFJDR discharge has decreased. While this would be expected to produce warming trends at stream monitoring stations, relatively few sites are experiencing significant increasing DH or TDH trends.

While stream temperature stability in the face of warming climate and decreasing flow is better than stream warming, it does not mean that the temperature mitigation goals of managers and restoration practitioners are being met. The objective is, ultimately, to reduce stream temperatures in spite of external factors, and generally this is not happening. Whether this is justification for adaptive management or modification of restoration strategy is under the purview of resource managers. The length of time for which the IMW has been active is relatively small in an ecological context, and some of the responses that would have the greatest effect on reducing stream temperature may require significant time to be realized. In particular, riparian restoration in the Middle Fork John Day is constrained by the growing season and the rate of vegetative loss due to ungulate browse (Beschta and Ripple 2005). Overall, there were relatively few significant trends detected given my use of a 0.1 significance level, and the locations and directions of trends didn't indicate strong, clear patterns. One reason for this could be the length of time over which restoration elicits a response in habitat condition, but could also be attributed to inherently noisy data, background effects, and unrestored reaches diluting the local effects of site-specific restoration.

My study attempted to identify response variables that related to two specific temperature-related metrics – the total amount of energy in the system, and the threshold at which salmonid stress occurs. Other metrics have been used for temperature analysis (Diabat 2014), and communication between researchers and managers/restoration practitioners should continue to fully define useful and relevant response metrics. The response metrics used in this analysis required fairly stringent data-availability and quality standards and ultimately precluded the use of many of the candidate sites. Alternative response metrics (e.g., average daily temperature, average daily TDH) could be adapted to datasets with small amounts of missing data and would expand analysis eligibility to a broader range of sites. The failure of many candidate sites to meet my analysis criteria was due to missing data rather than data quality issues; this highlights the need for diligent temperature logger maintenance, standardized monitoring protocols, and timely QA/QC procedures to identify corrective actions.

Altered landscapes and in-stream habitat have affected the regulation of stream temperature in the MFJDR, and the predicted effects of climate change, have the potential to depress fish distribution and population health in the Pacific Northwest (Beechie et al. 2012). Stream and riparian restoration has been proposed as a method for ameliorating the deleterious effects of climate change (Justice et al. 2017) and significant resources have been allocated towards these efforts in the MFJDR. Continued coordination between data-collectors and data-users will be required to evaluate the cumulative effectiveness of both the restoration and monitoring actions employed through the MFIMW.

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Adaptive Management

Evaluation of Lessons Learned and Recommendations from the 10-year Summary Report

Adaptive management is an important component of any restoration plan, and an adaptive management framework should be incorporated into the IMW structure (Bouwes et al. 2016). During writing of the 10-Year Summary Report we asked researchers and restoration practitioners to share lessons learned and future recommendations based on their involvement with the MFIMW. These lessons and recommendations extended beyond what was learned from study findings; they illustrated how participants would incorporate improved methodologies and strategies into subsequent phases of the IMW process and future IMW programs.

Partners were asked questions like:

- What have you learned based on scoping, designing, permitting, and implementing specific restoration projects in the IMW study area?
- What recommendations do you offer to do things differently in future projects?
- What have you or the MFIMW learned about restoration based on your monitoring and/or research?
- What recommendations can you offer to improve engagement?

Partners and authors provided 86 lessons learned and recommendations for the 10-year Summary Report and several similar themes emerged from the lessons learned and recommendations provided by partners. Therefore, lessons learned, and recommendations were generally organized by theme:

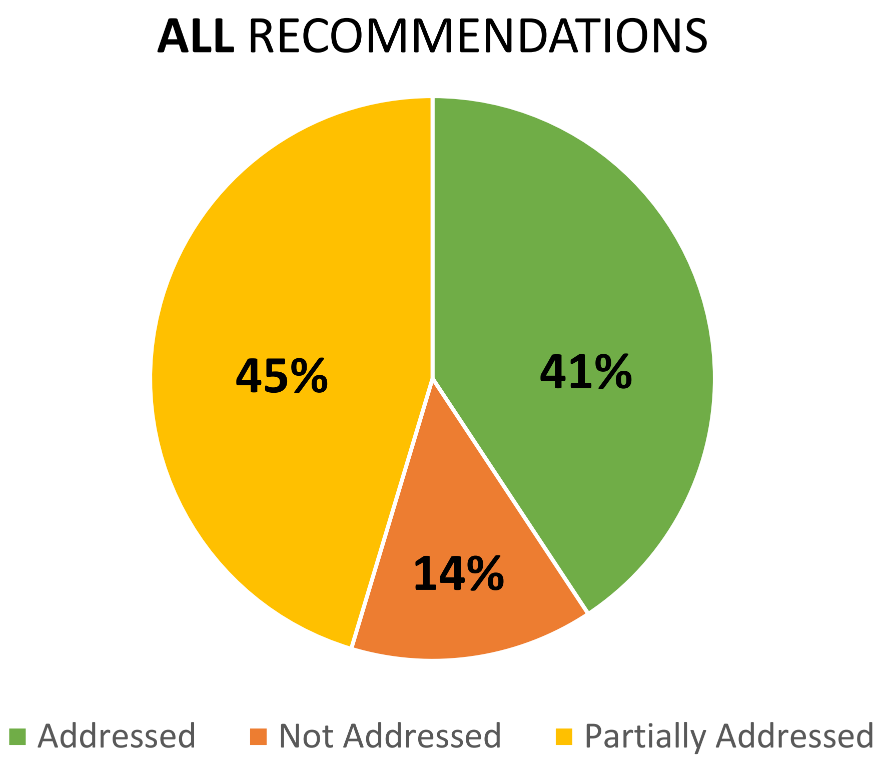
- PLANNING
- MONITORING
- RESTORATION

In this context, PLANNING refers to the planning, facilitation, and coordination of the MFIMW process and group itself. MONITORING refers to data collection, evaluation, and research. RESTORATION refers to practical recommendations for on-the-ground actions.

After completion of the 10-Year Summary Report, report authors communicated these lessons learned and recommendations to partners, and partners began incorporating these into monitoring and restoration within an adaptive management framework.

For this Summary Report we wanted to determine the degree that the lessons and recommendations from the 10-year Summary Report were implemented. To accomplish this, we asked a core representative group of MFIMW partners to rank whether each of the 86 recommendations from the 10-year Summary Report were addressed, partially addressed, not addressed, or are no longer applicable. We summarized responses based on all recommendations, and broke these down into Planning, Monitoring, and Restoration categories.

Overall partners felt that 86% of the recommendations were either addressed or partially addressed/ongoing (Figure 12). Broken down by each category, the recommendations that were addressed or partially addressed/ongoing were 91% for Planning, 61% for Monitoring, and 94% for Restoration. The full list of 10-year Report recommendations and categorizations from can be found in Appendix B.



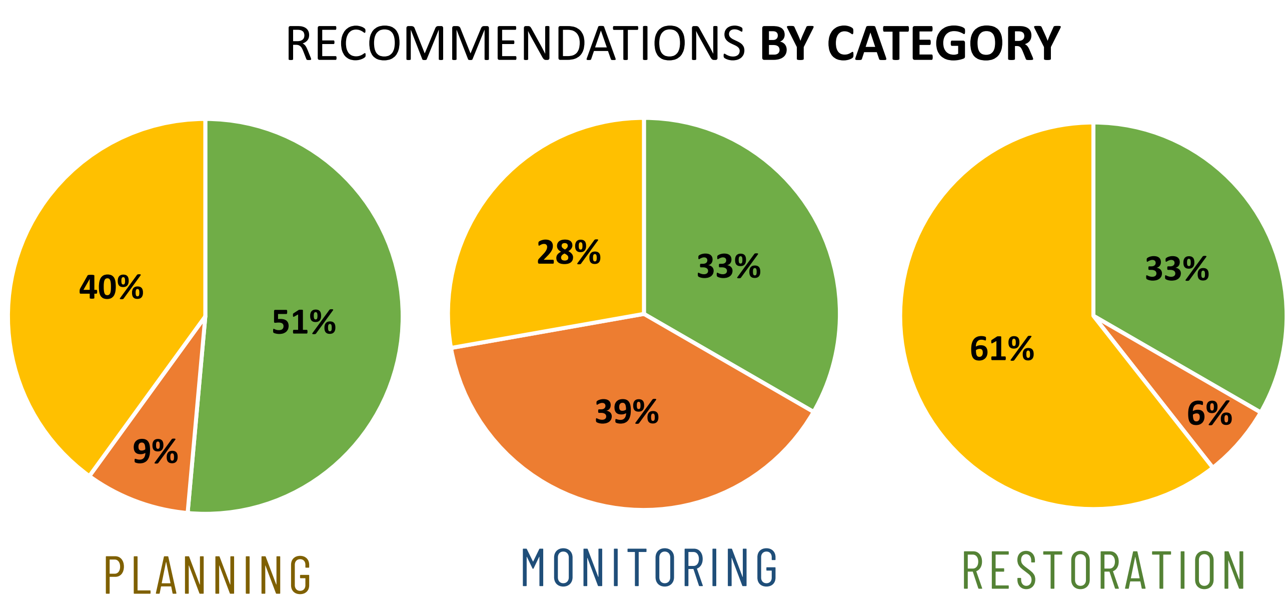


Figure 12. Results of ranking implementation of lessons learned and recommendations from the 10-year Summary Report, 2017.


Credit: NFJDWC
Riparian vegetation along Camp Creek.

Adaptive Management Examples

In addition to evaluating the recommendations that have been addressed from the 10-year Summary Report, we also identified and highlighted six examples of adaptive management actions completed by MFIMW partners. Here we present two examples each of adaptive management in Planning, Monitoring, and Restoration presented as the challenge, recommendation, and adaptation.

Water Temperature Monitoring Coordination

CHALLENGE

An uncoordinated approach to water temperature monitoring and data management was used with the vast amount of data collected in the MFIMW.

This limited the ability of partners and managers to assess water temperature trends and caused difficulties in data processing.

LESSONS LEARNED

Collecting data without linking the data needs to specific management or restoration questions produced datasets without consistent time series needed to document changes before and after specific actions.

Staff turnover and time lags resulted in site selection occurring in a reactive manner, with documentation often lacking.

Lack of communication among groups about their monitoring activities resulted in unnecessary duplication of effort.

The large amounts of data acquired during water temperature monitoring **requires a dedicated data manager** and agreed upon data management strategy.

Conducting quality control measures years after the data was collected was inefficient.

ADAPTATIONS

A water temperature database and software were purchased and a single agency was assigned as the water temperature **data manager** for the MFIMW.

A water temperature monitoring subgroup was formed to:

- Establish clear monitoring goals
- Document reasoning behind site selection
- Ensure consistent field methods and QA/QC measures.

A water temperature monitoring strategy was written and agreed to by the subgroup so that correct data handling procedures were followed. The strategy also assisted collaborators to maintain communication, avoid duplication, and create efficiencies.

Coordination with Monitoring & Restoration Practitioners

CHALLENGE

Communication between the restoration and monitoring practitioners was **not consistent** across all of the efforts.

LESSONS LEARNED

The MFIMW shows that **continued coordination is needed** when multiple agencies are implementing a variety of monitoring and restoration actions across a broad area.

It was challenging to compile all the information needed for the restoration inventory.

ADAPTATIONS

Starting in 2018, the MFIMW and the JDBP held **annual joint meetings to improve communication** across monitoring and restoration efforts. Additional restoration coordination meetings are held quarterly. The meetings helped collaborators learn where monitoring and restoration actions were planned and needed in the future.

Fish Movement

CHALLENGE

Despite increased spawning habitat capacity, **we did not observe a discernible watershed-scale response to restoration efforts within the first 10-years** of monitoring in the MFIMW.

LESSONS LEARNED

Change at the watershed scale takes a long time to realize. With restoration implemented at the reach scale, **monitoring needs to occur at matching reach scales where changes can be detected.**

Detecting initial and often subtle changes in population productivity **may require monitoring population metrics at a finer scale** than the watershed scale.

Watershed **enhancements must effectively address limiting life-stage threats** before other life-stage enhancements will show a detectible increase in population productivity.

ADAPTATIONS

Monitoring efforts were increased to quantify the spatial variation in Chinook Salmon and steelhead productivity metrics and habitat variables across mainstem rearing habitats.

Next, productivity metrics were linked to habitat conditions and water temperature in order to identify areas that were disproportionally impacting populations.

Macroinvertebrate Analysis

CHALLENGE

Previous approaches to analyze the macroinvertebrate data relied on a BACI study design that was **not spatially balanced.**

LESSONS LEARNED

The sample design was not balanced, minimizing the strength of the relationship between the predictor and response variables.

Additional and different analysis was needed to explore if functional group analysis and spatial models would support the hypothesis that management actions are affecting the biotic integrity of the MFJDR.

ADAPTATIONS

A contractor was hired to analyze the drift and benthic macroinvertebrate data sets for temporal and spatial trends and relate these to restoration actions that have been completed. The contractor provided written reports with recommendations for future research, monitoring, and analysis along with results of the trend analysis.

Riparian Plantings

CHALLENGE

Riparian plantings used a wide range of methods with **mixed success.**

LESSONS LEARNED

The restoration practitioners in the John Day Basin Partnership (JDBP) **needed focused conversations** to increase the success of planting projects.

ADAPTATIONS

A workshop was held in November 2023 to review what the JDBP learned through research, monitoring, and field experience with riparian plantings.

The workshop provided an opportunity for participants to make recommendations to improve future riparian planting projects.

A recommendations document was developed from the workshop to inform future work (Appendix D).

Multi-Phase Restoration at Oxbow Conservation Area

CHALLENGE

After five phases of restoration were implemented in the OCA from 2011- 2016, ongoing observations and monitoring **identified areas that were under performing.**

LESSONS LEARNED

Additional large wood placement is needed to supplement wood lost or not included in previous phases. Results from the MFIMW and other observations in the literature indicate that wood placement will increase fish productivity since the primary habitat need (temperature) is met in this particular reach.

Additional grading is needed to lower the elevation of a high terrace that is limiting floodplain inundation at higher flows.

Additional grading is also needed to activate side channel habitats, both perennial and intermittent.

Additional planting is needed due to the survival of installed plants in the mine tailings area being lower than anticipated.

ADAPTATIONS

Future phases of restoration will begin in 2025. The new phases will improve habitat conditions, floodplain inundation and riparian vegetation.

Credit: ODFW

ODFW staff installing PIT tag array near Ritter.

New Lessons Learned & Recommendations

In addition to evaluating progress on the 86 lessons learned and recommendations from the 10-year Summary Report, we also asked partners and authors to provide new lessons learned and recommendations based on their experiences in the last 5 (or 15 years). This section summarizes these new lessons learned and recommendations compiled from the restoration practitioners and each of the MFIMW partner projects, providing an integrated overview of key aspects of the project. Readers should refer to individual reports for details and supporting information.

We paired lessons learned with accompanying recommendations based on what we gleaned from experience. This section presents a compendium of lessons that are not prioritized, but should provide valuable insights for ongoing planning, monitoring, and restoration efforts to make improvements within the MFIMW and similar IMWs.

Credit: ODFW

Sampling for Chinook Salmon fry emergence timing in the MFJDR in early spring.

Restoration Practitioners: Compiled Lessons Learned and Recommendations

(USFS, CTWSRO, NFJDWC and BOR).

PLANNING

	LESSONS LEARNED	RECOMMENDATIONS
1	Community support for a project makes everything a lot easier but is also not always possible to come to consensus across the entire community.	Engage all potential interested parties before designing a project. For instance, for Forest Service projects, all the different resource experts will have to sign off on the project so be sure they fully understand the scope of the project. Effective and open communication can often help.
2	The MFIMW lacks a senior monitoring biologist that acts as a lead monitoring coordinator.	Assign a senior monitoring biologist as a point person or liaison to lead or recommend monitoring actions for each significant restoration action. Ideally, this lead biologist would have some independence from the practitioners.
3	Restoration is a variable process; no two restoration projects are the same. What works for one project might not necessarily work for the project directly upstream or downstream.	When scoping a project, take inventory of what is working in that certain location or a similar reference area. The restoration design should address the concern in that specific location.
4	While endangered or threatened species are critically important, this can cause too much focus at the microscale and ignore the watershed and hydrological processes that create productive aquatic habitats at their peril.	Restoration of large headwater meadows that can store and release water should be considered to increase climate resiliency over the long term.

PLANNING

5	Rushing the design and permitting process with the goal to implement projects every year may result in costly delays and require project alterations during implementation.	Think about budgets in the early stages of restoration project design by allowing ample time to work through the project design details and permitting process. It is important to take the necessary steps to engage local partners to get as many opinions as possible to receive their input before sitting down with the contracted design firm to save time and money.
6	Experienced restoration contractors are limited to perform work in the MFJDR and may not be available to submit a proposal on your project.	Get your request for proposals out early, so there is an opportunity to select a contractor with extensive experience.
7	There is a disconnect between restoration practitioners and researchers. It is challenging to understand what research is occurring on restoration projects. Practitioners are eager to apply research to help inform the restoration scoping and design process.	Create more opportunities for in person meetings and project site visits between the restoration practitioners and researchers to allow time to discuss and apply findings to inform restoration project scoping and design. Each group needs to take the time to attend these meetings to share perspectives to facilitate collective learning.
8	Research data has broadened the understanding of fish habitat use and timing in project areas and is being incorporated into project designs. Project designs are moving beyond adding wood for channel complexity. Designs now include more habitat for higher flows such as off channel habitat, side channels, islands, backwatered sedge flats so there is a variety of habitats available at multiple flow levels.	Research data such as fish habitat use, juvenile fish movement timing, and spatial distribution of fry from redds should be part of the discussion in all project scoping and planning. Be specific about habitat features that a particular life stage is using at different flow levels, so the restoration designers can incorporate information into designs.
9	Permitting and regulatory compliance including but not limited to National Historic Preservation Act (NHPA) Section 106 remains one of the larger hurdles to implement projects at a faster pace and with limited funding resources. A majority of the permitting and regulatory practices were developed as a one size fits all strategy and often not for evaluating stream restoration projects. This results in restoration planners spending a lot of extra time and money to fit projects into the existing permitting framework with very little proven benefits to fishery resources.	Project timeline and budget planning should anticipate permitting delays because the permitting process (associated reviews, surveys and reports) take time and additional resources to be completed.

PLANNING

#	Lessons Learned	Recommendations
10	Historic properties permitting through NHPA Section 106 is one of the biggest hurdles facing restoration on the MFJDR regarding removing legacy railroads grades which restrict floodplain access and reduce juvenile rearing capacity of streams on almost all tributaries.	The permitting process should not deter restoration practitioners from pursuing the removal of legacy railroad grades features on the landscapes to restore processes that form high quality fish habitats.
11	Despite extensive restoration in currently managed conservation properties, a significant numerical fish response at the population scale has not been achieved.	To address current limiting factors, additional actions need to occur beyond existing conservation properties. The acquisition of Phipps Meadows is a positive action, however, addressing limiting factors and application of additional restoration actions on private properties in the upper watershed would provide lift in addressing limiting factors affecting the MFJDR.
12	Subgroups that come together to address a particular monitoring topic can be very effective at refining and answering monitoring questions.	Utilize the organizational structure of the MFIMW Working Group to recruit interested individuals that are willing to participate in subgroups to make progress on a particular monitoring topic. These subgroups can help develop clear monitoring questions and monitoring methods.
13	It is challenging to ensure monitoring experts and restorationists are aware of all of the science produced over the life of the MFIMW.	Create and maintain an archive of publications annually. Work with experts who can assist us in developing an archive or repository such as the Columbia River Inter-Tribal Fish Commission (CRITFC)'s Columbia Basin Fish & Wildlife Library.

MONITORING

#	LESSONS LEARNED	RECOMMENDATIONS
14	A blanket approach to monitoring restoration actions may not yield useful information.	Ensure the monitoring design (sampling frequency, location and parameters) is suitable to measure the restoration project's intended ecological outcomes. Consider the climactic-bio-physiographic regime as well as the implemented restoration actions.
15	The recent information about Chinook Salmon fry emergence is great information, but it could be more specific. Identifying that fry are using sedge habitat is a great start but more details are needed.	Identify what sedge habitats fry are using. For example, are fry using sedge habitat along the edge of a bank while flow is below bank full, is that sedge/grass where flow is over bank, is it backwatered grassy/sedgy depressions, woody vegetation or debris along the edges? Annotated pictures of the example habitat features would convey the details the easiest to engineers/planners.

16	Having definitive recommendations based on fully vetted science completed in the area is really helpful. For example, the modeled effects of different stream shading scenarios to water temperature.	The restoration practitioners need more studies that produce similar, actionable results. For example, outside of water temperature, what is the next most important limiting factor for salmonids? Is it more important to cater habitat to the juveniles or the adults? What habitats are in low supply that improving would have a measurable impact on the fish population? Deep pools for holding? Areas for spawning? Are juveniles food limited in these systems? Are invasive predators having an outsized impact on the juveniles?
17	Research has been focused on the mainstem MFJDR within tribal property areas and channel features.	Expand monitoring and research into the tributaries and the headwaters of the MFJDR into the hydrological processes and ecological aspects of the MFJDR and tributaries including meadow areas. A more robust study is needed to determine tributary contribution/usage by juvenile fish including those streams in the headwaters which may have subsurface flow later in the season. Examine fish growth based on stream water temperature and future climate scenarios to understand if and when tributaries are used by juvenile fish to take advantage of available resources.
18	There has been limited hydrologic and geomorphologic monitoring and research in the MFJDR.	A better understanding of the sediment regime in the MFJDR and the actions which have altered that process would be helpful. Floodplain soil sampling and historical context within wide valley areas would be useful to help plan future restoration projects.
19	The finding that water temperature is the most limiting factor for salmonids and that stream shading is an effective way to mitigate the problem, greatly influenced restoration priorities to focus on increasing stream shade. Water temperature monitoring and understanding juvenile fish movement has helped us prioritize cold water tributaries to enhance juvenile summer habitat.	Increase learning opportunities for presentations from researchers to engage in discussions about how to apply their findings to future restoration actions.
20	There is limited availability of methods and tools necessary to collect drone imagery and to complete the final analyses for interpreting it to measure change associated with restoration actions.	Document the technical tools and resources necessary to conduct drone surveys and imagery analyses when establishing drone monitoring plans and protocols.

	Lessons Learned	Recommendations
21	Drones can be expensive but can simplify project monitoring and can be used for other unintended purposes.	The use of drones should be incorporated into all project aspects, when possible, from long-term photopoint monitoring to outreach materials and videos, aside from the standard monitoring practices.
22	It is important to remember that river systems are dynamic and the conditions that exist one year after the restoration project is completed can look different compared to what it looked like right after it was completed.	Take the time and resources to assess the success of the project yearly and gauge whether adaptive management is needed.
23	Implementation does not always go as designed; issues will always come up that need to be addressed in the field.	Always document changes that were made in the field during project implementation so you can share those adaptations across the partnership and apply them in an adaptive management approach. For example, relocating spring channels and the exact placement and the amount of large wood during project implementation needs to be modified, as site characteristics and restoration objectives dictate.
24	Funders consider a request for funds to adaptively manage a completed restoration project as a failure. As learning continues and systems evolve, more or different restoration actions may be needed.	Funders should view the need to return to a project to implement additional actions is part of the overall project adaptive management process and is not a sign of failure.
25	Project goal development has changed over the years including a better understanding of where a project location fits on the stream channel evolution model (Cluer and Thorne 2014).	Identify what stage the stream channel is currently in to help guide how to restore the channel to a more desirable stage.
26	Past riparian planting projects have had limited success.	Planting potted plants that were allowed to grow for one year prior to planting along with floodplain reconnection and ungulate exclosure fencing restoration actions can increase plant survival. Consider the use of biochar or other agents to increase soil moisture and soil water absorbing capacity on floodplains and meadows. All planting should be completed as a single contract where feasible.

RESTORATION

27	The depth to water table is a key component for vegetation response especially in dredge mined areas because the moisture holding capacity is much different in an un-dredged area.	Groundwater levels should be monitored prior to restoration design to ensure revegetation efforts are successful.
28	Tributary fans to the MFJDR are biological hotspots and have been altered in some manner by railroad grades, mining, homesteading, and roads. These are areas where cool groundwater enters the mainstem MFJDR.	Focus on tributary fans for restoring spawning habitat downstream and improve the ability for bedload to travel through culverts and be deposited on these fans. Areas where there are wide depositional valleys should be the focus for creating multi thread channels for juvenile fish habitat. Consider filling existing incised channels within these areas and utilizing grade controls such as tributary fans and geomorphic pinch points to inundate the entire valley floor and restore riparian vegetation. Consider installing BDAs in these depositional valleys where adequate vegetation is present.
29	Restoration of depositional features and processes are key to restoring vegetation on the MFJDR. Several examples of individual large trees aggrading the stream channel and reconnecting or creating side channels can be found in the mainstem MFJDR. Removal of features that constrict stream channels that are currently in an altered state can restore depositional processes and create a large benefit to support riparian vegetation and juvenile rearing habitat.	Placement of key large individual whole trees should be considered within the mainstem MFJDR to restore depositional features, habitat complexity, connect abandoned side channels and increase vegetation recruitment by allowing fine sediments to settle to support cottonwood germination. Consider installing whole channel spanning wood jams on the mainstem MFJDR within depositional areas to reduce stream power.
30	The spatial patterning of ownership and land management practices creates persistent challenges for restoration practitioners in the MFIMW. Impaired sections of the Middle Fork John Day River, including private land near Bates, OR and Bates Pond, occur upstream of ongoing and future large scale restoration projects on CTWSRO properties.	Watershed location should be kept in mind as a likely important determinant of the efficacy of restoration actions in the MFIMW

Credit: CTWSRO

Willow plantings on the Vincent to Vinegar reach of the MFJDR.

Riparian Planting Lessons Learned and Recommendations from the John Day Basin Partnership Riparian Planting Workshop Summary.

PLANNING

	LESSONS LEARNED	RECOMMENDATIONS
31	Understanding of the hydrologic and soil conditions is needed to ensure riparian plantings will survive.	Assess the hydrologic and soil site conditions prior to designing a planting plan including floodplain connectivity, depth to groundwater, high energy areas/flow rates and soil conditions including disturbance, sediment size, presence of organic materials, and compaction to determine the site's productivity potential.
32	Understanding the site conditions and establishing clear project objectives will inform a successful planting plan.	Plant opportunist species that can colonize quickly. Willows and cottonwoods need to have their roots to the water table and need point bars which is challenging in a sediment limited system. Consider that seral plant succession might be needed, particularly in degraded sites and where invasives are a problem. This means plantings might need to start by seeding grasses and forbs, preferably with locally sourced seeds (or carex mats, where appropriate). Consider planting browse-tolerant, flood-tolerant species. Some shrubs will drown if planted in overly wet conditions (e.g., wet meadow suited to sedges). If stream shade is a priority, consider fast growers over more traditional riparian species.

33

LESSONS LEARNED

Restoration actions along with or prior to riparian plantings are crucial to success. Plantings in less disturbed or improved soils have higher rates of survival and growth.

RECOMMENDATIONS

Connect the stream to its floodplain to raise the water table and provide water to plantings.

Treat weed and competing plants using a variety of approaches including mechanical and chemical removal.

Avoid planting in heavily compacted soils and if appropriate and possible, improve soils prior to planting.

Dig planting trenches in the streambank to get roots to water and improve sediment deposition and retention.

Add roughness to protect plants during high flows and improve sediment deposition and retention.

Protect plants from ungulate, cattle and rodent (beaver, muskrat) browse.

Credit: ODFW

Snorkel-herding juvenile salmonids on the MFJDR.

Steelhead and Chinook Salmon Monitoring and Evaluation.

Kasey Bliesner, Ian Tattam, Nadine Craft (ODFW).

PLANNING

	LESSONS LEARNED	RECOMMENDATIONS
34	The communication and coordination with restoration practitioners is key to long-term effectiveness monitoring.	Continue coordination with restoration practitioners to conduct pre-restoration monitoring in important locations of restoration activity like Camp Creek, Summit Creek and Phipps Meadow. Follow up as restoration is implemented to conduct post-restoration monitoring and analysis.
35	Many monitoring and research projects were referenced in the report as “unpublished.”	Prioritize and create the framework to support the publication of results to document monitoring and research, demonstrate effects of climate, habitat conditions, and restoration actions to reach wider audiences in a timely manner.
36	Investment in stable long-term funding and staffing for monitoring of all life-stages is crucial to the success of the MFIMW.	Continue to present findings to decision makers and funders to maintain support for this long-term monitoring project. Remind funders and partners the importance of a long-term established monitoring program and the benefits it provides to the MFIMW, and partner agencies.
37	The IMW network provides valuable results about fish response to restoration actions in other basins that may be applicable to the MFIMW.	Continue to participate in the Pacific Northwest Aquatic Monitoring Partnership (PNAMP) to help coordinate the broader IMW network and host opportunities to share information across IMWs.

MONITORING

LESSONS LEARNED

38 Restoration to improve high water temperatures is a slow process and it takes many years to affect changes in salmon and steelhead populations.

RECOMMENDATIONS

Continue monitoring to determine if restoration affects salmonid populations as riparian plantings mature and planting techniques improve success of plantings.

FUTURE ANALYSIS RECOMMENDATIONS

39 Investigate if higher densities of steelhead redds in Camp Creek (compared to other MFIMW reaches) are related to restoration.

Investigate other methods for estimating steelhead spawner abundance, including estimates using PIT tags and/or side-scanning sonar. These methods will require dedicated resources (i.e., fully funded staff in addition to current staff).

Investigate environmental variables that may be affecting watershed scale Chinook Salmon productivity using the residual analysis methods described by Warkentin et al 2022. Environmental variables to consider include flow, air temperature and water temperature metrics.

RESTORATION

LESSONS LEARNED	RECOMMENDATIONS
40 Distribution of juvenile salmonids, especially Chinook Salmon continues to be limited by summer stream temperatures.	Future restoration should continue to focus on improving thermal conditions throughout the watershed to increase salmonid distribution. Consider prioritizing restoration to reduce water temperature in areas of low abundance at the upper threshold of temperature limits.
41 Watershed-scale analysis helps us understand the overall picture of population status but may be too broadscale to pinpoint exact variables to affect change.	Investigate habitat or geographic characteristics of Chinook Salmon redd locations in restored reaches to better understand what might be affecting productivity. Use these results to prioritize restoration actions in areas to improve habitat conditions in marginal areas or in reaches where productivity could be increased.

Credit: ODFW

ODFW biologists conduct snorkel surveys for juvenile salmonid distribution in the MFJDR.

Quantifying riverscape productivity to inform limiting factor analysis and guide reach-based restoration goals.

Lindsy R. Ciepiela, Joseph T. Lemanski, and Ian A. Tattam.

PLANNING

	LESSONS LEARNED	RECOMMENDATIONS
42	Some presented findings represent conditions prior to restoration in the MFFCA.	Continue riverscape monitoring to track the effectiveness of recently implemented restoration activities at alleviating density-independent and density-dependent forces impacting population productivity across the riverscape.
43	The data presented in this report represents a small fraction of the available data on the relationships between juvenile salmonids and their rearing environments.	Pursue funding and find creative solutions for personnel to continue to analyze and publish additional available data to better understand the relationships between juvenile salmonids and their rearing environments.
44	Obtaining fine-scale information about steelhead can be challenging given their evasive behaviors during monitoring, complex life cycles and low abundance.	Using Chinook Salmon as an indicator species for steelhead is a pragmatic approach to overcome challenges in obtaining fine-scale information on steelhead due to limited sample sizes. Ensure there is consistent Chinook Salmon monitoring at sites where steelhead monitoring objectives exist.

MONITORING

#	LESSONS LEARNED	RECOMMENDATIONS
45	Tracking the spatial heterogeneity in density-independent and density-dependent factors impacting population productivity complemented and provided finer-scale restoration recommendations than population monitoring alone. Working towards a continuous view of how processes are interacting in the riverscape and across life-stages allows us to identify unique and rare features that are disproportionally impacting populations.	Expand the spatial extent of monitoring to better incorporate the spatial heterogeneity across the riverscape.
46	Restoration effectiveness will be maximized when it is implemented to 1) protect and expand thermal refugia or 2) address temperature limitations in central zones of temperature influence.	Incorporate thermal mapping and monitoring of thermal refugia and central zones of temperature influence into ongoing monitoring to further refine the location of where restoration will be most effective.

RESTORATION

#	LESSONS LEARNED	RECOMMENDATIONS
47	Within the MFJDR, thermal refugia is found at within and at the confluences of cool water tributaries (i.e., Granite Boulder, Vinegar, Davis, Dead Cow and Deerhorn creeks etc.) and in the MFJDR downstream of rkm 95.0 (Granite Boulder Creek).	Specific restoration actions to protect and expand cool-water thermal refugia should include: • Maintain or improve connectivity to cool-water tributaries (i.e, Caribou Creek). • Protect and expand tributary riparian corridors via riparian fencing and plantings near the stream margins. • Protect and expand MFJDR riparian corridors at, and downstream of, tributary confluences via riparian fencing and plantings near stream margins. • Strategically place wood structures to deflect mainstem water and capture tributary water at confluences, with the goal of expanding the volume of cool-water plumes created at confluences. • Increase the habitat complexity (to increase the carrying-capacity) in tributaries and in the MFJDR near tributary confluences. • Downstream of rkm 95.0 maintain the riparian fence and continue to invest in a viable planting strategy, which may include bringing in topsoil to improve the success of plantings.

48	Addressing stream temperature, through passive and active restoration actions, continues to be a vital strategy for recovery of salmonids in the MFJDR. Low parr-to-smolt survival upstream of rkm 109 (Vincent Creek) indicates this area is a central zone of temperature influence.	Implement restoration aimed at alleviating temperature within or upstream of the central zone of temperature influence. Specific restoration actions aimed at shrinking the central zone of temperature influence should include: • Convert long fast non turbulent habitat units into a series of pool/riffle habitat units. • Narrow the channel through island formations. • Reconnect the stream to the floodplain to facilitate the reestablishment of a riparian corridor. • Develop and execute a planting strategy that prioritizes planting species that will grow quickly and provide stream shade (i.e., alders), followed by those that will create a diverse and sustainable riparian community (i.e., willows and cottonwoods etc.).

Credit: ODFW

Kayaks ready to conduct steelhead spawning ground surveys on the MFJDR.

Credit: ODFW

Juvenile steelhead finding cover in and under rocks.

Patterns of Spring Chinook Salmon fry emergence and dispersal across the Middle Fork John Day River basin.

Melody J. Feden (ODFW).

PLANNING MONITORING

49

LESSONS LEARNED

Staff turnover makes data analysis and management very difficult.

RECOMMENDATIONS

Develop a process that lines out communication between outgoing and incoming project leaders to ensure consistency across long term monitoring projects.

50

LESSONS LEARNED

Analysis of fry emergence monitoring was made difficult by the lack of a documented study design. The fry emergence monitoring project was led by a succession of multiple project leaders and analysis of data made difficult by multiple research questions and an unclear study plan.

RECOMMENDATIONS

Have a very clear research question in mind before collecting data, and document and communicate this research question and study plan to ensure continuity throughout the project.

51

In our effort to collect data on many fish, we forget to collect data where there are few fish.

Think about biased sampling when designing. This will be important for determining habitat preferences. Choose sites randomly to prevent sampling biases that can affect results.

FUTURE ANALYSIS RECOMMENDATIONS

When looking at temperature trends and deploying temperature loggers, include winter temperature metrics and data collection to understand how water temperature may affect the development of eggs and fry emergence timing.

Future fry dispersal monitoring should include the tributaries, to learn when they start entering tributaries.

Determine what type of habitat use will increase survival at each life stage.

Quantify available flooded habitat for fry to occupy at different flow conditions to determine if there is a positive relationship with survival by year.

LESSONS LEARNED | RECOMMENDATIONS

RESTORATION

#	Lessons Learned	Recommendations
52	Redd placement can potentially impact parr survival, because redd placement can limit access to resources.	Consider all life stages of salmonids when planning and executing restoration projects.
53	Most fry do not disperse greater than 1 km from redds and have limited ability to swim in strong currents.	Restoration targeting the fry life stage should focus on areas downstream of high-density spawning areas to have the greatest benefit. Avoid developing off-channel habitat where fry can get carried with the flow and not be able to make it back to the main channel. Instead aim to develop off-channel habitat that creates a mix of high and low velocities that allows fry return to habitat they can survive in.

Credit: CTWSRO

Drone imagery of the Oxbow Conservation Area, post 2018 channel reconfiguration.

Juvenile Chinook Salmon Dispersal Patterns Across the MFJDR Watershed.

Matthew Kaylor, Jonathan Armstrong, Lindsy Ciepiela, Melody Feden, Casey Justice, Joseph T. Lemanski, Stefan Kelly, Shawn Narum, Benjamin Staton & Seth White.

MONITORING

#	Lessons Learned	Recommendations
54	Collecting samples from a high proportion of spawning adults, and in particular female adults, is critical. Carcasses degraded rapidly which substantially reduced genotyping success. Further scavengers removed many carcasses or live fish from the river.	A crew of 4-6 solely dedicated to surveying for carcasses daily is recommended during peak spawning.

55 | For juvenile sampling, it is important that the sample design try to achieve a random sample of the population. In other words, the number of parr sampled at each randomly selected site should have been proportional to abundance at that site. | Future studies could consider sampling for a fixed amount of time and processing all individuals, although bias may still arise due to differences in capture efficiency and methodology.

Juvenile sampling should be conducted in the shortest time window possible, as sampling across a wide temporal extent adds challenges to evaluate size vs dispersal relationships.

56 | The dispersal patterns in this study may not represent those in other years with different environmental and biological conditions. For example, snorkeling efforts in summer 2023 and prior ODFW surveys have revealed parr utilizing numerous small tributaries that we were not able to sample in 2021 (e.g., Big Creek, Bridge Creek, Deerhorn Creek). We may have underestimated the downstream extent of parr and parr in unsampled tributaries. | Evaluating dispersal patterns across a broader range of conditions may reveal different dominant drivers of dispersal (e.g., habitat quality vs temperature), which may be an important link between spawning and juvenile rearing distributions among years.

MONITORING

Credit: CTWSRO

Installing plants in the floodplain of the MFJDR.

Planting efficacy and groundwater monitoring on the Middle Fork Oxbow Conservation Area.

Lauren Osborne (CTWSRO) and Matt Kaylor (CRITFC).

	LESSONS LEARNED	RECOMMENDATIONS
57	The 2012 planting efficacy study showed variation in survival and additional recruitment within monitoring plots, whereas the 2021 study showed little survival of installed plants, with almost a fifth of the plants being lethally browsed by small rodents within the first-year post-installment.	Protection of established plants may result in quicker revegetation of the stream than installing new plants and fine-meshed rodent exclusionary fencing may be a necessary addition to protect newly installed plants from small-animal browse, especially when plants are sparse and immature.

RESTORATION

Credit: NFJDWC

Securing water temperature loggers to bricks attached to rebar is one method for water temperature data collection in the MFJDR.

Freshwater temperature trends in the Intensively Monitored Watershed of the Middle Fork John Day River, Oregon.

Stefan Kelly (CTWSRO)

PLANNING

	LESSONS LEARNED	RECOMMENDATIONS
58	Altered landscapes and in-stream habitat changes have affected the regulation of stream temperature in the MFJDR. Coupled with predicted effects of climate change, there is significant potential for further depressed fish distribution and population health.	Continued coordination between data-collectors and data-users will be required to evaluate the cumulative effectiveness of both the restoration and monitoring actions employed through the MFIMW.

MONITORING

	LESSONS LEARNED	RECOMMENDATIONS
59	The response metrics used in this analysis required stringent data-availability and quality standards and ultimately precluded the use of many of the candidate sites.	Alternative response metrics (e.g., average daily temperature, average daily TDH) could be adapted to datasets with small amounts of missing data and would expand analysis eligibility to a broader range of sites. The failure of many candidate sites to meet analysis criteria was due to missing data rather than data quality issues; this highlights the need for diligent temperature logger maintenance, standardized monitoring protocols, and timely QA/QC procedures to identify corrective actions.
60	Some of the responses that would have the greatest effect on reducing stream temperature may require significant time to be realized.	In the MFJDR, riparian restoration is constrained by the growing season and the rate of vegetative loss due to ungulate browse (Beschta and Ripple 2005), and riparian plantings should continue to be protected from browse through use of fencing, caging, and natural protections.

61 Overall, there were relatively few significant trends detected (using a 0.1 significance level) and the locations and directions of trends didn't indicate strong, clear patterns.

Eliciting a response in habitat conditions (that would subsequently elicit a response in water temperatures) requires a long period of time to be realized. Temperature monitoring should continue long-term within an organized framework that takes into account inherently noisy data, background effects, and unrestored reaches diluting the local effects of site-specific restoration.

RESTORATION

LESSONS LEARNED

62 Stream temperatures are not displaying either warming or cooling trends. While this stability in the face of warming climate and decreasing flow is better than stream warming, it does not mean that the temperature mitigation goals of managers and restoration practitioners are being met.

RECOMMENDATIONS

Further adaptive management or modification of restoration strategies may need to be implemented, under the purview of resource managers.

Credit: NFJDWC

Inside of a kick-net after macroinvertebrate sample collection.

MFIMW Macroinvertebrate Community Analysis.

Zee Searles Mazzacano, CASM Environmental, LLC Planning and Michael B. Cole, Cole Ecological.

PLANNING

LESSONS LEARNED

63 The macroinvertebrate sampling program was not established with a Before After Control Impact (BACI) design in mind, yet attempts have been made to use the data with this design. Too few pre-restoration data and control sites obviate the ability to effectively use this design.

RECOMMENDATIONS

Future analyses should continue to focus on trend monitoring and not assessing before-after/control-treatment effects.

64 Drift sampling has been performed without a clear understanding of how the data are to be used in ecological assessments, and drift data are less useful for measuring ecological condition than are benthic data.

Omit drift sampling from the MFIMW monitoring unless there is an explicit intended purpose for using them for assessing juvenile salmonid food abundance.

LESSONS LEARNED	RECOMMENDATIONS

65 Physical habitat and water quality data for the MFJDR are scant and are not collected at each macroinvertebrate monitoring site. PIBO habitat data are not collected at the same scale or frequency as the MFIMW macroinvertebrate data, and therefore are of limited utility for assessing habitats changes at individual MFIMW monitoring sites.

Consider adding physical habitat assessment to occur concurrently with benthic macroinvertebrate sampling. Do not attempt to use PIBO habitat data to assess changes at the reach scale.

Consider adding continuous temperature monitoring (i.e., HOBO loggers) at each macroinvertebrate monitoring location to better characterize the thermal regime. Because habitat assessment increases the effort and cost of monitoring, assessment could focus on select habitat attributes that are most likely to change more rapidly in response to restoration and that are more likely to elicit a response from the macroinvertebrate community. Such attributes could include selected substrate and riparian metrics such as particle size/embeddedness, percent composition of habitat types in the sampled reach (riffles, pools, glides, etc), % stream shading (measured with a densiometer), % cover of native and non-native plants in riparian zone at different levels (groundcover, understory, canopy).

Next Steps

Building from the long list of lessons learned and recommendations summarized in this document, the MFIMW Working Group will prioritize recommendations for Planning, Monitoring, and Restoration. One area of interest is to examine and implement new lessons learned and recommendations that have emerged in this 2024 Summary Report and how many are consistent with the 10-Year Summary Report. This can help identify areas of new learning that should be explored over the next five years.

Many participants are interested in developing an outreach strategy to report the MFIMW key findings to various audiences. These outreach efforts will likely span over a period of time to receive adequate input and develop the appropriate approach and materials to inform the different audiences that are identified. In the short term, partners presented MFIMW findings at the River Restoration Northwest Symposium and Oregon American Fisheries Society (ORAFS) annual gathering in February 2024 and plan to present again in 2025. Efforts will be made to update the MFIMW public website with new content. The September 2024 joint MFIMW/JDBP meeting will focus on sharing findings and recommendations that can guide restoration actions in the coming years. Finally, the MFIMW will work proactively with NMFS, the Pacific Northwest Aquatic Monitoring Project (PNAMP), and other IMWs in the PNW to reflect on the findings across the broader IMW network and determine how the MFIMW moves forward to provide needed information for decision-makers and practitioners.

With continued funding the MFIMW Working Group will continue to monitor priority data sets including Chinook Salmon and steelhead life cycle, water temperature, streamflow and benthic macroinvertebrates. An effectiveness monitoring effort has been recently initiated to monitor the effects of riparian exclosures on vegetation and thermal conditions in Camp Creek. This monitoring is a collaborative effort with the PNW Research Station, USFS Malheur National Forest and NFJDWC. In addition, a field crew from the USFS PNW Research Station in Logan, UT will be returning in 2024 to complete its fourth monitoring event to track watershed scale stream habitat condition changes following the Pacfish/Infish Biological Opinion Effectiveness Monitoring Program (PIBO) sampling methods in the mainstem and Camp/Lick creeks. In addition, ODFW will be analyzing existing juvenile steelhead abundance and distribution datasets to understand what the data is revealing and determine the future pathway for this type of monitoring.

Conclusion

The network of IMWs was established with the primary goal of testing whether physical habitat actions could create salmonid population responses. Results across the IMW network have varied. After 15 years of paired restoration and monitoring within MFIMW, we have observed that despite worsening global environmental conditions, stream temperature and salmonid freshwater productivity have been maintained, but have not increased (as hoped). While the Middle Fork IMW has achieved a putatively neutral result- in that salmonid productivity has neither increased, nor decreased, the process of IMW restoration implementation and monitoring has still generated extensive valuable learning, as demonstrated by the MFIMW 10-year Report and the Adaptive Management section of this report.

At this juncture in the IMW, it's important to assess our position in the adaptive management loop, and also re-evaluate whether continuing to answer the original question (Can habitat actions improve salmonid productivity?) is the only direction for this IMW. With restoration actions thus far maintaining temperature and productivity, but not changing either to desired levels, it's reasonable to hypothesize that a) more extensive and intensive channel reset processes may be needed to move out of the current state of watershed processes such as sediment routing and water storage, and advance floodplain reconnection, and b) multiple restoration work iterations will likely be needed to help achieve a more functional stream system. We were able to formulate these conclusions and work within an adaptive management framework because of the suite of low to medium intensity restoration projects that have been implemented over the past 15 years, allowing us to test the effect of those projects on habitat and salmonid population productivity. Hence, iteratively working through those projects has been an important contribution to our learning process. In this vein, preexisting restoration actions in a stream reach shouldn't remove those reaches from consideration for future restoration actions that may either build upon, or broadly alter, prior restoration actions. Potential large-scale restoration actions to recreate the historic water and sediment routing processes needed to achieve broad-sense recovery for salmonids will require increased floodplain connectivity and interaction.

Floodplain connectivity projects inherently create a high likelihood of complex socio-cultural interactions, due to their high visibility and often private land ownership. Further acknowledging this social component is a key component of the future of the MFIMW—if people are the slowest part of this system, we have to actively engage and change broad scale sentiments about floodplain restoration and both the timeline and magnitude of actions needed to achieve biologically meaningful floodplain restoration. Education and outreach in the next phase of the Middle Fork IMW has to be viewed as a mechanism to shift from reactive or opportunistic restoration to proactive and structured restoration through intentional engagement. Demonstrating potential co-benefits of restored floodplains and human socio-economic needs will be an important component of this process. By demonstrating that co-existence can occur, floodplain restoration would then become an exportable technology; which is similar to one of the original missions of the IMW network—identifying the mechanisms creating fish population response, so that those mechanisms/actions can be exported to watersheds in similar ecoregions that are not closely monitored. To accomplish this, the outreach and education component of the Middle Fork IMW needs to be funded and implemented on a pace and scale that is comparable to the physical habitat restoration and monitoring because this opens the door for more and better restoration on both public and private lands (including working lands).

While the original IMW network question was focused on salmonid population response to habitat restoration, an additional question could ask if floodplain reconnection and rebuilding in the Middle Fork IMW can keep up with a shifting baseline of global environmental conditions. Advancing the MFIMW into a socio-ecological demonstration and test of the hypothesis that increased floodplain reconnection can offset and/or reverse the effects of changing global climate conditions will require expanded coordination and outreach on a large scale. The MFIMW will work proactively with NMFS, the Pacific Northwest Aquatic Monitoring Project (PNAMP), and other IMWs in the PNW to reflect on the findings across the broader IMW network and determine how the MFIMW moves forward to provide needed information for decision-makers and practitioners. Beyond the IMW network, we need the capacity to build other systems into the IMW network for learning. Advancing the MFIMW into a socio-ecological demonstration project, not just a biological experiment, will require more resources to look across jurisdictions for learning— that is, look at all similar restoration projects, not just IMWs. The mechanisms and resources exist (e.g., PCSRF restoration project database, NWFSC PNSHP database). Reactivating and capitalizing on these resources will require support and staff capacity to engage in the broad-scale coordination (e.g., engage with restoration practitioners from Whychus Creek, Upper Grande Ronde, etc.) needed to accelerate learning from all ongoing stream restoration and expand the Middle Fork IMW into a vehicle guiding floodplain and fishery resiliency projects across this eco-region.

Appendix A: Restoration Inventory

[1] **Lead Entity:** See Abbreviations list at the beginning of the document.
[2]**Year:** indicates the year construction began.
[3]**Restoration Activity**: **BS:** Bank stabilization; CR: Channel reconfiguration; FP: Fish passage; FR: Floodplain reconnection; FI: Flow increase; IHI: Instream habitat improvement; RM: Riparian management; UM: Upland management.
***** Amount to be included in phase II.
n/a: amounts not available

Restoration Project Name	Lead Entity[1]	Year[2]	Restoration Activity[3]	Total Cost
Oxbow Aspen Fencing Project	CTWS	2017	RM	$ 5,583
Granite Boulder Riparian Planting	CTWS	2017	RM	$ 66,000
Granite Boulder Grazing Program Management	CTWS	2017	RM	$ 10,000
Vincent to Caribou: Phase 1	CTWS	2017	IHI; BS	$ 398,000
Davis Creek Rip Rap Removal and Large Wood Placement	USFS	2017	IHI; CR	$ 23,447
Phipps Powerline ROW Coarse Wood Project	USFS	2017	IHI; FR	$ 6,000
Forrest Conservation Area Riparian Planting	CTWS	2017	UM; RM	$ 17,000
Holmes Property Fencing	ODFW	2017	RM	$ 26,500
TNC Riparian Planting and Maintenance	CTWS	2017	UM	$ 27,500
Wiwaanaytt Creek Restoration and Fish Screens Project	USFS	2017	FP; IHI	$ 113,000
Bear Creek Restoration Large Wood Placement/Reconnect passage	USFS	2018	FR; FP; IHI; RM; UM	$ 152,263
Wiwaanaytt Creek and Meadow Restoration Phase III and 3b	USFS	2018	FP; IHI; FR	$ 442,008
Big Creek Reach 4	USFS	2018	FR; IHI	$ 209,869
Granite Boulder Grazing Program Management Lease Renewal	CTWS	2018	RM	$ 7,000
Bridge Creek ODOT Hazard Tree Removal Highway 26	USFS	2018	IHI	$ 10,000
Wiwaanaytt Creek Fish Passage and Screens Project	ODFW	2018	FP	$ 177,150
Long Creek-Neal Restoration/Large Wood Placement	USFS	2018	IHI	$ 12,332
Butte Fence Project	USFS	2018	RM	$ 28,824
Granite Boulder Browse Proof Caging	CTWS	2018	RM	$ 110,000
Clear Creek Restoration	USFS	2019	FR; IHI; RM	$ 110,000
Camp Valley Restoration Phase 1 (Reach 5)	USFS	2019	IHI; FR; RM	$ 381,216
Oxbow Conservation Area Riparian Planting	CTWS	2019	RM	n/a

Restoration Project Name	Lead Entity[1]	Year[2]	Restoration Activity[3]	Total Cost
Cottonwood Creek Culvert FS 36 Road	USFS	2019	IHI	$ 180,124
Bear Creek Restoration Riparian Planting and Exclosures	USFS	2019	FR; IHI; FP; RM	$ 4,000
Summit Creek Culvert FS Road 2622	USFS	2019	FP	$ 14,100
Deadwood Creek Culvert FS 4560 Road	USFS	2019	FP	$ 94,298
Galena Tree Tipping -on CTWS	USFS	2019	UM	n/a
Vinegar to Vincent Fish Habitat Improvement Project - Phase 1	CTWS	2020	IHI; RM	$ 1,146,919
Butte, Ruby, Beaver Creeks Phase 1	USFS	2020	FR; IHI; RM	*
Vincent to Caribou Phase II	CTWS	2020	IHI, RM, CR	$ 5,300
Camp Valley Restoration Phase 2 (Reach 4)	USFS	2020	FR; IHI; RM; UM	$ 154,493
Camp-Lick Riparian Restoration Phase I	USFS	2020	RM	*
Camp Valley Restoration Phase 3 (Reach 3)	NFJDWC	2021	CR; IHI, RM	$ 162,063
Riparian Plant Maintenance	CTWS	2021	RM	n/a
Camp-Lick Riparian Restoration Phase 2	USFS	2021	RM	$ 635,539
Plant Propagation/Planting Phase I	CTWS	2021	RM	n/a
Dead Cow instream habitat complexity/wood placement	CTWS	2021	IHI; RM	n/a
Dunstan Preserve Floodplain Enhancement Phase 2	CTWS	2021	IHI, RM	$ 9,585
Summit Creek Culvert FS Road 1940-281	USFS	2021	FP	$ 147,642
Vinegar to Vincent Phase 1.5	CTWS	2021	FR	$ 27,426
Big Creek Exclosures and Floodplain LWD phase 1	NFJDWC	2022	FR; IHI, RM	$ 206,440
Vinegar to Vincent Instream Restoration Phase 2	CTWS	2022	CR; FP, FR, IHI; RM	$ 2,845,328
Butte, Ruby, Beaver Creeks Phase 2	USFS	2022	FR; IHI; RM	$ 289,802
Riparian Plant Maintenance	CTWS	2022	RM	n/a
Camp-Lick Riparian Restoration Phase 3	USFS	2022	RM	$ 537,713
Plant Propagation/Planting Phase 2	CTWS	2022	RM	$ 518,052
Deep Creek Habitat Restoration Phase 1	USFS	2022	CR; FP; FR; IHI; RM	$ 258,555
Big Creek Exclosures and Floodplain LWD phase 2	NFJDWC	2023	RM	$ 232,836
Butte, Ruby, Beaver Creeks Phase 3	USFS	2023	FR; IHI; RM	$ 285,214

Appendix B: 10-Year Recommendations

P: Planning
M: Monitoring
R: Restoration

Green: Addressed
Yellow: Partially Addressed
Orange: Not Addressed

REPORT	TYPE	RECOMMENDATION
Analysis of Benthic and Drift Macroinvertebrate Samples	M	Future investigations should increase the number of macroinvertebrate collection sites within control reaches to better explore biotic integrity changes with stream restoration.
Analysis of Benthic and Drift Macroinvertebrate Samples	P	Carefully consider all attributes of the predictive model used to guide stream restoration.
Analysis of Benthic and Drift Macroinvertebrate Samples	M	Explore if functional group analysis and spatial models would support the hypothesis that management actions are affecting the biotic integrity of the MFJDR.
Analysis of Benthic and Drift Macroinvertebrate Samples	P	Ensure sufficient sample size and power to answer research questions. Statistical tests, particularly parametric tests, are most powerful with balanced designs.
Analysis of the Relationship between Macroinvertebrates, Streamflow, and Temperature in the Middle Fork John Day River, OR	M	Have a consistent data collection effort across data types, years, and sites to limit noise and variability and increase power of the analysis.
Camp Creek Restoration: A BACI Comparative Analysis	P	Alternative designs should be examined for future watershed scale restoration experiments. The paired-reach BACI design is promising. Alternative BACI designs should be researched through simulation and in the field.
Future Changes in Mainstem Water Temperatures in the Upper Middle Fork John Day River and the Potential for Riparian Restoration to Mitigate Temperature Increases	P	Plant faster-growing species such as cottonwood, alder, and aspen to achieve relatively large, closed canopy conditions within a few decades. Given these species can be susceptible to animal browsing, invest in efforts to exclude browsers, including deer, elk, and beaver.
Future Changes in Mainstem Water Temperatures in the Upper Middle Fork John Day River and the Potential for Riparian Restoration to Mitigate Temperature Increases	R	Given the importance of temperature in habitat quality, focus riparian revegetation efforts in streams where shade is currently limited. Use a long-term approach to measure the effects of riparian plantings given uncertainties around climate change.

REPORT	TYPE	RECOMMENDATION
Geomorphology and Physical Habitat	M	Results from the physical habitat surveys during the MFIMW further support the observation that it takes several years to show measurable results from restoration actions, and monitoring should be supported and evaluated throughout this timeline.
Geomorphology and Physical Habitat	M	Use remote sensing data to complement field measurements.
Geomorphology and Physical Habitat	P	It is important to think through potential processes and effects of vegetation change while designing active restoration and coupled monitoring projects.
Geomorphology and Physical Habitat	P	Develop in advance a plan for monitoring if a large flow event occurs.
Geomorphology and Physical Habitat	R	Incorporate the placement of log structures in existing or constructed pools to maintain depth as a restoration technique.
Geomorphology and Physical Habitat	R	Develop a long-term restoration plan before designing the monitoring plan that incorporates a communication plan.
Influence of Deer and Elk Browsing on the Success of Riparian Restoration Plantings	R	Consider ways to protect woody riparian species from browsing by deer and elk.
Influence of Deer and Elk Browsing on the Success of Riparian Restoration Plantings	R	When planning riparian plantings, consider the specific needs of plant species. If there is low ability to maintain or protect new plantings from browsing, focus on species known to be resistant to browsing.
Influence of Deer and Elk Browsing on the Success of Riparian Restoration Plantings	R	Planting species that are less affected by browsing, such as Ponderosa pine and thinleaf alder, may allow the establishment of a forested riparian canopy with a hardwood understory, achieving the desired condition of streamside shade.
Lessons Learned & Recommendations for Future Restoration Actions on the MFIMW	R	Future restoration actions should target flow enhancement in the upper reaches of the watershed where cool water originates including Meadow and pasture reaches of the mainstem MFJDR from Caribou Creek upstream through Phipps Meadow that remain with poorly developed riparian shade and altered channel profiles.
Lessons Learned & Recommendations for Future Restoration Actions on the MFIMW	R	MFJDR restorationists would benefit from a strategic plan that includes collaboration and coordination while also targeting actions suggested herein.
Lessons Learned & Recommendations for Future Restoration Actions on the MFIMW	R	Future restoration actions should target flow enhancement in the upper reaches of the watershed where cool water originates.
Lessons Learned & Recommendations for Future Restoration Actions on the MFIMW	R	Stream water surfaces need to be protected in tributary and upstream reaches from solar insolation to keep this cool water cool. Specific reaches to consider for restoration include:

REPORT	TYPE	RECOMMENDATION
Lessons Learned & Recommendations for Future Restoration Actions on the MFIMW	R	Future restoration actions should target flow enhancement in the upper reaches of the watershed where cool water originates including Cool-water tributaries such as Bridge Creek that have been particularly altered and no longer retain their cool water connection to the MFJDR.
Lessons Learned and Recommendations from MFIMW Contributors and Workgroup Discussions	P	Agree upon a list of the required information to be stored in the restoration inventory and update it annually with the restorationists.
Lessons Learned and Recommendations from MFIMW Contributors and Workgroup Discussions	P	Provide clear communication structures to develop Implementation and Experimental Design among all partners involved. Provide adequate time for restorationists to buy into the Experimental Design for treatment and control areas to be maintained as best as possible to allow long-term monitoring and statistical analyses.
Lessons Learned and Recommendations from MFIMW Contributors and Workgroup Discussions	P	Continue to prioritize funding this gage to allow long-term streamflow data collection.
Lessons Learned and Recommendations from MFIMW Contributors and Workgroup Discussions	P	For future IMW work, fund and maintain the research station for visiting researchers at the CTWSRO's Oxbow Conservation Area. Use RVs to complement available local housing or tent camping. The current RV may provide several more years of use but will need to be replaced eventually.
Lessons Learned and Recommendations from US Forest Service Restoration Efforts	R	Through collaborative working groups and a clear communication structure throughout the project, ensure adequate opportunities for all partners to learn where monitoring and restoration actions are planned.
Lessons Learned and Recommendations from US Forest Service Restoration Efforts	P	Consider stream gradient and valley confinement, riffle lengths, pool quality, and quantity in addition to existing large wood loading and recruitment to improve instream conditions.
Lessons Learned and Recommendations from US Forest Service Restoration Efforts	P	Place wood that interacts with low flow conditions and consider side channels and other human features that constrain valley processes. Consider treating the entire reach and valley, rather than patches with log weirs.
Lessons Learned and Recommendations from US Forest Service Restoration Efforts	P	Valuable tools and information, such as NetMap and BRAT, are available to evaluate various limiting factors or processes impacting riparian and instream conditions. Consider these tools when prioritizing actions in landscapes with riparian community and beaver issues.
Lessons Learned and Recommendations from US Forest Service Restoration Efforts	R	Be prepared for different public perceptions when implementing large-scale restoration projects and perform adequate community outreach to minimize negative responses from the community.

REPORT	TYPE	RECOMMENDATION
Lessons Learned and Recommendations from US Forest Service Restoration Efforts	R	Evaluate landscape restoration actions from ridgetop to ridgetop, considering resistance and resilience to biophysical processes and ecological functions from a top-down context. Integrate planning into the revegetation program. Consider valley characteristics and processes of solar radiation loading. Identify plants that are ecologically appropriate for the site, and plant at distances that can expand without management inputs through passive management.
Lessons Learned from Oxbow Conservation Area Dredge Tailings Restoration Implementation	R	Require a licensed landscape specialist to work with the contractor on plant salvage and planting operations.
Lessons Learned from Oxbow Conservation Area Dredge Tailings Restoration Implementation	R	Salvage and re-plant all native vegetation when possible. This ensures that new channels look natural sooner, and the vegetation holds soils and the banks together.
Lessons Learned from Oxbow Conservation Area Dredge Tailings Restoration Implementation	R	Install elk-proof fencing to protect investment in riparian plantings.
Lessons Learned from Oxbow Conservation Area Dredge Tailings Restoration Implementation	R	Invest in irrigation to keep riparian plantings alive through the first 2 to 3 growing seasons to establish their sustainability.
Lessons Learned from Oxbow Conservation Area Dredge Tailings Restoration Implementation	R	Ground-truth the LiDAR data set before the design process is initiated.
Lessons Learned from Oxbow Conservation Area Dredge Tailings Restoration Implementation	R	Channel design must conform to a profile where the riffle crest or head is the highest feature in the substrate. Riffles need fines washed in to ensure the matrix is hardened and stable.
Lessons Learned from Oxbow Conservation Area Dredge Tailings Restoration Implementation	R	Determine if future flooding flows will assist with sealing riffles substrates. It is possible that high flows may degrade riffle crests that are not adequately constructed.
Lessons Learned from Oxbow Conservation Area Dredge Tailings Restoration Implementation	R	Wherever possible, acquire appropriate baseline information specific to areas of interest.
Lessons Learned from Oxbow Conservation Area Dredge Tailings Restoration Implementation	R	Add design elements that would cause sediment deposition over time, as well as large wood and gravel placements to narrow the active low-flow channel.

REPORT	TYPE	RECOMMENDATION
Lessons Learned from Oxbow Conservation Area Dredge Tailings Restoration Implementation	R	To maintain scour and provide other benefits, place large wood structures out into the channel.
Lessons Learned from Oxbow Conservation Area Dredge Tailings Restoration Implementation	R	Ensure there are adequate personnel to transfer fish to decrease transfer time and reduce mortalities.
Monitoring and Assessment of Critical Thermal Dynamics in Upper Middle Fork of the John Day River, 2008-2016	P	Develop a plan to collect additional data over decadal scales to accurately assess how changes to vegetative cover (shading) might impact stream temperatures.
Monitoring and Assessment of Critical Thermal Dynamics in Upper Middle Fork of the John Day River, 2008-2017	M	Consider results in concert with other findings from the IMW to understand the apparent lack of hyporheic water exchange within the MFIMW.
Monitoring and Assessment of Critical Thermal Dynamics in Upper Middle Fork of the John Day River, 2008-2017	M	DTS can be implemented to identify locations and magnitude of groundwater influence.
Monitoring and Assessment of Critical Thermal Dynamics in Upper Middle Fork of the John Day River, 2008-2017	M	Collect thermal infrared/FLIR data throughout the day to evaluate the full temperature signature.
Monitoring and Assessment of Critical Thermal Dynamics in Upper Middle Fork of the John Day River, 2008-2019	R	The magnitude and location of cold water inputs into the MFJDR from tributaries and groundwater upwelling can be leveraged in restoration designs.
Monitoring and Assessment of Critical Thermal Dynamics in Upper Middle Fork of the John Day River, 2008-2020	R	Restoration should incorporate the reduction of exposed stream area to maximize salmonid productivity and restoration effectiveness.
Monitoring and Assessment of Critical Thermal Dynamics in Upper Middle Fork of the John Day River, 2008-2021	R	Managers need to consider how goals and factors interplay through adaptive management and prioritize actions as needed to achieve their priority goals.
Monitoring and Assessment of Critical Thermal Dynamics in Upper Middle Fork of the John Day River, 2008-2022	R	Future restoration efforts should include temperature analyses in their restoration impact assessments to maximize benefits to salmonids.
Monitoring and Assessment of Critical Thermal Dynamics in Upper Middle Fork of the John Day River, 2008-2023	R	Evapotranspiration for the restored system should be analyzed based on the changes in the riparian system. Greater shade requires larger plants, which consume water.

REPORT	TYPE	RECOMMENDATION
Projected Response of Riparian Vegetation to Passive and Active Restoration over 50 years	P	Expectations for restoration outcomes should be tempered with a realistic understanding of the rate at which natural systems can recover and account for relatively rare episodic events.
Projected Response of Riparian Vegetation to Passive and Active Restoration over 50 years	R	Our simulations suggest that active restoration will have a bigger impact on species that have a limited potential spatial distribution, and where a significant proportion of the available habitat is in poor condition.
Socio-Economic Indicators Follow-Up Study	P	Guidelines for how to track and analyze connections between ecosystem restoration and contributions to local economies should be established before restoration actions are implemented. Define what types of data are needed and how to extrapolate from unique characteristics and specific restoration projects.
Socio-Economic Indicators Follow-Up Study	P	Define indicators and outcome measures in consultation with local officials and residents, to gauge metrics that are important to them.
Socio-Economic Indicators Follow-Up Study	P	Use the measures to inform the general public about the socio-economic contribution of restoration efforts and as an input to public decision-making and action.
Socio-Economic Indicators Follow-Up Study	P	Use the measures to help private landowners as they make decisions about engaging in restoration work so they can put these decisions in the context of the local economy.
Steelhead and Chinook Salmon Monitoring and Evaluation	M	Improve understanding of juvenile Chinook movement and distribution during baseline (pre-treatment) conditions.
Steelhead and Chinook Salmon Monitoring and Evaluation	M	Operate rotary screw trap site continuously throughout the migration.
Steelhead and Chinook Salmon Monitoring and Evaluation	P	Expand the use of bioenergetics and life-cycle models to investigate influential mechanisms.
Steelhead and Chinook Salmon Monitoring and Evaluation	M	Include additional sampling events during winter to better understand juvenile salmonid movement throughout the year.
Steelhead and Chinook Salmon Monitoring and Evaluation	P	To investigate fish/habitat relationships, design paired study reaches across specific habitat variables to address specific questions
Steelhead and Chinook Salmon Monitoring and Evaluation	P	Couple habitat with fish monitoring to answer questions about fish survival, growth, and abundance in a paired experimental fashion using newly developed models that link habitat metrics to fish metrics.
Steelhead and Chinook Salmon Monitoring and Evaluation	R	Restoration actions take decades to achieve results. In the interim timeframe, evaluate restoration actions using habitat response variables and then use predictive models to link to fish responses.

REPORT	TYPE	RECOMMENDATION
Stream Habitat Condition for Middle Fork John Day River and Camp Creek Watershed	M	Future sampling of the MFJDR and Camp Creek sites should continue to occur at 5-year intervals. The next sampling event should occur in 2019 and 2024.
Stream Habitat Condition for Middle Fork John Day River and Camp Creek Watershed	M	Long-term monitoring should continue in the MFJDR and Camp Creek to track habitat changes. Maintain continuity of long-term sampling sites to enable trend detection using an established protocol that generates habitat metrics important to salmonids.
Stream Habitat Condition for Middle Fork John Day River and Camp Creek Watershed	P	Evaluate tasks across the scope of the entire project to identify economies of scale.
Stream Habitat Condition for Middle Fork John Day River and Camp Creek Watershed	P	A useful next step from this study would be to combine all existing PIBO data from the MFIMW. Analyze Camp Creek data with the other randomly established PIBO tributary sites that the USFS Research station has established throughout the MFJDR to better describe changes over a larger watershed scale.
Stream Habitat Condition for Middle Fork John Day River and Camp Creek Watershed	M	Have riparian plantings improved the vegetation and how does this compare to passive restoration actions (fencing and grazing management) alone?
Stream Habitat Condition for Middle Fork John Day River and Camp Creek Watershed	M	Has the change in riparian vegetation affected physical habitat attributes such as bank stability and percent fines in pools?
Stream Habitat Condition for Middle Fork John Day River and Camp Creek Watershed	M	Are invasive plant species more predominant; if so, which ones?
Stream Habitat Condition for Middle Fork John Day River and Camp Creek Watershed	M	Analyze the PIBO vegetation data to understand how riparian habitats have changed based on passive and active restoration actions in both geographic areas. Specifically, we suggest answering the following questions after the 2019 resurvey is performed:
Stream Habitat Condition for Middle Fork John Day River and Camp Creek Watershed	P	Collaborative partnerships need a point person to analyze data, streamline workflow, and create efficiencies to meet stated objectives for all partners involved.
Water Temperature Monitoring	M	Coordinate among water temperature data collection efforts, to promote collaboration, avoid duplication, and create efficiencies.
Water Temperature Monitoring	P	Identify an appropriate platform for storing temperature data and secure funding to purchase, develop, and maintain the platform. Clearly communicate consistent monitoring goals and written protocols for data collection, quality control, and analysis methods. This communication is especially important when multiple organizations are contributing data.
Water Temperature Monitoring	P	Perform consistent and timely quality control procedures every season after the data is downloaded. Develop a data collection protocol and quality control procedures in collaboration with all data collection entities to ensure its usefulness.

REPORT	TYPE	RECOMMENDATION
Water Temperature Monitoring	P	Establish clear monitoring goals, perform mid-project analysis, document reasoning behind site selection, and maintain communication with collaborators as the study continues. Use an adaptive monitoring approach, with clear documentation to help during times of staff and/or funding changes.
Water Temperature Monitoring	P	Form a committee of individuals invested in temperature monitoring to develop a Sampling and Analysis Plan for water temperature monitoring to ensure consistent field protocols and data QA/QC measures are followed. Consider a statistical site selection process like GRTS and contributing data to NorWest.
Water Temperature Monitoring	P	Identify statistical analyses that could include air temperatures and flow data to better understand watershed level water temperature changes. For example:
Water Temperature Monitoring	P	Complete 7DADM analysis on tributary loggers
Water Temperature Monitoring	P	Update HeatSource and/or ISEMP models. Use results to identify restoration activities that influence water temperature.
Water Temperature Monitoring	R	A complete evaluation of the influence of the current and restored Bridge Creek habitat potential should include a temperature analysis using the HeatSource model to understand impacts to fish using a bioenergetics modeling approach to fully understand the restoration alternatives.
Water Temperature Monitoring	P	Link monitoring projects with specific management or restoration questions. For example, identify specific restoration projects that are anticipated to affect water temperature and then document changes pre-/post-restoration.
Water Temperature Monitoring	P	Complete analysis incorporating air temperature and flow data.
Water Temperature Monitoring	P	Identify loggers with data before and after MFIMW inception (2008) and calculate differences – similar to SFJDR vs MFJDR analysis.

Appendix C: MFIMW Macroinvertebrate Community Analysis

Phase 1 Report Prepared for the North Fork John Day Watershed Council Attention: Javan Bailey, Restoration Project Manager

Date: 21 March 2023

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BACKGROUND

Restoration projects in the Middle Fork John Day River Intensively Monitored Watershed (MFIMW) have been implemented since the mid-1990s. The overarching goal is to improve degraded instream and riparian conditions and enhance ecological functions to benefit native fish and improve the ecological integrity of the watershed. Habitat changes resulting from restoration are also expected to improve conditions for macroinvertebrate populations in the Middle Fork John Day River (MFJDR). For example, reductions in sediment load and substrate embeddedness may be accompanied by decreased abundance of burrowing organisms and increases in clingers; improved riparian conditions can enhance populations of organisms that feed as shredders; increases in canopy cover and stream flow can support communities with more sensitive taxa or lower community temperature associations; and increased habitat stability may support organisms that are more sensitive to disturbance and/or have a longer egg to adult development time.

Analyzing results from a Before-After/Control-Impact design (BACI) involves separating effects of restoration from other sources of variation, such as the normal annual fluctuations in macroinvertebrate communities and longer-term stressors such as climate change. Typically, restoration that occurs at the stream or basin level is more effective in improving overall habitat conditions, compared to smaller scale reach-level restoration in a basin still experiencing multiple negative stressors. With this in mind, we investigated divergence in Oregon Department of Environmental Quality (ORDEQ) PREDATOR model scores, community composition, and multiple ecological metrics between restored MFJDR and control South Fork John Day River (SFJDR) sites over time, as well as long term changes at sampling sites, to answer the initial questions posed in Phase 1 of the project.

For macroinvertebrate communities collected by drift sampling, Phase 1 questions are:

1. How does macroinvertebrate biomass change through time and space at each site?
2. How does macroinvertebrate community composition change through time and space?

For macroinvertebrate communities collected by benthic sampling, Phase 1 questions are:

1. How do macroinvertebrate communities change through time at each individual site in the MFJDR?
2. How do macroinvertebrate communities vary across sampling sites for a given year in the MFJDR?
3. For each site in the MFJDR and SFJDR, in which direction are the O/E scores trending?
4. Are there differences or overall trends between or across the MFJDR and SFJDR watersheds?

A final Phase 1 question is: Are there are similarities between drift and benthic samples?

Additional metrics beyond those needed to answer the drift sampling Phase 1 questions were calculated to facilitate initial comparisons of community characteristics between the drift and benthic data sets.

METHODS

Drift data calculations

All years of drift data (2010-2022) were compiled into a single database and the taxonomy was brought into agreement with the most recent standard taxonomic effort established by the Pacific Northwest Aquatic Monitoring Partnership (PNAMP). Because the duration of sampling and stream flow rates differed between sites and years, each sample was standardized by calculating total organismal concentration (# individuals/m^3 of water) and biomass (mg biomass/m^3 water). The ORDEQ models used for the benthic data cannot be applied to drift data, as they were developed specifically for benthic macroinvertebrates collected in riffle habitats. However, some of the metrics relating to taxonomic richness, diversity, and tolerance that were calculated for benthic samples were also calculated for drift samples to facilitate detection and comparison of trends. Additionally, the relative abundance of terrestrial organisms in each sample was calculated, as this metric could be expected to respond to a subset of restoration activities that impact the riparian zone, such as livestock exclusion or re- vegetation.
Drift sample concentration and biomass were calculated according to Danehy et al. (2017). To standardize concentration and biomass in each sample, data from field sheets was used to calculate duration (seconds) that drift nets were left in place and mean water velocity (m/s) during the sampling period. The duration of the sampling period varied greatly between years, ranging from as few as two hours to more than 18 hours. The time of day during which sampling occurred also varied, being done in the late morning, afternoon, or evening/night in different years. These factors can all be expected to alter sample composition, as many insects that exhibit diel drift patterns are more likely to enter the drift at night, while some Trichoptera (caddisfly) and Acari (mite) taxa drift more during the day (Waters, 1972; Brittain & Eikeland, 1988).

The volume of water (m^3) that passed through the net during the sampling interval was calculated as [mean flow (m/s) x duration of sampling event (s) x net area (0.08 m^2)]. Total numbers of organisms per sample were calculated by multiplying the number of individuals picked from the sample by the inverse of the percentage of the entire sample that was sub- sampled. This was necessary because macroinvertebrate samples are routinely sub-sampled to an organismal count of 500 individuals, and the percent of the total sample picked among all drift samples across time ranged from 13-100%.

Individual abundance and organismal mass of each sample was summed and divided by total water volume to yield concentration (# individuals/m^3) and biomass (mg/m^3) metric values. Other community metrics calculated included:

- total richness (# of taxa in sample);
- Shannon Diversity Index H (measure of species diversity, with lower values reflecting less diversity);
- % terrestrial (relative abundance of terrestrial invertebrates);
- Ephemeroptera (mayfly), Plecoptera (stonefly), and Trichoptera (caddisfly) richness (individually and as total #EPT taxa);
- % diversity EPT (proportion of total richness comprised of EPT taxa);
- Community BI (biotic index; weighted average of individual taxa tolerance scores; note that these scores are not assigned to all taxa in a sample, as they do not apply to terrestrial taxa and are not known for all aquatic or aquatic/terrestrial taxa)
- % dominance of the top taxon (relative abundance of the most numerically abundant taxon in a sample);
- % small (0-6 mm), % medium (6-12 mm), and % large (12-100 mm) (relative abundance of organisms in different size classes in a sample)

Statistical analyses were done using PAST 4.0 (Hammer et al., 2001) and PRIMER-e v7 (Clarke et al., 2014) software. Comparisons of community composition were done as CLUSTER dendrograms run on Bray Curtis similarity indices of square root-transformed data. Because the first few years of samples were identified only to family level, all sample data were collapsed to this taxonomic level to standardize these community comparisons.

Statistically significant, unidirectional, long-term trends in community metrics were investigated by running Pearson Product Moment correlations between each of the 15 drift community measures and sampling years, with results reported at both alpha = 0.01 and alpha =

0.05. Results are reported at both significance levels to allow the authors and reviewers of the Phase 1 report to collectively review and discuss the results at different levels of stringency.

Following this overall assessment of differences, results of correlation analyses were used to assess whether temporal trends have occurred in each drift site across the 12-year monitoring period. These results were summarized as the number of significant correlations occurring in the Middle Fork watershed to assess the extent to which trends have occurred (as measured by the number of significant correlations relative to the total number of correlations run) and the direction(s) in which measured trends are occurring (i.e., increasing or decreasing).

The number of significant unidirectional trends was then summarized for each drift site, and drift sites with multiple significant trends were further examined to determine whether trends were consistently indicating site-specific improving or declining conditions. Such directionally consistent results were construed as several lines of evidence of either improving or declining community conditions at individual drift sites.

Drift community measures (mean/SD) were plotted from downriver to upriver for each metric to investigate longitudinal differences in drift conditions among individual sites in the Middle Fork watershed. The plots were examined and interpreted for potential downstream- upstream spatial trends and directional changes in trends.

Initial analysis of changes in drift community composition over time was done for each site using CLUSTER dendrograms to provide a broad assessment of similarity/difference of the drift community in different years at each site. Dendrograms were examined and interpreted for potential changes relating to restoration activities in specific years, but no additional taxonomic or community analyses were done at this point.

Benthic data calculations

As with the drift data, all available years of benthic data were first compiled into a single Access database. All benthic data were then reviewed to ensure that current taxonomic nomenclature and consistent taxonomic effort were applied across all sampling years and according to the most recent standard taxonomic effort established by PNAMP. Because the benthic data have historically been summarized each year into several community metrics and PREDATOR model O/E scores, these historical calculations were reviewed. From the historical community metrics and PREDATOR O/E scores, select community measures known to be particularly responsive to environmental stressors were chosen for these analyses of temporal and spatial differences and trends in benthic conditions in the MFIMW.

Specifically, the following community measures were selected for inclusion in these analyses:

- PREDATOR WCCP (Western Cordillera + Columbia Plateau) O/E scores;
- ORDEQ temperature stress scores;
- ORDEQ fine sediment stress scores;
- Total taxa richness;
- Ephemeroptera (mayfly), Plecoptera (stonefly), and Trichoptera (caddisfly) taxa richness
- Percent shredders (percent of individuals in sample belonging to the shredder functional feeding group)
- Shannon Diversity Index H (measure of species diversity, with lower values reflecting less diversity)

ORDEQ temperature and stressor models were applied to the complete set of 2010-2022 benthic data, as these had not been previously calculated. Statistical analyses of benthic data were performed using GraphPad Prism version 7.05 for Windows (GraphPad Software, San Diego, California USA, www.graphpad.com). Statistically significant, unidirectional, long-term trends in community conditions were investigated by running Pearson Product Moment correlations between each of the benthic community measures and sampling year, with results reported at both alpha = 0.01 and alpha = 0.05. Results are reported at both significance levels to allow the authors and reviewers of the Phase 1 report to collectively review and discuss the results at both levels. Phase 1 benthic analyses were conducted to characterize general spatial and temporal variability and trends in conditions within each watershed and temporal variability and trends at each individual benthic site.

For the Phase 1 benthic analyses, mean values of each community measure were first computed for each of the 20 benthic sites. Paired two-sample t-tests were then applied to the data to determine whether overall benthic community conditions differed between the Middle Fork and South Fork during the 12-year study period. Benthic community condition classes (good, fair, and poor) were then derived from PREDATOR WCCP O/E scores. The distribution of PREDATOR benthic community condition classes was plotted for each year for each study watershed to determine whether conditions differed between the watersheds and whether the distribution of condition classes changed over the study period in each watershed.

Following this overall assessment of differences between the two watersheds, results of correlation analyses were used to assess whether temporal trends have occurred in each watershed across the 12-year monitoring period. These results were first summarized as the number of significant correlations occurring in each watershed to assess the extent to which trends have occurred (as measured by the number of significant correlations relative to the total number of correlations run) and the direction(s) in which measured trends are occurring in each watershed.

The number of significant temporal trends was summarized for each benthic site, and those benthic sites with more than one significant trend were further examined to determine whether trends were consistently indicating site-specific improving or declining conditions. Such directionally consistent results were construed as several lines of evidence of either improving or declining benthic community conditions at individual benthic sites. Individual sites with more than one significant trend were further examined to assess the extent and nature of changes occurring at each site in question.

Benthic community measures (mean/SD) were plotted from upriver to downriver for each watershed to characterize differences in benthic conditions among individual sites within each watershed. Correlation analyses or other statistical tests were not applied to these data (i.e., between community measure values and site location); rather, the plots were examined and interpreted for potential downstream-upstream spatial trends and directional changes in trends occurring in each watershed.

RESULTS

Drift data

Evidence for long-term trends in drift samples

Statistically significant, long-term trends in individual measures of community condition occurred at several drift sampling sites within the Middle Fork watershed. Pearson Product Moment correlations were run between each of 15 community condition measures and year for each site (Table D 1). Owing to the large number of correlations, results are reported at both alpha = 0.01 and alpha = 0.05.

At a significance level of 0.05, 39 of the 210 correlations between individual community measures and year were significant; at a significance level of 0.01, only 11 of the 210 correlations between individual community measures and year were significant. Correlation analysis generally indicated improving conditions, with a few exceptions at individual sites (Table D 2).

Table D 1. Summary of results of Pearson Product Moment correlations run between each drift community measure and sampling year. Significance of test results is reported at both alpha = 0.01 and alpha = 0.05. Green = improving conditions, yellow = mixed results, red = declining condition.

		Significant?		
	Significance level (alpha)	**No**	**Yes**	**Trend**
CONCENTRATION (#/m3)	0.05	10	4	increasing
	0.01	14	0	
BIOMASS (mg/m3)	0.05	10	4	increasing
	0.01	14	0	
% TERRESTRIAL	0.05	14	0	
	0.01	14	0	
TOTAL RICHNESS	0.05	10	4	increasing
	0.01	13	1	increasing
EPHEMEROPTERA TAXA	0.05	6	8	increasing
	0.01	10	4	increasing
PLECOPTERA TAXA	0.05	12	2	increasing
	0.01	14	0	
TRICHOPTERA TAXA	0.05	10	4	increasing
	0.01	13	1	increasing
EPT RICHNESS	0.05	12	2	increasing
	0.01	12	2	increasing
RELATIVE DIVERSITY EPT	0.05	13	1	increasing
	0.01	13	1	increasing
COMMUNITY BI	0.05	11	3	2 increasing / 1 decreasing
	0.01	13	1	increasing

% TOP TAXON	0.05	12	2	decreasing
	0.01	14	0	
% SMALL	0.05	14	0	
	0.01	14	0	
% MEDIUM	0.05	13	1	increasing
	0.01	14	0	
% LARGE	0.05	12	2	increasing
	0.01	14	0	
SHANNON DIVERSITY H	0.05	12	2	increasing
	0.01	13	1	increasing
TOTALS	**0.05**	**171**	**39**	
	0.01	**199**	**11**	

The number of significant correlations was tallied for each site to determine which had several metrics that were significantly trending (Table D 2). Only one site had no trends in any community measures (D 005), while each of the remaining 13 sites had one to eight community measures trending at alpha = 0.05. Six sites had community measures trending at alpha = 0.01, with one to four measures trending. No trends were detected at any site for three community measures: Shannon Diversity Index H; % terrestrial; and % small. The community metric that trended significantly at the greatest number of sites (eight sites at alpha = 0.05, four at alpha = 0.01) was Ephemeroptera (mayfly) taxa richness, which increased among all sites where a trend was found. Trends generally suggest improving conditions, with increasing organismal concentration or biomass, increasing sample and/or EPT richness, more large-bodied organisms, and a more balanced community (i.e., lower relative abundance of the top taxon). Exceptions included a significantly increased community BI (indicative of more tolerant organisms) at D 367 (alpha = 0.05) and at D 006 (alpha = 0.01).

Table D 2. Summary of the number of significant unidirectional trends at both alpha = 0.01 (green) and alpha = 0.05 (yellow) between 15 benthic community measures across time at each of the 14 MFIMW drift sampling sites. Site numbers are presented in order from downstream to upstream. I = increasing, D = decreasing, N = no statistically significant trend.

	D 702	D 611	D 367	D 001	D 002	D 003	D 634	D 780	D 007	D 215	D 115	D 006	D 004	D 005
concentration	N	N	I	N	N	I	N	I	I	N	N	N	N	N
biomass	N	N	I	N	N	I	N	I	N	N	N	N	I	N
% terrestrial	N	N	N	N	N	N	N	N	N	N	N	N	N	N
sample richness	N	N	I	N	N	I	N	I	N	N	N	I	N	N
Shannon H	N	N	N	N	N	N	N	N	N	N	N	N	N	N
#Eph	I	N	I	I	I	I	N	I	N	I	N	N	I	N
#Ple	N	N	I	N	N	N	N	I	N	N	N	N	N	N
#Tri	I	N	I	N	I	N	I	N	N	N	N	N	N	N
EPT richness	I	N	I	N	N	N	N	N	N	N	N	N	N	N
rel. div. EPT	I	N	N	N	N	N	N	N	N	N	N	N	N	N
Comm. BI	D	N	I	N	N	N	N	N	N	N	N	I	N	N
%dom top taxon	N	D	N	N	N	N	N	N	N	N	N	D	N	N
%small	N	N	N	N	N	N	N	N	N	N	N	N	N	N
%medium	I	N	N	N	N	N	N	N	N	N	N	N	N	N
%large	N	N	N	N	N	N	N	N	N	N	I	N	I	N

Relationship between restoration years and changes in community composition at drift sites

Site D 001 experienced passive restoration prior to 2009, including riparian livestock fencing within 50 ft of the sampling start point and dam removal within 50-150 m of the sampling start. This site had a statistically significant trend in a single metric (alpha = 0.05), with increased numbers of Ephemeroptera taxa over time (Table D 2). The drift communities in earlier sampling years (2010-2013) differed more from each other and from all other years, but no strong patterns of community change were evident (Figure D 1). The most numerically abundant organisms in most years were adult chironomid midges.

Site D 002 experienced passive and active restoration, with riparian livestock fencing installed prior to 2009 within 50 m of the sampling start point, and channel reconfiguration, floodplain reconnection, instream habitat improvement, and bank stabilization done in 2016 within 500-1000 m from the sampling start point. This site had statistically significant trends in two metrics, with increased Ephemeroptera and Trichoptera taxa (alpha = 0.05), both of which suggest improving habitat conditions (Table D 2). No strong patterns of macroinvertebrate community change between years were evident (Figure D 1). The most numerically abundant organisms in most years were adult chironomid midges.

Site D 003 experienced both active and passive restoration, with riparian livestock fencing installed within 50 m of the sampling start point prior to 2009, and channel reconfiguration, floodplain reconnection, instream habitat improvement, and bank stabilization done in 2016 within 500-1000 m of the sampling start. This site had a statistically significant trend in four metrics (alpha = 0.05), with increased organismal concentration and biomass and more total and Ephemeroptera taxa (Table D 2), all of which suggest improving habitat conditions. The drift communities in earlier years (2010-2013) differed more from each other and from all other sampling years, but no other patterns were evident (Figure D 1). The most numerically abundant organisms in most years were adult chironomid midges and terrestrial Hemiptera (true bugs).

Site D 004 experienced active and passive restoration, with riparian management, instream habitat improvement, fencing, and planting done within 50 m of the sampling start point in 2013. This site had a statistically significant trend in three metrics, with increased organismal biomass,increased Ephemeroptera taxa (alpha = 0.05), and increased abundance of large organisms (alpha = 0.01) over time (Table D 2). The composition of drift communities in later sampling years was more similar, with an overall average Bray-Curtis similarity >62% among the 2016-2022 sample communities (Figure D 1). The community in samples taken closer to the year in which restoration was done (2013, 2015, 2017) differed more from other years, but it is not yet known if community composition changes can be related to restoration-driven habitat changes. The most numerically abundant organisms in most years were adult or larval chironomid midges.

Site D 005 experienced active and passive restoration, with riparian management, instream habitat improvement, fencing, and planting within 50 m of the sampling start point done in 2013. No statistically significant trends in any community metrics were seen at this site (Table D 2). No strong patterns in macroinvertebrate community composition between years were evident, although communities in most of the later sampling years (2016-2022) were more similar to each other than to earlier years (Figure D 1). The most numerically abundant organisms in most years were adult or larval chironomid midges.

Site D 006 experienced active and passive restoration. Prior to 2009, grazing management was done within 50 m of the sampling start point and riparian management, instream habitat improvement, and new side channel creation was done within 500-1000 m of the sampling start. In 2012, grazing management and planting was done within 50 m of the sampling start. This site had a statistically significant trend in three metrics, with increased total richness and lower abundance of the most numerically dominant taxon (alpha = 0.05) and increasing community BI (alpha = 0.01) (Table D 2). These results are mixed, as the first two trends suggest improving conditions, while an increasing biotic index suggests more organic pollution. No patterns in macroinvertebrate community composition between years were evident, although the communities in several early sampling years taken around the restoration time (2010, 2012, 2013) differed more from all other years (Figure D 1). The most numerically abundant organisms in most years were adult or larval chironomid midges.

Site D 007 experienced extensive active and passive restoration in multiple years. Prior to 2009, planting and fencing was done within 50 m of the sampling start point; in 2012, planting and grazing management was done within 150-500 m of the sampling start; in 2014, instream habitat improvement, channel reconfiguration, floodplain reconnection, bank stabilization, planting, and enclosures were done within 50 m of sampling start; in 2015, instream habitat improvement, channel reconfiguration, floodplain reconnection, riparian management was done within 50-150 m of sampling start; in 2017, riparian management and planting was done within 50 m of the sampling start and instream habitat improvement, channel reconfiguration, and flow modification was done within 500-1000 m of start; and in 2020, planting was done within 150- 500 m of the sampling start. Despite the extensive restoration, this site had a statistically significant trend in only a single metric, with increasing organismal concentration (alpha = 0.05) (Table D 2). No patterns in macroinvertebrate community composition between years were evident (Figure D 1). The most numerically abundant organisms in most years were adult or larval chironomid midges or Ephemeroptera nymphs.

Site D 115 experienced passive restoration as the result of upland fencing and riparian management done in 2009-2011 within 50 m of the sampling start point. This site had a statistically significant trend in a single metric, with increasing abundance of large-bodied organisms (alpha = 0.05) (Table D 2), which can be indicative of more stable habitat that supports the generally longer developmental time of these larger taxa. No patterns in macroinvertebrate community composition between years were evident, although communities in several earlier years (2010, 2012, 2013, 2015) differed more from all other years (Figure D 1). The most numerically abundant organisms in most years were adult or larval chironomid midges.

Site D 215 experienced passive restoration as the result of upland fencing and riparian management done in 2009-2011 within 50 m of the sampling start point. This site had a statistically significant trend in a single metric, with increasing Ephemeroptera taxa (alpha = 0.01) (Table D 2), which suggests improving habitat conditions. No patterns in macroinvertebrate community composition between years were evident, although communities in several earlier years (2010, 2012, 2013) differed more from all other years (Figure D 1). Taxa that contributed the greatest individual abundance to the sample varied annually more than at many other sites, with terrestrial Hemiptera, adult and immature Ephemeroptera and Trichoptera, aquatic worms (Oligochaeta), and terrestrial aphids dominating organismal abundance in different years.

Site D367 experienced passive restoration as the result of riparian livestock fencing done in 2012 within 150-500 m of the sampling start point. Somewhat paradoxically, despite having undergone very little restoration, this site had significant unidirectional trends in more metrics (eight) than any other drift site, with increased organismal concentration and biomass, more Plecoptera taxa, and increased community biotic index (alpha = 0.05), and more total, Ephemeroptera, and Trichoptera taxa and increased EPT richness (alpha = 0.01) (Table D 2). All trends except increased biotic index suggest improving conditions. Sample communities in several earlier years (2010, 2012, 2014) differed more from all other sampling years, and sample communities in 2019-2022 had the highest overall average similarity (>65%) (Figure D 1). Taxa that contributed the greatest individual abundance to the sample varied more than at many other sites, with terrestrial Hemiptera, adult and larval Chironomidae, aquatic Lepidoptera (butterfly) larvae, and Trichoptera larvae dominating organismal abundance in different years.

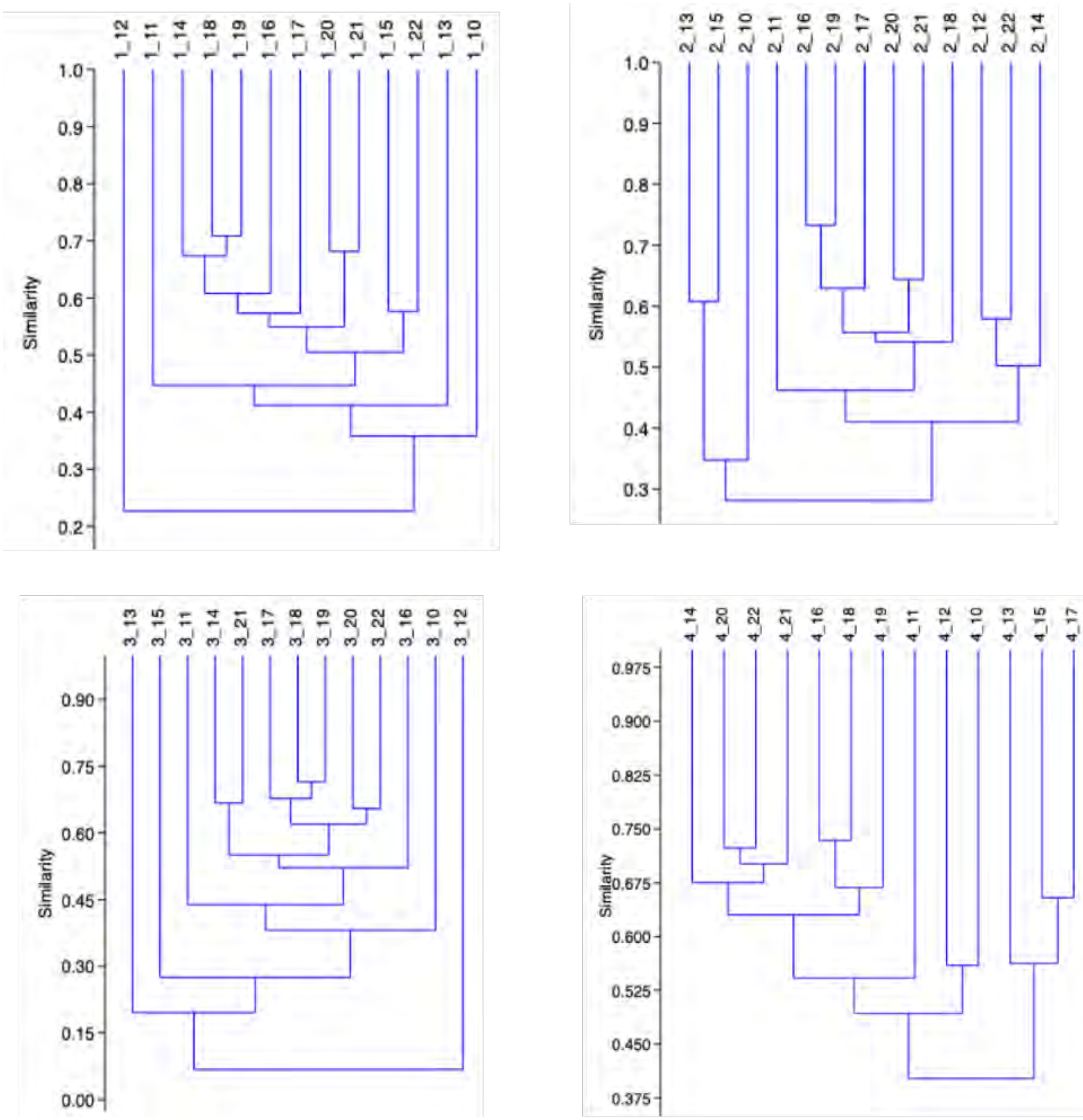
Site D 611 did not experience any restoration. This site had a statistically significant trend in a single metric, with decreasing abundance of the most numerically dominant taxon (alpha = 0.05) (Table D 2), which generally reflects less disturbed conditions. The sample communities in early years (2010-2012) differed more from all other sampling years (Figure D 1), but no patterns in community change were evident. The most numerically abundant organisms in most years were terrestrial Hemiptera or adult or larval Chironomidae.

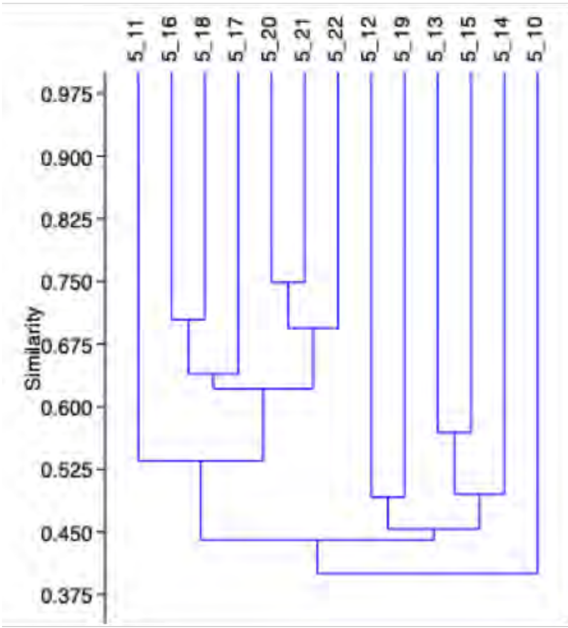
Site D 634 experienced active and passive restoration. Prior to 2009, livestock fencing & planting was done within 50 m of the sampling start point and channel reconfiguration, instream habitat improvement, and logjam placement was done within 150-500 m of the sampling start. In 2015, channel reconfiguration, floodplain reconnection, instream habitat improvement, and bank stabilization were done within 50 m of the sampling start. Despite extensive restoration in close proximity to the sampling reach, this site had a statistically significant trend in just a single metric, with increasing Ephemeroptera taxa (alpha = 0.05) (Table D 2). The communities in several earlier years (2010- 2015) differed more from each other and from all other years (Figure D 1) but no other patterns were evident. The most numerically abundant organisms in most years were adult chironomid midges.

Site D 702 experienced active and passive restoration within 500-1000 m of the sampling start site, with logjams and plantings installed in 2009, and exclosures placed in 2020. This site had statistically significant trends in six metrics: increasing Trichoptera taxa, decreasing community BI, and increasing abundance of medium-bodied organisms (alpha = 0.05); and increasing Ephemeroptera richness, EPT richness, and relative diversity of EPT (alpha = 0.01) (Table D 2), all of which generally suggest improving conditions. No patterns in macroinvertebrate community composition across time were evident, although the 2010 community was an outlier to all other years (Figure D 1). The most numerically abundant organisms in most years were terrestrial Hemiptera or adult Chironomidae.

Site D 780 experienced active restoration, with channel reconfiguration, floodplain reconnection, and instream habitat improvement in 2009 within 500-1000 m of the sampling start point, and riparian management done in 2017 within 50 m of the sampling start. This site had a statistically significant trend in five metrics: increasing organismal concentration and biomass, increasing total and Plecoptera taxa richness (alpha = 0.05), and increasing Ephemeroptera taxa (alpha = 0.01) (Table D 2), all of which suggest improving conditions. No pattern in community composition between years was evident (Figure D 1), although sample communities from the most recent three sampling years (2020-2022) were most similar to each other. The most numerically abundant organisms in most years were adult Chironomidae or terrestrial Hemiptera.

Figure D 1. CLUSTER dendrograms of macroinvertebrate community composition among drift sampling sites in each year. The number at the end of the site label indicates sampling year.

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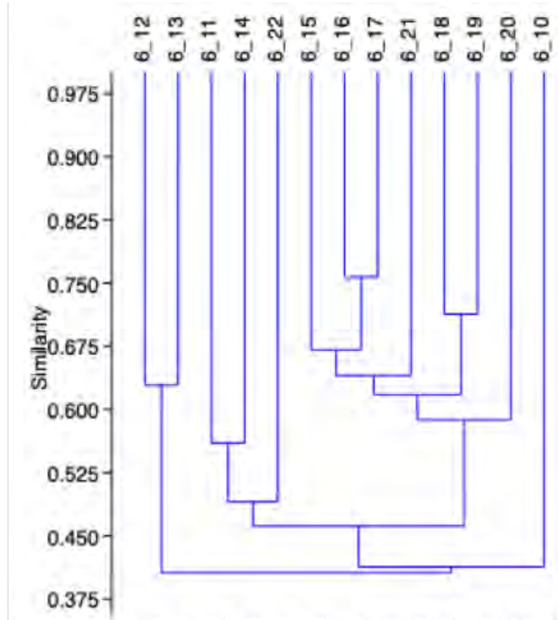
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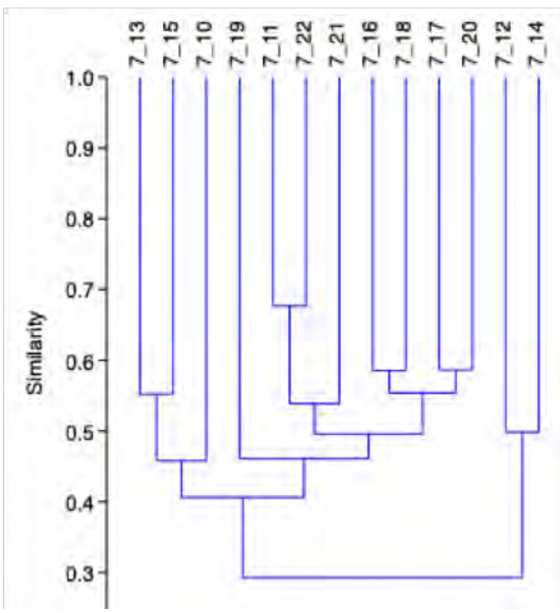
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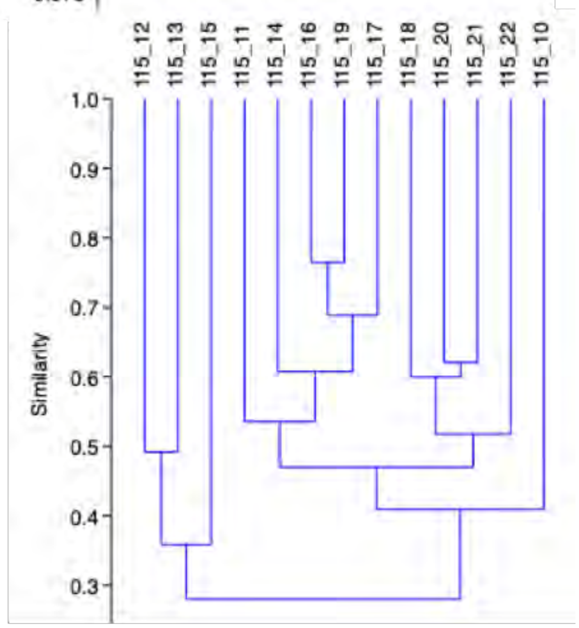
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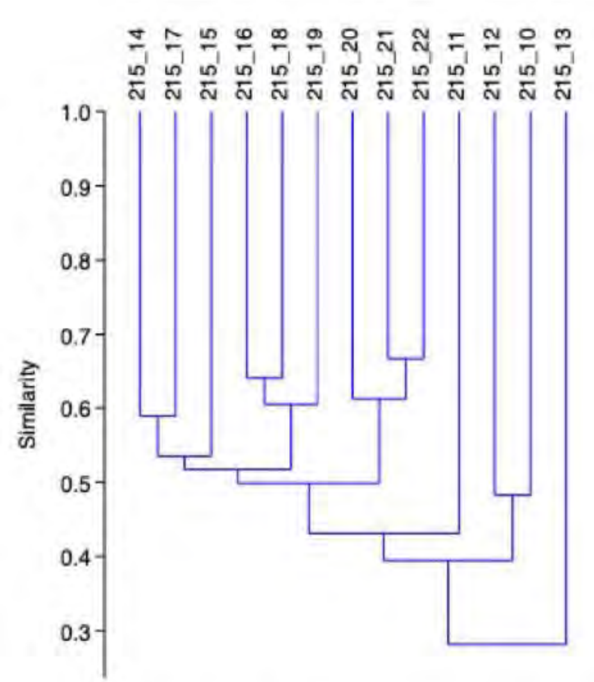
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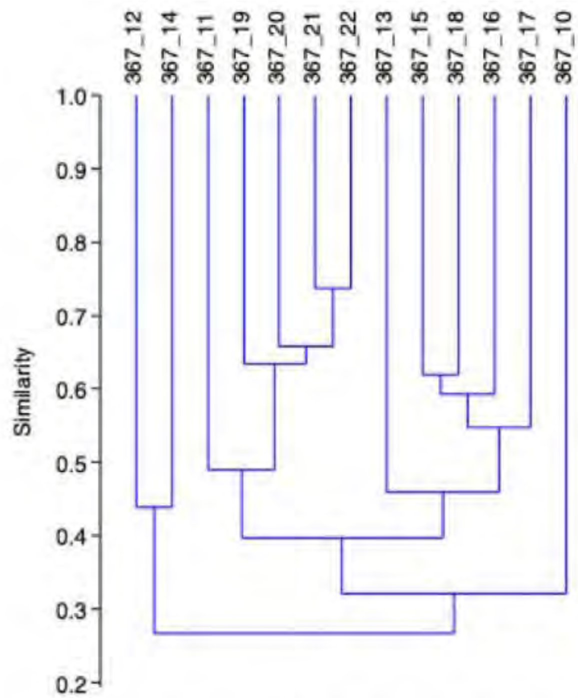
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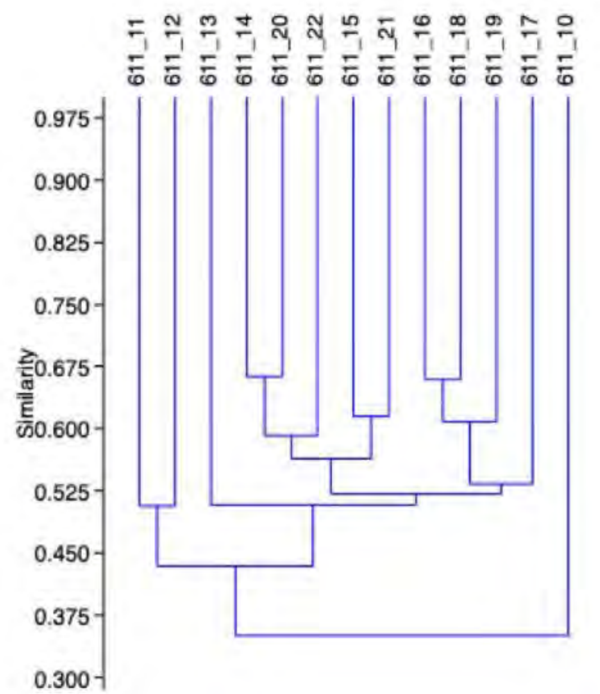
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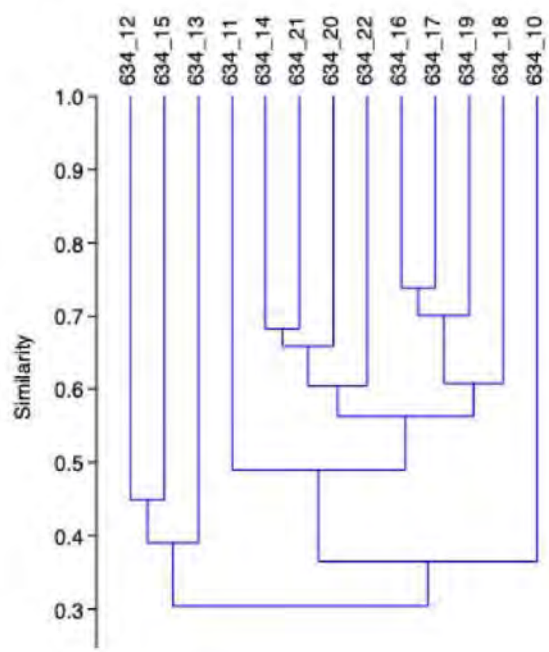
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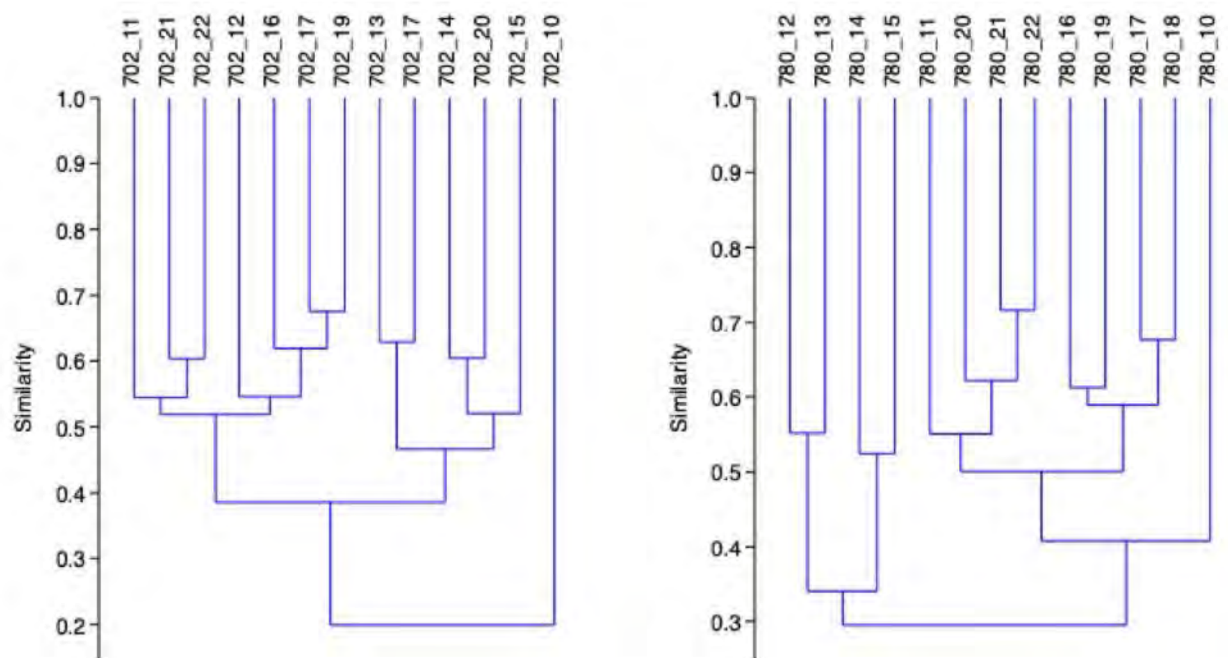
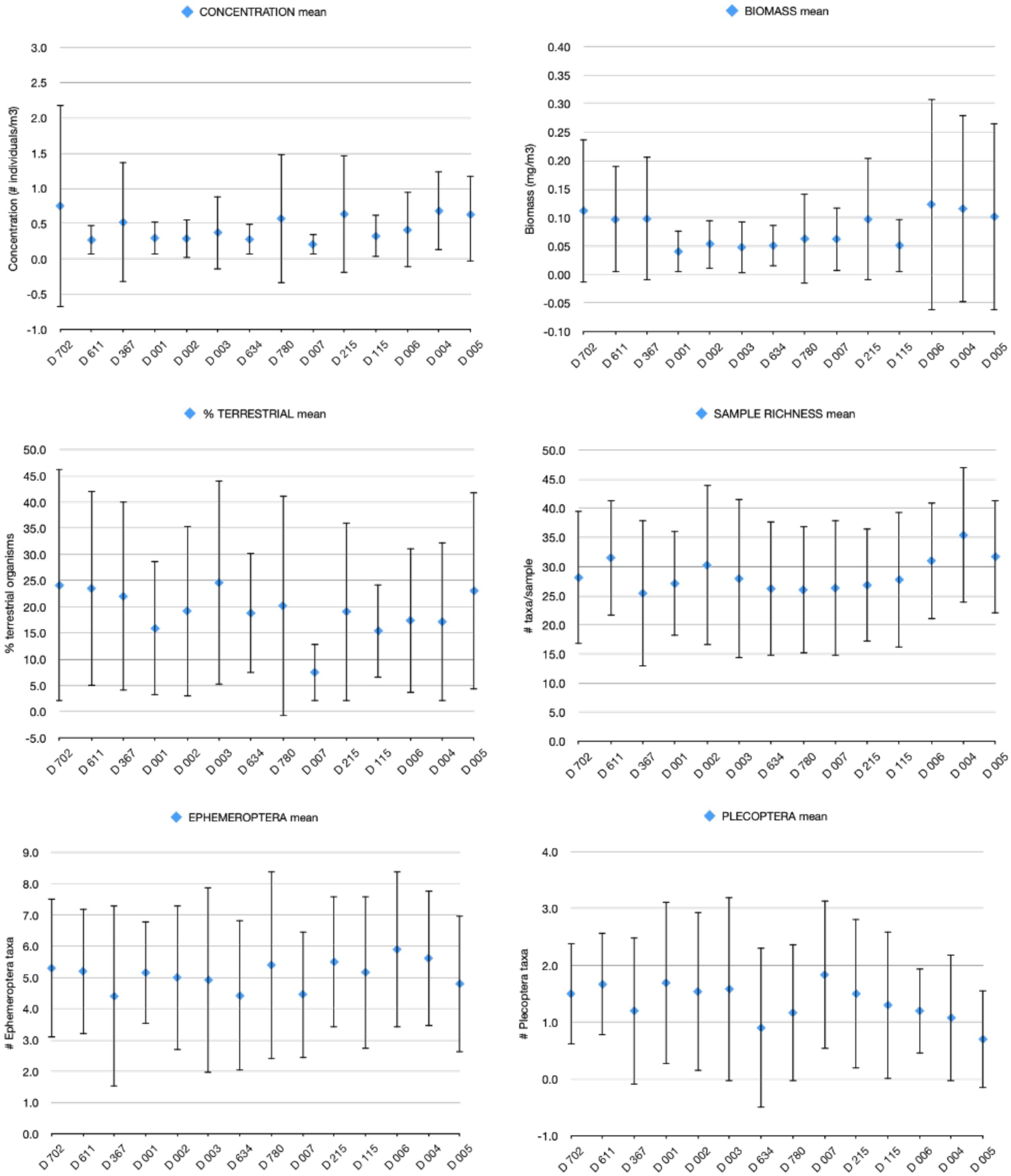
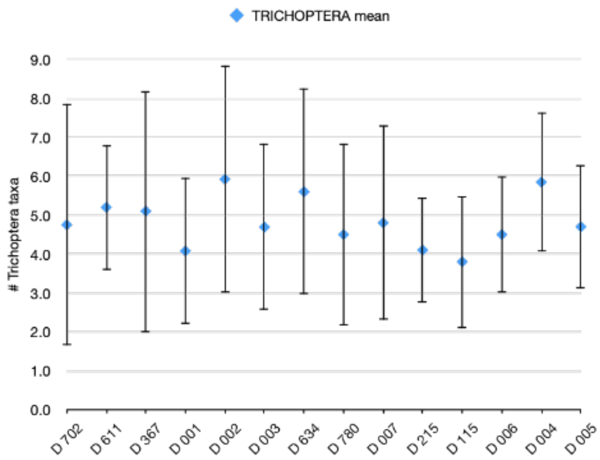


Spatial differences in metrics among drift sites

The values of each community measure exhibit variability, some of which could potentially occur as longitudinal trends along the length of each river. Community measures were examined individually to determine if any trends from downriver to upriver were evident, and to discern directional changes in these trends (Figure D 2). Drift sample metrics varied between sites, and most did not exhibit a longitudinal trend, but there were a few metrics that exhibited a slight spatial trend. Sites located the farthest upstream (D 006, D 004, D 005) had higher mean organismal biomass and concentration, while the mean number of Plecoptera and relative diversity of EPT taxa decreased moving upriver from site D 007 through site D 005.

Figure D 2. Means (±1 SD) of community metrics calculated at each drift site in the Middle Fork John Day River. Sites are arranged in order (left to right) from downstream to upstream.

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Trichoptera taxa
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8.0
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
D 702
D 611
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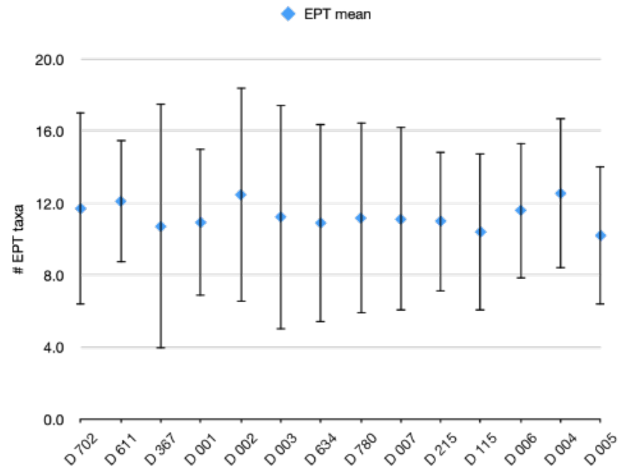
D 005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EPT taxa
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12.0
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0.0
D 702
D 611
D 367
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D 634
D 780
D 007
D 215
D 115
D 006
D 004
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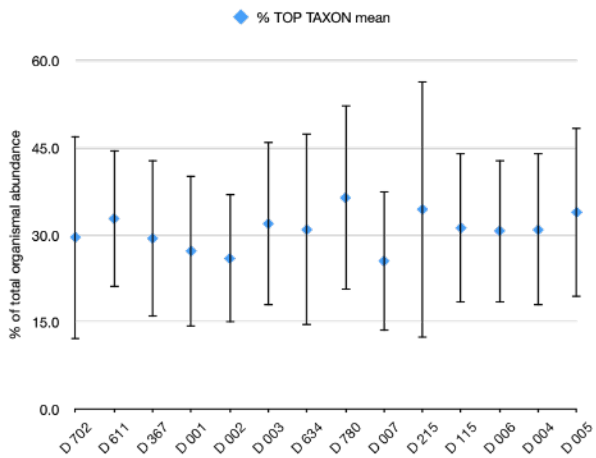
% TOP TAXON mean
% of total organismal abundance
60.0
45.0
30.0
15.0
0.0
D 702
D 611
D 367
D 001
D 002
D 003
D 634
D 780
D 007
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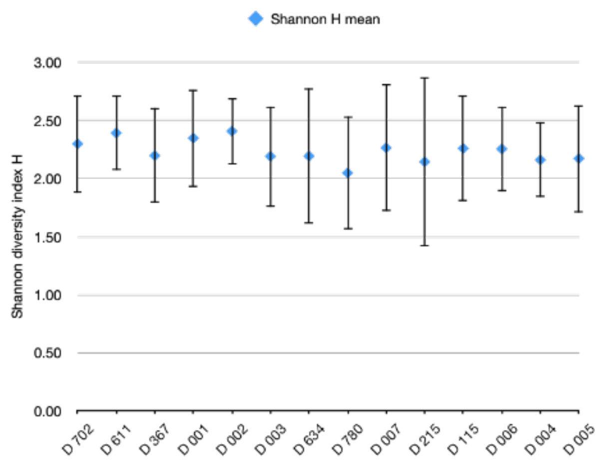
Shannon H mean
Shannon diversity index H
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2.50
2.00
1.50
1.00
0.50
0.00
D 702
D 611
D 367
D 001
D 002
D 003
D 634
D 780
D 007
D 215
D 115
D 006
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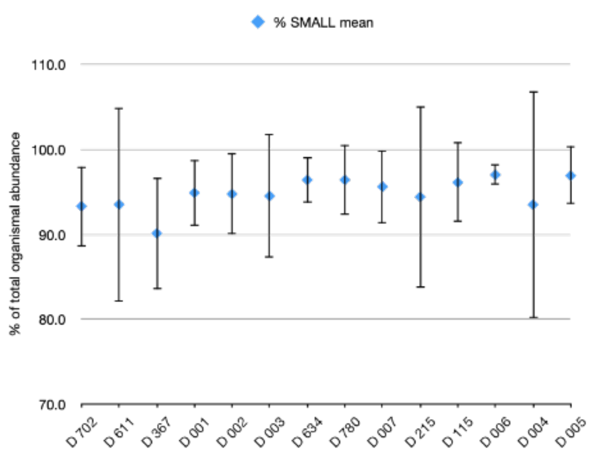
% SMALL mean
% of total organismal abundance
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100.0
90.0
80.0
70.0
D 702
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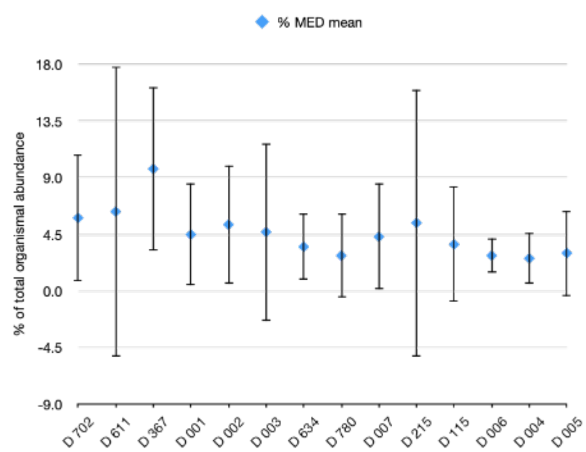
% MED mean
% of total organismal abundance
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13.5
9.0
4.5
0.0
-4.5
-9.0
D 702
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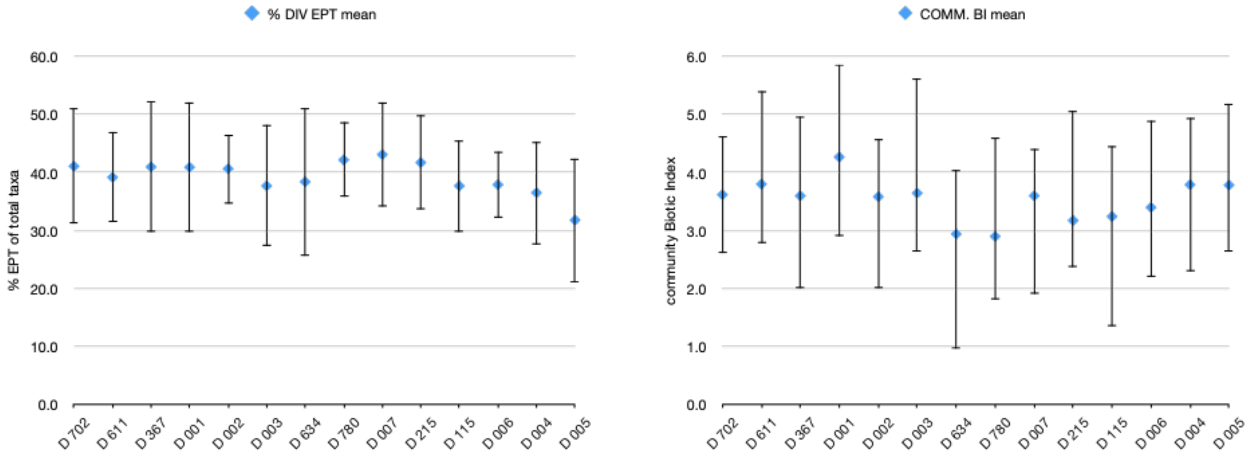
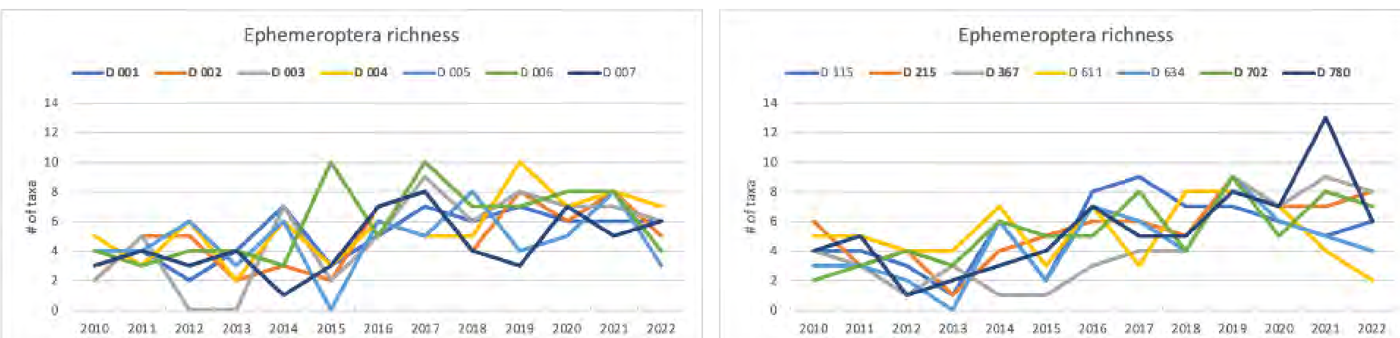


Evidence for long-term trends at individual drift sites over time

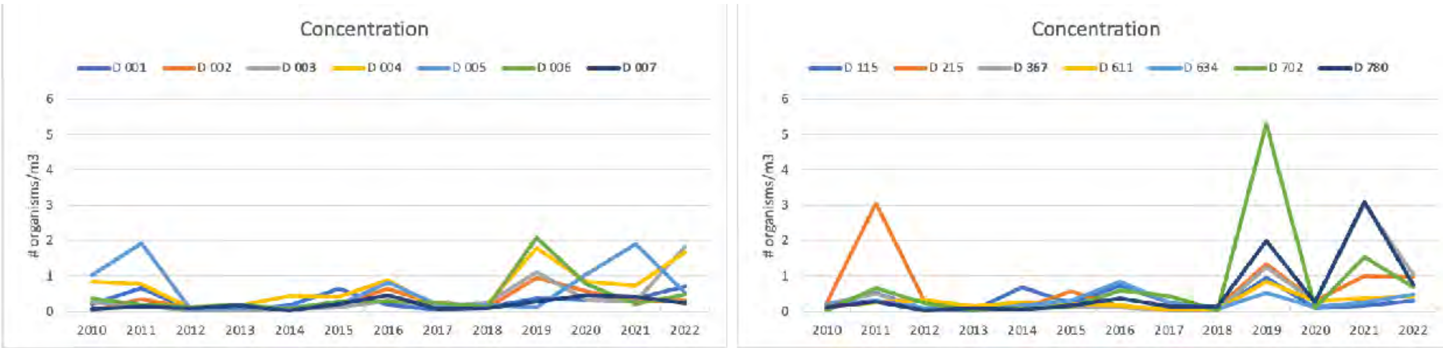
Ephemeroptera taxa richness (Figure D 3) showed a unidirectional increasing trend at the most individual drift sites (four at alpha = 0.01, eight at alpha = 0.05). There was a general overall trend towards increasing Ephemeroptera richness among most sites, especially in more recent sampling years.

Figure D 3. Ephemeroptera taxa richness at each drift site from 2010 to 2022. Bold-faced site codes in the legend are sites with significant correlations.



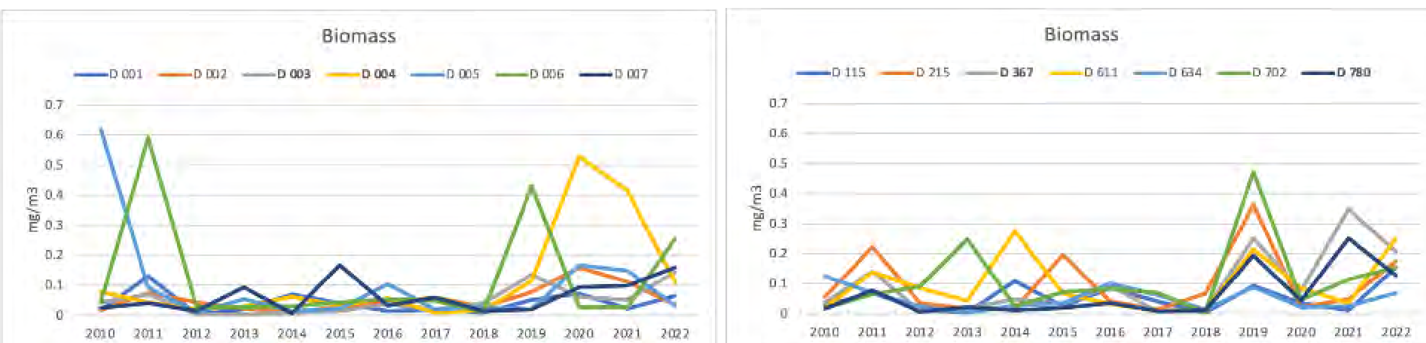
Organismal concentration (# individuals/m^3 water; Figure D 4) showed a statistically significant unidirectional increasing trend at four drift sites (alpha = 0.05). Values ranged from 0.002-5.29 individuals/m^3, which is within the range reported by Danehy et al. (2017) for drift studies done in Oregon's Calapooia River (0.7-13.7 ind./m^3). The highest concentration values at many sites occurred within the most recent sampling years (i.e., 2018-2022).

Figure D 4. Organismal concentration at each drift site from 2010 to 2022. Bold-faced site codes are sites with significant correlations.



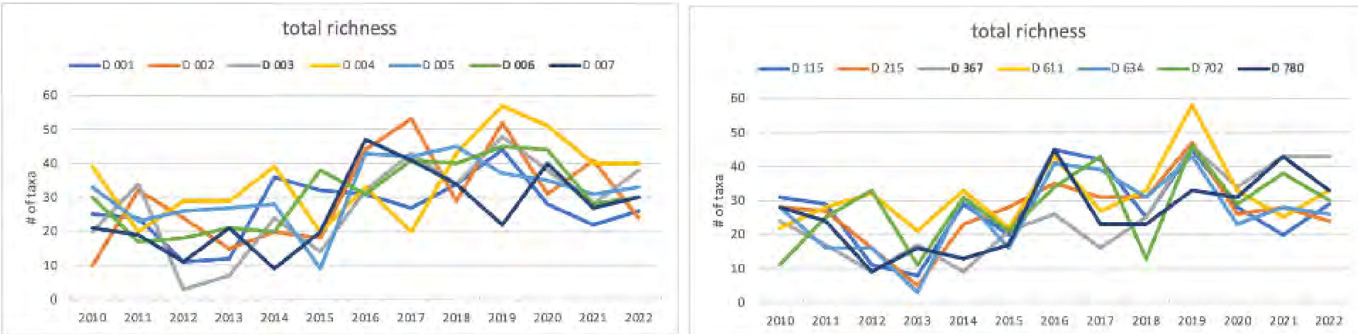
Biomass (mg/m^3 water sampled; Figure D 5) showed a statistically significant unidirectional increasing trend at four drift sites (alpha = 0.05), three of which were also sites at which concentration increased significantly. Values ranged from 0.002-0.62 mg/m^3, which is within the range reported by Danehy et al. (2017) for drift studies done in Oregon's Calapooia River (0.20-1.23 mg/m^3).

Figure D 5. Organismal biomass at each drift site from 2010 to 2022. Bold-faced site codes are sites with significant correlations.



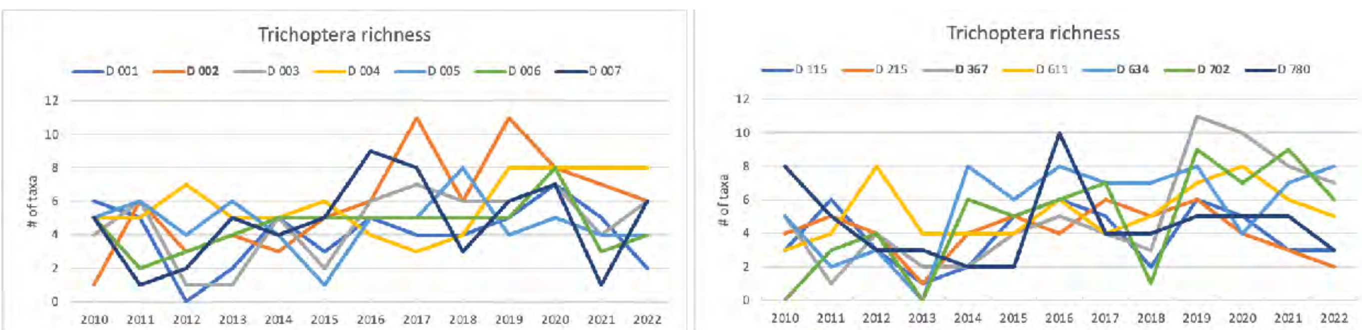
Sample richness (total taxa; Figure D6) showed a statistically significant unidirectional increasing trend at four drift sites (one at alpha = 0.01). There was a general overall increase in total sample richness among most sites, especially in more recent sampling years.

Figure D 6. Total richness (# taxa/sample) at each drift site from 2010 to 2022. Bold-faced site codes are sites with significant correlations.



Trichoptera richness (# caddisfly taxa; Figure D 7) showed a statistically significant unidirectional increasing trend at four drift sites (one at alpha = 0.01). Two of these sites also had a significant increasing trend in Ephemeroptera richness.

Figure D 7. Trichoptera richness (# taxa/sample) at each drift site from 2010 to 2022. Bold-faced site codes are sites with significant correlations. Note a trend towards increased richness at most sites.

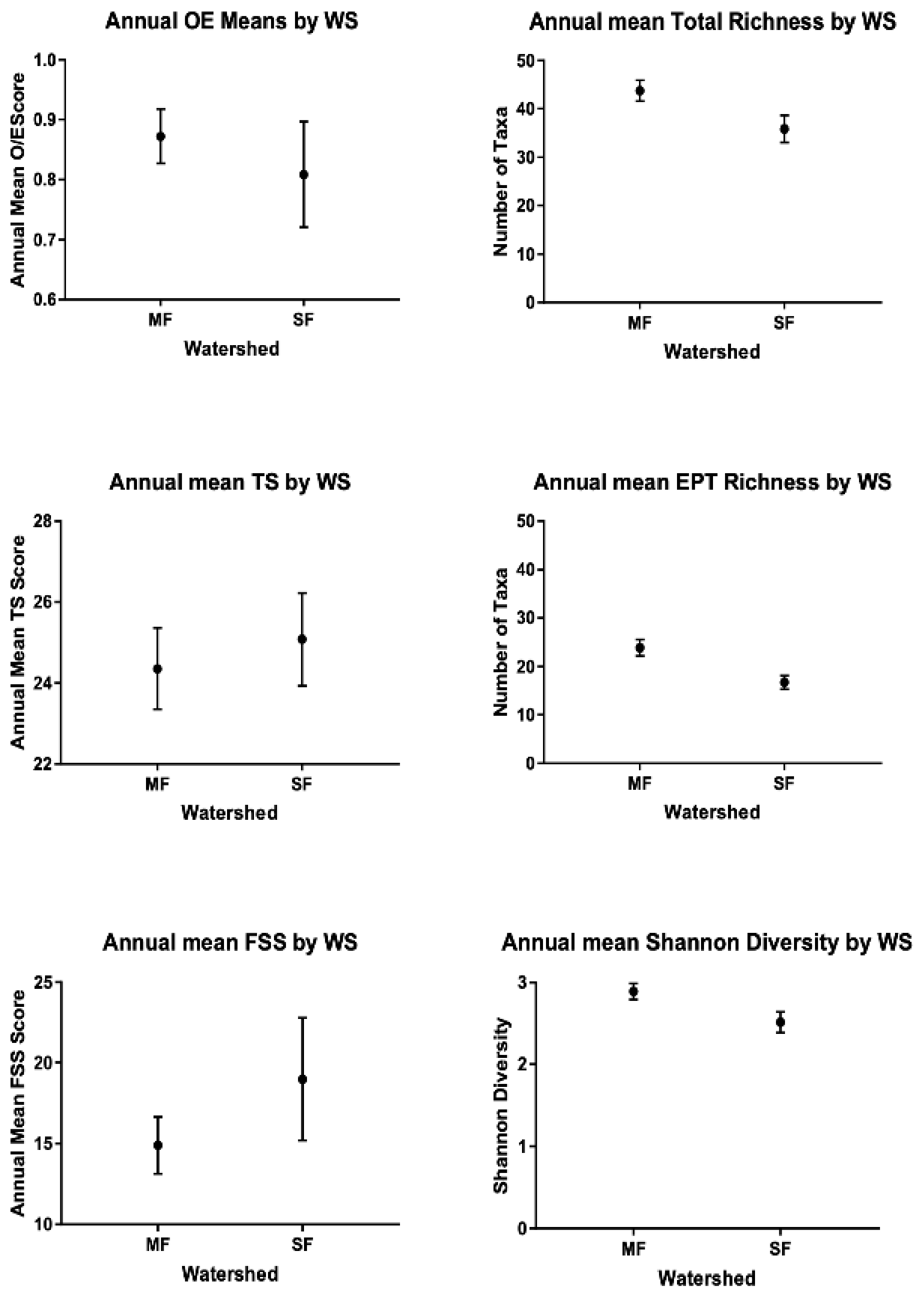


Benthic data

General comparison of benthic conditions between watersheds

O/E scores, temperature and sediment stressor scores, and taxonomic richness metrics are all significantly better (as indicated by paired two-sample t-tests on annual means from each watershed) in the Middle Fork watershed than in the South Fork watershed (Figure B 1). Five of the six t-tests were significant at alpha = 0.01. The temperature stress metric was significant only at alpha = 0.05; $p = 0.0187$). These results collectively suggest that the Middle Fork supports benthic communities characterized by more species-rich and more diverse assemblages that are less tolerant to fine sediment and thermal stress than those in the South Fork.

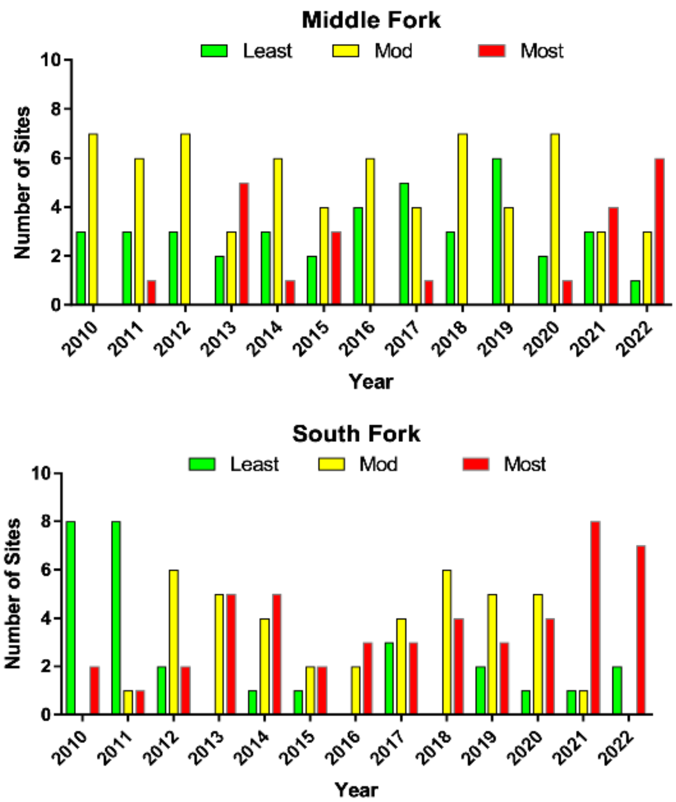
Figure B 1. Annual means (±1 SD) of six community metrics calculated watershed-wide for the Middle Fork John Day River (MF) and South Fork John Day River (SF).



PREDATOR O/E condition classes also indicate less disturbed benthic ecological conditions in the Middle Fork than in the South Fork. The Middle Fork generally includes a higher number of sites in the “least disturbed” condition class (MF mean = 3.1 sites per year versus SF mean = 2.3 sites per year), a higher number of sites in the “moderately disturbed” condition class (MF mean = 5.2 sites per year versus SF mean = 3.2 sites per year), and a lower number of sites in the “most disturbed” condition class (MF mean = 1.7 sites per year versus SF mean = 3.8 sites per year; Figure B 2).

The distribution of PREDATOR O/E condition classes also differed among years within each watershed. The distribution of O/E condition classes exhibits inter-annual variability in each watershed, and particularly in the South Fork, where a general trend of decreasing overall condition appears to potentially be occurring as indicated by a shift in the distribution of scores resulting from a decrease in the number of least disturbed scores and in increase in the number of most disturbed scores. This potential trend is most evident considering the past two years of data. The next few years of monitoring should reveal if this apparent trend continues.

Figure B 2. Distribution of PREDATOR O/E condition classes across sites in the Middle Fork John Day River (upper) and South Fork John Day River (lower) between 2020 and 2022.



<u>Evidence for long-term benthic trends in each watershed</u>

Statistically significant, long-term trends in individual measures of community condition occurred at individual sites within both the Middle Fork and South Fork watersheds. Within each watershed, correlations were run between each of seven community condition measures and year for each site, resulting in 70 individual correlations run within each watershed (Table B 1).

Owing to this large number of correlations, results are reported at both alpha = 0.01 and alpha = 0.05.

At a 0.05 significance level, 15 of 70 correlations between individual community measure and years were significant in the Middle Fork, and 14 of 70 correlations were significant in the South Fork. At a significance level of 0.01, only six of 70 correlations between individual community measure and years were significant in the Middle Fork, and only four of 70 correlations were significant in the South Fork. Across each watershed, correlation analysis did not consistently indicate improving or declining conditions, thus a closer look at individual sites was required.

Table B 1. Summary of results of Pearson Product Moment correlations run between each of seven benthic community measures and sampling year. Significance of test results is reported at both alpha = 0.01 and alpha = 0.05.

	Significance level (alpha)	Significant?		Middle Fork (10 sites)		South Fork (10 sites)	
		No	Yes	Yes MF	Condition trend	Yes SF	Condition trend
WCCP O/E	0.05	17	3	0		3	declining
	0.01	19	1	0		1	declining
TEMP STRESS	0.05	17	3	3	declining	0	
	0.01	18	2	2	declining	0	
FINE SED STRESS	0.05	16	4	1	declining	3	2 improving/1 declining
	0.01	18	2	1	declining	1	improving
TOTAL RICHNESS	0.05	14	6	2	1 improving/1 declining	4	declining
	0.01	18	2	0		2	declining
EPT RICHNESS	0.05	17	3	2	improving	1	declining
	0.01	19	1	1	improving	0	
PERCENT SHREDDERS	0.05	14	6	6	declining	0	
	0.01	18	2	2	declining	0	
SHANNON DIVERSITY	0.05	16	4	1	improving	3	improving
	0.01	20	0	0		0	
TOTALS	**0.05**	**111**	**29**	**15**	of 70 correlations at 0.05	**14**	of 70 correlations at 0.05
	0.01	**130**	**10**	**6**	of 70 correlations at 0.01	**4**	of 70 correlations at 0.01

The number of significant correlations was tallied for each site to determine which had more than one measure that was significantly trending (Table B 2). Three individual sites in the Middle Fork had at least two community measures trending at 0.05; results at two of these sites (MF-2 and MF-3) indicated mixed results, while correlation results at MF-7 indicated potentially declining benthic community conditions between 2010 and 2022 (Table B 2). Five sites in the South Fork had at least two community measures trending at alpha = 0.05; results at four of these five sites showed generally declining conditions (Table B 2These results generally suggest non-trending conditions at most sites in the Middle Fork, yet potentially declining conditions at several sites in the South Fork.

Table B 2. Summary of the number of significant correlations (at both alpha = 0.01 and alpha = 0.05) between each of seven benthic community measures and sampling year at each of the 20 long-term MFIMW sampling sites.

Number of sig correlations (of 7)

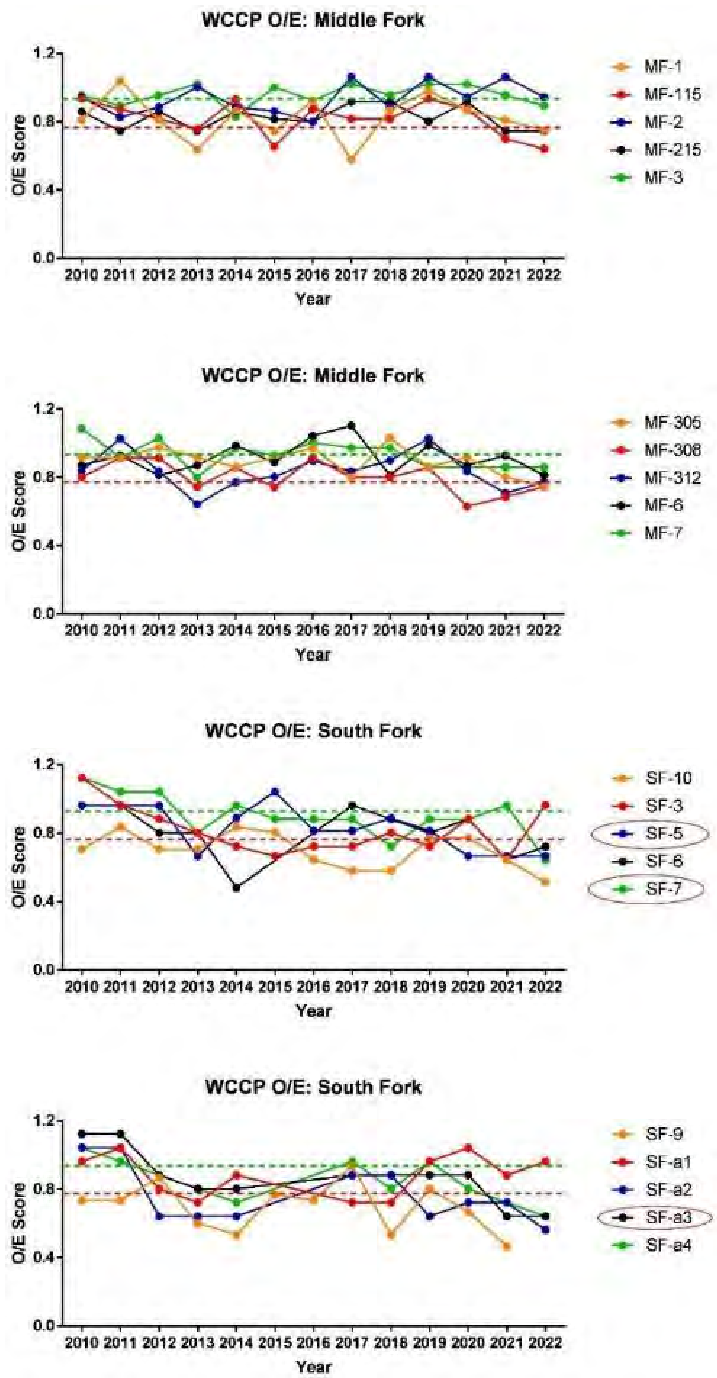
Site	alpha = 0.05	alpha = 0.01	Comments
MF-1	1	1	
MF-115	0	0	
MF-2	3	1	mixed results
MF-215	0	0	
MF-3	5	2	mixed results
MF-305	1	1	
MF-308	1	0	
MF-312	1	0	
MF-6	1	0	
MF-7	2	1	declining conditions
SF-10	1	0	
SF-3	2	0	mixed results
SF-5	4	2	generally declining
SF-6	0	0	
SF-7	3	0	declining conditions
SF-9	0	0	
SF-a1	0	0	
SF-a2	0	0	
SF-a3	2	1	declining conditions
SF-a4	2	1	declining conditions

Evidence for long-term trends at individual benthic sites over time

O/E scores generally do not show unidirectional trends at most individual sites. Results of correlation analyses between O/E scores and year were significant at one of 20 sites at alpha =

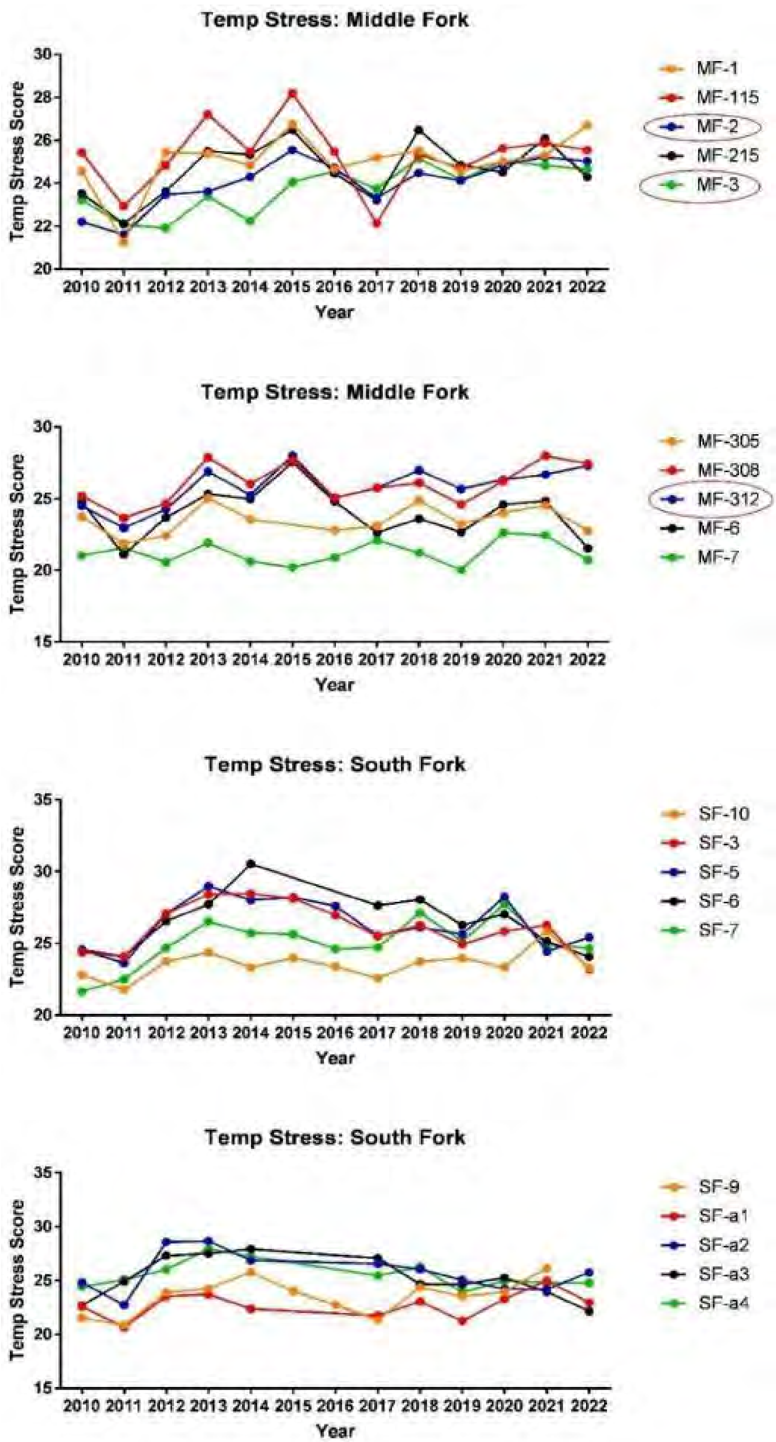
0.01 (SF-a3) and at three sites at alpha = 0.05 (SF-5, SF-7, and SF-a3). No MFIMW sites showed a significant correlation between O/E scores and sampling year (Figure B 3).

Figure B 3. PREDATOR WCCP O/E scores at each site from 2010 to 2022. Green-circled site codes are sites with significant correlations indicating improving ecological condition; red-circled site codes are sites with significant correlations indicating declining ecological condition.



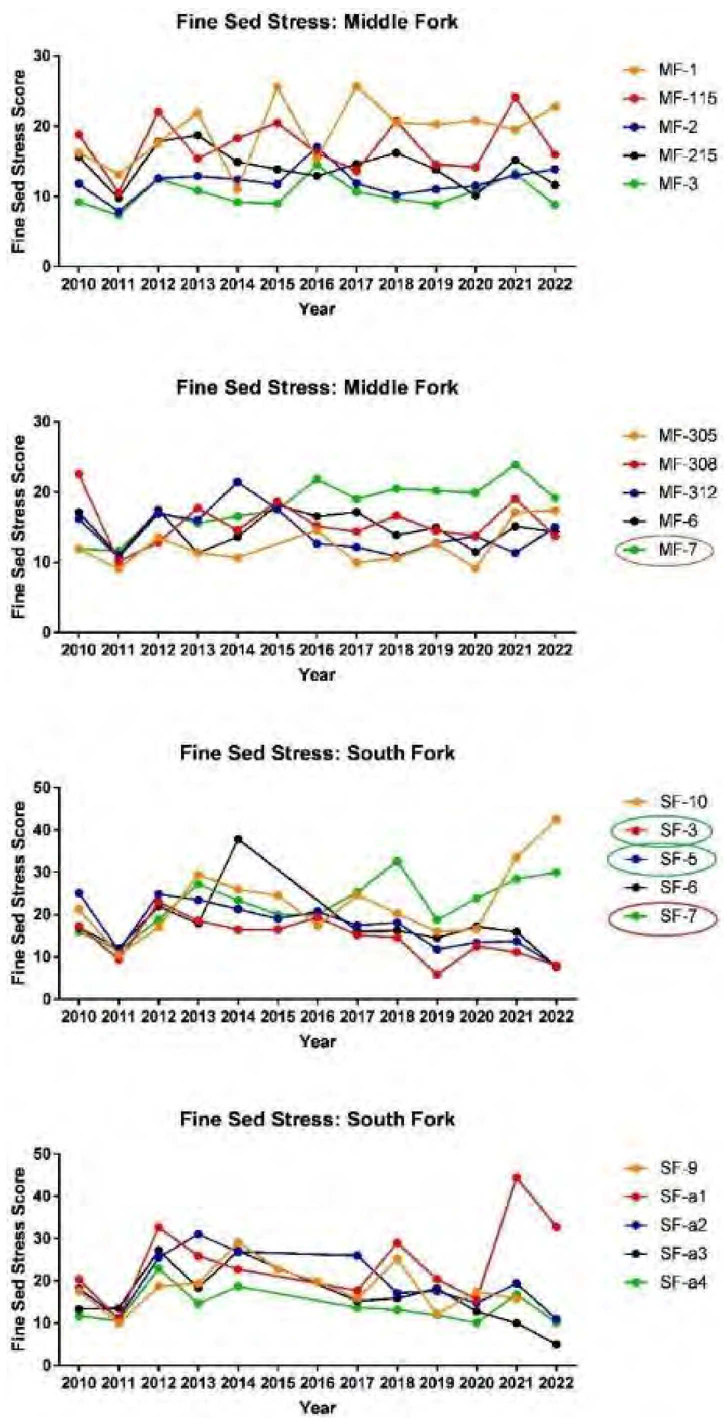
Temperature stress scores generally do not show unidirectional trends at individual sites (Figure B 4). Results of correlation analyses between TS scores and year were significant at two of 20 sites at alpha = 0.01 (MF-2, MF-3) and at three sites at alpha = 0.05 (MF-2, MF-3, MF- 312). Results suggest a statistically significant increase in temperature stress scores during the 12-year period at these sites.

Figure B 4. Temperature stress scores at each site from 2010 to 2022. Green-circled site codes are sites with significant correlations indicating improving ecological condition; red-circled site codes are sites with significant correlations indicating declining ecological condition.



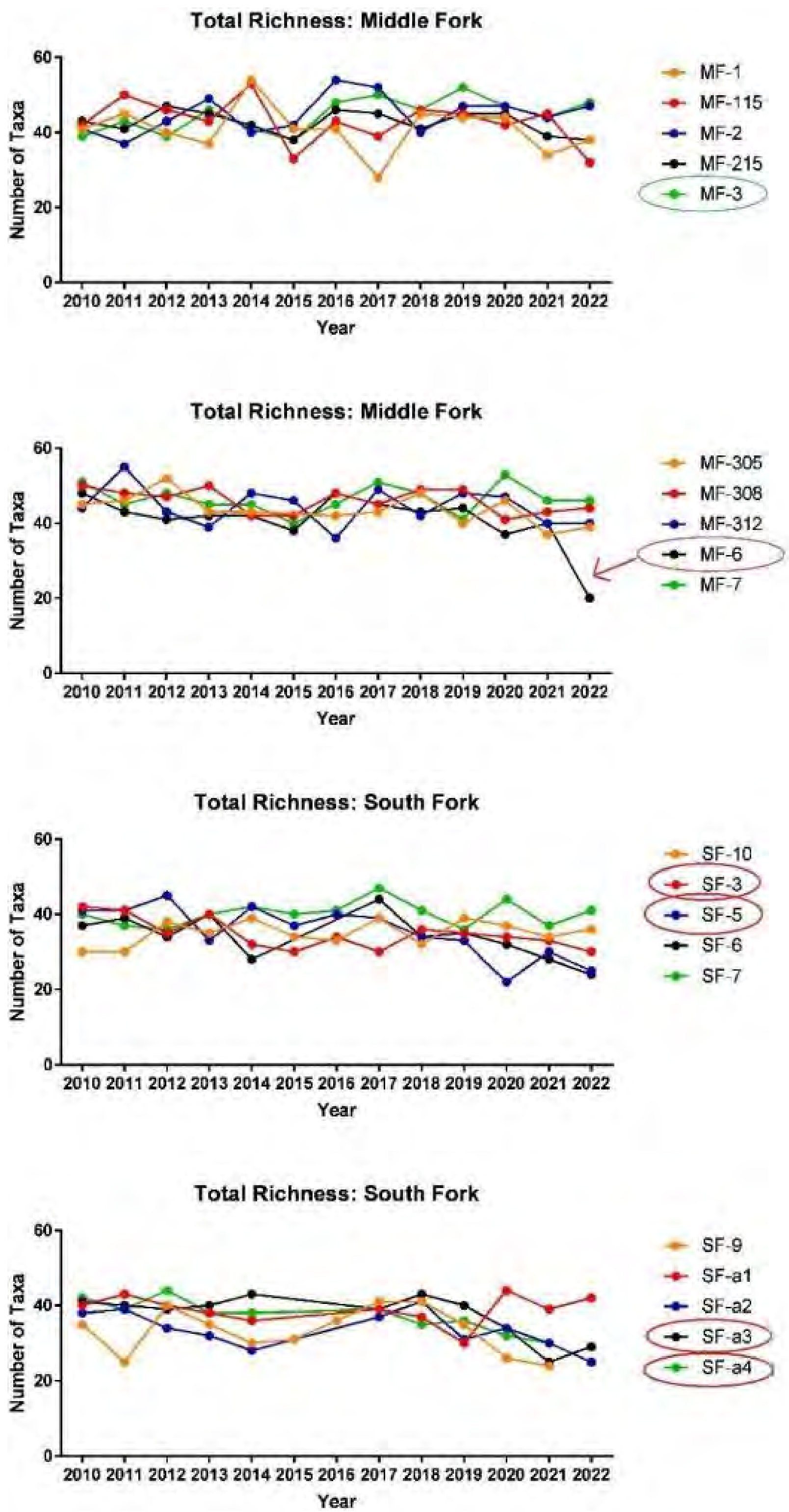
Sediment stress scores are generally not showing unidirectional trends at individual sites (Figure B 5). Results of correlation analyses between FSS scores and year were significant for two sites at alpha = 0.01 (significantly increasing scores at MF-7 and SF-5) and for four sites at alpha = 0.05 (significantly increasing scores at MF-7, SF-3, SF-5, SF-7). Correlation results at two sites (MF-7 and SF-7) suggest potentially declining conditions with respect to fine sediment stress, while results at two sites (SF-3 and SF-5) suggest potentially improving conditions with respect to this particular stressor (Figure B 5).

Figure B 5. Fine sediment stress scores at each site from 2010 to 2022. Green-circled site codes are sites with significant correlations indicating improving ecological condition; red-circled site codes are sites with significant correlations indicating declining ecological condition.



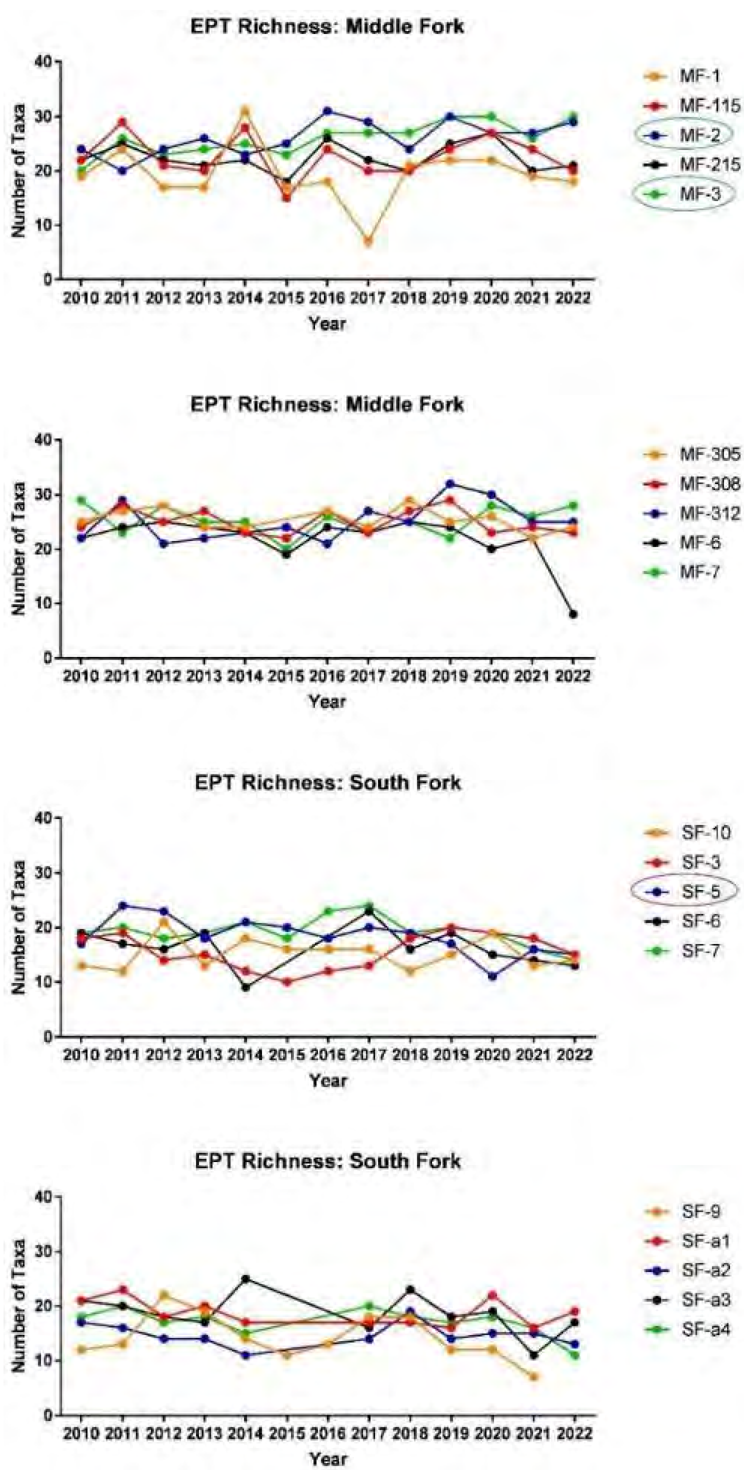
Total taxa richness generally does not show unidirectional long-term trends at most sites (14 of 20 sites at alpha = 0.05; Figure B 6). Results of correlation analyses between total richness and year were significant at alpha = 0.01 at two sites (significantly negative trends at SF-a4 and SF-5) and at six sites at alpha = 0.05 (MF-3, MF-6, SF-3, SF-5, SF-a3, SF-a4). Trends were indicative of declining ecological condition at five of 6 sites (all but MF-3).

Figure B 6. Total taxa richness at each site from 2010 to 2022. Green-circled site codes are sites with significant correlations indicating improving ecological condition; red-circled site codes are sites with significant correlations indicating declining ecological condition.



EPT taxa richness is not showing unidirectional trends at most individual sites (Figure B 7). Results of correlation analyses between EPT richness and year were significant at one site at alpha = 0.01 (MF-3, significantly increasing richness) and at three sites at alpha = 0.05 (MF-2, MF-3 and SF-5). Correlation results suggest that EPT taxa richness has increased at MF-2 and MF-3 from 2010 to 2022 and has decreased during the same period at SF-5 (Figure B 7).

Figure B 7. EPT taxa richness at each site from 2010 to 2022. Green-circled site codes are sites with significant correlations indicating improving ecological condition; red-circled site codes are sites with significant correlations indicating declining ecological condition.



Percent abundance of shredders is not showing unidirectional trends at most individual sites (Figure B 8). Results of correlation analyses between percent shredders and year were significant at one site at alpha = 0.01 (MF-305) and at six sites at alpha = 0.05 (MF-1, MF-2, MF-3, M-305, MF-308, MF-7). Results of all six significant correlations suggest a decrease in the relative abundance of shredders. These apparent trends are evidently arising from a large proportion of shredders occurring in North Fork samples at these six sites between 2011 and 2013 (Figures B 8 & B 9) and are not likely indicative of a longer-term trend occurring in the watershed.

Figure B 8. Percent abundance of shredders at each site from 2010-2022. Green-circled site codes are sites with significant correlations indicating improving ecological condition; red-circled site codes are sites with significant correlations indicating declining ecological condition.

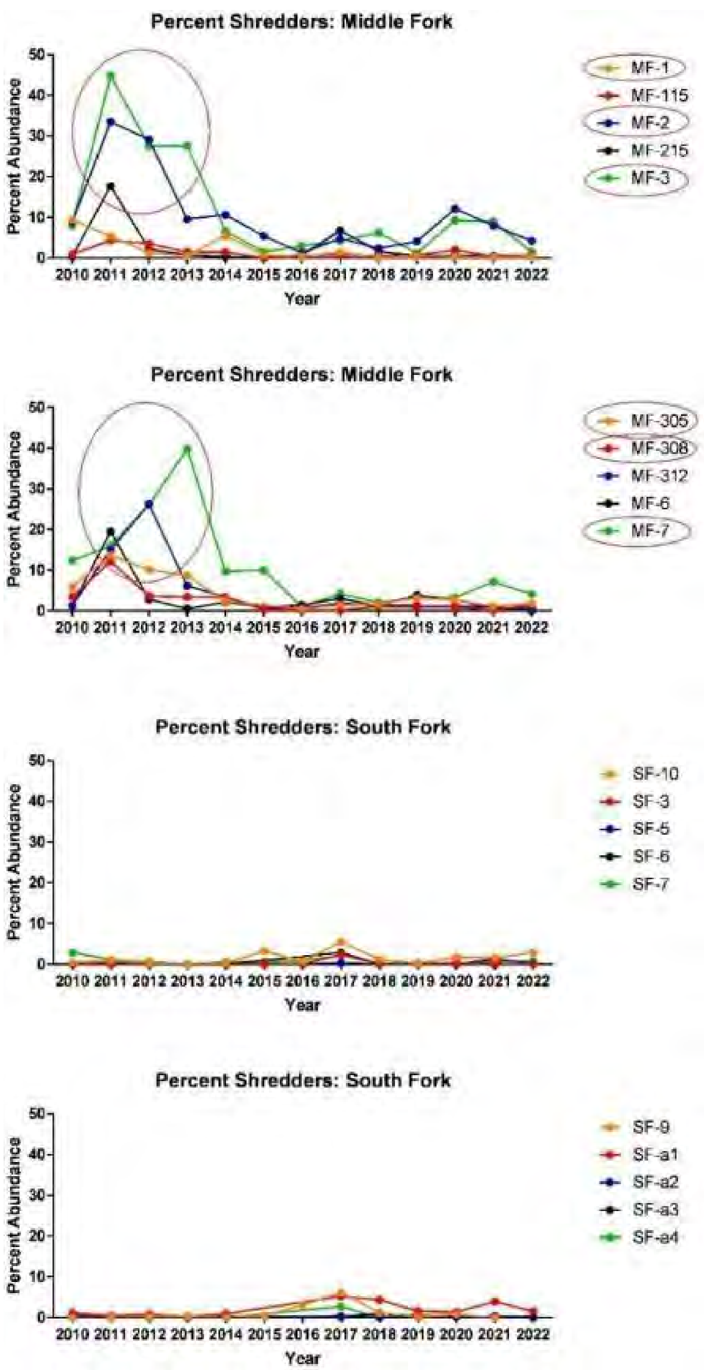
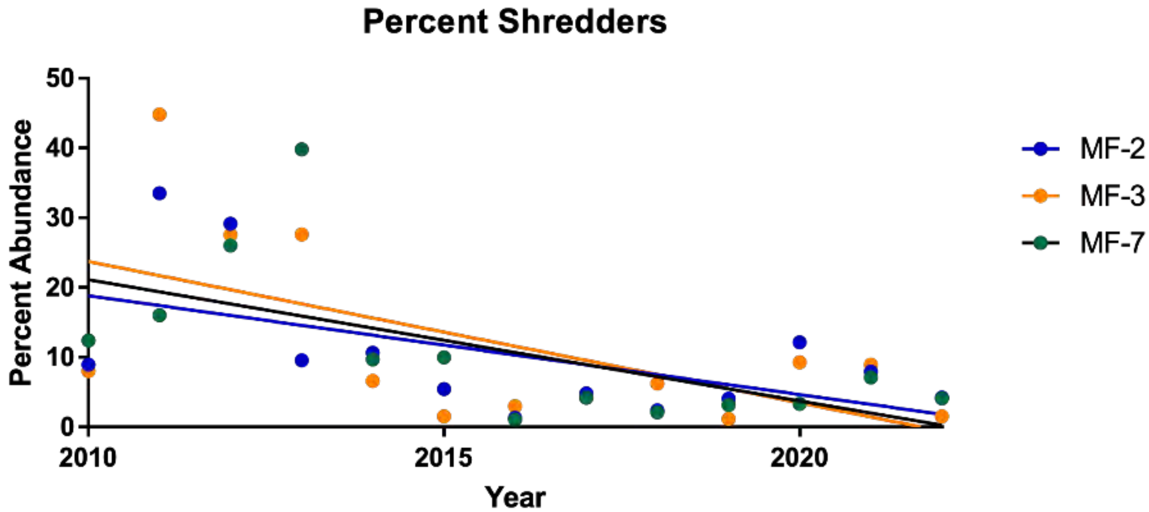
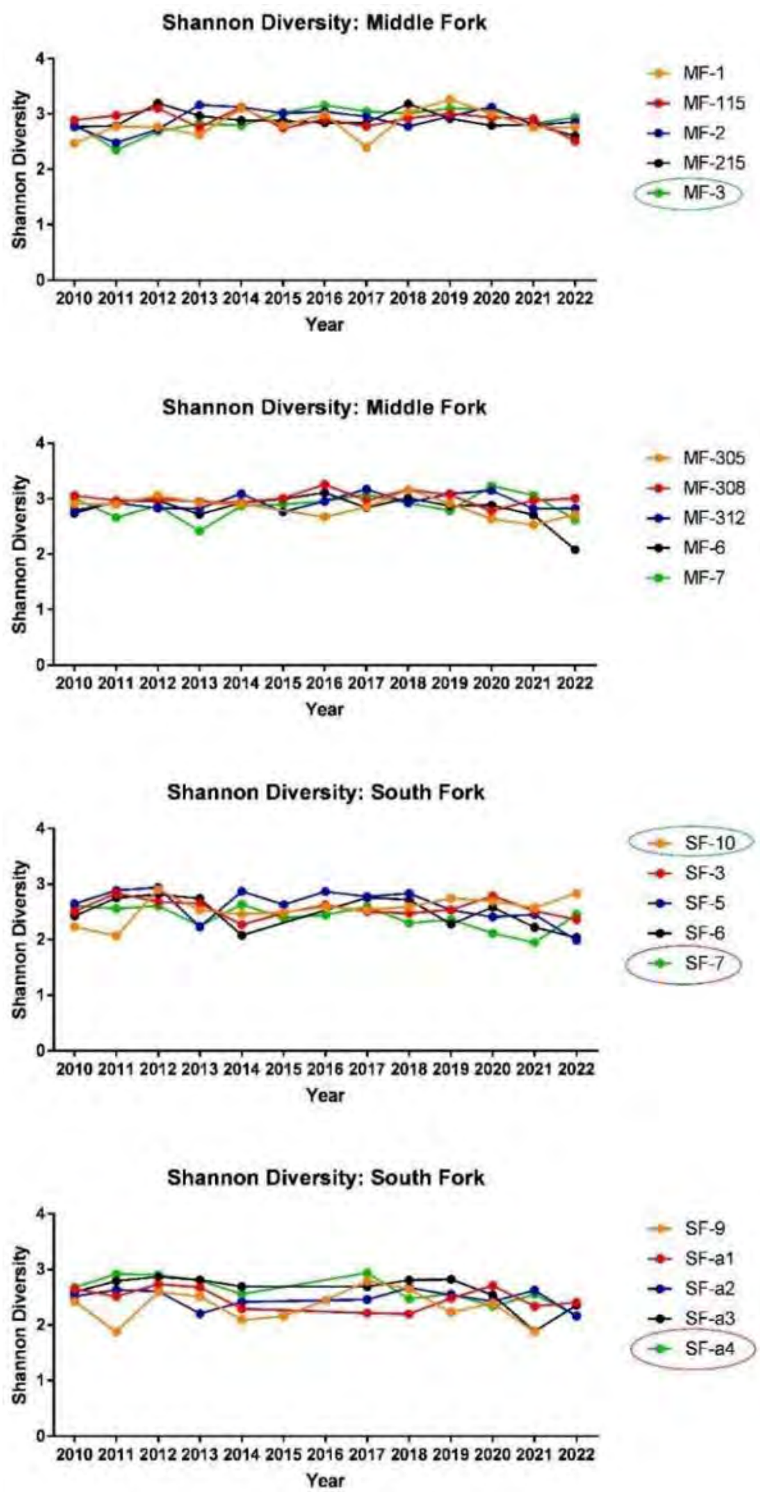


Figure B 9. Percent abundance of shredders at the three Middle Fork sites where two more community measures exhibited statistically significant trends between 2010 and 2022. Note that a large proportion of shredders occurred exclusively between 2011 and 2013, driving the significant results of the correlations.



Correlation analysis suggests that Shannon Diversity H is not showing unidirectional trends at most individual sites (Figure B 10). Results of correlation analyses between Shannon Diversity H and year were significant at no sites at alpha = 0.01 and at four sites at alpha = 0.05 (positive at MF-3, SF-10 and negative at SF-7, SF-a4)

Figure B 10. Shannon Diversity (H) at each site from 2010 to 2022. Green-circled site codes are sites with significant correlations indicating improving ecological condition; red-circled site codes are sites with significant correlations indicating declining ecological condition.



A closer look at sites showing several lines of evidence of unidirectional change

Among the 20 long-term benthic monitoring sites in the two watersheds, seven were found to have at least two community measures showing significant long-term trends at alpha =

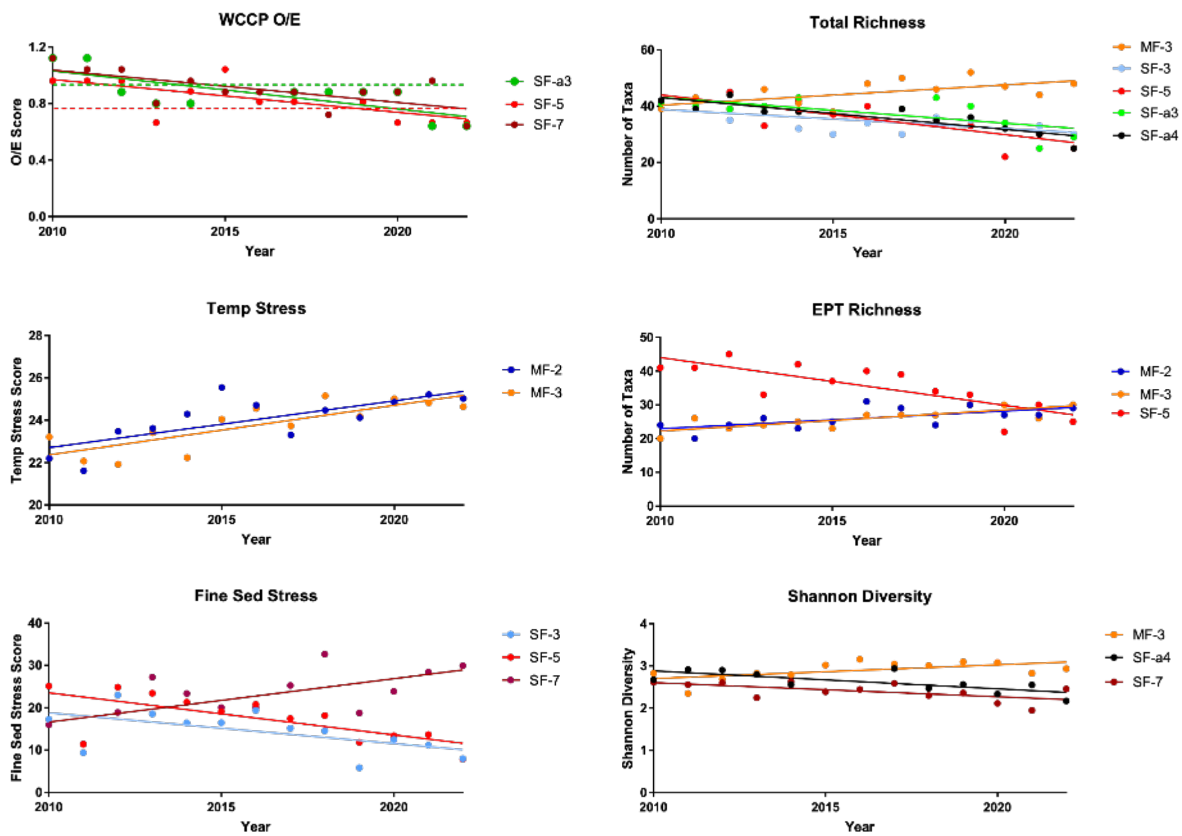
0.05 (Figure B 11). Two of these sites (MF-2, MF-3) occur in the Middle Fork, while five sites (SF-3, SF-5, SF-7, SF-a3, SF-a4) occur in the South Fork. Middle Fork site MF-2 showed mixed correlation analysis results. Temperature stress scores increased at this site, indicating potentially increasing thermal stress, but EPT taxa also increased, indicating potentially improved ecological condition (Figure B 11). Site MF-3 also showed mixed results; temperature stress scores at MF-3 increased, suggesting a potential increase in thermal stress, while EPT taxa, total richness and Shannon Diversity H all increased, collectively indicating improving community conditions (Figure B 11). PREDATOR O/E scores did not significantly increase or decrease at either of these sites over the 2010-2022 period, suggesting that overall ecological conditions at these two sites did not significantly improve or decline.

Among the five South fork sites showing at least two significant correlations, SF-3 exhibited the most ambiguous results. Here, fine sediment stress scores decreased, while total richness also decreased (Figure B 11). No significant trend in O/E scores occurred, indicating unchanged overall ecological conditions during the 12-year period. Site SF-5 correlation results indicated generally declining ecological conditions: O/E scores, EPT richness, and total richness decreased, all indicating a potential decline in benthic ecological conditions, while fine sediment stress scores also decreased, suggesting a potential decrease in fine-sediment stress (Figure B 11).

Correlation results for sites SF-7, SF-a3, and SF-a4 consistently indicated declining community conditions. At SF-7, O/E scores decreased, fine sediment stress increased, and Shannon Diversity H scores decreased. At SF-a3, both O/E scores and EPT richness decreased while at SF-a4, total richness and Shannon Diversity H both decreased (Figure B 11).

Overall, longer-term data from the Middle Fork do not consistently indicate trends in ecological conditions at any sites across the 12-year monitoring period. Longer-term data from the South Fork suggest that trends in declining ecological conditions are potentially occurring at several sites (SF-5, SF-7, SF-a3, and SF-a4). These apparent declining trends in the South Fork are largely the result of the past few years of data. Data collected in the next several years should indicate whether these apparent trends will persist. Importantly, these trends identified are only an indication of changing ecological conditions over the entire 12-year monitoring period and are not intended to be any indication of the potential effects of restoration activities in the MFIMW. Rather, these initial results allow insight into overall benthic trends in each of the watersheds and at individual sites within each watershed. Phase 2 of this analysis will examine what effect restoration activities have had on the benthic macroinvertebrate community in the context of the spatial and temporal variability and trends identified in this first phase of the MFIMW benthic macroinvertebrates analysis.

Figure B 11. Summary of statistically significant temporal trends occurring at sites with two or more significant correlations between community measures and monitoring year.



Differences in conditions among benthic sites

Each measure of community condition exhibits variability among sites within each watershed. Some of this variation appears to occur as longitudinal trends along the length of each river. Several community measures appear to trend from downriver to upriver and to show directional changes in these trends. PREDATOR O/E scores in the Middle Fork appear to trend upwards from MF-308 to MF-3 and then trend downward from MF-3 to MF-115 (Figure B 12). Temperature stress scores appear to show similar trends in the Middle Fork: TS scores trend downward (improve) between MH-308 and MF-7 and then trend upward between MF-7 and

MF-115 (Figure B 13). Fine sediment stress scores also appear to follow this general trend pattern in the Middle Fork, but the trends are punctuated with an apparent outlier condition at MF-7 (Figure B 13).

Trends in richness metrics in the Middle Fork are more subtle, but both metrics indicate a slightly decreasing trend in taxa richness and EPT richness between MF-7 and the most upriver sites (Figure B 14). Percent shredders was the most variable community measure evaluated in the Middle Fork, both among sites and across years. Mean values and inter-annual variability in percent shredders both generally increased from MF-308 to MF-3 and then decreased from MF-3 to MF-115 (Figure B 15).

In the South Fork, PREDATOR O/E scores appear to remain similar through much of the length of the river, with a noticeable decrease in O/E scores at SF-a2 and at the two most upriver sites, SF-9 and SF-10 (Figure B 12). Both total richness and EPT richness appear lowest at these three sites in the South Fork as well (Figure B 14). Temperature stress scores appear to show a general decrease from downriver to upriver (Figure B 13), suggesting generally lower thermal stress in the reaches further upriver on the South Fork. Contrary to an apparent general decrease in thermal stress, sediment stress scores generally increase from downriver to upriver in the South Fork (Figure B 13). The value of the percent shredder individuals measure was generally low in the South Fork, and did not vary appreciably among sites (Figure B 15). The considerably lower relative abundance of shredders in the South Fork than in the Middle Fork suggests that coarse particulate organic matter such as leaves, needles, and other vegetative debris is a less common food source in the South Fork than in the Middle Fork and may relate to different riparian conditions along the two rivers.

Figure B 12. Mean (±1 SD) PREDATOR O/E scores from each monitoring site in the Middle Fork John Day River (left) and South Fork John Day River (right).

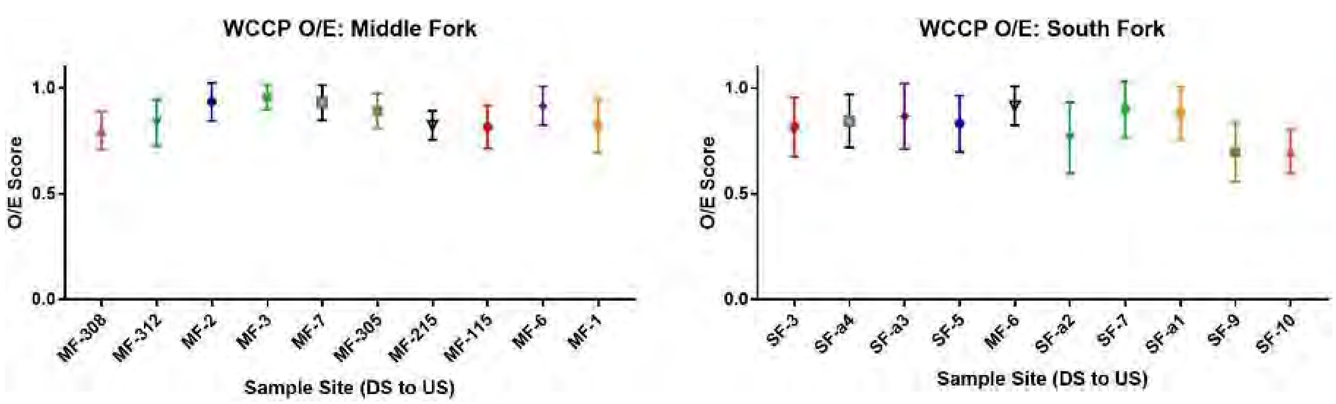


Figure B 13. Mean (+1 SD) temperature (upper) and fine sediment (lower) stressor scores from each monitoring site in the Middle Fork John Day River (left) and South Fork John Day River (right).

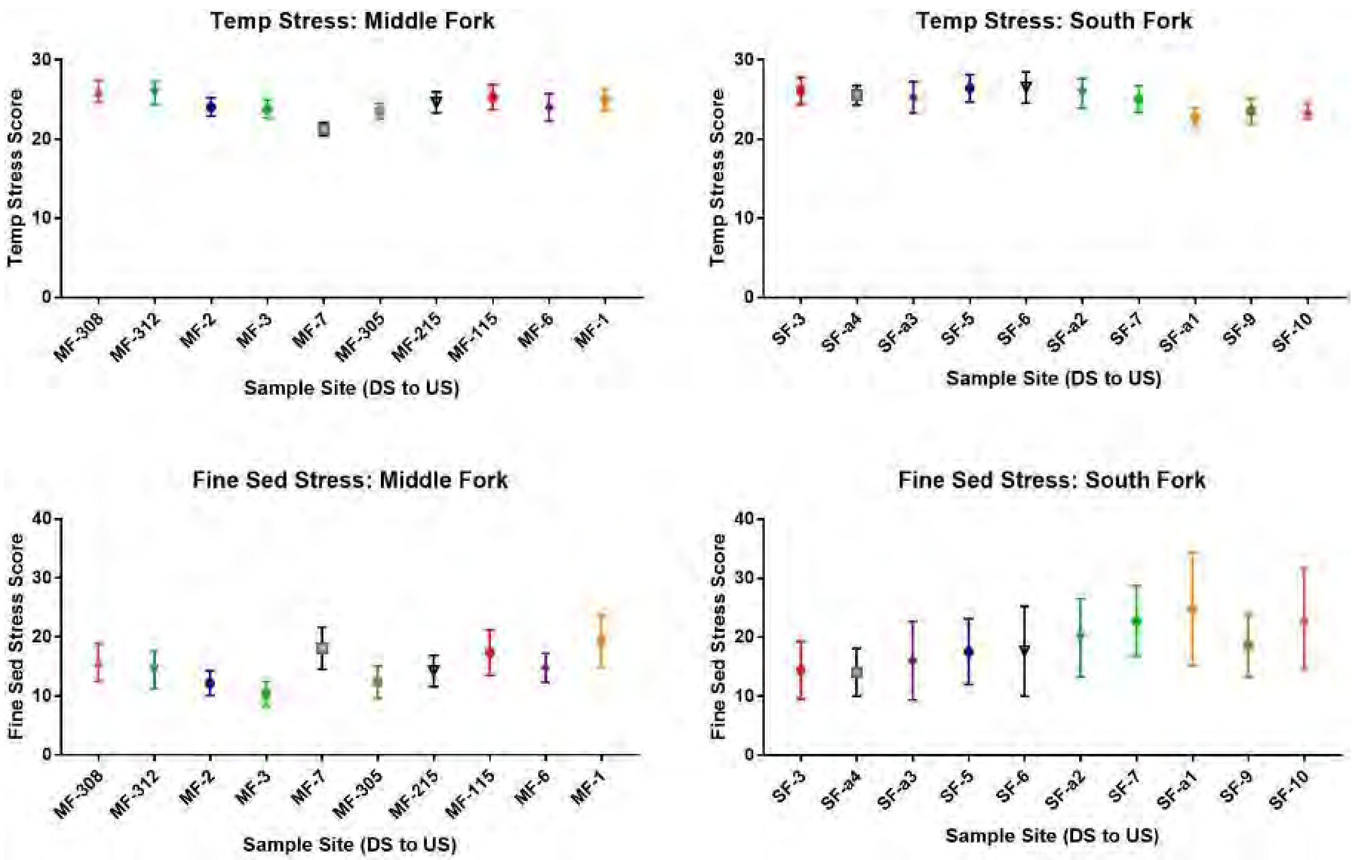


Figure B 14. Mean (±1 SD) total taxa richness (upper) and EPT taxa richness (lower) from each monitoring site in the Middle Fork John Day River (left) and South Fork John Day River (right).

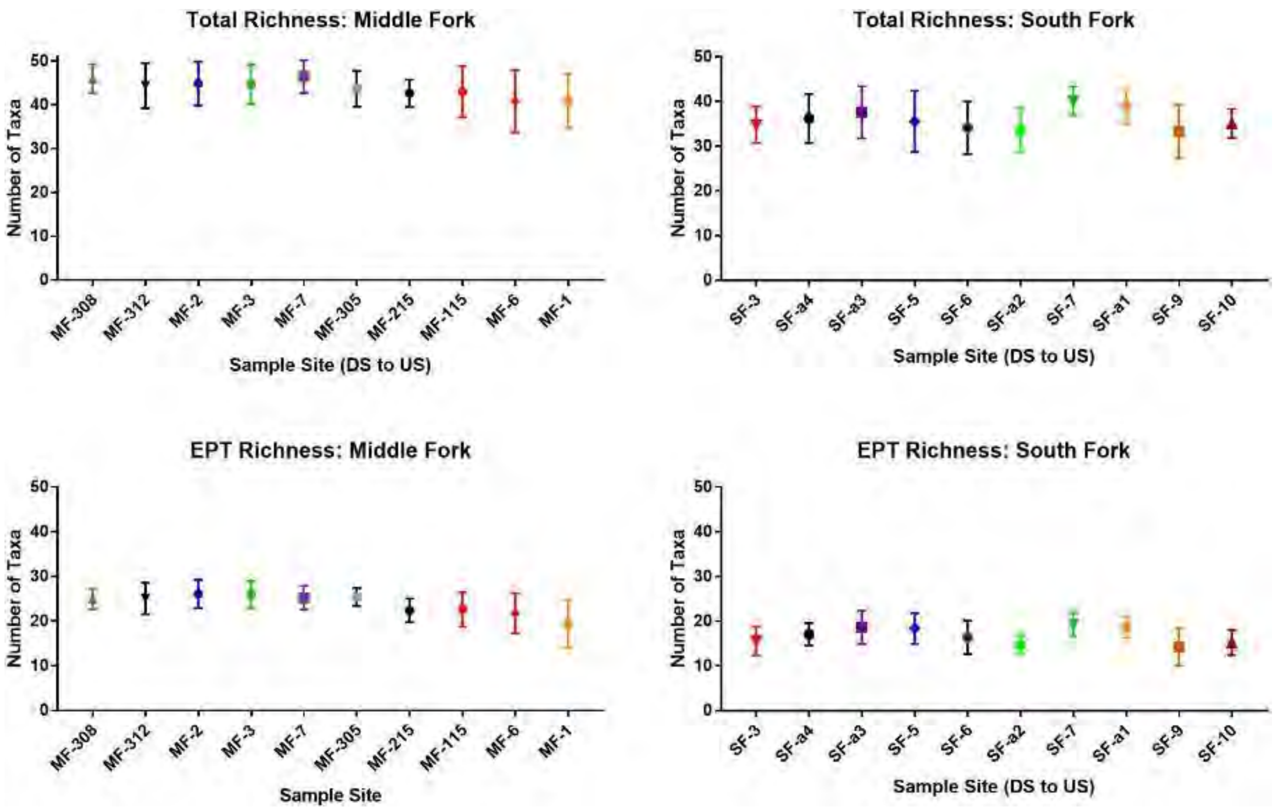


Figure B 15. Mean (±1 SD) percent abundance of shredder individuals from each monitoring site in the Middle Fork John Day River (left) and South Fork John Day River (right).

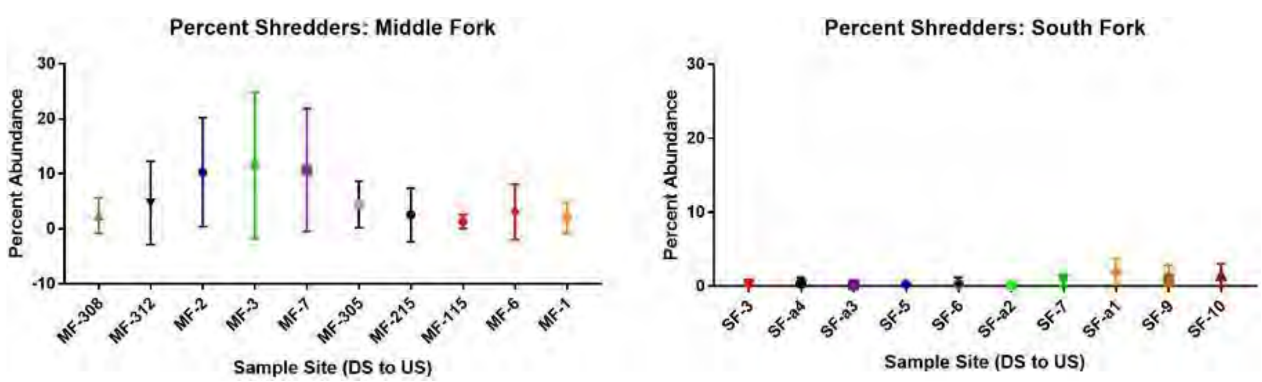
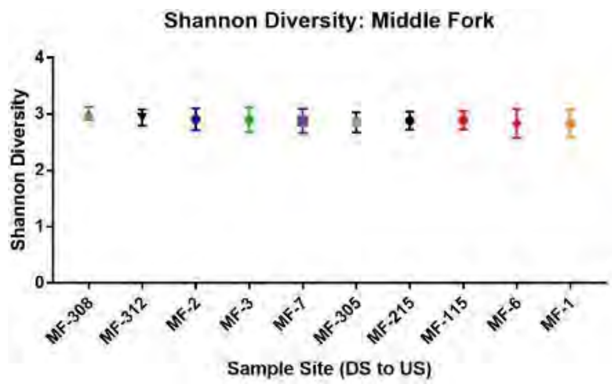
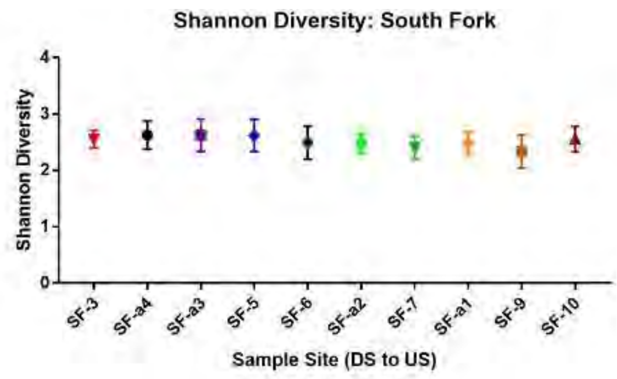


Figure B 16. Mean (±1 SD) Shannon Diversity Index H scores from each monitoring site in the Middle Fork John Day River (left) and South Fork John Day River (right).





General comparison of trends between spatially related drift and benthic samples

All drift samples were taken in the Middle Fork watershed, so between-watershed comparisons are not possible. However, because several benthic and drift samples were taken in the same or closely contiguous reaches, between-site comparisons can be made. Fewer benthic sampling sites had significant trends in any community measures; only three Middle Fork benthic sites, MF-2, MF-3, and MF-7, had significant trends in community measures, and trend directions indicated mixed results. MF-2 showed increased temperature stress scores, indicating potentially increasing thermal stress, but the increased EPT taxa trend at the same site suggested improved ecological conditions (Figure B 11). MF-3 also had increased temperature stress scores, suggesting a potential increase in thermal stress; however, total and EPT taxa richness and Shannon Diversity H all increased, collectively indicating improving community conditions (Figure B 11). The D 003 drift site is in the same reach as MF-2 and downstream of MF-3; this site had significant trends indicating improved conditions for four metrics, with increased concentration, biomass, total richness and Ephemeroptera richness, all at alpha = 0.05.

Correlation results at the benthic MF-7 site indicated potentially declining benthic community conditions between 2010 and 2022, with significant decrease in percent shredder organisms and increased sediment stress. This site is located slightly downstream of the D 007 drift site, which had a statistically significant correlation in just a single metric, concentration (increasing), which suggests potentially improving conditions.

SUMMARY

Drift data

Drift behavior is influenced by both biotic and abiotic factors; drift activity also varies by season and with the body size and life stages of individual taxa within the macroinvertebrate community. There are several types of drift behavior associated with aquatic invertebrates: catastrophic drift, where individuals are dislodged as the result of a stressor or disturbance, i.e., abrupt changes in flow, sedimentation, pollution; passive (accidental) drift, where organisms are dislodged due to changes in physical conditions of the stream; and active (intentional) drift, which is initiated by the organism for a variety of reasons, i.e., to avoid predation, find new food resources, etc. (Waters, 1972; Brittain & Eikeland, 1988; Wooster & Sih, 1995; Svendsen et al., 2005). Active drift often has a diel periodicity; many taxa drift more actively at night, potentially to avoid predation, while others drift more actively during the day.

The drift data collected for the Middle Fork must be approached with a certain level of caution, as there is substantial variation in sampling across the 12-year period. Drift samples were collected across a wide range of times in each year, varying from as few as two hours to as much as 24 hours. They were also collected at different times of day, from mid-morning to afternoon to evening; given the diel periodicity of drift for different taxa, these sampling periods may have captured different populations of macroinvertebrate taxa based on timing of sampling.

The following Phase 1 questions were posed for the drift macroinvertebrate data:

How does macroinvertebrate biomass change through time and space at each individual site?

Statistically significant unidirectional trends for biomass over the 2010-2022 sampling period were seen for four of the 14 drift sampling sites (D 003, D 004, D 367, and D 780), all with an increasing trend. A statistically significant trend of increasing organismal concentration was also seen for three of these four sites (D 003, D 367, and D 780). The sites at which biomass increased significantly are not closely related spatially, as they are distributed along the entire stream sampling length, and they have also experienced different types, extents, and timing of restoration.

How does macroinvertebrate community composition change through time and space?

Analyses of within-site changes of drift sample community composition across time revealed few patterns. Because drift samples will necessarily capture macroinvertebrate taxa that are more likely to become entrained in the drift, a high degree of similarity in community composition may be expected from year to year at the same site. Most samples had a high abundance of aquatic Diptera (true flies; especially Chironomidae and Simuliidae), terrestrial Hemiptera (true bugs), Ephemeroptera (mayflies; especially Baetidae and Ephemerellidae), and Trichoptera (caddisflies; especially Helicopsychidae and Lepidostomatidae). This generally accords with findings from other studies regarding the taxa most likely to be collected in drift samples (Waters, 1972; Brittain & Eikeland, 1988). The most consistent trend noted from community analyses was that community composition in samples from earlier years (i.e., 2010- 2015) at individual sites tended to differ more from each other and from samples taken in later years, while samples taken in the latter part of the project period (i.e., 2017-2022) often had higher overall average similarity (>60%). When years in which restoration at individual sites was conducted were considered, there was no clear pattern of before/after change.

In addition, there was no consistent relationship between the degree of restoration at an individual drift site and the number of statistically significant trends observed in community metrics. For example, a significant trend in the greatest number of community metrics was seen at a site that had experienced only passive restoration in a single year (eight metrics; D 367), while a site that experienced extensive active and passive restoration in multiple different years had a statistically significant trend in only a single metric (D 007). Thus, while community composition has changed among all sites over time regardless of the number, type, or extent of restoration projects, it is not possible at this point to relate changes in overall community composition to restoration activities, especially in the absence of data regarding changes in physical habitat.

Are there are similarities between drift and benthic samples?

Drift samples are usually collected to provide a measure of macroinvertebrate production and food availability for fish, or to characterize taxon-related differences in entrainment, and are not generally analyzed for the same community measures as benthic data. In addition, due to the presence of many terrestrial taxa collected in drift samples (i.e., bugs, beetles, caterpillars, wasps, aphids, spiders), community composition between drift and benthic sites that are co-located or adjacent would be expected to diverge widely. However, community metrics applicable to the drift data were calculated and subjected to trends analysis to facilitate comparisons between the drift and benthic datasets. More statistically significant unidirectional trends across time were seen for drift than for benthic samples, and most observed trends in drift community metrics were suggestive of improving habitat conditions. Many fewer significant trends were seen for benthic community metrics in the Middle Fork, with some indicating mixed results or declining habitat conditions. In addition, there was no concordance between the type and direction of trends that were seen at benthic sites and spatially related drift sites.

At this point, none of the trends in drift community metrics or changes in community composition can be directly correlated to restoration activities. In Phase 2, the proximity and timing of restoration activities at individual sites will be examined to enable us to create specific hypotheses regarding if and how restoration outcomes would be expected to impact drift invertebrate communities. This will include examining available physical habitat data, such as PIBO, that is co-located or in close proximity to drift monitoring sites. Habitat data that suggest improving physical conditions in the MFJDR would allow for more confidence in expectations of improving biological conditions in response to restoration activities. However, evidence that physical conditions have not meaningfully changed following restoration would temper expectations for changes or improvements in macroinvertebrate community composition or metrics. If Phase 2 analysis reveals trends or changes potentially related to restoration activities, individual site data will be examined more closely to determine the ways in which the community shifted and how those shifts (i.e., functional feeding groups, sensitivity, contributions to biomass) may be related to expected restoration outcomes.

Benthic data

The following Phase 1 questions were posed for the benthic macroinvertebrate data:

Are there differences or overall trends between or across watersheds (MFJDR and SFJDR)?

Benthic macroinvertebrate communities in the MFJDR are generally in better condition than those in the SFJDR, as indicated by significant differences in several community measures between the two watersheds. O/E scores, temperature and fine sediment stressor scores, and taxonomic richness metrics were all significantly higher in the MFJDR than in the SFJDR. Furthermore, correlation analyses suggested that benthic macroinvertebrate conditions are potentially declining at several locations in the SFJDR, while those in the MFJDR have generally remained stable over the 12-year monitoring period, showing obvious evidence of neither decline nor improvement.

How do macroinvertebrate communities change through time at each individual site in the MFJDR?

Benthic communities in the MFJDR generally do not exhibit trends in community characteristics or conditions at most sites over the 12-year monitoring period. Some evidence of trending is occurring at three sites in the MFJDR, but the results were ambiguous at two sites (MF-2 and MF-3) and indicated a potential decline in macroinvertebrate community condition at one site (MF-7). Data suggest that benthic community conditions at most sites in the MFJDR have remained unchanged during the 12-year monitoring period. In contrast, data suggest that benthic community conditions have potentially declined at four of 10 sites in the SFDJR, as two or more measures of community condition showed significantly worsening trends over the 12- year period at each of these four sites.

How do macroinvertebrate communities vary across sampling sites for a given year in the MFJDR?

Benthic community conditions varied among individual sample sites in the MFJDR, as each measure of community condition exhibits variability among sites. Some of this variation appears to occur as longitudinal trends along the length of the river: O/E scores, temperature stress scores, and fine sediment stress scores all indicated that conditions potentially improve from the lower MFJDR sites to the middle sites and then potentially decline between the middle and upper MFJDR sites. Within most years, macroinvertebrate community conditions in the MFJDR, based on PREDATOR O/E scores, ranged from “least disturbed” to “most disturbed”, while in four of the monitoring years, conditions ranged only between “moderately disturbed” and “most disturbed”. While Phase 1 identified and characterized this variation, phase 2 of these analyses will seek to elucidate why such variation in condition occurs among sites.

For each site in the MFJDR and SFJDR, in which direction is the O/E score trending?

PREDATOR O/E scores showed no significant temporal trends at MFJDR sites. O/E scores showed significant trends at only three SFJDR sites during the 12-year period and were significantly declining at all three. While Phase 1 analysis results generally suggest that MFJDR benthic community conditions have largely remained unchanged over the 12-year monitoring period, Phase 2 will more closely examine these data for site-specific trends and changes in relation to restoration and enhancement activities. At each monitoring site, the proximity and timing of implementation of restoration activities will be determined to enable specific predictions to be made about when and how macroinvertebrate communities could be expected to change in response. The data will then be more closely examined with consideration to the likely timing, nature, and magnitude of biological responses.

Because restoration activities are implemented at different times, are of different sizes, and are expected to accrue different physical, hydrologic, and geomorphic changes, the size and nature of biological responses could be expected to vary widely among individual MFJDR sites. For Phase 2, we will first assess each site for its potential to accrue improving biological conditions. This will include examining any available physical habitat data (PIBO or other) that is co-located with or in close proximity to benthic monitoring sites. Habitat data that can show evidence of improving physical conditions in the MFJDR would allow for more confidence in expectations of improving biological conditions in response to restoration activities. However, evidence that physical conditions have not meaningfully changed or improved following restoration would suggest that expectations for improvements in biological conditions to date should be tempered.

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Appendix D: JDBP Riparian Planting Workshop Summary

Contents

INTRODUCTION

The John Day Basin Partnership's riparian planting adaptive management pilot project began with the recognition that elevated stream temperature is a significant salmonid population limiting factor, and riparian planting is one strategy to reduce solar input and thus stream temperatures. Riparian plantings have occurred throughout the basin using a wide range of methods. The objectives of the pilot project were to evaluate and learn from previous planting projects, provide recommendations for future planting projects, identify knowledge gaps and discuss ways to address them, and identify riparian planting technical guides and other resources.

The November 8, 2023 workshop provided an opportunity for people working in the John Day Basin to collectively review what they've learned through research, monitoring, and field experience with riparian plantings and make recommendations for improving future riparian planting projects. Workshop participants included knowledge experts from the Confederated Tribes of Umatilla Indian Reservation, Confederated Tribes of Warm Springs, Gilliam Soil and Water Conservation District, Malheur National Forest, Monument Soil and Water Conservation District, Oregon Department of Fish and Wildlife, Oregon State University, North Fork John Day Watershed Council, and South Fork John Day Watershed Council.[1]

The morning session was a roundtable discussion of project implementers who shared their lessons from experience with riparian plantings in the John Day Basin. Roundtable participants were asked to address the following questions:

1. What is your experience with riparian plantings to provide stream shade in the John Day Basin?
    a. Site conditions (generally, e.g. mine tailings)
    b. Planting specs (species planted, fencing, irrigation, etc.)
    c. Implementation
    d. Monitoring specs and results (where available))
2. What have you found improves riparian planting survival and growth rates? (What are your top 3-5 riparian planting success factors?)
3. What are things to watch out for (that reduce planting success)?
4. Where do you wish you had more information (knowledge gaps)?
5. What planting guide(s) do you use/recommend?

The afternoon session included presentations from botany and restoration ecology scientists who addressed broader considerations and related research.

This report provides summary recommendations for people planning and implementing riparian restoration projects to improve native fish habitat in the John Day Basin, drawn from lessons learned from the workshop presentations and discussions. It also includes a summary of outstanding uncertainties that could be addressed through future research synthesis or monitoring and next steps for the John Day Basin Partnership.

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[1] Workshop participants: Javan Bailey – NFJDWC, Kristen Walz – NFJDWC, Herb Winters - Gilliam SWCD, Amanda Hardman - Malheur NF, Ann Moote - Mamut Consulting, Nicole Lexson – CTWS, Erik Rook - Monument SWCD, Cody Lund - Malheur NF, Becky Long - Malheur NF, Dan Armichardy - Malheur NF, John Clark – CTUIR, Roger Lathrop - Gilliam SWCD, Zach Cunningham – CTWS, Robert Warren – BEF, Lindsy Ciepiela – ODFW, Ian Tattam – ODFW, Adrienne Averett – ODFW, Josh Averett – OSU, Hannah Latzo - SFJDWC

SUMMARY RECOMMENDATIONS

Start by assessing site conditions

- Hydrology: floodplain connectivity, depth to groundwater, high energy areas/flow rates
- Soil conditions (disturbance, especially compaction; productivity)
- Species already growing on or near the project site (desirable and undesirable)
- Evidence of ungulates and rodents (browse)

Prep site for planting success

- Connect floodplain, raise water table
- Treat weeds / competing plants
- If appropriate improve soils; avoid heavily compacted soils
- Add roughness to protect plants during high flow and improve sediment deposition, retention
- Dig planting trenches in streambank to get roots to water and improve deposition and retention
- Add browse protection

Choose plants appropriate to site conditions and project goals

- Favor species already growing on or near the site
- Match plants' water requirements to hydrologic conditions
- If a primary goal is stream shade, favor faster-growing species
- If the project goal is to improve fish habitat, consider protecting bryophytes
- Consider early seral plants (forbs, annual grasses, sedges)
- Consider bryophytes

Communicate objectives and outcomes

- Temper expectations around short-term outcomes

LESSONS LEARNED

Improve hydrologic conditions to ensure adequate water for plants.

- Research has shown that proximity to water and access to water in the driest summer periods are the greatest limiting factor to riparian plants east of the Cascades.
- In most cases, plants with roots reaching the water table have the highest rates of survival and growth.
    - Plant survival and growth are better when water table is higher.
    - Willows and cottonwoods need to reach the water table.
    - Digging a hole or trench to the water table and planting there trench was effective.
    - Irrigation challenging and less effective.
- Fixing hydrological processes before planting improves outcomes.
    - Reconnect the floodplain to raise the water table.
    - Getting water up on the floodplain and dissipating the energy helps reduce plant loss in high flows.

Plantings in less disturbed or improved soils have higher rates of survival and growth.

- It is hard (maybe impossible) to successfully grow plants in compacted soil. Haul roads used for heavy equipment during restoration probably won't grow plants.

- In general, planting in tailings is a beast: it's hard to get plants established, even with irrigation and several rounds of plantings.
- Planting behind roughness (and add roughness – e.g. woody debris) encourages more soil deposition, which benefits plants, and can also protect plants during high flow periods.
- Consider soil amendment, e.g. fine soils or biochar.

Site prep: remove competing plants, add roughness, and protect plants from browse.

- Mechanically treat weeds/competing plants before planting.
    - Treat prior to planting because treating post planting is really difficult.
    - If waiting a season or two before planting, plant early seral species to improve soils and protect against invasives.
    - If using herbicides, consult with botanists.
- Plant behind roughness to encourage fine sediment accumulation and protect plants during high flows.
- Protect plants from ungulate, cattle and rodent (beaver, muskrat) browse.
    - Browse is a major contributor to poor plant survival and growth.
    - Impact from ungulates and rodent browse is cumulative.
    - Use exclusion fencing to protect from elk, deer, and cattle.
        - Rebar and elk wire are cheap and fast, but should not be installed in areas with high velocity flows.
        - If fencing's not an option, consider planting more densely.
        - Consider planting species that help create exclosures (e.g, conifers, hawthorne).
    - When recently planted plantings are the only food item available rodents have the capacity to significantly negatively impact plantings. May need cages for rodents.
    - Consider planting in floodplain and adjacent uplands to reduce browse near streams.

When choosing species to plant, consider site conditions and project objectives.

- Examining what is already there will inform what can grow there.
    - Expectations on what should grow should be tempered by what can actually grow in these highly altered systems.
        - If conifers grow, plant conifers. If alders grow, plant alders. If it's a meadow, plant meadow species.
        - Willows and cottonwoods need to have their roots to the water table and need point bars which is challenging when you are working a sediment limited system.
        - Some shrubs will drown in planted in overly wet conditions (e.g., wet meadow suited to sedges).
    - Consider that seral plant succession might be needed, particularly in degraded sites and where invasives are a problem. This means plantings might need to start by seeding grasses and forbs, preferably with locally sourced seeds (or carex mats, where appropriate).
    - Use opportunist species that can colonize quickly
    - Consider browse-tolerant, flood-tolerant species
- If stream shade is a priority, consider fast growers over more traditional riparian species

    - Willow and cottonwood are slow growers if they are not in ideal conditions and need adequate water – will not get shade in 30 years with these unless you already have a functioning system.
    - Alder grow more quickly and have higher survival rates.
    - Consider planting conifers to get more shade faster.
        - Conifers can also help provide exclosures from ungulate browse.
        - Pines will eventually fall and add roughness – can be considered an interim system improvement.
    - Hawthorne sometimes grow where nothing else will and create great exclosures.
- Choose plants suited to available amount of water
    - E.g., willows need to reach the water table, but can also drown if too wet.
- Larger, older (a few years old) seedlings do better than smaller, younger (1-2 years) ones – have been found to better outcompete undesirable/invasive plants.
- Bryophytes improve native fish habitat by providing food (macroinvertebrates) and refugia.
    - Bryophytes need to be protected from browse and trampling.

Riparian plantings take years to reach full height and provide stream shade

- Takes 10-20 years to get mature alders, cottonwood, full riparian community.
- Conifers may be desirable for faster growth and more shade in the first 30 years.
- Funders may not understand these timelines, and want demonstrated outcomes.

Public perception is important

- Need public, and particularly landowners', support for future projects.
- A planting project that leaves behind a lot of debris or fencing that goes down and doesn't get repaired looks really bad, and the public notices.

OUTSTANDING QUESTIONS/UNCERTAINTIES

Prioritizing planting projects/sites

- Focus on less disturbed sites with higher likelihood of fast plant growth?
- Focus on areas where you can do the whole suite of restoration (including floodplain reconnection etc. to restore ecosystem processes)?
- If the goal is to shade existing channels in the shortest timeframe, should work be focused on near-channel over broader floodplain?
- Would planting the uplands also reduce ungulate and rodent browse in the riparian zone?

What to plant

- Avoid planting willow and cottonwood unless you have a functioning hydrologic system?
    - Plant pine or alder until you can get the hydrology restored?
    - Other beneficial species (grow fast, resist browse…)?

When to plant

- Spring or fall?
- How long after site prep?

Irrigation (whether, where, how)

- See Stromberg's research out of the Southwest – directly addresses pros and cons of irrigation.

Herbicides for weed treatment (whether, what type, how much)

NEXT STEPS

1. What questions do we need to be asking during site assessments?
    a. It would be really nice to have a flow chart to walk through these questions.
    b. Question include:
        i. What site prep is required. Weed removal? Soil amendments? Soil additions?
        ii. What plants should be planted at a specific site – there is a need to learn a quick and repeatable protocol
        iii. How do you get perennial forbs to outcompete other species
        iv. How to know when to spray and when not to spray for weeds
        v. Consider the timing of the site visit (high vs low flow)

2. How to continue/follow up on this discussion?
    a. Power of group discussion – created a learning lab and opportunity to learn from each other. Want to continue/build on that? If so, how?

3. What questions could be shared with OSU researchers?
    a. Are revamping their research at Starkey to be more hydrology-focused, stage zero type work and are open to incorporating questions practitioners need answered.

4. How to communicate complexity to decision-makers and funders?
    a. The further you get away from lessons learned in the field the more the nuances get lost.
    b. How to align planning and funding cycles (including benchmarks/targets) with lessons learned?

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