Middle Fork John Day River Intensively Monitored Watershed

Final Summary Report

Appendices B – M

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Prepared by the Middle Fork IMW Working Group

Forrest Conservation Area. *Courtesy of ODFW.*

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Appendix B – Steelhead and Chinook Salmon Monitoring and Evaluation

Kirk Handley, Oregon Dept. of Fish and Wildlife, John Day OR
Jim Ruzycki, Oregon Dept. of Fish and Wildlife, La Grande OR

Abstract

We monitored the response of steelhead *Oncorhynchus mykiss* and Chinook salmon *O. tschawytsha* to restoration actions in the Middle Fork John Day River at several spatial and temporal scales. Monitoring included measures of abundance, survival, distribution, and productivity. Results at the watershed scale indicated limited response by the steelhead and Chinook populations. Freshwater productivity, measured as smolts/spawner, has not increased since inception of the IMW.

Monitoring of juvenile steelhead and Chinook indicates abundance varied both seasonally and annually among sites and streams. Survival models for juveniles rearing within the IMW indicate survival was influenced by streamflow, juvenile parr density, and varied by season.

Although factors limiting freshwater production by salmonids were likely improved through restoration actions in the MFIMW, the primary limiting factor of temperature was not (to date) significantly altered. We conclude that elevated stream temperatures continue to limit the freshwater production of salmonids by limiting their summer distribution and causing poor early life-stage survival.

Introduction

Background

Chinook Salmon and steelhead monitoring in the John Day River has been ongoing for >50 yrs. Index surveys of adult steelhead and Chinook spawning activity and redds throughout the John Day River Basin was initiated during the 1960’s. More recently, in 2004, steelhead redd monitoring in the John Day River Basin employed a spatially balanced approach (Generalized Random Tessellation Survey design; GRTS) while smolt monitoring, initiated in 2002, employed rotary screw traps (RST). The data derived from these long-term trends have been crucial for understanding fish responses within the MFIMW. These efforts to monitor salmonids in the MFIMW, which relied heavily on status and trend monitoring, was conducted outside the of the funded MFIMW partnership and primarily funded by Bonneville Power Administration (BPA). For the fish monitoring component funded through the MFIMW, we chose to focus on juvenile survival, growth, habitat distribution, and productivity.

Goals and objectives

The goal of the fish monitoring in the MFIMW was to evaluate the effectiveness of implemented restoration actions for recovering the depressed anadromous steelhead and Chinook Salmon populations residing there. We also sought to understand how habitat conditions influenced fish performance metrics at both watershed and subwatershed scales. Our main objectives included:

1) Estimating spawner escapement (abundance) of steelhead and Chinook populations to the MFJDR.
2) Estimating freshwater productivity (smolts/redd) of Chinook and steelhead populations.
3) Estimating parr-to-smolt survival for steelhead and Chinook.
4) Comparing the above MFIMW fish metrics to reference populations within the John Day River basin.
5) Delineating juvenile Chinook (parr) seasonal rearing habitat.
6) Investigating juvenile fish passage on Bridge Creek and Bates Pond.

Site Selection

We selected sample sites in response to the goal of the MFIMW to enhance depressed fish populations and to understand how these actions influenced fish performance metrics at both watershed and sub-watershed scales. Within the MFIMW, several types of restoration actions were implemented across a wide range of temporal and spatial scales. Given this complexity, multiple comparisons implemented within a hierarchical design framework were necessary to fully evaluate our objectives.

The hierarchal framework is composed of a watershed scale evaluation at the anadromous fish population scale. It also includes a nested comparison within the larger framework that targeted specific restoration actions in the Camp Creek sub-watershed; referred to as the Camp and Granite Boulder Creek comparison. Both the watershed scale and nested comparisons were designed as Before-After-Control-Impact (BACI) experiments. The watershed scale design compared the MFIMW to either the South Fork John Day River (SFJDR) watershed (steelhead) or a combination of the Upper Mainstem John Day River (UMJDR) and North Fork John Day River (NFJDR) watersheds (Chinook). These watersheds were chosen because of their proximity to the MFIMW and because they had both historic and ongoing fish monitoring. The initial nested design evaluated restoration efforts in the Camp Creek sub-watershed against the control watershed, Granite Boulder Creek. Both of these tributaries are located within the MFIMW. The thermally stressful environment of Camp Creek was targeted for extensive restoration actions while the colder Granite Boulder Creek was to remain mostly unaltered except for some fish passage restoration actions. After Granite Boulder Creek was determined to be a poor reference watershed, we switched to using Murderer's Creek, a tributary to the SFJDR, as our control.

Methods

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

Trends in adult salmonid escapement in the MFIMW is correlated with basin and region wide adult escapement. Region wide trends in adult escapement are often driven by ocean and climactic conditions. For this reason, adult steelhead escapement alone is not a reliable metric to determine the success or effectiveness of restoration actions in freshwater but an estimate of adult escapement is necessary to assess freshwater productivity.

Spawner escapement for steelhead and Chinook in both the MFIMW and the three reference watersheds, was measured using redd surveys of spawning grounds. Census surveys were conducted annually to monitor adult Chinook spawning escapement over the entire spawning habitat in the MFIMW, UMJDR, and NFJDR during August and September of each year. Surveys were conducted by walking upstream through identified sampling reaches and counting observed redds, live fish, and sampling of carcasses. See McCormick et al. (2010) for a complete description of Chinook redd survey methods.

Steelhead redd surveys, based on standard ODFW methods (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 (RKM 24) were excluded since this area was outside of the MFIMW area. 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 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:

$$R_D = \sum_{i=1}^{n} r_i/d_i \quad (1)$$

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:

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

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

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

where C is an annual fish per redd constant (e.g. 1.6 fish/redd for 2010) 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).

Objective 2. Estimate freshwater productivity (smolts/redd) of Chinook and steelhead populations.

For the measurement of recovery of listed fish species, NMFS is primarily interested in estimates of fish production or survival which relate directly to their recovery. Estimating smolts/adult is the most direct approach we currently have to estimate freshwater production. Adults were measured using the above stated methods. Productivity of the steelhead and Chinook populations was measured as smolts produced per adult spawner. At the watershed scale, we measured responses of the MFJDR populations and compared these to neighboring reference populations. A BACI-like design was employed to provide spatial and temporal contrast and account for out-of-basin effects. For comparisons, we chose control watersheds inhabited by nearby fish populations where significant background information was already being collected. While some restoration was occurring in the control watersheds, we assumed that the amount of restoration implemented for the MFJDR would be significantly more extensive. The SFJDR watershed, used for the steelhead control has had similar steelhead population metrics (Figure 6). The SFJDR watershed is also dominated by public lands and few large scale restoration actions. The NFJDR watershed was used as a Chinook control because it also is dominated by public lands and wilderness. We later decided to include the UMJDR watershed to increase our temporal comparison for Chinook productivity (Figure 7) because the historic dataset there was longer than that from the NFJDR.

Out-migrating juvenile salmonids (primarily smolts) were monitored and enumerated using rotary screw traps (RST). Juvenile spring Chinook and steelhead migrants from the MFIMW were captured using a 1.52 m RST operated on the MFJDR near Ritter. Complementary RSTs were operated on the SFJDR, NFJDR, and UMJDR reference watersheds (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 and steelhead migrants were anesthetized with tricane methane sulfonate (MS-222), 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, FL; mm). A subsample of fish was released above the trap to estimate migrant abundance using mark-recapture techniques. We used linear extrapolation to account for un-sampled nights. Further details of our RST operation are available in Wilson et al. (2010).

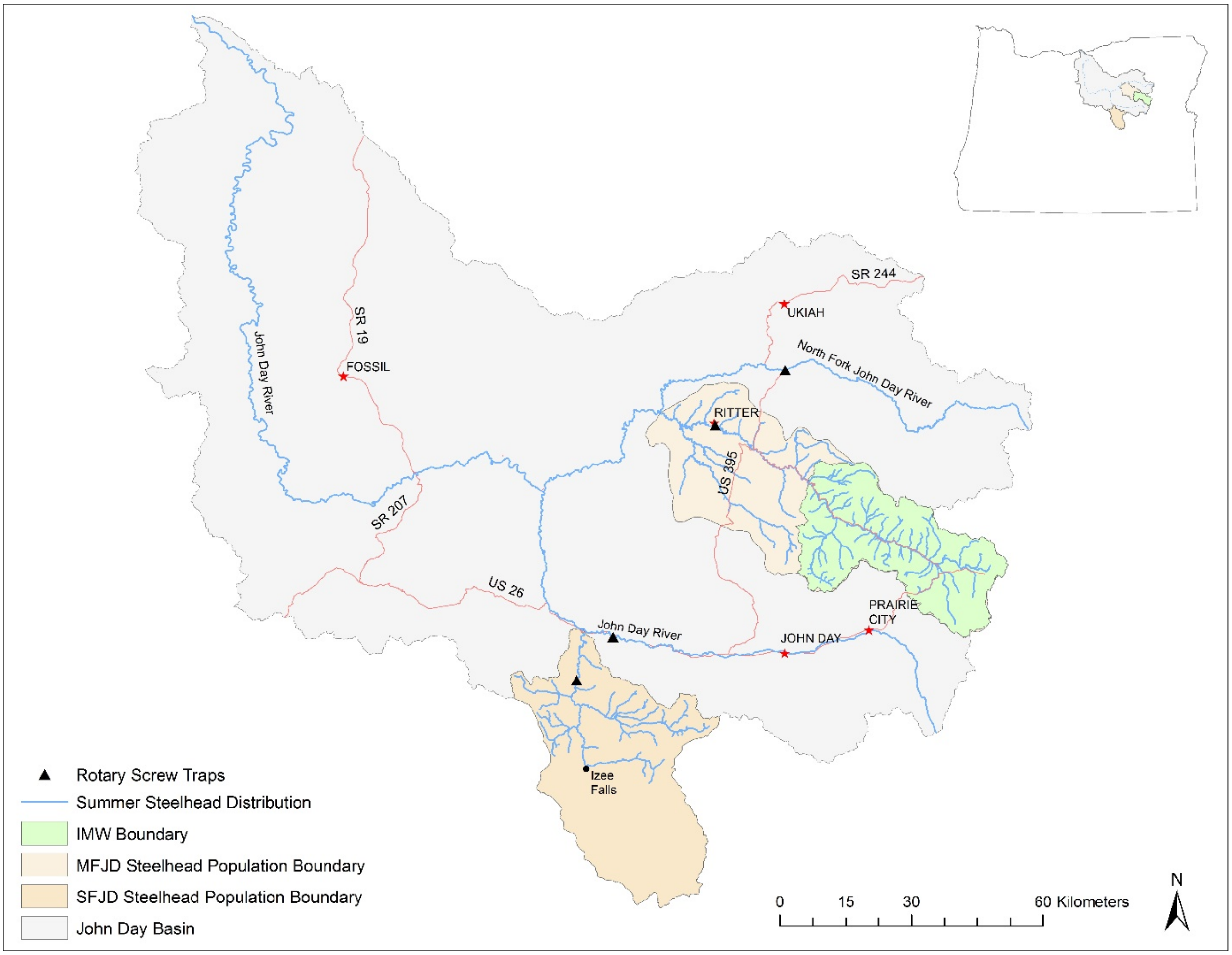

Figure 1. Map showing the John Day River basin, the Middle Fork IMW, the entire Middle Fork John Day watershed, and the control watershed for steelhead, South Fork John Day River, used for the experimental BACI design.

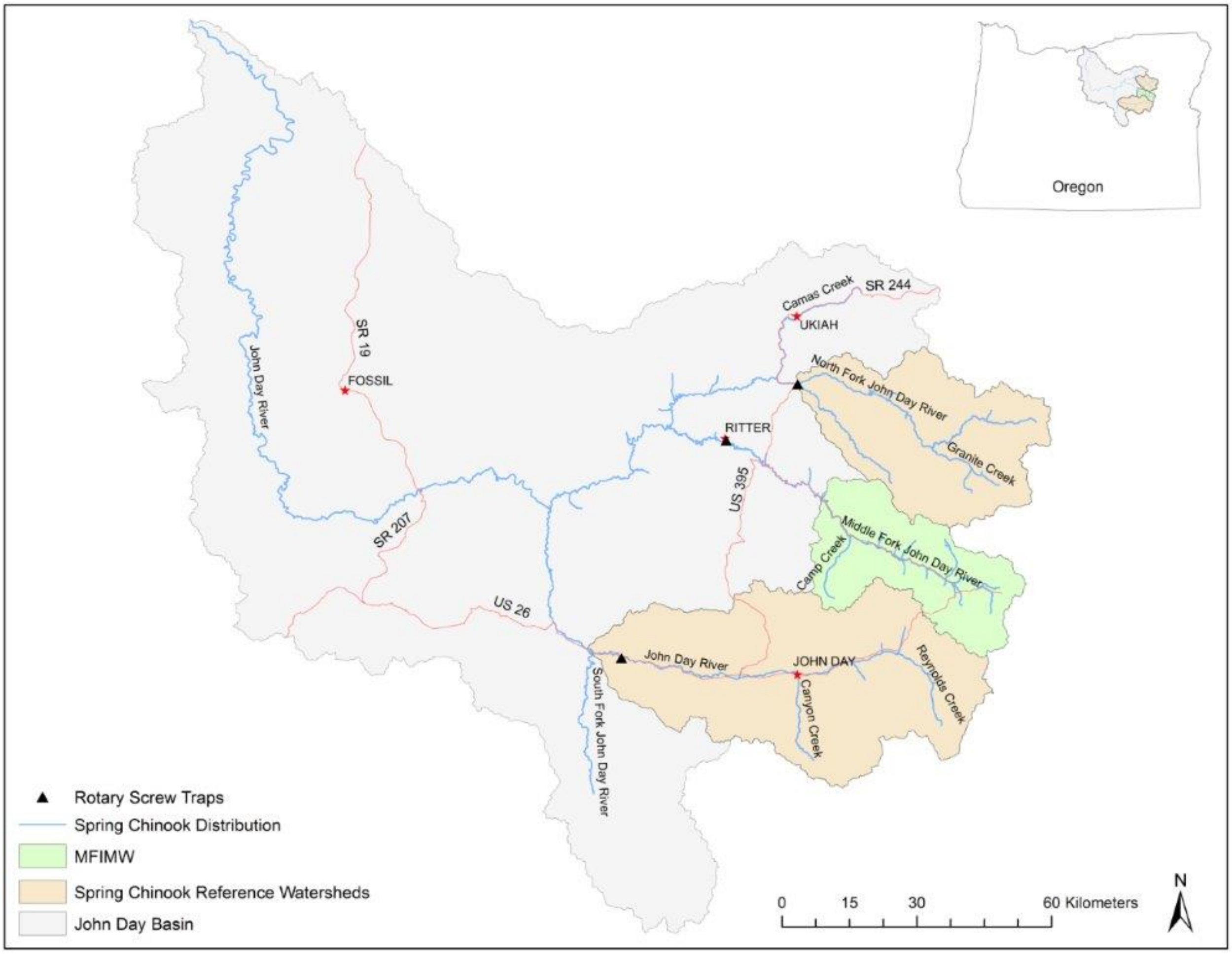

Figure 2. Map showing the John Day River basin, the Middle Fork IMW, the entire Middle Fork John Day watershed, and the control watersheds for Spring Chinook Salmon, North Fork John Day River and the Upper Mainstem John Day River, used for the experimental BACI design**.**

Objective 3. Estimate parr-to-smolt survival for steelhead and Chinook.

Steelhead

Parr-to-smolt survival was estimated by PIT tagging parr in various habitat areas and detecting tagged fish in our RST and PIT antenna array as they migrate out of the basin as smolts. Survival to emigration was compared among tagging reaches and tributaries to determine differential survival rates. Tributaries were selected to provide a wide range of rearing temperatures. Granite Boulder Creek and Camp Creek were selected for survival monitoring because of their widely different temperature regimes during the summer rearing season. Camp Creek, with several active restoration projects, is warmer than Granite Boulder Creek during summer months. Each stream was divided into reaches based on the current summer steelhead distribution and topographical features from 1:24,000 quad topographic maps. Although both summer steelhead and spring Chinook were targeted in this sampling, steelhead distribution was utilized for both species because their distribution encompasses the entire known distribution of Chinook. Within each reach, three sites were selected for monitoring (Figure 4). Sites were determined by utilizing the GIS layer developed for steelhead spawning surveys in the MFIMW. Specifically, the first point encountered in each reach proceeding in an upstream direction was selected as a sampling site. Depending on whether that point was in the first, middle, or latter third of the reach, all other site locations in the reach were located a distance equal to 1/3 of the reach distance from the other sampling points within that reach, resulting in one sampling site occurring in each third of the reach.

Site lengths were 20 times the average active channel width (ACW) measured at five locations near the site point. The site point was considered the mid-point of the sampling section, however in some instances the section was moved upstream or downstream to avoid constraints from secondary channels or tributaries. Block nets were employed at the upstream and downstream extents of each sample section to eliminate fish movement during sampling. Sites were sampled once a day for three consecutive days. Block nets remained in place until sampling was completed on the third day. Juvenile fish were collected using a backpack electrofisher (Smith-Root LR20B). In larger stream sections, e.g. the MFJDR, fish were collected by snerding, where a snorkeler would enter at the head of a pool and attempt to herd fish downstream into a seine anchored at the pool tailout.

Sampled fish were placed into an aerated 5 gal. bucket and transferred to in-stream live boxes where they were held until the entire site was sampled and tagging operations commenced. Captured juvenile spring Chinook, steelhead, and Bull trout were anesthetized with MS-222, interrogated for PIT tags, weighed to the nearest 0.1 g, and fork length (FL) measured to the nearest millimeter (mm). Scales were taken from a subsample of steelhead collected that were larger than 60 mm. Subsamples were grouped into 10 mm bins and 15 samples were collected in each bin during summer sampling and 15 samples collected during fall sampling. All bull trout were sampled for scales. All anesthetized fish were allowed to recover in an aerated bucket until they regained equilibrium (~5-10 min). Once recovered, fish were released in small groups throughout the site and allowed to distribute themselves naturally within the sampling reach.

Ages of steelhead and bull trout were determined by counting scale annuli (Alvord 1954). A length at age key was then used to assign ages to unaged fish based on 10 mm length bins (Guy 2007).

Encounter histories were developed for each tagged steelhead to estimate population abundance. A closed capture model (Otis et al. 1978) was used to analyze the encounter histories by site in Program MARK (White and Burnham 1999). This analysis utilizes a log maximum likelihood probability to estimate both capture (p) and recapture (c) probabilities as well as population abundances (N). Model variables for capture and recapture estimates can vary temporally, or can be constant, either together or separately. For each site, three potential models were fit to the data (Table 1). The most parsimonious model was selected based on the lowest Akaike Information Criteria (AICc) value. When AICc values of two or more potential models differed by less than two, the model with fewer parameters was selected.

We assessed PIT-tag detection histories of all fish tagged as part of the MFIMW project by querying tagging and interrogation files for observation of these fish. Fish tagged in the MFIMW have the potential to be interrogated at remote in-stream antenna arrays located in the MFJDR near Mosquito Creek, in the John Day River near McDonald's Ford, and at John Day Dam, Bonneville Dam, and the Columbia River estuary. Other observations are also possible during collection events within streams where surveys are being conducted as well as at the RST on the MFJDR near Ritter (MFRST). Detection histories were grouped by species (spring Chinook or summer steelhead), tag site (Camp Creek, Granite Boulder Creek, or the MFJDR), and by tag year. Subsequent interrogations were grouped by observation site and year of observation where observation year began on 1 July and ended on 30 June the following year to incorporate in-stream tagging events and align with migratory years that overlap from fall to spring. Operation of the in-stream antenna array in the MFJDR also allowed us to interrogate returning adult fish that cross the antenna to spawn upstream. This information allowed us to assess the origin of these fish as they migrate past our array by querying tag files within PTAGIS (www.PTAGIS.org).

Table 1. Models fit to encounter history data, description of the models, and the number of parameters in the associated model. All models also parameterized population abundance, which is not included in this table.

Model	Model Description	# of Parameters
p(.),c(.)	Capture and recapture are constant but not equal	2
p(.)=c(.)	Capture and recapture are constant and equal	1
p(t)=c(t)	Capture and recapture vary temporally but equal during individual sampling events	3

Within the MFIMW we also compared the juvenile salmonid density, growth and survival in contrasting tributaries from 2008 to 2016. Camp Creek, a relatively warm tributary where extensive restoration actions were occurring, was compared to the colder Granite Boulder Creek where few restoration actions were planned. Murderers Creek, a tributary to the SFJDR, was used as a control or reference since it was similar to Camp Creek. Both juvenile Chinook salmon use and juvenile steelhead production were expected to increase in Camp Creek after restoration actions were implemented. Unlike the larger scale watershed scale evaluation, the Camp and Granite Boulder Creek comparison targeted the response of specific restoration actions within Camp Creek, providing a platform to investigate causal mechanisms between these specific restoration actions and fisheries outcomes.

Steelhead parr abundance at closed capture sites was translated to parr density (#/100 m of stream) for each creek so that streams with differing sampled areas could be compared. Murderers Creek abundance and density estimates were obtained from the ISEMP group at three sites in lower Murderers Creek (Figure 4). Sites in Murderers Creek and Granite Boulder Creek were classified as control sites and treatment sites in Camp Creek were used to evaluate restoration actions in that stream, most of which occurred in 2011.

We tested the relationship of stream flow, spawner density, and parr density to the age-1 parr density/spawner ratio (hereafter referred to as age-1 productivity) of the following year to assess the relative effects of each of these three variables on age-1 densities during spring sampling the following year. We assumed summer base flow or lowest recorded stream flow at the USGS gauging station near Ritter, OR was an acceptable measure of low flow conditions for these streams. Brood year spawner density estimates (#/km) were calculated based on our annual fish per redd estimates and redds observed within random steelhead spawning ground surveys in each stream. The Murderers Creek spawner densities were calculated using spawning ground surveys downstream of a partial upstream passage barrier near RKM 18 which likely limits upstream spawner distribution during lower stream flows (Ian Tattam, ODFW, personal communication). Parr density of older conspecifics in these streams the previous fall was used to test the hypothesis that older parr cannibalize or outcompete age-0 steelhead.

Chinook

Chinook parr survival was monitored at eight sites in the mainstem MFJDR, one site in Vinegar Creek, Camp Creek, and one site in Coyote Creek (Figure 3). Camp Creek Chinook parr sampling was concurrent with steelhead monitoring twice annually at closed sample sites (Figure 3). Each of the other sites was sampled four times annually: the first three sampling occasions were done at three week intervals starting the second week of July with a final six week interval and the final sampling occasion in October. Each captured parr was interrogated for PIT tags upon capture and marked with a 12mm PIT tag if not previously captured. Open population models such as the Cormack-Jolly Seber model (Cormack, 1964; Jolly, 1965; Seber, 1965) are capable of estimating apparent survival but if emigration from the study area is likely, true survival is confounded with permanent emigration. We assume that parr move throughout the summer and more frequently as conditions change late in summer and into the fall season. To model survival at the watershed scale we sought a model that could account for emigration from our relatively short sampling sites (150 m). In order to take advantage of PIT tag detection infrastructure throughout the migration corridor, interrogation information was included after instream sampling was finished. One potential problem with including interrogation information is that not all individuals are migrating at the same rate or time past interrogation sites. This was the case at the Middle Fork PIT Array (MFA) and the RST near Ritter where in some years, as early as October, fish initiated downstream movements prior to the traditional spring smolt migration period. For this reason a multi-state model (Arnason, 1972, 1973) was chosen to estimate true survival of individuals marked at these sites from a “parr” to “migrant” state. This model was run through Program Mark (White et al., 1999) using code from package Rmark (Laake, 2013) in R (R Development Core Team, 2008). Using the multistate model we were able to include all records of capture and integration for each marked parr.

Interrogation or detection sites included the MFA, MFRST (Figure 5), John Day Dam, Bonneville Dam, and the Columbia River estuary trawl sampling. When initially tagged, all individuals were identified as “parr”, when individuals were detected at one of the downstream interrogation sites they assumed the “migrant” state. The multistate model is only capable of handling encounters at discrete encounter occasions but in the case of detections at interrogation sites encounters occur over a three month period. These detections were grouped into one encounter occasion prior to the start of the three month interval because survival to at least the start of the interval could be assumed if individuals were detected afterward. This biased migrant survival low for the next interval but did not influence survival estimates for parr which was the primary interest. Detection probability at the MFRST and MFA was modeled with stream discharge between each interval because detection efficiency at the MFA and MFRST declines as flows increase. The final two encounter occasions (7 and 8) represented detections at John Day Dam and Bonneville Dam, respectively.

We tested models for probability of capture (p) at both the parr and migrant state which included covariates for flow and capture technique (Table 2). Transition probability from migrant to parr was fixed at 0. Transition probability from parr to migrant was modeled with time (interval) and proportion of the total migrants from each migration year captured at the RST over each interval (Table 2). Chinook parr survival was modeled with covariates for streamflow, temperature, reach condition, treatment reach, and brood year redds to account for environmental factors and density dependence (Table 2).

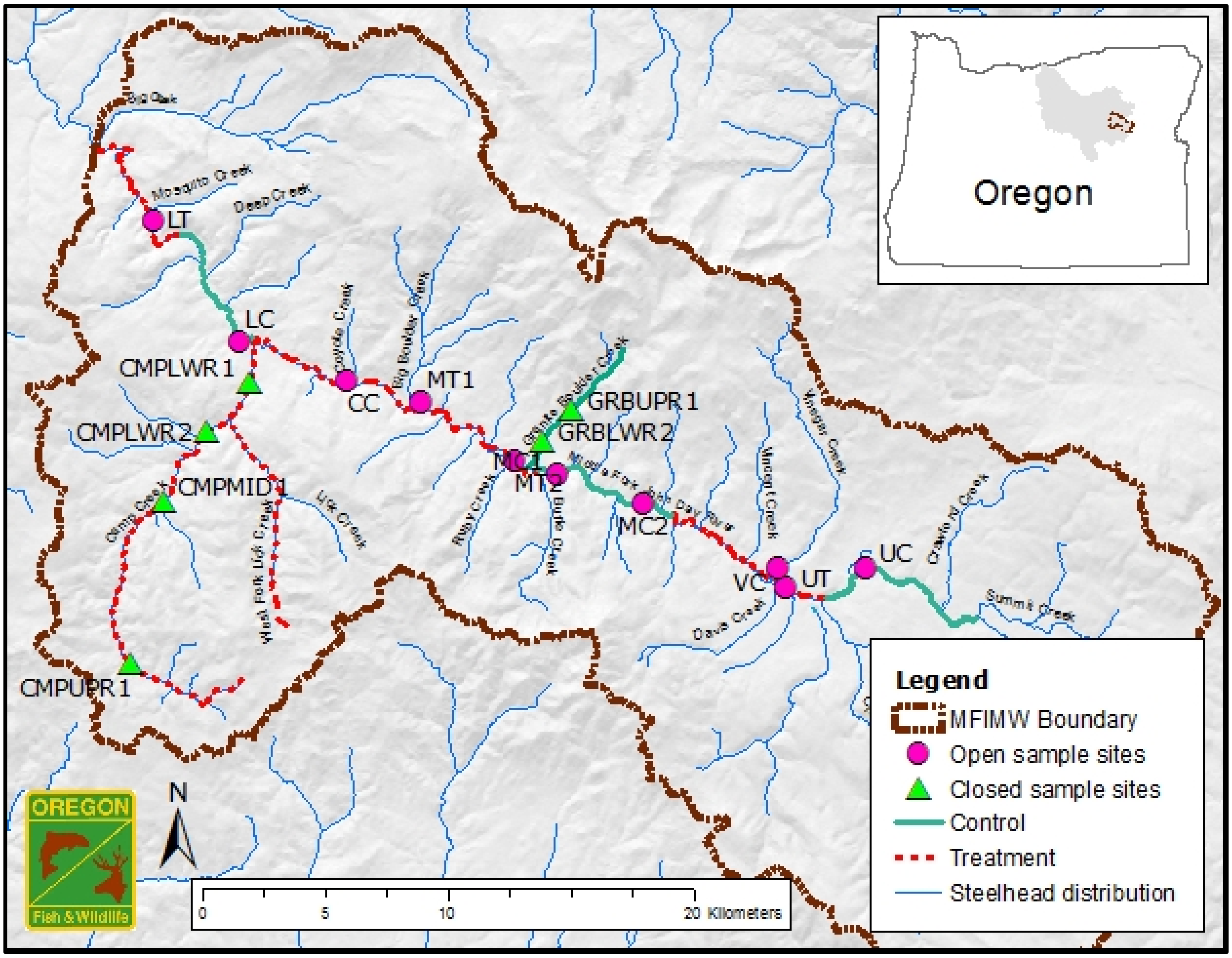

Figure 3. Parr mark-recapture sites throughout the MFIMW. Open sample sites were sampled years 2011 through 2015, closed sample sites were sampled 2008–2015. Open sites are abbreviated as follows: Lower Treatment (LT), Lower Control (LC), Coyote Creek (CC), Mid Treatment 1 (MT1), Mid Treatment 2 (MT2), Mid Control 1 (MC1), Mid Control 2 (MC2), Vinegar Creek (VC), Upper Treatment (UT), and Upper Control (UC).

Hypothesis testing

Different hypotheses were tested by fitting models with covariates and varying model structure. We tested models to determine if survival was related to annual streamflow trends, temperature exceeding stressful levels, average temperatures over intervals, and intra-specific density (brood year redds) against site and time specific models. With the assumption that summer temperature is a limiting factor, we hypothesized survival would be inversely related to 7DAM temperature, 7DA temperature, 7DAmin temperature, and average temperature over each interval during the summer season (Table 2). We also assumed streamflow limited habitat over the summer season and hypothesized a positive relationship between average flow over each interval and summer survival. Flow and temperature were not necessarily influenced by restoration actions over the short term monitoring that we conducted but these relationships were important to understand when trying to determine any observed changes to survival related to stream restoration in or near our sample sites.

In an attempt to identify effects of restoration actions, we tested models including time and reach covariates (Table 2). We also included covariates for reach condition of each sampling site based on a geomorphological classification system (O'Brien et al. 2014). As an example, to test the hypothesis that treatments influenced survival in reaches that were thermally suitable for Chinook parr we tested the model: season x treatment x 7DA temp, and compared it to: season x 7DA temp. The model with the highest rank (rank is inversely related to AICc score) best fits the data and is the most likely description of the observed data.

Summer steelhead parr abundance and survival over the summer growing season were monitored at Camp (treatment) and Granite Boulder (control) Creeks. Sites were sampled once early in the summer and again in October using three pass closed capture techniques (Otis et al. 1978) to determine abundance and to mark and recapture individuals using PIT tags. Abundance was estimated separately for each seasonal sampling occasion using a closed capture model in Program Mark (White et al. 1999) which calculates abundance based on capture probabilities from observed recapture rates.

Summer steelhead parr survival was estimated using a Barker model which includes information from both instream recaptures of live individuals, recoveries of dead individuals, and downstream detections of smolts at PIT tag interrogation sites during migration (Barker 1997). Interrogation sites included the MFA, MFRST, John Day Dam, Bonneville Dam and the estuary trawl. We also included instream detections and PIT tag recoveries from scans with a mobile PIT tag antennae extending from 250 m above to 250 m below the sample sites. These reaches were scanned with the mobile antennae during mid-summer in 2013, 2014, and 2015.

In the final analysis of survival data, we included mark recapture data from a similar study conducted in Murderers Creek (ISEMP-EcoLogical Research Inc.). In this study, sampling was conducted at three sites, three times annually: once in mid-June, again in late September, and once in late winter (Figure 4). To eliminate bias in survival estimates and make the two datasets comparable we excluded individuals captured during the winter sampling event in Murderers Creek. Fall sampling in Murderers Creek was typically conducted two weeks earlier than sampling in the MFIMW. This may have biased survival estimates slightly higher for summer intervals and slightly lower for winter intervals in Murderers Creek when compared to Camp and Granite Boulder Creek.

Table 2. Covariates tested in Multi-State Chinook Parr survival models and corresponding parameters. Type refers to how covariates were used, group covariates model each group separately, number covariates are assigned as individual covariates to each individual and a line is fit to describe parameter estimates.

Covariate	Use	Description	Type	Sample Size	Min	Ave	Max
Season	S,p,Psi	summer=July–October, winter= November–May	group	165,220	na	na	na
condition	S	Good, Moderate, Poor (see O'Brien et al. 2014)	group	175,175, 35	na	na	na
Ave	S	Avg. temperature during interval	#	117	5	16	22
aveflow	S,p	Avg. flow of MFJDR near Ritter during interval	#	30	18	138	557
redds	S	brood year redds for each cohort	#	5	113	365	516
EarlyM	Psi,S	Proportion of total migrants passing MFRST	#	5	0	0.14	0.99
capture	p	sample method: shock, snerd, MFA +MFRST, Col. Dams	group	69,96,11 0,110	na	na	na

Steelhead Parr Survival

Models were tested in a similar fashion to the Chinook parr survival models but sites and streams were monitored prior to the occurrence of large scale restoration projects. This enabled us to test models for pre and post treatment against models that did not include treatment effects. We were able to compare MFIMW estimates to a similar study in Murderers Creek in the South Fork John Day Basin which was assumed to have similar climatic conditions.

The best fit model for probability of capture (p) included stream and capture (Table 3). The best fit for probability of detection outside of the sample reach (R) included streamflow, season, mobile antennae scan, and watershed. Site fidelity (F) was modeled with age of individuals, season, and stream because of the relatively consistent age structure of smolts captured at the rotary screw trap (Bare et al. 2016). To avoid confounding survival with emigration, fidelity for individuals captured in Murderers Creek and Camp Creek were grouped by life history type. Granite Boulder Creek was modeled as a separate group because of observed differences in anadromy compared to the other two streams. Permanent emigration was assumed for fish detected outside of the site (F'=0). This simplified model structure and made sense for anadromous fish that migrate permanently out of the study site when they smolt. It was only possible to detect an assumed dead individual during a mobile scan so r was fixed to 0 for intervals where no mobile antennae PIT tag scans occurred (Barker 1997). We also removed individuals from the study that were detected outside of the study area by adding a 1 to the encounter history for the sampling interval prior to detection and coding a negative sign before the freq. column in the encounter history. This was done to avoid a "trailing 0 effect" in the encounter history caused by a near 0 probability of detection after smolting (Conner et al. 2015).

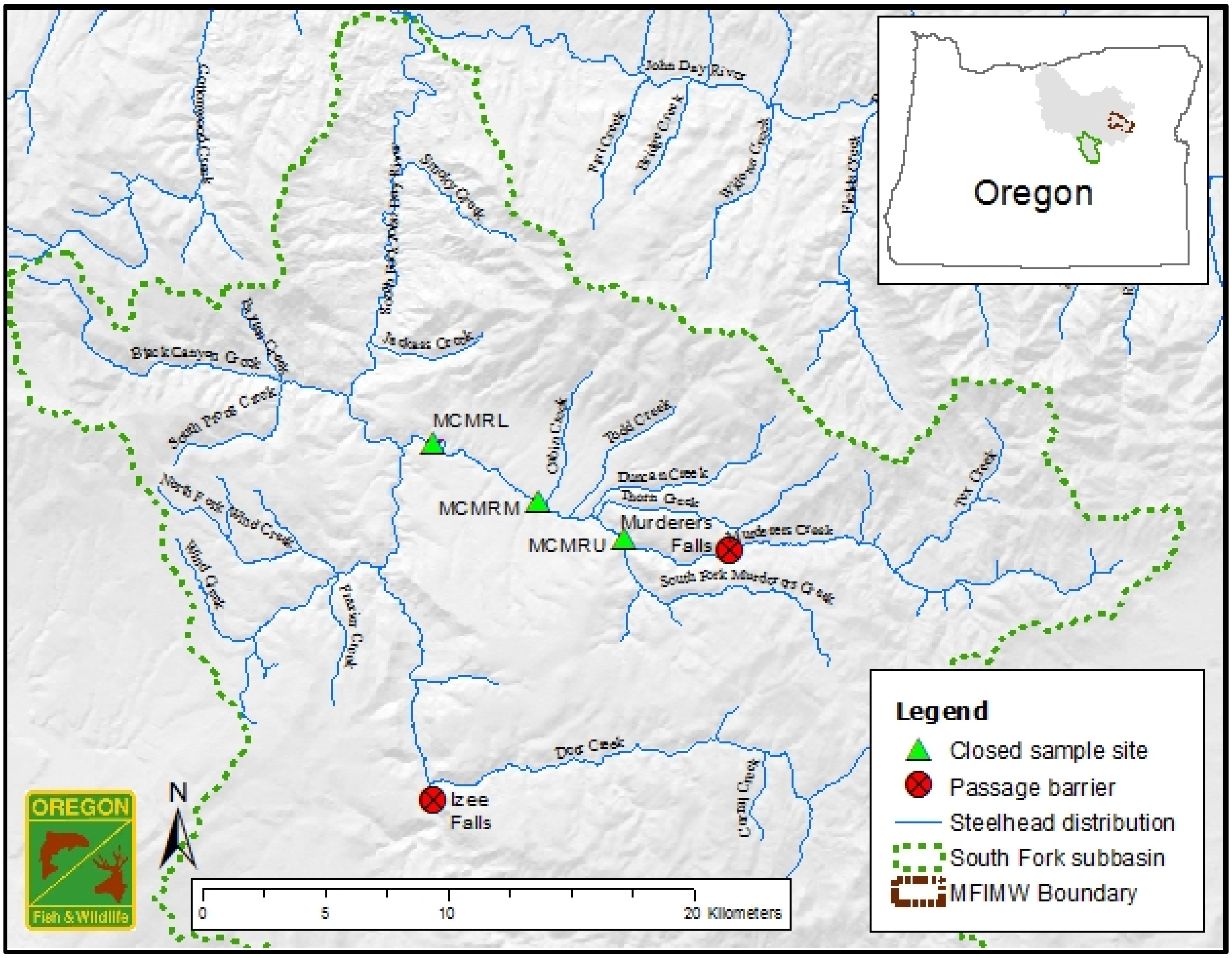

Figure 4. ISEMP steelhead monitoring sites in the South Fork John Day River used to compare survival and growth estimates from the MFJD IMW. Information collected at these sites from 2008–2015 was compared to steelhead parr survival, growth, and abundance at sites monitored in the MFIMW.

We tested survival models with covariates for average flow, age, treatment, temperature, reach condition and density to determine effects of these covariates on survival in summer and winter seasons. Temperature covariates were missing for many of the intervals which made it impossible to test the relative fit of these models when ranked using AICc scores because no sites had continuous temperature monitoring for the duration of this study. To correct for missing covariates, data were grouped by a dummy variable for covariate present (t=1) or covariate absent (t=0) Linear modeling could then be performed using model predictions from the covariate present group (t=1). The relative fit of different temperature covariates was tested for intervals and locations where temperature was recorded continuously over the entire interval. The same procedure was used for density estimates where d=1 and d=0 for when density estimates were and weren't available, respectively. Discharge, measured at the MFJDR near Ritter was used for Camp and Granite Boulder Creeks. Stream discharge measurements taken from the SFJDR USGS stream gage near Dayville, OR were used to calculate flow covariates for Murderers Creek sites. To simplify discharge conditions for each interval, a binned discharge covariate was created (flowbin; Table 3). Mean flow data for the MFJDR was binned for the Middle Fork tributaries as <1.42 CMS (cubic meters per second), 1.42-2.83 CMS, >2.83 CMS and classified as drought, low, and high, respectively. For Murderers Creek, mean discharge from the South Fork gauge was binned as <.85 CMS, .85-1.42 CMS, >1.42 CMS and classified as drought, low, and high, respectively.

Parr Growth

Growth can be assessed through known length at age or through mark recapture of individual fish. Both methods produce reliable metrics of growth assuming fish can be accurately aged. However, mark recapture of individual fish provides knowledge of the location specific growth of fish. Using the mark recapture method, the location of the fish at the initial capture (start of growth) and recapture (end of growth period) is known. For the length at age method where only the initial location of the fish is known. In this analysis we used mark recapture to compare temporal and reach specific individual growth of Chinook and steelhead parr at our mark-recapture sites throughout the MFIMW. For a portion of our analysis we were interested in determining if specific stream treatments influenced growth rates. In this case using the individual mark recapture growth information allows us to assume if a fish was marked and recaptured at the same location it's growth was likely obtained in that reach of stream. We assumed that covariates specific to each reach of stream and time between initial mark and recapture occasions contributed to observed growth of individual fish.

We used a mixed effects growth model to assess the fit of models including covariates determined likely to influence growth (Tables 4, 5, & 6). In these models, we used the starting time of each growth interval as the random effect which was equal to time. Recaptures could occur at any of the subsequent sampling events. Growth and time between marking and recapture were calculated for each individual. This allowed us to assess differences in growth among control and treatment reaches and among reach condition rankings from the river styles framework (O'Brien et al. 2014). Some covariates tested were not available for all intervals between marking and recapture runs. We used a subset of the original dataset for intervals that included all covariates to test relative fit of each covariate. We then re ran models using all available data to obtain parameter estimates and model predictions.

Table 3. Covariates used in Barker model for steelhead parr captured at sites in Camp, Granite Boulder, and Murderers Creek from 2008 through 2015.

Covariate	Use	Description	Type	Sample Size	Min	Ave	Max
lifehist	F	assumed dominant life history of each stream: anadromous or resident	group	1296, 384	na	na	na
trueage	S,F	age of each individual during each interval: 0, 1, 2, 3 ,4 , 4+	group	continuous	na	na	na
season	S,F,R	summer=June through October, winter= October through June	group	840, 840	na	na	na
watershed	S,R	Middle Fork John Day River or South Fork John Day River	group	1088, 592	na	na	na
flow	S,R	average flow of MFJDR near Ritter OR or SFJDR near Dayville OR	number	16	15	181	655
flowbin	S	binned mean flow over interval: drought, low, high (see definition)	group	315, 247, 1118	na	na	na
>18	S	hours stream temperature exceeded 18^{0}C over interval	number	61	0	175	1363
>24	S	hours stream temperature exceeded 24^{0}C over interval	number	61	0	18	150
7Dmin	S	highest seven day average minimum temperature over interval	number	61	5.7	14	18.8
Average	S	average temperature over interval	number	61	2.5	12	19.2
7DA	S	highest seven day average temperature over interval	number	61	7.3	17	22.5
7DAMax	S	highest seven day average maximum temperature over interval	number	61	9	21	26.9
tempbin	S	binned average temperatures: poor, fair, good	group	286, 265, 1125	na	na	na
treatment	S	intervals treatment occurred stream specific: pre, post, control	group	308, 396, 976	na	na	na
treat	S	intervals treatment occurred not stream specific: pre, post	group	735, 945	na	na	na
sitetreat	S	intervals treatment occurred site specific: control, treatment	group	176, 1504	na	na	na
density	S	steelhead density (individuals >60mm FL/100 meters) at start of interval	number	129	27	141	610
scan	R,r	intervals scanned with mobile antennae: no scan, scan	group	1476, 204	na	na	na
capture	p	number of passes used to sample sites: 2 pass, 3 pass, not sampled	group	323, 1300, 57	na	na	na
condition	S	Good, Moderate (see O'Brien et al 2014 for definitions)	group	1328, 352	na	na	na

Table 4. Group covariates for Chinook and steelhead parr used in mixed effects growth model. Sample sizes for both species are shown in the left two columns. These data were collected from 2008 through 2015 at parr monitoring sites.

Covariate	Group	Chinook	Steelhead
Stream	Camp Creek	33	1852
	Coyote Creek	58	202
	Granite Boulder Creek	12	507
	Middle Fork John Day River	2591	517
	Murderers Creek	NA	605
	Vinegar Creek	177	462
Condition	Good	1683	2571
	Moderate	669	1546
	Poor	519	23
Treatment	Treatment	1077	1115
	Control	1794	2025
Temp Group	8°C ± 2°C	3	319
	12°C ± 2°C	205	761
	16°C ± 2°C	1176	1093
	20°C ± 2°C	547	309
Season	Summer (June 1 through Oct 31)	2870	3099
	Winter (Nov 1 through May 31)	NA	506

Table 5. Covariates used in mixed effects growth models for Chinook parr captured and recaptured throughout the Middle Fork John Day River from 2008 through 2015.

Covariate	Description	n	Min	Ave	Max
AverageT	average temperature over interval	1931	10.1	16.3	20.9
AveFlow	average flow of MFJDR	2871	19.8	59.3	259.3
FlowV	total of average daily discharge measurements of MFJDR	2871	456.0	2038.6	72089.0
FLowTrib	average discharge over interval for each stream	39	1.3	66.0	385.0
VolTrib	sum of average daily flow of each stream	39	145.0	1763.0	72089.0
BYCHS	brood year redds observed for each cohort	2871	113.0	378.6	516.0

Table 6. Covariates used in mixed effects growth models for steelhead parr captured and recaptured throughout the Middle Fork John Day River Intensively Monitored Watershed and Murderers Creek from 2008 through 2015.

Covariate	Description	n	Min	Ave	Max
AverageT	average temperature over interval	2482	1.5	14.2	20.8
AveFlow	average flow of MFJDR	4145	14.9	109.6	655.1
FlowV	total of average daily discharge measurements of MFJDR	4145	456.0	25356.8	384244.0
FLowTrib	average discharge over interval for each stream	158	1.3	14.3	663.0
VolTrib	sum of average daily flow of each stream$_2$	158	145.0	1650.0	138857.0
BYSTS	Adult spawner escapement	4145	432.0	2992.3	4859.0
BYSTSD	Steelhead spawner densities (adults/km)	4145	1.7	7.0	10.8
ParrD	steelhead parr/100 meters	2928	13.0	134.6	539.1

Objective 5. Delineate Chinook parr seasonal rearing habitat

Distribution surveys

Regional managers requested a comprehensive study of the summer distribution of Chinook parr in the watershed. We currently have information indicating that parr vacate warmer reaches of Mainstem habitats and enter cooler tributaries. This suggested that temperature is limiting smolt production in the basin. Hence, we sought to determine the upstream and downstream limits of Chinook parr summer rearing distribution in the mainstem MFJDR and tributaries throughout the MFIMW. We also sought to determine the temperature of reaches occupied by juvenile Chinook to aid in modelling future occupancy based on stream temperatures.

Summer rearing distribution of juvenile Chinook salmon within the MFIMW was assessed by snorkeling or electro-fishing pools in tributaries of the Middle Fork John Day River. Sampling proceeded upstream from the tributary mouth noting the presence or absence of juvenile Chinook, steelhead, or Bull trout based on reported juvenile Chinook data (Figure 2). Locations of all pools sampled were recorded with a handheld GPS along with focal fish presence/absence. Within tributary streams, we sampled every fifth pool beginning at the first pool upstream of the tributary confluence. In the event that no juvenile Chinook were observed in a sampled pool, we proceeded to sample every pool encountered, until a juvenile Chinook was encountered, at which point we returned to sampling every fifth pool. If no juvenile Chinook were encountered after sampling five consecutive pools, sampling ceased in that tributary. This information provided an estimate of the amount of habitat not currently used by rearing juveniles during the critical summer period. This in turn allowed for a measure of the potential for increasing freshwater production.

Tributary streams were sampled with one snorkeler moving in an upstream direction from the stream mouth or previous known distribution point. The snorkeler carefully moved in an upstream direction in small streams and reported the presence or absence of juvenile steelhead, Chinook, and any bull trout to a data recorder stationed on the streambank. If stream depth allowed the snorkeler to float without touching the streambed, the snorkeler would calmly float in a downstream direction rather than crawling upstream. GPS waypoints were recorded for each sampled pool in tributary streams and for each sampled reach in the MFJDR (James et al. 2008, Handley et al. 2011). In addition to basic presence-absence information, counts were also made for each target species in reaches sampled in the MFJDR in years 2014, 2015, and 2016 to compare densities among different reaches of stream (Handley et al 2015). Reach lengths were 150 m for survival monitoring sites and 100 m for snorkel distribution surveys.

We assumed that stream temperature loggers within one km of fish sampling locations without a major tributary between the logger and the fish sampling/observation location provided a reasonable representation of stream temperatures for each sampled reach. Daily maximum and daily average stream temperature readings were averaged for the week prior to fish sampling for loggers nearest each sampling location. We used a logistic regression model to describe the relationship between 7DAM (seven day average maximum stream temperature) and 7DA (seven day average stream temperature) to predict Chinook and steelhead presence based on observations throughout the MFJDR. In this model we included all of our fish sampling and distribution survey sites and reaches from the mainstem MFJDR with available temperature information corresponding to each sampling date.

Bates Pond

PIT Tag Antennae

Bridge Creek is a potential cool-water tributary to the MFJDR. However, Bates Pond, a relic sawmill pond, has an unnatural warming influence on Bridge Creek (see temperature monitoring report) that creates an impediment to juvenile fish passage and upstream migration. Because this site has significant potential for restoration, we monitored juvenile fish passage through Bates Pond using two PIT tag antennae arrays in Bridge Creek (Figure 6). The lower array was located approximately 30m downstream of the fish ladder and consisted of one 67 x 33 cm flat panel antenna. The Upper array was located approximately 10m upstream of Bates Pond (Figure 6) under the foot bridge and consists of one 67 x 33 cm flat panel antennae and one 80 x 30.5 cm rectangular pass through antennae. Antennae were securely anchored to the streambed to detect tagged fish movement across most of the wetted width of the channel at base flow. All PIT tag antennae were controlled by Biomark FS2001 transceivers which store the tag code, date, and time of each detected tagged fish. Transceivers were powered by 12v batteries and photovoltaic panels.

We placed antenna arrays both above and below Bates Pond. Initially during August 2010, one antenna each was installed in Bridge Creek immediately above and below the pond. Another antenna was added to the upper site in March of 2011 to improve coverage in Bridge Creek as it enters Bates Pond. Due to fewer daylight hours and shading during the winter months the upper antennae site could only be operated seasonally from March/April (depending on snowpack) through the following November. This period does encompass likely migration timing of smolts, upstream movement of adult salmon and steelhead, and any movement during the summer rearing season of salmon and steelhead parr. The array below the pond was not as shaded during the winter months and received enough sunlight to operate all year.

To evaluate juvenile passage through the fish ladder and pond we captured and PIT tagged steelhead and Chinook parr both downstream and upstream of Bates Pond. We also tagged parr at various other sites throughout the Middle Fork IMW which could be detected if they pass the arrays. Tagging of parr in Bridge Creek began in the summer of 2010 just prior to antenna installation and continued through 2014.

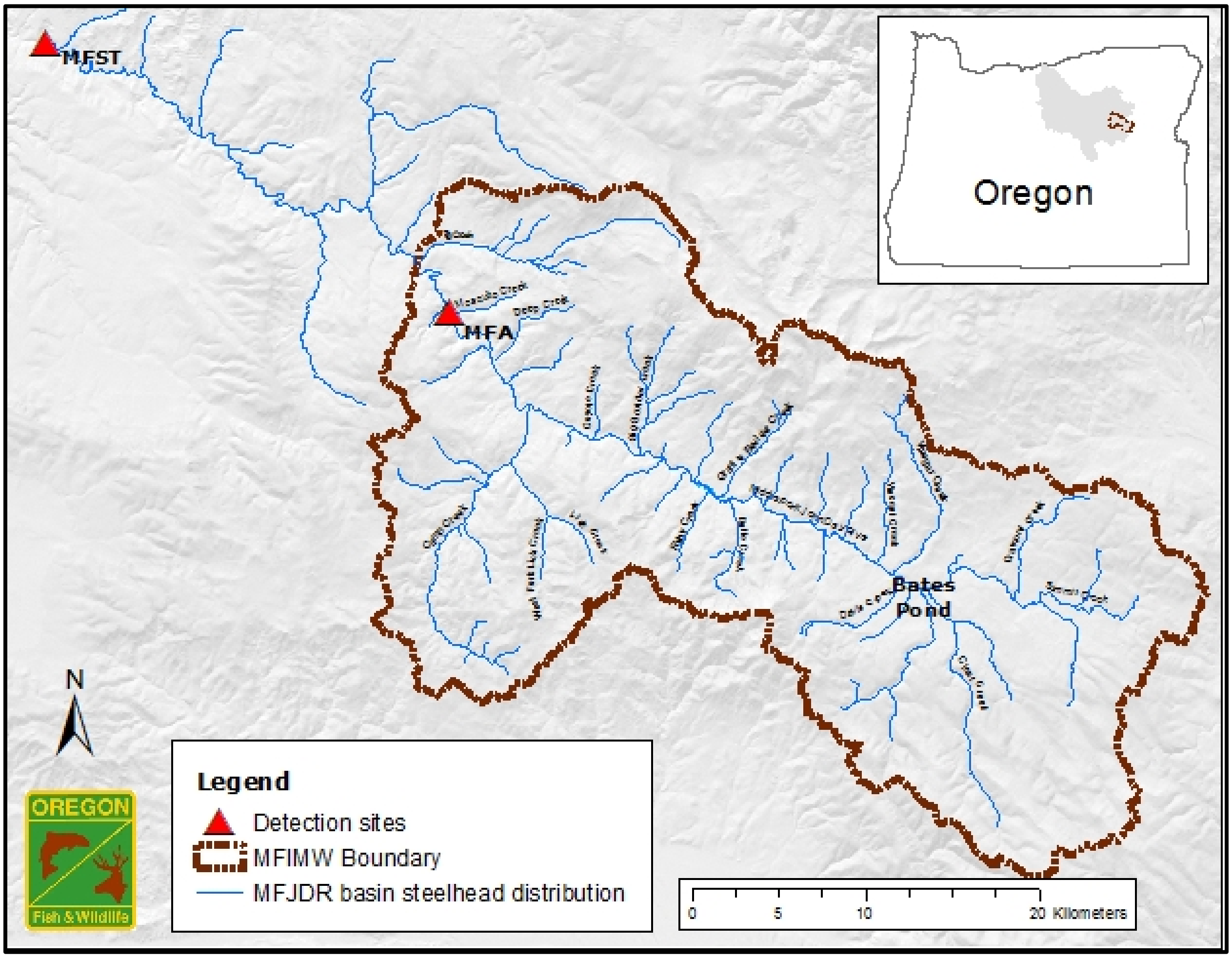

Figure 5. Map of the location of Juvenile detection and recapture site arrays operated in the MFJDR. Two PIT tag antennae arrays were located near Bates Pond in Bridge Creek. One array (MFA) was located near the downstream end of the MFIMW in the MFJDR near Mosquito Creek.

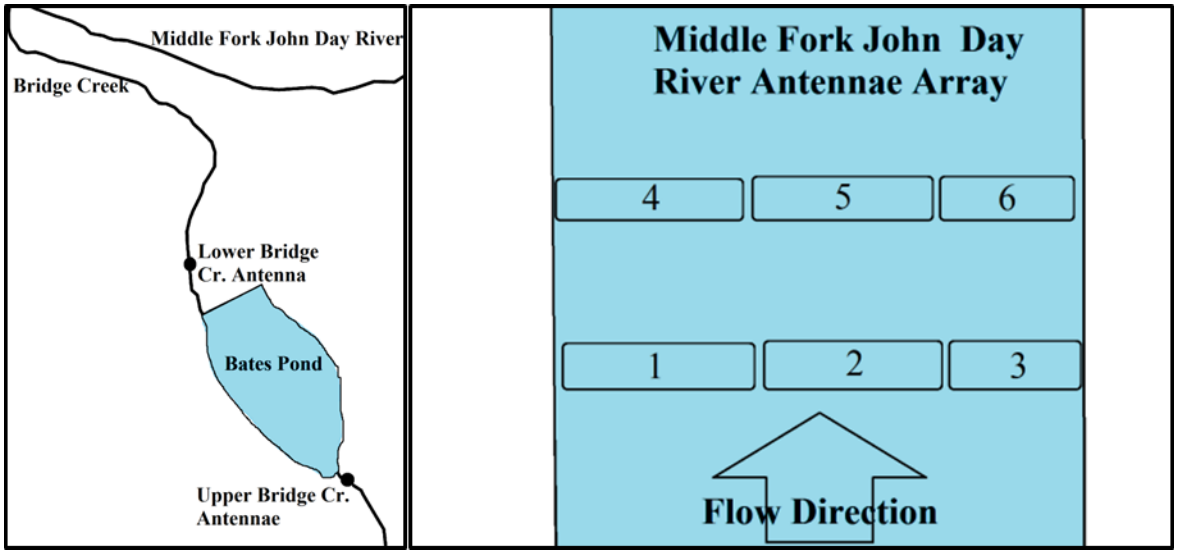

Figure 6. PIT tag antennae locations near Bates Pond (left) and schematic of antenna array near Mosquito Creek in the MFJDR.

Juvenile Chinook Distribution Bates Pond

In addition to using the PIT tag arrays to monitor passage of parr near Bates Pond, we also conducted snorkel surveys in Bridge Creek to evaluate the distribution of Chinook parr. Chinook do not typically spawn in Bridge Creek, therefore, parr that are encountered likely entered and migrated upstream from downstream locations in the MFJDR. During these surveys, snorkelers carefully entered pools and visually identify parr. The location of parr observations were then recorded using hand-held GPSs and later mapped in a GIS. Assuming no adult Chinook had spawned upstream of Bates Pond the previous year, we could conclude that all Chinook parr observed upstream of Bates Pond originated downstream and successfully migrated past both the pond and ladder. Parr distribution was also surveyed downstream of Bates Pond in Bridge Creek in 2012 and 2014 to determine their presence in the tail water.

The John Day District office has conducted steelhead spawning ground surveys on a 4km reach of upper Bridge Creek annually since the completion of the fish ladder in 2000. As part of MFIMW monitoring, John Day Fish research personnel also surveyed a random two kilometer reach in Bridge Creek upstream of Bates Pond every odd year. A two kilometer section of Bridge Creek from USFS road 2416 downstream to Bates Pond has also been surveyed each September for adult Chinook salmon and redds since 2011.

Results

Adult Escapement and Freshwater Productivity

Steelhead

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 IMW was established (Pearson Product Moment Correlation = 0.66, $P < 0.01$; Figure 7). The difference in escapement estimates between these two watersheds is greater after the MFIMW was established potentially a result of the more reliable and spatially balanced sampling protocol initiated during this time period (Figure 8).

Freshwater productivity (smolts produced per spawner) in the MFIMW has not increased relative to the SFJDR (Figure 9). Freshwater productivity has trended downward in the MFIMW since 2008 (Figure 9).

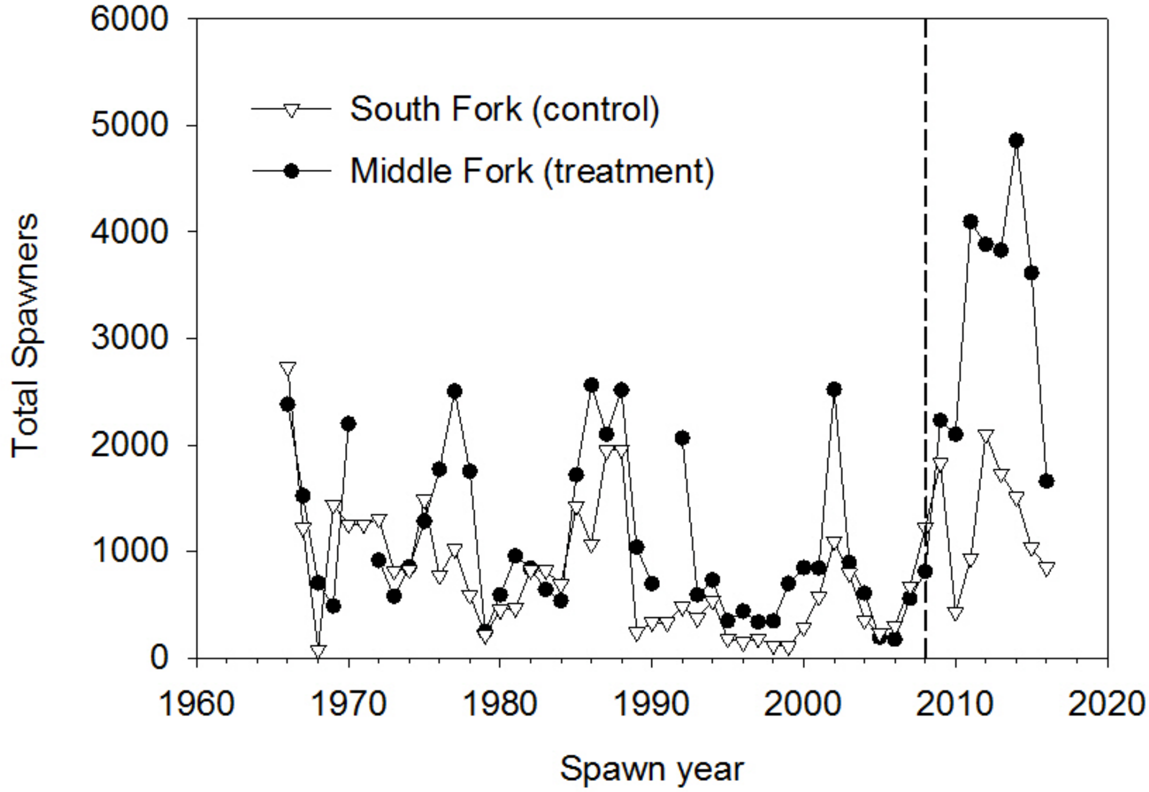

Figure 7. Long-term trends in adult spawner abundance for the Middle Fork and South Fork John Day steelhead populations. Vertical dashed line indicates initiation of the MFIMW experimental period (2008).

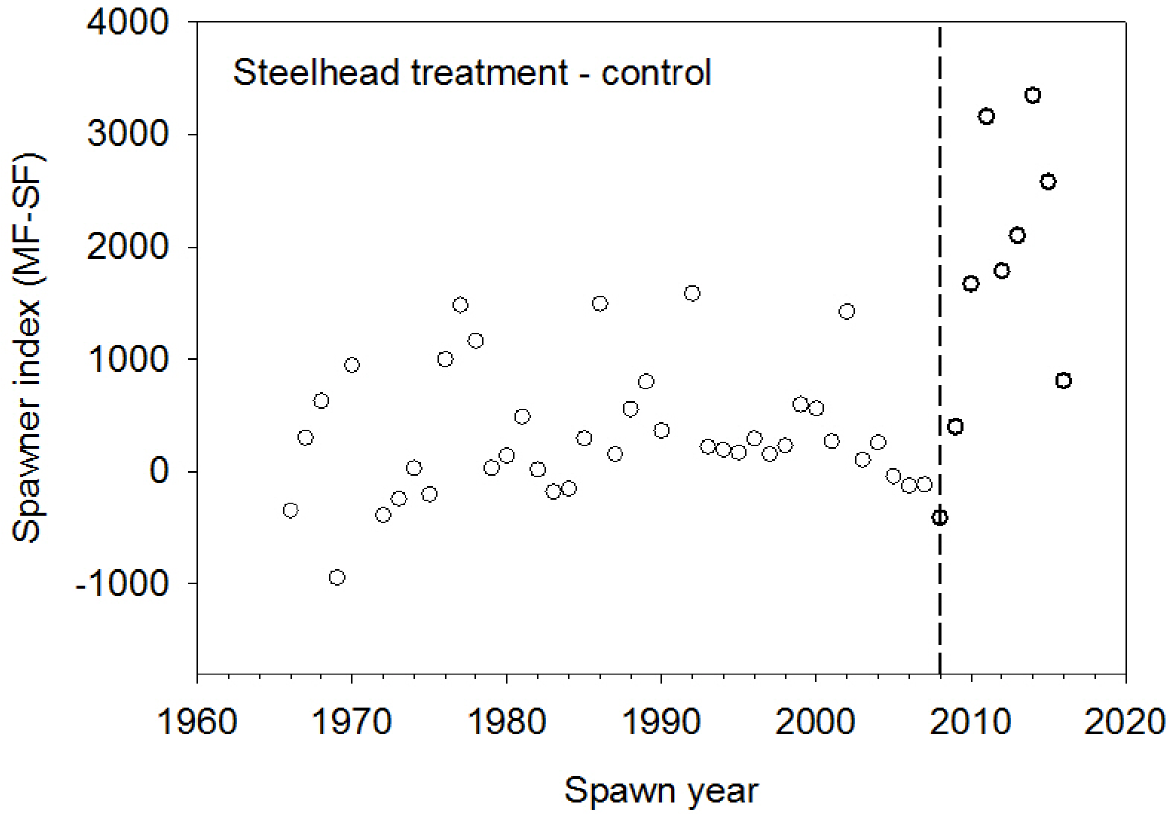

Figure 8. Index of adult steelhead spawner abundance before and after initiation of the MFIMW. The index shown is the difference of spawner abundance of the Middle Fork (treatment) population and the South Fork (control) population. Vertical dashed line indicates initiation of the IMW experimental period (2008).

Chinook

Trends in adult Chinook population escapement in the John Day Basin are driven by ocean and climactic conditions similar to steelhead (Figure 10). Adult Chinook escapement in the MFIMW followed a similar trend to the adult escapement in the Upper Mainstem and North Fork John Day prior to 2008 when the IMW was established (Pearson Product Moment Correlation ≥ 0.61, $P < 0.01$; Figure 10). Adult Chinook escapement is measured on the spawning grounds after these fish have spent the summer holding in locations near their spawning grounds. The MFJD basin population of adult Chinook 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. These events occurred in 2007, 2013, and 2015 and reduced the number of adults building redds on spawning grounds during those years (Figure 11). Chinook Spawner abundance in the MFJD basin has not increased relative to the combined Mainstem and North Fork John Day population segments (Figure 12).

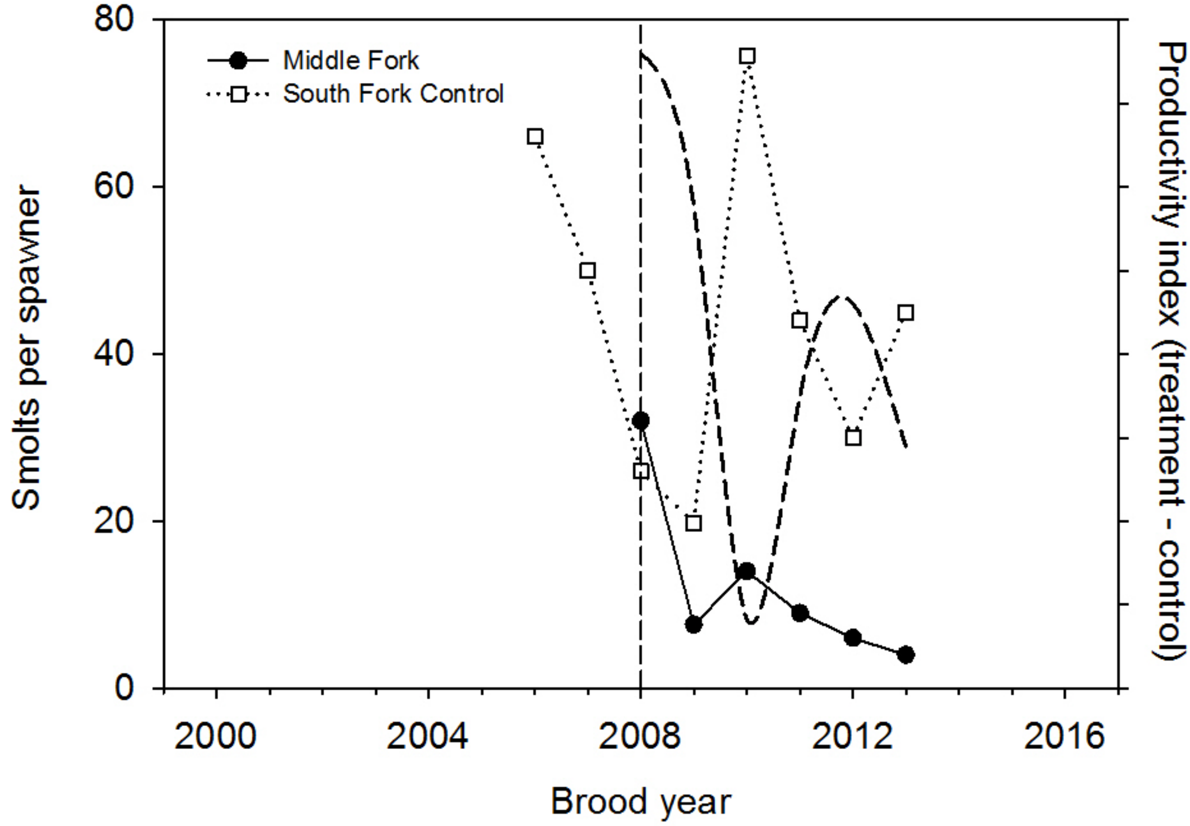

Figure 9. Trends in juvenile freshwater productivity for the Middle Fork and South Fork John Day steelhead populations. An index of the influence of the MFIMW restoration actions is shown (interpolated dashed line) as the difference of productivity of the Middle Fork (treatment) population and the South Fork (control) population. Vertical dashed line indicates initiation of the MFIMW experimental period (2008).

Trends in smolt abundance in the MFJDR basin relative the Upper Mainstem John Day Basin were relatively stable (Figure 13). Freshwater productivity of both basins has declined since the first year of the MFIMW (Figure 14).

Steelhead Parr Density

Steelhead parr densities showed more annual variation in Camp and Murderers Creek than Granite Boulder Creek (Figure 15). We were unable to detect a significant increase in steelhead parr density within our closed capture sites in Camp Creek post treatment (Figure 16). We also did not observe a significant change in density in Granite Boulder Creek or Murderers Creek over this same period.

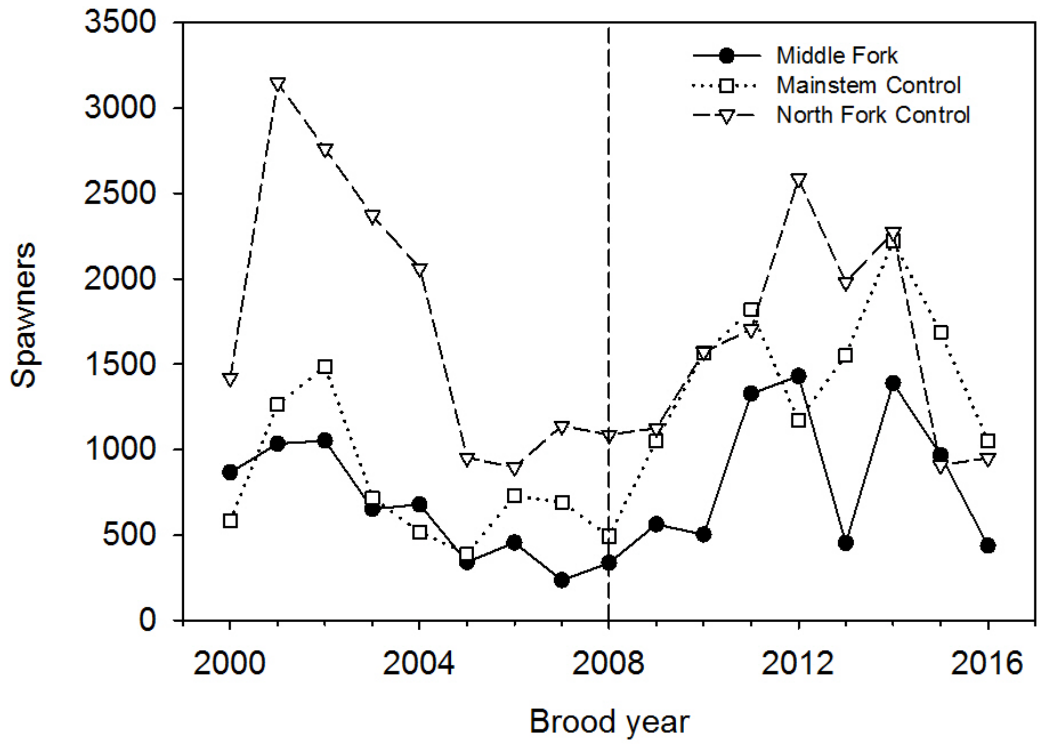

Figure 10. Long-term trends in adult spawner abundance for the Middle Fork, Upper Mainstem, and North Fork John Day Chinook populations. Vertical dashed line indicates initiation of the MFIMW experimental period (2008).

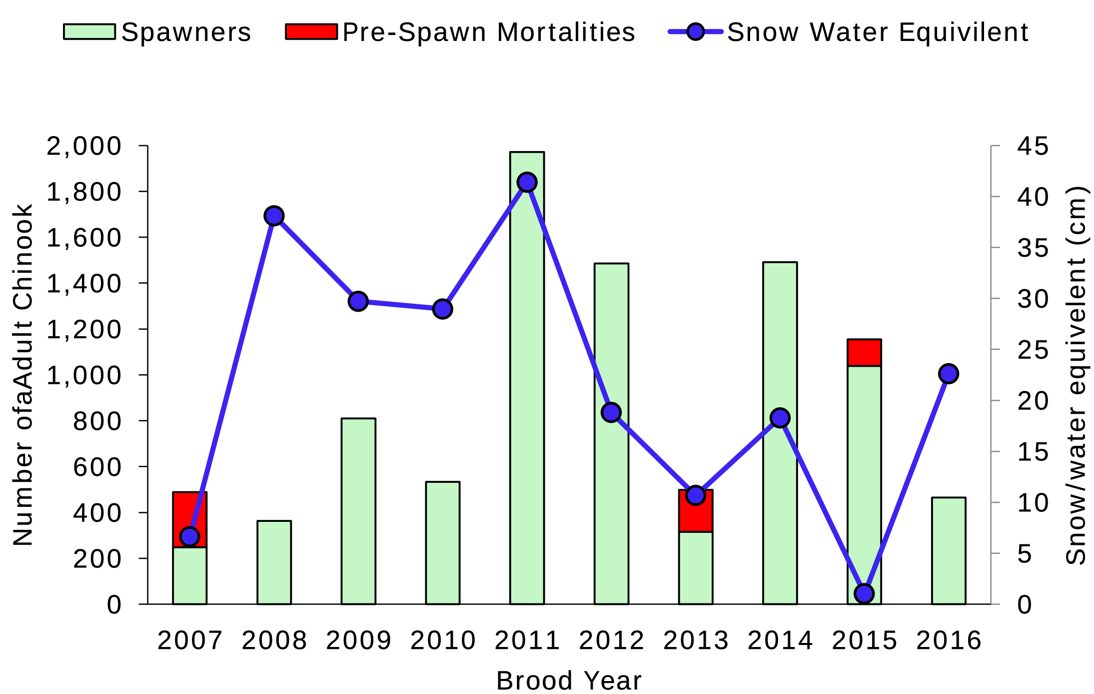

Figure 11. Total adult spring Chinook returns to the MFIMW and pre spawn mortality estimates for 2007, 2013, and 2015. April 15 Snow water equivalent estimates from the Tipton SnoTel site corresponding to each year are also shown.

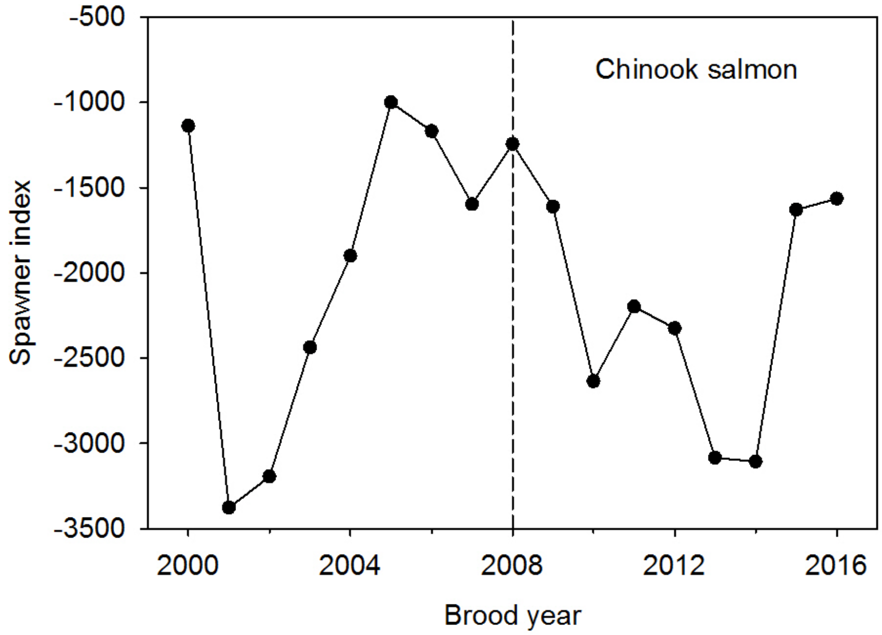

Figure 12. Before-after-Control-Impact comparison of adult Chinook spawner abundance before and after initiation of the MFIMW. The index shown is the difference of spawner abundance of the Middle Fork (treatment) population and the Mainstem and North Fork (control) populations. Vertical dashed line indicates initiation of the IMW experimental period (2008).

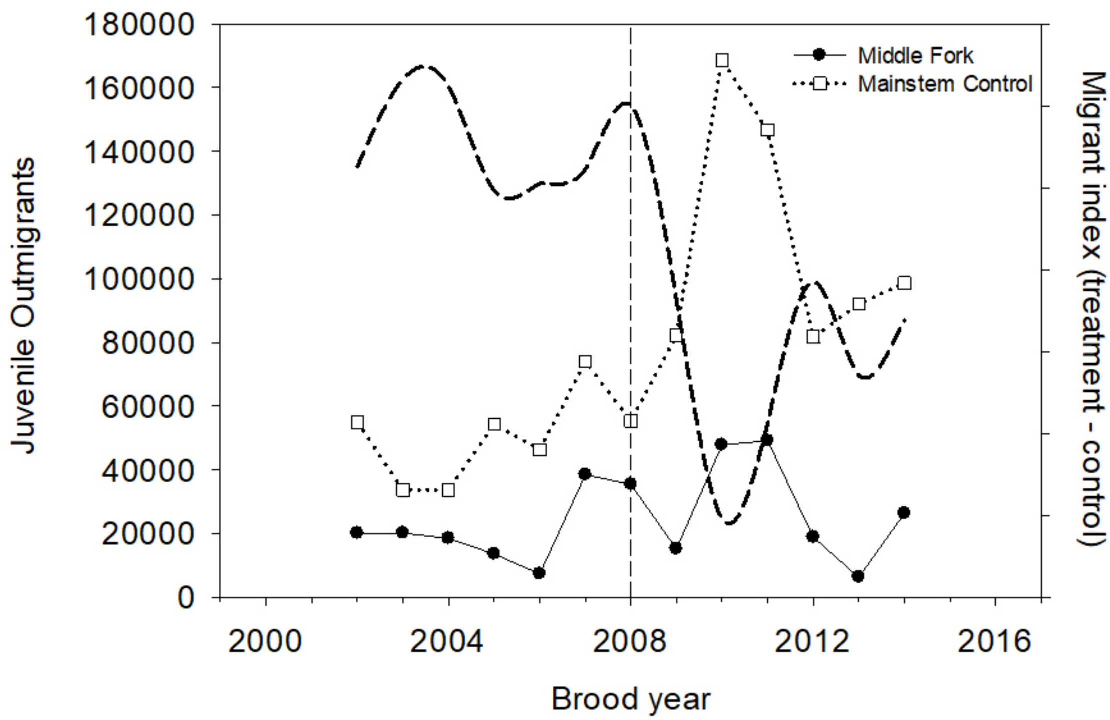

Figure 113. Trends in juvenile out-migrant abundance for the Middle Fork and Upper Mainstem John Day Chinook populations. An index of the influence of the MFIMW restoration actions is shown (interpolated dashed line) as the difference of migrant abundance of the Middle Fork (treatment) population and the Mainstem (control) population. Vertical dashed line indicates initiation of the MFIMW experimental period (2008).

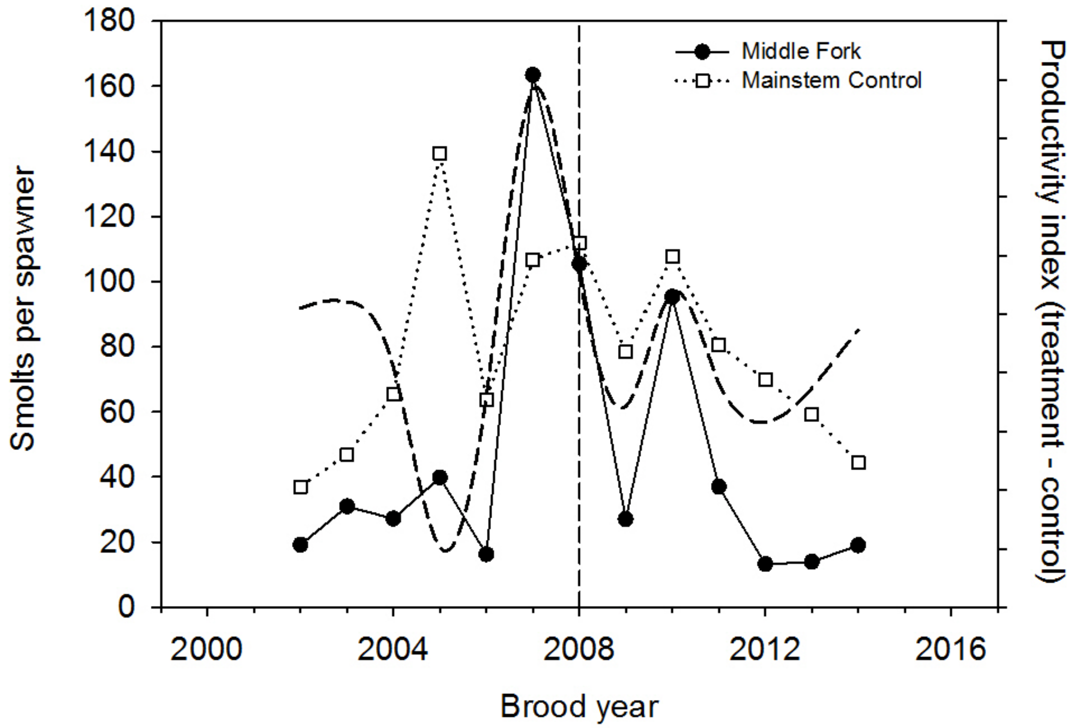

Figure 14. Trends in juvenile freshwater productivity for the Middle Fork and Upper Mainstem John Day Chinook populations. An index of the influence of the MFIMW restoration actions is shown (interpolated dashed line) as the difference of productivity of the Middle Fork (treatment) population and the Mainstem (control) population. Vertical dashed line indicates initiation of the MFIMW experimental period (2008).

In Camp Creek, age-1 steelhead productivity was most closely correlated with previous season base flows (Table 7). In Granite Boulder and Murderers Creek age-1 productivity was negatively correlated with brood year spawner densities (Table 7). Age-1 steelhead in all streams showed a negative relationship between spawner densities and reach specific age-1 productivity. Murderers Creek and Camp Creek showed a positive relationship for age-1 productivity and base flow during the summer of their first year (Figure 17, Table 7). Age-1 productivity did not show a consistent pattern among streams nor correlation with parr density (Table 7).

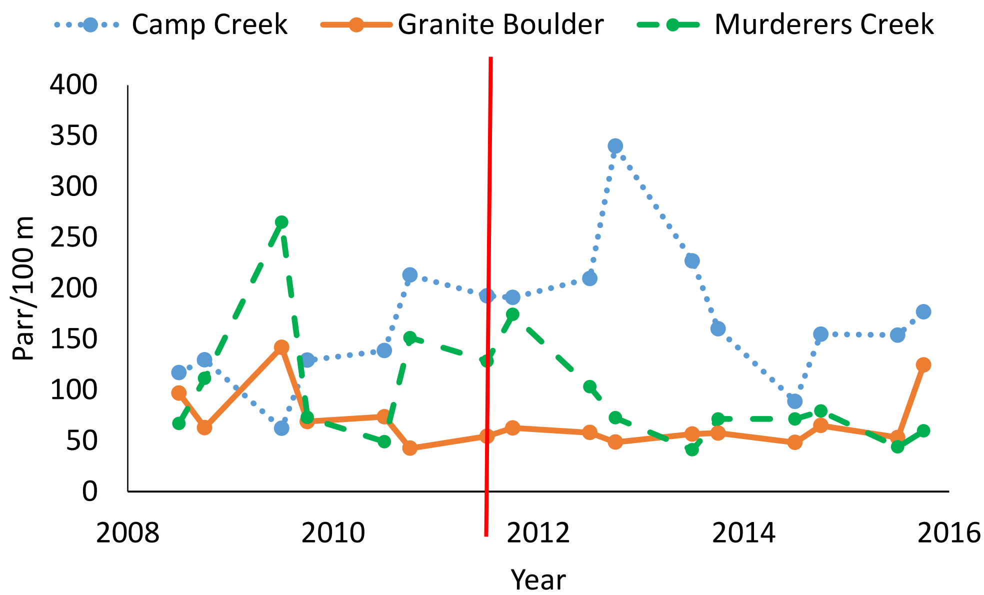

Figure 12. Steelhead parr density estimates at closed capture sites in Camp Creek, Granite Boulder Creek, and Murderers Creek from 2008 through 2015. The red line between 2011 and 2012 shows when the main treatment occurred

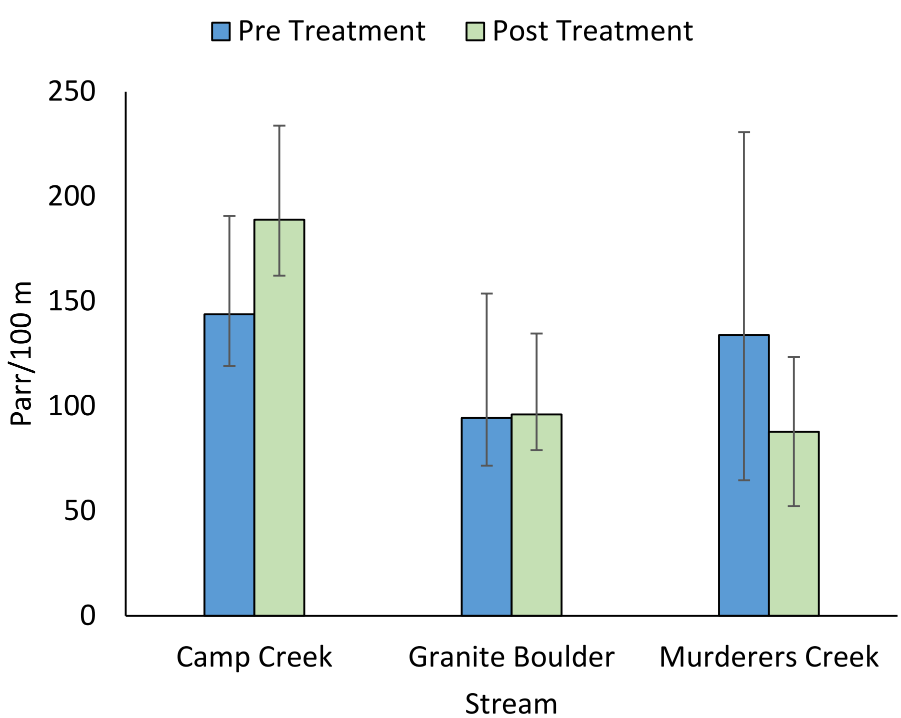

Figure 13. Average steelhead parr density within closed capture sties sampled from 2008 through 2015. Pre-Treatment period refers to 2008 through the summer of 2011 and post treatment densities refer to the period from the fall of 2011 through the fall of 2015.

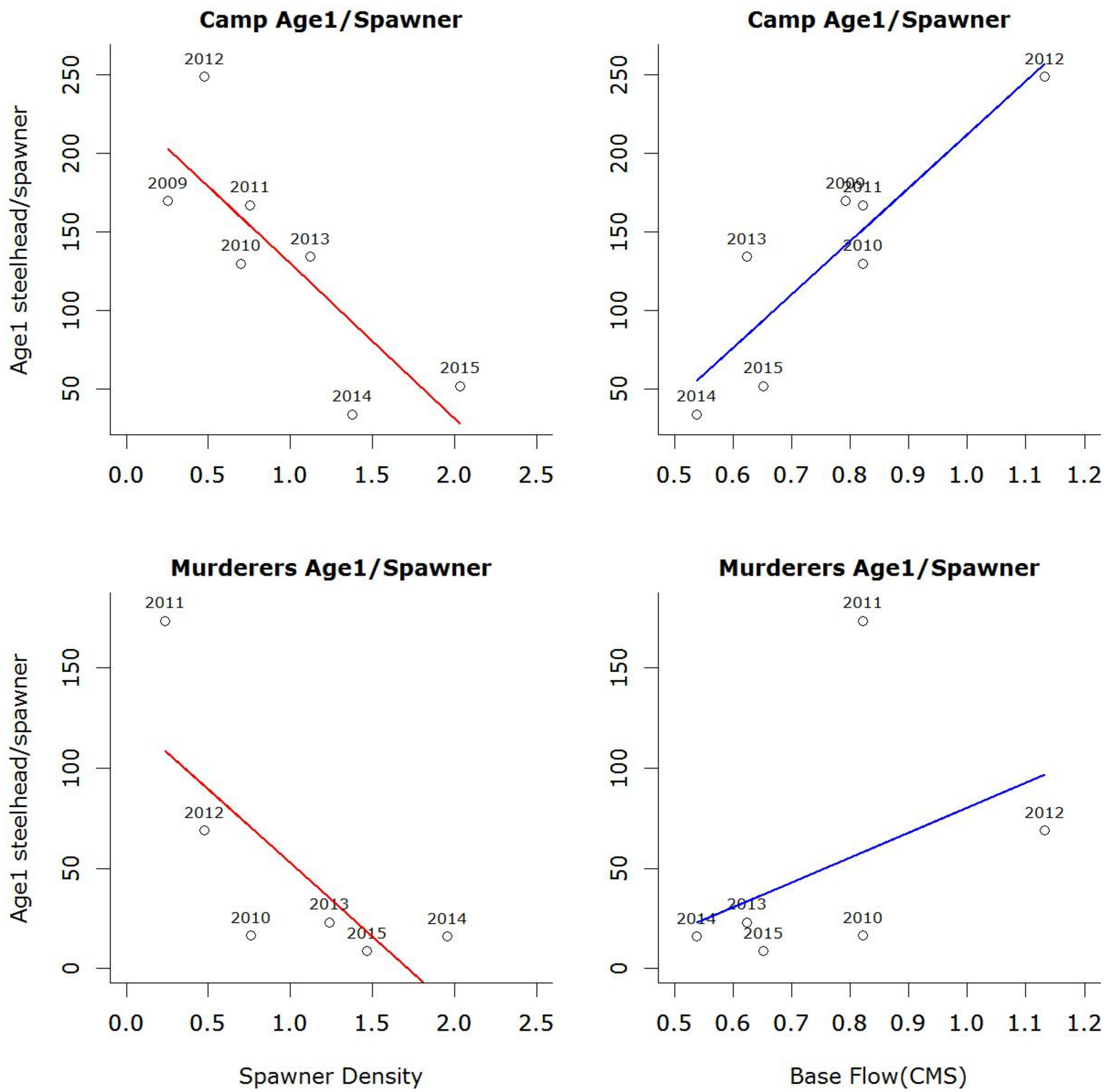

Figure 14. Actual (points) and modeled (lines) of age-1 parr/spawner in Camp and Murderers Creek. The left panels show age-1 steelhead produced per adult relative to brood year adult densities. Left panels show parr/spawner produced relative to base flows at the USGS Ritter gauge for the corresponding brood year. Points are labeled corresponding to the year of the juvenile density estimates in each stream.

Table 7. Correlation coefficients of age1 steelhead productivity (#/broodyear adult) at closed capture sites in each stream with broodyear spawner density, lowest discharge of MFJDR at the Ritter gauge for sample year (Base Flow), and parr density (#/100m) at closed capture sties from the previous fall (Parr Density).

Stream	Spawner Density	Base Flow	Parr Density
Camp Creek	-0.81	0.90	0.15
Granite Boulder Creek	-0.92	-0.31	-0.56
Murderers Creek	-0.75	0.42	0.65

Survival

Chinook Parr

The best fit model for Chinook parr survival suggests that survival is more variable by year than by location (Table 9). The second best model fit included covariates for brood year redds, time, and stratum but does not compete with the top model. Brood year redds represent the number of eggs deposited for each cohort of parr and may be a good measure of how much intraspecific competition will occur the following season (Table 9). Model 3 grouped individuals by year, again indicating survival varies to a great degree annually (Table 9). Model 4 included average stream flow and time testing the hypothesis that stream flow influences survival differently through the course of a year (Table 9). Model 8 grouped individuals based on a reach condition ranking from the river styles classification system (O'Brien et al 2014) but considering it's similar structure to the other models (grouping by sites and time) reach condition ranking at sites we sampled apparently did not affect parr survival.

Table 8. Age specific steelhead densities from closed capture sites in Camp Creek (CC), Granite Boulder Creek (GBC), and Murderers Creek (MC) from 2008 through 2011. Brood year adult (BYSpawners) correspond to the age-1 parr cohort.

Stream	Year	Season	BYSpawners	fish/100m	Age0	Age1	Age2	Age3	Age4	Age5
CC	2008	Spring	NA	103	0	80	23	0	0	0
CC	2008	Fall	NA	114	52	53	7	2	0	0
CC	2009	Spring	2.54	55	0	43	11	1	0	0
CC	2009	Fall	2.54	114	37	59	16	1	0	0
CC	2010	Spring	6.99	122	0	91	27	5	0	0
CC	2010	Fall	6.99	188	89	79	18	2	0	0
CC	2011	Spring	7.55	168	0	126	39	2	0	0
CC	2011	Fall	7.55	167	48	90	26	2	0	0
CC	2012	Spring	4.75	185	0	118	58	6	2	0
CC	2012	Fall	4.75	299	154	100	41	2	1	0
CC	2013	Spring	11.20	200	0	150	39	9	1	0
CC	2013	Fall	11.20	141	5	102	28	5	1	0
CC	2014	Spring	13.75	78	2	47	27	2	1	0
CC	2014	Fall	13.75	136	56	55	24	1	0	0
CC	2015	Spring	20.36	136	1	106	28	1	0	0
CC	2015	Fall	20.36	156	66	74	15	0	0	0
GBC	2008	Spring	NA	112	0	76	25	7	4	0
GBC	2008	Fall	NA	74	6	48	14	4	1	0
GBC	2009	Spring	0.00	145	0	70	57	13	4	1
GBC	2009	Fall	0.00	71	15	22	24	7	3	1
GBC	2010	Spring	NA	109	0	60	27	19	3	1
GBC	2010	Fall	NA	63	11	33	12	5	2	1
GBC	2011	Spring	3.18	108	0	65	30	13	1	0
GBC	2011	Fall	3.18	125	37	57	23	6	1	1
GBC	2012	Spring	NA	86	0	47	26	8	2	2
GBC	2012	Fall	NA	72	25	28	12	6	2	0
GBC	2013	Spring	1.55	84	1	66	16	1	1	0
GBC	2013	Fall	1.55	85	23	50	9	2	1	0
GBC	2014	Spring	NA	72	0	42	24	5	1	1
GBC	2014	Fall	NA	96	14	54	25	4	0	0
GBC	2015	Spring	6.68	79	0	61	16	2	0	0
GBC	2015	Fall	6.68	185	104	65	13	3	0	0
MC	2008	Spring	NA	73	0	16	39	15	2	0
MC	2008	Fall	NA	111	5	45	44	15	1	0
MC	2009	Spring	NA	265	1	62	162	32	9	0
MC	2009	Fall	NA	106	5	44	47	11	0	0
MC	2010	Spring	7.62	49	0	13	23	12	1	0
MC	2010	Fall	7.62	151	14	79	46	12	0	0
MC	2011	Spring	2.39	128	1	41	68	16	2	0
MC	2011	Fall	2.39	174	15	90	55	14	0	0
MC	2012	Spring	4.75	103	0	33	58	11	1	0
MC	2012	Fall	4.75	134	3	69	52	10	0	0
MC	2013	Spring	12.36	72	1	28	36	5	1	0
MC	2013	Fall	12.36	101	32	40	24	5	0	0
MC	2014	Spring	19.58	72	1	31	32	7	1	0
MC	2014	Fall	19.58	79	10	43	21	5	1	0
MC	2015	Spring	14.69	55	9	13	25	7	1	0
MC	2015	Fall	14.69	85	29	36	15	5	0	0

Table 9. Rank of models tested for Chinook parr survival using the multi-state model at sites throughout the MFIMW from 2011 through 2015. The covariates modeled with survival are ranked from low to high (Model Rank) Akaike Information Criterion scores adjusted for small sample size (AICc). Parameter counts are shown in the table in the npar column. Model structure of probability of capture and transition probabilities remained the same for each survival model tested. Covariate definitions are shown in Table 2.

Model Rank	Survival covariates	npar	AICc	ΔAICc	weight	Deviance
1	Year + time*stratum	41	27364	0	1	4161
2	redds + time*stratum	38	27579	215	0	4383
3	Year*stratum	34	27639	275	0	4451
4	aveflow + time*stratum	38	27683	319	0	4486
5	MarkSite + time*stratum	47	27801	437	0	4586
6	stream + time*stratum	38	27836	472	0	4639
7	Average + time*stratum	39	27848	484	0	4650
8	condition + time*stratum	39	27853	489	0	4654
9	treatment + time*stratum	38	27858	494	0	4661
10	Average + T*stratum	29	28132	768	0	4953
11	aveflow*stratum	28	28164	800	0	4988
12	MarkSite*stratum	46	28183	819	0	4971
13	stream*stratum	28	28247	883	0	5071
14	CapSite*stratum	48	28287	923	0	5070
15	condition*stratum	30	28297	933	0	5116
16	condition + treatment*stratum	30	28302	938	0	5122
17	treatment*stratum	28	28304	939	0	5127

Annual variability in Chinook parr survival is driving juvenile population trends at the basin wide scale. Multi-state model parameter estimates show significant differences in survival between 2011 and all other years we monitored (Figure 18). Model predictions for Chinook parr survival rates from model 2 (Table 9) suggest survival is also density dependent at the basin wide scale. Slopes of model predictions for survival suggest that density dependence has a greater influence on survival during July and August than in September (Figure 19). Model predictions for flow derived from model 4 (Table 9) show that survival is positively related to streamflow (Figure 19). This model is not ranked as a competing model based on AICc scores but considering the drastic differences in predicted survival at different flow volumes (Figure 19) it likely explains a great degree of the annual variation in survival rates for model 1 parameter estimates (Figure 17). Parameter estimates from model 9 (Table 9) show that survival does not vary significantly between control and treatment reaches as defined by the MFIMW (Figure 18).

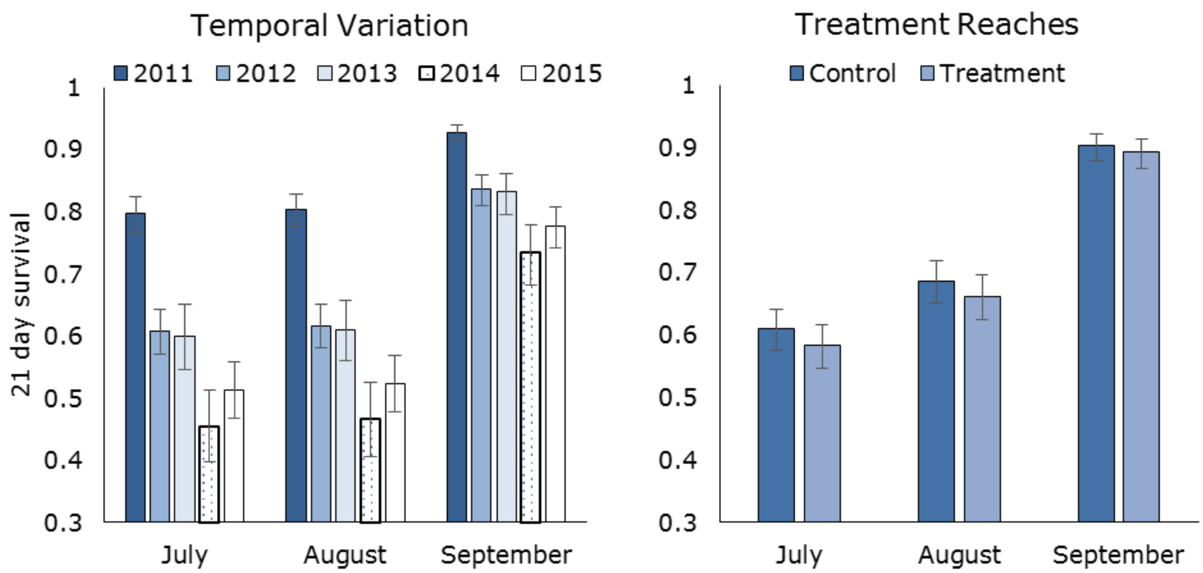

Figure 15. Best fit model parameter estimates of 21 day survival rates for Chinook parr captured at mark recapture sites throughout the MFIMW. Error bars represent 95% confidence bounds.

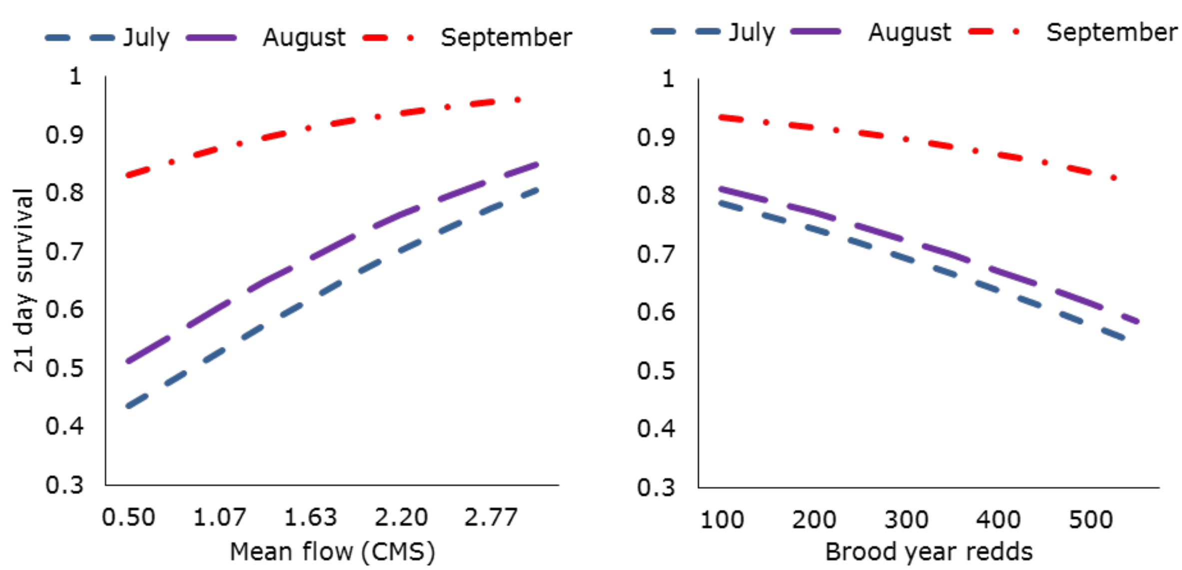

Figure 16. Multi-state model predictions for Chinook parr three week survival rates throughout the summer with varying mean streamflow rates (left panel) and brood year redd counts from the previous year (right panel).

Steelhead Parr

Model fit results from juvenile steelhead survival modeling suggest that survival rates varied between each sampling site and throughout the period we monitored from 2008 through 2015. Model ranking results suggest that individual age and restoration actions had less influence on survival than individual site characteristics and annual and seasonal variation (Table 10).

Table 10. Rank of Barker survival models tested for steelhead parr captured at sites in Camp Creek, Granite Boulder Creek, and Murderers Creek from 2008-2015. The covariates modeled with survival are ranked from low to high (Model Rank) Akaike Information Criterion scores adjusted for small sample size (AICc). Parameter counts are shown in the table in the npar column. Model structure of probability of capture, movement, detection, and site fidelity remained the same for each survival model tested. Covariate definitions are shown in Table 3.

Model Rank	Survival Model	npar	AICc	△AICc	weight	Deviance
1	stream * RKM + time	55	153054.4	0.00	0.81	133783.7
2	trueage + time * ta	66	153057.4	2.96	0.19	133764.5
3	trueage + treatment * ta	40	153137.2	82.77	0.00	133896.6
4	trueage + season * ta	38	153306	251.65	0.00	134069.5
5	flow * season * stream	40	153353	298.63	0.00	134112.5
6	treatment + time)	46	153430.4	376.02	0.00	134177.8
7	time + stream	46	153433.4	378.99	0.00	134180.8
8	time	44	153497.4	442.99	0.00	134248.8
9	Average * season * t	36	153671.7	617.29	0.00	134439.1
10	treatment + season	32	153685.5	631.14	0.00	134461.0
11	stream + season	32	153695.9	641.54	0.00	134471.4
12	density + season * d	33	153722.4	668.00	0.00	134495.9
13	treat + season	31	153788.4	734.02	0.00	134565.9
14	sitetreat + season	31	153814.1	759.66	0.00	134591.5

Results from model the treatment and age model (trueage + treatment * ta; Table 10) show lower survival rates for all age classes of steelhead after the 2011 treatment but they remain higher than survival rates at control sites over this same period (Figure 20).

Model predictions of steelhead parr survival from model 5 (Table 10) suggest improvement in survival with higher average summer streamflow in all streams modeled (Figure 21). Model predictions from model (stream* RKM+time; Table 10) show a positive relationship between parr density and survival rates during the summer and winter seasons (Figure 22). Model predictions for survival compared with average four month temperatures from June 15 to October 15 indicate decreasing survival rates with increases in average summer temperatures (Figure 21).

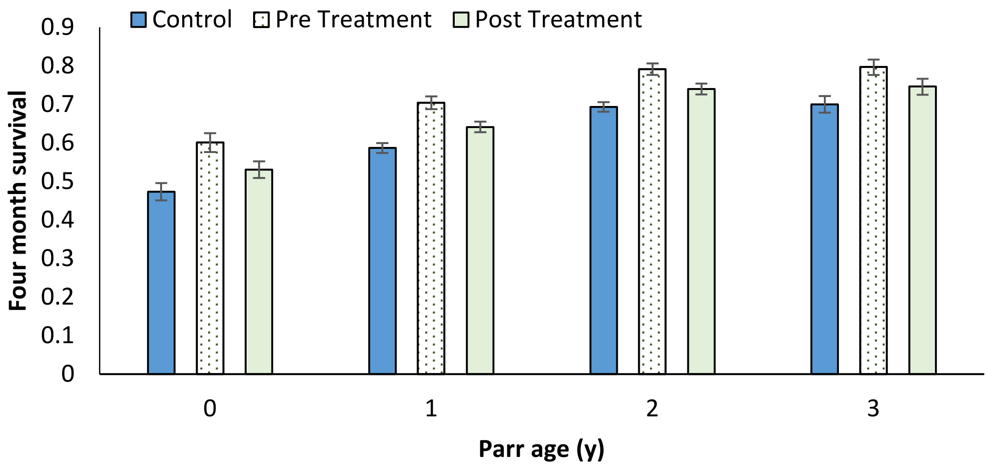

Figure 17. Age specific steelhead parr survival estimates from 2008 through 2015 at control and treatment streams. Control streams include Murderers Creek and Granite Boulder Creek. Pre and Post Treatment indicates survival before and after the 2011 treatment in Camp Creek. Error bars represent 95% confidence intervals.

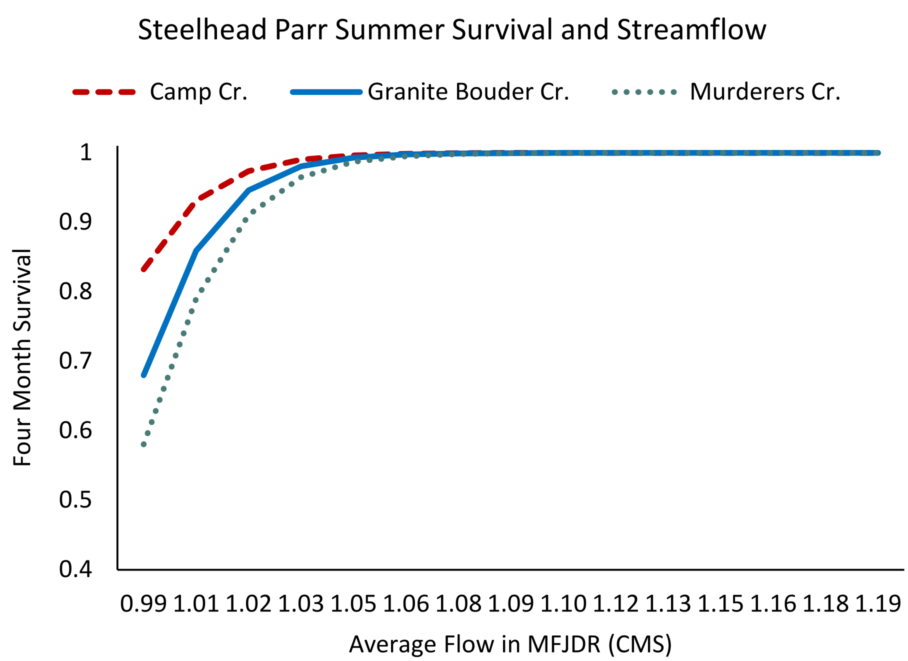

Figure 18. Barker model predictions for steelhead parr survival at closed capture sites and average stream flow between summer and fall mark recap runs for sites within the MFJDR watershed. Streamflow units are cubic meters per second from the USGS gauge near Ritter Oregon.

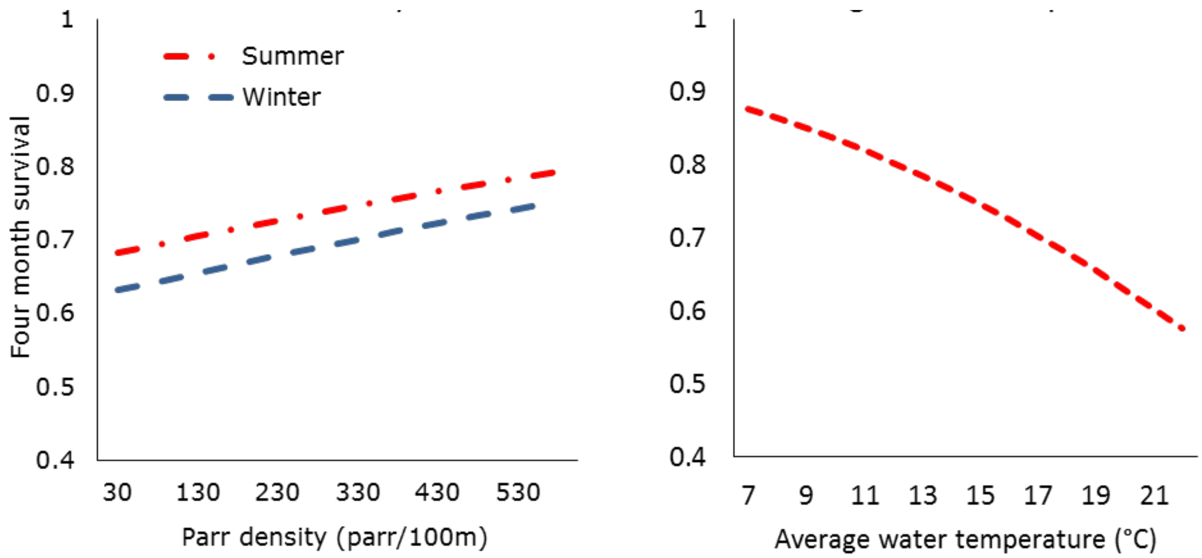

Figure 19. Barker model predictions of four month survival rates for steelhead parr at varying parr densities (left) within closed capture sites for summer and winter periods. Model predictions of four month steelhead parr survival rates as a function of water temperature (right) at closed capture sites in Camp Creek, Granite Boulder Creek, and Murderers Creek from 2008 through 2015.

Growth

Mixed Effects Model Results

Chinook Parr

Model ranking based on AICc scores suggest that Chinook parr growth varied more among streams than by treatment or reach condition assignment. However, the models that grouped growth by treatment and reach condition are ranked lower than models with no reach assignment (Table 11).

Model predictions from the best fit model in (Table 11) indicate that growth rates in tributary streams are not significantly different among streams (Figure 23). Growth predictions based on the “Treatment” model (Table 4) indicate no significant difference in growth between control and treatment reaches (Figure 24). When grouped by reach condition the only reach classified as “poor” showed higher growth rates than reaches in “moderate” condition however this difference was not significant (Figure 24).

Table 11. Model Rank of mixed effect growth models tested for Chinook parr growth tested for all individuals marked and recaptured from 2008 through 2015 throughout the MFJDR IMW.

Model	Rank	K	AICc	∆AICc	ML	AICcWt
Stream	1	13	-1135.93	0.00	1.00	1.00
Treatment X Condition	2	13	-667.04	468.89	0.00	0.00
Treatment	3	7	-350.28	785.65	0.00	0.00
Condition	4	9	-229.20	906.73	0.00	0.00
AveFlow X BYCHS	5	8	-132.09	1003.84	0.00	0.00
AveFlow + BYCHS	6	8	-131.65	1004.28	0.00	0.00
AveFlow	7	7	-131.58	1004.35	0.00	0.00
FlowV	8	7	-90.54	1045.39	0.00	0.00
BYCHS	9	7	-45.30	1090.63	0.00	0.00
Length	10	5	-43.23	1092.70	0.00	0.00
Site	11	21	3047.99	4183.92	0.00	0.00

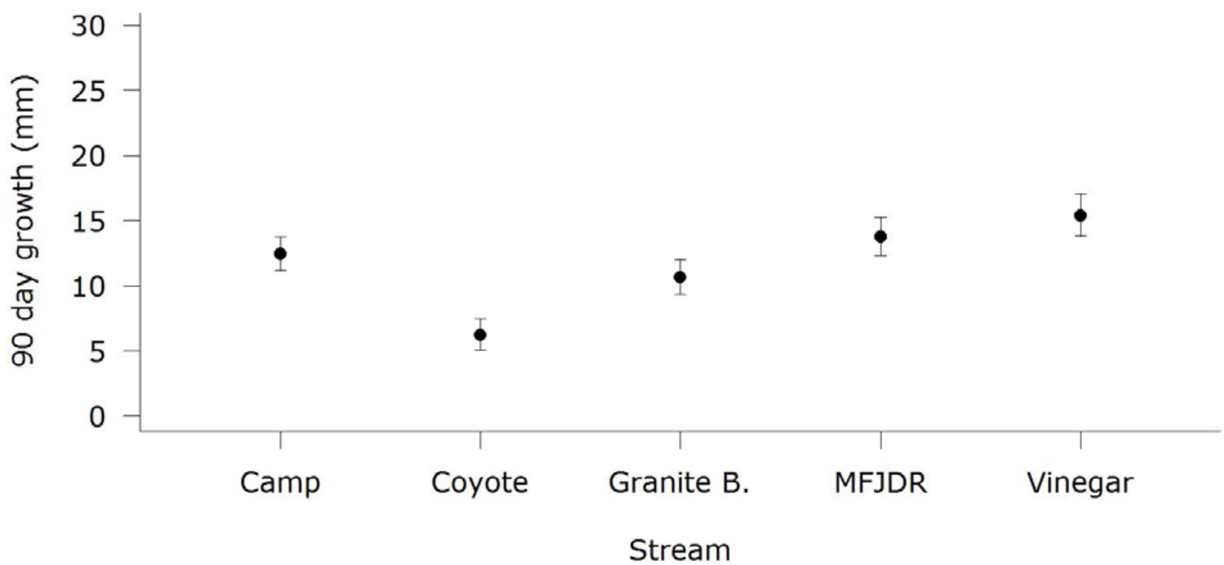

Figure 20. Best fit model estimates for Chinook parr growth grouped by stream. Growth estimates were obtained from Chinook parr captured and recaptured throughout the MFIMW from 2008–2015. These model predictions represent 90 day growth rates for a 65 mm Chinook parr, error bars represent ± 95% confidence intervals. Model estimates for Chinook parr growth grouped by treatment reach.

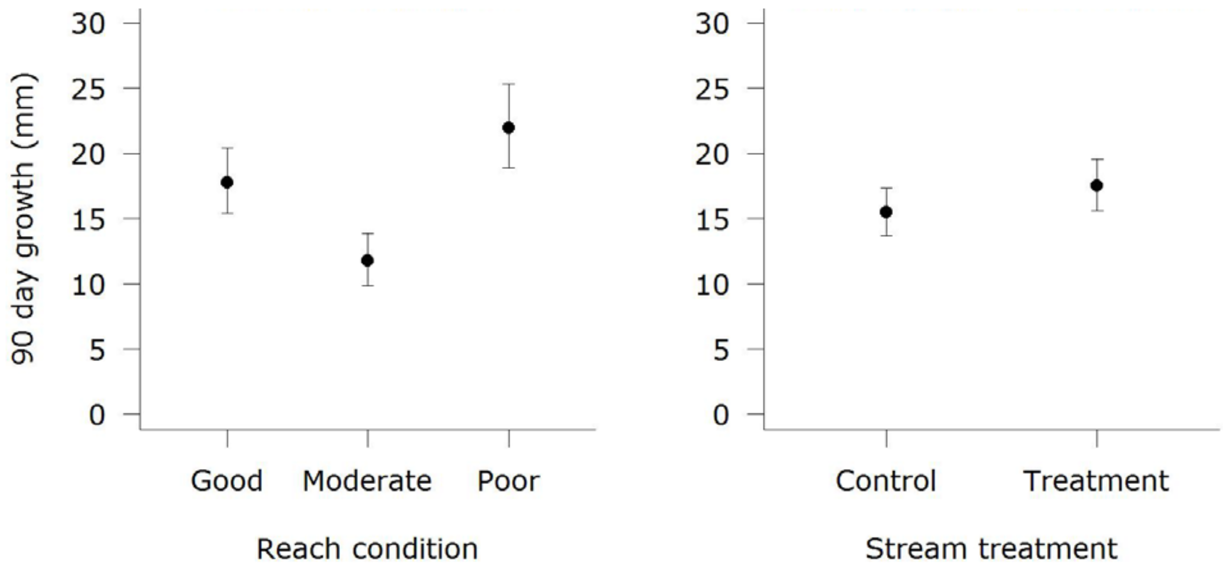

Figure 21. Mixed effects model estimates for Chinook parr growth grouped by reach condition, and treatment. Growth estimates were obtained from Chinook parr captured and recaptured throughout the MFJDR IMW from 2008–2015. These model predictions represent 90 day growth rates for a 65 mm Chinook parr, error bars represent ± 95% confidence intervals. Model estimates for Chinook parr growth grouped by treatment reach. Growth estimates were obtained from Chinook parr captured and recaptured throughout the MFJDR IMW from 2008–2015.

Mixed effects growth models including covariates showed tributary flow volume had the greatest influence on juvenile Chinook growth. The next best three models all included covariates for stream flow. Models including covariates for brood year redds and temperature ranked lower (Table 12).

Predictions from the best fit model in Table 12 indicate a positive relationship between flow volume and Chinook parr growth (Figure 25). This model predicts growth from tributary specific flow volumes estimated from the MFJDR, Camp Creek, and Granite Boulder Creek. Model predictions based on temperature suggest growth is negatively influenced by temperature at our sample sites (Figure 25). Predictions for growth from the density dependence model "BYCHS" which predicts growth from brood year redd counts indicates a negative relationship between high numbers of brood year redds and parr growth, suggesting density dependent growth (Figure 25).

Steelhead Parr

Model rank for mixed effects growth for steelhead parr indicate steelhead growth is best described by the stream individuals were marked and recaptured in. FLowV is the next best predictor of steelhead growth followed by brood year spawner density. Reach condition and reach treatment groupings did not fit these data as well (Table 13).

Table 12. Model Rank of mixed effect growth models tested for Chinook parr growth for subset of individuals associated with covariates for temperature and tributary flow marked and recaptured from 2008–2015 throughout the MFIMW.

Model	Rank	K	AICc	△AICc	ML	AICcWt
VolTrib	1	7	-746.69	0.00	1.00	1.00
FlowTrib	2	7	-689.79	56.90	0.00	0.00
AveFlow	3	7	-669.83	76.86	0.00	0.00
AveFlow X BYCHS	4	8	-668.16	78.53	0.00	0.00
XAverageT	5	7	-665.09	81.60	0.00	0.00
AveFlow + BYCHS	6	8	-652.97	93.71	0.00	0.00
FLowV	7	7	-572.01	174.68	0.00	0.00
Temp Group	8	10	-555.41	191.28	0.00	0.00
BYCHS	9	7	-507.15	239.54	0.00	0.00
Length	10	5	-495.84	250.85	0.00	0.00
+AverageT	11	6	-494.68	252.01	0.00	0.00

Table 13. Model Rank of mixed effect growth models tested for steelhead parr growth for all marked and recaptured individuals. These models were tested with all available steelhead mark recapture data from sites sampled throughout the MFIMW and Murderers Creek 2008 through 2015.

Model	Rank	K	AICc	△ AICc	ML	AICcWt
Stream	1	15	3200.76	0.00	1.00	1.00
FlowV	2	7	3538.62	337.85	0.00	0.00
AveFlow X BYSTSD	3	8	3634.48	433.72	0.00	0.00
BYSTSD	4	7	3664.33	463.56	0.00	0.00
AveFlow	5	7	3665.05	464.29	0.00	0.00
AveFlow + BYSTSD	6	8	3671.20	470.44	0.00	0.00
Treatment X Condition	7	15	3775.23	574.47	0.00	0.00
Condition	8	11	3784.10	583.33	0.00	0.00
Length	9	5	3793.44	592.68	0.00	0.00
Treatment	10	9	3795.33	594.57	0.00	0.00

Model predictions for steelhead growth from the best fit model (Table 13) show that growth rates are highest for steelhead captured and recaptured in Murderers Creek and the MFJDR (Figure 26). Growth rates were lowest for steelhead in Coyote Creek which was the smallest of the streams sampled (Figure 26). When grouped by treatment for the two streams with recent restoration activities growth rates were higher in treatment reaches of the MFJDR but reduced post treatment at Camp Creek sites (Figure 26).

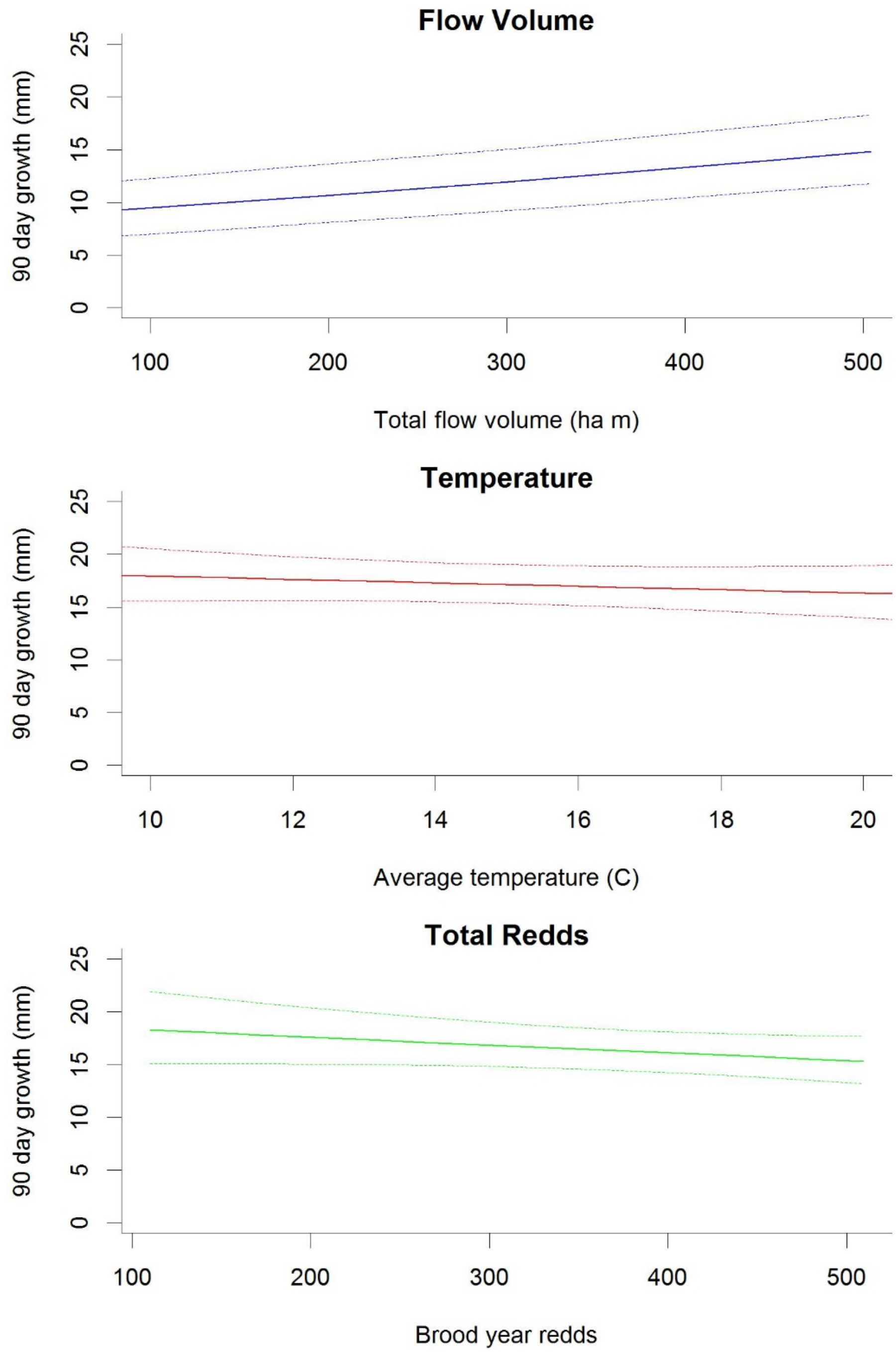

Figure 22. Mixed effects model predictions for Chinook parr growth relative to flow volume, average temperature, and brood year redds throughout the MFIMW. Growth estimates were obtained from Chinook parr captured and recaptured from 2011–2015. These model predictions represent 90 day growth rates for 65 mm Chinook parr, dashed lines represent ± 95% confidence bounds.

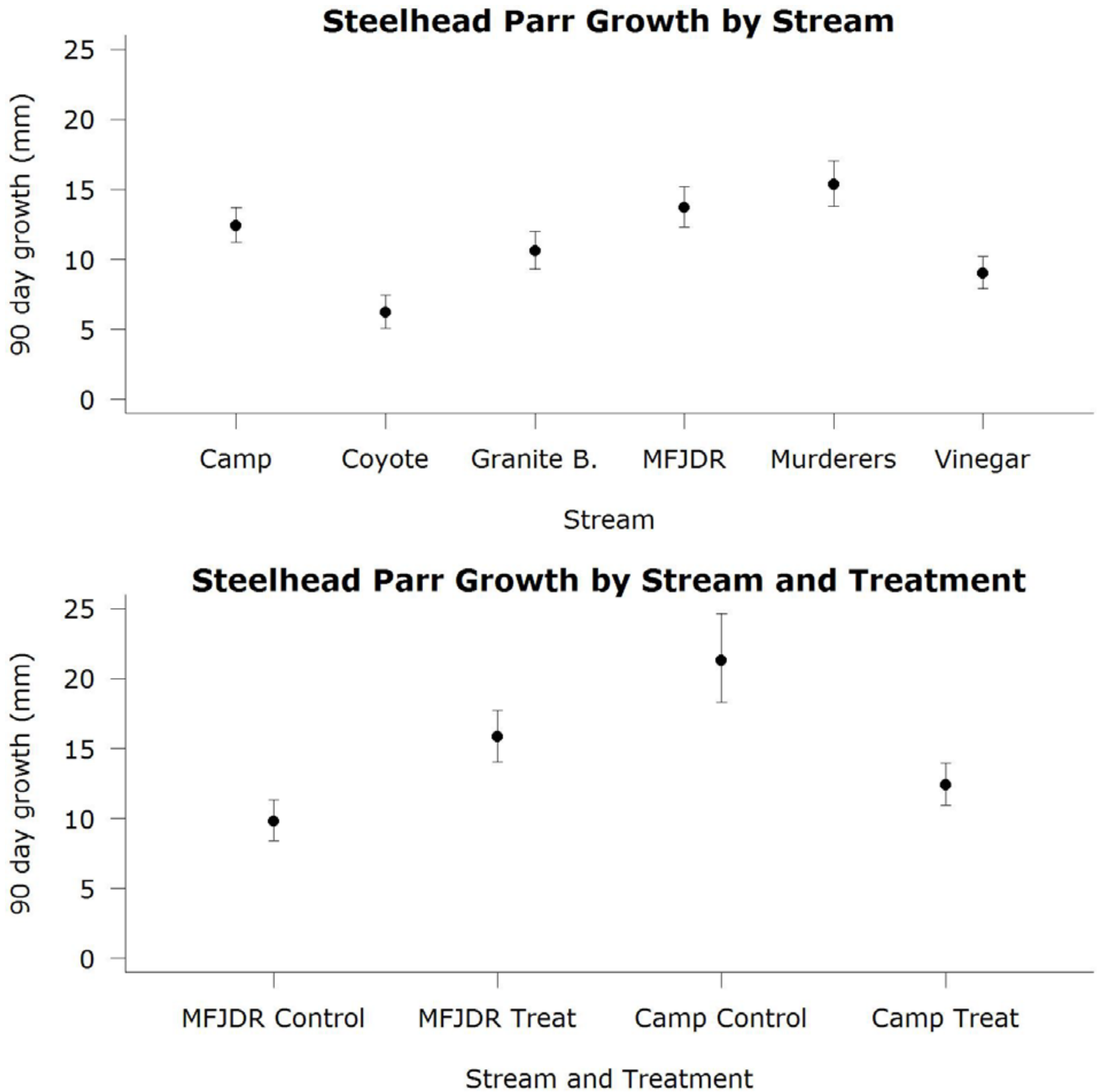

Figure 23. Growth rates for treatment and control groups for steelhead parr marked and recaptured throughout the MFJDR and in Camp Creek. MFJDR Control and MFJDR Treat refer to individuals marked in control and treatment reaches respectively. Camp Creek control group refers to individuals marked and recaptured prior to the 2011 treatment while Camp Treat refers to individuals marked and recaptured from 2012-2015. These point estimates represent 90 day growth rates for 105 mm steelhead parr, error bars represent ± 95% confidence intervals.

Models tested on steelhead growth data with available covariates for parr density, stream temperature, and tributary specific stream flow indicated stream temperature was the best predictor of growth at our sample sites followed by flow covariates (Table 14). Best fit model estimates for steelhead parr growth show that steelhead parr growth increases with temperature until temperatures exceed 16°C

(Figure 27). These results suggest optimal temperature for steelhead parr growth at our sample sites is approximately 14–18°C. Steelhead parr growth shows a negative relationship with parr density at closed capture sites (Figure 28).

Table 14. Model Rank of mixed effect growth models tested for steelhead parr growth. These models where tested with a subset of individuals associated with covariates for temperature, tributary flow, and parr density estimates from Camp Creek, Granite Boulder Creek, and Murderers Creek from 2008-2015.

Model	Rank	K	AICc	△ AICc	ML	AICcWt
Temp Group	1	11	1097.20	0.00	1.00	1.00
FlowV	2	7	1164.82	67.62	0.00	0.00
AveFlow	3	7	1167.86	70.66	0.00	0.00
AverageT	4	7	1169.60	72.41	0.00	0.00
VolTrib	6	7	1177.74	80.54	0.00	0.00
FlowTrib	7	7	1178.46	81.26	0.00	0.00
BYSTS	8	7	1181.97	84.77	0.00	0.00
ParrD	9	7	1215.93	118.73	0.00	0.00
BYSTSD	10	7	1221.30	124.10	0.00	0.00
Length	11	5	1236.98	139.78	0.00	0.00

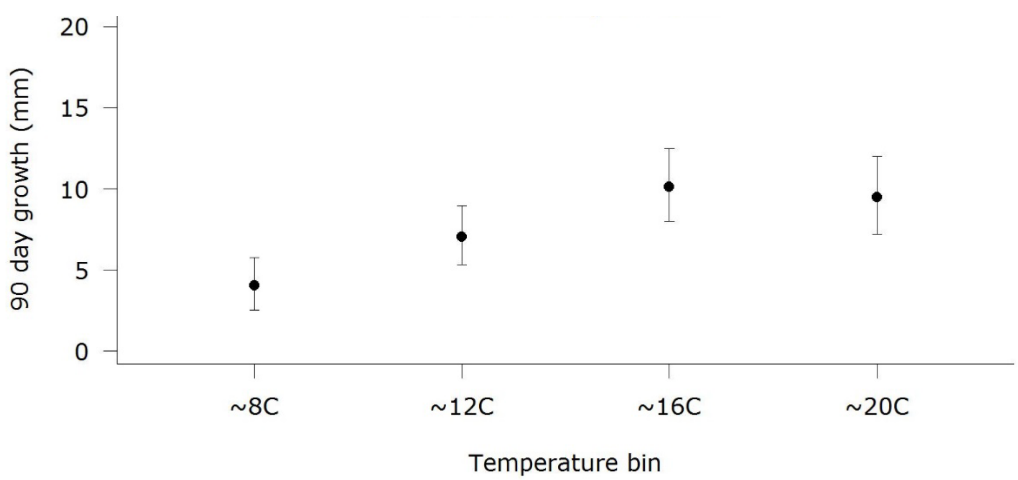

Figure 24. Best fit model parameter estimates for steelhead parr growth at binned modal temperatures observed between mark and recapture occasions from a mixed effects growth model. Point estimates represent model predicted 90 day growth rates for a 105 mm steelhead parr, error bars represent ±95% confidence intervals.

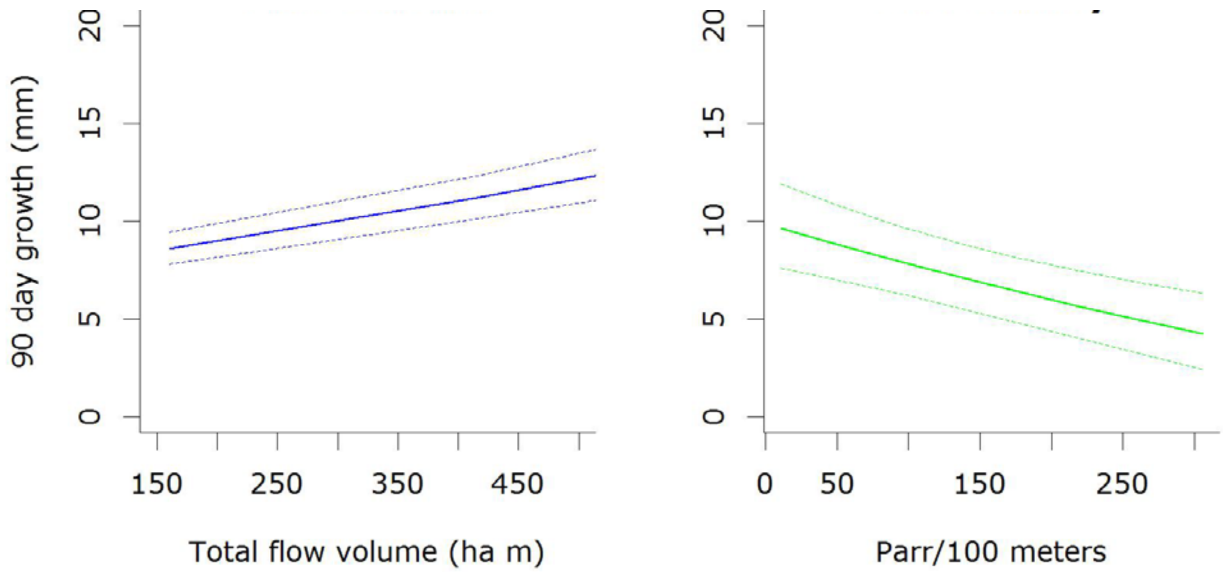

Figure 25. Mixed effects model predictions for steelhead parr growth relative to total flow volume (FlowV) on the left, and parr density (ParrD) on the right. Point estimates represent model predicted 90 day growth rates for a 105 mm steelhead parr, error bars represent ±95% confidence intervals.

Parr Distribution

Chinook parr occupied the lower reaches of many tributary streams throughout the MFIMW and were observed in the mainstem MFJDR from the headwaters near Phipps Meadow downstream to the USFS boundary near Mosquito Creek. Upstream distribution of Chinook salmon in tributaries of the MFJDR is limited by access and barriers to migration. In most tributaries to the MFJDR that we sampled it appeared that distribution ended at or near natural stream obstacles either in the form of channel spanning logs or cascades which apparently exceeded the jumping capabilities for juvenile salmon (Figure 29). Maximum jump heights for juvenile Coho salmon of comparable size are near 26 cm (Mueller et al. 2008). Juvenile steelhead or resident *O. mykiss* occupied every reach of stream that we sampled.

Temperature Influence on juvenile distribution

Chinook parr observations increased as snorkelers moved upstream with densities from less than one Chinook per km in the Mosquito to Deep Creek reach, to a high of 438/km in the Little Butte to Little Boulder reach near RKM 99 in the MFJDR (Table 15). We observed higher densities in the lower reaches of the MFJDR during the summer of 2016 compared to the previous two years (Table 16). This coincided with a slightly lower 7DA water temperature prior to sampling (Table 14). Logistic model results for Chinook presence (Figure 30) were fit by the equation:

$$p(C) = \frac{1}{e^{(-0.76T-15.14)}}$$

where T is the seven day average temperature, and p(C) is the probability of observing a Chinook in a 100 m reach of stream. Steelhead presence (Figure 31) was predicted using the equation:

$$p(S) = \frac{1}{e^{(-0.34T-7.55)}}$$

where T is the seven day average temperature, and p(S) is the probability of observing a juvenile steelhead in a 100 m reach of stream.

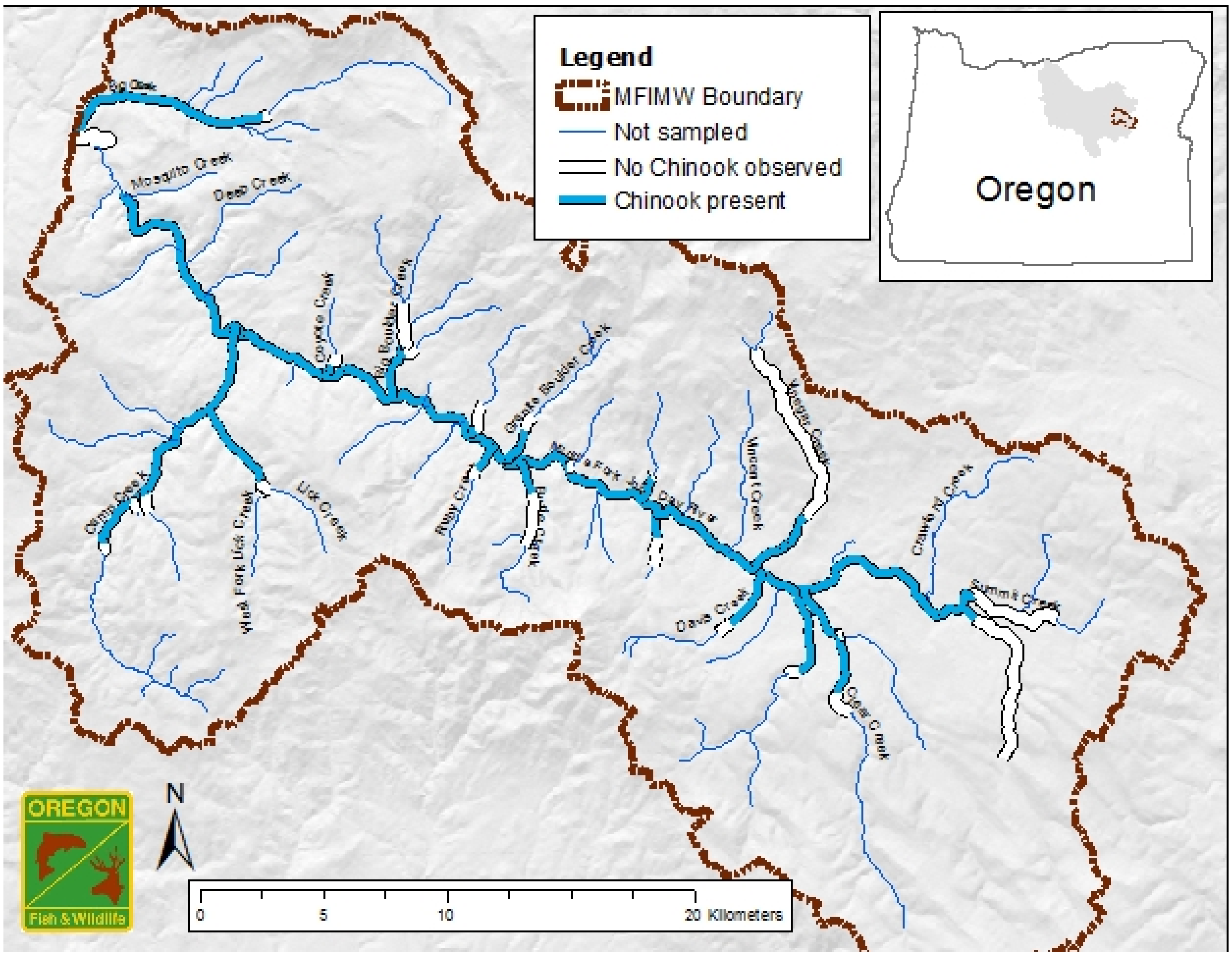

Figure 26. Map showing juvenile Chinook presence and absence based on observations from electrofishing and snorkeling surveys conducted during 2014–2016.

Table 13. Chinook and steelhead parr observations and density estimates (parr/km) during snorkel surveys at different reaches throughout the mainstem MFJDR. Surveys were completed during July and August each year in the mainstem MFJDR. Distance refers to the length of stream snorkeled in each reach.

Reach	2014			2015			2016		
	Distance (km)	Chinook Density	Steelhead Density	Distance (km)	Chinook Density	Steelhead Density	Distance (km)	Chinook Density	Steelhead Density
Mosquito to Deep	6.6	1.4	15.0	6.6	1.5	20.9	5.6	56.4	27.9
Camp to Galena	4.6	0.2	6.1	4.6	14.8	56.1	4.6	18.7	56.1
Balance to Big Boulder	4.8	49.8	85.2	3.9	1.8	5.4	3.3	86.7	101.8
Ruby to Butte	1.3	163.8	63.1	-	-	-	-	-	-
Butte to Windlass	2.4	126.7	65.4	-	-	-	-	-	-
Windlass to Little Butte	1.7	408.8	190.6	-	-	-	-	-	-
L. Butte to L. Boulder	0.9	437.8	145.6	-	-	-	-	-	-

Table 14. Seven day average stream temperatures (°C) calculated from temperature loggers within snorkeled reaches prior to survey date each year. Dashes indicate locations with no available temperature information, ns refers to reaches not sampled.

Reach	Year		
	2014	2015	2016
Mosquito to Deep	21	21	20
Camp to Galena	-	-	-
Balance to Big Boulder	20	-	-
Ruby to Butte	15	ns	ns
Butte to Windlass	-	-	-
Windlass to Little Butte	18	ns	ns
L. Butte to L. Boulder	-	ns	ns

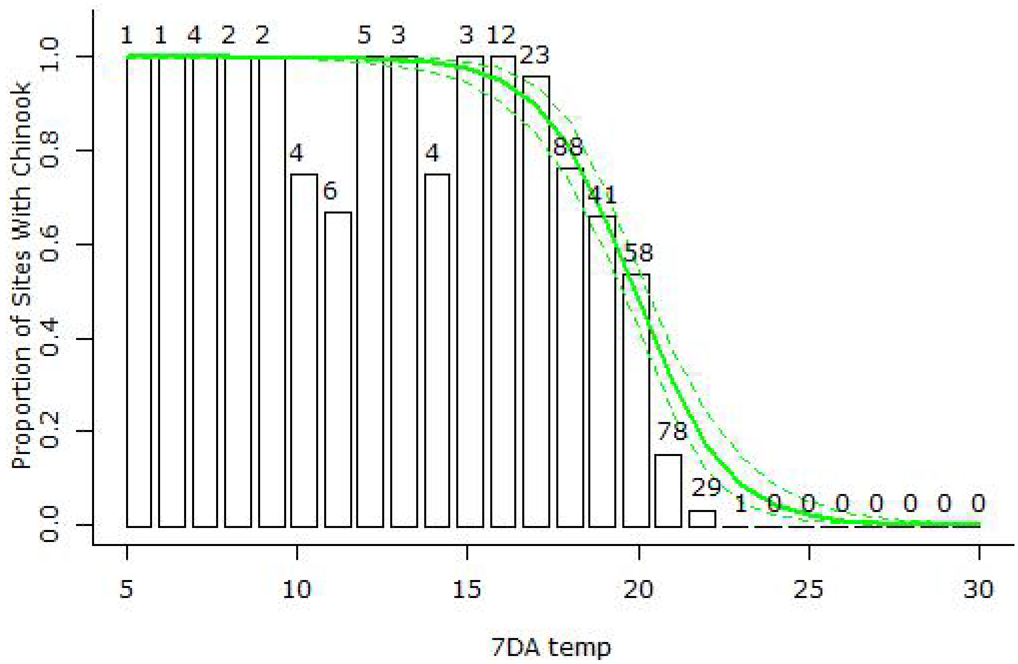

Figure 30. Logistic model results for Chinook presence or absence and seven day average water temperatures within one km of sampling locations at sites within the mainstem MFJDR from 2011 through 2015. Bars represent the proportion of sites from each 1°C bin with observed Chinook parr. Sample sizes for each bin are shown at the top of the bars. Model predictions for Chinook presence and ±95% confidence bounds are represented by the green line and dashed green lines respectively.

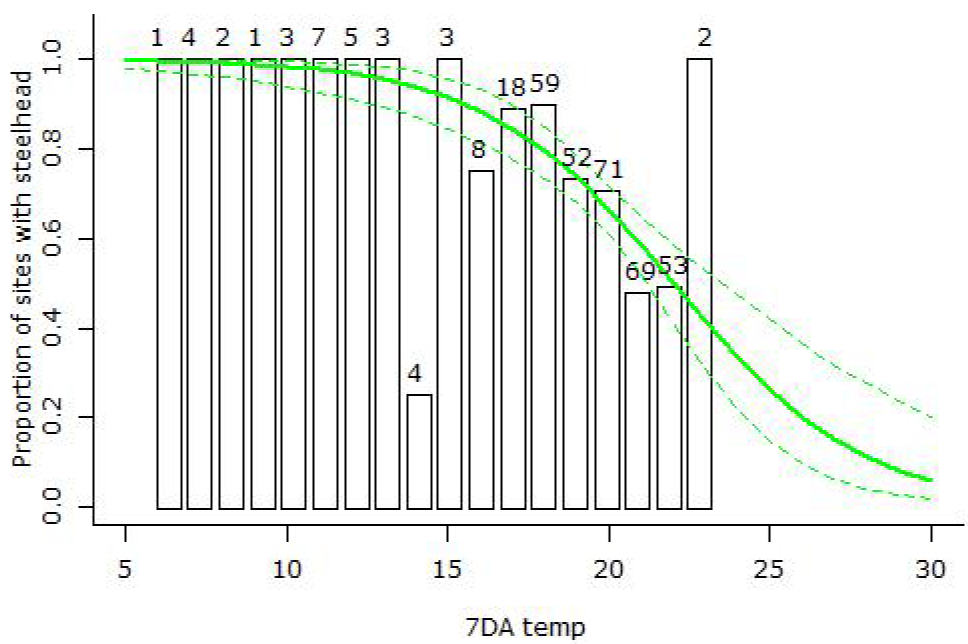

Figure 27. Logistic model results for steelhead presence or absence and seven day average water temperatures within one km of sampling locations at sites within the mainstem MFJDR from 2011 through 2015. Bars represent the proportion of sites from each1°C bin with observed steelhead parr. Sample sizes for each bin are shown at the top of the bars. Model predictions for steelhead presence and ±95% confidence bounds are represented by the green line and dashed green lines respectively.

Logistic model results for Chinook parr indicated a 50% occupancy of sites at 20°C and no occupancy above 24°C. Similarly, the model predicted a decline in steelhead presence with increasing 7DA stream temperatures. Fifty percent occupancy occurred at 22°C however, steelhead parr were observed in reaches of stream sampled with temperatures exceeding 23°C.

Bates Pond Passage

PIT tag detections

Detections of PIT tagged parr at the antennae array on Bridge Creek just upstream of Bates Pond indicate that at least some Chinook and steelhead parr are able to ascend the fish ladder and navigate above Bates Pond during certain times of the year. Sixteen steelhead parr tagged below the fish ladder have been documented above the ladder and pond since 2010 (Table 17). Only six Chinook parr tagged downstream of Bates Pond have been detected at the Upper antennae (Table 18).

Table 17. Location of parr and resident steelhead tagged by ODFW personnel and subsequent detections at the Bridge Creek PIT tag antennae from 2010 through August 28, 2014.

Tagging Location	Year Tagged	# Tagged	Lower Antennae					Upper Antennae				
			2010	2011	2012	2013	2014	2010	2011	2012	2013	2014
Lower Bridge Cr.	2010	85	13	4	1	0	0	0	0	0	1	0
	2011	45	-	0	0	0	0	-	0	0	0	0
	2012	52	-	-	8	0	0	-	-	2	0	0
	2013	44	-	-	-	15	0	-	-	-	1	1
	2014	39	-	-	-	-	2	-	-	-	-	0
Upper Bridge Cr.	2010	256	0	0	0	0	0	97	21	7	1	0
	2011	30	-	2	5	1	0	-	1	9	0	0
	2012	94	-	-	7	3	1	-	-	44	12	0
	2013	56	-	-	-	7	3	-	-	-	24	4
	2014	51	-	-	-	-	0	-	-	-	-	19
Outside Bridge Cr.	2010	2089	0	0	0	0	0	0	0	0	0	0
	2011	1995	-	2	5	1	0	-	0	6	1	0
	2012	3120	-	-	7	3	1	-	-	2	0	1
	2013	1509	-	-	-	7	3	-	-	-	1	0
	2014	774	-	-	-	-	0	-	-	-	-	0

Table 18. Location of spring Chinook salmon parr tagged by ODFW personnel and subsequent detections at the Bridge Creek PIT tag antennae from 2010 through August 28, 2014.

Tagging Location	Year Tagged	# Tagged	Lower Antennae					Upper Antennae				
			2010	2011	2012	2013	2014	2010	2011	2012	2013	2014
Lower Bridge Cr.	2010	50	6	2	0	0	0	0	0	0	0	0
	2011	3	-	3	0	0	0	-	0	0	0	0
	2012	35	-	-	6	0	0	-	-	2	0	0
	2013	11	-	-	-	3	0	-	-	-	1	0
	2014	5	-	-	-	-	2	-	-	-	-	0
Upper Bridge Cr.	2010	3	0	1	0	0	0	2	0	0	0	0
	2011	0	-	0	0	0	0	-	0	0	0	0
	2012	1	-	-	0	0	0	-	-	0	0	0
	2013	1	-	-	-	0	0	-	-	-	0	0
	2014	0	-	-	-	-	0	-	-	-	-	0
Outside Bridge Cr.	2010	1920	0	0	0	0	0	0	0	0	0	0
	2011	3651	-	1	1	0	0	-	0	3	0	0
	2012	4623	-	-	8	1	0	-	-	0	0	0
	2013	2200	-	-	-	10	0	-	-	-	0	0
	2014	504	-	-	-	-	0	-	-	-	-	0

Chinook Parr Distribution

Chinook distribution surveys in Bridge Creek above Bates Pond indicated the presence of juvenile Chinook in 2010, 2011, 2012, and 2014. Chinook parr were also present in nine of eleven snorkeled pools below Bates Pond in the Bridge Creek tail-water in 2012 and nine of the twelve pools surveyed during 2014.

No Chinook redds have been documented upstream of Bates Pond since surveys began in 2011, however, one adult female Chinook carcass with a partially constructed redd was found in 2012 approximately 30m upstream of the pond. This carcass was identified as a pre spawn mortality that had not successfully spawned. Steelhead redds have been observed upstream of Bates Pond each year that it has been surveyed since 2001.

Discussion

Our survival and abundance models, based on empirical evidence from monitoring juvenile salmonids, indicate that stream flow and water temperature are limiting freshwater production of both steelhead and Chinook salmon in the MFIMW. The limited summer distribution of juveniles in the mainstem MFJDR, and the movement of juveniles into cool-water tributaries also demonstrates that water temperature is limiting the availability of productive habitat for rearing both Chinook and steelhead. Further, juvenile Chinook are more vulnerable to temperature limitation in the MFIMW due to their lower tolerance of water temperatures above 20°C.

We found significant relationships between Chinook parr summer survival and both parental density (negative) and summer discharge (positive). The strength of the discharge effect was approximately double that of the density effect, however. This indicates that habitat actions that increase summer discharge will increase Chinook survival, and hence population productivity. Although we did not see an influence of habitat actions on these effects at the population scale during our monitoring period, we did establish a flow-survival relationship that can be used to estimate long-term effects as the habitat projects continue to mature. Collection of these data was done over a relatively short period. Our understanding of restoration action effectiveness could be improved with increased pre-treatment monitoring.

Stream discharge likely influences almost all aspects of a stream dwelling fish’s life. Higher flow volume increases velocity, depth, and width of a stream so it can be assumed that higher flow volume translates into more space for fish to occupy. More flow increases cover, invertebrate drift, facilitates movement between stream reaches, and decrease competition for all of these resources between individuals. During our monitoring, flow conditions changed more frequently over the winter interval than the summer season but low summer flow volume (lowest stream discharge over the course of the season) was the most likely indicator of survival. Average flows over a three month period could potentially obscure an extremely harsh summer if stream levels rise in the fall or flash early in the season. It may therefore, be necessary to fit a model with minimum flow to help explain mortality associated with seasonal extremes.

Steelhead parr density within closed capture sties showed no significant increase after the 2011 treatment in Camp Creek but when compared to Murderers Creek over the same time period, Camp Creek showed an increase in mean density of 31% while Murderers Creek showed a 35% decrease in parr density (Figure 15). Although these results are encouraging, most of this apparent increase in parr density in Camp Creek was caused by extremely high densities during the fall of 2012 following the best water year and highest adult escapement from 2008 through 2017 (Figure 14).

Survival

Model results suggest that steelhead parr survival declined after the 2011 treatment in Camp Creek. This decline was not apparent in either of the other two streams sampled during the same time period in the models we tested. Post treatment monitoring coincided with increased adult escapement and drought which likely effected the Camp Creek watershed to a greater degree than the other streams included in this analysis (Figure 8). Density dependent effects are somewhat confounded with drought in that extremes of both occurred over the same time period that we monitored. A stage logger was placed in Camp Creek to monitor flow for most of 2016, these data are correlated with Middle Fork Discharge and may potentially be useful to back-calculate Camp Creek flows over this monitoring period to help explain the decreased survival estimates post 2011.

It seems unlikely that restoration activities in this watershed was the cause of the observed decline in survival but rather a decline in resources caused by drought. Monitoring these Camp Creek sites during better flow conditions could help explain the decline in survival that we observed from pre and post treatment. Ideally we would have been able to monitor the same site for at least a few years before and after treatments occurred. Treatments of different reaches of stream near our monitoring sites were ongoing as we conducted this study and may have effected parr movement and distribution. Although these sites are not set up in an ideal fashion to monitor reach scale changes we feel they are representative of the watershed and can provide an indication of what is limiting production at the watershed scale. This is especially true when considering the hypothesis that parr select habitats that best suit their needs at different seasons. For instance if parr are living in a warm stream during the summer with temperatures approaching their maximum metabolic rate they will likely select reaches of stream that have abundant macroinvertebrate drift that can meet their high metabolic demand (faster water near riffles). If they are living in a cooler environment (fall and winter) they may select reaches of stream based on abundance of slower velocity water and cover. As stream flow declines, velocities and stream power decline and the habitat in each reach of stream changes. As parr grow they reach the carrying capacity of their habitat at some point and some are forced to explore new reaches of stream. Assuming movement is occurring prior to a final smolt migration a multistate model allows us to evaluate survival throughout the IMW by including downstream detections. To better understand when parr leave particular reaches it could be useful to include additional sampling occasions throughout the winter but this may not be possible in the Middle Fork Basin due to snow and ice throughout most of the winter season.

Growth

Chinook parr growth varied by stream and was influenced by variation in streamflow, temperature, and to a lesser extent density dependence. However, Chinook parr growth did not vary significantly between treatment and control reaches that we monitored. To gain a greater understanding of the reach level effects on parr growth, it may be useful to collect information on food availability, and parr densities throughout the study area. It would be beneficial to study which habitat types produce the most food and support the most parr during the summer months. Future parr monitoring efforts may answer some of these questions by monitoring specific habitat types and incorporating a bioenergetics modeling approach similar to the bioenergetics component of the McCugh et al. (2017) steelhead life-cycle model.

Steelhead growth rates varied with temperature among all sites and showed a slight decrease when exposed to temperatures above 18°C suggesting temperatures exceeding this level are stressful. Streams with 7DAM water temperatures exceeding 18°C do not meet Oregon Department of Environmental Quality standards for rearing salmonids. Camp Creek, Coyote Creek, Murderers Creek, and the MFJDR are listed as impaired by temperature on the ODEQ 303d list.

Chinook Distribution

Model results from Chinook parr distribution and 7DA temperatures in the mainstem MFJDR suggest that Chinook parr do not frequently occupy reaches of stream where 7DA water temperatures exceed 20°C. Chinook parr where only observed at 20% of sites with 7DA temperatures exceeding 20^{0}C. When reach specific water temperatures were less than 18°C Chinook parr occupied more than 90% of the locations we sampled. Decreasing summer water temperatures throughout the MFIMW would increase the downstream distribution of juvenile Chinook in the MFJDR during the warmest times of the year thereby improving watershed productivity.

Moreover, the volume and productivity of this downstream habitat would likely have significant influence on overall productivity if it was habitable during all times of the growing season.

Bates Pond Passage

The presence of juvenile Chinook upstream of Bates Pond without any confirmed adult spawning activity in that reach the previous year suggests all observed Chinook Parr in upper Bridge Creek originated downstream of the pond. We have never confirmed the presence of a live adult Chinook within the survey reach above Bates Pond during spawning season. Adult Chinook have been observed in Bates Pond in late spring and summer. While these observations indicate that both adult and juvenile salmonids are able to use the fish ladder to migrate into Upper Bridge Creek, these data are insufficient to determine the efficiency of the structure and habitability of the pond to allow fish passage. In addition, we have insufficient data to evaluate the efficiency of downstream passage of these fish. The ladder and pond do appear to allow more efficient passage of steelhead versus Chinook salmon. This is likely influenced by the migration timing of these two species. Adult steelhead pass the dam in early spring during high flows and cold water temperatures while adult salmon do not arrive in Bridge Creek likely until late May and June when flows begin to recede. Juvenile passage is more of a concern because these fish are attempting to access cold water refugia during summer months when temperatures are approaching their thermal maximum both in the MFJDR and Bridge Creek below Bates Pond. Due to high surface water temperatures in Bates Pond during summer months, lower Bridge Creek is warmer than the MFJDR during most of the summer. During our distribution surveys 7 August 2012 we recorded a water temperature of 14°C in the mainstem Middle Fork at the mouth of Bridge Creek and a temperature of 20°C in Lower Bridge Creek. This likely offers little incentive for juvenile and adult fish seeking thermal refugia to move into Bridge Creek during the heat of the summer and ascend the fish ladder. If the thermal condition of Bridge Creek through and below Bates Pond was improved, more steelhead and salmon would utilize Bridge Creek as cool water refugia during periods of heat stress that limit salmonid productivity in the MFJDR.

Lessons Learned

Future Restoration

Distribution of juvenile salmonids, especially Chinook continues to be limited by summer stream temperatures. Future work should focus on improving thermal conditions throughout the watershed to increase salmonid distribution downstream. This could be achieved by restoring natural riparian function where it is impaired with the goal of creating a self-maintaining riparian area which produces shade and cover for salmonids. In tributaries, this may require exclusion of cattle/ungulates via riparian fencing in strategic reaches to reduce grazing/browsing pressure with the hopes of encouraging deciduous tree and shrub species to grow.

Fencing projects should focus on sections of stream with limited canopy cover within grazing allotments or areas where wild ungulate presence has been shown to limit riparian vegetation recruitment. Fencing projects should be set up with contracts/plans that include a regular maintenance schedule until riparian areas have been restored. These projects should consider fencing short reaches of stream to allow livestock and wildlife passage across the landscape and access to water.

Monitoring

Short term reach scale monitoring will be most effective if stream study treatment reaches are paired with control reaches that match specific environmental and physical conditions. We suggest pairing future study reaches based on answering specific questions about fish and habitat relationships rather than a simple control and treatment classification. Control and treatment study designs work well when monitoring can be achieved before and after treatment reaches have been treated but this often presents challenges for researchers as much of the restoration work is done opportunistically with available funding making it difficult to plan effective pre-treatment reach scale monitoring. Future research should focus on answering specific questions about reach productivity based on habitat characteristics assumed to be limiting factors. More effort should be placed on using habitat information with fish monitoring. This information could be used to answer specific questions about survival, growth, and abundance in a paired experimental fashion using some of the modeling structure we have developed thus far.

Juvenile salmonids rearing in the Middle Fork watershed are effected by climactic, density dependent, and seasonal fluctuations in temperature and streamflow at the watershed scale. Population trends and freshwater productivity are driven by these environmental factors. Action effectiveness monitoring should consider pairing productivity before and after treatments based on these watershed wide conditions and point in time relative to restoration activities rather than a simple before and after approach. Restoration actions aimed at improving watershed function may take decades to mature or improve conditions for salmonids but could be evaluated using habitat metrics or riparian condition to determine effectiveness in a shorter time frame in the MFIMW.

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Appendix C – Stream Habitat Condition for Middle Fork John Day River and Camp Creek Watershed

K. Fetcho, Oregon Watershed Enhancement Board (OWEB) Salem, OR
E. Archer and J. V. Ojala, PacFish/InFish Biological Opinion (PIBO) Monitoring Program USDA Forest Service, Logan, UT

Abstract

Stream habitat in the Middle Fork John Day River (MFJDR) and its tributaries has been impacted by various historic land management practices. Steelhead and spring Chinook habitat is targeted for restoration efforts in the John Day Basin. Insufficient habitat quantity and diversity have been identified as key limiting factors affecting their recovery (CBMRCD 2005 and Carmichael and Taylor 2010). To detect changes in stream habitat at the watershed scale, the MFIMW commissioned the USFS to establish PIBO sampling sites. We estimated trends by measuring changes in individual stream habitat metrics in Camp Creek and the MFJDR to investigate the effectiveness of restoration efforts implemented throughout the MFIMW. In addition, we used a "habitat index approach" to compare individual aspects of habitat conditions in the Camp Creek watershed to reference sites established in the Blue Mountains Ecoregion and in the Upper Columbia River Basin. Results indicate that most individual aspects of habitat condition in the MFIMW are stable or improving. Overall habitat index, large woody debris frequency, and percent undercut banks in both Camp Creek and the MFJDR showed statistically significant improving trends. The only measure in which we observed an undesired trend in both geographic areas was percent fines in pools, which increased in both Camp Creek and MFJDR. Results show that habitat in the Camp Creek watershed is in poorer condition than reference sites in the Blue Mountains and the Upper Columbia River Basin. The improving trend in the overall habitat index and most individual habitat attributes shows that restoration and current management efforts have a measureable positive impact at the watershed and sub watershed scale. The current status of the Camp Creek habitat condition, while improving, highlights the need for additional restoration actions. Long-term monitoring should continue to track how past and future restoration actions improve habitat as riparian vegetation is established and floodplain processes and functions are restored.

Introduction

Background

The Pacific Anadromous Fish Strategy (PACFISH) and Inland Fish Strategy (INFISH) Biological Opinion (PIBO) for bull trout and steelhead was developed in 1998. PIBO includes a monitoring program for aquatic and riparian resources, for specific habitat attributes that define aquatic conditions, and for their relationship with listed fish species. This list has been translated into measurable criteria and specific sampling protocols for stream channel attributes and vegetation parameters. The UMFWG establish PIBO sampling sites within the MFIMW to understand how habitat conditions change at a watershed scale over time. PIBO was selected because the approach offers a wide variety of physical attributes to characterize stream habitat at each site and because the results can be compiled into a single index for comparison among sites. PIBO measures habitat attributes that influence the production or survival of native salmonids, are sensitive to land-use changes, and can be measured consistently.

Goals and Objectives

The goal of this study is to detect stream and riparian habitat changes at a watershed scale using a monitoring protocol at numerous sites in the Camp Creek watershed and the MFJDR. This monitoring effort complements other site specific habitat monitoring efforts applied in the MFIMW that focus on geomorphologic responses to specific restoration actions at a smaller scale. This study also assessed the status of stream habitat conditions of the combined sampling sites in the Camp Creek watershed by comparing the habitat characteristics to reference streams over the entire PIBO study area and the ecoregion. This monitoring effort also documents changes in habitat conditions (e.g. “trend”) over a given time period in both the Camp Creek watershed and the MFJDR.

Hypotheses

We expect this analysis to show that the aquatic habitat attributes in the MFJDR and Camp Creek watershed improve over time after restoration actions were implemented. In addition, we expect that the status of Camp Creek watershed habitat attributes will more closely resemble reference site conditions after restoration actions were implemented. If stream habitat condition is improving, we would expect a positive trend in the overall habitat index, residual pool depth, pool percent, percent undercut banks, bank stability, large wood frequency, and median substrate size. In addition, we would expect a negative trend in bank angle and percent fines in pools.

Site Selection

The UMFWG established 10 PIBO monitoring sites (Figure 1) in the Camp Creek subwatershed that were surveyed in 2008 and 2014. The PIBO monitoring sites are being utilized to evaluate the watershed restoration actions designed to improve aquatic habitat including the removal of stream spanning log weirs installed in the Camp Creek watershed in the 1980s. The 5 control sites never had log weirs while the 5 experimental sites had log weirs removed. It is important to note that in addition to the log weir removals a variety of restoration actions, such as riparian plantings and instream habitat improvements, were implemented across the Camp Creek watershed. Therefore, when combined, these 10 sites collectively describe the habitat conditions status and trends for this watershed based on all restoration actions that have occurred to date.

The UMFWG also established 15 PIBO sites (Figure 1) in the MFJDR that were surveyed in 2009 and 2014. These sites were selected to help evaluate trends in important aquatic variables throughout the study area over time. Overall, this distribution of sampling sites throughout the watershed helps evaluate habitat condition in response to individual restoration actions and other contributing factors such as: forest management, land use changes, high water events, wildfires, etc.

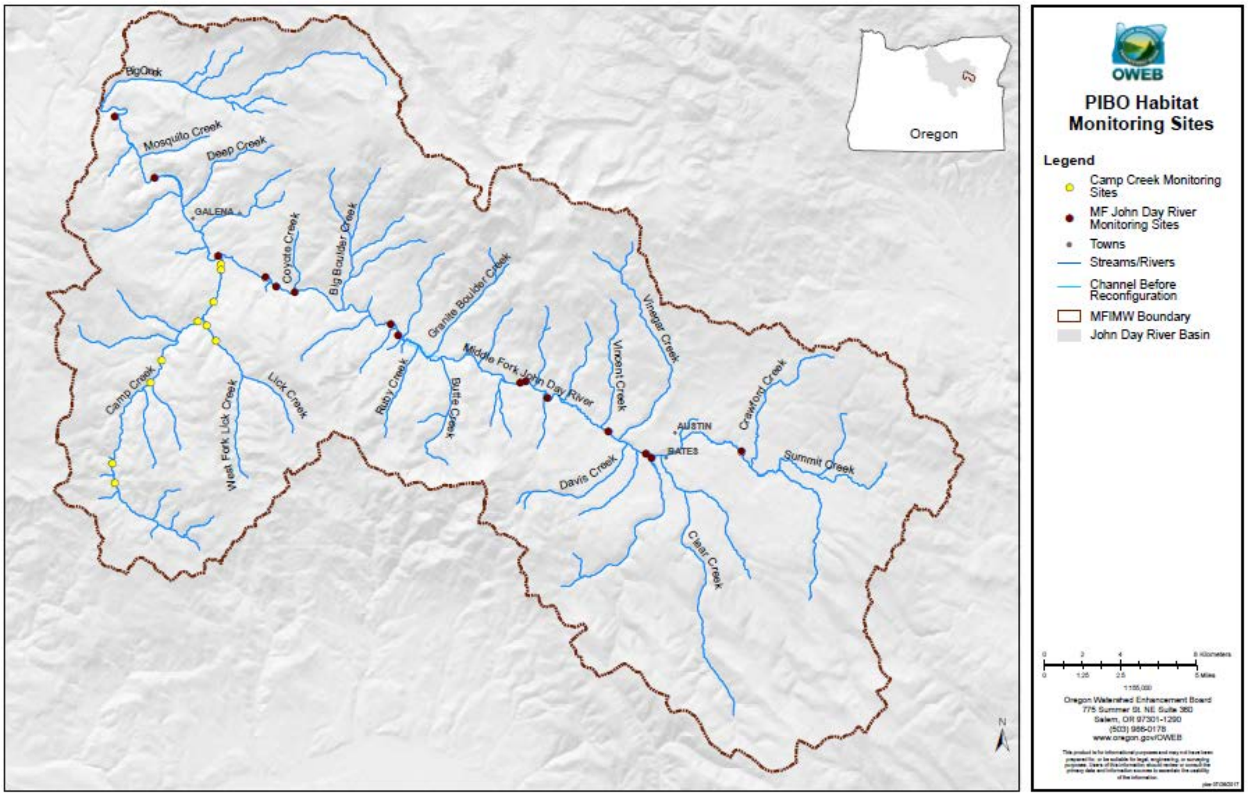

Figure 1. PIBO sampling sites in the MFIMW.

Methods

Status and Trend

PIBO's approach is to compare the status of stream habitat conditions at sites in 'managed' watersheds (those that have experienced disturbance from various management actions) to habitat conditions at sites within 'reference' watersheds, which are used as a benchmark of expected condition. All streams experience natural disturbances. In an ideal study design, a range of stream habitat conditions at managed sites would be compared with expected conditions for streams that experienced only natural disturbance. To evaluate status, we created an index of habitat condition that combines several stream habitat attributes and helps account for natural variability among sites (Al-Chokhachy et al. 2010). While an index is good for determining status, it may be less sensitive when detecting trend in habitat condition over time because it averages conditions of several attributes that may be more individually responsive. Therefore we estimate trends by measuring changes in individual stream habitat metrics, such as bank stability or large wood frequency, at the sites over the duration of PIBO sampling (2008/9-2014).

Reach sampling

PIBO began collecting physical stream habitat and macroinvertebrate data at the reach scale (160-400 meter stream lengths) within the interior Columbia River and Upper Missouri River basin in 2001. Approximately 300 sub-watersheds (6th field HUCs) are randomly selected each year for sampling. Over a five year period, 1,300 sub-watersheds are sampled in the Columbia River basin and 250 sub watersheds are sampled in the Missouri basin. These sub-watersheds have been resampled on a five year rotation, and the data are used to assess status and trend of aquatic and riparian conditions. In addition, the USFS Forestry Sciences Lab samples additional PIBO sites upon request. The sites sampled for the MFIMW were not selected as part of this random draw described above, and were identified by request to support the MFIMW.

Sub-watershed and Reach Types

The sub-watersheds that are randomly selected by PIBO to be sampled every year are divided into two groups based on management history (i.e., livestock grazing, mining, roads): "Reference" (minimally managed) or "Managed". Reference sites are primarily located in wilderness areas or in sub-watersheds with no obvious mining, no recent grazing (within 30 years), minimal timber harvest (< 5%) and minimal road density (<0.5 km/km^2). There are 217 reference sites within the larger PIBO study area and 19 reference sites within the Blue Mountains ecoregion for comparison with Camp Creek sites. Because the MFJDR does not offer adequate references size due to its large size, only the Camp Creek sites were compared to the reference sites to determine status.

Field Data Collected for Status and Trend

Physical Habitat Attributes

To estimate status of physical stream habitats at each site, we focus on six stream channel attributes that (1) influence the production or survival of native salmonids; (2) are sensitive to land-use changes; and (3) can be measured consistently by observers (Table 1). For a complete description of these variables and field methods used, see Kershner et al. (2004) and Archer et al. (2013).

Biological Attributes

To evaluate a biological component of habitat status, we sample macroinvertebrates using the protocol recommended by the Center for Monitoring and Assessment of Freshwater Ecosystems, Utah State University (Hawkins et al. 2000). Macroinvertebrates are sampled from 8 fast-water habitats per site and combined into a composite sample. Macroinvertebrate taxa are identified by the BLM/USU National Aquatic Monitoring Center in Logan, UT.

Attributes Used for Trend

We estimate trend using the same six physical stream habitat attributes and one biological attribute (macroinvertebrate O/E) used for status, plus two additional metrics: bank stability and percent undercut banks (Table 1).

Table 1. Stream habitat attributes measured by PIBO.

Stream Habitat Attributes	Status	Trend
Average Bank Angle	*	*
D50 (medium substrate particle size	*	*
Percent fine sediment (<6mm diameter, in pool tails)	*	*
Large Wood frequency (pieces/km)	*	*
Residual pool depth (m)	*	*
Percent pool habitat	*	*
Bank stability (% bank covered with plants or rock)		*
Percent of bank with undercuts (bank angle <90°)		*
Macroinvertebrate taxa (Observed/Expected)	*	*

Calculating Physical Habitat Index Scores to Assess Status

To evaluate the status of stream habitat conditions at a given site, we first developed an index score for each physical habitat attribute. We used multiple linear regressions to explain inherent differences among sites. To account for local differences in stream type and geographic location we included landscape 'predictor' variables, such as average precipitation, percent forested and slope of the valley (Table 2), as well as some measures of stream power (reach gradient, and catchment area). These variables were used as covariates in the regression models. We selected the best multiple regression model to fit each attribute using data only from the reference sub-watersheds (n = 217; 10% of reference were set aside to verify model performance) to provide 'expected' stream habitat conditions in the absence of land management activities (Al-Chokhachy 2010).

Table 2. Landscape predictor variables used in model development.

Catchment area (km^2)
Average precipitation (m)
Slope of valley along reach (%)
Percent forested along reach (%)
Drainage density in catchment (km/km^2)
Reach gradient (%)
Elevation (m)
Dominant geology type (categorical)

We then compared observed conditions to what would be expected after controlling for local and landscape characteristics. We created an index for each stream habitat attribute by re-scaling the regression residuals from 0-10, using the 5^{th} and 95^{th} percentiles of the residuals at reference reaches as floor (index score = 0) and ceiling (index score =10) values. This process was repeated for each physical stream habitat attribute used to estimate status in Table 1. A site scored high (closer to 10) if the measure of observed habitat condition was better than expected and low if it was lower than expected (closer to 0). The distribution of index scores for a particular area represents the scatter around the line. Sites with sub-watershed areas less than $3km^2$ or greater than $300km^2$ were excluded from the analysis because they were outside the range of conditions present at reference sites.

For reference sites, residuals are considered to represent natural variation due to natural disturbances, such as fire, beetle kill, climate, or variance unexplained by our models. For managed sites, residuals are considered to represent a combination of natural factors, unexplained variation in the model, and a management effect. A significant difference between the reference prediction and the actual managed site index scores could be attributed to management.

To create an overall index of physical habitat condition for a site, we summed the individual attribute scores included in the index and then rescaled this sum from 0-100. For complete details of our statistical methods, see Al-Chokhachy et al. 2010.

Calculating a Macroinvertebrate Taxa Index O/E score to assess Status

To assess biological status at each site, we compared the macro-invertebrate taxa 'observed' at managed reaches (O) to the assemblages 'expected' to be found in relatively pristine reference reaches (E) based on a modeling exercise similar to that used for stream habitat (see Hawkins et al. 2000 for more specific details). The PIBO O/E model was developed using macro-invertebrate samples collected at 201 reference reaches between 2001 and 2005. Taxa were identified by the BLM/USU National Aquatic Monitoring Center. The O/E index score for each reach was estimated by dividing the number of expected taxa by the number of observed taxa. A monitored site with an O/E value of '1' indicates that all of the macroinvertebrate taxa expected at a reference site were found, while a value of '0' indicates that none of the taxa expected were found. Scores > 0.8 are generally considered similar to references reaches. Scores > 1 are either equivalent to what would be expected at a reference location or may have an enhanced insect community as a result of restoration action such as instream habitat improvements or riparian plantings.

Sites in the Camp Creek watershed were analyzed by comparing them to reference reaches within the Blue Mountains ecoregion (n= 17) and reference reaches throughout the PIBO study area (n = 215). The ecoregions included were the Blue Mountains, Idaho Batholith, Middle Rockies, Canadian Rockies, and Northern Rockies (for details, see Omernick 1987). If at least one managed site was located within a given ecoregion, then we included all reference sites from that ecoregion in our analysis. At least five managed reaches for a given area were necessary to run the analysis. In addition, at least five reference reaches had to be present in the area of interest in order to make a comparison at that scale.

We used a Welch’s t-test to determine if differences between index scores for each metric at managed and reference reaches were statistically significant; a p-value < 0.10 was considered significant. For the purposes of this study the “managed” sites refer to the special PIBO sites established in the MFIMW.

Estimating Trends in Stream Habitat Conditions

To estimate trends in stream habitat condition, we used actual measured values (and not index scores) for eight stream habitat attributes (see Table 1). We compared data collected at the first sampling visit with data from the last visit using the Wilcoxon signed rank summed test, a non-parametric statistical test that evaluates repeated measurements at the same site to determine if there has been a change in the metric value. A p-value < 0.10 indicates that the change is significant. Desirable changes could be either in a positive or negative direction (i.e., either increased bank stability or fewer fine sediments). The desired direction of change (positive or negative) for each habitat attribute is shown in Table 3 for the Camp Creek watershed and Table 4 for MFJDR sites. Summary tables also show the mean value for each attribute for the first and last sampling events, and the percent change in the metric over the evaluation period.

Results

Status of Physical Habitat Index Scores

Camp Creek Watershed Sites

The following figures compare the Camp Creek watershed habitat attributes to the ecoregion and entire study area reference sites. The first graph in the series below reports the median score and range of scores for all three datasets. The second graph in the series below reports the distribution of values for Camp Creek "managed" reaches (histogram) compared with expected values at reference reaches (the line graph). Close matches between histogram height and line generally indicate that conditions are similar when comparing managed and reference reaches.

Our findings indicate that aquatic habitat in the Camp Creek watershed is in worse condition when compared to both reference sites in the Blue Mountains ecoregion and across the entire PIBO study area (Figures 2-17). However, both median substrate size and percent fines in pools demonstrated better condition than at reference watersheds (Figures 9 and 11). In addition, the median of these two attributes scored higher than the Blue Mountains ecoregion and across the entire PIBO study area (Figures 8 and 10).

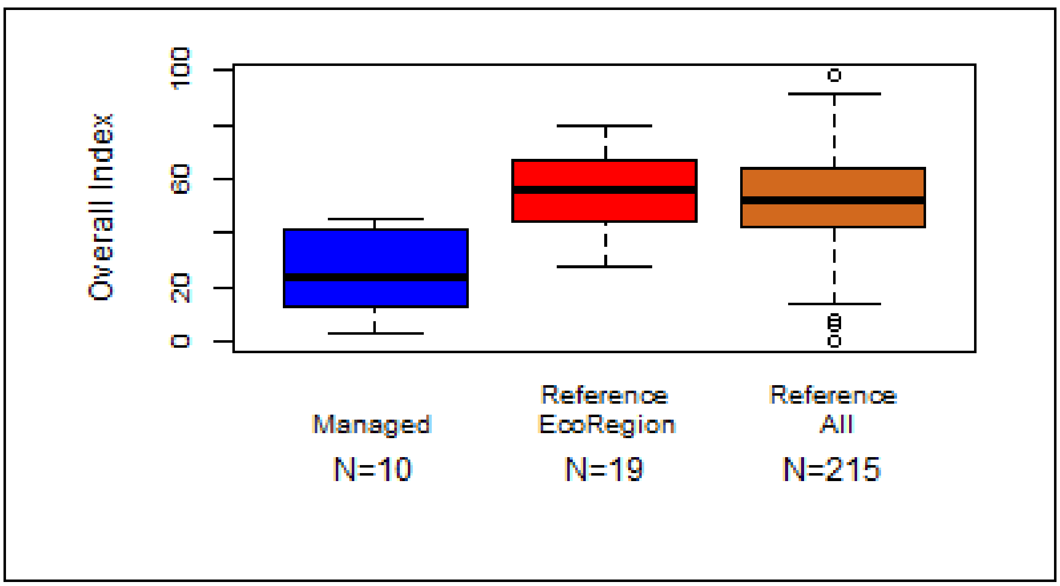

Figure 2. Overall index values across the Camp Creek watershed sites. Median and range of index values for managed sites, reference sites within the ecoregion, and reference sites for the entire PIBO study area.

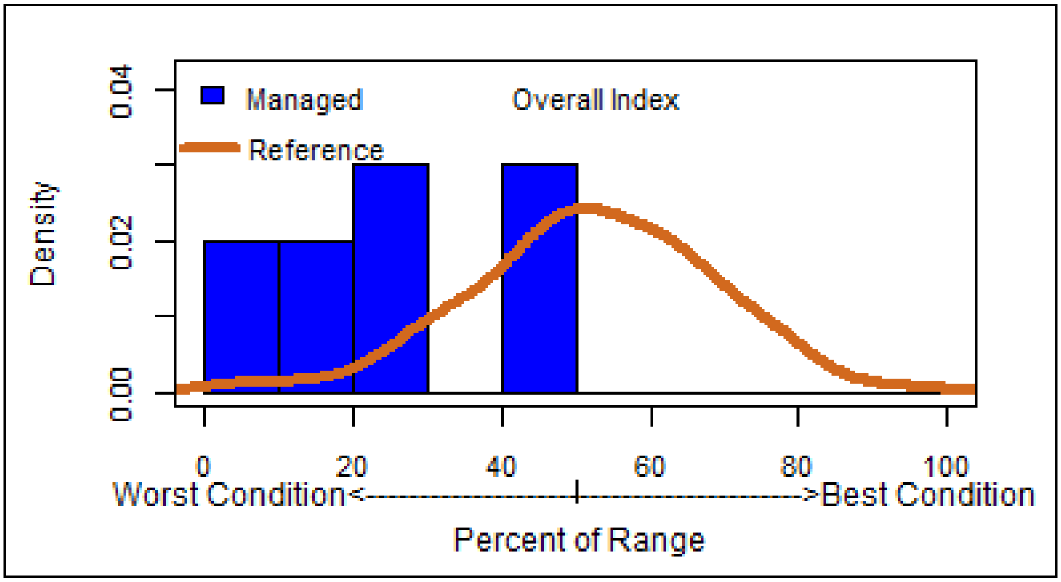

Figure 3. Overall Index values across the Camp Creek watershed sites. Distribution of index values for managed reaches (histogram) compared to expected values at reference reaches (the line graph).

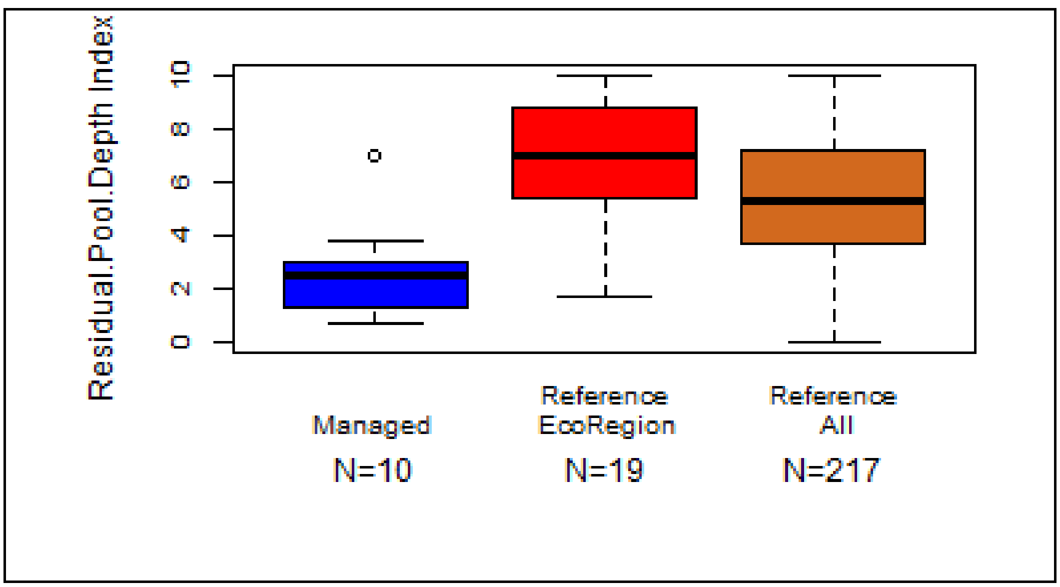

Figure 4. Residual Pool Depth Index values across the Camp Creek watershed sites. Median and range of index values for managed sites, reference sites within the ecoregion, and reference sites for the entire PIBO study area.

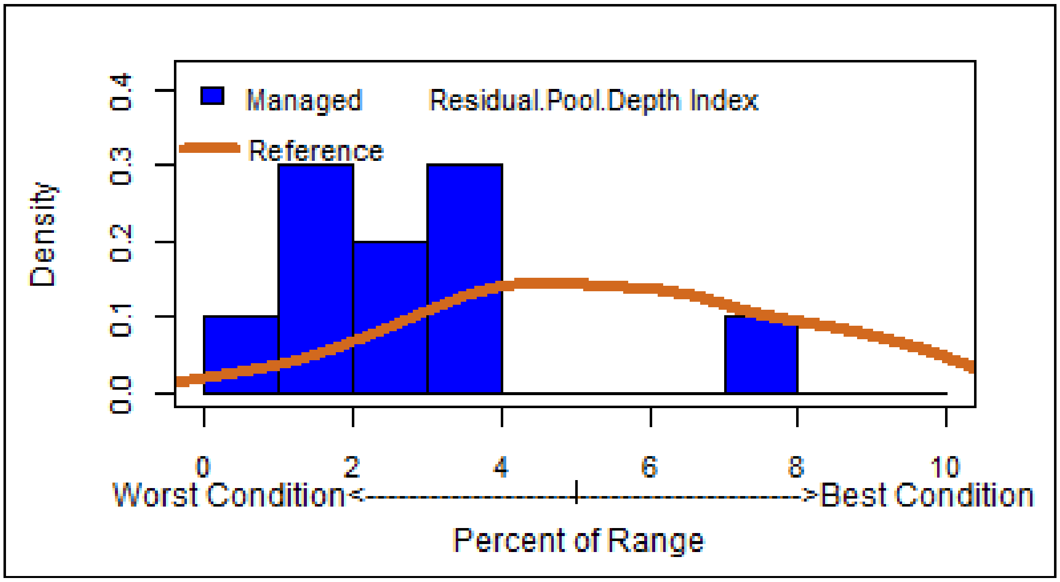

Figure 5. Residual Pool Depth Index values across the Camp Creek watershed sites. Distribution of index values for managed reaches (histogram) compared to expected values at reference reaches (the line graph).

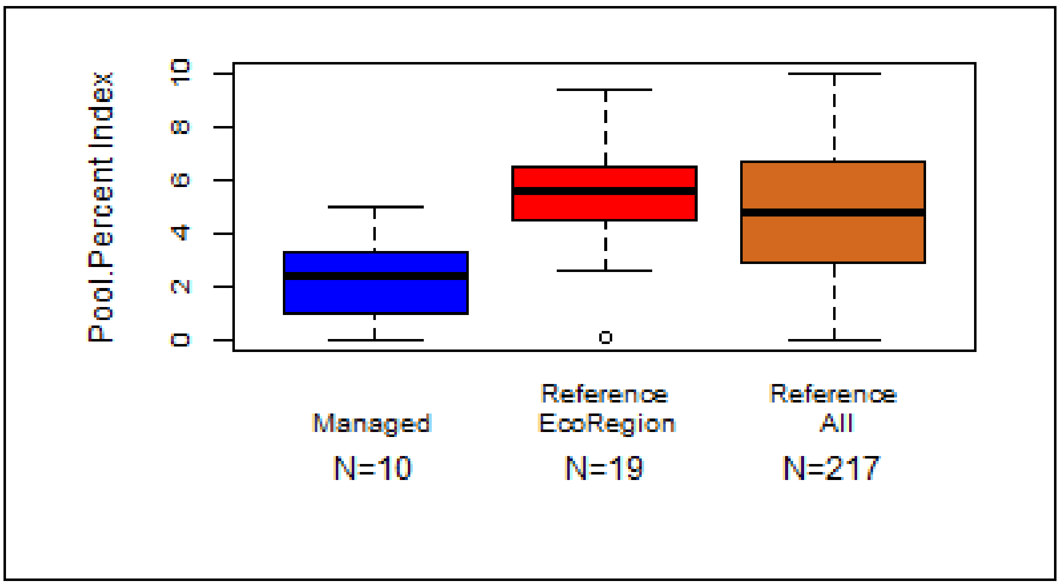

Figure 6. Pool Percent Index values across Camp Creek watershed sites. Median and range of index values for managed sites, reference sites within the ecoregion, and reference sites for the entire PIBO study area.

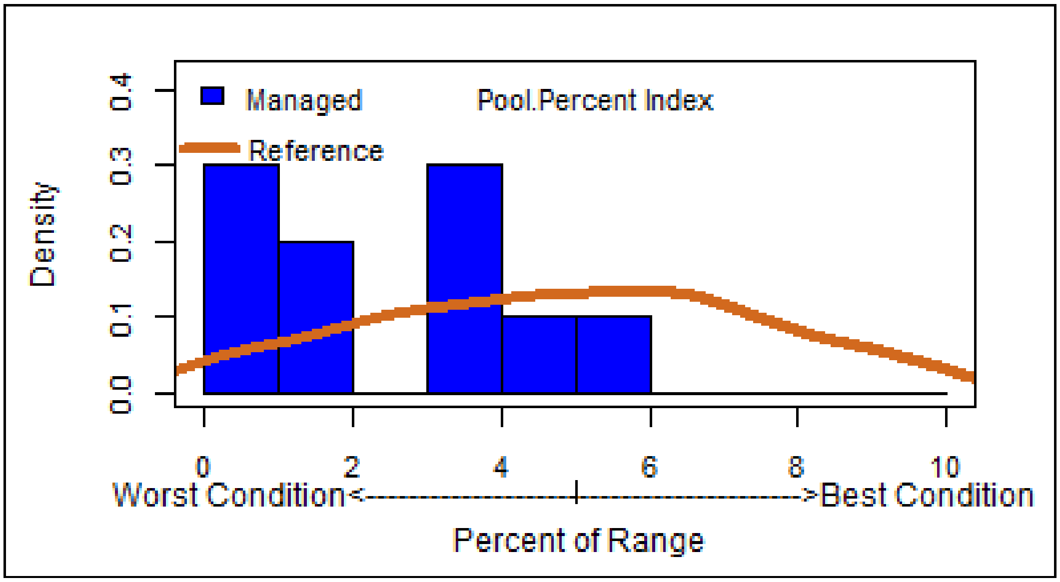

Figure 7. Pool Percent Index values across the Camp Creek watershed sites. Distribution of index values for managed reaches (histogram) compared to expected values at reference reaches (the line graph).

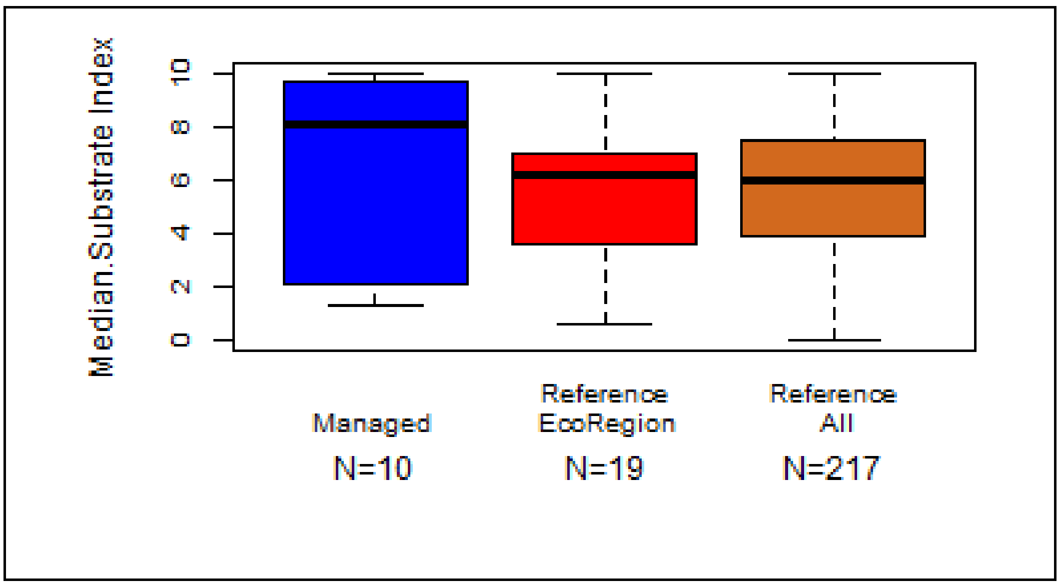

Figure 8. Median substrate index values across the Camp Creek watershed sites. Median and range of index values for managed sites, reference sites within the ecoregion, and reference sites for the entire PIBO study area.

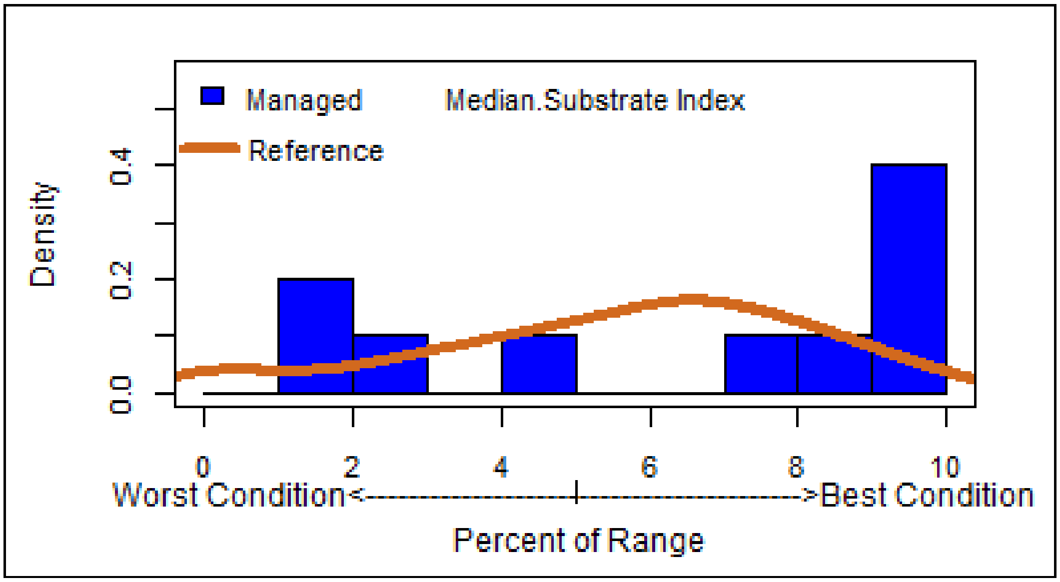

Figure 9. Median substrate index values across the Camp Creek watershed sites. Distribution of index values for managed reaches (histogram) compared to expected values at reference reaches (the line graph).

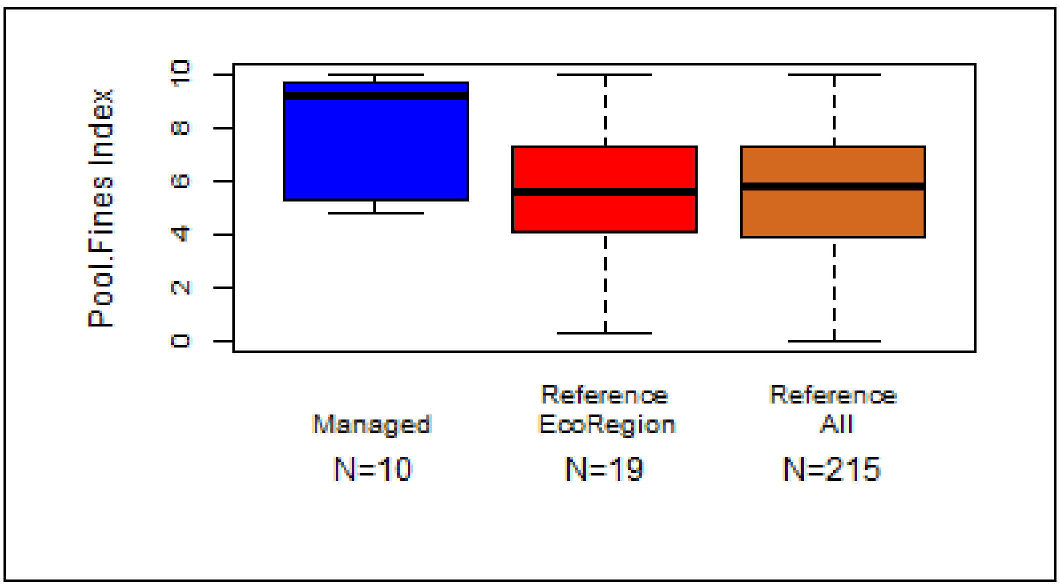

Figure 10. Pool Fines < 6 mm Index values across the Camp Creek watershed sites. Median and range of index values for managed sites, reference sites within the ecoregion, and reference sites for the entire PIBO study area.

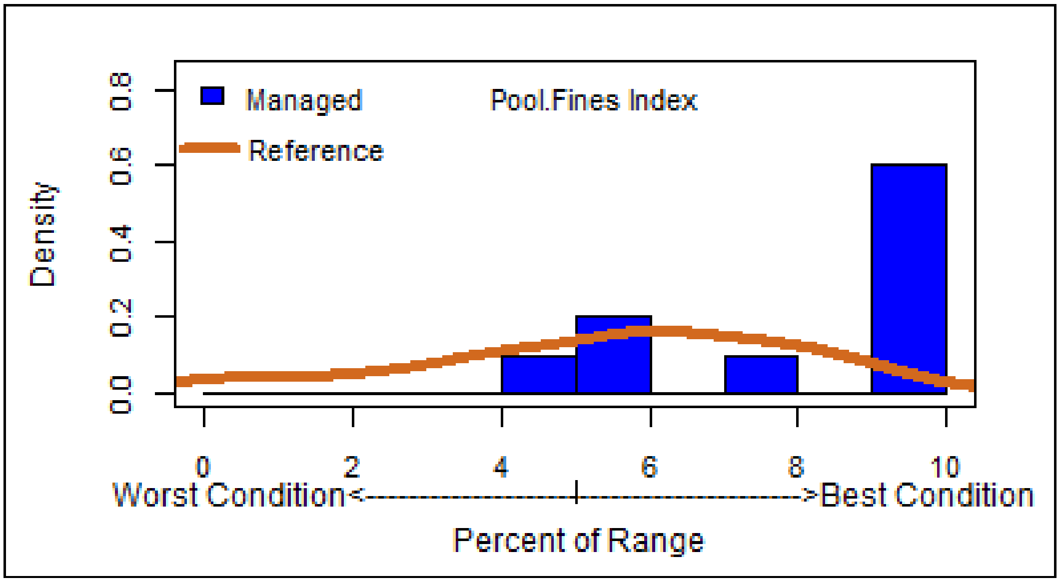

Figure 11. Pool Fines < 6 mm Index values across the Camp Creek watershed sites. Distribution of index values for managed reaches (histogram) compared to expected values at reference reaches (the line graph).

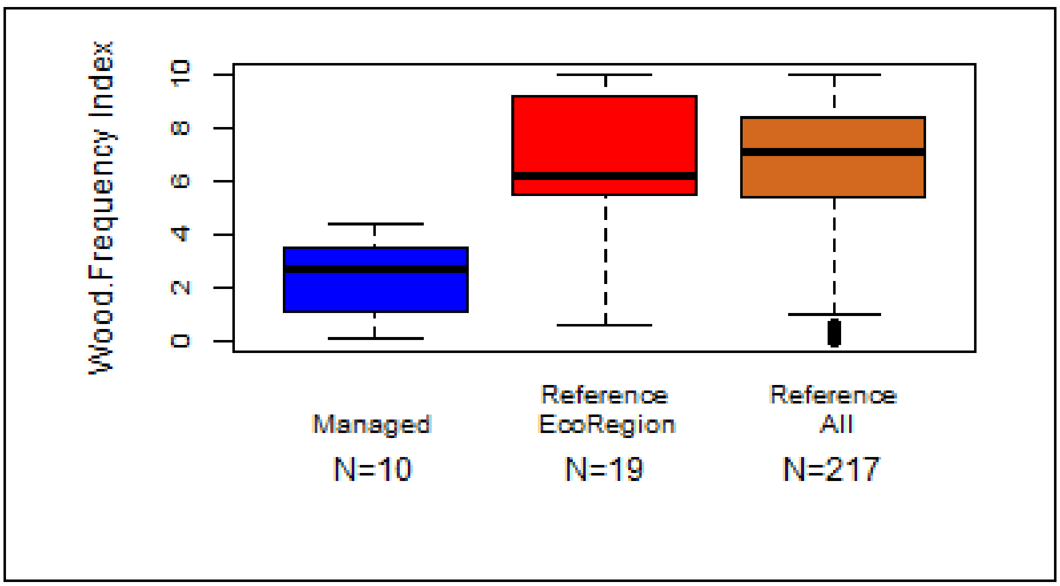

Figure 12. Wood Frequency Index values across the Camp Creek watershed sites. Median and range of index values for managed sites, reference sites within the ecoregion, and reference sites for the entire PIBO study area.

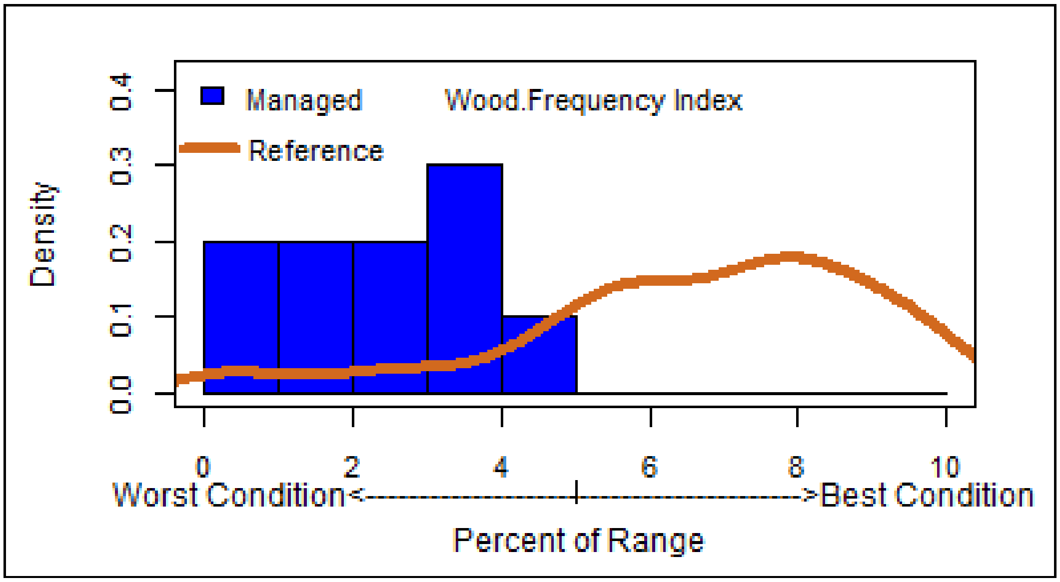

Figure 13. Wood Frequency Index values across the Camp Creek watershed sites. Distribution of index values for managed reaches (histogram) compared to expected values at reference reaches (the line graph).

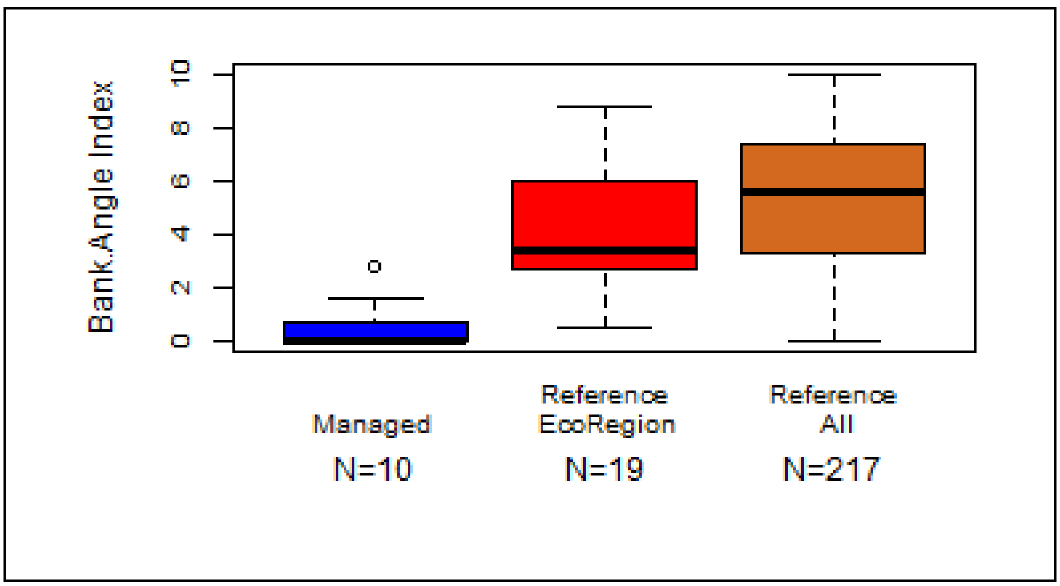

Figure 14. Bank Angle Index values across the Camp Creek sites. Median and range of index values for managed sites, reference sites within the ecoregion, and reference sites for the entire PIBO study area.

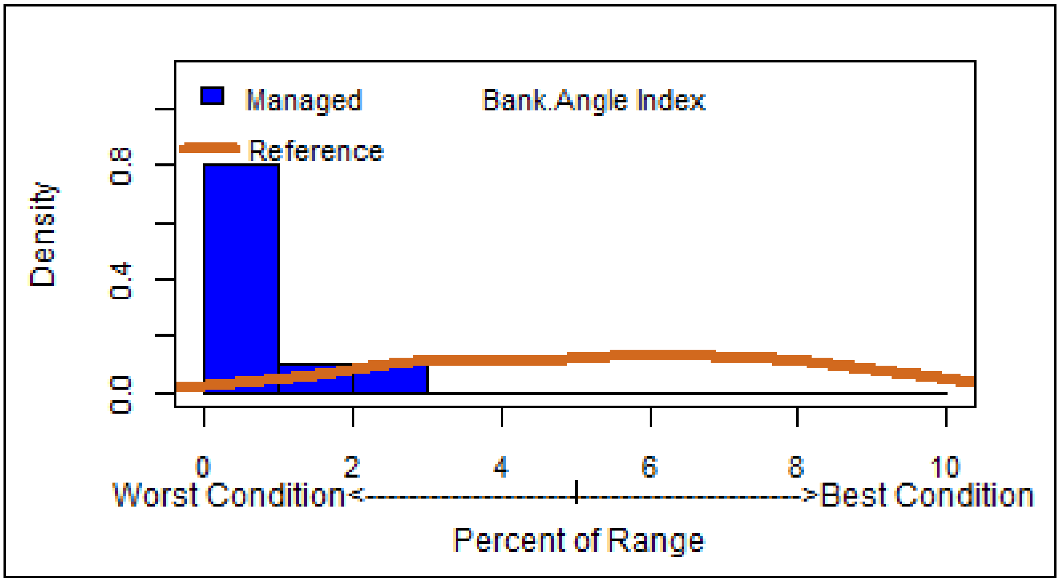

Figure 15. Bank Angle Index values across the Camp Creek sites. Distribution of index values for managed reaches (histogram) compared to expected values at reference reaches (the line graph).

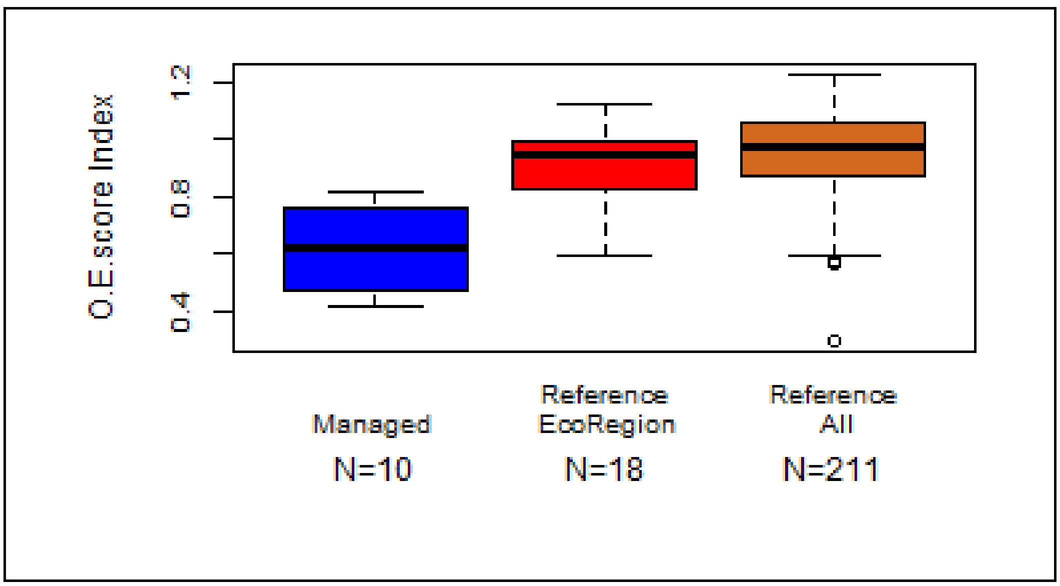

Figure 16. O/E Macroinvertebrate score Index values across the Camp Creek watershed sites. Median and range of index values for managed sites, reference sites within the ecoregion, and reference sites for the entire PIBO study area.

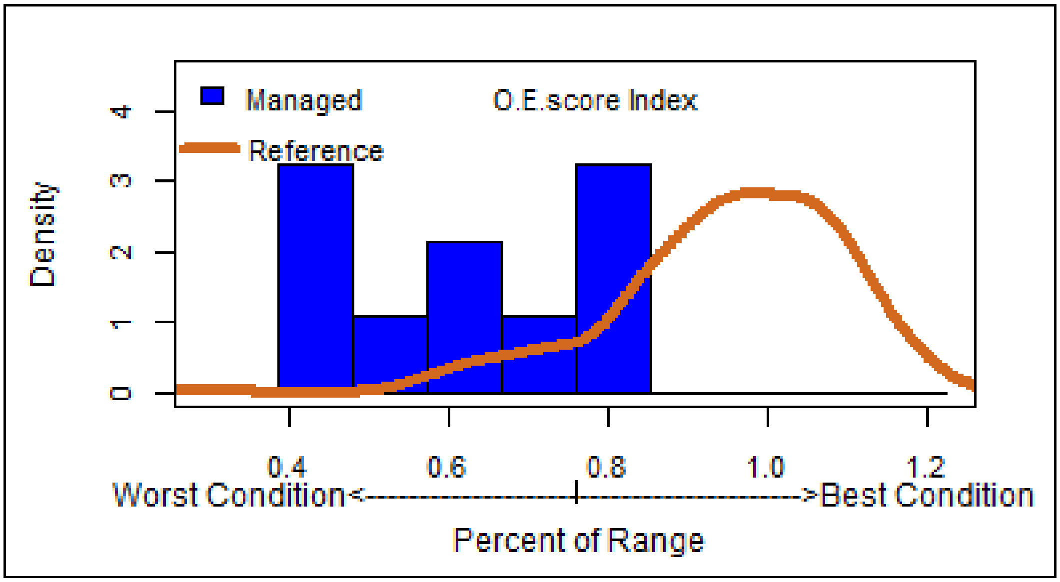

Figure 17. O/E Macroinvertebrate score Index values across the Camp Creek watershed sites. Distribution of index values for managed reaches (histogram) compared to expected values at reference reaches (the line graph).

Status of Physical Habitat Index Scores

Middle Fork John Day River Sites

MFJDR sites could not be compared with PIBO reference sites, which are not appropriate for comparison because they are located in smaller order streams and positioned higher in the watershed. Therefore, the status of MFJDR sites was not determined.

Trend of Physical Habitat Index Scores

Camp Creek Watershed Sites

The trends in physical habitat metrics are summarized in the following table. The PIBO sites in the Camp Creek watershed show an improving trend from 2008-2014 for the majority of the physical habitat metrics (Table 3). The overall habitat index score improved by 30.2%, with 8 of the 10 sites in Camp Creek showing a positive increase. Five of the ten metrics (overall habitat index score, bank stability, percent undercut banks, large wood frequency and residual pool depth) moved in the desired direction. Three of the ten metrics (observed/expected macroinvertebrate score, bank angle, and percent pools) showed no significant difference. Two metrics (percent fines in pools and median substrate size) moved in an undesirable direction over six years.

The biggest improvements observed over the two sampling events was the large woody debris frequency and percent undercut bank metrics that increased by 95.7% and 42.9%, respectively. Median substrate size and the percent fines in pools metrics both worsened over the sampling events. While the decrease in median substrate size and the increase in percent fines in pools is of concern, these metrics are in better condition than reference sites sampled throughout the ecoregion (Figures 8 and 10). Despite the lack of a statistically significant difference, the percent of pools and bank angle metrics trends are moving in a desirable direction.

Figure 18. Photo of site 518-04-I, in Camp Creek, bottom of reach facing upstream, 7/21/14.

Figure 19. Photo of site 518-04-I, in Camp Creek, bottom of reach facing upstream, 7/24/08.

Table 3. Trend in stream habitat attributes across the Camp Creek watershed sites. Including: Overall_Index score, O.E. (Observed/Expected macroinvertebrate score), VegStab (bank stability), UnCutPct (percent undercut banks), LWFrq (large wood frequency), Bank Angle, PTFines6 (percent fines in pool tails), D50 (median substrate size), RPD (residual pool depth), and PoolPct (percent pools). Time1 = mean during first visit; Time2 = mean value for last visit; Percent Change = Percent change in the mean values between the first and last visit; Desired Direction = direction of change in the mean that would result in an improved habitat condition, which can be either + or -; Actual Change = actual direction of change in the mean, which can be either +, -, or not statically significant (NS).

Metric	**Time1 Value**	**Time Value**	**Percent Change**	**Desired Direction**	**Actual** Change
Overall_Index	21.42	27.89	30.2	+	**+**
O.E.	0.71	0.54	-23.6	+	**NS**
VegStab	85.26	95.67	12.2	+	**+**
UnCutPct	10.64	15.2	42.9	+	**+**
LWFrq	51.54	100.87	95.7	+	**+**
BankAngle	135.8	133.4	-1.8	-	**NS**
PTFines6	2.64	6.72	154.3	-	**+**
D50	0.0818	0.0598	-26.8	+	**-**
RPD	0.24	0.32	32.2	+	**+**
PoolPct	29.32	30.52	4.1	+	**NS**

Middle Fork John Day River Sites

The PIBO sites in the MFJDR also show an improving trend from 2009-2014 for the majority of the physical habitat metrics (Table 4). The overall habitat index score improved by 17% with 9 of the 15 sites showing improvement in habitat conditions and 5 sites showing no change over the two sampling events. Five of the ten metrics (overall index score, percent undercut banks, large woody debris frequency, bank angle and percent pools) moved in the desired direction; four of ten metrics (observed/expected macroinvertebrate score, bank stability, median substrate size and relative pool depth) showed no significant difference; and only one metric (percent fines in pools) moved in the undesirable direction over five years.

The biggest improvements detected were large woody debris frequency and percent undercut banks metrics that increased by 160.8% and 52.7%, respectively, over the two sampling events. Despite the large amount of restoration actions that occurred along the MFJDR, the three habitat attributes (bank stability, median substrate size, and relative pool depth) that lacked a statistically significant difference in trend moved in a direction opposite to what was expected.

Figure 20. Photo of site 522-07-I, in MFJDR, top of reach facing downstream, 8/13/14.

Figure 21. Photo of site 522-07-I, in MFJDR, top of reach facing downstream, 7/26/09.

Table 4. Trend in stream habitat attributes across the MFJDR sites. Including: Overall_Index score, O.E.(Observed/Expected macroinvertebrate score), VegStab (bank stability), UnCutPct (percent undercut banks), LWFrq (large wood frequency), Bank Angle, PTFines6 (percent fines in pool tails), D50 (median substrate size), RPD (residual pool depth), and PoolPct (percent pools). Time1 = mean during first visit; Time2 = mean value for last visit; Percent Change = Percent change in the mean values between the first and last visit; Desired Direction = direction of change in the mean, which can be either + or -; Actual Change = actual direction of change in the mean, which can be either +, -, or not statistically significant (NS).

Metric	**Time Value**	**Time Value**	**Perce Chang**	**Desired Directi**	**Actual** Change
Overall_I			17	+	**+**
O.E.	0.47	0.51	8.2	+	**NS**
VegStab	[illegible]	[illegible]	-1.7	+	**NS**
UnCutPct	[illegible]	[illegible]	52.7	+	**+**
LWFrq	[illegible]	[illegible]	160.8	+	**+**
BankAngle	[illegible]	[illegible]	-8.9	-	**-**
PTFines6	2.61	4.98	91.1	-	**+**
D50	[illegible]	[illegible]	-2.2	+	**NS**
RPD	0.58	0.46	-19.7	+	**NS**
PoolPct	44.06	52.6	19.4	+	**+**

Discussion

The improving trend in the overall habitat index score for the majority of PIBO sites in Camp Creek and the MFJDR confirms our hypothesis that the aquatic habitat has improved at a watershed scale after restoration actions were implemented over the last ten years in the MFIMW study area. Two individual habitat attributes (percent undercut banks and large woody debris frequency) improved in both Camp Creek and the MFJDR. This is likely because large woody debris was placed in Camp Creek and the MFJDR for restoration. Depending on the specific placement, large woody debris can assist in creating undercut banks, and may have contributed to that trend being observed in both geographic areas.

Some habitat attributes did not improve as predicted. For example, percent fines in pools both increased in Camp Creek and the MFJDR. Results in both geographic areas were mixed, suggesting that existing impairments are influencing the habitat conditions and the system has not fully responded to restoration. Recent restoration actions may have mobilized fine sediments, or results may be caused by impairments in the watershed such as erosion associated with roads that may still need to be addressed.

Residual pool depth has not increased along with the significant amount of large wood that has been placed in the MFJDR, although the percent of pools has increased. These results may be affected by site (reach) selection as it was unexpected to see residual pool depth decrease by 20% (although not statistically significant). It is also possible that this change may not be associated with the restoration actions and is consistent with general conditions throughout the watershed. The lack of a trend being observed for the macroinvertebrate metric in both geographic areas indicates that the biotic integrity of the macroinvertebrate community has not yet responded to the restoration actions. Macroinvertebrates may be highly sensitive to water year variability or to conditions resulting from historic land management practices.

Our hypothesis that the condition of Camp Creek watershed habitat attributes will more closely resemble reference site conditions after restoration actions were implemented was not supported. Although several restoration projects were implemented in Camp Creek, the aquatic habitat is still well below reference desired conditions. This important finding highlights the need for additional restoration actions to improve the aquatic habitat at a watershed scale. The site-specific and combined data in Camp Creek can serve as a planning tool to target and design restoration projects that can improve the aquatic habitat.

Lessons Learned

Monitoring

Look for efficiency and economies of scale in large scale monitoring efforts. Water temperature was not monitored as part of the PIBO monitoring effort because the MFIMW study had already invested in monitoring this parameter. This slight modification resulted in a cost savings that allowed the PIBO crews to perform the habitat monitoring and meet the MFIMW's budget constraints.

Maintain continuity of long term sampling sites to enable trend detection using an established protocol that generates habitat metrics important to salmonids. Long term data sets that are sampled at regular intervals are essential to detect trends. It was important that the UMFWG members resampled the PIBO sites in 2014 after the initial investment of establishing these sites in 2008/9. Future sampling of these sites should continue to occur at five year intervals to determine the response of past and future restoration actions that are likely to affect the stream habitat attributes that PIBO measures. The next sampling event should occur in 2019.

Designating a point person helps to streamline workflow and create efficiencies for all partners involved. Collaborative partnerships need a point person to analyze data. While the UMFWG made a collective decision to fund this data collection effort, no specific agency (or person) was identified to analyze the data collected in the different geographic areas. Although the USFS Forestry Sciences Lab produced data and provided a brief summary report, they did not analyze the data to answer specific questions for the Camp Creek sites.

Additional analyses of the existing PIBO data could provide a clearer picture of what is happening at specific reaches in Camp Creek. Analyses of the data from the control and treatments sites are needed to reveal how the removal of the log weirs and subsequent addition of large woody debris may have changed the physical habitat attributes. This information will also serve to provide information to design and implement future restoration actions. To provide additional time for the stream to respond to the restoration actions, this analysis should happen after the 2019 resurvey.

The existing vegetation data also offers an opportunity to understand how riparian habitats have changed based on passive and active restoration actions in both geographic areas. Although the bank stability attribute incorporates the vegetation data, it is difficult to understand how the riparian vegetation has changed by looking at this attribute.

We suggest answering the following questions after the 2019 resurvey is performed:

- How has the riparian vegetation changed in passive restoration sites?
- Has the change in riparian vegetation affected physical habitat attributes such as bank stability and percent fines in pools?
- Have riparian plantings improved the vegetation and how does this compare to passive restoration actions alone?
- Are invasive plant species more predominant, if so which ones?
- PIBO data from the MFIMW should be combined with the larger PIBO data set that is from the randomly established sites in the MFIMW study area by the USFS.

Further analyses could help understand how status and trends compare over a larger spatial scale. The Camp Creek data could be analyzed with the other PIBO tributary sites in the MFIMW to understand how the tributaries are comparing to reference conditions. This larger data set could also be analyzed to show changes over time. These randomly established sites could also be pooled with the Camp Creek and MFJDR sites to look at trends over time to better describe changes over a larger watershed scale.

Finally, this information could be utilized to understand changes in the MFJDR at a finer spatial scale. The PIBO sites that are within or downstream of restoration projects could be analyzed to see how specific habitat attributes changed over time. Did residual pool depth increase at or downstream of restoration projects? Is the riparian vegetation responding to grazing management? Has riparian plantings improved the vegetation? By repeating the PIBO survey in 2019 it would provide additional time for the stream to respond to restoration actions and another data point to establish trends.

Restoration

This sampling effort was not designed to be able to recommend specific restoration actions to address limiting factors or locations, as that is beyond the scope of PIBO.

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Appendix D – Geomorphology and Physical Habitat

Pat McDowell, University of Oregon, Department of Geography, Eugene, OR

Abstract

Changes in channel geomorphology, sinuosity, pool depth, bed material, and fish cover were monitored over about seven years in six reaches, three project reaches where active restoration projects were installed, and three control reaches. We also monitored changes in channel morphology at individual log structures. The goal was to test specific restoration goals of the projects, such as increasing pool depth or narrowing channels. In addition to the active restoration projects, removal of livestock grazing spurred increases in vegetation within the active channel that have had important influences on channel morphology and habitat. Channels did not narrow and deepen or become more sinuous in response to restoration as hypothesized. This may be because not enough time has elapsed since restoration for fluvial response to be fully developed. Restoration did produce a significant increase in pool depth. Bed material was generally in good condition at the beginning of the study. Both project reaches and control reaches experience a significant decrease in the percentage of embedded gravels. The 2011 flood, one of the largest floods ever recorded on the MFJD, did not cause significant net erosion or deposition, indicating the channel is relatively stable and in dynamic equilibrium.

Introduction

Background

River channels are open systems that are shaped by throughput of water and sediment, and they are dynamic, continually changing at short and long time scales (Beechie et al. 2013). The restoration projects studied in this project were intended to improve habitat for native salmon, and to improve overall ecological function in the river, by building structures in the channel (called log structures or engineered log jams) that mimic natural wood accumulations, removing inhibiting features such as bank rip-rap, and encouraging increases in riparian vegetation through both passive (remove cattle grazing) and active (planting) approaches. These restoration techniques are expected (hypothesized) to interact with flow and sediment either to produce additional ongoing geomorphic changes, or to stabilize or dampen undesirable changes. Geomorphic processes such as bed scour, bar aggradation and bank aggradation can improve aquatic habitat by creating and deepening pools, narrowing channels to create deeper summer flows and shading, mobilizing gravel beds to flush out accumulated fine sediment, increasing channel sinuosity, and modifying channel cross sections to produce a diversity of fast and slow, deep and shallow flows in a reach. Vegetation growing within the channel, on bed, banks and bars, is often an active agent in scour and aggradation and may be more active than direct hydraulic processes (Corenblit et al. 2007). There has been, however, little study of whether specific restoration techniques produce the intended results, and restoration projects in some watersheds have had neutral or negative effects (Kondolf et al. 2001; Mika et al. 2010). Therefore, for each restoration project, these intended effects can be considered hypotheses, and the role of this monitoring effort is to test them.

The goal of the geomorphology and physical habitat monitoring was to evaluate whether the restoration projects are achieving specific goals for improvement of geomorphology and physical habitat. We used a control-impact approach to monitoring, as opposed to a before-after approach, because the time and funding available did not allow for pre-restoration monitoring, and it was not clear at the beginning of the study what reaches would be restored during the study period. However, most of the hypothesized changes (area, width and W:D decreasing, sinuosity increasing) were expected to develop over a period of years following restoration as the channel responded to changed hydraulics due to the restoration structures. Therefore, our monitoring for seven years following restoration does provide a preliminary test of these hypotheses.

Monitoring focused on detecting expected direct effects of restoration projects at the reach and sub-reach levels, in contrast to the watershed-scale PIBO monitoring. Although some effects of restoration, such as changes in fish populations and water temperature, may spread beyond the restoration site, changes in channel morphology, fish cover, and bed material characteristics are not expected to spread spatially beyond the individual restoration feature or the project reach. Therefore, we monitored at restoration project reaches and control reaches selected to be as similar to the project reaches as possible. We monitored at the same spatial locations each time to ensure the restoration context was known (i.e., at a restoration log structure or not). We used high resolution measurement techniques and spatial control to optimize our chances of detecting effects.

We monitored at six reaches (Fig. 1, Table 1), including three project reaches and three control reaches. The project reaches were selected because they were the first three reaches restored after 2007. All of the reaches are unconfined, low-gradient reaches, although they have some differences in land use history.

Table 1. Characteristics of project reaches. Year of construction for each project is given in parentheses.

Reach name	Reach type	Land owner	Drainage area, km^2	Reach length, km	Channel gradient, %	Sinuosity (2006)
VIBR(P)	Project (2008)	CTWS Forrest CA	305	1.19	0.52	1.20
BUTI(C)	Control	CTWS Oxbow CA	465	0.83	0.38	1.32
BEBU(P)	Project (2011)	CTWS Oxbow CA	470	1.72	0.40	1.46
RABE(P)	Project (2009)	CTWS Oxbow CA	555	0.93	0.59	1.08
DRRA(C)	Control	CTWS Oxbow CA	570	1.05	0.58	1.14
JUCA(C)	Control	US Forest Service	735	1.20	0.50	1.07

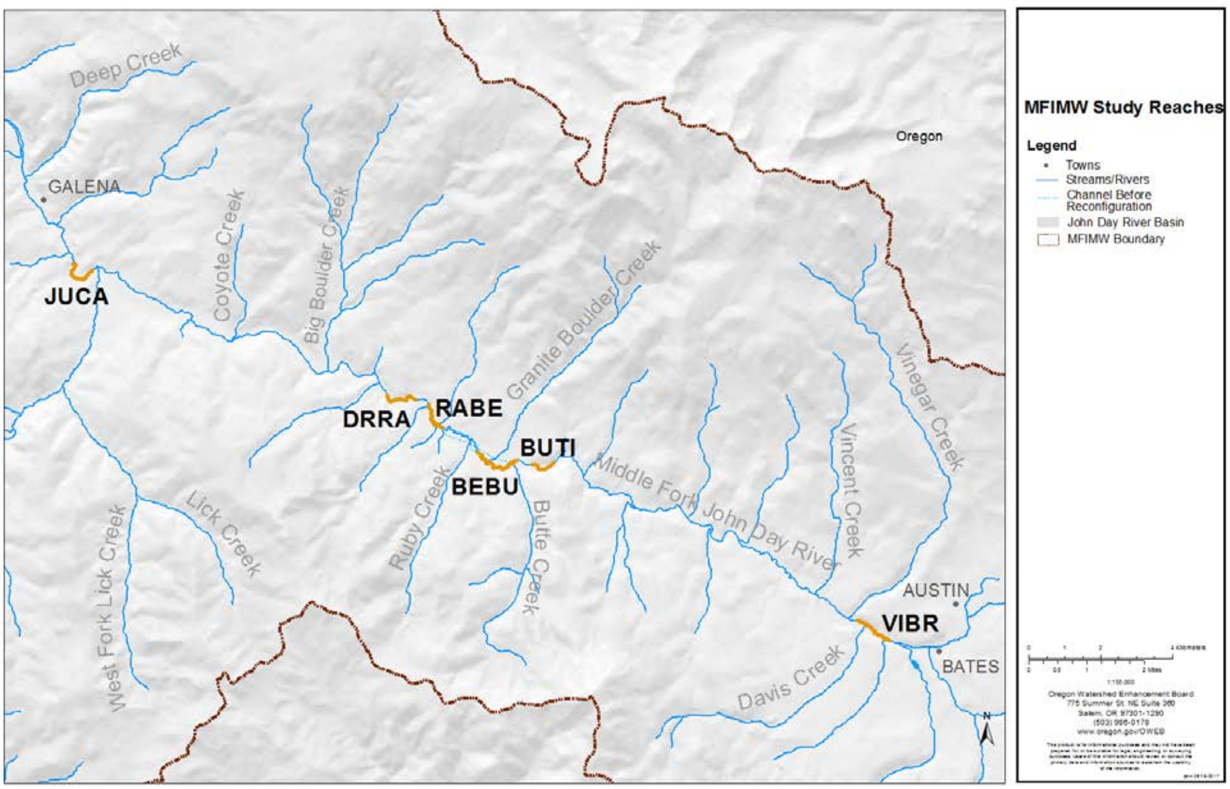

Figure 1. Map of study area showing reaches monitored.

Processes shaping geomorphology and physical habitat in the study area have been influenced since 2000 by both active restoration and passive restoration. Each of the project reaches was the site of an active restoration project built between 2008 and 2011. The primary restoration techniques included removal of rock spurs and bank rip-rap; construction of log structures anchored into the channel banks; addition of unanchored large wood pieces (large woody debris; LWD) in the channel and on the floodplain; LWD anchored on the floodplain to provide floodplain roughness; and enlargement of upstream mouths of intermittent side channels. Log structures redirected flow of water and affected hydraulics, potentially leading to erosion and deposition and changes of channel form. Side channels were enlarged to encourage high flow into them and development of secondary channel habitat. Most of the log structures were positioned over pools or were intended to create pools, and during construction the pools were dug up to 1 – 1.5m deep.

In addition, both project and control reaches were affected by passive restoration (Kaufman et al. 1996), mainly the removal of cattle grazing after 2000. The control reaches did not have any active restoration in recent years, but they were subject to passive restoration to some extent. Cattle grazing was removed from two of them (BUTI(C) and DRRA(C)) in 2000. Cattle grazing continued in the third control reach (JUCA(C)), but stocking level and grazing season were probably reduced since the 1990s. Grazing by wild ungulates (deer, elk) has continued in all reaches.

Significant effects from the removal or reduction of cattle grazing have been observed in the study area. Vegetation cover increased on the channel banks and point bars. One native sedge species, torrent sedge (*Carex nudata*), greatly increased in abundance, occupying formerly bare point bars, the edge of the low-flow channel bed and mid-channel positions on riffles, and forming islands (Fig. 2). Torrent sedge is an extremely tough tussock-forming sedge that is anchored tightly into the gravel bed by roots and ably survives high flows (Levine, 2000). For example, there was very little torrent sedge mortality or dislodgement in the 2011 flood. The spread of torrent sedge has stabilized point bar surfaces and the channel bed, and redirected flow, causing localized scour and deposition. Torrent sedge is found in the Middle Fork John Day River in all the study reaches except VIBR(P) (from about Deerhorn Creek downstream). In addition to the vegetative effects of removing cattle grazing, there was also active planting of woody shrubs on channel banks in all reaches except JUCA(C).

Goals and objectives

In 2008, the first year of the monitoring program, we extracted goals and objectives for the planned restoration projects from the project plans that were available at that time, including plans for the VIBR(P) and RABE(P) projects. (The goals and objectives expressed in the later restoration plan for BEBU(P) were not significantly different from those of the early plans, so we continued to monitor based on the initial goals, objectives, indicators, and hypotheses.)

These goals and objectives are tied to active restoration projects, where the channel was being purposefully modified through removing rip-rap and rock spurs that had been added in earlier years, adding LWD structures, opening side channels and oxbows, and planting vegetation on banks and floodplains. The goals and objectives were correlated across projects, and were used to select monitoring indicators for geomorphology and physical habitat, and to develop hypotheses for those indicators. We initially developed a long list of monitoring indicators that would describe the restoration objectives, including indicators measured in the field and others to be measured from aerial imagery in GIS. We selected a subset of these indicators that we thought would be feasible to monitor with the limited funding and time available. Restoration objectives, indicators and hypotheses used in monitoring are summarized in Table 2.

Figure 2. View of channel in BEBU(P) reach before and after removal of cattle grazing. Note the expansion of torrent sedge by 2008 on the 1996 bare gravel bar, and on the island in the foreground in 2008.

Table 2. Goals, indicators and hypotheses for changes over times as a function of active restoration

Restoration Goals	Monitoring indicator	Hypothesis for restoration effects
Log structures increase roughness and overbank flows to increase channel-floodplain connectivity	Cross-section area	Area decreases
Log structures and revegetation add roughness and obstruction, resulting in channel scour	Cross-section area	Area increases
Move toward natural channel morphology	Width:depth ratio	W:D decreases (channels become narrower and deeper)
Move toward natural channel morphology Revegetation and bank stabilization will result in sediment trapping and narrowing.	Width	Channels narrow
Log structures create an obstruction, inducing lateral scour	Width	Channels widen
Channel narrowing and scour under log structures will result in deepening	Depth	Log structures create and maintain deep pools
Move toward natural channel morphology, and increase lateral migration, by removing rock, placing LWD, and reconnecting oxbows	Sinuosity Lateral migration rate	Reach-level sinuosity increases Lateral migration rate increases
Reaches are dynamically stable (the channel may shift but dimensions do not change significantly)	Area, width, W:D Channel morphology at log structures Gravel size (D50, D95)	Area, width, and W:D are relatively stable over years; incision does not occur Channel cross-sections at log structures are dynamically stable D50 and D90 do not increase
Increase aquatic habitat quality by placing LWD, removing rock	Number of pools/km, maximum pool depth, residual pool depth	Pools/km, deep pools/km increase Reach-level residual pool depth increases Deep pools form and are maintained at LWD structures
Increase aquatic habitat quality by placing LWD, removing rock	% fines in gravel beds, gravel embeddedness	% fines decrease; gravel embeddedness decreases
Increase fish cover by placing LWD	% area of fish cover	Fish cover increases

As the project progressed, it became clear that there were also effects from passive restoration (removal of cattle grazing). We identified possible effects from passive restoration based on previous studies in the scientific literature (Table 3). Expected passive restoration effects are generally in the same direction as the corresponding hypotheses for active restoration effects. The fact that the dual effects of passive restoration and active restoration are parallel is positive, because the chances of seeing a positive restoration effect is increased. However, the dual effects make it impossible to determine whether observed changes are due to active or passive restoration, or a combination of the two. Furthermore, we did not begin monitoring until eight years or more after the beginning of passive restoration, so it is possible that most changes due to passive restoration had already occurred.

Table 3. Hypotheses related to passive restoration (removal of cattle grazing)

Restoration effect	**Monitoring indicator**	**Hypothesis for restoration effects**
Increased vegetation cover on channel banks	Width Width:depth	Channels narrow W:D decreases
Reaches are dynamically stable due to reduced disturbance	Area, width, W:D	Area, width, and W:D are relatively stable over years; incision does not occur
Increased fish cover due to increased channel and floodplain vegetation	% area of overhanging vegetation cover, % area of aquatic vegetation cover	% area of overhanging vegetation cover and aquatic vegetation increases
Reduced fine sediment in transport and in channel bed due to reduced soil erosion	% fines Embeddedness	% fines decreases Embeddedness decreases

In addition to these general effects of increased vegetation cover, we observed some specific geomorphic effects from the spread of torrent sedge, such as increased roughness, formation of mid-channel islands, and stabilization of active bars. Because torrent sedge primarily occupies the channel bed rather than banks, and it forms protruding tussocks (commonly 0.5m high or more), we expected the effects to be different from those listed in Table 3, but we did not have a clear model of the effects. This topic is the subject of a separate ongoing research project (M. Goslin, doctoral dissertation, University of Oregon, in progress).

In addition to the spread of torrent sedge, a large flood event occurred in April-May 2011. Flows were extensively overbank, and we observed that gravels and cobbles were mobilized and redeposited in all the study reaches. At the Middle Fork John Day River at Ritter gage (USGS 14044000), it was the largest peak flow event recorded in 76 years of record. The recurrence interval was approximately 30 to 40 years. (The gage at Middle Fork John Day River above Camp Creek (USGS 14043840) did not go into service until fall 2011.) Large floods have the potential to erode banks, mobilize and re-shape bed morphology, remove bank vegetation, and even produce avulsions in many rivers (Knighton 1998). We wanted to determine if geomorphic change we observed over the course of this study was due to the flood event, or to incremental change through the period. Although we were not able to monitor geomorphology and physical habitat immediately before and after this flood event, we did make as many measurements as possible in summer 2011 to capture flood effects. As part of analyzing change over the period of this study, we analyzed change between our last measurement date before the flood and our first measurement date after the flood event, in an attempt to capture the effects of the flood as closely as possible.

Hypotheses

The monitoring indicators that were selected based on the restoration goals fall into six groups: channel morphology, channel planform, pool characteristics, morphologic change at log structures, bed material, and fish cover. Based on the goals and objectives described above, we identified specific hypotheses for the direction of change (increase, decrease) in each monitoring indicator. Hypotheses are listed in Tables 2 and 3. Table 4 shows the indicators and the field protocols used to measure each indicator.

Table 4. Monitoring indicators and methods. (Methods are described in detail in the appendix.)

Group	Monitoring indicators	Methods
Channel morphology	Cross-section area Width Width:depth ratio Depth Bed aggradation/degradation Bank erosion/deposition	Cross-section surveys
Channel planform	Sinuosity Lateral migration rate	GIS analysis of aerial imagery
Pools	Pools/km Deep pools/km Residual pool depth	Residual pool depth
Log structures	Bed aggradation/degradation Bank erosion/deposition	Detailed log structure survey; Log structure cross-section
Bed material	D50, D95 % Fines % Gravel embeddedness Embeddedness height ratio	Gravel count
Fish cover	% Area of fish cover, % Area of fish cover by type	Fish cover

Site Selection

Six reaches were selected for monitoring for geomorphology and physical habitat, three project reaches and three control reaches. Characteristics of the six reaches are summarized in Table 1 and their locations are shown in Figure 1.

Control reaches were selected to be as similar as possible to the restoration reaches in terms of watershed level controls of geomorphology (valley width, slope, drainage area and geology). Only a limited number of potential control reaches met the criteria. Because the restoration projects were implemented in reaches with unconfined channels and space for meander development, unconfined reaches were selected for the control reaches. The limited number of unconfined reaches in the upper Middle Fork John Day watershed guided selection of control reaches. We recognize that the control reaches are not perfect matches for the project reaches. Project and control reaches were not similar enough to pair reaches for analysis, therefore analysis compared all project reaches to all control reaches.

Table 5. Reaches and timing of monitoring. The year of construction of each project reach is given in parenthesis in the first column.

Reach	Reach type	Year of project	First year monitoring	Subsequent year monitoring
VIBR(P)	treatment	2008	2008	2010, 2011, 2013, 2014, 2015
BEBU(P)	control (2009); treatment (after 2011)	2011	2009 (C), 2013 (P)	2011, 2016
RABE(P)	treatment	2009	2009	2011, 2013, 2014, 2016
BUTI(C)	control		2008	2010, 2012, 2015
DRRA(C)	control		2009	2011, 2015
JUCA(C)	control		2010	2012, 2016

Methods

Sampling Plan

Within each reach, monitoring sites (channel cross-sections, fish cover plots, and gravel count plots) were selected by regular sampling of appropriate channel units for the monitoring protocol, i.e., every other pool, every other riffle. The number of monitoring points selected for each reach varied to achieve a similar density of points/km in each reach. All of our fieldwork was conducted in summer, primarily in July and August. The aerial imagery used was also flown in summer and represents summer flow conditions.

Detailed field and data analysis methods are in the Appendix.

Channel Morphology

We monitored channel form through repeated channel cross-section surveys that followed standard protocols (Harrelson et al., 1994), using an auto-level or total station. Each channel cross-section was monumented by driving a piece of rebar 0.5-0.75m long into the ground, and repeat surveys were set up on the same monuments for high replicability. We selected a bankfull elevation for each cross-section, based on bankfull indicators and consistency within the reach, as a standard elevation for measuring channel dimensions across repeat surveys. For each survey, we plotted the data and measured bankfull cross-section area and bankfull width (top width), and calculated mean depth (area/width) and width:depth ratio.

We visually examined plots of the surveyed channel cross-sections to identify the dominant modes of behavior. We recorded the presence of seven types of channel adjustment (bank erosion, bank slumping, bank aggradation, point bar aggradation, lateral migration, bed aggradation and bed incision) for each cross-section by comparing the first and last years of cross-section survey. We estimated the amount of vertical or horizontal movement to the nearest 0.1m.

Pool Depth

Pools were counted and measured in 2008, before active restoration projects, as part of a US Forest Service stream habitat inventory (U.S. Forest Service, 2008). We replicated these measurements, using the same protocol (U.S. Forest Service, 2007), in 2015 and 2016 on five reaches, with spatial locations collected using GPS and manually. 2008 pool depth data were not available for JUCA(C). From field data, we calculated residual pool depth (maximum pool depth minus pool tail crest depth), and pools per km. To compare 2008 and 2015-16 pool depths, we first incorporated the 2008 pools data in our GIS using a limited number of spatial locations noted in the 2008 survey. We then matched locations of 2015-2016 pools to the closest pool in the 2008 stream habitat survey. There is probably some error in the locations of the 2008 pools, so these correlations are not highly reliable. However, we were able to match almost all the pools in the three reaches, so in effect this is almost a comparison of the complete sample of pools in 2008 and 2015-16.

Channel Morphology at Log Structures

We monitored channel form and post-restoration adjustments at log structures in project reaches VIBR(P), RABE(P) and BEBU(P) . In VIBR(P) and RABE(P), we did a detailed survey of bed and bank morphology at selected log structures, with an RTK_GPS surveying set-up, and repeated this survey after two years and again two to three years after the second survey, for a total five to six year span. At each log structure, we surveyed an area that encompassed the complete log structure, covered the bed to the opposite bank, and extended 5 or more meters upstream and downstream of the structure. We created a gridded digital elevation model from the survey points for each year and assessed change by subtracting the elevation of the early years from the elevation of the final year. Methods are detailed in Duffin (2015). In BEBU(P) , because of limited field time available, we surveyed three to five cross-sections (following our standard cross-section survey procedure) across the channel at each log structure in 2011 within a few weeks following construction. We repeated these surveys in 2013 and 2016, for a total five year span. Pool depth at log structures before and after restoration were monitored using the pool depth methods described above.

Channel Planform

We measured channel sinuosity by digitizing the channel centerline on high-resolution aerial imagery (Guerrant, 2014). Sinuosity is calculated as meandering channel length divided by valley centerline length. Imagery used for the sinuosity measurements included 2006 (Watershed Sciences, 2006) and 2013 (Dietrich, 2014, 2016).

Fish Cover

We used two methods to assess fish cover. First, in the three project reaches we directly measured the total area of fish cover attributable to log structures and to *Carex nudata*, by digitizing cover boundaries from the 2013 aerial imagery. We digitized the total wetted channel area, and calculated percent area in fish cover of each of these types.

Second, we measured fish cover at several plots in each of the six reaches, following the protocol of U.S. Environmental Protection Agency (2004). Plots 10 m long and spanning the wetted channel were established in pools and glides. We visually estimating the percent of area in each of ten types of fish cover (filamentous algae, aquatic macrophytes, large woody debris, brush and small woody debris, live trees and roots, overhanging vegetation, undercut banks, boulders, artificial structures (log structures, weirs, etc.), and turbidity) by section within each plot. Aquatic macrophytes includes emergent plants rooted in the water, such as torrent sedge, other sedges and reeds, and submerged aquatic plants. We excluded turbidity from the analysis because it was almost always zero. We excluded filamentous algae from analysis because values at a sampling plot were highly variable from year to year, it was unclear what processes were controlling this variation, and mechanisms for restoration to influence algae were not obvious.

Bed Material

We sampled the surface layer of the channel gravel bed at selected riffles using a modified version of the Wolman pebble count. At each selected site, we measured the size of bed material at 300 points, using a gridded sampling scheme (Harrelson et al., 1994) and a gravel template (gravelometer). All diameters were recorded in millimeters. Points where the diameter was less than 2mm (including sand, silt and clay) were recorded as fines. For each clast, we recorded whether or not it was embedded in fine sediment, and the depth of embeddedness. Descriptive variables, including percent embedded, embeddedness ratio (total particle height, percent fines, mean, and diameters of the 50th (D50), 84th (D84) and 95th (D95) percentiles.

In addition we took and analyzed volumetric samples (Bunte and Abt 2003) of the surface layers (armor layer) and the sub-armor layer at two to three sites in each reach. Using a barrel sampler, the entire surface layer of sediment and then an equivalent depth of the subsurface layer were each removed and analyzed by size class. All sediment sizes except suspendable fines (silt and clay, which were lost in the flow) were collected and measured. Similar particle size variables as for the pebble count method were calculated.

Data Analysis

We used graphs to visually, qualitatively evaluate the differences between years and among reaches, and statistical tests to test for significance of differences. T-tests were used to test hypotheses by evaluating the differences between project and control reaches. If data were not normally distributed and of equal variance, they were transformed using a log10 transformation, before statistical analysis. Results of statistical analysis are given in the Appendix.

Results

Analysis and Interpretation of Channel Morphology Data

Table 6 summarizes channel morphology for each reach in the initial year of measurement. While cross-section dimensions tend to increase slightly going downstream, there is large within-reach variation. A few cross-sections are narrow and deep (W:D<10), but most cross-sections are intermediate (10<W:D<40) or slightly wide and shallow (W:D>40) in shape.

Table 6. Summary of bankfull channel dimensions of the reaches, in order from upstream (VIBR(P)) to downstream (JUCA(C)), based on initial year measurements.

Reach name	VIBR(P)	BUTI(C)	BEBU(P)	RABE(P)	DRRA(C)	JUCA(C)
Reach type	project	control	project	project	control	control
No. of cross-sections	11	8	14	10	10	12
Median area, sq. m.	6.30	6.17	9.50	8.93	17.03	11.16
Range of area, sq. m.	4.70-9.97	2.01-16.90	4.26-16.81	5.51-19.08	9.09-21.34	7.17-42.27
Median width, m.	14.64	13.70	15.50	17.55	23.95	20.48
Range of width, m.	10.08-19.97	8.01-27.05	8.26-30.80	10.09-33.41	15.50-29.28	13.50-33.11
Median depth, m.	0.43	0.59	0.64	0.55	0.71	0.60
Range of depth, m.	0.33-0.91	0.25-0.90	0.25-1.08	0.34-0.85	0.49-0.92	0.36-1.28
Median W:D ratio	36.90	30.45	26.44	34.11	35.32	28.75
Range of W:D ratio	12.17-55.84	14.52-43.30	8.05-90.83	11.90-60.13	22.06-46.61	22.92-75.85

Our repeat measurements of cross-sections over a six to seven year period allowed qualitative observations on how cross-sections are changing in response to restoration. By visually comparing the same cross-section through repeat surveys, the dominant modes of channel adjustment were summarized for each of the six reaches (Table 7). Each reach shows distinctive and relatively consistent behavior. For example, reaches dominated by bed aggradation had no or perhaps one cross-section that incised, and vice versa. Three reaches (one project and two control) were dominated by bed aggradation and three by bed incision. The magnitudes of aggradation and incision were small, typically 0.2-0.1m total over the period of study. Bank erosion and point bar aggradation were commonly observed. None of the reaches showed widespread bank aggradation or bank slumping, although there were a few isolated cases of bank slumping. BEBU(P)-XS06 experienced 7 m of bank erosion, probably due to collapse of an undercut or piped bank. JUCA(C)-XS11 and JUCA(C)-XS13 both experienced 3 m of bank erosion, probably through normal fluvial scour. The JUCA(C) reach visually appeared to be the most active reach, showing lots of fresh scour and deposition after the 2011 flood and in later years.

Based on the relatively modest amounts of erosion and deposition, especially given the large flood in 2011, we conclude that both project and control reaches are dynamically stable, meaning they are eroding and depositing but are not experiencing major changes. BEBU(P) is the only reach that we were able to survey before restoration, so the channel adjustment represents a before-after comparison. BEBU(P) was dominated by bed aggradation, but the amount of aggradation was modest, typically about 0.1m over a seven year period. At most BEBU(P) cross-sections, the aggradation was well underway between 2009 and 2011, so it is not attributable to the restoration actions later in 2011.

Table 7. Summary of dominant modes of channel change for each reach, over the period of study. The dominant modes of adjustment are those displayed by one-third or more of the cross-sections in the reach.

Reach	Reach type	Bank erosion	Point bar aggradation	Lateral migration	Bed aggradation	Bed incision
VIBR(P)	Proj	X	X			X
BEBU(P)	Proj				X	
RABE(P)	Proj	X	X	X		X
BUTI(C)	Cont	X				X
DRRA(C)	Cont		X		X	
JUCA(C)	Cont	X	X	X	X	

We also analyzed qualitative changes in channel dimensions based on the repeat cross-section surveys. The BEBU(P) 2011 and 2016 data represent a before-after comparison. Results showed that, for BEBU(P) overall, area and depth decreased slightly, while width and W:D increased slightly (Appendix Figs. CM-1 to 3). Variability was high among the cross-sections in the reach, with some cross-sections increasing and others decreasing on each monitoring indicator. The differences between 2011 and 2016 in area, width, depth, and W:D were not statistically significant (Appendix Table CM-1). These results do not support the hypotheses for restoration.

For VIBR(P) and RABE (P) where we did not have before measurements, our surveys after restoration reflect restoration effects developing over a period of years through fluvial processes influenced by the restoration projects. We examined trends in channel cross-section dimensions in all reaches over the period of study, to determine whether the project reaches showed different trends from the control reaches, and whether the trends were consistent with our hypotheses or not. There was no consistent pattern of change in cross-section area over the period of monitoring (Fig. 3), and there was no consistent difference in trends between project and control reaches (Appendix Table CM-2).

For the project reaches, cross-section area increased slightly in VIBR(P), remained about the same in BEBU(P) , and decreased slightly in RABE(P). Behavior was also inconsistent in the control reaches. Similarly, VIBR(P) increased in width while BEBU(P) and RABE(P) predominately decreased in width (Fig. 4). Changes in width:depth ratio were relatively small across all reaches (Fig. 5). Eighty-two percent of the cross-sections widened or narrowed by less than 1 W:D unit per year. None of the reaches had statistically significant changes in channel dimensions over the period of study (Appendix Table CM-3). .

Overall, the hypotheses about the effects of restoration on cross-section area, width and W:D stated in Table 2 are not supported by the data. Project and control reaches also are not different in the processes of adjustment (Table 7). Reach response seem to be controlled by factors such as individual reach slope, stream power, bed material, local sediment inputs from tributaries, etc., rather than by restoration. Despite our intent to select reaches that were geomorphically similar to each other, we did not succeed; subtle geomorphic differences are producing differences in behavior between reaches. This illustrates the difficulty of finding true control reaches for comparison with treatment reaches.

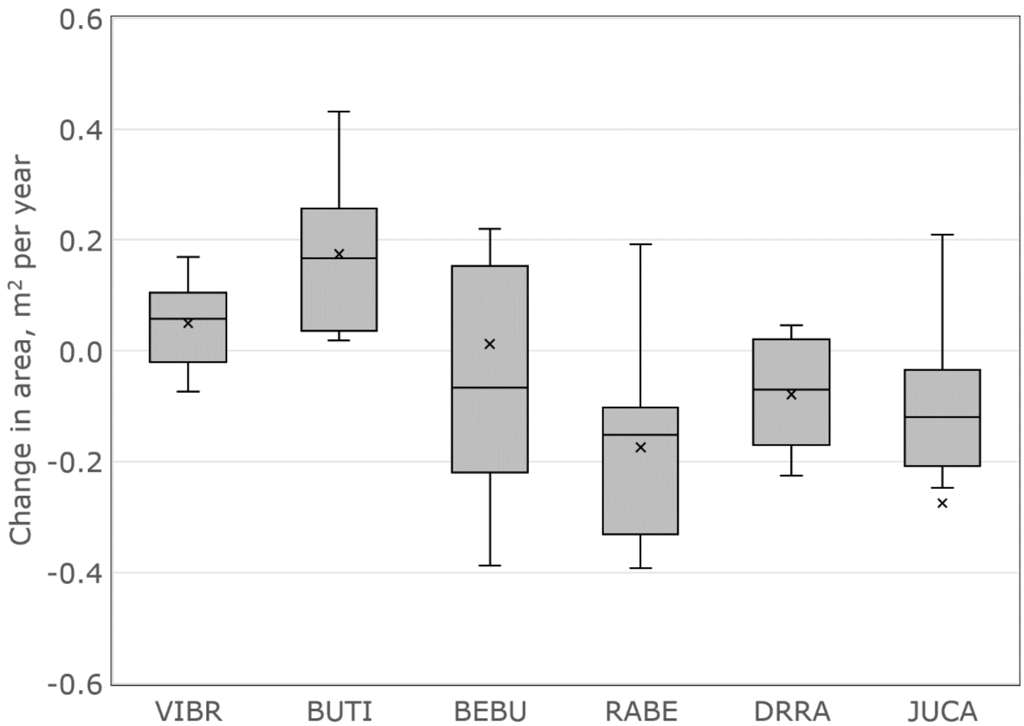

Figure. 3. Change per year in cross-section area, over the monitoring period. (Two large outliers are not shown.) The box extends from the 25th to 75th percentile. The whiskers capture the largest quartile and smallest quartile. The horizontal line represents the median, and the x represents the mean.

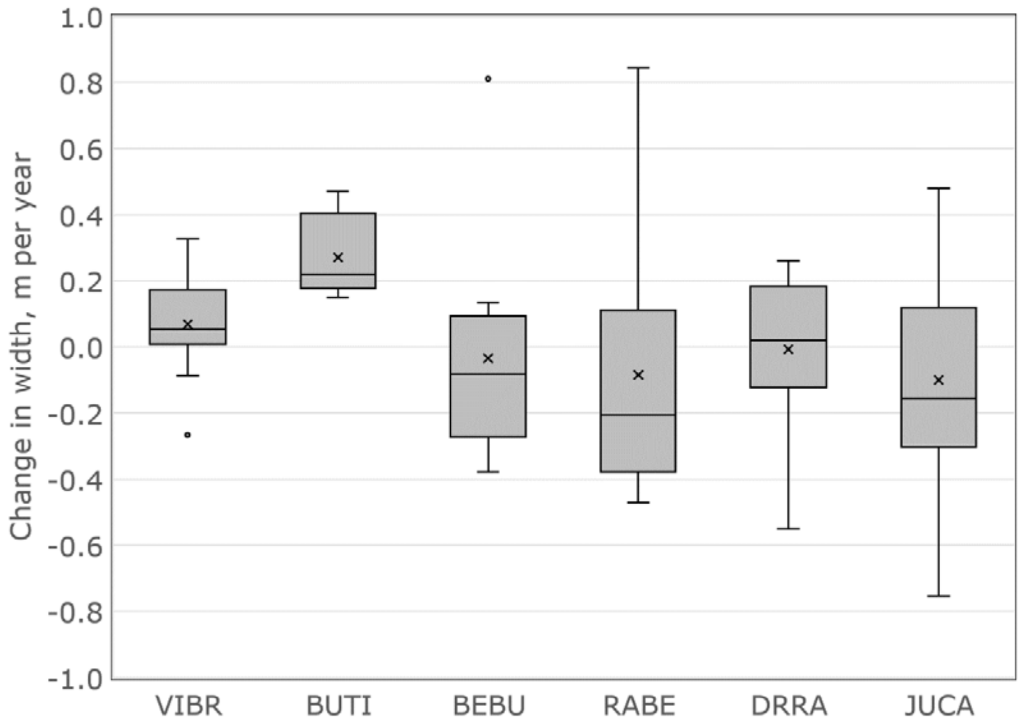

Figure 4. Change per year in width over the monitoring period.

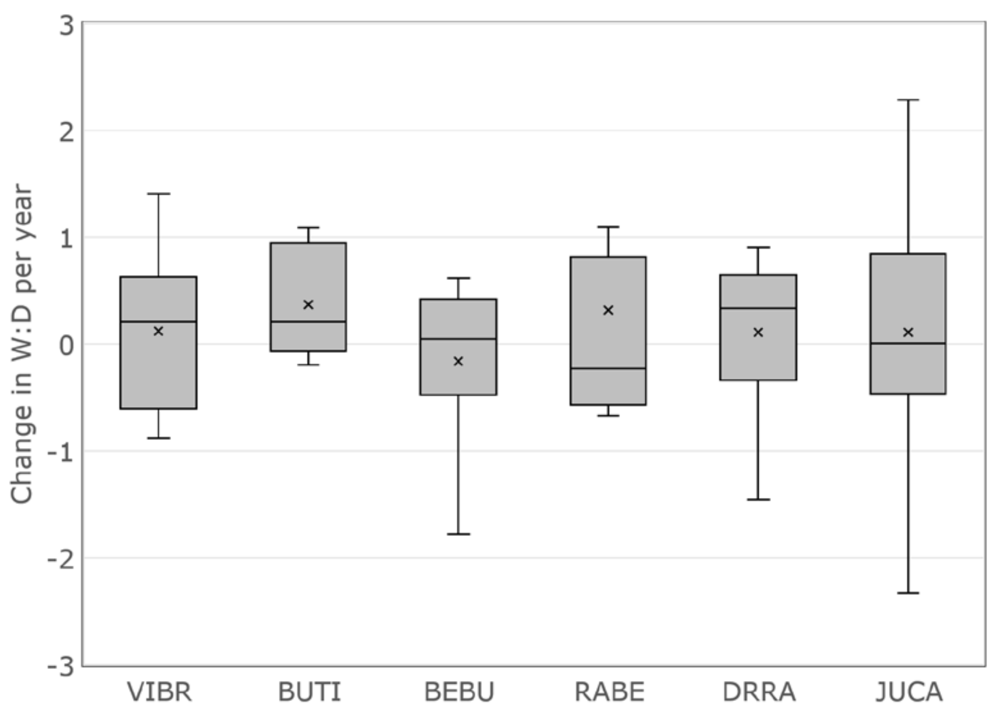

Figure 5. Change per year in width:depth ratio over the monitoring period, across reaches. One large outlier, RABE(P)_XS09 (which experienced major aggradation during the 2011 flood), is not shown. This cross-section is at the mouth of Beaver Creek, so aggradation may have been related to sediment delivery from Beaver Creek.

Effects of the 2011 Flood on Channel Morphology

We also examined channel change due to the large flood of 2011. From visual examination of the channel in the summer of 2011, we know that the flood was geomorphically effective in mobilizing bed material and doing erosion. We observed fresh gravel accumulations on many point bars. Some torrent sedge tussocks were uprooted and transported downstream as a unit, to be redeposited in the channel or occasionally on the floodplain. However, most torrent sedge tussocks survived the flood in place and put out leaves as normal in summer 2011. We saw few gaps in the torrent sedges, and we feel that the vast majority of torrent sedges survived the flood in place.

We analyzed change in channel dimensions between the nearest year prior to the flood, and the nearest year after the flood for which survey data were available. We attempted to survey all reaches in summer 2011 following the flood, but we were not able to complete all reaches. Data about the flood measurements are summarized in Table 8.

Table 8. Cross-section survey years bracketing 2011 flood

	Pre-flood measurement	**Post-flood measurement**
VIBR(P)	2010	2011
BUTI(C)	2010	2012
BEBU(P)	2009	2011
RABE(P)	2009	2011
DRRA(C)	2009	2011
JUCA(C)	2010	2012

The main effects of the flood were modest increases in width and width:depth ratio in some reaches. Most reaches showed no consistent change. BUTI(C) and DRRA(C) tended to widen (Fig. 6) and increased their width:depth ratios (Fig. 7), while other reaches showed little difference. The results show little change in channel cross-section area, with the exception of BUTI(C) which tended to increase area in the flood period (Fig. 8). Changes in channel area, width, depth, and W:D over the flood period were not significantly different between project reaches and control reaches, based on statistical analysis (Appendix Table CM-4). It is likely that the main effects of the flood were to mobilize sediment, do modest bank erosion, and scour and redeposit bed material without large net change in channel morphology.

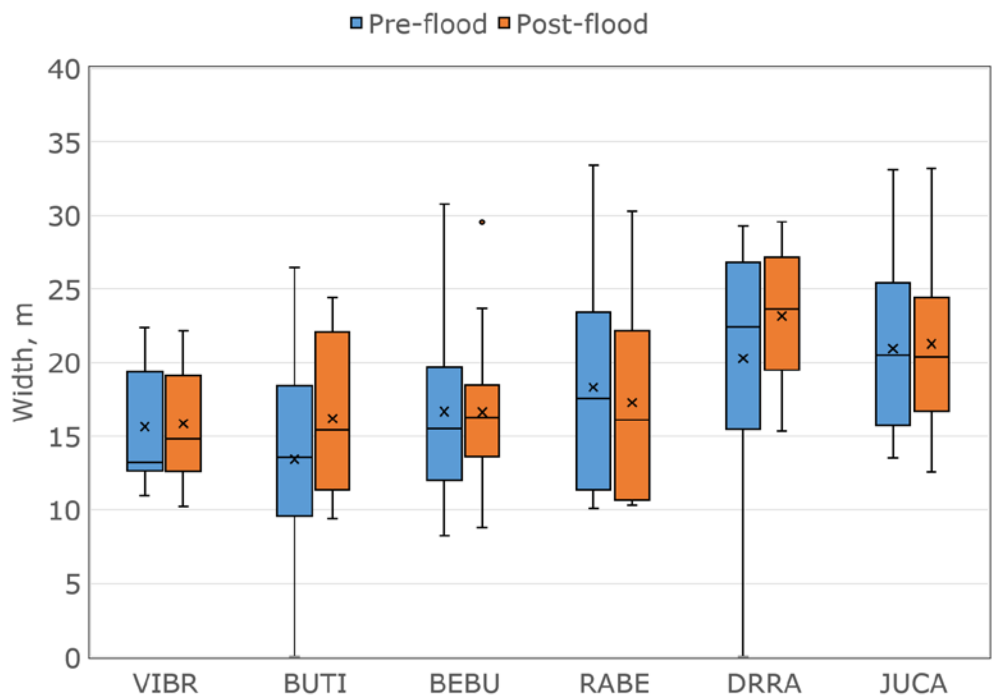

Figure 6. Comparison of pre- and post-flood channel width.

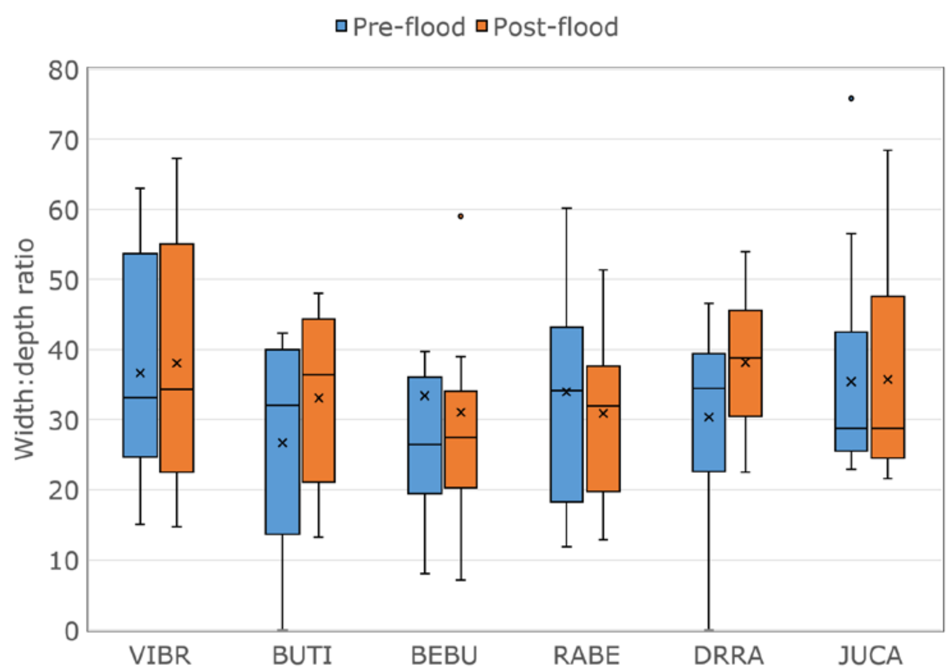

Figure 7. Comparison of pre-and post-flood width:depth ratio

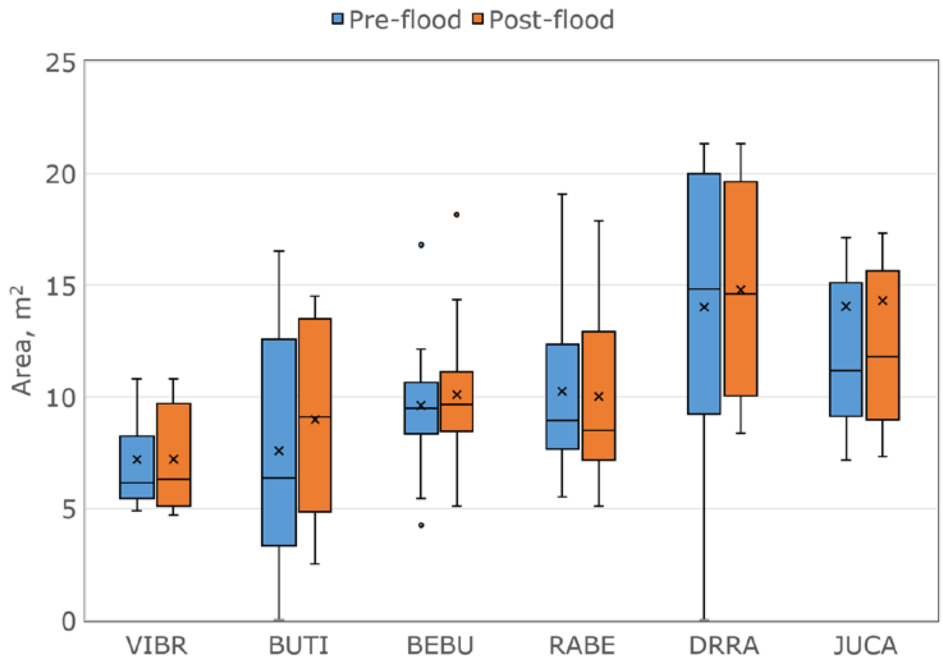

Figure 8. Comparison of pre- and post-flood channel cross-section area.

Analysis and Interpretation of Pools Data

These data provide a before-after restoration comparison for all three project reaches, VIBR(P), BEBU(P) and RABE(P). The VIBR(P) pool survey was completed in late July 2008, just before the VIBR(P) restoration project was constructed. Pool data for JUCA(C) in 2008 are not available at this time. Residual pool depth increased significantly between 2008 and 2015-16 in two of the project reaches, VIBR(P) and BEBU(P) (Fig. 9; Appendix Table PD-1), supporting the hypothesis that restoration will increase pool depth. Pools were dug deeper when log structures we installed, and this strategy appears to have been effective. In the RABE(P) project reach, residual pool depth did not change significantly between 2008 and 2015, although the variability was greater in 2015. Of the two control reaches, BUTI(C) showed no change in pool depth and DRRA(C) showed a decrease.

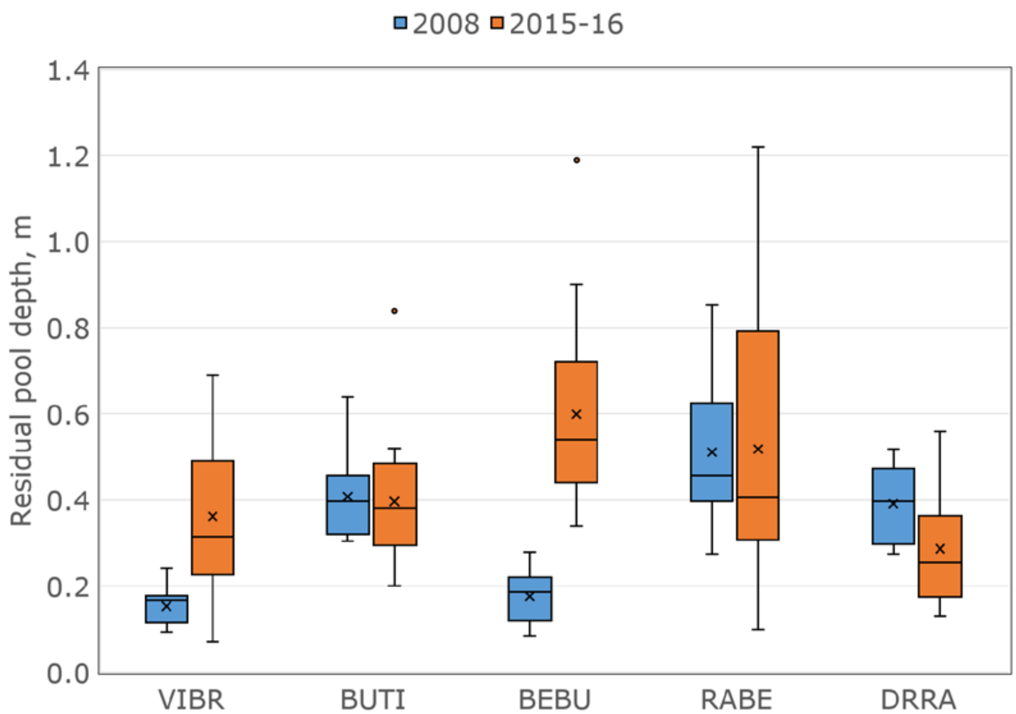

Figure 9. Boxplot of residual pool depth for five reaches, pre-restoration (2008) and after restoration (2015-16).

VIBR(P) and RABE(P) increased pool frequency (pools/km) between 2008 and 2015-16, but BEBU(P) did not (Table 9). BEBU(P) is a more sinuous reach than VIBR(P) or RABE(P), and in 2008 it had a high pool frequency, with good pool development at meander bends. In VIBR(P) and RABE(P), a number of the pools in 2008 were associated with boulders and rock barbs, about 75% in RABE(P) and 50% in VIBR(P). Most of these rock barb pools were also associated with stream bends, so they may not be due to rock barbs alone. The rock barbs were removed in restoration of VIBR(P) and RABE(P). In BEBU(P) in 2008, all the pools were associated with stream bends and only one pool was associated with boulders. The two control reaches, BUTI(C) and DRRA(C), also increased pool frequency. This may be due to reshaping of the channel bed during the 2011 flood.

We also examined the frequency of deep pools. Deep pools are usually defined as those with residual pool depth greater than 1 m, but only four pools meeting this criterion (two in BEBU(P) , one in RABE(P), and one in JUCA(C)) were found. For this analysis, we defined deep pools as those deeper than 0.5m. The frequency of deep pools increased significantly in the project reaches and did not increase in the control reaches (Table 9).

Log structure pools are deeper than the average of all pools in VIBR(P) and RABE(P) (Table 9), indicating that the construction of log structures enhanced pool depth. In BEBU(P) , the overall pool depth is deeper than the average log structure pool depth. This is probably due to the presence of two very deep pools at the downstream end of BEBU(P).

Table 9. Comparison of pool frequency and depth among reaches. Deep pools are those deeper than 0.5m. All depths are given in residual pool depth. LS pools = pools influenced by log structures.

	pools/ km, 2008	pools/ km, 2015-16	deep pools/km, 2008	deep pools/km, 2015-16	mean res. depth, m, 2008	mean res. depth, m, 2015-16	LS pools mean res. depth, m, 2015-16
VIBR(P)	14.3	28.6	0.0	6.7	0.15	0.36	0.41
BUTI(C)	15.7	20.5	2.4	2.4	0.41	0.39	na
BEBU(P)	18.6	16.3	0.0	10.5	0.18	0.62	0.57
RABE(P)	12.8	23.6	4.3	9.6	0.51	0.52	0.72
DRRA(C)	5.7	13.3	1.0	1.0	0.39	0.29	na
JUCA(C)		15.0		3.3		0.40	na

Overall, restoration has increased pool depth, supporting the hypothesis in Table 2. Almost all of the increase is due to digging deeper pools during construction. These deep pools are largely being maintained, with only modest filling since construction.

Analysis and Interpretation of Changes at Log Structures

Structural changes (losses and additions of logs) in the log structures in VIBR(P) and RABE(P) were assessed by comparing aerial photos from 2009 and 2013. Out of 34 total structures examined, two structures were completely removed, one structure was completely re-arranged, and four structures experienced minor changes (Duffin 2015). The two structures removed were mid-channel gravel bars with a log buried at the apex. They were the only two structures of this type constructed; other structures were all located on banks and largely anchored in the banks. Most of the changes in structures occurred during the 2011 flood. The log structures constructed in these two reaches therefore are quite robust.

Log structures act as porous obstructions to flow and therefore are expected to change the hydraulics of flow by reducing velocity in some places and increasing it in others, and by shifting the zone of highest velocity (and scour) either downward under the log structure or laterally beyond the edge of the log structure. The log structures were commonly located at meander bends (12 out of 17 total structures in VIBR(P), 13 of 19 in BEBU(P) and 6 of 15 in RABE(P)). At VIBR(P) and RABE(P) structures, we used changes in bed elevation to assess whether net aggradation or degradation was occurring at log structures. Most log structures displayed some areas of aggradation (increase in bed elevation through deposition) and some areas of decrease in bed elevation (scour). The range of aggradation varied between 0 and 0.2m, and the range of scour varied between 0 and 0.15m, but most values were less than 0.05m. Change is dominated by net aggradation (Fig. 10). Overall, the log structures are producing net bed aggradation (more aggradation than scour), indicating they are acting as flow obstructions.

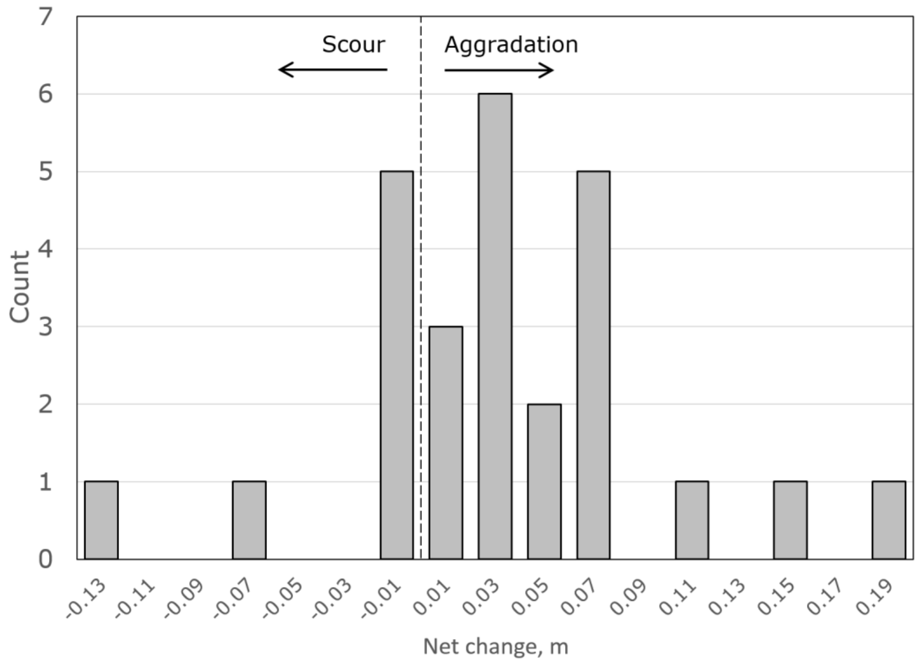

Figure 10. Net change in bed elevation measured at log structures in VIBR(P) and RABE(P).

Many of the log structures were intended to create or deepen existing pools, by forcing scour under the log structure. Existing pools were dug deeper when the structure was constructed. Pool depths are clearly deeper now than before restoration in 2008 (Fig. 11). In VIBR(P) and BEBU(P), virtually all log structure pools are deeper now than in 2008. In RABE(P), more than half the pools are deeper now. In 2008 RABE(P) had relatively deep pools, deeper than in BEBU(P) and VIBR(P); this may explain why there is not a stronger difference between 2008 and 2016. In the first five to six years following construction in RABE(P) and BEBU(P) , the pools have shallowed slightly, typically by 0.01 to 0.20m (Fig. 12). Pools were dug somewhat over deep during construction, and it was anticipated that pools might shallow slightly as flow adjusted to the log structure. This process appears to be occurring. Although pools at log structures are shallowing, the area of individual pools is predominately increasing (Duffin 2015).

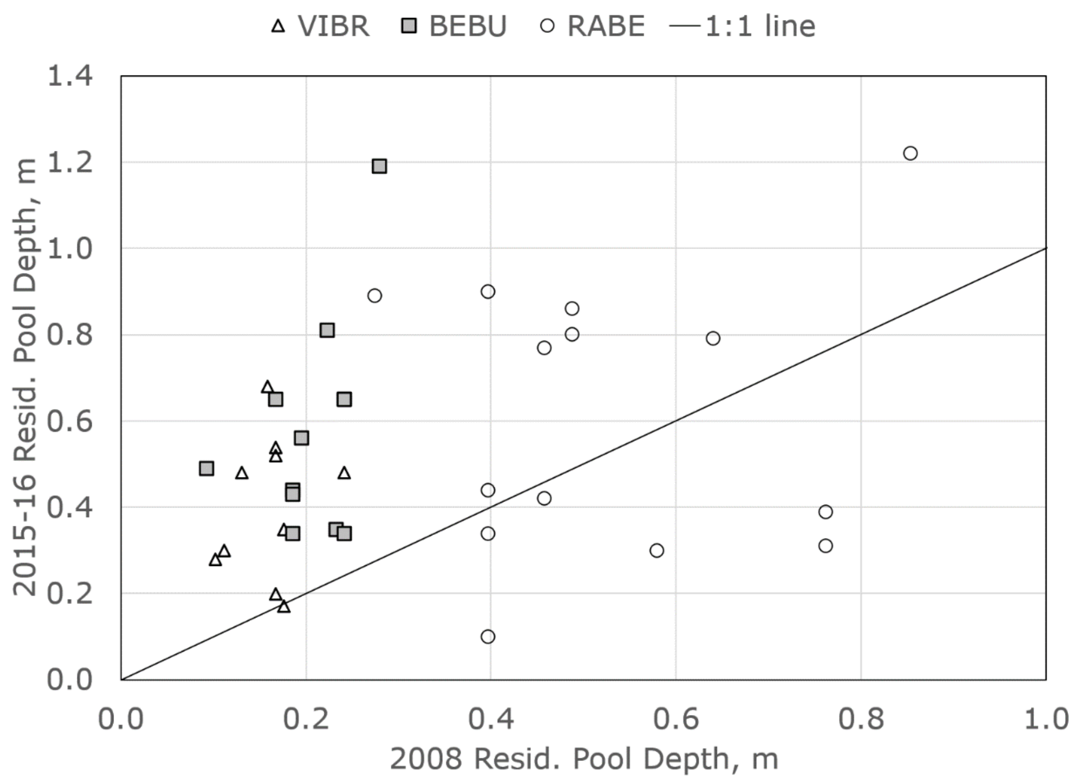

Figure 11. Comparison of residual pool depths before and after restoration, at log structures.

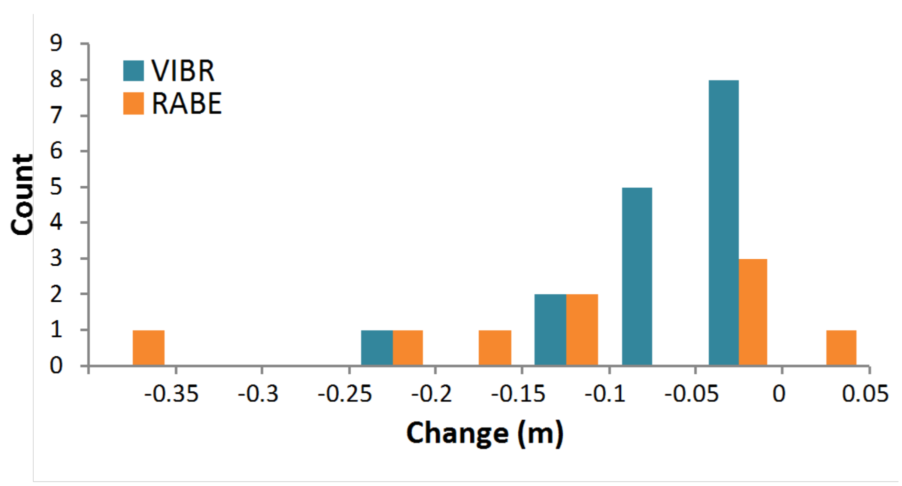

Figure 12. Change in residual pool depth at log structures over the period of study.

We qualitatively assessed direction of change in eight log structures in BEBU(P) using the repeat cross-sections (Appendix Table LS-2). Changes in BEBU(P) were generally consistent with changes in RABE(P) and VIBR(P). The dominant pattern of change in BEBU(P) was some erosion/retreat of vertical banks under the log structure (the bank in which the structure was installed), and aggradation on the bar opposite the log structure. Although some log structures showed slight aggradation in the deepest part of the bed, this response was not very consistent across the BEBU(P) structures.

Overall, the log structures have been stable and have performed well. Structures in VIBR(P) and RABE(P) survived the 2011 flood in good condition, with the except of two mid-channel bar structures in RABE(P). Construction of the log structures produced significantly deeper pools than before. Since construction, some of the pools have shallowed slightly but are still deeper than pre-structure. Some structures display slight bank erosion and bar aggradation, which indicates that normal lateral migration processes are still operating, despite the potential stabilizing effect of the structures.

Analysis and Interpretation of Channel Planform Data

Sinuosity adjustment in river channels typically occurs more slowly than adjustments in channel cross-sections and in bed material (Knighton 1998). Therefore we did not expect large sinuosity changes in the MFJD. In two of the project reaches, VIBR(P) and RABE(P), banks had previously (probably in the 1970s) been hardened with rock barbs and rip-rap placed on banks, in an effort to reduce bank erosion. Most of the barbs and rock material was located on the outside banks of meander bends. As part of the restoration projects in these two reaches, almost all of this artificial rock material was removed so that over the long term channel migration could occur. Log structures were installed on most of the outside meander banks, to provide fish habitat in the short term (20 years or so), with the idea that natural channel migration would occur as these log structures decay. In the short term, the log structures seemed likely to prevent migration of meander bends and increase of sinuosity through this mechanism. However, results for BEBU described above show that some bank erosion is still occurring under the log structures.

We measured sinuosity changes in this study to determine current rates of migration and trends in sinuosity. Measured sinuosity increased slightly in all reaches over the period of study (Table 10). The changes are small, and they may be due to differences in the resolution of the imagery. It is notable that reaches with higher initial sinuosity (VIBR(P), BEBU(P)) showed the largest increases in sinuosity. In RABE(P), channel length was increased by opening and reshaping an inactive side channel. This side channel now carries flow throughout the summer. The side channel increased the channel length in RABE(P) by about 140m, producing an increase in total sinuosity (primary channel plus side channel length) of 15%. The three project reaches all experienced more sinuosity increase than the control reaches. The increase was accomplished by natural channel adjustments in VIBR(P) and BEBU(P) , and by additional of a side channel in RABE(P). Compared to BEBU(P) and VIBR(P), RABE(P) is a straight, relatively steep, more confined reach, so its ability to increase sinuosity of the main channel is limited.

Table 10. Sinuosity changes 2006-2013

Reach	Reach type	Sinuosity (2006)	Sinuosity % change	Secondary channel added, m (% increase)
VIBR(P)	Project	1.20	1.90	
BUTI(C)	Control	1.32	0.93	
BEBU(P)	Project	1.46	1.21	
RABE(P)	Project	1.08	0.08	139 (15%)
DRRA(C)	Control	1.14	1.19	
JUCA(C)	Control	1.07	0.24	

We also examined channel lateral migration. Much of the channel did not show any lateral migration, but 21 sites of migration were detected over five of the six reaches (all except RABE(P)). Data on the migration points are summarized in Table 11. The length of channel experience lateral migration is very small. The rates of migration are moderate, around 0.1m/year. Two of the project reaches, VIBR(P) and BEBU(P), have the highest lateral migration rates, but it is difficult to attribute this effect to restoration. Sinuosity is a relatively stable geomorphic characteristic, and it changes slowly except where directly modified by constructing new channel. Also, in both reaches the restoration actions are more likely to stabilize sinuosity in the short term (10-20 years) than to increase it, because log structures were commonly placed on meander bends where they would inhibit expansion of the meanders.

Table 11. Summary of all areas of lateral migration between 2006 and 2011, by reach. Data include only those sections of channel where lateral migration was observed.

Reach	# Sites of Change	Average Area of Change (m^2)	Average Annual Lateral Migration (m/year)	% of reach exhibiting lateral migration
VIBR(P) (project)	3	4.71	0.129	1.183
BUTI(C)	5	4.58	0.081	1.791
BEBU(P)	5	18.25	0.158	1.866
RABE(P) (Project)	0	--	--	--
DRRA(C)	2	1.20	0.060	0.226
JUCA(C)	6	3.77	0.072	3.111
Overall	21	7.30	0.101	1.363

The results on sinuosity change and lateral migration indicate that the Middle Fork is adjusting channel planform modestly, but it is not a very laterally active river. Opening the side channel in RABE(P) had a direct positive effect on sinuosity and habitat availability.

Analysis and Interpretation of Bed Material Data

Based on the surface pebble counts on riffles, all of the reaches are dominated by gravel-cobble bed material, with typically 40 to 70% gravel, 25 to 55% cobbles, the median (D50) between 40 and 60 mm (coarse to very coarse gravels) (Fig. 13), and the 95th percentile (D95) typically between 100 and 250mm (cobbles). About half the samples have at least one boulder. (Bed material characteristics are summarized in the appendix.)

The content of fine sediment grains (sand, silt and clay, <2mm diameter) is quite low in all reaches, generally less than 5% of total bed material (Fig. 14). Along with low content of fines, embeddedness of gravels is also low. Data from the first year of measurement show the percentage of embedded grains was typically 5 to 25% (Fig. 15), and embeddedness ratios (ratio of total particle height to embedded height) typically 2 or higher.

In most gravel-bed rivers, the bed material size becomes finer going downstream; the MFJD does not have a trend of downstream-fining in the study area (Fig. 13). The lack of downstream-fining indicates the influence of local sediment sources from tributaries and adjacent hillslopes, and the relatively small variation in reach gradient within the study area. RABE(P), the steepest reach, has the coarsest D50 values, and BUTI(C) and BEBU(P), the lowest-gradient reaches, have the finest D50 (Fig. 13). Under initial conditions (first year of measurement in each reach), there is no systematic difference between the project and control reaches in the sediment characteristics of the coarse sediment (See Appendix, Tables BM-1 and BM-3).

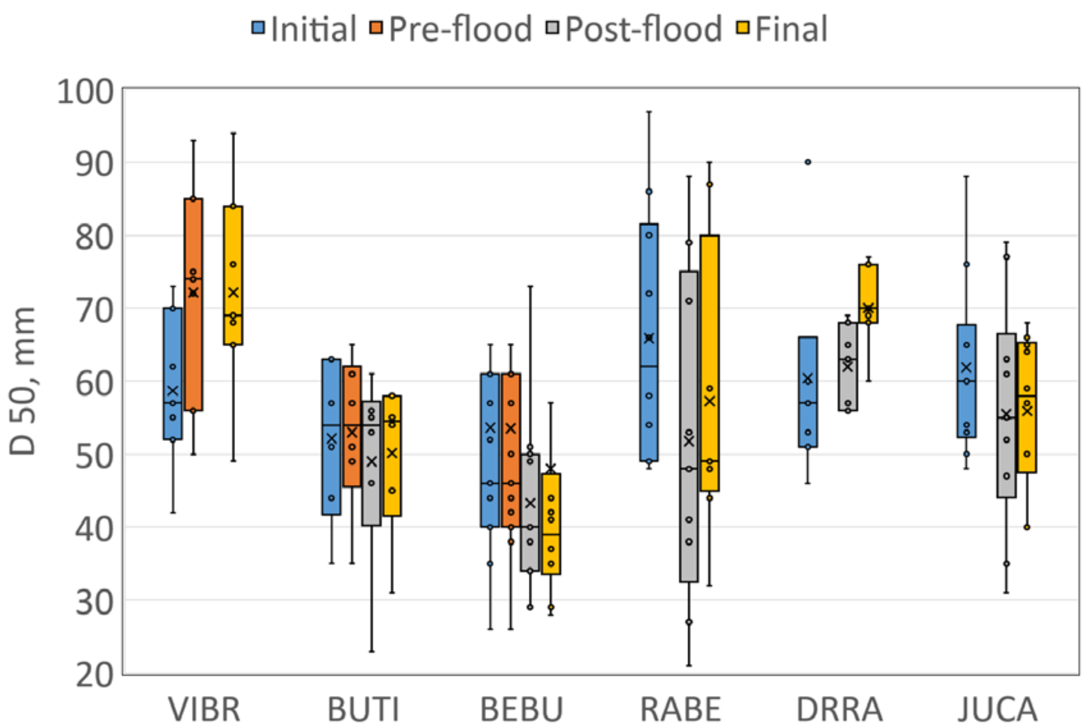

Figure 13. Changes in D50 over the period of study. Initial shows the initial year of measurement, which occurred in 2008 to 2010 in different reaches. Pre-flood shows values for reaches where a second measurement was completed before 2011. Post-flood shows measurements taken in 2011 or later and before the final measurements. Final shows measurements taken in 2015 to 2017 in different reaches. Data based on surface pebble count method.

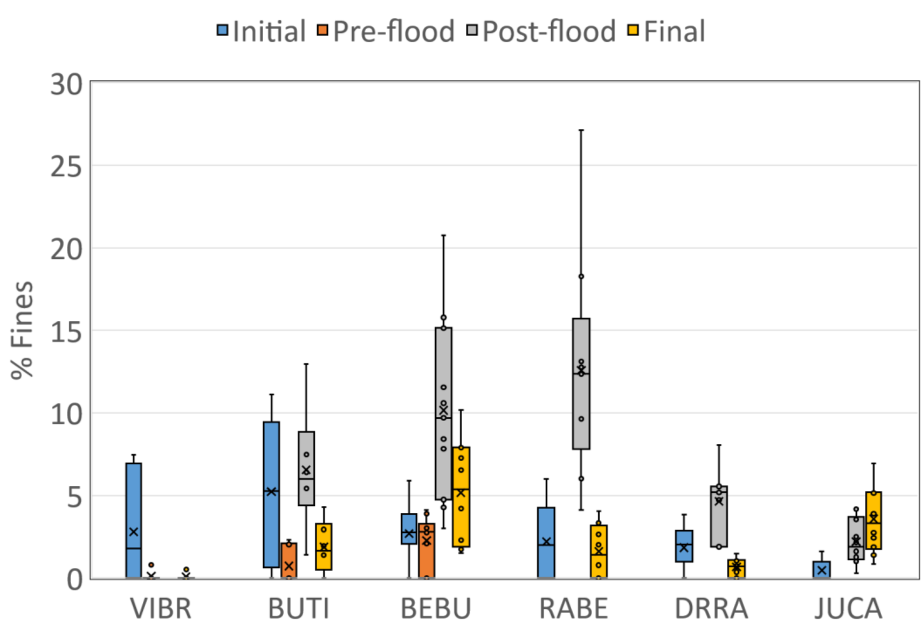

Figure 14. Changes in percent fines (finer than 2mm) over the period of study. Data based on surface pebble count method.

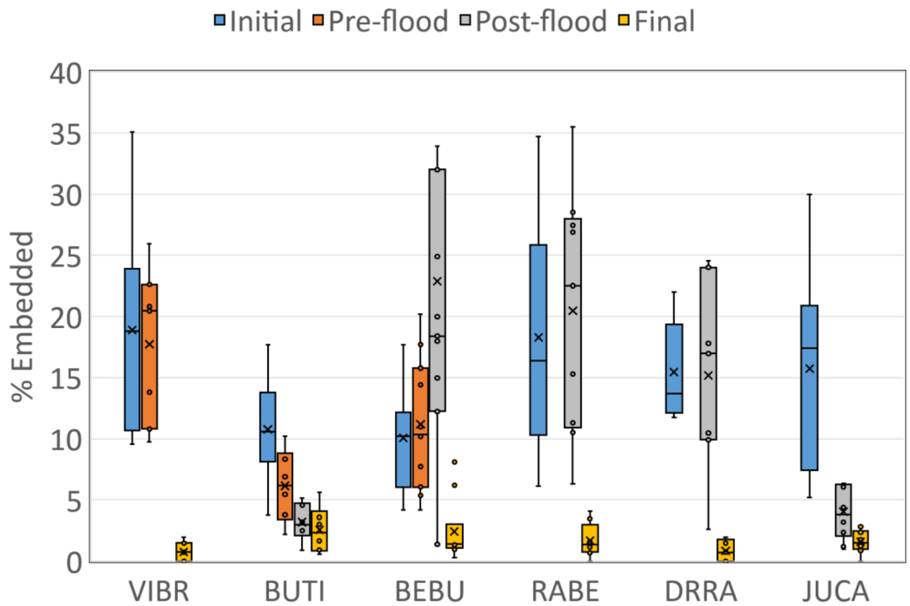

Figure 15. Changes in percent of grains embedded, over the period of study. Data based on surface pebble count method.

We examined the data for changes in both the gravel-cobble distributions, and the fines and embedded grains, over two time periods, the whole period of study and before and after the 2011 large flood. Over the period of study, median grain sizes showed no significant change in most reaches, with the exception of slight increases in VIBR(P) and DRRA(C) (Appendix Table BM-4). The coarse fractions (D84 and D95) showed a similar pattern. In contrast, there is a clear decrease in percentage of embedded grains at all sites (Fig. 15; Appendix Table BM-5). This is one of the strongest responses we observed in all of the geomorphic and physical habitat analysis. All reaches decreased in percentage of embedded grains between the initial and final measurements, and final measurements are typically less than 5%. The percentage of fines decreased at two reaches (VIBR(P) and DRRA(C)) but increased in JUCA(C) (Fig. 15; Appendix Table BM-5). This indicates that the decrease in embedded grains is a reflection of sediment transport and reworking of the channel bed, rather being than driven by an overall decrease in fines. There is no systematic difference in percent embedded and percent fines between project and control reaches, so the changes are not due directly to management. The embeddedness ratio (reflecting the depth of embeddedness) did not show any trend (Appendix Table BM-5).

The low fines content and embeddedness indicates generally good bed conditions for salmonids, and the trend is generally improving over time. The large flood in spring 2011 potentially changed bed material and embeddedness, but our data do not indicate the flood is the cause of changes in embeddedness (Fig. 14, 15). Two reaches, BUTI(C) and JUCA(C), appear to have decreased in % embedded during the flood, but most reaches did not show much change during the flood. Much of the decrease in % embedded appears to have happened since 2012. Therefore it must be due to flushing of fines during high flow events in the years after the 2011 flood. Changes in percent fines are erratic over time.

In addition to surface gravel counts, we also completed two to seven volumetric samples per reach (Appendix Table BM-2). Percent fines content in volumetric samples is typically higher than in surface pebble count samples because large gravels hide fines in surface pebble counts (Bunte and Abt 2001), and this is shown in our data. Very little suspendable fine sediment (very fine sand and finer) was flushed from the gravels during volumetric gravel field sampling, consistent with our data from surface gravel counts. About 60% of the volumetric gravel sites are armored (D50 surface/D50 subsurface > 1.5). All reaches have some armored sites and some unarmored sites. This suggests that the reaches are not significantly sediment-supply limited.

Analysis and Interpretation of Fish Cover Data

We directly assessed the fish cover area contributed by log structures and by torrent sedge (passive restoration) by measuring area using GIS. These data represent all the wetted channel area in the reach (Table 12). Log structures cover about 5-6% of each reach. Torrent sedge coverage is higher, 8-12%. There is a difference between cover provided by torrent sedge and log structures, however. Log structures cover pools and provide cover in deep water. Water depths under torrent sedge canopy at the edge of the channel are generally shallower (0.1-0.3m), although depths of 0.5-0.7m are observed under many mid-channel torrent sedges.

Table 12. Fish cover in three reaches from log structures and torrent sedge in 2013. Values are percentages of total summer water area in the channel in the reach.

Reach	Reach type	*C. nudata* % cover	Log structures % cover
RABE(P)	Project	7.8	4.8
BEBU(P)	Project	12.1	6.0
BUTI(C)	Control	10.1	0

In addition, we monitored fish cover at plots within each reach. These plots were stratified by channel unit type and did not necessarily capture log structures. In the initial year of measurement, four of the reaches had about 5 to 15% total fish cover area, and two reaches, VIBR(P) and BEBU(P) had significantly higher fish cover (Fig. 16). In VIBR(P) the most important type of fish cover was aquatic macrophytes, while in BEBU(P) both aquatic macrophytes and overhanging vegetation were important. Most of the aquatic macrophytes recorded in all reaches were emergent plants, although submerged macrophytes were present to some extent in most reaches. Three cover types, aquatic macrophytes, overhanging vegetation, and boulders, accounted for most fish cover. The remaining types (not shown on Fig. 16) each had less than 2% fish cover area. Two reaches, VIBR(P) and BEBU(P), were relatively high in aquatic macrophytes (mainly sedges and reeds growing at the edge of the channel). In the other four reaches, aquatic macrophytes, overhanging vegetation and boulders more equally contributed to fish cover.

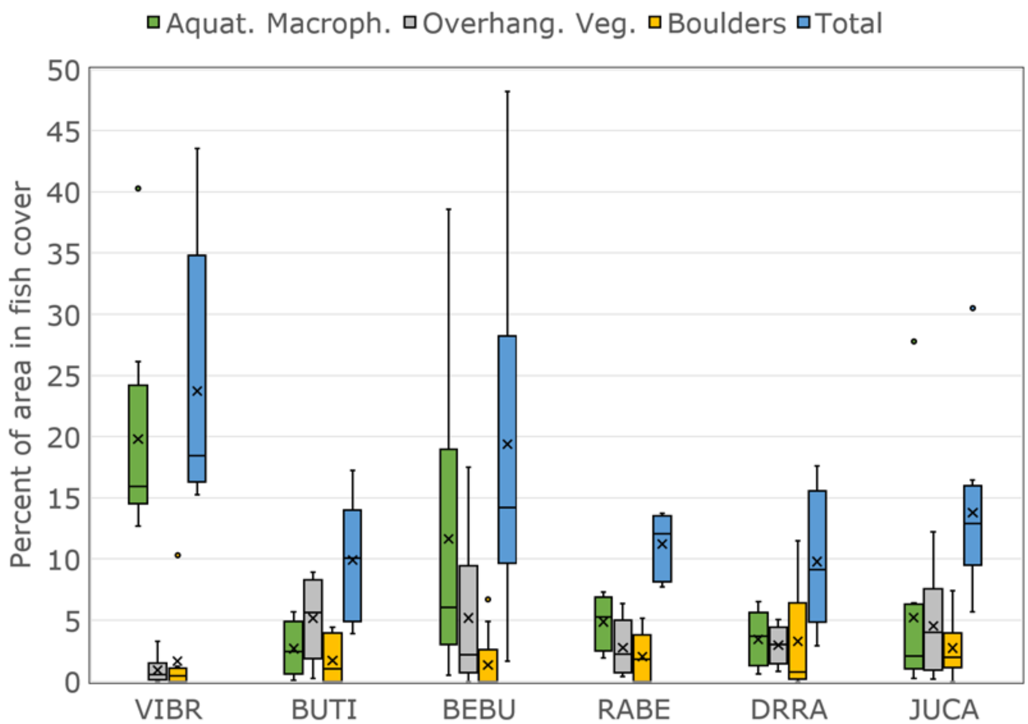

Figure 16. Comparison of major fish cover types in initial year of measurement.

Change in fish cover over the period of study was relatively modest. Total cover generally increased in three reaches, BUTI(C), BEBU(P) , and RABE(P), decreased in one reach, VIBR(P), and remained about the same in the other two reaches (Fig. 17). But these changes were not statistically significant (Appendix Table FC-1). The increases in BUTI(C), BEBU(P) and RABE(P), and the decrease in VIBR(P) were mainly due to changes in aquatic macrophytes. VIBR(P) is the only reach that does not have torrent sedge. Field photos indicate abundant emergent sedges and reeds, so probably the decrease is due to these plant types rather than submerged aquatics. Aquatic macrophytes tended to increase across all reaches except VIBR(P), but it is not clear which type of aquatic macrophytes are driving this change. Aquatic macrophytes have been little studied in this region, and we do not know what factors are likely to be driving changes in their abundance. In general, changes in fish cover were not statistically significant, except for increase in aquatic macrophyte cover in DRRA(C) and JUCA(C), and a decrease in overhanging vegetation cover in JUCA(C) (Appendix Table FC-1).

The hypotheses about increased fish cover from active restoration (log structure cover) and passive restoration (overhanging vegetation cover) are supported by our measurements from aerial imagery. The field plot sampling of fish cover does not support these hypotheses in terms of statistical results. This is probably due to the more complete measurement done from aerial imagery, in contrast to the smaller portion of the reach that is sampled in fish cover plots.

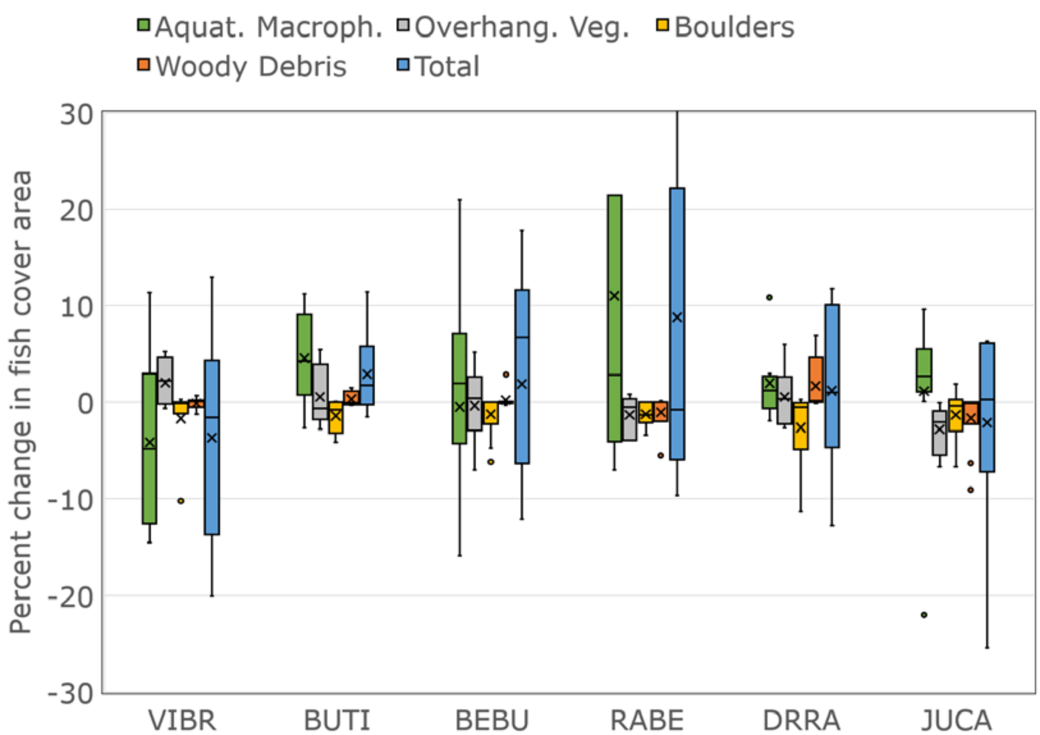

Figure 17. Change in fish cover from initial year to final year. Woody debris includes both large and small debris.

Discussion

Although the restoration projects in the MFJD are thought of as primarily active restoration projects, passive restoration has been very important in this river. Passive restoration has encouraged growth of non-woody and shrubby vegetation within the bankfull channel. This vegetation appears to be playing an influential role in post-restoration response in the study area.

Our data do not support most of the hypotheses about change in channel dimensions in response to restoration. In BEBU(P) (before and after restoration), the channel did not narrow or deepen as a result of restoration. The channel deepened at pools under log structures, but not at other cross-sections. In other reaches, we did not observe a different response in channel dimensions between control and project reaches, and the project reaches did not consistently change in the direction hypothesized. The lack of response may be due to inadequate time to observe a response, or to inadequate density of monitoring.

Restoration was more successful in affecting pool characteristics than channel dimensions. In general, restoration increased residual pool depth, pool frequency, and frequency of deep pools. This response is due to the direct effects of digging deeper pools when the log structures were constructed. The deepest pools are associated with log structures. Since restoration, deep pools at log structures have been maintained in general, with only modest aggradation. This suggests that the pool strategy is successful and is likely to persist for the near future.

There was very little change in sinuosity over the period of study. This is not surprising, since sinuosity is a slow-changing property of river channels. Reaches with highest sinuosity were able to increase their sinuosity slightly more over the period of study than lower-sinuosity reaches.

Bed material in the MFJD is in relatively good condition, with a low percent fines, and low gravel embeddedness. One of the strongest trends we observed is a decrease in gravel embeddedness in project reaches over the period of study. The mechanisms causing this change are not clear. It may be due partly to changes in hydraulics and mobilization of gravels, and/or to vegetation effects.

Fish cover measured at plots is provided mainly by emergent and submerged aquatic plants and overhanging terrestrial vegetation. Woody debris contributes only small amounts, but there is the potential for this to increase over a period of decades. Fish cover increased modestly in two out of three project reaches and showed little change in the control reaches over the period of study. Thus, the hypothesis for increased fish cover from restoration is weakly supported. The mechanisms for this positive response are primarily vegetation driven, attributable to the increase on non-woody and shrubby vegetation within the active channel. This the response is due mainly to passive rather than active restoration. Both log structures (active restoration) and torrent sedge (passive restoration) contribute substantially to fish cover when the reach as a whole is measured.

Most of the observed changes in channel morphology and bed materials in both project and control reaches over the period of study were modest. The 2011 flood, one of the largest floods in the historical record for the MFJD, did not cause significant net erosion or deposition in either project or control reaches. This indicates that the river channel is relatively stable and in dynamic equilibrium. This is a healthy state (as opposed to a river system that is in disequilibrium or near a major threshold of change). It also suggests that the MFJD is a relatively robust system. Resistance to change in the MFJD may be due to several factors, including coarse bed materials for its size, relatively low channel gradients (less than 1% slope), and modest stream power.

The research design and goals for PIBO monitoring were quite different from those of this study. In particular, the PIBO monitoring was intended to document watershed-scale effects, as opposed to effects of specific projects, and to compare the state of the watershed to reference reaches that have not experienced any significant management impacts. However, the PIBO monitoring on the mainstem MFJD measured some indicators that are equivalent or similar to those we monitored. PIBO values reported for D50, percent fines, and residual pool depth are generally consistent with our values. The PIBO monitoring found a slight but not significant decrease in D50 over time; we found no trend in D50 over time. PIBO found an increase in percent fines over time, while we found a decrease at three reaches and an increase at two reaches. If our results from all six reaches are pooled, they also show no net change in percent fines. PIBO found an increase in residual pool depth over time. This result is consistent with our results from project reaches, but not with our results from control reaches. Comparison of the results from our study and PIBO monitoring indicates that PIBO monitoring is most valuable for comparing watershed-scale trends with reference reaches, while our approach provides information on specific processes and mechanisms of response to restoration. The PIBO monitoring reported a 17% improvement in the habitat index score for the mainstem MFJD. Our results suggest a more muted level of success of the restoration projects.

Lessons Learned

Passive restoration actions are very important in improving aquatic habitat and in understanding how active restoration improves habitat. In this study passive restoration had been implemented well before active restoration, and passive restoration has had important effects on channel geomorphology and habitat. Passive restoration in this region mainly occurs through removal or reduction of livestock grazing, and vegetation within the channel and in the nearby riparian zone is the main driver of response. It is important to think through potential processes and effects of vegetation change while designing an active restoration project and a monitoring project.

Develop a long-term restoration plan before designing the monitoring plan. The monitoring plan designed at beginning of the study was influenced by new restoration projects during the course of the study that were not anticipated. For example, BEBU(P) started out as a control reach, but became a project reach in 2013. From the beginning, we were not able to establish a before-after monitoring study. In our first year, the first restoration project on the CTWS property was installed. We did not have advance information about where new projects were likely to be built or when they would be built. While we were not able to collect one year of data before most restoration projects, a robust before-after monitoring design would involve several years of monitoring before restoration because some monitoring indicators may be affected by flow levels, time since a significant high flow event, and other factors. Thus a robust before-after-control-impact monitoring program would require substantially more planning and more investment of time and money than was available in the MFJD.

Control-impact analysis with a before-after component can produce useful monitoring results. Despite the fact that our project was primarily a control-impact design, our data did yield important insights into the effectiveness of restoration actions. While a few restoration actions had effects immediately, right after construction (such as pool depth and fish cover due to log structures), most of the expected restoration effects were expected to develop over a period of a few years following restoration, as flow and sediment transport reshaped the channel, and vegetation recovery contributed habitat. We were able to report on these effects by following trends in channel change over a six to seven year period after restoration.

Develop in advance a plan for monitoring if a large flow event occurs. The flood of 2011 was unanticipated. Fortunately we had quite a bit of monitoring from one to two years before the event. In the summer immediately following the flood, we were able to monitor more than half our reaches. It would have been useful to determine in advance what the priorities were for monitoring after a big flood event, and to be able to tap additional funds for extra monitoring following a large flood.

Use remote sensing data to complement field measurements. In the course of this project, we experiments with high-resolution aerial imagery from helicopters and unmanned aerial vehicles, high-resolution LiDAR elevation data, and high-resolution elevation data derived photogrammetrically from structure-from-motion analysis of aerial images (Dietrich, 2016). These tools greatly enhanced our collection of field data. They also allowed extraction of measurements directly from data sets without field measurement, at many more sample points than we could measure in the field. These are new technologies just beginning to be applied to restoration planning, design and monitoring. Our results indicate that they should continue to be applied and developed.

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Appendix – Geomorphology and Physical Habitat Field and Analysis Methods

Cross-Section Survey

By P. McDowell and M. Goslin, University of Oregon

This method is based on Harrelson et al (1994). In most cases cross-sections were surveyed using an auto-level and fiberglass tape, but in some cases we used a total station or RTK GPS instrument. These instructions are for using the auto-level. All measurements were in meters, to the nearest 0.01m.

All cross-sections were surveyed between permanent monuments (3/8” rebar 0.6 – 1m long driven into the floodplain so that 1-3cm were exposed above the ground. Monuments were installed the first year of survey and relocated in subsequent years using GPS and a metal detector. Left bank is on your left looking downstream. Zero meters on the cross-section survey is at the left bank monument.

1. Set up a tape (30-m or 50-m) on the cross-section line. The 0-m mark on the cross-section tape is on the left bank (left side when looking downstream). Pin the 0-m end immediately beside the LB monument. Stretch the tape across the channel. At the RB monument, pull the tape as horizontal as you reasonably can without putting too much stress on the tape. Pin it at the RB monument by locking the tape handle and pinning it to the ground through the handle. It will not be perfectly horizontal but should have relatively little sag.
2. Identify bankfull stage on the cross-section line. (See the instructions for identifying bankfull stage.) You may find more than one point that you think might be bankfull – that is OK. It's good to identify bankfull stage on both the left and right banks; the elevations won't be exactly the same, but this will help to identify bankfull stage when the data are plotted. Note down the criteria/reason for identifying this as bankfull stage. Stick a pin flag in the ground at the bankfull point. Be sure to include this pin flag(s) in the photos that will be taken later. If you see two or three possible bankfull points, you should survey all of them and record the reason for each one.
3. Set up the auto-level, using the auto-level instructions. Set it up off the cross-section tape line, but in a position where you can see the whole cross-section easily. It should be at least 3m away from the cross-section line at the nearest point.
    a. In some cases you will not be able to see the whole cross-section tape from one instrument station. In these cases, you will need to move the auto-level to a second instrument station, using the turning point method or re-shooting the LB and RB monuments.

4. There is no need to shoot a back sight unless it is a new cross-section not surveyed in previous years. The cross-sections should all be controlled by a common datum so elevation from one cross-section to another can be calculated. To establish the true height of the instrument in m above sea level, we will use the monuments elevation controls. You will survey the top of each monument, and we will use these points for the calculations.
    a. If the cross-section is being surveyed for the first time, it is necessary to establish the elevation of both monuments. This can be by (1) shooting a back sight with the auto-level to a control point; (2) surveying with the total station; or (3) surveying the monuments with the survey GPS.
5. Survey along the cross-section tape. You can start from either end, although usually you start at the zero end. If using an auto-level, the cross-section tape is used for horizontal distance and vertical elevation is read using the auto-level. (If using a total station, survey along the cross-section tape and record the tape location for each surveyed point, although the total station will read both horizontal location and vertical elevation.)
6. Survey along the tape, recording rod reading and tape distance for each point. Begin and end with readings on top of the left and right bank monuments as well as on the ground immediately adjacent to the monuments. Survey each inflection point, and each change in sediment or vegetation characteristics (see list below). Note these changes in bed material (i.e. from what to what, be specific about the substrate change).

Don't survey every meter (i.e. at fixed, regular intervals). The points will be irregularly spaced and located on inflection points (or points of substrate change) to capture the morphology of the cross-section accurately. On floodplain surfaces where there is little change in elevations over long distances (e.g. anywhere from 1-20m), you may want to simply take an additional measurement or two in between inflection points or vegetation change points These measurements may be widely spaced. When surveying the channel bed, you will want a higher density of measurements. In addition to inflection points and points of substrate change, you may need to survey points in between these change points with the goal of having spacing between measurements around .5 – 1 m.

7. As you survey, record in the notes column what surface you are on (floodplain, swale on floodplain, bank, bar, wetted channel, etc.). You don’t need to record the surface for every point in a series of points on the same surface, but record when you move onto a different surface. Record a note for all significant points or changes (such as those listed

in the list below). (See table below and data dictionary for all acceptable abbreviations)

8. When you reach the wetted channel, survey a point at the water edge. Continue surveying the submerged channel bed. For every submerged point, also read water depth (off the rod) and record it. While in the wetted channel, if the bed sediment changes from sand to gravel, survey that point.
9. The surveyed points should include the top edge of the bank, the bottom edge of the bank, the edges and top surface of any bars in the channel, the thalweg, the floodplain surface, and any other topographic breaks you notice. Be sure to measure at the left and right water edges.
   a. If you encounter a swale or dry side channel, note whether it is connected to the main channel at either the upstream or downstream ends, or both. Is it likely to carry water (through flowing) from the main channel when the main channel is below bankfull stage?
10. Be sure to survey the previously identified bankfull stage points.
11. When torrent sedge (*Carex nudata*) patches are encountered, we will typically take a point at the edge of the patch (typically water's edge) and at the top of the tussock mound.
12. Survey all of the following points. Some codes to use in the notes column are given in parentheses.

Features to be surveyed	**Suggested code for notes column**
LB monument top, RB monument top	"lb mon", "rb mon"
ground surface immediately adjacent to LB and RB monuments	"ground at lb mon, "ground at rb mon"
top edge of any bank	top of bank
bottom of any near vertical bank	bottom of bank
edge of permanent vegetation (continuous sod, as opposed to sparse pioneer vegetation found on bar surfaces	Edge of grass, edge of willows, etc.
Left and right water edges	"lwe", "rwe"
edges and tops of bars	bar edge, bar top
thalweg	thalweg
At least a couple of points on the floodplain surface on right and left sides of the channel	"fp"
Swales or dry channel on the floodplain	Top of swale, bottom edge of swale, center of swale, etc.
Bankfull point	"lbf", "rbf"

13. Photos: The purposes of the photos are (1) to record the position of the XS tape and monuments so they can be relocated in subsequent years, and (2) to record the form and roughness conditions of the cross-section (for example, to estimate Manning's n). Keep these goals in mind when taking photos and make sure the photos will meet these goals. Take photos of the cross-section with the tape and the bankfull flags visible, as follows: looking down XS tape from RB and from LB; upstream of the XS on one bank looking down toward it; downstream of the XS on one bank looking upstream toward it. IN the field notebook or data form, record each photo number and from what position is taken (for example, LB looking downstream at XS, RB looking across at XS, etc.). If the cross-section monument is obscured by tall grass, take additional photos to show each monument (flagged) with surroundings or background that would be useful in finding the monument in future years.

Reference

Harrelson, C. C., Rawlins, C. L., Potyondy, J. P., 1994. Stream Channel Reference Sites: An Illustrated Guide to Field Technique. U. S. Forest Service General Technical Report RM-245.

Cross Section Data Analysis

By A. Zettler-Mann and P. McDowell, University of Oregon

The analysis procedures generally follow the guidelines of Harrelson et al (1994).

1. Bankfull stage was identified for each cross-section, to normalize the cross section area across years of observation. The bankfull elevation did not change over time. To identify bankfull elevation, we used using field observations, geomorphic features visible in the cross section profiles, and the longitudinal slope profile generated using the 2006 LiDAR data.
    a. We had a number of candidate bankfull elevations for each cross-section, and our goal was to select the most reasonable elevation for each cross-section. For each cross-section, several possible bankfull elevations were identified during field survey. In addition, plotting the cross-sections revealed breaks in slope that could also be bankfull elevation.
    b. Each cross section was plotted in Excel along with all bankfull elevations from field observations associated with that cross section. Bankfull elevations that did not at least approximately match the location of geomorphic surfaces in the cross section were eliminated.

c. Initially we selected a bankfull elevation for each cross-section such that the cross section area through the reach remained relatively consistent. This approach produced bankfull elevations varied erratically within the reach, and it was abandoned.

d. We then selected bankfull elevation that produced a relatively consistent water-surface profile for the reach. We produced a longitudinal profile for each reach using the 2006 LiDAR DEM, which probably represents the channel bed or low-flow water surface. We plotted the candidate bankfull elevations for each cross-section on this profile. For each cross-section location, we selected the bankfull elevation that produced a line roughly parallel to the longitudinal profile. Where there were several candidate elevations that were close to the reach-level bankfull surface, we selected the elevation that was most reasonable geomorphically (corresponding to the clearest geomorphic break in slope or surface). We iterated between the profile, cross-section plots and field notes to select the most appropriate choice.

2. After deciding on a bankfull elevation, we found that in some cases the bankfull elevation was higher than the extent of the field survey on one or both banks. For these cases, we extracted cross sections along the field cross-section line using the LiDAR DEM in ArcGIS and the 3D Analyst package. We then extracted elevation data along this cross-section beyond the extent of the field survey, and added these data to the field survey data.
3. We exported the Excel cross-section data for each cross-section to Grapher 8, and used Grapher 8 to calculate cross-section bankfull area for each year.
4. Some cross-sections had a dry swale separate from the main channel. If these swales appeared to have both upstream and downstream connection to the main channel and had evidence of recent flow, we assumed that were part of the bankfull channel and included their area.
5. Width was calculated each cross-section using Excel. We interpolated between survey points to find the tape distance where the bankfull elevation intersected the cross-section on both right and left banks. We subtracted left bank tape distance from right bank tape distance.
6. Average channel depth (m) was calculated by dividing the area (m^2) by the width (m).
7. Width-to-depth ratio was calculated by dividing the width by the depth.
8. The change in area between surveys at a cross-section was calculated by subtracting the earlier year's area from the later year's area. Negative values represent deposition over time, and positive values represent erosion over change.

9. The change in width-to-depth ratio was calculated by subtracting the earlier year's area from the later year's area. Negative values represent an increase in the width-to-depth ratio and positive numbers indicate a decrease in the width-to-depth ratio.
10. We characterized channel change processes by visually examining the plotted cross-sections from different years in Excel.

Reference

Harrelson, C. C., Rawlins, C. L., Potyondy, J. P., 1994. Stream Channel Reference Sites: An Illustrated Guide to Field Technique. U. S. Forest Service General Technical Report RM-245.

Residual Pool Depth Survey

Protocol by: P. McDowell, July 23, 2015

Objective: Measure maximum pool water depth and pool tail water depth to calculate residual pool depth for comparison to 2008 stream habitat inventory data.

Equipment: stadia rod, Trimble GPS unit, field maps, paper form for data, pencils, clipboard

Team: 2 people, one wading in the stream and another on the bank recording and taking GPS points.

1. Start at the downstream end of the reach and work upstream.
2. At the first smooth-water unit (not a riffle), determine whether it is a pool or a glide / run, using the following criteria.
    a. Pools are depressions in the streambed that are concave in profile, laterally and longitudinally.
    b. Pools are bound by a 'head' crest (upstream break in streambed slope) and a 'tail' crest (downstream break in streambed slope).
    c. Only consider main channel and side channel pools, where the thalweg runs through the pool, and not backwater pools.
    d. Pools span at least 50% of the wetted channel width at any location within the pool. So a pool that spans 50% of the wetted channel width at one point, but spans <50% elsewhere is a qualifying pool.
    e. If there is a pool-shaped concavity but it is narrower than 50% of wetted channel width, do not include it.
    f. Side channels: measure pools in side channels that are carrying flow during our summer season.
    g. Maximum pool depth is at least 1.5 times the pool tail depth.
    h. If all criteria were not met, it is a Glide / Run, not a pool.

3. If it is a pool, enter a number for it in the numbering sequence. 1 = downstream-most pool of the reach.
4. You may encounter a pool that has a sill (pool tail crest) within it. Is it 2 pools or 1?
   a. The middle sill does not necessarily have to break the water surface, but it should be a distinct rise to a crest that divides two pool low spots.
   b. If the upstream portion has a pool tail that is ≤10cm deeper than the downstream pool tail, split the unit into two pools (each with its own sequence number and depths).
   c. If the upstream pool tail depth is >10cm deeper than the downstream pool tail depth, it is one pool.
5. Measure and record pool-tail depth, to the nearest cm.
   a. PTD is measured at the maximum depth along the pool tail crest, normally but not always the thalweg.
   b. To find this point, imagine that the water in the stream is 'turned off'. You want to measure the depth of the last spot that would have flowing water before the stream stopped flowing.
6. Measure and record maximum pool depth, to the nearest cm.
   a. This is the deepest point in the pool. Locate it by probing the pool with the rod.
   b. Estimate maximum depth if it is unsafe to measure.
7. While the wader is finding and measuring depths, the recorder should take a GPS point on one bank (bank edge), halfway between the downstream and upstream boundaries of the pool.
   a. Follow our usual conventions for GPS file name.
   b. In the GPS point code, record the pool sequence number, i.e., pool 16.
8. Before moving to the next pool, check the data form to make sure all data are filled in.
9. Continue to the next smooth-water unit upstream and determine whether it is a pool. This unit will have the next number in the sequence.
10. Continue with measurements and GPS.
11. Calculate residual pool depth by subtracting tail crest depth from maximum pool depth.

References:

This method is based on the PIBO and AREMP methods.

Heitke, Jeremiah D.; Archer, Eric K.; and Roper, Brett B. 2010. Effectiveness monitoring for streams and riparian areas: sampling protocol for stream channel attributes. Unpublished paper on file at: http://www.fs.fed.us/biology/fishecology/emp

Anon., 2013. Protocol: AREMP 2013 Field Season - Regional Interagency Monitoring for the Northwest Forest Plan v1.0. Method: Channel Morphology: Pools v1.0. https://www.monitoringmethods.org/Protocol/Details/1953

Log Structure Survey

By P. McDowell and D. Tu

The goal is to survey adequately to capture changes in morphology 5-10 years from now, or after the first major flow event. The purpose is to understand how individual structures perform. This kind of survey will help us understand how the structures performed and why, with the goal of informing future designs form this knowledge. We are monitoring individual log structures because the hydraulic response is at the scale of the individual structures (although we hope to see improved characteristics at the reach-level as the cumulative result of individual structures).

1. Surveying will be done with a total station or RTK GPS instrument.
2. Surveying will be irregularly spaced with particular attention paid to geomorphic areas of interest which include the dug pool, log structure and changes in slope. Field surveys using total station and/or survey grade GPS will require set up time. Make sure to take the time to set up your backshot or base station. Record accuracy readings in the notebook.
3. As you survey, each point is to be coded. Below is a summary of the major survey codes used.

Survey Code	Description
Bk	Bank
BT	Bank Top
DP	Dug Pool
WtEd	Wetted Edge
ChBd	Channel Bed

4. Survey an area extending streamwise from about 10-20m upstream of the edge of the log structure to about 10-20m downstream of the log structure, and extending laterally from 3 m onto the floodplain from the bank top to bankfull stage on the opposite bank.
5. If the structure has been surveyed in a previous year, make a map of the previous survey and use this in the field to delineate the boundaries of your survey.

6. To ensure that the survey area is adequately covered, survey systematically along informal transect lines running perpendicular to the centerline of the channel. Set up flag pins on either side of the bank will help you stay on track and in a relative line as you record in-stream survey points. Transects are set up about 1m apart.
7. For general riverbed and sloping surfaces up to bankfull, point spacing should be about 0.5-1m along transect lines. (In practice, our initial year surveys had an interpoint distance of 1 to 3m, and our later surveys had an interpoint distance of 0.3 to 1m.) Determine your natural stride length (walking in the channel), and use our stride to estimate point spacing as you survey. Survey specific features as described below.
8. Get riverbed points to represent the bed outside the dug pool, and the bed near the bank across from the rootwads. Get enough bed points to represent the major aspects of the form. This might means an approximate spacing of 0.7m for riverbed survey points (which do not include dug pool survey points). Bed points will be obvious, they should be coded ChBd. The channel bed points along the water's edge should be coded as WtEd.
9. For geomorphic areas of interest, points should be spaced approximately 0.5m apart.
10. In dug pools (typically under the log structure), space points about 0.5m apart. They can be more dense to capture the complexity of the pool. Get points on channel bed at the lip of the dug pool, enough to represent the shape of the pool adequately. Get several points in the bottom of the pool, being sure to get the deepest point. Code these as dug pool, DP.
11. It is important to delineate the top edge of any steep or vertical bank. Survey of the bank top and base is usually done separately from the transects designed to capture the bed. Record bank top points every 2-5 meters, and code them BT. The points are not evenly spaced; they should be spaced to capture the form. Survey a few points on the bank surface. Be sure to capture the base of the bank (usually underwater), with extra points between transect lines if necessary.

Log Structure Data Analysis Methods

By J. Duffin, 2015

This method is extracted from Duffin (2015).

1. Import field survey data for a site into ArcGIS as a point cloud. Converted the point cloud into a TIN (Triangulated Irregular Network), using break lines along the water surface edge.

2. Interpolated the TIN into a DEM (Digital Elevation Models) with a resolution of 0.1 m x 0.1 m, using nearest neighbor interpolation. Clip the DEM to a channel area polygon that was covered by the surveys for all survey years.
3. Create a difference of DEM (DoD) using the methodology of Milan et al. (2011), through the following steps.
4. Established a linear relationship between the elevation error (survey point elevation minus the interpolated elevation) and the local topographic roughness (standard deviation of elevations within a radius around each point—the radius was determined for each survey based on point density so that no more than about 7 survey points were encompassed).
5. Applied the linear regressions to the map of topographic roughness, which creates a spatial error grid.
6. Calculated the root mean square error (RMSE) by combining the error grids from each year's DEM to create a spatially distributed LoD grid.
7. Subtracted the LoD from a basic DoD to create a thresholded surface with spatially distributed error.
8. The area of the clipped DoDs, referred to as the survey area, is used to normalize the change in sediment volumes to make it comparable among surveys.
9. Extract quantitative information on topographic change from the DoDs and DEMS, including total volume of sediment aggraded and degraded for each survey area, net change in volume, pool area and volume for each survey, changes in pool area and volumes over time.
10. Extract qualitative variables from the DoDs by visual inspection, including the dominant location and direction of change for each log structure, shifts in pool shape or location, and channel narrowing or widening. The location of change was categorized relative to the log structure as downstream, upstream, outboard, under structure, along opposite bank, or along same bank as the structure.

References

Duffin, J.L., 2015. Effects of engineered log jams on channel morphology, Middle Fork of the John Day River, Oregon. M.S. thesis. Department of Geography, University of Oregon.

Milan, D. J., Heritage, G. L., Large, A. R., & Fuller, I. C. (2011). Filtering spatial error from DEMs: Implications for morphological change estimation. Geomorphology, 125(1), 160–171.

Channel Planform Analysis Method

By C. Guerrant and P. McDowell

This method is based directly on Guerrant (2014).

We used imagery from 2006, 2011 and 2013 to analyze changes in channel planform. Imagery is listed in the table below, and sources are listed in the references.

Year	Source	Resolution (pixel size)
2006	Watershed Sciences 2006	0.1524 m
2011	USDA 2012	1 m
2013	Dietrich 2014	0.10 m

Centerlines were digitized by hand from each set of imagery using heads-up digitization, typically at a scale of about 1:1500. Because the centerlines were digitized at different times by different operators, we examined vertex density is each centerline, and we found the vertex density was similar across all three centerlines. Typical vertex density was 50 to 60 vertices per km, although in reaches with more complex channel form the density was between 50 and 100 vertices/km.

To assess what sort of georectification error existed between the sets of imagery, we used the translational error rectification method outlined by Hughes and McDowell (2006). We identified the same ground control points on each set of photos and compared their coordinates. We used the average difference of all of the control points for each reach to determine how much to shift the 2011 and 2013 centerlines to match the 2006 centerline. We found georectification error of between 0.3 and 2.6m; separate georectification errors and corrections were determined for each reach.

To assess migration rates, we used an approach outlined by Urban and Rhoads (2004) and Hughes and McDowell (2006). Each centerline was buffered using a distance of 1.5m, to represent human error in digitization. When centerlines were overlaid, difference in position within the buffered zones is thought of as noise, or false change, while any differences between the buffered areas can be thought of as real change.

With two buffered centerlines overlaid, we drew polygons of the areas of difference, in order to calculate area of change. We divided the area of real change (m^2) by the length of each study reach (m) to get distance of lateral migration ($m^2/m = m$) within each reach. Lastly, we divided this number by the time between the two sets of images to calculate the annual migration rate.

After completing the GIS work, we field checked the areas of lateral migration. At a subset of mapped migration areas, we looked for and described any physical evidence that change had occurred, such as actively eroding banks, undercut banks, recent accretion on the bank opposite the eroding bank, or the presence/absence of established perennial vegetation. The fieldwork supported the evidence of lateral migration based on imagery.

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Dietrich, J.T., 2014. Applications of structure-from-motion photogrammetry to fluvial geomorphology. Doctoral dissertation. Dept. Of Geography, University of Oregon.

Hughes, M., and McDowell, P., 2006. Accuracy assessment of georectified aerial photographs: Implications for measuring lateral channel movement in a GIS. *Geomorphology*. 74.: 1-16.

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Urban, M., and B. Rhoads, 2004. Catastrophic Human-Induced Change in Stream-Channel Planform and Geometry in an Agricultural Watershed, Illinois, USA. *Annals of the Association of American Geographers*. 93.4 (2004): 783-796.

Watershed Sciences, 2006. *Remote Sensing Data Collection: Desolation Creek, Middle Fork John Day River, & John Day River, Oregon.* Portland, Oregon. Data available at: http://www.oregongeology.org/lidar/.

Gravel Count (surface) Method

By P. McDowell and M. Goslin, University of Oregon

This procedure is based on Bunte and Abt (2001) and Harrelson and others (1994). The basic idea is to pick up a gravel at a number of points on a riffle surface, and measure the embeddedness and diameter. We will use a regular grid sampling method in which several tape lines are established across the channel, parallel and regularly spaced, and gravels are measured at regular intervals along these lines. We will aim for counts of 300 gravels, but in some cases we will not be able to count this many. We record gravel size in size classes as <u>coarser than a specific size limit</u>; this is consistent with the approach in the STREAMS (Ohio Dept. of Natural Resources, n.d.) and the USFS Pebble Count Analyzer (Potyondy and Bunte, 2002).

The riffle surface to be measured is the channel bed and bar area of sediment that may be mobile at bankfull flow. At low flow, part of the mobile sediment surface will be submerged, but some part may be dry. Include all submerged area and any adjacent dry area that is not stabilized by vegetation and looks like it experiences active bed sediment transport at higher flows (such as active bar surfaces). Usually we measure to the base of a vertical bank. If measuring on a bar surface, as the bar rises it may become occupied by vegetation cover dense enough that the gravel does not appear to be regularly transported. Place the edge of your measured area here. Patches of rushes and other emergent aquatics that are on submerged areas are included (although the sediment found here is often fines). In the Middle Fork John Day, torrent sedge (*Carex nudata*) is a tough, tussock-forming sedge that grows on the active channel bed. While patches of torrent sedge within the submerged area or an otherwise active bar surface may be found within the overall grid design, sample points that land within torrent sedge root masses or tussocks should not be sampled.

You will take photos and a GPS point before taking down the bank tape (detailed in steps 19-20).

Equipment: Two 30-m or 50-m tapes; 6-10 surveyors pins; gravelometer; plastic ruler in cm and mm; 3-m hand tape; data collector or field form; camera; GPS unit; lawn chair (optional)

Team: A minimum of two people, one to wade the channel and measure, and the other to record data. If three people are available, two can wade and measure at the same time, on separate lines. The recorder needs to be careful to keep the two lines separate.

Setting up the sampling grid

1. Visually examine the riffle and decide on a minimum grid spacing, the distance between sampling points when moving along a cross tape (transect). The minimum spacing should be at least two times the size (B-axis) of the larger clasts present. By larger clasts we mean largest 10% of the population, not the largest single boulder present. The minimum grid spacing should be a round number (10s of cm). A number rounded to the nearest 50 cm is best because it will allow you to easy find the tape point to be sampled but this may not always be feasible in smaller riffles.
    a. Enter the minimum grid spacing as **Spacing_XS** in the **GC_Site** worksheet
2. Measure the channel width of the riffle and determine the number of points that can be sampled per transect, i.e. the number of gravels that can be counted along each cross tape. To determine this number, divide the channel width by the minimum grid spacing and then add one (since there will be a sample point at each end). Once you have determined the number of points along a transect, divide 300 by this number of points, round to the nearest whole number, and you have the number of transects we need to lay out to ensure that 300 gravels are sampled. If the riffle varies in width, measure 2 or 3 widths and calculate an average channel width.
    a. Enter the channel width as **Site_Width** in the **GC_Site** worksheet

3. Next, visually identify the end and beginning of the riffle. Set up a tape along the bank (the bank tape), pinned with surveyor's arrows at both ends, with 0 m at the downstream end of the riffle.
   a. Curved riffles: If the bank is curved, use additional pins to fit the tape smoothly to the curve of the riffle. The bank tape should be set up on the inside bank of the curve. Cross tape points will be evenly spaced on this side and will fan out on the opposite side.
   b. Record the bank ("l" or "r") along which the tape has been laid in **Bank_Long_Tape** in the **GC_Site** worksheet
4. Determine the length of the riffle. Determine the spacing between transects by dividing the riffle length by number of transects. Unless the riffle is very short, this spacing is typically greater than the minimum grid size (spacing between points along the cross tape for each transect). Leave the bank tape in place – you'll use it to set up your cross tapes.
   a. Enter the spacing between transects along the bank tape as **Spacing_Long** in worksheet **GC_Site**
5. Now calculate the total number of gravels you expect to count by multiplying the number of gravels counted per transect (cross tape, step 2) by the number of transects. Make sure you will count more than 300 gravels; if not, try to adjust your spacing to get 300 gravels. If your cross tape point spacing and bank tape spacing are very unequal, you can adjust them up and down (to round numbers) to get a more square grid size, so long as you have more than 300 grid points.
   a. Adjusting to a finer grid spacing: If the minimum grid size is large and the riffle is very short, use the minimum grid size (rounded to the nearest 10 cm) as the spacing on both the cross tapes and the bank tape. This requires more care in finding the sampling points on both tapes. For example the sampling points will be 0.70, 1.40, 2.10, 2.80m, etc., rather than 1.0, 2.0, 3.0, etc. To make it easier, the recorder can calculate all the points in advance and call them out to the measurer.
6. In the worksheet in which the gravel count data will be entered (**Data_GC_***), enter the distances along the bank tape where cross tapes will be located (column: **D_Long_Tape**). You can also fill in the distances along the cross tape for the first couple of transects (**D_XS_Tape**). Having these filled in advance will make it easier for the recorder to keep the gravel counter on track.
7. Set up the first cross tape. It will be pinned at the bank tape and on the opposite bank. Adjust the position on the opposite bank so that it is perpendicular to the channel. This line should be at 0 m on the bank tape (or whatever point you decided to make a regular grid across the

riffle). Record which bank side the 0 m end of the cross tape is staked (**Bank_XS_0** in the **GC_Site** worksheet). Typically this will be on the same bank as the bank tape but occasionally, it will be more practical to stake the 0 m end of the cross tape on the bank opposite the bank tape.

Example: Setting up the Sample Grid

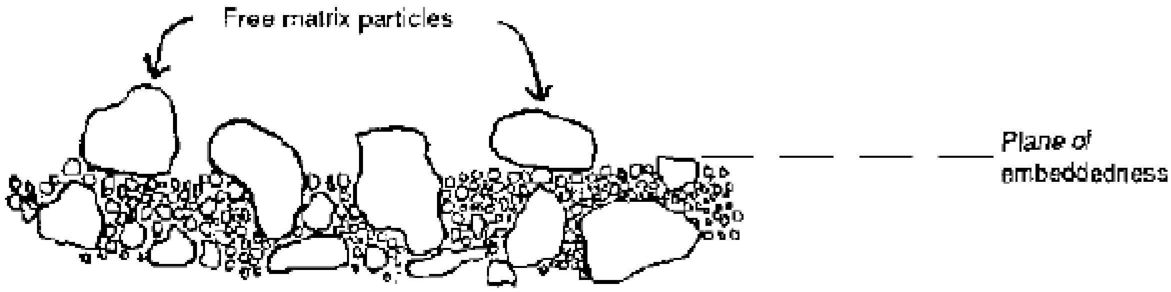
You decide the largest (10%) of the clasts is about 20 cm in the B-axis: use a .5 m minimum grid spacing. The average width of the channel is 5.5 m. Divide 5.5 by .5 m and then add 1. This gives you (11+1 = 12) sampling points along each 5.5 m transect with .5 m spacing along the cross tape. Divide 300 (the minimum needed gravel samples) by 12 (sampling points per transect). The results means that you will need 25 (300/12) transects along the length of the riffle to get 300 gravel samples. The riffle length is 15 m. Divide 15 m by 25 and this yields .6 m, the spacing along the bank ("long") tape. Therefore, sampling grid will have 25 transects with .6 m spacing between these transects along the bank tape and .5 m spacing across each cross tape (transect).

Sampling a gravel

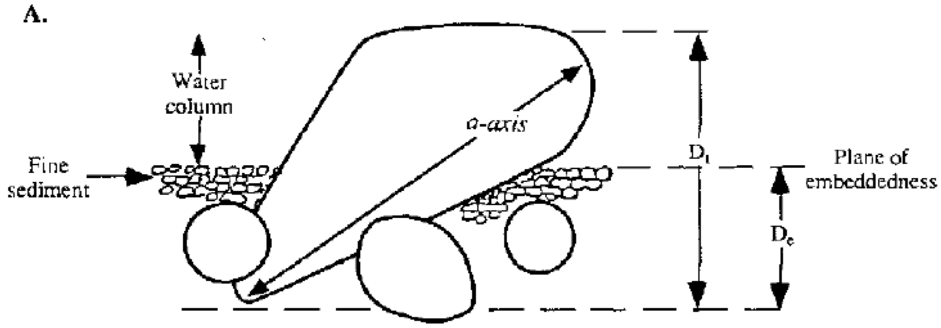
8. Now start sampling. The recorder gets in position on the bank on the lawn chair. The measurer goes to the sampling point on the cross tape line, and pick up the clast directly below the sampling point on the tape <u>without looking at it</u> (to avoid bias in sampling). A good way to do this is to use a surveyor pin. Place your foot so that the front edge of your toe is directly under the point on the tape. With your eyes averted (so as not to bias the sample), slide the pin perpendicularly down toward the bed at the toe of your boot. When the pin touches the bed, slide your finger down and pick up the gravel that the point is touching.

Embeddedness

9. Measure and record embeddedness before measuring gravel size. Embeddedness is measured perpendicular to the plane of embeddedness (the surface of fine sediment grains between gravel particles). Embeddedness (E) is the ratio of the total vertical extent of the particle (D_t) to the vertical extent of the particle <u>below</u> the plane of embeddedness (D_e). Vertical extent is measured perpendicular to the plane of embeddedness, not along the long axis of the particle (a-axis).



A.



10. Pick up the particle, keeping track of its orientation relative to the plane of embeddedness. Examine the particle to determine whether or not it is embedded. If it is embedded, you will generally find some fine sediment or algae clinging to the sides of the particle, and often this fine sediment has a clearly defined upper surface (a bathtub ring around the gravel). If the particle comes out easily (no suction or tightness felt from fine sediment), and no fine sediment or algae is visible on its sides, it is not embedded. Call out "not embedded". Sometimes you may need to check the hole the gravel came from to see if it is formed by fine sediment. Particles that are surrounded by a matrix of coarse sands and gravels should not be considered embedded. The matrix around an embedded particle should include a significant proportion of fines.
    a. Record “e” for ‘embedded’ or “n” for ‘not embedded’ in the **Embed** column in the worksheet **Data_GC_***. It may be useful to enter all “n”s when beginning a transect and then change these to “e” when embedded particles are encountered
11. If the particle is embedded, call out embedded. Measure D_e and call out the number to be recorded. Then measure D_t and call out the number to be recorded.
    a. Record D_e as **Ht_Embed** and D_t as **Ht_Total** in the worksheet **Data_GC_***

Gravel size

12. Measure gravel size using the gravelometer. Try to push the gravel through various holes in the gravelometer, rotating the gravel as necessary to try to get it through the hole. Determine the largest hole that the gravel **cannot pass through** – the gravel is coarser than this size. This is the gravel size that will be recorded. Don't force it or bend the gravelometer.
    a. Enter the size in the **Size** column in the worksheet **Data_GC_***
13. If you find loose fine sediment (finer than 2mm) at your sampling point, record it as "f" in the **Size_Code** column and do not enter any size in the **Size** column. These data are used to calculate % fines. Record numerical sizes down to 2mm. Everything finer than 2mm is recorded as "f" (if it is loose mobile sediment). If the fine sediment is compacted and hard, it probably not sediment in active transport but a eroded surface. Do not record these as fines. Record it as "fhp" in the **Size_Code** column -- fine hardpan. (This is the term used in the STREAM spreadsheet.)
14. When you encounter particles larger than 180mm (the largest size hole on the gravelometer), you will enter "l" (large) in the **Size_Code** column, and you will need to measure the B-axis and C-axis using a ruler or hand tape and enter these values in the **B-axis** and **C-axis** columns. We will correct these to gravelometer sizes using the equation in Bunte and Abt (2001, Ch. 2, p. 21).
15. If a particle is encountered that cannot be dislodged, write "utd" (unable to dislodge) in the **Notes** column. You will also measure whichever axes you are able to measure or estimate.
    a. Look at the rock and try to determine which axes are which, that is, which are visible and able to be measured and which are buried. Which is the longest axis? This is the A-axis. In most cases, the A-axis will be readily available and visible along the stream bed surface, horizontal relative to the plane of the channel bed. Measure or estimate this and record the value in the **A-axis** column. Next, look at the axis perpendicular to the longest axis (i.e. also visible and horizontal to the plane of the channel bed). Does this appear to be the second longest axis (B-axis)? or does the second longest axis appear to be buried and vertical to the plane of channel bed? Decide whether this available-to-be-measured axis that is perpendicular to the first longest axis is the B-axis or C-axis and record it in the appropriate column (**B-axis** or **C-axis**). For utds, judgment calls are necessary to guess which axes are which and you may not be able to measure more than one axis. The important thing is to use your best judgment, obtain what measurements you can and record these in the appropriate columns.

b. In the column **Size Code**, enter "ul" (unable to dislodge, large) if the particle appears to be greater than 180 cm in size (B-axis) or "um" (unable to dislodge, medium) if the particle appears to be less than 180 cm in size. Enter "uk" (unable to dislodge, unknown) for the **Size_Code** if it is impossible to get any useful measurements of the particle. If the entry of "ul" vs "um" vs "u" is slowing the recording process, it is acceptable to simply enter "u" in the Size_Code and we can later determine which code is appropriate depending upon the data entered in the A-axis, B-axis and C-axis columns.

16. In some cases, a very large rock will be encountered twice within the sampling grid in spite of our use of a minimum grid spacing to prevent such occurrences. This occurs when there is an extremely large boulders within the sampling grid that is larger than the minimum grid spacing. In these cases, record a "d" (duplicate) in **Size_Code** the second time the rock is encountered and do not make a measurement of it the second time.
17. When recording data, don't use any symbols in the **Size** column other than numbers and don't use any symbols in the **Size_Code** column other than "f", "fhp", "l", "ul", "um", "uk," "u" or "d". If you do, these will have to be edited out manually to analyze the data. All notes should go in the notes column. Commonly used notes include: "utd" (unable to dislodge), "wv" (wetted vegetation, where the point actually fall on a patch of vegetation; usually the sediment is fines, but we don't include these in the gravel size analysis since it's not active bed), "CANU" (for a point on a torrent sedge tussock, *Carex nudata*). It is also common practice to note water edge along the transect ("rwe" or "lwe"), since transects may extend into dry areas that include gravels that could potentially be mobilized. Also note any unusual circumstances.
18. When the first transect is completed, move the cross tape to the next position along the bank tape and begin the second transect. Continue until you have counted all the transects (cross tape lines) you decided upon in steps 1 through 5. Count all the transects you laid out in steps 1 through 5, even if you exceed 300 particles. We need to sample the entire riffle.
19. Check data sheet at the end of each transect.
20. Check the data sheet to make sure that all numbers are filled in and that all numbers make sense.

Photos and GPS:

21. Take a minimum of two photos from the bank, one at the upstream end l of the bank tape ooking downstream at the riffle, and one at the downstream end of the bank tape looking upstream at the riffle. Try to

get the bank tape in the photos. Record the photo number, bank and tape positions for each photo.

22. Take two GPS points at each gravel count site, on the bank tape side, one at the upstream end and one at the downstream end. Record the bank and tape positions for each GPS point.

References

Bunte, K., and Abt, S. R., 2001. Sampling Surface and Subsurface Particle-Size Distributions in Wadeable Gravel- and Cobble-Bed Streams for Analyses in Sediment Transport, Hydraulics, and Streambed Monitoring. U. S. Forest Service Rocky Mountain Research Station General Technical Report RMRS-GTR-74. Available at: http://www.stream.fs.fed.us/.

Harrelson, C. C., Rawlins, C. L., and Potyondy, J. P., 1994. Stream Channel Reference Sites: An illustrated guide to field technique. U. S. Forest Service Rocky Mountain Forest and Range Experiment Station General Technical Report RM-245. Available at: http://www.stream.fs.fed.us/.

Ohio Department of Natural Resources, Division of Soil and Water Resources, n. d. Spreadsheet Tools for River Evaluation, Assessment and Monitoring. Available at: http://www.dnr.state.oh.us/soilandwater/water/streammorphology/default/tabid/9188/Default.aspx.

Potyondy, J., and Bunte, K., 2002. Analyzing pebble count data collected by size classes. Spreadsheet available at: http://www.stream.fs.fed.us/.

Gravel Count Data Analysis

By P. McDowell, University of Oregon

1. We excluded from quantitative analysis all points coded as wetted vegetation, Carex nudata, aquatic vegetation, etc. We excluded "unable to dislodge" (UTD) points if we did not get a measurement of B-axis. We included only those points for which we got a particle size measurement or F (fines).
2. Using the gravelometer, all clasts are recorded in 0.5 phi size classes. We recorded "coarser than" size classes – the size of the largest opening the clast would not pass through. The size classes larger than the largest opening (180mm) are 256, 362, 512, 724, 1024mm, etc.
3. For rocks larger than the largest gravelometers opening (>180mm), we used the B-axis value to determine the size class, and manually entered the value for the "coarser than" size class boundary.
4. To calculate percentiles and other statistics, we used a spreadsheet from the Ohio Department of Natural Resources (2010; STREAM module; Reference_Reach_Survey_4_2_T.xlsm; http://www.dnr.state.oh.us/soilandwater/water/streammorphology/default/tabid/9188/Default.aspx, 11/05/2010).
    a. We developed a spreadsheet which converted our field data to counts by size class in the STREAM module.
    b. The STREAM module spreadsheet calculated % gravel, % cobble, % boulder, D16, D35, D50, D65, D84, D95 and mean.
    c. Our fines size class corresponds to <2mm (sand, silt and clay). Since this is larger than a single size class, we excluded fines from the calculations in the STREAM module spreadsheet. Therefore, our values for % gravel, % cobble and % boulder sum to 100%
5. We calculated % fines by combining the fines counts with the gravel, cobble, and boulder counts.
6. We calculated % embedded as a percentage of the total count of fines, gravels, cobbles and boulders.
7. We calculated the embeddedness ratio for each grain recorded as embedded by dividing the total grain height by the embedded height (Bunte and Abt, 2001).
8. We calculated average embeddedness ratio for each year sampled at each site.

References

Bunte, K., and Abt, S. R., 2001. Sampling Surface and Subsurface Particle-Size Distributions in Wadeable Gravel- and Cobble-Bed Streams for Analyses in Sediment Transport, Hydraulics, and Streambed Monitoring. U. S. Forest Service Rocky Mountain Research Station General Technical Report RMRS-GTR-74. Available at: http://www.stream.fs.fed.us/.

Ohio Department of Natural Resources (currently at Ohio Department of , 2010. STREAM Modules: Spreadsheet Tools for River Evaluation, Assessment and Monitoring. Available at: http://water.ohiodnr.gov/water-conservation/stream-restoration#SPR, and http://www.agri.ohio.gov/divs/SWC/SWC.aspx#tog, accessed Aug. 29, 2017.

Fish Cover Survey

By P. McDowell, P. Lind, and M. Goslin, University of Oregon

This protocol follows the method of USEPA (2004) with minor modifications. The goal is to estimate percent area fish cover in representative glides and pools. The survey is done on a 10 meter long section of a representative channel unit. Left bank is always on your left when looking downstream, and right bank on your right when looking downstream.

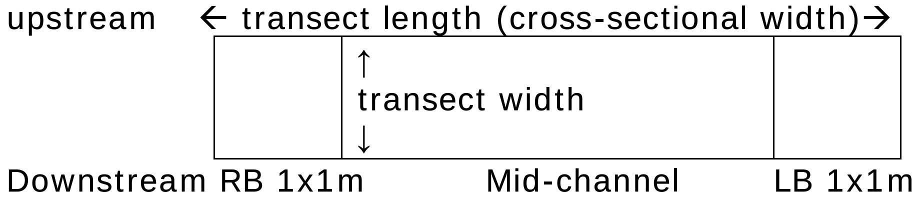
Equipment: list of GPS coordinates for the fish cover sites to be sampled; 3 (or more) 30-m tapes; 6 or more chaining pins to hold the tapes in place; data collector or data form; field notebook; camera, GPS unit. 1-m square of pvc piping is recommended but optional.

Team: at least two people. One person focuses on recording data, GPS and photos, while the other one focuses on estimating fish cover.

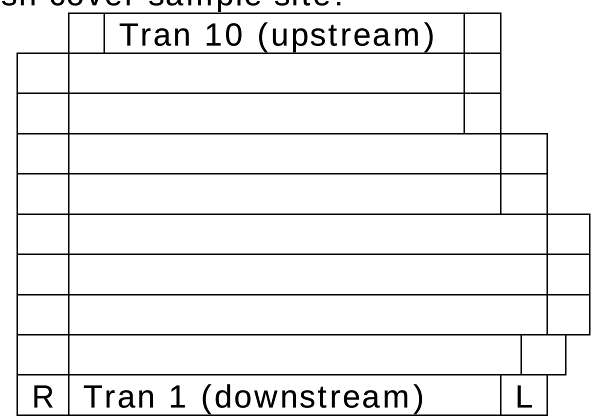
1. Navigate to the site and visually identify the upper and lower boundaries of the pool or glide and its midpoint. The midpoint will not necessarily be at the coordinate recorded for the midpoint several years ago, because the pool may have shifted upstream or downstream. Set up a ten meter tape longitudinally along the wetted edge of the channel on one side. Meter “5” on the tape is placed at the approximate mid point of the channel unit, and 0m is downstream.
2. Only the wetted portion of the channel is surveyed for fish cover. The ten meter long survey is divided into ten cross-sectional transects with each transect 1 m in width. Usually you will start at the downstream end and work upstream, so that sediment you kick up will not obscure the section you are working on. Transects are numbered from downstream to upstream. The first transect (0-1 m at the downstream end) is labeled 1, the second transect (1-2 m) is 2, and so on going upstream. Each transect is subdivided across the channel into three channel sections (left bank edge, mid-channel, right bank edge). The LBE and RBE sections are the 1 m^2 areas at the wetted edge of the channel,

while the mid-channel section is the remaining area of the 1m wide transect between the LBE and RBE. Mid-channel area varies depending on the transect length (the cross-sectional water surface width), but LBE and RBE are consistently 1 m^2.

A single transect:



Ten transects of a fish cover sample site:



3. Where the channel unit is curved, set up the longitudinal tape on the inside of the bend. Then set up each transect as a 1-m wide band (1-m wide at both ends). The transects will fan out, and there will be slivers between them that are not sampled. The goal is to sample the same area as you would sample for a straight unit.
4. At transect 1, stretch the second tape across the channel and measure the wetted cross-sectional width. Record the cross-sectional width (the transect length) in the XS_width column. If the transect has slanted edges, just measure the cross-sectional width at an average point. Leave the second tape pinned across the channel in order to help visually estimate the LB, middle and RB sections. Stretch and pin a second tape across the channel at the boundary between transect 1 and 2, to help in visualizing area.
5. Identify the RB section of transect 1 visually. You can simply visualize the boundary between the RB and mid-channel sections (use the cross-channel tape to see the 1-m point) or you can lay down the 1-m pvc square to mark the RB section.
    a. ***Remember you are facing upstream, so the RB section is on your left and the LB section is on your right.***

b. We are sampling water that is deep enough for small fry or deeper.

6. Now visually estimate the percentage of area in the RB section for each type of fish cover. Fish cover is considered water surface or bed area covered by vegetation, undercut banks, or objects in channel. (See list of cover types below.)
   a. Estimate to the nearest 10%. If the area is between 5 and 15%, record 10%. If it is between 15 and 25 %, record it as 20%. And so on. If there is 5 to 1%, enter 2. If there is less than 1%, enter 0. The estimator calls out the percent area for each type and the recorder records the data.
   b. The area for each type is estimated independently of other types. For example, there may be an area that has both emergent aquatic vegetation and filamentous algae. That area is counted under each of these two columns.
   c. For vegetation area, you should be estimating canopy area, not basal area or root area. Estimate the entire area covered by canopy even if the canopy does not provide 100% cover. There may be some small clear spots within the canopy, but these are likely to be too small for prey to be observed.

Fish cover types (Field Name in data worksheet)

Filamentous Algae (**FAlgae**): refers to long streaming algae

Aquatic Macrophytes (**AMacro**): emergent and submerged aquatic plants (including mosses), but not including filamentous algae

Carex nudata (**CANU):** Include areas with CANU canopy over the water. Do not include CANU canopy that over land. Estimate area of the entire circle of CANU over water, including the tussock. (We'll subtract a percentage for tussock area later.) (In some years, **CANU** was included in **AMacro**.)

Other Overhanging Vegetation (**OverVeg**): Include both woody and non-woody plants (excluding CANU) that is not growing in the water but overhangs within 1 m vertically above the low-flow water surface.

In-channel Live Trees or Roots (**LiveTR**): Include live woody plants rooted or lying within the wetted channel, such as fallen still-alive submerged trees; small woody plants, e.g. willow saplings; and plants that may have been uprooted from a bank upstream and deposited mid-channel among other debris. Estimate area of the portion of the plant that is currently inundated or would be inundated at bankfull flow (within 1m of low-flow water surface). Do not count non-woody plants within the channel.

Large Woody Debris (**LWD**): Pieces of wood at least 0.1 m in diameter at the small end and at least 1.5 m in length. If it is smaller in either one of these dimensions, it is Small Woody Debris (SWD). Do not include constructed log structures.

Brush & Small woody debris (**SWD**): pieces of wood in the channel that are smaller than 0.1 m in diameter at the small end and 1.5 m in length.

Undercut Banks (**UCutBnk**): Reach under the undercut with a ruler or rod and get approximate depth and length.

Boulders (**Boulders**): Include boulders within or partly in the wetted channel. Cover provided by boulders is primarily a narrow strip under the overhang of the boulder. This is the area under boulder overhang or around the edges of the boulder which a fish might occupy and be provided some kind of protective cover. Much of the area occupied by a very large boulder is rock and not available to fish. In a group of small boulders close to each other, the entire area between these small boulders should be estimated as boulder cover (lots of available areas underneath and among the boulders). Do not include boulders placed by humans.

Artificial Structures (**ArtStruct**): Include both boulders and LWD that have been artificially placed (e.g. rip rap or engineered log jams); also include rip-rap boulders (very large, angular) that have fallen in from the road edge or the bank. These artificial boulders and LWD are not included within **LWD** or **Boulders**.

Turbidity (**Turbid**): Water is too turbid to determine cover.

Total Area of Fish Cover (**Total**): Don’t count overlapping types twice.

7. Now independently estimate the total percentage of area with fish cover in the RB section. If an area has two types, it is not counted twice. The sum of the percentages for the individual types may sum to a larger value than the total percentage.
8. Repeat the procedure for the LB section of transect 1.
9. Repeat the procedure for the middle section of transect 1.
10. Repeat the procedure at each of the other nine transects.
11. Channel width is measured separately at each transect for calculation of total transect area and total survey area.
    Take photographs of the surveyed area with the longitudinal (bank) tape visible, from upstream and downstream. Record photo numbers and where taken (for example, from US LB looking DS, or from mid channel DS looking US).
12. Take a GPS point on the bank at channel’s edge at the 5-m mark on the longitudinal tape (middle of the site). Record which bank you are on (RB or LB). Record GPS file name and coordinates.

Reference

USEPA. 2004. Wadeable Stream Assessment: Field Operations Manual. EPA841-B-04-004. U.S. Environmental Protection Agency, Office of Water and Office of Research and Development, Washington, DC.

Fish Cover Analysis

Cover types recorded as a trace in the field are those with <5% cover. Where trace was recorded in the edge boxes, we counted it as 2%. Where trace was recorded in the middle box (larger area), we counted it as 0%. Cover types recorded in the field (in some years) as <10% were converted to 5%.

To calculate percentage of area in each cover type for each 1m-strip within a site, we weighted each percent cover value recorded in the field by the area sampled, and averaged these values for the strip. To calculate percent cover for each type for the entire plot, we averaged values from each 1-m strip weighted by the area of the strip.

We calculated total fish cover percentage in two ways, with and without filamentous algae, and used these values in different ways in interpretation.

Supplemental Analysis and Results

Channel Morphology

Figures CM-1 to CM-3 show restoration effect on channel dimensions in BEBU(P).

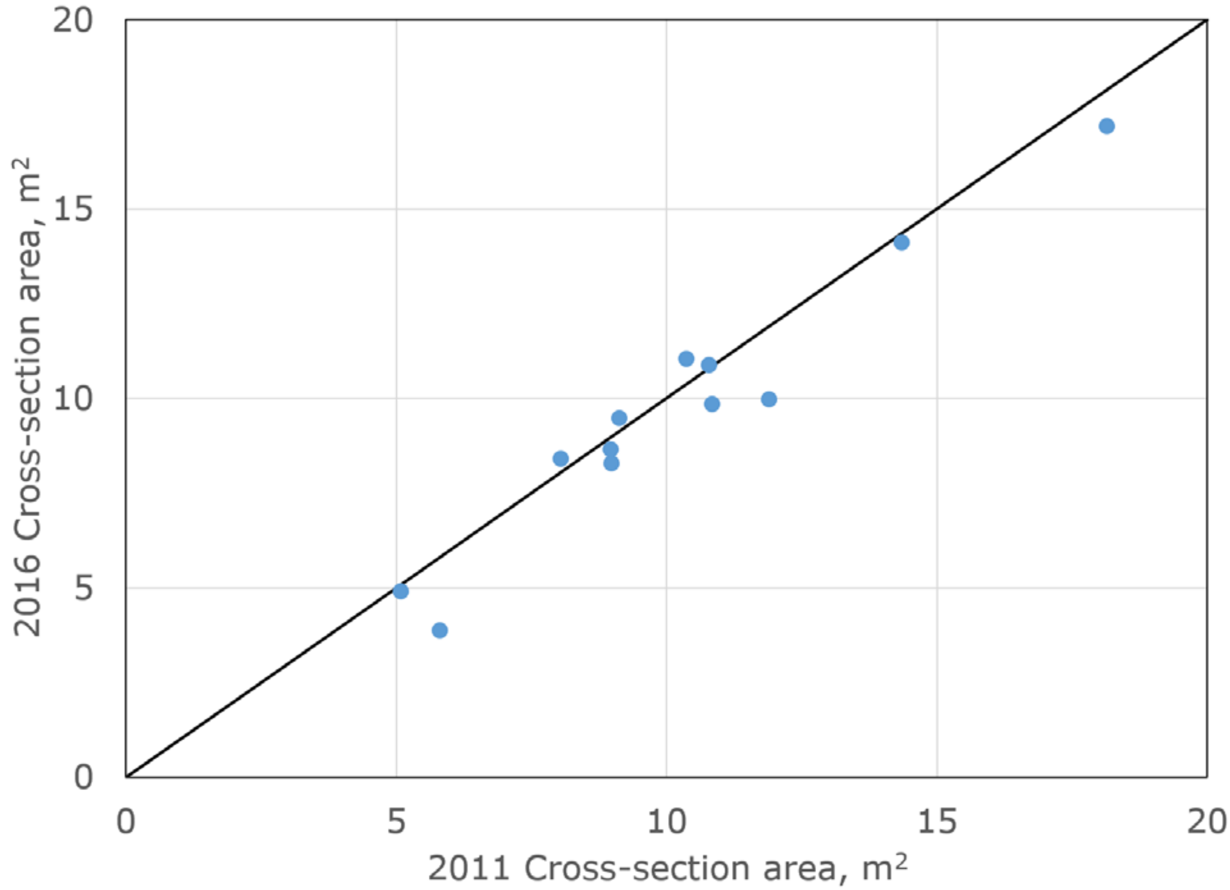


Figure CM-1. Comparison of channel cross-section area in BEBU pre-restoration (2011) and post-restoration (2016).

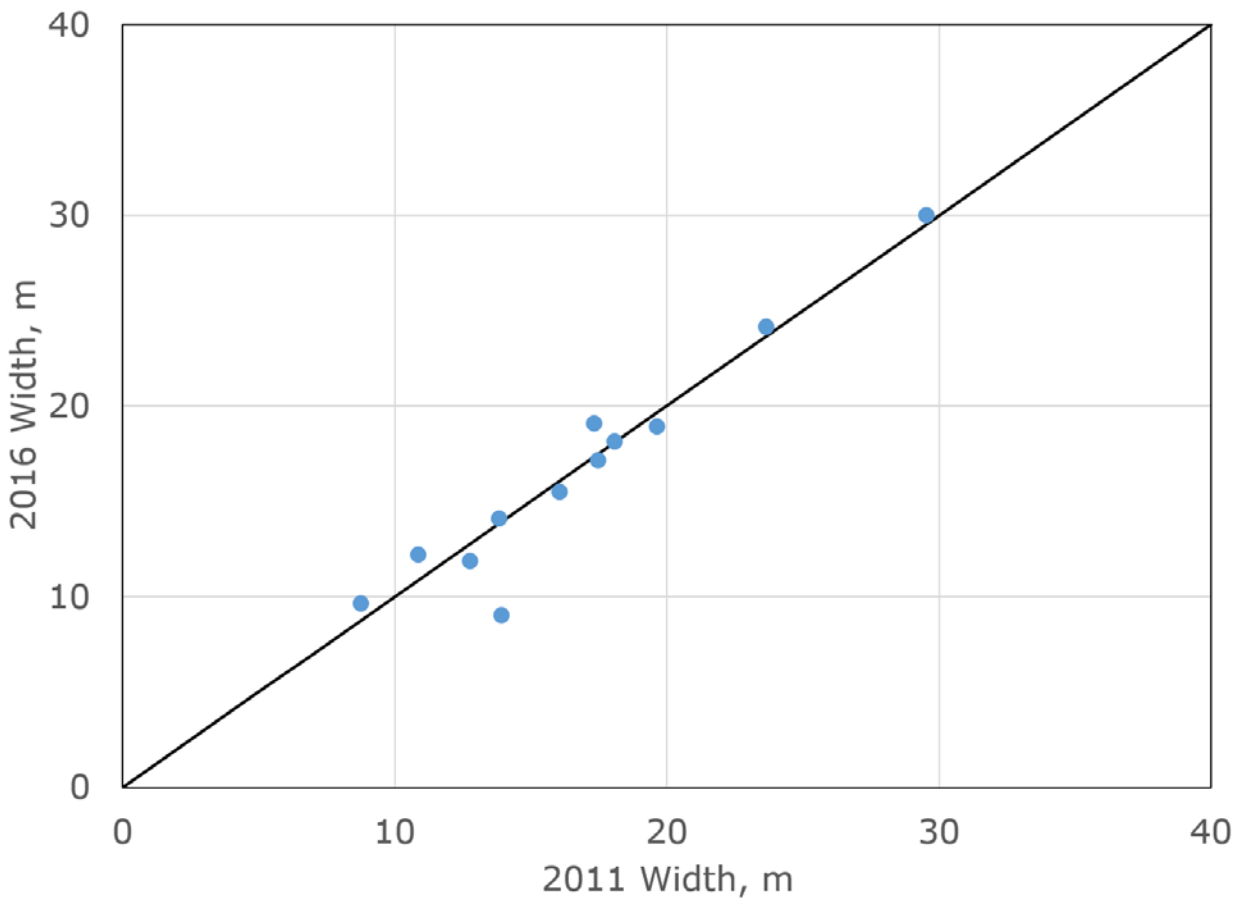


Figure CM-2. Comparison of channel width in BEBU pre-restoration (2011) and post-restoration (2016).

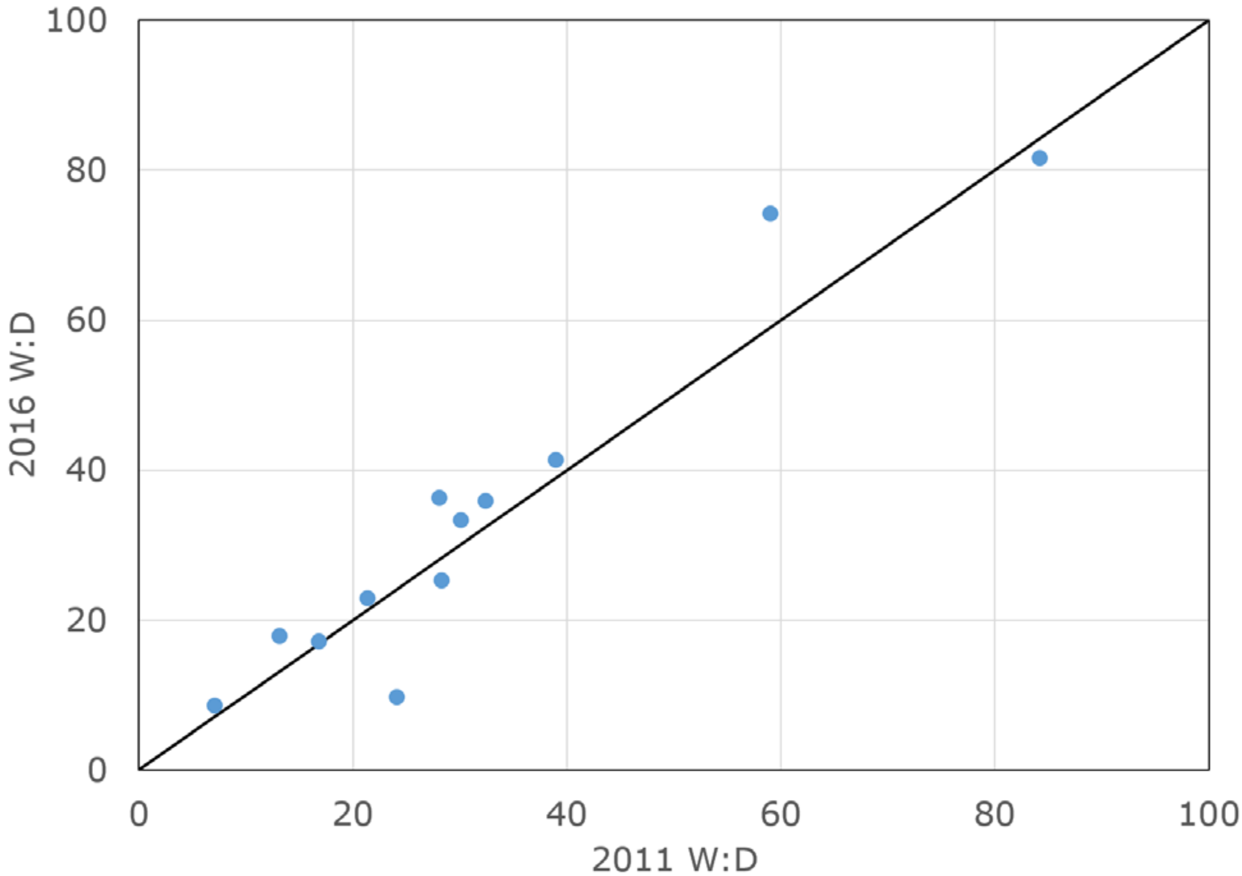


Figure CM-3. Comparison of channel width to depth ratio in BEBU pre-restoration (2011) and post-restoration (2016).

Table CM-1. Differences between pre-restoration and post-restoration channel morphology in BEBU(P)

Variable	log10 Area (m^2)	log10 Width (m)	log10 Depth (m)	log10 W:D
Initial mean	0.9591	1.1945	-0.2202	1.4299
Final mean	0.9574	1.1952	-0.2457	1.4330
Initial variance	0.0248	0.0312	0.0299	0.1075
Final variance	0.0310	0.0242	0.0395	0.0933
P(one-tail)	0.4746	0.4857	0.1261	0.4280
P(two-tail)	0.9492	0.9713	0.2522	0.8560

Notes on Table CM-1: The time span is 2011-2016. The original variables were bankfull cross-section area (m2), bankfull width (m), mean bankfull depth (m), and bankfull width:depth ratio. The original variables were log-10 transformed and adjusted to eliminate negative values. The test is a paired, two-sample t-test. The degrees of freedom equals 11. None of the variables show significant differences between project and control reaches at the level P=0.05.

Table CM-2. Differences between project and control sites in channel morphology change over the period of study.

Variable	Area change	Width change	Depth change	W:D change
Project sites mean	0.6619	0.5196	0.7341	0.7795
Control sites mean	0.6768	0.4548	0.8279	0.6863
Project sites variance	0.0341	0.0560	0.2679	0.0500
Control sites variance	0.1084	0.3193	0.0193	0.2819
Degrees of freedom	44	38	22	38
P(one-tail)	0.4209	0.2942	0.2134	0.2045
P(two-tail)	0.8417	0.5883	0.4268	0.4089

Notes on Table CM-2: The original variables were the same as in Table CM-1. Each variable was calculated as percent change between the final and initial measurements at a cross-section. Each variable was then log-transformed and adjusted to eliminate negative values. The test was a t-test, two-sample, assuming unequal variances. The project sites sample was all cross-sections from VIBR(P) and RABE(P), and the control sites sample was all cross-sections from BUTI(C), DRRA(C), and JUCA(C). None of the variables show significant differences between project and control reaches at the level P=0.05.

Table CM-3. Differences between initial and final measurements for four channel dimensions in each reach. Each variable was then log-transformed and adjusted to eliminate negative values. None of the reaches show significant differences for any variable at the level P=0.05.

Reach	Variable	Log10 Area	log10 Width	log10 Depth	log10 W:D
VIBR	Initial mean	0.8271	1.1710	-0.3476	1.5148
VIBR	Final mean	0.8449	1.1866	-0.3417	1.5283
VIBR	Initial variance	0.0141	0.0093	0.0152	0.0373
VIBR	Final variance	0.0174	0.0075	0.0183	0.0343
VIBR	Degrees of freedom	20	20	20	20
VIBR	P(one-tail)	0.3718	0.3469	0.4583	0.4345
VIBR	P(two-tail)	0.7436	0.6938	0.9166	0.8689
BUTI	Initial mean	0.8285	1.1256	-0.2971	1.4226
BUTI	Final mean	0.9031	1.1860	-0.2829	1.4689
BUTI	Initial variance	0.0907	0.0303	0.0329	0.0358
BUTI	Final variance	0.0750	0.0246	0.0260	0.0262
BUTI	Degrees of freedom	12	12	12	12
BUTI	P(one-tail)	0.3181	0.2540	0.4399	0.3159
BUTI	P(two-tail)	0.6363	0.5080	0.8797	0.6317
RABE	Initial mean	0.9845	1.2321	-0.2644	1.4797
RABE	Final mean	0.9218	1.2223	-0.3132	1.5228
RABE	Initial variance	0.0242	0.0298	0.0141	0.0541
RABE	Final variance	0.0306	0.0251	0.0111	0.0361
RABE	Degrees of freedom	18	18	18	18
RABE	P(one-tail)	0.2042	0.4482	0.1720	0.3277
RABE	P(two-tail)	0.4085	0.8965	0.3440	0.6553
DRRA	Initial mean	1.1693	1.3422	-0.1729	1.5152
DRRA	Final mean	1.1561	1.3363	-0.1802	1.5165
DRRA	Initial variance	0.0235	0.0109	0.0075	0.0132
DRRA	Final variance	0.0233	0.0166	0.0058	0.0214
DRRA	Degrees of freedom	16	16	16	16
DRRA	P(one-tail)	0.4288	0.4582	0.4262	0.4913
DRRA	P(two-tail)	0.8576	0.9164	0.8525	0.9827
JUCA	Initial mean	1.0928	1.3053	-0.2125	1.5177
JUCA	Final mean	1.0565	1.2919	-0.2354	1.5273
JUCA	Initial variance	0.0420	0.0152	0.0190	0.0264
JUCA	Final variance	0.0322	0.0158	0.0130	0.0255
JUCA	Degrees of freedom	22	22	22	22
JUCA	P(one-tail)	0.3242	0.3972	0.3308	0.4431
JUCA	P(two-tail)	0.6484	0.7943	0.6615	0.8863

Table CM-4. Differences between project and control reaches in change during the 2011 flood event.

Variable	Area % change	Width % change	Depth % change	W:D % change
Project sites mean	1.0937	1.0247	1.1256	1.2811
Control sites mean	1.2451	1.0882	1.2001	1.3183
Project sites variance	0.3388	0.0498	0.1702	0.0466
Control sites variance	0.0057	0.0192	0.0106	0.0284
Degrees of freedom	21	31	22	37
P(one-tail)	0.1247	0.1298	0.2135	0.2585
P(two-tail)	0.2495	0.2597	0.4271	0.5171

Notes on Table CM-4: The original variables were bankfull cross-section area (m2), width (m), depth (m) and width:depth ratio. The pre-flood and post-flood values were the measurements closest in time to spring 2011. The time span varied from 1 to 2 years. Each variable was calculated as percent change between the pre-flood and post-flood measurements at a cross-section. Each variable was then adjusted to eliminate negative values and log-transformed to ensure the distribution was normal. The test was a two-sample t-test assuming unequal variances. None of the variables show significant differences between project and control reaches at the level P=0.05.

Pool Depth

Table PD-1. Differences in pool depth from 2008 to 2015-16, by reach. Bold and underlined indicates statistically significant values at the P=0.05 level.

	VIBR(P)	BEBU(P)	RABE(P)	BUTI (C)	DRRA (C)
2008 mean	-0.4892	-0.2339	-0.3591	-0.4329	-0.5820
2015-16 mean	-0.8264	-0.7585	-0.3123	-0.4004	-0.4179
Direction of change in pool depth	deeper	deeper			shallower
2008 variance	0.0468	0.0225	0.0741	0.0182	0.0370
2015-16 variance	0.0137	0.0238	0.0189	0.0100	0.0109
Degrees of freedom	49	57	32	28	17
P(one-tail)	<u>**0.0000**</u>	<u>**0.0000**</u>	0.2551	0.2273	<u>**0.0125**</u>
P(two-tail)	<u>**0.0000**</u>	<u>**0.0000**</u>	0.5102	0.4547	<u>**0.0250**</u>

Notes on Table PD-1: The original variables were residual pool depth (m) for each pool in the reach. The values were log10 transformed. For each reach, a t-test of two-samples, assuming unequal variance, was done. A test that is significant means that there is a difference in 2008 and 2015-16 residual pool depth in that reach.

Log Structures

Table LS-1: Years of log structure survey

	2008	2009	2010	2011	2012	2013	2014
VIBR LS 1	x		x				x
VIBR LS 2	x					x	
VIBR LS 3	x		x			x	
VIBR LS 4	x		x			x	
VIBR LS 5	x		x			x	
VIBR LS 6	x						x
VIBR LS 7	x						x
VIBR LS 8	x		x			x	
VIBR LS 9	x		x			x	
VIBR LS 11	x						x
VIBR LS 12	x		x			x	
VIBR LS 13	x					x	
VIBR LS 14	x						x
VIBR LS 15	x					x	
VIBR LS 16	x					x	
VIBR LS 17	x						x
RABE LS 1		x		x		x	
RABE LS 2		x				x	
RABE LS 4		x		x			x
RABE LS 6		x		x			
RABE LS 7		x		x		x	
RABE LS 8		x				x	
RABE LS 9		x				x	
RABE LS 10		x				x	
RABE LS 11		x		x			x
RABE LS 12		x				x	
RABE LS 16		x					x

Table LS-2. Summary of qualitative changes observed in cross-section at BEBU (P) log structures.

Log structure number	XS	Bed	Bar	Bank
LS03	1	N	N	E
LS03	2	A	A	N
LS03	3	A	N	S
LS05	1	A	A	N
LS05	2	A/D	N	E/A
LS05	3	A	A	E
LS05	4	D	A	N
LS05	5	D		
LS06	1	A/D	A	N
LS06	2	A	N	A
LS06	3	D	N	N
LS06	4	D	N	N
LS09	1	A	A	E
LS09	2	A	A	E
LS09	3	A	A	E
LS09	4	D	A	N
LS09	5	A	A	N
LS10	1	A	A	N
LS10	2	A	D/A	E
LS10	3	D	A	E
LS10	4	N	N	E/A
LS10	5	A	A	E/A
LS11	1	D	A	E
LS11	2	D	A	E
LS11	3	N	N	E
LS11	4	D	A	E
LS-12	1	N	N	E
LS-12	2	A	N	N
LS-12	3	N	D	N
LS16	1	N	A	E
LS16	2	A	N	E
LS16	3	A	D	S/A
LS16	4	A	A	A

Notes on Table LS-2: Log structures are numbered from 1 (upstream) to 16 (downstream). At each log structure, cross-sections are numbered from 1 (upstream) to 3 or 5 (downstream). Bed refers to the near-horizontal surface that is all or partly underwater in summer. Bar refers to a surface with low slope on one side of the cross-section. Bank refers to a vertical or near-vertical surface on one side of the cross-section. Codes: A = aggradation; D = degradation of bed or bar. N = no clear change. E = erosion of near-vertical bank. S = bank slump. D/A = degradation in part of cross-section and aggradation in another, or degradation is some time period, aggradation in another. E/A in vertical banks usually indicates erosion in upper part of bank and deposition near base of bank. S/A indicates slump in upper part of bank and aggradation (possibly by slump block) in lower part.

Bed Material

Table BM-1. Summary of bed material characteristics in initial year of measurement

Reach		% Gravel	% Cobble	% Boulder	D16, mm	D50, mm	D84, mm	D95, mm	% Fines	% Embedded	Embed ratio
VIBR(P)	median	54	46	0	16	57	110	160	1.8	18.8	2.6
	range	42 - 79	21 - 58	0 - 1	11 - 36	42 - 70	74 - 110	110 - 190	0 - 7.4	9.6 - 35.1	1.8 - 3.0
BUTI(C)	median	62	38	0.5	22	54	88.5	125	5.3	10.6	2.2
	range	50 - 85	28 - 49	0 - 3	19 - 25	35 -63	62 - 120	95 - 210	0 - 11.1	3.8 - 17.7	1.6 - 2.9
BEBU(P)	median	63	36	0	20	46	88	120	2.8	10.2	1.9
	range	19 - 88	12 - 68	0 - 12	13 - 58	26 - 120	53 - 230	91 - 310	0 - 5.9	4.2 - 17.7	1.6 - 3.0
RABE(P)	median	50.5	48	0	22	62	120	175	2.0	16.4	2.0
	range	33 - 67	33 - 65	0 - 0	7.7 - 37	48 - 97	85 - 170	150 - 330	0 - 6	6.1 - 34.7	1.7 - 5.6
DRRA(C)	median	57	42	0	27	57	100	160	2.0	13.7	2.3
	range	31 - 68	30 - 69	0 - 2	17 - 46	51 - 90	84 - 130	120 - 230	0 - 3.8	11.8 - 22.0	1.7 - 2.6
JUCA(C)	median	53.5	43.5	2	24.5	60	125	215	0.0	17.4	2.2
	range	38 - 67	32 - 53	0 - 11	17 - 37	48 - 88	89 - 230	150 - 310	0 - 1.6	5.2 - 30	1.7 - 3.4

Table BM-2. Data from volumetric gravel samples

sample_id	D_{95} surface, mm	D_{95} subsurface, mm	D_{50} surface, mm	D_{50} subsurface, mm	D_{50surf}/D_{50sub}	% Finer than 2mm surface	% Finer than 2mm subsurface	% Finer than 5.6mm surface	% Finer than 5.6mm subsurface
BEBU_BG_01	43.20	106.69	24.95	32.96	0.76	7.00	7.99	16.24	17.09
BEBU_BG_02	91.92	67.08	48.09	27.80	1.73	2.31	10.25	4.97	18.64
BEBU_BG_03	284.89	292.01	82.98	33.56	2.47	6.82	9.43	14.02	20.34
BEBU_BG_04	109.54	117.55	31.36	23.95	1.31	3.72	6.53	14.97	20.58
BEBU_BG_05	88.49	109.51	32.93	26.83	1.23	5.23	7.93	13.90	19.07
BEBU_BG_06	155.26	134.89	75.30	31.10	2.42	1.20	8.49	5.28	18.91
BEBU_BG_07	227.47	169.53	99.00	48.68	2.03	2.24	7.01	6.86	17.50
BUTI_BG_01	117.18	155.51	64.10	64.40	1.00	1.69	6.11	4.29	13.72
BUTI_BG_02	164.03	324.37	77.29	42.27	1.83	2.90	8.99	8.38	18.10
BUTI_BG_03	124.85	111.13	26.63	19.85	1.34	5.60	12.00	15.72	25.43
DRRA_BG_01	157.64	162.79	67.52	40.63	1.66	1.93	7.57	7.48	19.07
DRRA_BG_02	235.24	124.41	65.91	18.70	3.52	5.46	10.65	14.89	25.80
JUCA_BG_01	159.64	109.56	60.34	37.12	1.63	3.32	7.22	7.21	13.69
JUCA_BG_02	168.25	116.14	95.66	48.00	1.99	2.97	10.41	4.76	16.85
JUCA_BG_03	168.68	108.53	39.91	16.57	2.41	8.23	14.15	15.97	27.91
JUCA_BG_04	164.79	240.52	66.36	52.39	1.27	4.77	6.72	12.14	14.83
RABE_BG_01	155.36	117.16	59.11	51.23	1.15	8.07	9.91	14.01	19.46
RABE_BG_02	170.39	162.03	65.05	24.48	2.66	4.56	11.03	10.71	25.82
VIBR_BG_01	159.84	166.68	66.55	55.10	1.21	2.23	4.89	7.21	12.91
VIBR_BG_02	114.31	54.65	53.09	14.21	3.74	4.46	13.48	10.68	28.73
VIBR_BG_03	136.85	144.22	47.61	31.55	1.51	7.42	8.63	15.03	18.28
VIBR_BG_04	112.14	161.40	45.64	50.77	0.90	5.93	6.28	13.90	14.24

Table BM-3. Results of ANOVA tests for differences among the six reaches in coarse grains, in initial year of measurement.

Variable	**F**	**F crit**	**P-value**
D50	0.8232	2.4221	0.5398
D64	1.1867	2.4221	0.3308
D85	1.5198	2.4221	0.2027
D95	2.8503	2.4221	**0.0255**

Notes on Table BM-3: There are no significant difference among the reaches in D50, D64, or D85. There are significant differences in D95. Figure BM-1 (below) indicates that JUCA is different from the upstream reaches in D95.

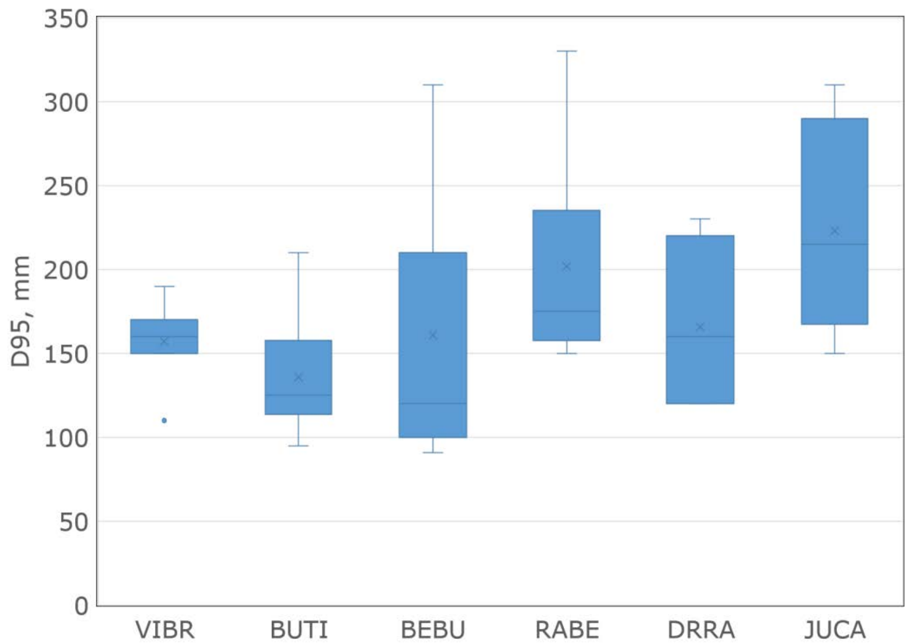


Figure BM-1. Boxplot of D95 for each reach.

Table BM-4. Initial to final differences in coarse fractions, by reach. Bold and underlined numbers show those variable where there is a significant difference between initial and final values at P=0.05.

Variable	VIBR	BUTI	BEBU	RABE	DRRA	JUCA
Degrees of freedom	6	5	9	7	6	9
log D50						
Mean initial	1.762329	1.708274	1.690419	1.809465	1.771774	1.784431
Mean final	1.850384	1.690698	1.633933	1.734356	1.843859	1.739993
Variance initial	0.006661	0.010004	0.031786	0.015566	0.009022	0.006765
Variance final	0.008162	0.011185	0.036329	0.022756	0.001276	0.007467
P (one-tail)	**<u>0.0047</u>**	0.2792	0.0625	0.0900	**<u>0.0314</u>**	0.0558
P (two tail	**<u>0.0094</u>**	0.5583	0.1249	0.1799	0.0627	0.1115
Direction of change	increase				increase	
log D84						
Mean initial	2.042557	1.955168	1.993908	2.10927	2.04655	2.11256
Mean final	2.071623	1.9734	1.950486	2.061148	2.087365	2.059124
Variance initial	0.007243	0.010102	0.040001	0.017792	0.010529	0.015782
Variance final	0.007633	0.010822	0.045604	0.014814	0.002117	0.008398
P (one-tail)	0.1485	0.2923	0.1642	0.2331	0.1376	0.0529
P (two tail	0.2970	0.5846	0.3284	0.4663	0.2752	0.1057
log D95						
Mean initial	2.191292	2.119729	2.176372	2.29682	2.206173	2.335422
Mean final	2.170437	2.120451	2.147473	2.254568	2.199597	2.289764
Variance initial	0.005396	0.013037	0.038191	0.013899	0.013202	0.012479
Variance final	0.006297	0.006787	0.056404	0.022433	0.000652	0.014694
P (one-tail)	0.1702	0.4923	0.2883	0.2983	0.4345	0.1098
P (two tail	0.3404	0.9847	0.5767	0.5967	0.8689	0.2197

Notes on table BM-4:

The original variables were D50, D84 and D95, all in mm, for each sampling site. The variables were log 10 transformed. For each reach, a t-test of two-samples, assuming unequal variance, was done. A test that is significant means that there is a difference in initial and final values in that reach.

Table BM-5. Initial to final differences in embeddedness and fines, by reach. Bold and underlined numbers show those variable where there is a significant difference between initial and final values at P=0.05.

Variable	VIBR	BUTI	BEBU	RABE	DRRA	JUCA
Degrees of freedom	6	5	9	7	6	9
Log % embeddedness						
Mean initial	1.2377	0.9923	0.9699	1.2155	1.1774	1.1387
Mean final	0.0504	0.2851	0.2050	0.1761	0.0810	0.1916
Variance initial	0.0391	0.0501	0.0438	0.0341	0.0119	0.0627
Variance final	0.0194	0.1431	0.1667	0.0748	0.0255	0.0384
P (one-tail)	**<u>0.0000</u>**	**<u>0.0081</u>**	**<u>0.0000</u>**	**<u>0.0000</u>**	**<u>0.0000</u>**	**<u>0.0000</u>**
P (two tail	**<u>0.0000</u>**	**<u>0.0162</u>**	**<u>0.0001</u>**	**<u>0.0000</u>**	**<u>0.0000</u>**	**<u>0.0000</u>**
Direction of change	decrease	decrease	decrease	decrease	decrease	decrease
Log Embeddedness Ratio						
Mean initial	0.3723	0.3462	0.3239	0.3605	0.3471	0.3717
Mean final	0.2049	0.3042	0.3361	0.2390	0.1992	0.2534
Variance initial	0.0084	0.0121	0.0070	0.0284	0.0053	0.0091
Variance final	0.0410	0.0141	0.0042	0.0140	0.0404	0.0267
P (one-tail)	0.0739	0.2235	0.3625	0.0978	0.0842	**<u>0.0195</u>**
P (two tail	0.1478	0.4469	0.7249	0.1957	0.1683	**<u>0.0390</u>**
Direction of change						decrease
Log % fines						
Mean initial	0.3354	0.3462	0.3239	0.3195	0.2404	0.0128
Mean final	-0.0405	0.3042	0.3361	-0.2618	-0.0549	0.4753
Variance initial	0.1456	0.0121	0.0070	0.1239	0.0573	0.0094
Variance final	0.0115	0.0141	0.0042	1.1873	0.0386	0.0864
P (one-tail)	**<u>0.0359</u>**	0.2235	0.3625	0.0844	**<u>0.0410</u>**	**<u>0.0004</u>**
P (two tail	**<u>0.0718</u>**	0.4469	0.7249	0.1688	0.0820	**<u>0.0009</u>**
Direction of change	decrease				decrease	increase

Notes on table BM-5:

The original variables were % embeddedness, embeddedness height ratio, and % fines (smaller than 2mm) for each sampling site. The variables were log 10 transformed. For each reach, a t-test of two-samples, assuming unequal variance, was done. A test that is significant means that there is a difference in initial and final values in that reach.

Fish Cover

Table FC-1. Initial to final differences in fish cover, by reach. Bold and underlined numbers show those variable where there is a significant difference between initial and final values at P=0.05.

Variable	VIBR	BUTI	BEBU	RABE	DRRA	JUCA
Degrees of freedom	6	5	12	5	7	9
Log % Total (all types)						
Mean initial	1.3574	0.9442	1.1414	1.0386	0.9201	1.0992
Mean final	1.2995	1.0886	1.2390	1.0572	0.9682	1.0167
Variance initial	0.0343	0.0590	0.0123	0.0123	0.0782	0.0382
Variance final	0.0271	0.0220	0.2495	0.2495	0.0816	0.0499
P (one-tail)	0.2370	0.0907	0.4655	0.4655	0.3794	0.2044
P (two tail	0.4739	0.1815	0.9311	0.9311	0.7588	0.4088
Log % Aquatic Macrophytes						
Mean initial	1.2778	0.1572	0.7912	0.6369	0.4263	0.3626
Mean final	1.1674	0.7879	0.8989	0.6792	0.6241	0.7396
Variance initial	0.0308	0.4699	0.2971	0.0587	0.1311	0.3469
Variance final	0.0521	0.0833	0.1680	0.7384	0.0896	0.0618
P (one-tail)	0.1317	0.0559	0.2693	0.4610	**<u>0.0366</u>**	**<u>0.0219</u>**
P (two tail	0.2634	0.1117	0.5386	0.9220	0.0731	**<u>0.0438</u>**
Direction of change					increase	increase
Log % Overhanging Vegetation						
Mean initial	-0.2170	0.5221	0.4239	0.2596	0.4020	0.4818
Mean final	0.2717	0.5369	0.5843	0.0913	0.4178	-0.1744
Variance initial	0.4560	0.3386	0.4111	0.2215	0.0810	0.3163
Variance final	0.3790	0.3379	0.1379	0.0767	0.1532	0.7519
P (one-tail)	0.1335	0.4515	0.1359	0.1912	0.4649	**<u>0.0045</u>**
P (two tail	0.2671	0.9029	0.2719	0.3825	0.9298	**<u>0.0090</u>**
Direction of change						decrease

Notes on Table FC-1:
The original variables were percent of area in aquatic macrophytes, overhanging vegetation, and total fish cover (all types), for each sampling site. Percent of area in boulder fish cover was not analyzed because of many zero values. The variables were log 10 transformed. For each reach, a t-test of two-samples, assuming unequal variance, was done. A test that is significant means that there is a difference in initial and final values in that reach.

Appendix E – Influence of Deer and Elk Browsing on the Success of Riparian Restoration Plantings

Steven M. Wondzell, USDA Forest Service, Pacific Northwest Research Station, Corvallis OR

Brian R. Cochran, Confederated Tribes of the Warm Springs Reservation of Oregon, Warm Springs, OR

Abstract

We studied the effects of wild ungulate browsing on native woody riparian species (both hardwoods and conifers) planted as part of the overall effort to restore aquatic and riparian ecosystems within the Middle Fork John Day River (MFJD). Unconstrained stream reaches along the river have been highly modified to support forage production for domestic livestock. Today, the MFJD is poorly shaded and summer stream temperatures can exceed 28 ^{o}C. To restore shade, thousands of seedlings were planted in 2006, but planting has had limited success, even in areas fenced to exclude cattle. We established small browsing exclosures in spring 2009 and remeasured the exclosures after one and two growing seasons. Our results showed that browsing by deer and elk suppressed the growth of most hardwoods. Only ponderosa pine and thinleaf alder showed consistent growth over two years. Overall, our results indicate that, in the absence of grazing by domestic livestock, browsing pressure from deer and elk may limit the potential to restore native riparian forests.

Introduction

Background

Many riparian zones throughout the western United States no longer support woody vegetation, especially native riparian hardwood species such as cottonwood, aspen, and willow (Figure 1). A variety of land-use activities over the preceding 100+ years is usually implicated in the loss of woody species, including riparian logging, dredge mining, over grazing, intentional removal to improve forage production for livestock, and the introduction of exotic pasture grasses that outcompete native woody species. In recent decades there has been increased interest in reestablishing woody riparian vegetation, in large part to restore stream shade and thus help mitigate warm stream temperatures that exceed water quality criteria and also impact cold-water dependent salmonids, many of which have been listed as threatened or endangered under the Endangered Species Act in recent years (Nehlsen et al. 1991).

Figure 1. Forrest Conservation Area (left), where native hardwoods are entirely lacking from the riparian area; the mouth of a small tributary, Vinegar Creek, where it enters the Forrest Conservation Area (right) showing expected dense growth of native riparian hardwoods.

Grazing by domestic livestock is often identified as the major factor controlling the current condition of native woody vegetation and is widely recognized as limiting regrowth of these species in riparian zones throughout the western United States (Belsky et al. 1999). A number of exclosure studies throughout the western USA help support this conclusion (Schulz and Leininger 1990) although results of exclosure studies may not be simple to interpret (Sarr 2002). At any rate, substantial time and money has been invested into fencing riparian zones to exclude cattle and promote recovery.

A small exclosure was established in the riparian zone of Camp Creek, a major tributary to the upper Middle Fork John Day River in 1977 (Figure 2). The fences were more than 8-ft tall and excluded deer and elk as well as domestic livestock. In the 3 decades since the exclosure was fenced, woody riparian species grew markedly, suggesting that there was substantial potential to restore riparian vegetation within the IMW area.

The Confederated Tribes of the Warm Springs fenced most of the valley floors of the Forrest and Oxbow Conservation Areas to exclude livestock (but not deer and elk), and in 2006, some 72,000 native riparian hardwoods and conifers (Table 1) were planted. Substantial mortality occurred within a few years of planting, but at least two-thirds of the planted seedlings and cuttings survived through 2009. However, the surviving plantings had not grown substantially since planting. All showed effects of severe browsing, even species such as Ponderosa pine and thinleaf alder. Further, hoof-prints, outlined in dried mud on the landscaping tarps, showed that both deer and elk were common within the livestock-excluded riparian zone (Figure 3).

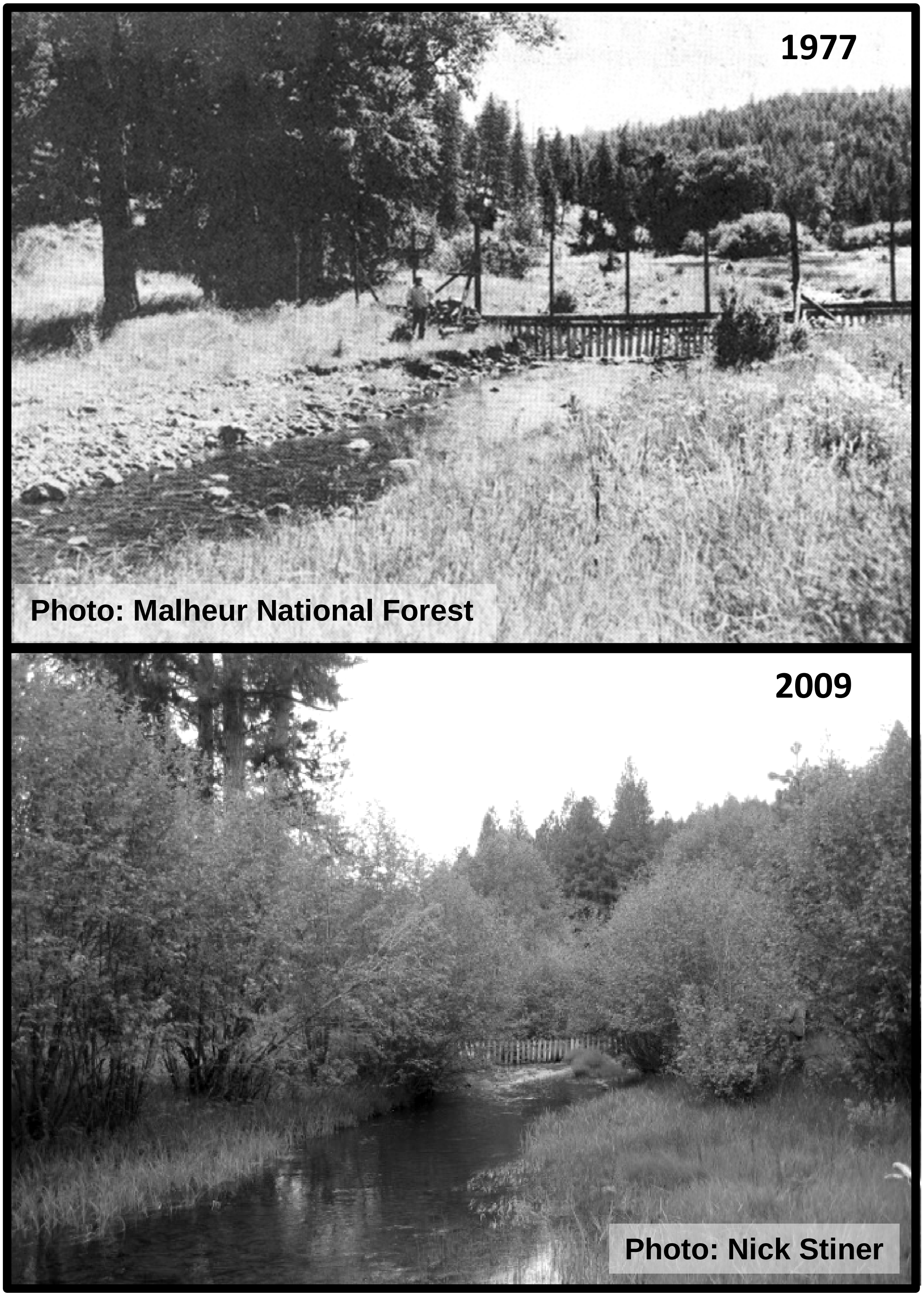


Figure 2. Repeat photographs of the Camp Creek ungulate exclosure. Camp Creek is a tributary of the upper Middle Fork John Day River (USDA 1989).

Figure 3. Heavily browsed thinleaf alder growing through landscaping tarp showing hoof prints of deer and elk.

Thus, in the absence of livestock grazing, it appeared that browsing by deer and/or elk might be limiting restoration of woody riparian vegetation within much of the riparian zone of the upper Middle Fork John Day River.

Goals and objectives

To document browse effects on riparian plantings, the Confederated Tribes of the Warm Springs Reservation partnered with the US Forest Service, Pacific Northwest Research Station to establish a number of small browsing exclosures surrounding riparian plantings on both the Forrest and Oxbow Conservation Areas.

Hypotheses

If browsing by deer and elk was, in fact, the primary factor limiting the growth of the riparian plantings, we hypothesized that substantial growth would occur once deer and elk were excluded.

Site Selection

Sites were selected in locations previously planted with native riparian seedlings and cuttings.

Methods

Most of the valley floor within both the Forrest and Oxbow Conservation Areas was fenced to exclude livestock, and in 2006, some 72,000 native riparian hardwoods and conifers (Table 1) were planted to restore woody riparian vegetation. In order to control competition from grasses, strips parallel to the channel were tilled using a disc-harrow and then covered in landscaping cloth. Cuttings and seedlings were planted through the landscaping cloth.

Browse measurements

Each strip of landscaping cloth and associated plantings was numbered and 22 strips were randomly selected on both the Forrest and Oxbow Conservation Areas. On each randomly selected tarp, we randomly selected the upstream half, or downstream half of the tarp, as the location for the browse exclosure; the unfenced "control" plot was located in the opposite direction. The exclosures were established in late spring of 2009, at the start of the growing season.

Exclosures measured 20-ft wide and 35 feet long. Eight-foot tall fences were constructed with steel T-posts and net-wire fencing and thus excluded deer, elk, as well as beaver. Unfenced control plots were located 50 feet from the fenced exclosure and also measured 20-ft wide and 35 feet long. On Average, 20 planting locations were present within each exclosure with a similar number within each matching unfenced "control". Because plantings were regularly spaced and because the landscaping cloth had been cut to allow planting through the cloth, we could easily identify locations where the plantings had already died by 2009. In most cases, we could not identify the dead plantings to species.

We began monitoring in early summer 2009. We used ground staples and numbered tags to identify each individual within our plots. We collected data on initial conditions in early June 2009 and remeasured the plots at the end of the growing season in October 2009 and again in October 2010. On each measurement date we relocated each numbered individual and measured the height and diameter of its canopy. Height was recorded as the distance from the ground surface to the highest point of the canopy; diameter was measured twice, once parallel to the length of the landscaping strip and once perpendicular. We chose this method, rather than attempting to measure the widest and narrowest diameters, because it would be repeatable, ensuring that measurements would be comparable across dates.

Table 1. Numbers of native riparian hardwoods, by species, contracted for planting across the Oxbow and Forrest Conservation Areas (Number Planted) and the number of living plantings recorded on the browse-excluded fenced plots (Number Unbrowsed) and the unfenced control plots (Number Browsed). The percent contribution of each species, to the total number planted or sampled, is also provided. All plantings were seedlings except that 8,600 cuttings were planted for black cottonwood and all willow and redoiser dogwood were planted as cuttings.

Species	Number Planted	%	Number UnBrowsed	Number Browsed	%
Black Cottonwood *Populus balsamifera trichocarpa*	16,100	22.27%	12	3	1.98%
Black Hawthorn *Crataegus douglasii*	1,000	1.38%	24	28	3.96%
Chokecherry *Prunus virginiana*	5,600	7.75%	117	107	19.31%
Common Snowberry *Symphoricarpos albus*	3,500	4.84%	81	58	13.37%
Golden Currant *Ribes aureum*	3,500	4.84%	45	62	7.43%
Thinleaf Alder *Alnus incana tenuifolia*	5,000	6.92%	53	60	8.75%
Ponderosa Pine *Pinus ponderosa*	8,500	11.76%	121	117	19.97%
Quaking Aspen *Populus tremuloides*	6,970	9.64%	78	86	12.87%
Rose (Nootka & other spp) *Rosa nutkana*, etc.	3,500	4.84%	46	41	7.59%
Willow Spp. (Pacific, Arryo,Greanleaf,Coyote) *Salix* spp.	8,600	11.89%	22	21	3.63%
Blue Elderberry *Sambucus nigra cerulea*	500	0.69%	0	2	0.00%
Douglas Fir *Pseudotsuga menziesii*	900	1.24%	2	1	0.33%
Mallow Ninebark *Physocarpus malvaceus*	10	0.01%	0	0	0.00%
Redosier Dogwood *Cornus sericea*	8,600	11.89%	5	1	0.83%
Redstem Ceanothus *Ceanothus sanguineus*	10	0.01%	0	0	0.00%
Snowbrush Ceanothus *Ceanothus velutinus*	10	0.01%	0	0	0.00%
SUM =	**72,300**	**100.00%**	**606**	**587**	**100.00%**

Here we report volume growth, where the volume of each individual was calculated as an ellipsoid whose dimensions were height and cross diameters, so that:

Volume = 4/3 * Pi * Height * Diameter1 * Diameter2

To calculate “volume growth”, we subtracted the initial volume for June 2009 from the ending volume in October 2010. The change in volume over this time period represented the growth accumulated over two growing seasons. We then averaged the volume growth for all individuals of a single species for the browse-excluded individuals and the unfenced control individuals.

We also calculated a simple “browse index” to compare among species, based on the relative difference in the average volume growth between the browse excluded plants and the unfenced controls.

Survival surveys

The riparian plantings were monitored to determine the rate at which different species survived over the first few years. Seedlings were planted in three rows along the length of the landscaping strips. To estimate seedling survivorship, the number of surviving individuals were tallied, by species, along the outer row of the landscaping tarp. This count was multiplied by 3 (for the 3 rows) to estimate the total number of surviving individuals. This estimate was compared to planting records which recorded the number of seedlings contracted to be planted, giving an overall estimate of survival. These tarp counts did not include the cuttings and thus do not include either willow or redosier dogwood. Also, the survivorship of black cottonwood was only calculated from the number of planted seedlings.

Survival rates were also estimated from the browse plots (fenced and unfenced controls, combined). We recorded the number of dead or missing plants when we laid out our plots in 2009 and added this to the total number of live plants measured to get an expected total of 1,729 individuals planted in our plots. We then used the number of individuals of each species originally contracted to be planted to calculate the expected proportion of individuals of each species. We multiplied the total number of plantings (1,729) by this proportion to estimate the expected number of individuals that should have been encountered in our plots. Finally, we estimated survival by dividing the number of live individuals of each species recorded in the plots with the expected number planted.

Results

Summary of Analyses

Excluding deer and elk resulted in substantial growth of the plantings as is obvious in the repeated photo pairs (Figure 4). The measurements, however, are able to provide quantitative illustration of the severe impacts of browsing in the absence of livestock.

Unfenced controls of three species – aspen, black cottonwood, and snowberry – actually decreased in size in the 2 growing seasons over which they were monitored (Figure 5). Initial measurements were taken in June 2009, at the time of the initial flush of spring growth. By October, that growth had been browsed away so that, on average, these species were slightly smaller in October 2010 than they had been in June 2009. In comparison, most browse-excluded individuals of these species grew rapidly and the average volume growth was substantial over the two year study (Figure 5).

Change in the average volume of unfenced control individuals of hawthorne and wild rose were very small whereas the average volume of browse-excluded individuals increased substantially over the two year study (Figure 5).

Choke cherry, willow, golden current, wild rose, all showed substantial volume growth – both for browse-excluded individuals as well as unfenced controls. However, that growth was substantially greater in the browse-excluded plots than in the unfenced controls (Figure 5).

While field observations showed that both Ponderosa pine and thinleaf alder were heavily browsed in June 2009, growth during the subsequent two growing seasons showed that the accumulated volume of these species was relatively little effected by browsing (Figure 5). It is important to note that the reduced browse effect was not due to these plants growing too tall to be browsed. By October 2010, Ponderosa pine averaged 100.7 cm tall and the tallest individual measured only 210 cm tall; thinleaf alder averaged 133.8 cm tall and the tallest individual measured 270 cm tall. Clearly, while very young, these two species were heavily browsed but with increasing age the browsing effect diminished such that unfenced control plants grew at very nearly the same rate as those protected from browsing (Figure 5).

Survival data appeared problematic. For several species, survivorship exceeded 100% - that is – more individuals were recorded in the years after planting than were expected, given the number of individuals of each species contracted to be planted (Figure 6). As such, the data are difficult to interpret. However, a few species showed extremely low survivorship under both methods – namely black cottonwood and redosier dogwood, and to a lesser extent, blue elderberry and willow. All other species showed relatively high survivorship (Figure 6).

Figure 4. Repeat photographs of riparian plantings in 2009 (top) and 2015 (bottom, photo from Emily Davis). Native woody seedlings were originally planted in 2006, with a ~10 foot spacing, using landscaping cloth to limit competition, and the entire riparian area was fenced to exclude livestock. Ungulate-exclusion cages (back ground, top photo) were erected in the spring of 2009, prior to the beginning of the growing season. By 2015, the landscaping cloth had been removed, but the exclosure fencing was left in place and shows how deer browsing is influencing the survival and growth of planted seedlings.

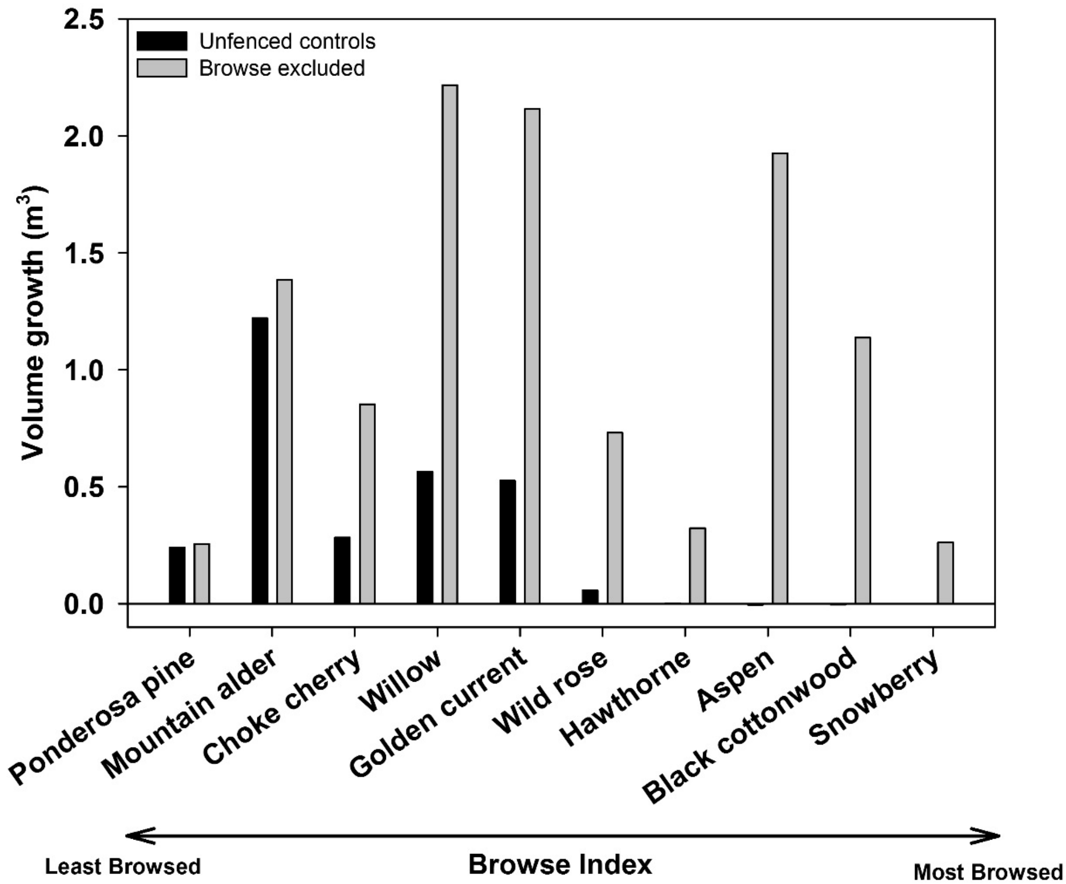


Figure 5. Comparison of volume growth between browsed and unbrowsed native woody riparian species planted on the Forrest and Oxbow Conservation Areas (all plots combined) along the upper Middle Fork John Day River. Species are sorted along a “browse index”, a simple measure of the difference in average volume growth between the browsed and unbrowsed individuals. Black bars are present, but too small to be visible for hawthorne, aspen, black cottonwood, and snowberry.

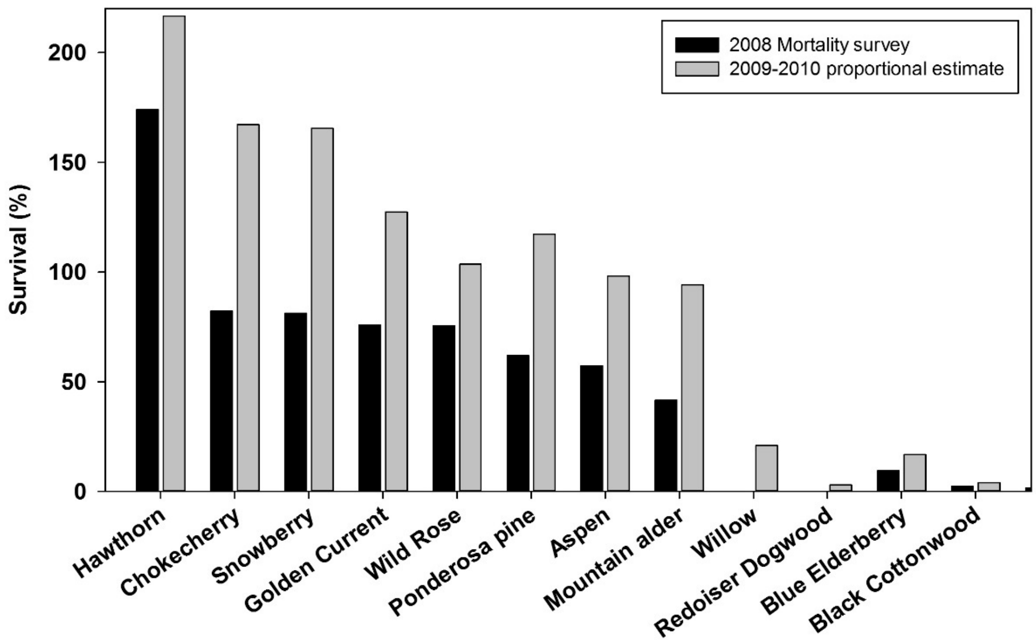


Figure 6. Estimated survival, by species, from a 2008 mortality survey and from a proportional estimate based on number of living individual encountered on the browse study plots (combining both the fenced plots and the unfenced control). Survival estimates, for both methods, were based on the number of individuals of each species that were originally contracted to be planted. It is likely that more individuals of hawthorn, chokecherry, and snowberry were planted than were originally contracted. Also, the 2008 mortality survey only included seedlings. Willow and redosier dogwood were only planted as cuttings and thus there is no estimate for survival of these species in the 2008 mortality survey.

Interpretation of findings

There are two critical caveats to interpreting these results: First, the seedlings and cuttings were planted through landscaping traps arranged in long strips. While this effectively controlled potential competition with other vegetation, is also may have influenced how deer and/or elk responded to the vegetation planted through the tarps. It may have increased susceptibility to browsing because the plants were easily found and completely exposed to browsing. Alternatively, lack of any ground cover (providing hiding cover, for example) may have dissuaded deer and elk from using these area as much as they might have if the landscaping tarps were not present. In any event, we did not have any study plots in areas lacking landscaping tarps so the effects of the tarps cannot be determined. Second, domestic livestock were entirely excluded from the study area – thus this study cannot determine the relative impact of deer and elk versus livestock (most commonly cattle) on the establishment and growth of the riparian plantings nor can it help inform how deer and elk might respond to riparian plantings in the presence of domestic livestock.

Discussion

Grazing by domestic livestock has long been identified as a major impact to riparian vegetation. There is growing awareness, however, that browsing by deer and elk may severely impact riparian vegetation. This study clearly demonstrates that, under present day conditions, deer and elk can have substantial impact on young riparian vegetation in actively managed landscapes. Thus, restoration efforts that use plantings of native woody riparian vegetation may have limited success if those plantings are not protected from deer and elk.

The survivorship data indicate that several species may prove difficult to establish, especially black cottonwood, redosier dogwood, blue elderberry, and willow. If these species are highly desirable components of the future restored woody riparian vegetation, additional methods should be explored to increase their survival.

Given the relatively high impact of browsing shown in this study, and if methods to limit browsing by deer and elk are not available, planting species that are less effected by browsing and have relatively high survivorship, such as Ponderosa pine and thinleaf alder, may allow the establishment of a forested riparian overstory with a hardwood understory. The historic composition of riparian vegetation in reaches with wide valley floors may not have been dominated by conifers, however. If so, restoring open meadows to a Ponderosa pine forest may not reflect restoration toward a historic reference condition. Further, a riparian zone dominated by only the few species that can be readily established in the face of browsing may lack the shrub and tree diversity necessary to meet other riparian management objectives.

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Appendix F – Projected Response of Riparian Vegetation to Passive and Active Restoration over 50 years

Steven M. Wondzell, USDA Forest Service, Pacific Northwest Research Station, Corvallis OR

Miles A. Hemstrom, Institute for Natural Resources, Oregon State University, Portland OR

Peter A. Bisson (retired), USDA Forest Service, Pacific Northwest Research Station, Olympia WA

Abstract

We modeled historical, current, and potential future conditions of riparian plant communities and salmon habitat quality in the upper Middle Fork John Day River of eastern Oregon using state and transition models. We focused our modeling efforts on stream reaches that had high intrinsic potential to support Spring Chinook Salmon or Steelhead. Using the models, we examined alternative management strategies for passive versus active restoration of riparian vegetation and salmon habitat quality. The results of our model projections appeared reasonable, data are not available to conduct a rigorous validation, thus model results should be interpreted with caution. Specifically, the results of modeled management alternatives should be interpreted as hypotheses of likely management outcomes. Simulation results suggested that recovery toward historic conditions occurs under both passive and active strategies. Recovery was relatively slower under passive restoration. Simulation results also varied by species of interest. Overall, our models suggested that restoration efforts significantly changed riparian and aquatic habitat quality over the time periods of decades. Our simulations also suggested that streams would not fully recover to the historical condition within 50 years (the duration of our simulations), even in the most aggressive restoration scenario we examined. These results indicate that river restoration investments need to be planned and evaluated over long time periods. Expectations for restoration outcomes needs to be tempered with a realistic understanding of the rate at which natural systems can recover from more than a century of Euro-American land-use.

Introduction

Background

Riparian and aquatic habitats in the Inland Northwest have been drastically altered by human activity after Euro-American settlement in the 1850-1900s (Hessburg and Agee 2003). Degradation of spawning and rearing habitat for salmon and steelhead has resulted in population declines or elimination of several salmonid species from their historical range (Nehlsen et al. 1991) and listing of several populations under the U.S. Endangered Species Act (USDA and USDI 2000). Protection of existing high quality habitat or restoration of essential streams is critical for conservation of these salmonids. Prioritization of restoration efforts and evaluation of their effectiveness across stream networks or large landscapes, however, is challenging. Further, predicting potential management effects on riparian and salmonid habitat quality is best done within a common framework that conceptualizes complex ecological relationships which change over time, natural disturbance dynamics, and management actions.

Few landscape-scale planning and assessment models exist for forecasting fine to mid-scale [i.e. 5th field hydrologic unit (USGS and USDA 2013)] riparian-aquatic habitat changes from plant succession, hydro-geomorphic processes, natural disturbances and land management. Wondzell et al. (2006) explored the use of state-and-transition models (STMs) to forecast the dynamics of aquatic and riparian habitats in eastern Oregon river networks. They conceptualized states, which were defined by riparian vegetation structure and composition, and transitions (plant succession, natural and anthropogenic disturbances) within a channel geomorphic classification system (Montgomery and Buffington 1997, 1998). Wondzell et al. (2006) demonstrated that the aquatic-riparian STMs could be useful for projecting riparian and aquatic habitat responses under different disturbance regimes.

Goals and objectives

The goal of this project was to examine the likely long-term outcomes of passive and active riparian restoration alternatives in stream reaches with the potential to provide high quality rearing habitat for cold-water dependent salmonids. We used STMs to simulate potential changes in riparian plant communities, stream attributes and salmonid habitat quality in the upper Middle Fork John Day River, northeastern Oregon. Our specific objectives were (1) to model historic, current, and future riparian vegetation and stream conditions, (2) to translate results into salmonid habitat indices for Spring Chinook Salmon (*Oncorhynchus tshawytscha*) and Steelhead (*O. mykiss*), and (3) to compare the outcome of different restoration alternatives on riparian vegetation structure and salmonid habitat quality over 50 years of implementation.

Methods

The aquatic-riparian STMs described here simulate the temporal dynamics of riparian vegetation. To apply these models, the stream network must be delineated into relatively homogeneous stream reaches classified into geomorphic types. The riparian zone around each reach must then be mapped and assigned to a potential vegetation type (PVT). Each reach polygon must also be attributed with the current vegetation type to provide a starting point for the model simulation. A brief description of these steps is provided below.

We used a 5-m LIDAR Digital Elevation Model (DEM) to delineate the upper Middle Fork John Day River stream network with the NetStream drainage routing tool (ESI 2009), a component of NetMap (Benda et al. 2007) and hydraulic geometry coefficients (Castro and Jackson 2001) calculated from field data collected in 2009. We classified stream reaches into six geomorphic types (Montgomery and Buffington 1997, 1998) based on channel gradient, contributing drainage area, and valley confinement thresholds derived from field data. These included cascade, step-pool, wood-forced step-pool, planebed, pool-riffle narrow and pool-riffle wide channel types.

The riparian PVTs were derived from upland STMs developed for the Blue Mountains. Riparian polygons in coniferous forest state classes were assigned to the same PVT as the adjacent upland stand and a new PVT was developed for riparian meadows. The final list of PVTs included dry mixed conifer forests, moist mixed conifer forests, dry ponderosa pine forests, riparian meadows and cold upper montane communities. State classes in each PVT were based on both successional age as well as structure and other stand attributes. These included early seral, non-forest states (barren, herbaceous native or exotic grass-forbs, and riparian shrubs) and forested states (conifer or cottonwood dominated forests of different ages and structure) and tracked single or multi-canopy layers, the presence and density of the shrub understory, and the abundance of large wood. Combining all possible geomorphic types with all PVTs resulted in 28 separate STMs, each of which was composed of tens of state classes.

Current riparian vegetation composition and structure within each reach was mapped with spatial modeling techniques and high density LIDAR point data acquired in 2008. Statistical models related remotely sensed data to field measurements from quantitative sub-plots to map vegetation attributes. Linear regressions were used to estimate continuous variables such as tree diameter at breast height and canopy cover (DBH) while RandomForest (Breimann 2001) algorithms were used to classify categorical variables such as cover type and strata. Methods used to classify the remotely sensed data were described in detail in Wondzell et al. (2012).

We built riparian state and transition models in the Vegetation Development Dynamics Tool (ESSA 2007) and simulated model runs with the Path Landscape Tool (ApexRMS 2013). These plant community succession models link vegetation communities along multiple pathways. Transitions are deterministic for successional development with transitions occurring at specified ages. Transitions are probabilistic for natural disturbances (e.g., fire, insects, disease) or management practices (e.g., grazing, thinning, forest harvest). The STMs are not spatially explicit.

Each state class in the STMs was linked to an expected channel morphologic condition. We used a 4-factor scale for each state class in the models to qualitatively rank their expected channel morphologic conditions: shade, erosion, undercut banks, large wood, pools, large pools, off-channel habitat, width-depth ratio, and riparian shrub abundance. We inferred the relative abundance of large wood, pools, undercut banks, and erosion from the cover type and structural stage of each state class in the models. These variables were then used in an expert systems model to rank the habitat quality (poor, fair, good, excellent) for both Spring Chinook Salmon and Steelhead. Additional details on the model development can be found in Wondzell et al. (2012) and at https://www.fs.fed.us/pnw/lwm/aem/projects/ar_models.html (accessed 7/27/2017).

We first simulated the historic condition by turning off all anthropogenic disturbances and running the model for 3,000 years under our simulated natural disturbance regime. Note that we do not assume that the natural disturbance regime has not changed for 3,000 years. Rather, the 3,000 year simulation was sufficiently long that our simulated historic condition was completely independent of the initial conditions. That is, regardless of the initial distribution of state classes, every 3,000 year simulation will converge toward the same simulated historic condition.

We modeled future riparian vegetation and salmonid habitat under four, 50-year long management alternatives by developing different sets of probabilities for disturbance transitions and management scenarios. We used current 2010 LIDAR-mapped conditions to initialize each modeled scenario. Each scenario was iterated 30 times to capture possible variation in model outputs and we used mean values to compare results of each simulation. Since the majority of the upper Middle Fork John Day watershed is managed by the USFS, we developed management scenarios that would be consistent with the agency's management prescriptions. We used expert opinion to parameterize current levels and trends of natural disturbance and management including wildfire dynamics, livestock grazing, wild ungulate browse pressure, timber harvest, stand management, and different river restoration approaches. Our primary goal was to evaluate how passive versus active management approaches might affect future vegetation conditions and salmonid habitat quality. The main management alternatives of interest were 1) a current management scenario reflecting typical US Forest Service management of eastern Oregon riparian zones between 1994 and 2010, 2) different livestock grazing prescriptions, 3) different levels of deer and/or elk browsing, and 4) intensive active restoration of stream channels coupled with riparian plantings.

Results reported here summarize results for Spring Chinook and Steelhead, independently. For Spring Chinook we only analyze results for reaches ranked with moderate to high intrinsic potential (IP; Burnett et al., 2007) to provide rearing habitat. That is, reaches that have geomorphic attributes that allows the potential to develop moderate or high quality habitat irrespective of the current quality of that habitat. For Steelhead, we only analyze results for reaches ranked with high IP. We often refer to these as either “Chinook reaches” or “Steelhead reaches”, and they compose only 33 and 129 km of the entire upper MFJD stream network, respectively (Fig. 1 and 2). Chinook reaches primarily encompass the wide valley floor portions of the mainstem upper MFJD whereas Steelhead are much more widely distributed throughout the network, including both the mainstem and many larger tributaries.

Results

Summary of Analyses

Current Management Scenario

Under the current management scenario, the 50-yr model runs were parameterized with moderate level of cattle grazing, high deer and/or elk browsing, no wildfire suppression, no forest management prescriptions, no anthropogenic vegetation or channel alteration, low active restoration of channels (0.01 km/y), and some conifer and riparian hardwoods planting. This parameterization reflects continuing changes in land management that have occurred in the latter decades of the 1900s. Our models suggest that the landscape, under current management conditions, will continue to change slightly into the future in response to changes in land use relative to the more intensive riparian land-uses of the early- and mid-1900s. Specifically, the current management scenario increased the abundance of riparian hardwoods and resulted in small improvements in spring Chinook rearing habitat quality even while cattle grazing continued at moderate intensity and deer and/or elk browsing was high (Fig. 3A-B). Combining all reaches with moderate to high IP for Spring Chinook, showed that the abundance of herbaceous states declined, whereas those with medium (15-40%) to dense (>40%) shrub cover, and either hardwood or conifer overstories increased. Excellent spring Chinook rearing habitat quality increased by about 10% in 50 years (Fig. 3B). In reaches with high IP for Steelhead, there was no change in abundance of herbaceous or shrub-dominated states whereas large changes occurred in the abundance of conifer states with medium density shrub understories (Fig. 3C). There was some improvement in steelhead rearing habitat from good to excellent condition but the amount of poor and fair habitat remained unchanged (Fig. 3D).

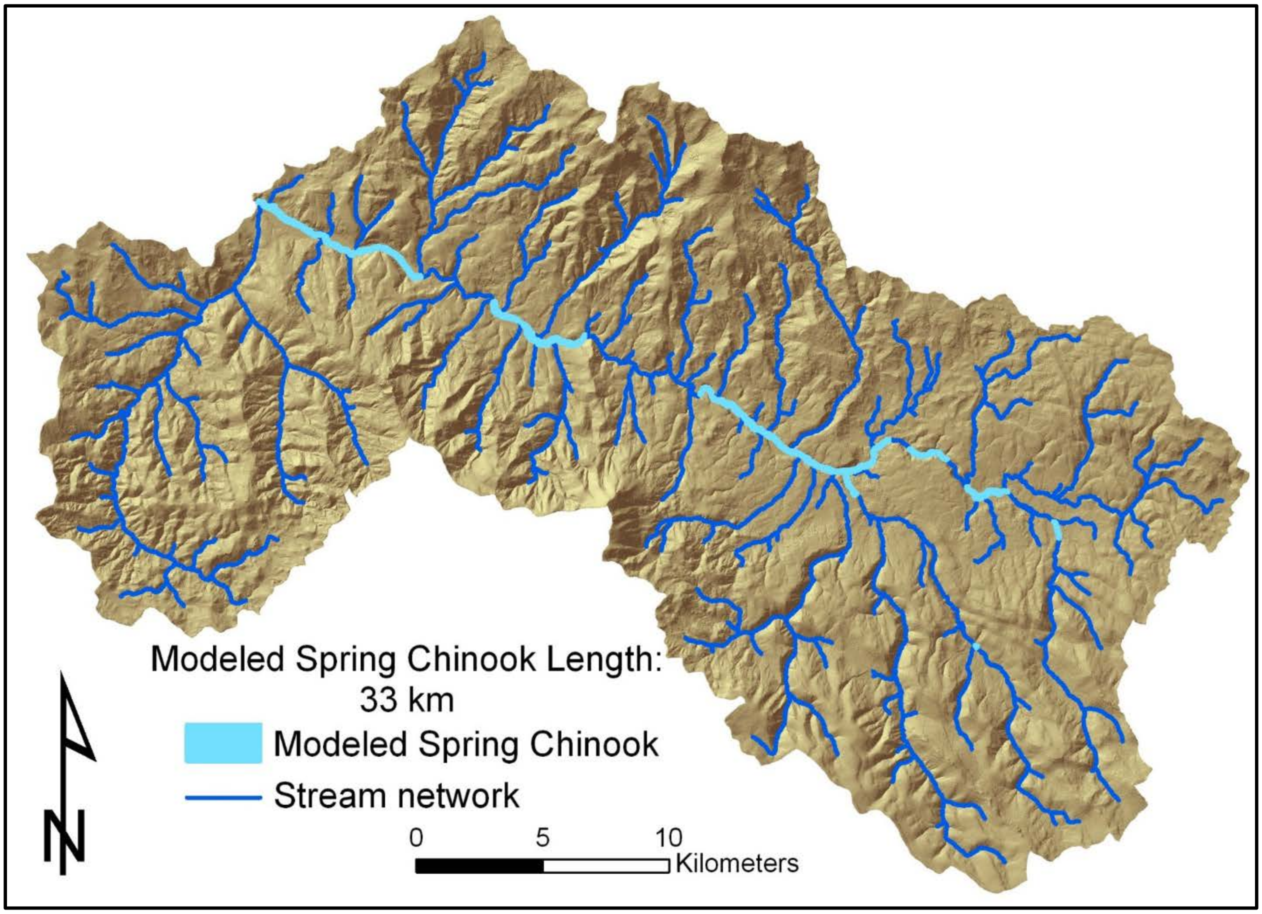

Figure 1. Portion of the stream network over which intrinsic potential models suggest the stream and valley floor attributes are such that they can provide moderate or high quality rearing habitat for Spring Chinook Salmon.

Grazing and Browsing Scenarios: We evaluated reduced grazing intensity and exclusion of deer and/or elk browsing as passive restoration approaches to improve riparian vegetation conditions and salmonid habitat quality. We first limited livestock, permitting only light grazing in riparian zones without changing browsing or other natural disturbances (Figure 4) and contrasted this with a scenario in which we completely excluded deer and/or elk from riparian areas (Figure 5).

Differences between the scenario with light intensity grazing and the current management scenario were relatively modest (Fig. 3 vs. Fig. 4). In the Chinook reaches, the largest differences were that exotic herbaceous state classes were not present in the lightly grazed scenario and the length of reach dominated by cottonwood was higher and by the end of the 50-year simulation there was substantially more reach length ranked as good or fair rearing habitat. Differences between these scenarios for the Steelhead reaches were also modest. The most obvious difference is that conifer with open shrub states did not persist in the lightly grazed scenario.

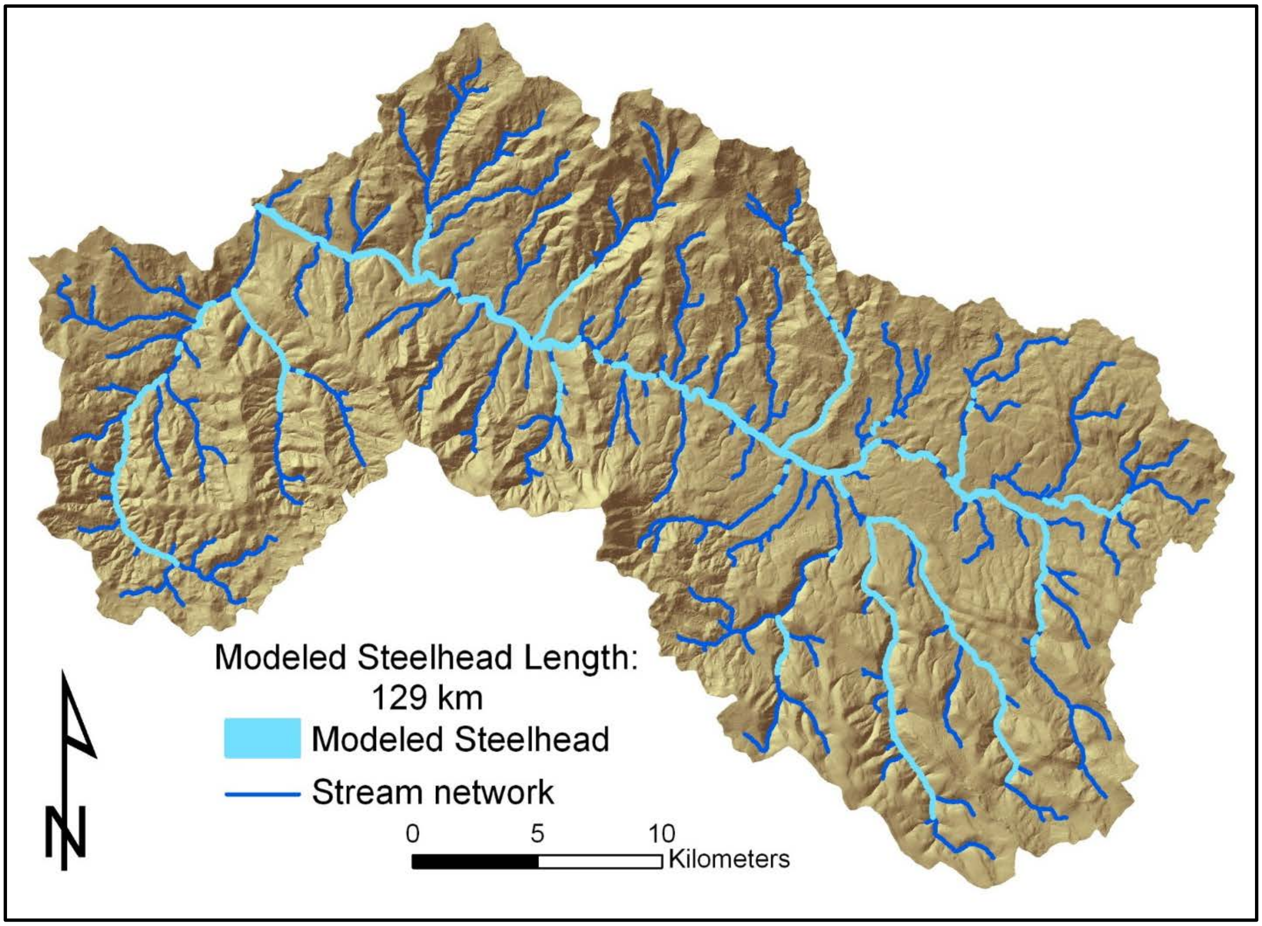

Figure 2. Portion of the stream network over which intrinsic potential models suggest the stream and valley floor attributes are such that they can provide high quality rearing habitat for Steelhead.

Eliminating deer and/or elk browsing in combination with reduced grazing intensity led to larger changes in the riparian vegetation when compared to either the current management scenario or the reducing grazing scenario (Fig. 5 vs. Fig. 3 and 4). In Chinook reaches, for example, all shrub state classes, whether in shrub dominated states or with shrub understories in forested states, transitioned to dense shrub cover (>40% canopy cover). After 50 years under this management scenario, the length of stream channel flowing through reaches dominated by cottonwood was also much greater and the length of conifer dominated stream channel declined. While the changes in riparian vegetation structure and composition were large, these did not lead to large changes in stream habitat quality for Spring Chinook (Figure 4B vs. 5B). Changes in the riparian vegetation in Steelhead reaches were similar to those in Chinook reaches, with dense shrub states becoming much more abundant and substantial increases in the amount of cottonwood. Again, the simulated changes in riparian vegetation did not lead to substantial changes in steelhead habitat quality.

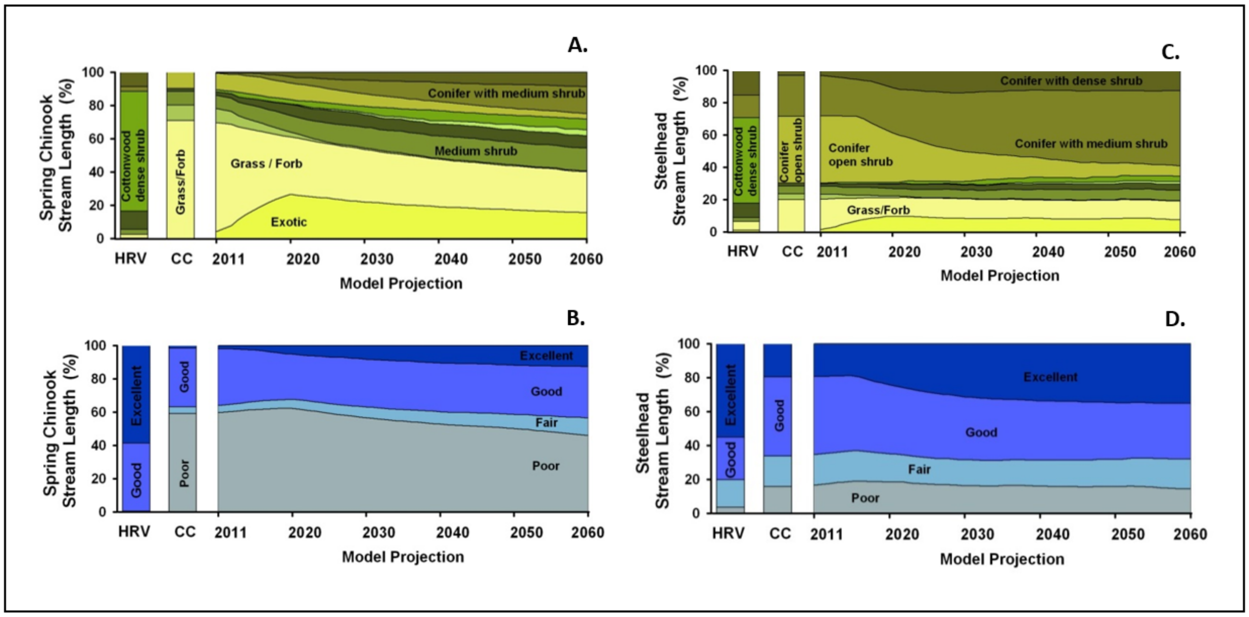

Figure 3. Simulated changes in riparian vegetation and stream habitat quality for portions of the stream network ranked with moderate or high intrinsic potential for Spring Chinook (A, B) and high intrinsic potential for Steelhead (C, D). The simulated historic condition (HRV) and LIDAR-derived current condition (CC) are given as bars on the left side of each graph. The time series composing the main body of each graph shows results of a 50-yr model simulation initialized with the current condition.

The current management scenario is shown here and includes moderate level of cattle grazing, high deer and/or elk browse, no wildfire suppression, no forest management prescriptions, no anthropogenic vegetation or channel alteration, low active restoration of channels, and some conifer and riparian hardwoods planting.

Combined active channel and passive riparian restoration scenario:

While we refer to this scenario in the text as the active restoration scenario, the restoration activities included both active channel restoration, reduction of grazing intensity, exclusion of deer and elk, and planting of riparian woody vegetation. This scenario implemented relatively high rates of active channel restoration (0.4 km/yr) as opposed the passive restoration and current management scenarios (0.01 km/yr). All other management prescriptions were the same as in the previous scenarios. The active channel restoration scenario led to substantial reduction in the abundance of grass/forb states and substantial increases in the abundance of cottonwood states in the Chinook reaches (Fig. 6A), when compared to any of the other scenarios (Fig 3A, 4A, and 5A). Active channel restoration also led to substantial increases in the proportion of Chinook reaches in good or excellent habitat condition (Fig. 6B). It is important to note that, in 50 years, active channel restoration at the rate of 0.4 km/yr would result in restoration of 20 km of channel and moderate to high Chinook IP reaches only include 33 km of the upper mainstem of the Middle Fork John Day.

Thus, 60% of the available Spring Chinook rearing habitat could be restored in 50 years under this scenario.

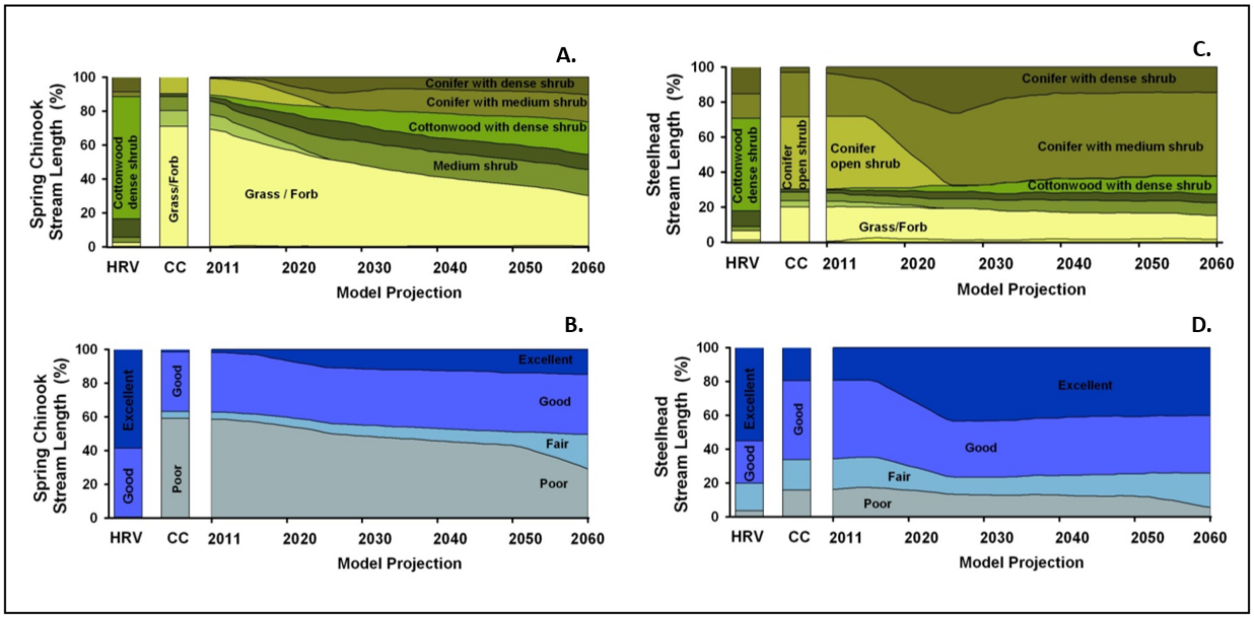

Figure 4. Simulated changes in riparian vegetation and stream habitat quality for portions of the stream network ranked with moderate or high intrinsic potential for Spring Chinook (A, B) and high intrinsic potential for Steelhead (C, D). The simulated historic condition (HRV) and LIDAR-derived current condition (CC) are given as bars on the left side of each graph. The time series composing the main body of each graph shows results of a 50 yr model simulation initialized with the current condition.

The light grazing intensity scenario is shown here and includes light cattle grazing, high deer and/or elk browse, no wildfire suppression, no forest management prescriptions, no anthropogenic vegetation or channel alteration, low active restoration of channels, and some conifer and riparian hardwoods planting.

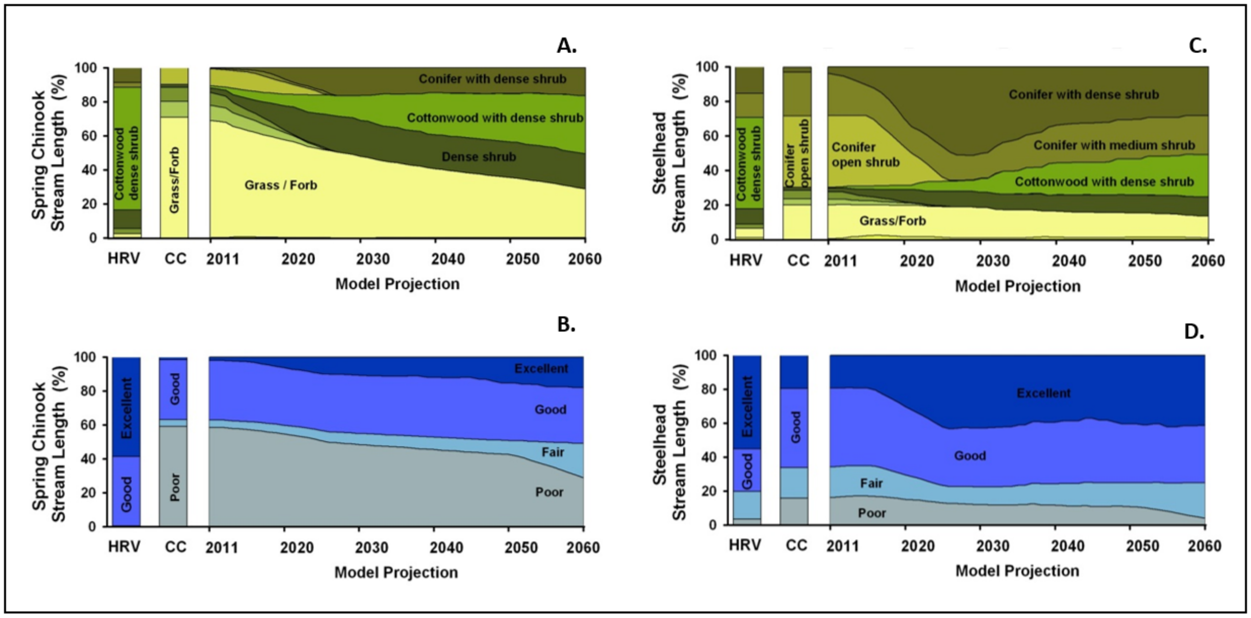

Figure 5. Simulated changes in riparian vegetation and stream habitat quality for portions of the stream network ranked with moderate or high intrinsic potential for Spring Chinook (A, B) and high intrinsic potential for Steelhead (C, D). The simulated historic condition (HRV) and LIDAR-derived current condition (CC) are given as bars on the left side of each graph. The time series composing the main body of each graph shows results of a 50 yr model simulation initialized with the current condition.

The light grazing and deer and/or elk exclusion scenario is shown here and includes light cattle grazing, deer and/or elk exclusion, no wildfire suppression, no forest management prescriptions, no anthropogenic vegetation or channel alteration, low active restoration of channels, and some conifer and riparian hardwoods planting.

Active restoration had little effect on riparian vegetation or rearing habitat quality for the Steelhead reaches (Fig. 6C-D). It is important to note, however, that 129 km of the upper Middle Fork John Day river network has high IP for Steelhead and a substantial number of those reaches are already ranked as good or excellent habitat. Thus, implementing active channel restoration at the rate of 0.4 km/yr will only treat 15% of the Steelhead reaches. This is a relatively small change, considering the length of channel already in good or excellent condition and thus, active channel restoration has little impact on the quality of the available Steelhead rearing habitat.

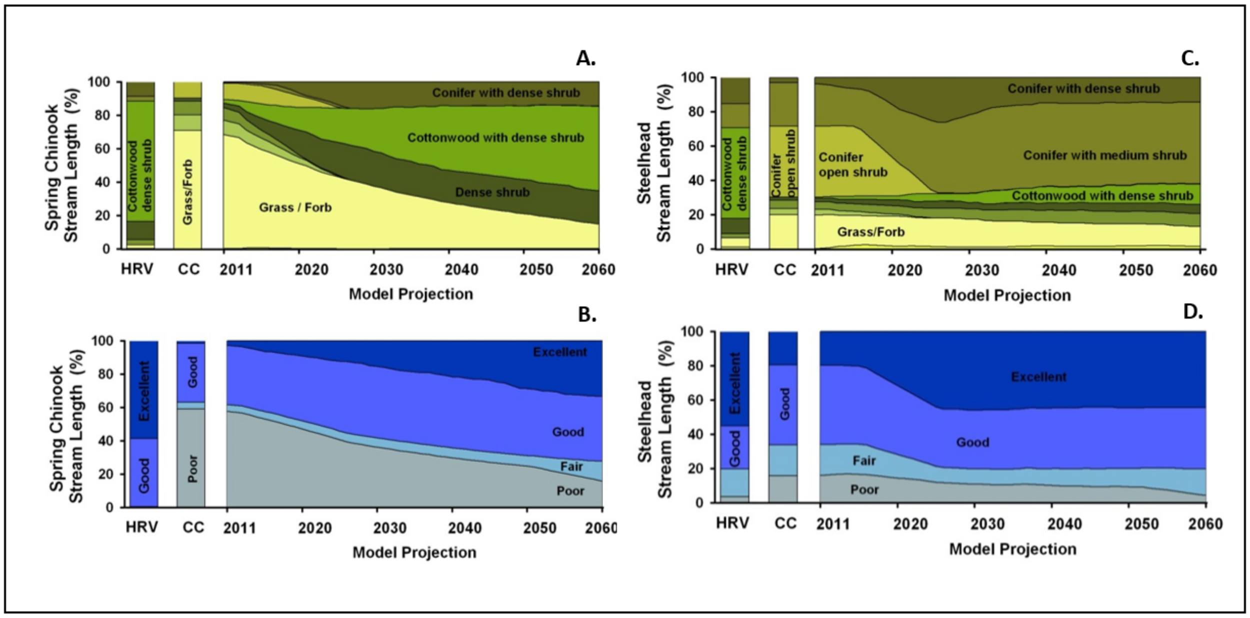

Figure 6. Simulated changes in riparian vegetation and stream habitat quality for portions of the stream network ranked with moderate or high intrinsic potential for Spring Chinook (A, B) and high intrinsic potential for Steelhead (C, D). The simulated historic condition (HRV) and LIDAR-derived current condition (CC) are given as bars on the left side of each graph. The time series composing the main body of each graph shows results of a 50 yr model simulation initialized with the current condition.

The active channel restoration with light grazing and deer and/or elk exclusion scenario is shown here and includes high active channel restoration, light cattle grazing, deer and/or elk exclusion, no wildfire suppression, no forest management prescriptions, no anthropogenic vegetation or channel alteration and some conifer and riparian hardwoods planting.

Discussion

The aquatic-riparian STMS are based on expert opinion, are not spatially explicit, do not account for uncertainty well, and rely on derivation of stream habitat indices from vegetation attributes. While the results of our model projections appeared reasonable wherever data were available for comparison, these comparisons did not provide detailed validation of all factors simulated in our models and results should be interpreted with caution. The models are not intended to provide detailed predictions of specific outcomes at the scale of a single reach or for a specific restoration project involving 100s of meters or a few kilometers of stream channel. Rather, the results of modeled management alternatives should be interpreted as hypotheses of likely management outcomes at the scale of a watershed (1 or several 5th-field hydrologic units).

The management alternatives we examined represent only a few of many management questions that could be explored. The models can be readily modified by other users to explore aquatic, riparian and terrestrial management scenarios, and to conceptualize watershed-specific riparian plant communities, channel morphologies, stream conditions, disturbance and management regimes. This adaptability makes the models useful for exploring various ways in which policy decisions or changes in disturbance regimes may influence future riparian vegetation and stream habitat conditions.

The use of the STMs to explore historic, current, and future conditions of riparian vegetation and stream habitat quality showed that current riparian vegetation communities, aquatic habitats, and disturbance regimes in the upper Middle Fork John Day River differ from historical conditions. Management strategies can be designed to move conditions closer to historical conditions and to improve habitat quality for species of concern. The scenarios we examined with our models suggested that restoration efforts could significantly change riparian and aquatic habitat quality over the time periods of decades, relative to our “current management scenario”. Critical factors were the exclusion of herbivores to allow establishment of woody riparian vegetation and active restoration of the stream channel where it had been impacted by dredge mining, levees, and boulder barbs and riprap.

Our simulations did show that restoration could be substantially accelerated through active restoration practices. However, active restoration is more expensive than passive restoration. Consequently, the choice between active and passive restoration needs to be made carefully. Our models simulated substantial increases in aquatic habitat quality for Spring Chinook Salmon under the active restoration scenario relative to passive restoration. In contrast, active restoration simulated relatively little influence on habitat quality for Steelhead. Our simulations suggest that active restoration will have a bigger impact on species (such as Spring Chinook) that have a limited potential spatial distribution within a landscape or stream network, and where a significant proportion of the available habitat is in poor condition. Under these conditions, a relatively small investment in restoration can have a large impact on the amount of high quality habitat available. In contrast, using active restoration techniques to improve habitat for a widely distributed species (such as Steelhead) seems less feasible. The IP models suggest that Steelhead habitat is abundant throughout the basin and our model analysis of current conditions suggests that a substantial portion of the Steelhead reaches are currently in good or excellent condition. If our simulation results accurately reflect the habitat conditions for Steelhead within the upper MFJD stream network, then large portions of the river network would need to be restored to substantially increase the proportion of the stream network that is in good or excellent condition for steelhead.

Our simulations suggested that streams would not fully recover to the historical condition within 50 years (the duration of our simulations), even in the most aggressive restoration scenario we examined. This should not be surprising. Full recovery will require regrowth of riparian trees, recruitment of large wood, and fluvial transport and reworking of sediment throughout the river corridor. It takes time for trees to grow. Further, the flood events that reshape river corridors are relatively rare episodic events. The results clearly indicate that river restoration is a long-term investment. Expectations for restoration outcomes need to be tempered with a realistic understanding of the rate at which natural systems can recover from more than a century of Euro-American land-use.

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Appendix G – MFIMW Water Temperature Monitoring

Kasey Bliesner, Oregon Department of Fish and Wildlife
Emily Davis, Confederated Tribes of the Warm Springs Reservation of Oregon
Justin Rowell, North Fork John Day Watershed Council

Abstract

Elevated water temperature has been identified as the primary limiting factor for salmonids to be addressed in the MFJD subbasin. Water temperature loggers were placed in the mainstem MFJD and tributaries in order to provide temperature data to calibrate a Heat Source model and to monitor EPA-defined total maximum daily load (TMDL) for water temperature of 18°C. Between 2005 and 2016, 122 water temperature loggers were operated in the mainstem MFJD between Bridge Creek and Big Creek and in 26 tributaries. Seven day average daily maximum temperatures and proportion of summer days with maximum temperatures exceeding 18°C were calculated for mainstem MFJD and Bridge Creek loggers. Summer water temperatures reported as maximum seven day average daily maximums were above the recommended 18°C for coldwater salmonids for all locations and all years. Restoration activities in the MFJD designed to improve water temperatures are recent, within the last five years, and have not matured over this short time period to effect a watershed-level change in temperature values. Water temperature monitoring should continue, with a clear monitoring plan in place, in order to detect changes due to restoration, update temperature models, and determine TMDL compliance.

Introduction

Background

Past land management activities in the MFIMW, including grazing, mining, logging, clearing forest for pastureland, and irrigation, have combined to decrease riparian vegetation, decrease floodplain connectivity, and decrease instream habitat complexity, which in turn has decreased summer low flows and increased summer water temperatures (CBMRCD 2005, BOR 2008, DEQ 2010). Water temperature has been identified as the primary limiting factor to be addressed in the MFJD subbasin by the UMFWG, in the John Day Subbasin Plan and in the Mid-Columbia Steelhead Recovery Plan (UMFWG 2011; CBMRCD 2005; Carmichael 2010).

In 2003, the EPA developed and published regional water quality standards intended to assist agencies in adopting appropriate temperature standards consistent with obligations under the Clean Water Act and the Endangered Species Act, and to protect cold-water habitats. Water temperature standards were published as "total maximum daily load" (TMDL), with a recommended 18°C as the threshold for coldwater species, including salmonids (EPA 2003). For monitoring TMDL threshold compliance, the EPA recommends water temperature measurements be taken in key areas on an hourly basis throughout the summer (June-October) so that average weekly maximum temperatures (7DADM) can be calculated. The 7DADM and maximum 7DADM are the preferred metrics because they reflect the maximum temperatures in a stream that a fish would be exposed to over a week-long period (EPA 2003).

Goals and objectives

Water temperature loggers were placed in the mainstem MFJD and tributaries in order to provide temperature data to calibrate the HeatSource model (Diabat 2014; Hall 2015). The HeatSource model addresses how restoration might influence water temperatures and an understanding of the casual mechanisms linking stream habitat restoration, water temperature, and changes in salmonid production at the watershed level. At a more basic, observational (non-statistical) level, water temperature loggers in the MFIMW were used to consider the following questions:

- Do water temperatures meet the EPA TMDL threshold of 18°C (EPA 2003)?
- Which tributaries appear to have a cooling or warming influence on the mainstem MFJDR?
- What is the temperature pattern in Bridge Creek?
- Does Bates Pond influence temperatures in Bridge Creek?
- Has there been any change in summer water temperatures at a watershed scale since the beginning of the MFIMW?
- This question will be difficult to answer without incorporating air temperatures and flow data.
- As compared to the reference SFJDR?
- Before/After initiation of restoration in the MFIMW?
- Certain restoration activities are more likely than others to affect water temperatures. Investigating specific restoration activities is outside the scope of this current report.

Site Selection

In general, loggers were placed in the mainstem MFJD between Bridge Creek and Big Creek, above and below major tributaries, and in key restoration areas. Additional loggers were placed in 26 tributaries, including the major tributaries Big, Bridge, Camp, Davis, Granite Boulder, Ruby and Vincent creeks (Figure 1; Table 1). Three temperature loggers were placed on the mainstem MFJD approximately 20 km downstream of the MFIMW area.

Methods

A total of 122 water temperature loggers were deployed in the Middle Fork IMW study area between 2005 and 2016 (Figure 1; Table 1), 74 of which were collecting data in 2016. The majority of these loggers were in the mainstem between Bridge Creek and Summit Creek, many of them near upstream and downstream locations of Camp, Granite Boulder, Bridge, and Vinegar Creeks. There were also loggers deployed in 26 tributaries and three loggers on the mainstem MFJD 20 km downstream of the MFIMW. Responsibility for logger deployment and retrieval was dependent on land ownership. Three different agencies conducted the majority of the fieldwork over the 12 year monitoring period: the NFJDWC, CTWSRO, and TNC.

Before deployment, loggers were tested for accuracy following protocols provided by ODEQ. Loggers were typically deployed in April and extracted in late October to capture summer water temperatures. Specific protocols for logger placement and deployment can be found in Appendix B.

Water temperature data were typically downloaded once in late fall when loggers were removed for the winter. Downloads were completed using HOBO software and converted to .csv files (Onset 2017), and provided to the NFJDWC for upload to an Access database created by ISEMP. After upload, data quality control of the data was completed by either the Monitoring Project Lead at the NFJDWC or by the MFIMW data steward.

Data was quality checked using graphics tools built in to the Access database. Each data collection event was graphed and visually inspected. Individual data points were ranked as “Accepted”, “Uncertain” or “Failed”. When possible water temperature data was graphed alongside air temperature data. Data points were failed when extreme high temperatures were observed (greater than 32°C), or when the hourly pattern matched air temperatures, indicating that the logger was likely out of the water or exposed to sunlight. Data points were marked “Uncertain” when temperatures were recorded below -1°C (indicating the logger was frozen) or when temperatures appeared out of range, but the individual conducting QA/QC had no context for failing the data point

Following guidelines from the EPA and ODEQ, seven day average of the daily maximum (7DADM) was calculated for the mainstem MFJD and Bridge Creek logger locations for all years available. For a logger to be included in analysis, the dataset had to include at least 90 days of concurrent summer data and data had to be marked as “accepted” during quality control. For this analysis, summer was considered June-September. The 7DADM was calculated by first determining the maximum daily temperature, then averaging maximum daily temperatures for that day and the previous six.

Following the EPA cold-water threshold of 18°C, the proportion of summer days with maximum daily temperatures above 18°C was calculated. The total number of summer days, and the number of those days where the daily maximum was greater than 18°C were determined. The proportion of summer days greater than 18°C was calculated by dividing the number of days greater than 18°C by the total number of summer days. Days greater than 18°C were plotted by rkm and by year for loggers on the mainstem MFJD and Bridge Creek.

Locations and years in which loggers were placed above and below tributaries on the MFJD were identified and daily maximum temperatures were calculated. The difference between the upper logger and the lower logger was calculated and plotted for each year of available data.

The proportion of days greater than 18°C and 7DADMs were calculated for 229 logger-year combinations in the mainstem MFJD, and 71 logger-years combinations in Bridge Creek. Where loggers were located on the mainstem MFJDR above and below tributaries, differences in water temperatures between the upstream and upstream loggers was calculated for 32 logger-year combinations (Appendix Table 1).

For the comparison between the SFJD and MFIMW, 7DADMs were calculated for two MFJD loggers at the downstream end of the MFIMW, and 7DADMS were calculated for the logger on the rotary screw trap on the SFJD. Difference between the MFJD and SFJD 7DADMs were calculated and a mean line of the differences was plotted.

Results

Water temperature data were uploaded into the MFIMW database from 2005-2016 for a total of 122 sites which resulted in 2,684,764 individual temperature records. The most consistent datasets occurred at two locations on the TNC properties, with data collected from 2007-2015 (Figure 1; Appendix Table 2).

Also included in the database were air temperature data for 12 sites; two in conjunction with TNC water temperature loggers, and 10 sites were added in 2016 in associated with CTWRSO restoration sites.

Considerable effort was undertaken in 2016 by co-managers from NFJDWC, CTWRSO, OSU and ODFW to verify logger metadata; resulting in accurate locations, defined responsibilities, logger status (active or inactive), and linear stream referencing for each logger. Summaries of logger locations and years of available data were created to assist co-managers in determining what data were available for cross-analysis (Appendix Table 2).

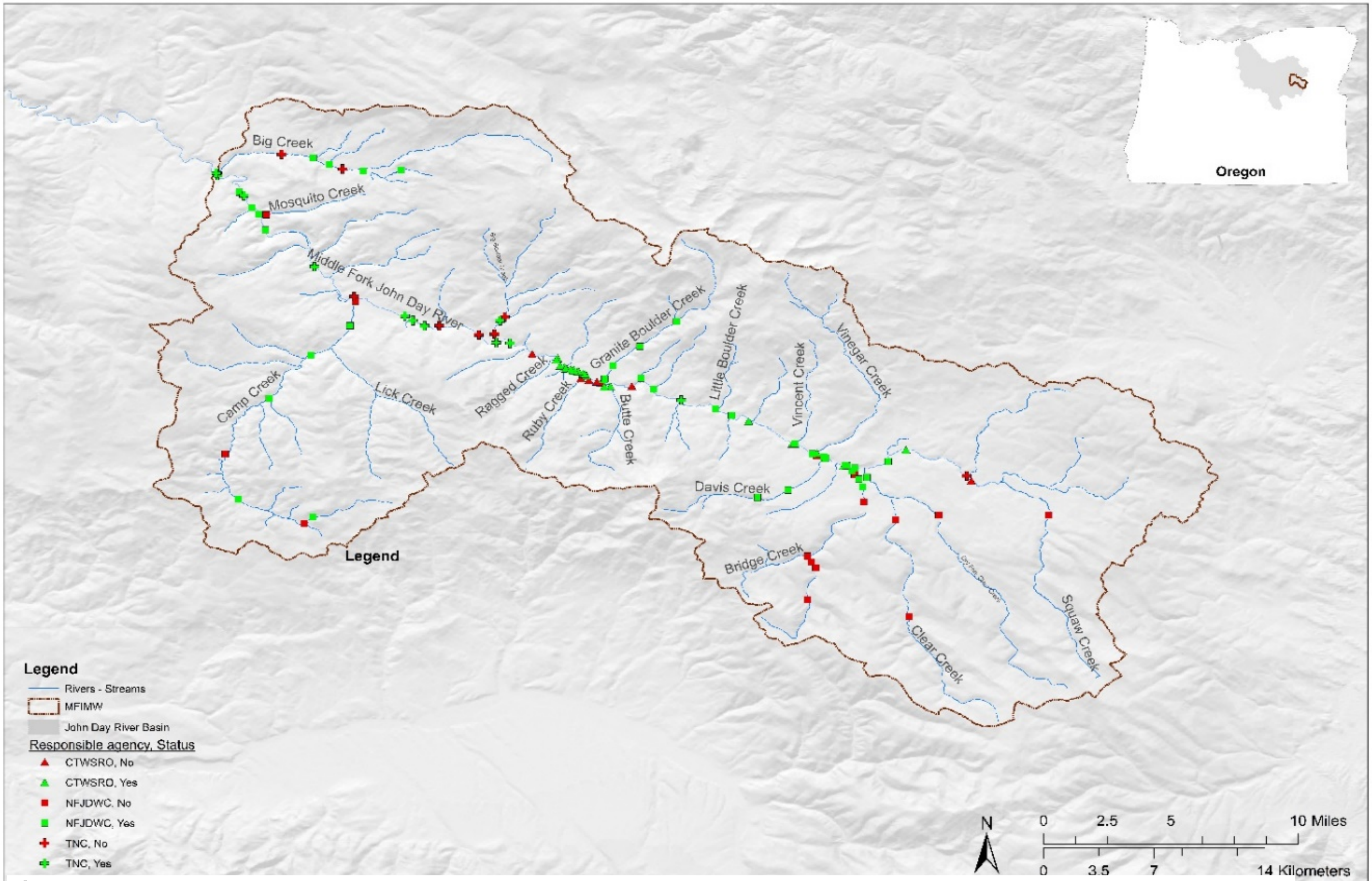

Figure 1. Map of MFIMW water temperature loggers by agency responsibility and active or in-active status

MFJD Maximum 7DADM

A summary of yearly maximum 7DADM in the MFJD shows highest 7DADMs occurring in the lower sections of the MFJDR below rkm 88, and lower 7DADMs occurring between rkm 90 and rkm 100 (Figure 2). The lowest maximum 7DADMs were observed in 2011, and the highest occurred in years 2013 and 2015 (Figure 3).

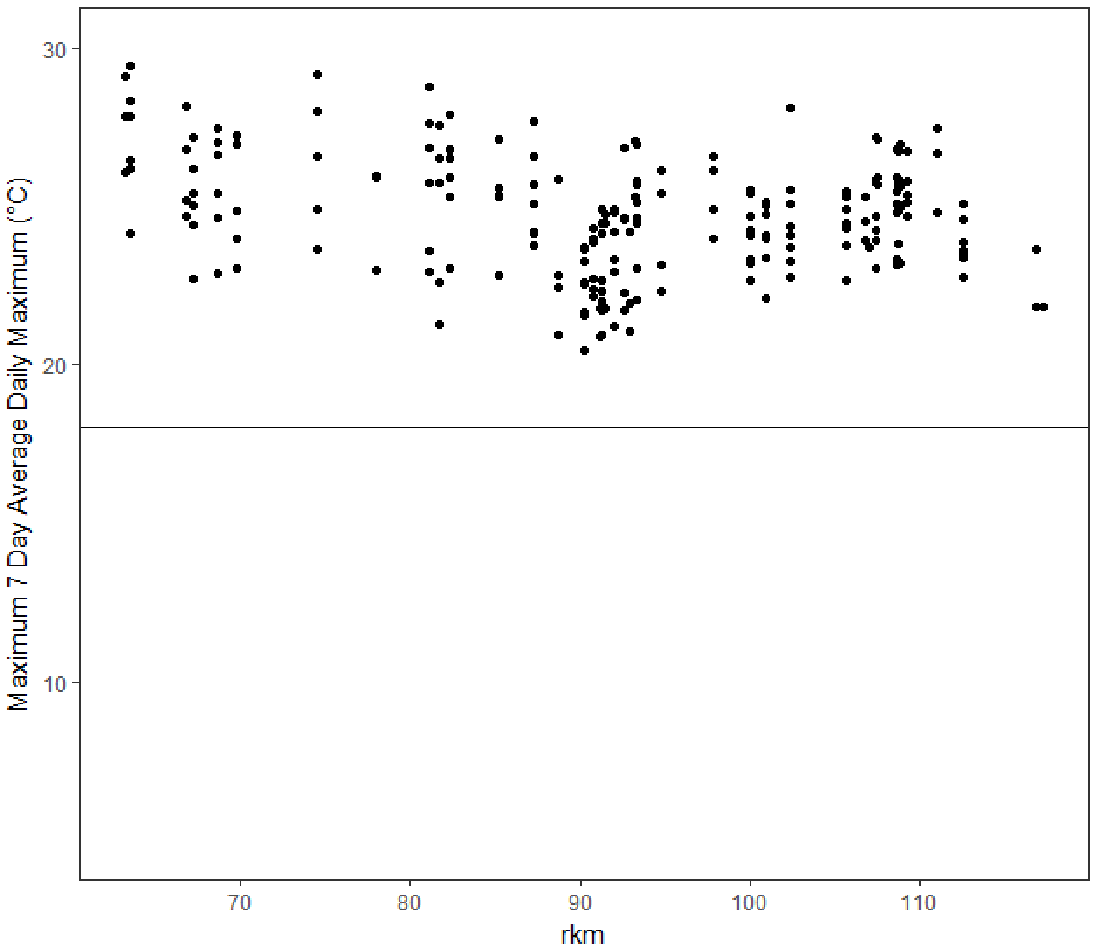

Figure 2. Maximum seven day average daily maximum (7DADM) by rkm (left to right, downstream to upstream) on the MFJD in the MFIMW area for all loggers and years. Rkm 0 represents the mouth of the MFJD. Black line is 18°C, the EPA cold-water threshhold.

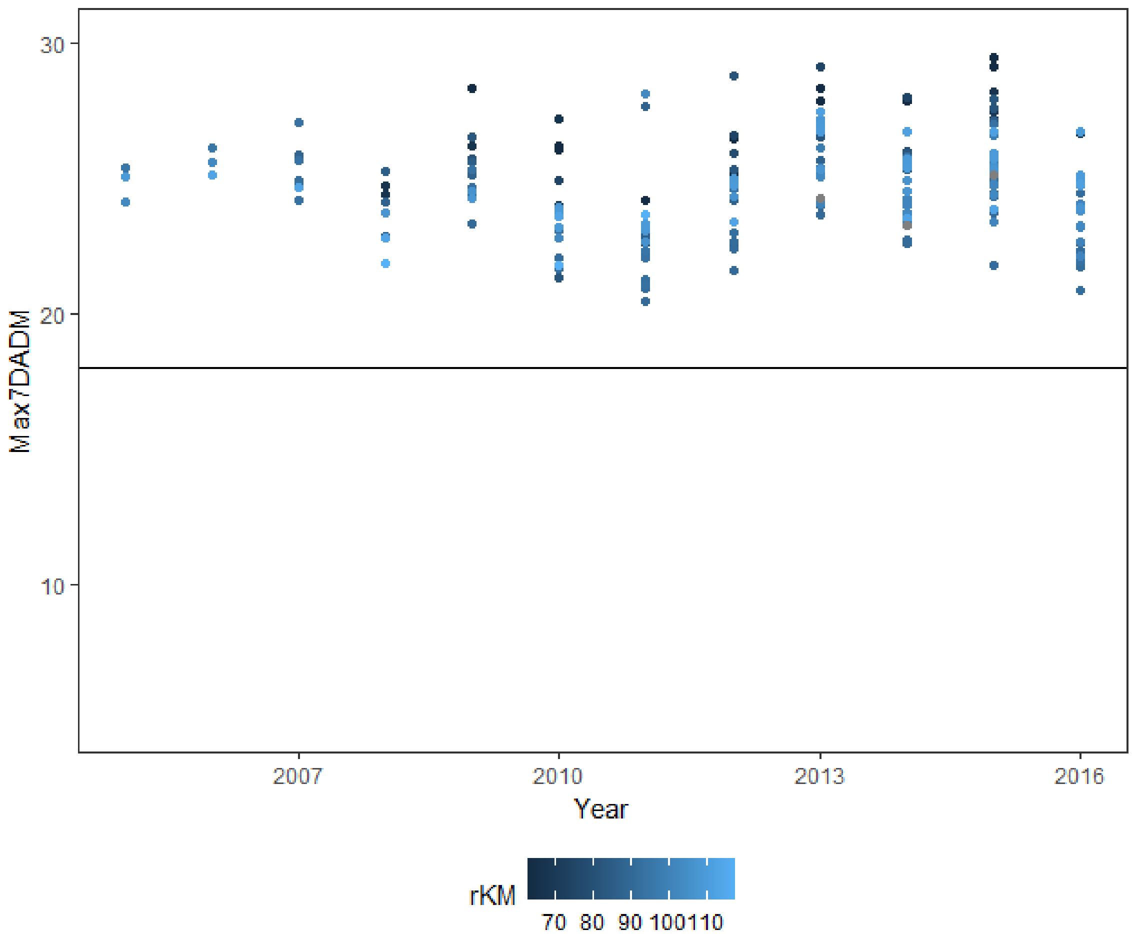

Figure 3. Maximum seven day average daily maximum temperature (°C) (7DADM) by year on the MFJD in the MFIMW area for all loggers and years. Rkm 0 represents the mouth of the MFJD. Black line is 18°C, the EPA cold-water threshhold.

In general, maximum 7DADMs occurred in the MFJD in August. For the years 2013 and 2015, the maximum 7DADMs occurred in July (Figure 4).

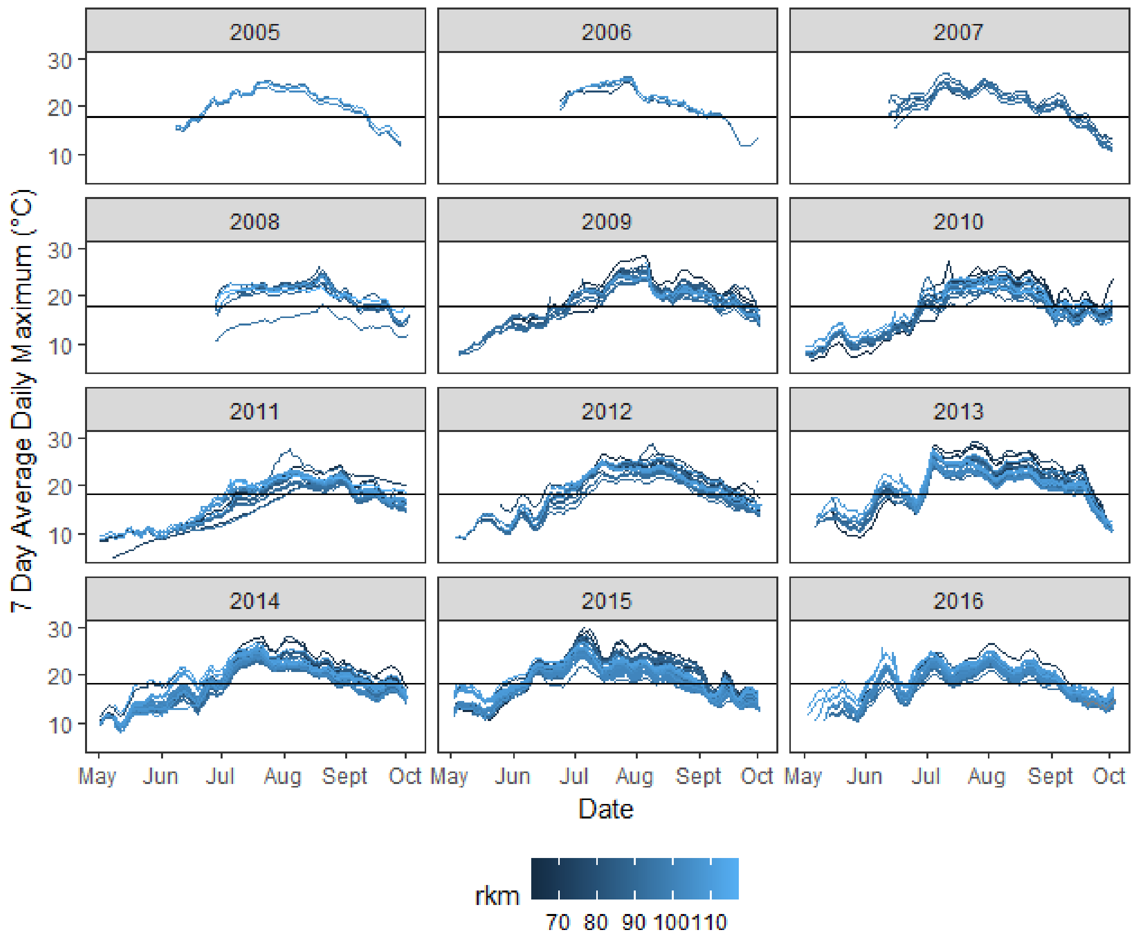

Figure 4. Seven day average daily maximum (7DADM) temperatures °C for loggers on the mainstem MFJD. Rkm 0 is at the mouth of the MFJD. Black line is 18°C, the EPA cold-water threshhold.

MFJD proportion of days with maximum daily temperatures greater than 18°C

The proportion of summer days with a maximum temperature greater than 18°C shows a similar pattern to the maximum 7DADM analysis. Sites with more days greater than 18°C occurred in the upper and lower sections. Sites with fewer days greater than 18°C temperatures occurred between rkm 91 and rkm 100(Figure 6). The highest and lowest proportion of summer days with maximum daily temperatures above 18°C occurred in 2011 at two different locations both (Figure 6).

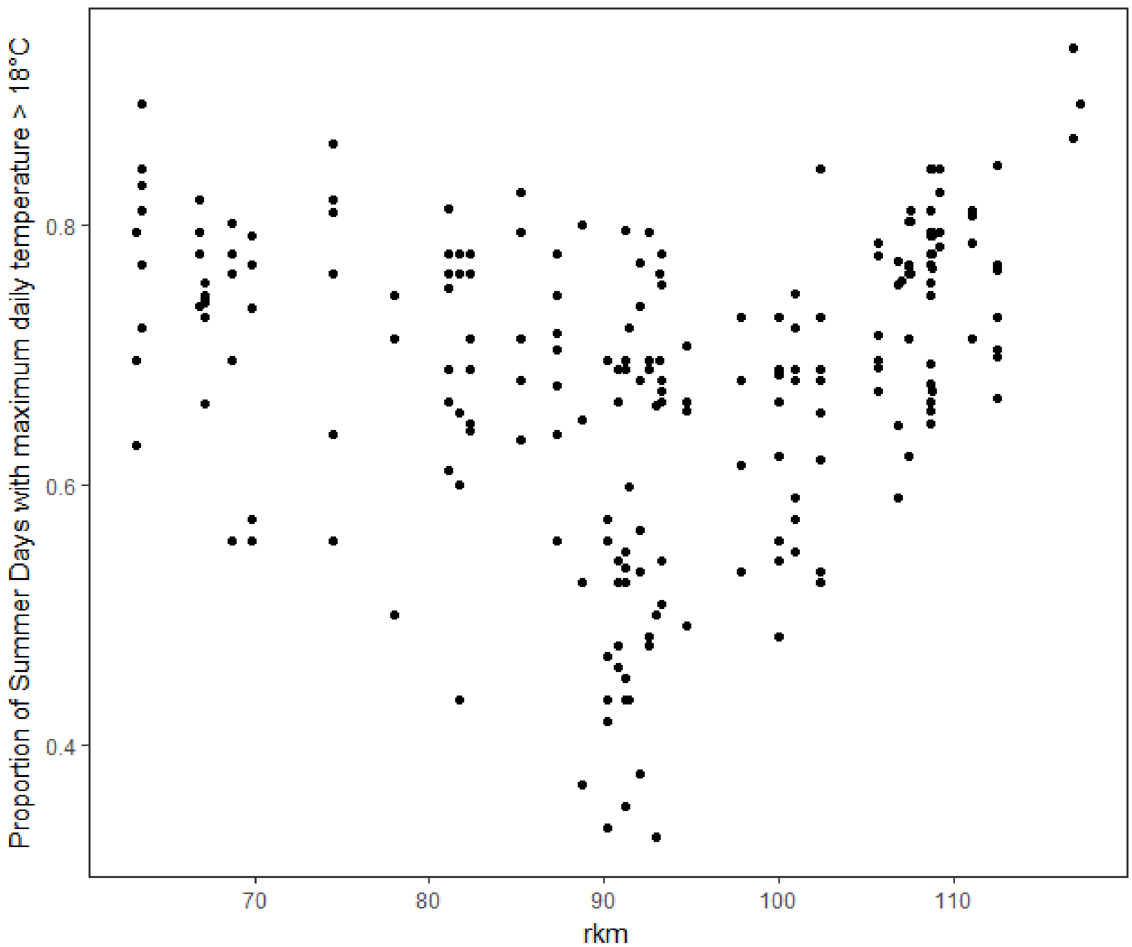

Figure 5. Proportion of summer days with maximum daily temperatures greater than 18°C by rkm (left to right, downstream to upstream) on the MFJD in the MFIMW area for all loggers and years. Rkm 0 represents the mouth of the MFJD.

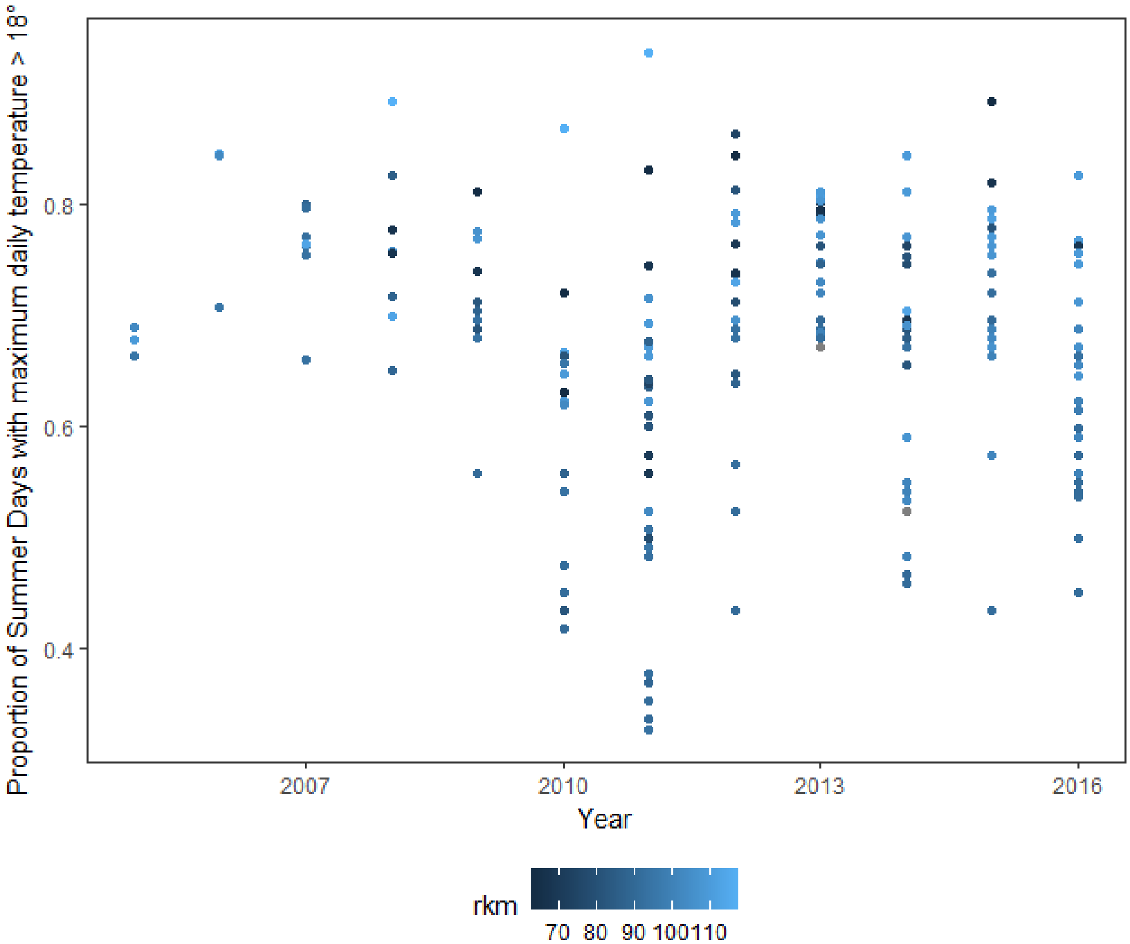

Figure 6. Proportion of summer days with maximum daily temperatures greater than 18°C by on the MFJD in the MFIMW area for all loggers and years. Rkm 0 represents the mouth of the MFJD.

Bridge Creek Maximum 7DADM

Loggers were placed in Bridge Creek beginning in 2009. Two years of data were measured for loggers located in Bates Pond (rkm 0.84). Higher 7DADMs occur in most years below Bates Pond, with much cooler temperatures occurring the further upstream the logger is from Bates Pond. However, compared to water temperatures in the mainstem MFJD, Bridge Creek was on average cooler, with the furthest upstream loggers nearing the 18°C threshold (Figure 7; Appendix Figure 2). The lowest maximum 7DADMs occurred in 2011, and the highest occurred in years 2013-2015 (Figure 8).

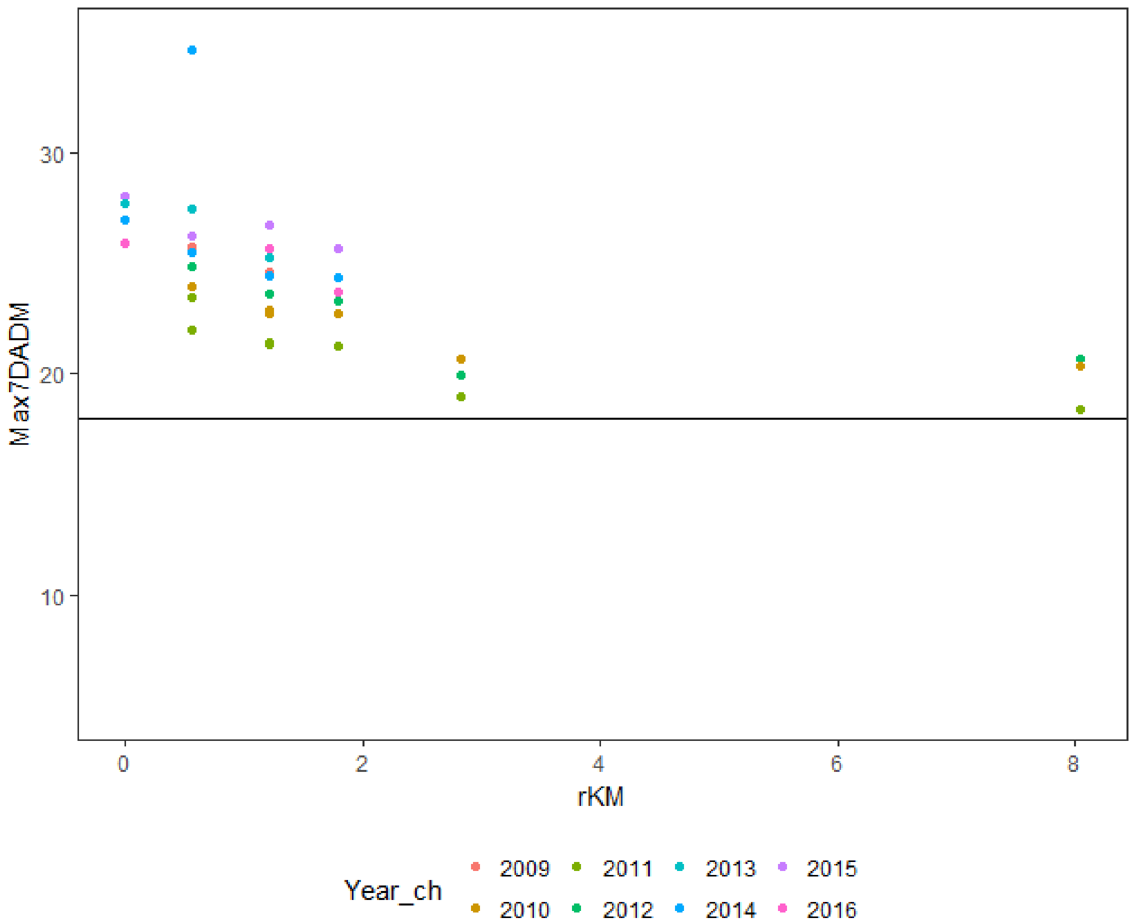

Figure 7. Maximum seven day average daily maximum temperature (°C) (7DADM) by rkm on Bridge Creek for all loggers and years. Rkm 0 represents the mouth of Bridge Creek; Bates Pond is located at rkm 0.89.

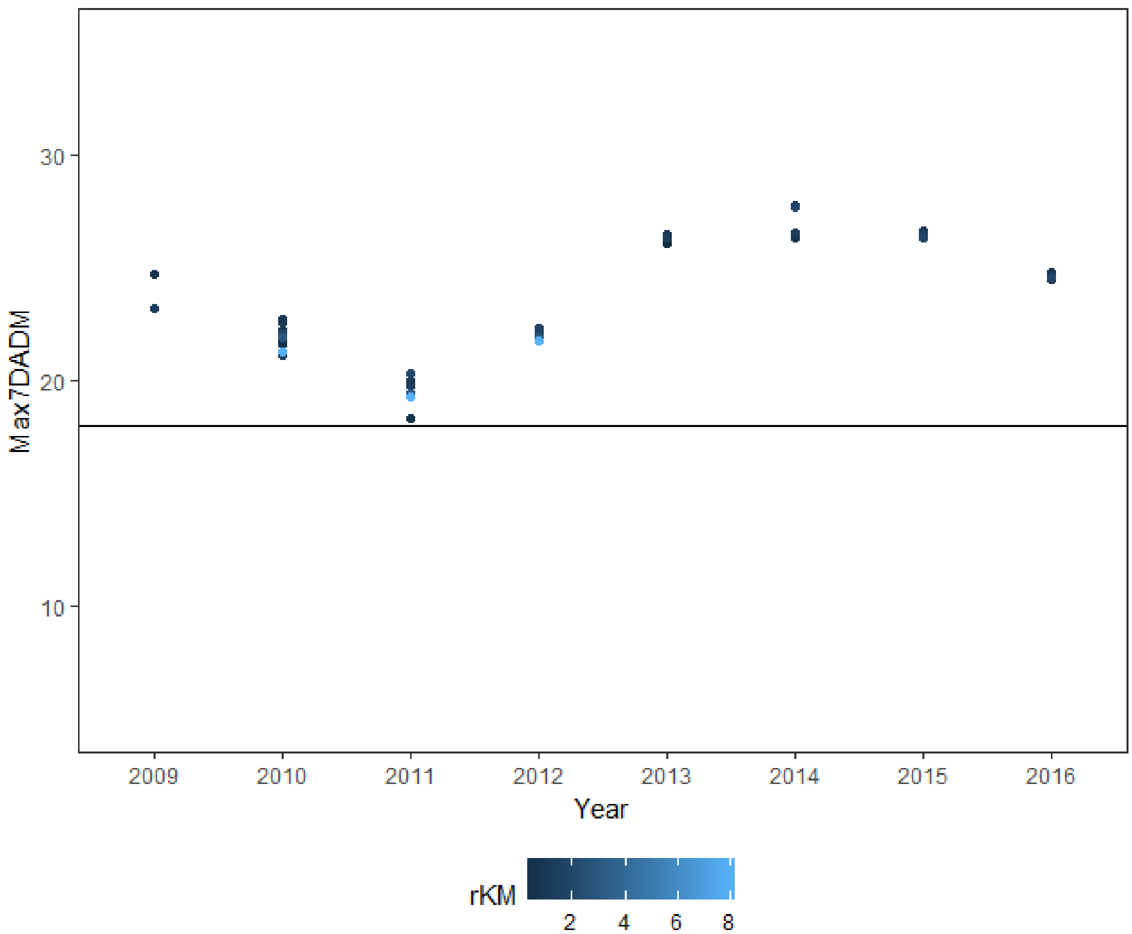

Figure 8. Maximum seven day average daily maximum temperature (°C) (7DADM) by year on Bridge Creek for all loggers and years. Rkm 0 represents the mouth of the MFJD; Bates Pond is located at rkm 0.89.

Bridge Creek proportion of days greater than 18°C

The proportion of summer days with a maximum temperature greater than 18°C shows a similar pattern to maximum 7DADM analysis. Sites with more days greater than 18°C occurred downstream of Bates Pond. Sites with fewer days greater than 18°C temperatures occurred upstream of Bates Pond (Figure 9). The highest proportion of days greater than 18°C occurred in 2014, while lowest proportion of days greater than 18°C occurred in 2011 (Figure 10).

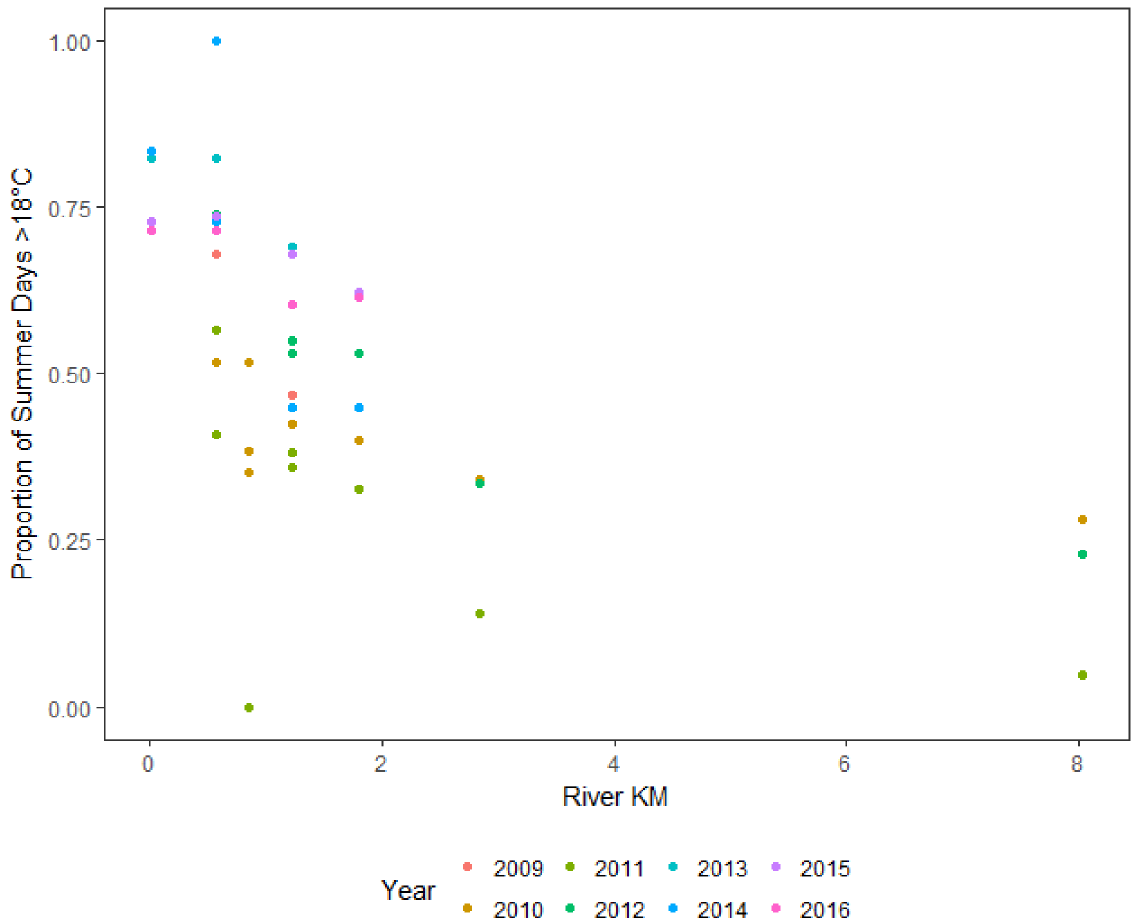

Figure 9. Proportion of summer days with maximum daily temperatures greater than 18°C by rkm (left to right, downstream to upstream) on Bridge Creek. Rkm 0 represents the mouth of the Bridge Creek; Bates Pond is located at rkm 0.89.

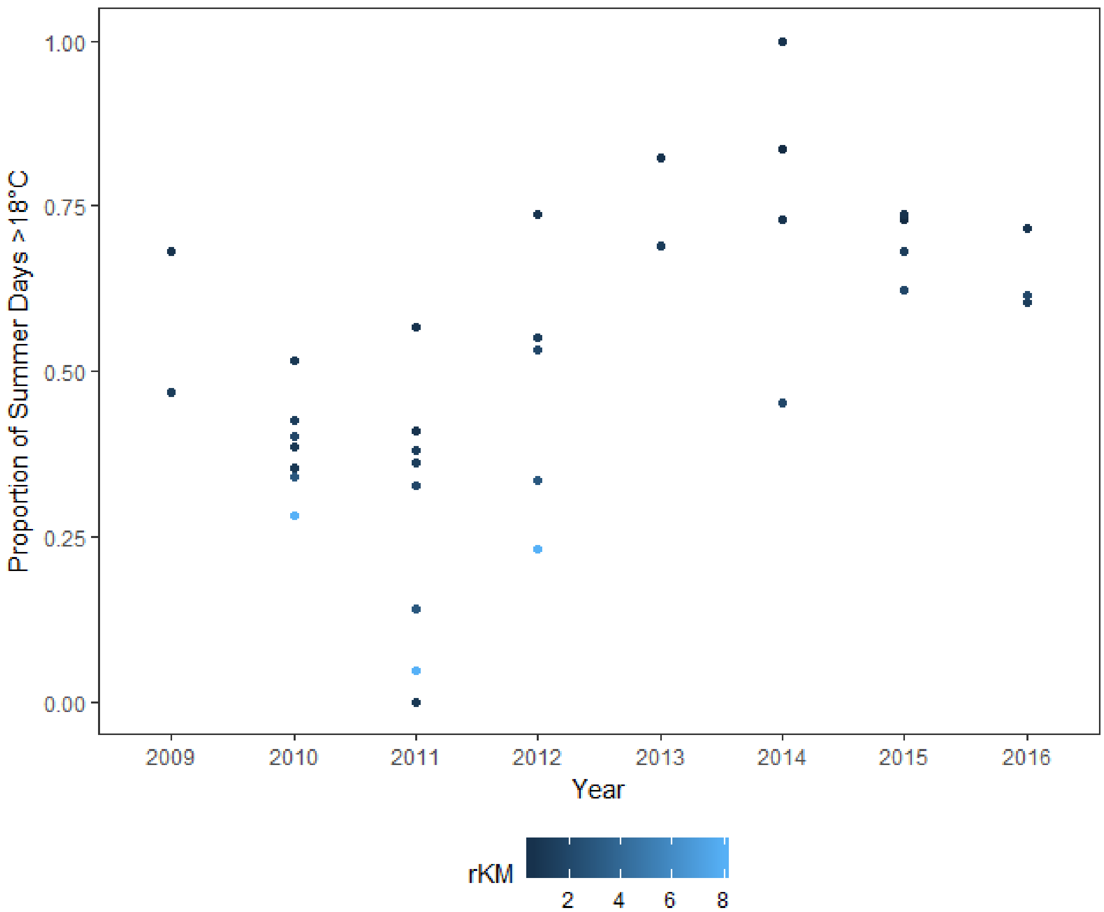

Figure 10. Proportion of summer days with maximum daily temperatures greater than 18°C by year on Bridge Creek. Rkm 0 represents the mouth of the Bridge Creek; Bates Pond is located at rkm 0.89.

MFJD Above/Below Tributary analysis

Data was available to calculate the differences in daily maximum temperatures between the upstream and downstream loggers on the MFJD for eight tributaries. Tributaries with a cooling influence will show a negative difference, while a positive difference indicates a warming influence on the mainstem MFJD. Big Creek, Granite Boulder Creek and Vinegar Creek all showed a cooling influence, while Beaver and Bridge Creeks showed a neutral influence, and Deerhorn Creek showed a warming influence (Figure A-3).

Differences between SFJD and MFJD

Maximum temperature differences between the SFJD rotary screw trap and loggers at the bottom of the MFIMW near the confluence with Big Creek showed no difference between the two watersheds. There was no significant trend in MFIMW temperatures through the sample years after they were corrected using the South Fork control (Figure 11).

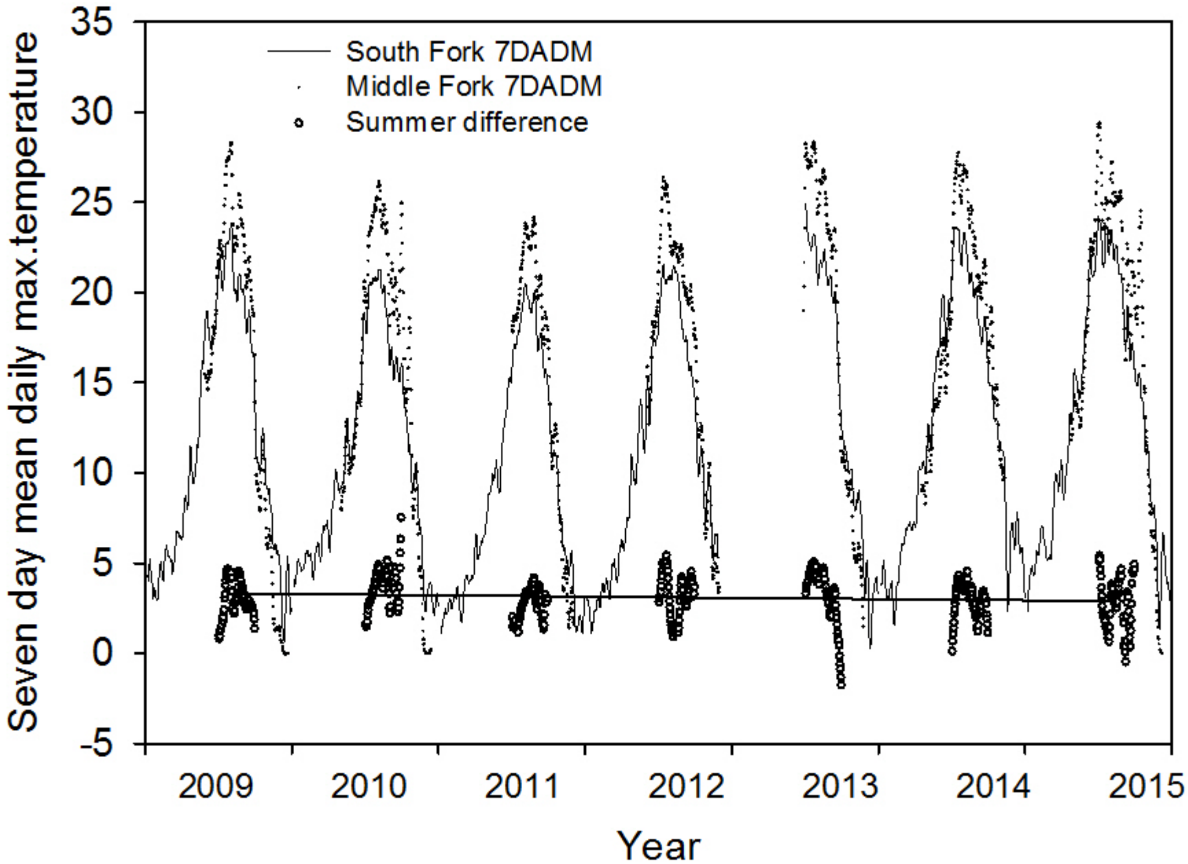

Figure 11. Difference in 7DADM between the loggers on the MJFD near Big Creek and the logger on the rotary screw trap on the SFJD. A linear regression line is also fitted to the annual difference in summer temperatures.

Discussion

Most notably, water temperatures within the MFIMW remain warm. All measured maximum 7DADMs for all years were above the EPA's 18°C 7DADM threshold and continued to remain above the 20°C water temperatures observed by FLIR flights in 2003 (UMFWG 2011).

Our analyses did not indicate any decline in MFJD temperatures when measured against the SFJD reference watershed. Major restoration projects in the MFJD began in 2009, and the time period of temperature monitoring in the MFJD has not been long enough to detect a difference from restoration alone. Restoration activities likely to affect water temperatures include channel reconstruction, as occurred in the Oxbow properties, and riparian plantings. The HeatSource model and work conducted by OSU (see the Groundwater, HeatSource, and DTS MFIMW report) show that lengthy time-periods (upwards of 25 years for riparian plantings) are required to influence water temperatures. Specific restoration projects that likely affected stream temperatures (i.e. restoring water to the South Channel) were not specifically addressed in this report.

The MFJD showed an unusual longitudinal pattern of temperatures along the stream length, with low temperatures occurring midway through the MFIMW and higher temperatures at both upstream and downstream directions. The cooler temperatures upstream of rkm 91 coincide with a known cold-water spring and is also near the mouth of Granite Boulder Creek, which has a cooling influence on the MFJD (see the OSU MFIMW report).

Observations of temperature trends on Bridge Creek show an increase in maximum 7DADM temperatures downstream of Bates Pond, compared to upstream. The tributary analysis on the mainstem MFJD indicates that Bridge Creek currently has an overall neutral effect on the mainstem temperatures. Proposed changes by Oregon State Parks to Bates Pond to increase shading and decrease water temperatures would change Bridge Creek from having a neutral influence to a cooling influence, especially if the pond was altered or removed to significantly reduce solar heating. If the proposed changes to Bates Pond also included complete fish passage, then Bridge Creek could also serve as a significant cold-water refuge for rearing juvenile salmonids and provide a moderating influence on mainstem water temperatures.

Observation of year to year trends show differences in timing of maximum 7DADMs. These differences are most likely due to yearly variation in air temperature and stream flow. The years 2013 and 2015 had low stream flows and high air temperatures, while 2011 generally had higher stream flows and lower air temperatures. Observations in 2007 when approximately 50% of the Spring Chinook died from a pre-spawn event (Ruzycki et al. 2008), showed 7DADMs occurring earlier in the year, in July rather than August, and those early high temperatures likely contributed to the pre-spawn event.

In conclusion, despite recent restoration activities in the MFJD, summer water temperatures continue to be well above the recommended 18°C for coldwater salmonids and temperatures remain a bottleneck to fish recovery. Restoration activates are however, recent, within the last five years, which is likely too short a time-period to see a watershed-level change in temperature values. Further nuanced analysis, including updating temperature models, conducting statistical analysis including environmental variables could help identify restoration activities that influence water temperatures. In addition, full analysis of tributary temperature data has not yet been performed, and many of the tributaries appear to be important cold water refuges for salmonids.

Lessons Learned

The amount of temperature data assembled for the MFIMW was substantial, and is housed in a large Access database. While attempts were made to deal with as many gaps and outliers as possible, numerous gaps and outliers continue to exist within the dataset, including unusually high water temperatures, water temperature data collected in minutes rather than hours, loggers that were labelled incorrectly in the database, or data that was uploaded more than once. Inaccurate data were corrected or excluded as it was discovered, but additional quality control still needs to be completed. There appeared to be many more outlier points than were expected. Whether this is a common occurrence in temperature monitoring or was due to poor logger placement, logger failure, or some other criteria, care should be taken to avoid this many outlier situations in future data collection efforts. However, for the temporal and spatial scales over which this analysis was completed, we believe the patterns and relationships are acceptable.

Unfortunately exact decision points around site selection prior to about 2014 were lost due to staff turnover and time, thus water temperature logger site selection appear to have occurred in a somewhat reactive manner; dependent on specific investigator needs and available funding. Clear monitoring goals, mid-project analysis, site selection documentation, and communication with collaborators would have helped co-managers determine which loggers locations were important to keep.

Clear consistent monitoring goals and written protocols for field, quality control, and analysis methods are vital for a data collection effort as large as this one. Conducting quality control measures years after the data was collected was problematic; the process was overly time consuming and complicated for the person performing quality control. Consistent and timely quality control procedures would have allowed more time for extensive and complex analysis and without these controls it is very possible that nuances in the temperature data may have been lost. It would have been useful to know through thorough logger documentation, if extremely high temperature points were due to the logger out of water or due to a dry channel. This is a fine point, but an important one when considering the effects of water usage and allotments and the habitat needs of salmonids.

Suggestions for Future Monitoring

- Continue monitoring water temperature data in the MFJD.
    - With clear goals and a monitoring plan in place!
- Form a committee of individuals invested in temperature monitoring, including monitoring projects, data steward, and other to be identified.
- Identify clear goals for water temperature monitoring. Goals need not be exactly the same between collecting agencies, but having goals in place will assist with the other tasks.
    - For instance, one goal may be to be able to contribute data to NorWest.

- Develop a clear site selection process, including documenting decision making around selections, and a monitoring plan for each logger deployed.
- Develop a QAPP for water temperature monitoring.
    - QAPP's are required for all ODEQ volunteer water quality monitoring programs.
- Collaborate with other data collectors in MFIMW to identify where and over what time period (i.e. year-long including winter) to leave loggers in place.
    - Consider a statistical site selection process, like GRTS.
- Develop and document agreed upon field protocols:
    - Consider following ODEQ Volunteer Monitoring Guidelines (http://www.oregon.gov/deq/FilterDocs/volunteerQAplan.pdf)
- Recommended physical placement and anchoring dependent on site.
- Logger calibration
- In-season maintenance
- Downloading data
- Develop and document quality control protocols and analytical steps:
    - Processes for failing, accepting, records
    - Dealing with outliers, extreme high temperatures, and nuances of decision making around quality checking temperature data
    - Utilizing written protocols, experience, and tools of other projects that monitor water temperature, including DEQ and CHaMP (https://www.champmonitoring.org/)
    - Adopt guidelines for having consistent data stewardship for temperature data.
- Identify appropriate analysis to answer questions and meet goals.
- Adopt an adaptive monitoring approach, identifying mid-project analysis, and defining clear rules for dealing with inevitable lost loggers, environmental changes, and staff and funding changes.
- Identify an appropriate platform for storing temperature data and securing funding to purchase, develop, and maintain the platform.
- Coordinate with other water temperature data collection efforts in the MFJD, including CHaMP and ODFW, to promote collaboration and avoid duplication.
- ODFW working on a statewide temperature monitoring framework to make temp data collection more consistent, less duplicative, and functional for large-scale regional collection efforts (NORWEST).

Future Analysis

- Identify specific restoration projects that likely affected water temperature and look for changes pre/post restoration.
- Identify statistical analysis that could include air temperatures and flow data to better understand watershed level water temperature changes.
- What kind of effects do the tributary temperatures have on the mainstem MFJD?
- Which ones? How far downstream do the effects seem to last? At what times of the year is this most apparent? Etc.
- Complete 7DADM analysis on tributary loggers
    - o Adapt R script for calculating batch 7DADM (script written and available from Joe Lemanski's (CWTRSO)).
- Identify loggers with data before and after IMW inception (2008) and calculate differences – similar to SFJD vs MFJD analysis.
    - o Perhaps use logger on ODFW's rotary screw trap?
- Analysis of air temperature, flow data, and water temperatures…
- Update Heat Source and/or ISEMP models.
- Daily fluctuations in temperature – i.e. how do night-time temps compare to day-time temps. Do nighttime summer temperatures provide any refuge?
- Incorporate water temperature logger data collected at other sites and/or by other agencies
    - o Stage-height logger
    - o CHaMP loggers
    - o ODFW rotary screw trap
- Complete 7DADMs on air temperature data
- Calculate time frames and lengths of cold and hot time periods

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Appendices

Appendix A.

Table A-5. Available water temperature loggers and years of data available for water temperature data collected for the MFIMW.

Stream Name	Site Name	River km	Agency	Active	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Beaver Cr	BeaverCr_aOldBridge	0.053	CTWSRO	Yes			1		1	1	1	1	1		1	1
Big Boulder Cr	BigBoulderCr_lwr	0.056	TNC	Yes				1	1	1	1	1	1	1	1	
Big Boulder Cr	BigBoulderCr_mid	0.644	TNC	No							1					
Big Boulder Cr	BigBoulderCr_upr1	1.616	TNC	Yes				1	1	1	1	1	1	1	1	
Big Boulder Cr	BigBoulderCr_upr2	1.976	TNC	No			1				1		1			
Big Creek	BigCr_aBlowout	6.926	TNC	No						1						
Big Creek	BigCr_atBoundary	3.766	TNC	No						1						
Big Creek	BigCr_mouth	0.015	TNC	Yes				1	1	1	1	1	1	1	1	
Bridge Creek	BatesPond_linebottom	0.848	NFJDWC	No						1	1					
Bridge Creek	BatesPond_linemiddle	0.848	NFJDWC	No						1	1					
Bridge Creek	BatesPond_linetop	0.848	NFJDWC	No						1	1					
Bridge Creek	BridgeCr_1	2.835	NFJDWC	No						1	1	1				
Bridge Creek	BridgeCr_2	8.042	NFJDWC	No						1	1	1				
Bridge Creek	BridgeCr_amouth	0.001	NFJDWC	Yes									1	1	1	1
Bridge Creek	BridgeCr_apond1	1.217	NFJDWC	Yes					1	1	1	1	1	1	1	1
Bridge Creek	BridgeCr_apond2	1.217	NFJDWC	No						1	1	1				
Bridge Creek	BridgeCr_bdam1	0.566	NFJDWC	Yes					1	1	1			1	1	1
Bridge Creek	BridgeCr_bdam2	0.566	NFJDWC	No						1	1	1	1	1	1	
Bridge Creek	BridgeCr_pnt5mileapond	1.801	NFJDWC	Yes						1	1	1		1	1	1
Butte Creek	ButteCr_bculvert	0.118	CTWSRO	Yes			1		1	1	1	1	1	1	1	1
Camp Creek	CampCr_a733Rd	21.60	NFJDWC	No						1						
Camp Creek	CampCr_FSBoundary	0.385	NFJDWC	No						1						
Camp Creek	CampCr_lwr1	2.126	NFJDWC	Yes				1		1	1	1	1	1	1	1
Camp Creek	CampCr_lwr2	5.075	NFJDWC	Yes				1	1	1	1	1	1	1		1
Camp Creek	CampCr_mid1	0.047	NFJDWC	Yes				1	1	1	1	1	1	1	1	
Camp Creek	CampCr_mid3	14.12	NFJDWC	No				1	1	1						
Camp Creek	CampCr_upr1	17.72	NFJDWC	Yes				1	1	1		1	1	1	1	1
Camp Creek	CampCr_upr3	22.26	NFJDWC	Yes				1	1	1		1	1	1	1	1
Clear Creek	ClearCr_1	4.245	NFJDWC	No						1	1	1				

Clear Creek	ClearCr_2	11.60	NFJDWC	No				1	1	1				
Clear Creek	ClearCr_aCORd20	0.662	NFJDWC	Yes			1	1			1	1	1	1
Clear Creek	DryForkClearCr	2.800	NFJDWC	No				1	1					
Coyote Creek	CoyoteCr_bRoad	0.271	TNC	No		1	1	1	1	1	2	1		
Davis Creek	DavisCr_amouth	0.001	CTWSRO	Yes	1	1	1		1	1	1	1	1	1
Davis Creek	DavisCr_lwr	0.068	NFJDWC	Yes				1	1	1	1	1	1	1
Davis Creek	DavisCr_mid	2.928	NFJDWC	Yes				1	1	1	1	1	1	1
Davis Creek	DavisCr_upr	4.617	NFJDWC	Yes				1	1	1	1	1	1	1
Dead Cow Gulch	DeadCowGulch_amouth	0.037	CTWSRO	Yes	1	1	1		1	1	1	1	1	1
Deadwood Creek	DeadwoodCr_lower	0.122	NFJDWC	Yes									1	1
Deadwood Creek	DeadwoodCr_Upper	2.405	NFJDWC	Yes										1
Deerhorn Creek	DeerhornCr_amouth	0.004	NFJDWC	Yes							1	1	1	1
East Fork Big Cr	EastFork_BigCr_Mouth	0.077	NFJDWC	Yes									1	1
Granite Boulder Cr	GraniteBoulderCr_lwr1	0.369	NFJDWC	Yes		1	1	1		1	1	1	1	1
Granite Boulder Cr	GraniteBoulderCr_lwr2	1.333	NFJDWC	Yes		1				1	1		1	1
Granite Boulder Cr	GraniteBoulderCr_mouth	0.290	CTWSRO	Yes							1	1	1	1
Granite Boulder Cr	GraniteBoulderCr_up3	5.555	NFJDWC	Yes		1	1	1	1	1		1	1	1
Granite Boulder Cr	GraniteBoulderCr_upr1	3.169	NFJDWC	Yes		1	1	1	1		1	1	1	1
Hawkins Creek	HawkinsCr_bRoad	0.395	NFJDWC	Yes		1	1	1	1	1	1	1	1	
Little Boulder Cr	LittleBoulderCr_amouth	99.99	NFJDWC	Yes							1		1	1
Lunch Creek	LunchCr_1	0.433	NFJDWC	No				1	1	1				
Lunch Creek	LunchCr_2	0.883	NFJDWC	No				1						
Lunch Creek	LunchCr_3	3.082	NFJDWC	No					1	1				
MJFD	MFJD_2.3rm	107.0	CTWSRO	No	1	1								
MJFD	MFJD_4.8rm	88.73	CTWSRO	No	1	1		1	1	1				
MJFD	MFJD_a12MileCr_Plemmo	27.43	NFJDWC	Yes				1	1	1	1	1	1	1
MJFD	MFJD_a395BLM	41.99	NFJDWC	Yes				1	1	1	1	1	1	1
MJFD	MFJD_aAlcove1	92.69	CTWSRO	Yes										1
MJFD	MFJD_aAlcove2	92.14	CTWSRO	Yes										1
MJFD	MFJD_aAlcove4	92.04	CTWSRO	Yes										1
MJFD	MFJD_aAlcove5	91.78	CTWSRO	Yes										1
MJFD	MFJD_aBallanceCr	81.69	TNC	Yes			1	1	1	1	1	1	1	
MJFD	MFJD_aBeaver	91.25	CTWSRO	Yes								1	1	1
MJFD	MFJD_aBigCr	63.50	TNC	Yes			1	1	1	1	1	1	1	
MJFD	MFJD_aBridgeCr	108.8 3	NFJDWC	Yes			1	1	1	1	1	1	1	1
MJFD	MFJD_aCBIV	85.24	TNC	No		1	1	1	1		1	1	1	

MJFD	MFJD_aClearCr	111.0	NFJDWC	Yes								1	1	1	1
MJFD	MFJD_aDavisCr	0.031	NFJDWC	Yes								1	1	1	
MJFD	MFJD_aDeepC	74.48	TNC	Yes					1	1	1	1	1	1	
MJFD	MFJD_aDeerhornCr	100.9	NFJDWC	Yes								1	1	1	1
MJFD	MFJD_aGraniteBoulderCr	93.21	CTWSRO	No		1	1	1	1						
MJFD	MFJD_aLittleBoulderCr	99.99	NFJDWC	Yes								1	1	1	1
MJFD	MFJD_aLowestAlcoveOCA	0.013	CTWSRO	Yes									1		1
MJFD	MFJD_aMosquitoCr	68.68	NFJDWC	Yes					1	1	1	1	1	1	1
MJFD	MFJD_aOldBridge	91.26	CTWSRO	No		1		1	1	1	1	1			
MJFD	MFJD_aRelocationFS	69.82	NFJDWC	Yes					1	1	1	1		1	
MJFD	MFJD_aRelocationRPB	67.17	TNC	Yes			1	1	1	1	1	1	1	1	
MJFD	MFJD_aS/NChannelConflu	1.503	CTWSRO	No				1	1	1	1	1			
MJFD	MFJD_aUpperAlcoveOCA	91.44	CTWSRO	Yes									1	1	
MJFD	MFJD_aVinegarCr	107.4	NFJDWC	Yes				1	1	1		1	1	1	1
MJFD	MFJD_b12MilleCr_Tillay	24.49	NFJDWC	Yes					1	1	1	1	1	1	
MJFD	MFJD_bBeaverCr	90.80	CTWSRO	Yes				1	1		1	1	1	1	1
MJFD	MFJD_bBigCr	63.21	TNC	Yes					1	1	1	1	1	1	
MJFD	MFJD_bBridgeCr	108.7	NFJDWC	Yes				1	1	1	1	1	1	1	1
MJFD	MFJD_bButteCr	2.639	CTWSRO	Yes		1		1	1	1	1	1	1	1	1
MJFD	MFJD_bClearCr	109.2	NFJDWC	Yes				1		1	1	1	1	1	1
MJFD	MFJD_bCoyoteCr	82.37	TNC	Yes				1		1	1	1	1	1	
MJFD	MFJD_bCrawfordCr	116.8	TNC	No					1	1					
MJFD	MFJD_bDeerhornCr	100.9	NFJDWC	Yes								1	1	1	1
MJFD	MFJD_bGraniteBoulderCr	92.94	CTWSRO	No		1				1					1
MJFD	MFJD_bLittleBoulderCr	99.99	NFJDWC	Yes								1	1	1	1
MJFD	MFJD_bLittleButteCr	97.87	TNC	Yes								1	1	1	1
MJFD	MFJD_bNewGraniteBoulde	92.94	CTWSRO	Yes								1	1	1	
MJFD	MFJD_bRaggedCr	90.21	CTWSRO	Yes				1	1	1	1	1	1	1	1
MJFD	MFJD_bRelocation_RPB	66.86	NFJDWC	Yes			1				1	2		1	
MJFD	MFJD_bRubyCr	92.04	CTWSRO	Yes		1		1	1	1	1	1	1	1	1
MJFD	MFJD_bVincentCr	105.7	CTWSRO	Yes				1		1	1	1	1	1	1
MJFD	MFJD_bVinegarCr	106.7	NFJDWC	Yes								1	1	1	1
MJFD	MFJD_CampCr_Gauge	0.026	TNC	No			1		1	1	1	1	1		
MJFD	MFJD_hwy7camp	112.4	CTWSRO	Yes	1	1	1		1		1	1	1	1	
MJFD	MFJD_inAlcove1	92.67	CTWSRO	Yes											1
MJFD	MFJD_inAlcove2	92.53	CTWSRO	Yes											1

MJFD	MFJD_inAlcove4	91.79	CTWSRO	Yes												1
MJFD	MFJD_inAlcove5	91.58	CTWSRO	Yes												1
MJFD	MFJD_inLowestAlcoveOCA	91.19	CTWSRO	Yes										1	1	1
MJFD	MFJD_inUpperAlcoveOCA	91.44	CTWSRO	Yes										1	1	1
MJFD	MFJD_lwrForrestCAbound	102.3	CTWSRO	Yes	1	1				1	1		1	1	1	1
MJFD	MFJD_PhippsMeadow	117.2	CTWSRO	No				1								
MJFD	MFJD_TNC_Eboundary	87.27	TNC	Yes				1	1	1	1	1	1	1	1	
MJFD	MFJD_TNC_Wboundary	81.06	TNC	Yes				1	1	1	1	1	1	1	1	
MJFD	MFJD_uprForrestCAbound	108.6	CTWSRO	Yes	1	1				1	1		1	1	1	1
MJFD	MFJD_uprOxbowCAbound	94.75	CTWSRO	No	1	1				1	1		1			
Mosquito Creek	MosquitoCr_lwr	0.230	NFJDWC	No						1						
Pizer Creek	PizerCr	0.148	NFJDWC	Yes											1	1
Ragged Creek	RaggedCr_lwr	0.082	CTWSRO	Yes					1	1		1	1	1	1	1
Ruby Creek	Ruby_aMouth		CTWSRO	Yes												1
Ruby Creek	RubyCr_aSlough	0.032	CTWSRO	No										1		
Ruby Creek	RubyCr_road	0.227	CTWSRO	No									1	1	1	
Squaw Creek	SquawCr	4.535	NFJDWC	No							1					
Tincup Creek	TinCupCr_lwr	0.303	NFJDWC	Yes					1	1	1	1		1	1	1
Vincent Creek	VincentCr_amouth	0.030	NFJDWC	Yes									1	1	1	1
Vinegar Creek	VinegarCr_amouth	0.168	CTWSRO	Yes					1		1	1	1	1	1	1
Windlass Creek	WindlassCr_lwr	0.080	NFJDWC	Yes					1	1	1	1			1	1

Table A-6. Maximum 7DADM (°C) water temperatures and proportion of summer days with temperatures greater than 18°C for water temperature locations in the mainstem MFJD.

Site Name	River km	Year	Maximum 7DADM	Proportion days >18°C
MFJD_uprOxbowCAboundry	94.75347	2005	25.39	0.6639
MFJD_lwrForrestCAboundry	102.39067	2005	24.10	0.6891
MFJD_uprForrestCAboundry	108.65567	2005	25.06	0.6780
MFJD_hwy7camp	78.04340	2006	25.08	0.8462
MFJD_uprOxbowCAboundry	94.75347	2006	26.13	0.6944
MFJD_lwrForrestCAboundry	102.39067	2006	25.54	0.8444
MFJD_hwy7camp	78.04340	2007	24.60	0.7521
MFJD_4.8rm	88.73492	2007	25.83	0.7863
MFJD_aOldBridge	91.26369	2007	24.89	0.7692
MFJD_bRubyCr	92.03710	2007	24.81	0.7521
MFJD_bGraniteBoulderCr	92.94740	2007	24.18	0.6441
MFJD_aGraniteBoulderCr	93.20977	2007	27.05	0.7500
wtr_temp_MFJD_bButteCr	93.85157	2007	25.66	0.7295
MFJD_aRelocationRPB	67.17839	2008	24.39	0.5702
MFJD_hwy7camp	78.04340	2008	22.76	0.6050
MFJD_aCBIV	85.24026	2008	25.27	0.6179
MFJD_TNC_Eboundary	87.26607	2008	24.11	0.5447
MFJD_4.8rm	88.73492	2008	22.83	0.5982
MFJD_bRelocation_RPB	90.21753	2008	24.70	0.5868
MFJD_aGraniteBoulderCr	93.20977	2008	18.13	0.0252
MFJD_2.3rm	107.02444	2008	23.67	0.6555
MFJD_PhippsMeadow	117.25466	2008	21.85	0.8288
MFJD_aBigCr	63.50145	2009	28.31	0.6471
MFJD_aRelocationRPB	67.17839	2009	26.17	0.5755
MFJD_TNC_Wboundary	81.06850	2009	25.72	0.5490
MFJD_aBallanceCr	81.69881	2009	32.12	0.6122
MFJD_bCoyoteCr	82.37437	2009	26.51	0.5686
MFJD_aCBIV	85.24026	2009	25.57	0.5686
MFJD_TNC_Eboundary	87.26607	2009	25.10	0.5621
MFJD_bRaggedCr	90.21863	2009	23.27	0.5484
MFJD_aS/NChannelConfluence	92.56400	2009	24.66	0.5903
MFJD_aGraniteBoulderCr	93.20977	2009	25.29	0.6855
MFJD_bButteCr	93.85157	2009	24.47	0.6694
MFJD_bVincentCr	105.64612	2009	24.49	0.7604
MFJD_aVinegarCr	107.40655	2009	24.24	0.5926
MFJD_bClearCr	109.20864	2009	23.07	0.4946
MFJD_bBigCr	63.21483	2010	26.05	0.5163
MFJD_aBigCr	63.50145	2010	26.17	0.6209
MFJD_aRelocationRPB	67.17839	2010	27.15	0.5000

MFJD_aRelocationFS	69.82872	2010	23.97	0.4444
MFJD_aDeepCr	74.48182	2010	24.92	0.4444
MFJD_hwy7camp	78.04340	2010	23.57	0.6000
MFJD_TNC_Wboundary	81.06850	2010	23.58	0.5425
MFJD_aBallanceCr	81.69881	2010	21.27	0.3464
MFJD_TNC_Eboundary	87.26607	2010	23.73	0.4444
MFJD_4.8rm	88.73492	2010	22.18	0.4516
MFJD_bRaggedCr	90.21863	2010	21.67	0.3333
MFJD_bBeaverCr	90.79900	2010	23.84	0.3856
MFJD_aOldBridge	91.26369	2010	22.01	0.3595
MFJD_aS/NChannelConfluence	92.56400	2010	21.69	0.3791
MFJD_bButteCr	93.85157	2010	23.03	0.4314
MFJD_uprOxbowCAboundry	94.75347	2010	23.16	0.5909
MFJD_lwrForrestCAboundry	102.39067	2010	22.76	0.5534
MFJD_CampCr_Gauge	106.77847	2010	24.93	0.6703
MFJD_aVinegarCr	107.40655	2010	23.92	0.4967
MFJD_uprForrestCAboundry	108.65567	2010	23.16	0.5909
MFJD_bBridgeCr	108.82884	2010	23.79	0.5294
MFJD_bCrawfordCr	116.82225	2010	21.80	0.7130
MFJD_aBigCr	63.50145	2011	24.15	0.6270
MFJD_aRelocationRPB	67.17839	2011	22.69	0.5600
MFJD_aMosquitoCr	68.68502	2011	22.87	0.4658
MFJD_aRelocationFS	69.82872	2011	23.02	0.4795
MFJD_aDeepCr	74.48182	2011	22.24	0.5310
MFJD_TNC_Wboundary	81.06850	2011	22.91	0.4603
MFJD_aBallanceCr	81.69881	2011	22.58	0.4524
MFJD_bCoyoteCr	82.37437	2011	23.05	0.4841
MFJD_aCBIV	85.24026	2011	22.82	0.4803
MFJD_TNC_Eboundary	87.26607	2011	27.64	0.5118
MFJD_4.8rm	88.73492	2011	20.92	0.3191
MFJD_bRaggedCr	90.21863	2011	20.46	0.2789
MFJD_aOldBridge	91.26369	2011	20.93	0.2925
MFJD_bRubyCr	92.03710	2011	21.23	0.3151
MFJD_aS/NChannelConfluence	92.56400	2011	22.26	0.4014
MFJD_bGraniteBoulderCr	92.94740	2011	19.96	0.2721
MFJD_bButteCr	93.85157	2011	22.03	0.4218
MFJD_uprOxbowCAboundry	94.75347	2011	22.32	0.4255
MFJD_lwrForrestCAboundry	102.39067	2011	22.60	0.4338
MFJD_bVincentCr	105.64612	2011	22.65	0.5397
MFJD_CampCr_Gauge	106.77847	2011	22.97	0.4326
MFJD_aVinegarCr	107.40655	2011	23.04	0.5205
MFJD_uprForrestCAboundry	108.65567	2011	23.29	0.5940
MFJD_aBridgeCr	108.82884	2011	23.20	0.5655

MFJD_bBridgeCr	108.82884	2011	23.18	0.5548
MFJD_bClearCr	109.20864	2011	22.80	0.6000
MFJD_bCrawfordCr	116.82225	2011	23.64	0.7165
MFJD_aBigCr	63.50145	2012	26.42	0.6797
MFJD_aRelocationRPB	67.17839	2012	25.01	0.5817
MFJD_aMosquitoCr	68.68502	2012	24.65	0.5957
MFJD_aRelocationFS	69.82872	2012	24.86	0.5745
MFJD_aDeepCr	74.48182	2012	26.56	0.6738
MFJD_hwy7camp	78.04340	2012	23.38	0.6692
MFJD_TNC_Wboundary	81.06850	2012	28.77	0.6623
MFJD_bCoyoteCr	82.37437	2012	25.29	0.5163
MFJD_TNC_Eboundary	87.26607	2012	24.20	0.5098
MFJD_4.8rm	88.73492	2012	22.45	0.4812
MFJD_bRelocation_RPB	90.21753	2012	25.18	0.5882
MFJD_bRaggedCr	90.21863	2012	21.55	0.3464
MFJD_bBeaverCr	90.79900	2012	22.38	0.4183
MFJD_aOldBridge	91.26369	2012	22.64	0.4183
wtr_temp_MFJD_bRubyCr	92.03710	2012	22.95	0.4510
MFJD_aS/NChannelConfluence	92.56400	2012	24.55	0.5490
MFJD_bButteCr	93.85157	2012	24.61	0.5425
MFJD_bVincentCr	105.64612	2012	24.31	0.5556
MFJD_CampCr_Gauge	106.77847	2012	25.88	0.5686
MFJD_aBridgeCr	108.82884	2012	24.99	0.6197
MFJD_bBridgeCr	108.82884	2012	24.87	0.6197
MFJD_bClearCr	109.20864	2012	24.69	0.6127
MFJD_bBigCr	63.21483	2013	27.84	0.5556
MFJD_aBigCr	63.50145	2013	28.33	0.6471
MFJD_aMosquitoCr	68.68502	2013	27.03	0.6268
MFJD_aRelocationFS	69.82872	2013	26.94	0.6197
MFJD_aDeepCr	74.48182	2013	29.13	0.6338
MFJD_hwy7camp	78.04340	2013	23.23	0.5913
MFJD_TNC_Wboundary	81.06850	2013	26.82	0.6078
MFJD_aBallanceCr	81.69881	2013	26.49	0.6078
MFJD_bCoyoteCr	82.37437	2013	26.80	0.6078
MFJD_TNC_Eboundary	87.26607	2013	25.65	0.5948
MFJD_bRelocation_RPB	90.21753	2013	26.80	0.6340
MFJD_bRaggedCr	90.21863	2013	23.64	0.5556
MFJD_bBeaverCr	90.79900	2013	23.99	0.5490
MFJD_aOldBridge	91.26369	2013	24.11	0.5490
MFJD_bRubyCr	92.03710	2013	24.21	0.5425
MFJD_aS/NChannelConfluence	92.56400	2013	26.81	0.6340
MFJD_bNewGraniteBoulderCr	92.94740	2013	24.25	0.5359
MFJD_bButteCr	93.85157	2013	25.14	0.5826

MFJD_bLittleButteCr	97.87071	2013	26.10	0.5704
MFJD_aLittleBoulderCr	99.98987	2013	25.38	0.5704
MFJD_bLittleBoulderCr	99.98987	2013	24.07	0.5352
MFJD_aDeerhornCr	100.94401	2013	25.14	0.5845
MFJD_bDeerhornCr	100.94401	2013	25.04	0.5634
MFJD_lwrForrestCAboundry	102.39067	2013	25.06	0.5817
MFJD_bVincentCr	105.64612	2013	25.28	0.6275
MFJD_bVinegarCr	105.64662	2013	25.30	0.6028
MFJD_CampCr_Gauge	106.77847	2013	26.44	0.6053
MFJD_aVinegarCr	107.40655	2013	27.16	0.6294
MFJD_aDavisCr	107.51940	2013	27.09	0.6294
MFJD_uprForrestCAboundry	108.65567	2013	26.79	0.6471
MFJD_aBridgeCr	108.82884	2013	26.96	0.6224
MFJD_bBridgeCr	108.82884	2013	26.83	0.6224
MFJD_bClearCr	109.20864	2013	26.72	0.6224
MFJD_aClearCr	111.02589	2013	27.47	0.6286
MFJD_aBigCr	63.50145	2014	27.81	0.6471
MFJD_aRelocationRPB	67.17839	2014	25.43	0.5948
MFJD_aMosquitoCr	68.68502	2014	25.38	0.5556
MFJD_aDeepCr	74.48182	2014	27.97	0.6078
MFJD_hwy7camp	78.04340	2014	23.51	0.5621
MFJD_aBallanceCr	81.69881	2014	25.71	0.5229
MFJD_bCoyoteCr	82.37437	2014	25.88	0.5490
MFJD_aCBIV	85.24026	2014	25.34	0.5425
MFJD_bRaggedCr	90.21863	2014	22.56	0.3725
MFJD_bBeaverCr	90.79900	2014	22.73	0.3660
MFJD_aBeaver	91.25857	2014	23.13	0.2054
MFJD_bRubyCr	92.03710	2014	23.32	0.4248
MFJD_bNewGraniteBoulderCr	92.94740	2014	23.22	0.4183
MFJD_bButteCr	93.85157	2014	25.77	0.5359
MFJD_bLittleButteCr	97.87071	2014	24.92	0.4248
MFJD_aLittleBoulderCr	99.98987	2014	24.23	0.4314
MFJD_bLittleBoulderCr	99.98987	2014	23.29	0.3856
MFJD_aDeerhornCr	100.94401	2014	24.03	0.4379
MFJD_bDeerhornCr	100.94401	2014	23.98	0.4379
MFJD_lwrForrestCAboundry	102.39067	2014	23.69	0.4248
MFJD_bVincentCr	105.64612	2014	24.93	0.5390
MFJD_bVinegarCr	105.64662	2014	24.51	0.4706
MFJD_CampCr_Gauge	106.77847	2014	25.97	0.5948
MFJD_aVinegarCr	107.40655	2014	25.71	0.6144
MFJD_aDavisCr	107.51940	2014	25.69	0.6471
MFJD_uprForrestCAboundry	108.65567	2014	25.45	0.6471
MFJD_aBridgeCr	108.82884	2014	25.62	0.6732

MFJD_bBridgeCr	108.82884	2014	25.57	0.6732
MFJD_bClearCr	109.20864	2014	25.35	0.6732
MFJD_aClearCr	111.02589	2014	26.67	0.6471
MFJD_bBigCr	63.21483	2015	29.09	0.6340
MFJD_aMosquitoCr	68.68502	2015	27.45	0.6209
MFJD_aRelocationFS	69.82872	2015	27.20	0.6144
MFJD_aDeepCr	74.48182	2015	30.17	0.6536
MFJD_hwy7camp	78.04340	2015	23.84	0.6144
MFJD_TNC_Wboundary	81.06850	2015	27.58	0.6209
MFJD_aBallanceCr	81.69881	2015	27.57	0.6209
MFJD_bCoyoteCr	82.37437	2015	27.88	0.6209
MFJD_aCBIV	85.24026	2015	27.11	0.6340
MFJD_TNC_Eboundary	87.26607	2015	26.55	0.6275
MFJD_bRelocation_RPB	90.21753	2015	28.16	0.6536
MFJD_bRaggedCr	90.21863	2015	23.72	0.5556
MFJD_bBeaverCr	90.79900	2015	24.33	0.5294
MFJD_inLowestAlcoveOCA	91.19592	2015	21.76	0.3464
MFJD_aBeaver	91.25857	2015	24.47	0.5556
MFJD_aUpperAlcoveOCA	91.43892	2015	24.72	0.5752
MFJD_inUpperAlcoveOCA	91.44121	2015	21.76	0.3464
MFJD_bRubyCr	92.03710	2015	24.92	0.5882
MFJD_bNewGraniteBoulderCr	92.94740	2015	25.10	0.5556
MFJD_bButteCr	93.85157	2015	26.96	0.6209
MFJD_bLittleButteCr	97.87071	2015	26.54	0.5425
MFJD_aLittleBoulderCr	99.98987	2015	25.51	0.5490
MFJD_bLittleBoulderCr	99.98987	2015	24.71	0.5294
MFJD_bDeerhornCr	100.94401	2015	24.77	0.5425
MFJD_aDeerhornCr	100.94401	2015	23.35	0.4575
MFJD_lwrForrestCAboundry	102.39067	2015	24.35	0.5425
MFJD_bVincentCr	105.64612	2015	25.47	0.5359
MFJD_bVinegarCr	105.64662	2015	25.32	0.6013
MFJD_aVinegarCr	107.40655	2015	25.83	0.6078
MFJD_aDavisCr	107.51940	2015	25.92	0.6078
MFJD_uprForrestCAboundry	108.65567	2015	25.89	0.6144
MFJD_aBridgeCr	108.82884	2015	25.82	0.6209
MFJD_bBridgeCr	108.82884	2015	25.72	0.6209
MFJD_bClearCr	109.20864	2015	25.79	0.6340
MFJD_aClearCr	111.02589	2015	26.69	0.6275
MFJD_aMosquitoCr	68.68502	2016	26.61	0.6078
MFJD_bRaggedCr	90.21863	2016	22.58	0.4575
MFJD_bBeaverCr	90.79900	2016	22.18	0.4314
MFJD_aLowestAlcoveOCA	91.12900	2016	22.33	0.4379
MFJD_inLowestAlcoveOCA	91.19592	2016	20.86	0.3595

MFJD_aBeaver	91.25857	2016	21.73	0.4184
MFJD_inUpperAlcoveOCA	91.44121	2016	24.46	0.4771
MFJD_bGraniteBoulderCr	92.94740	2016	21.93	0.3987
MFJD_bButteCr	93.85157	2016	25.11	0.5294
MFJD_bLittleButteCr	97.87071	2016	24.00	0.4902
MFJD_aLittleBoulderCr	99.98987	2016	23.19	0.4967
MFJD_bLittleBoulderCr	99.98987	2016	22.66	0.4444
MFJD_bDeerhornCr	100.94401	2016	24.07	0.5490
MFJD_aDeerhornCr	100.94401	2016	22.11	0.4706
MFJD_lwrForrestCAboundry	102.39067	2016	23.23	0.5229
MFJD_bVincentCr	105.64612	2016	23.77	0.5359
MFJD_bVinegarCr	105.64662	2016	23.89	0.5102
MFJD_aVinegarCr	107.40655	2016	24.69	0.5616
MFJD_uprForrestCAboundry	108.65567	2016	24.82	0.5948
MFJD_aBridgeCr	108.82884	2016	24.96	0.6054
MFJD_bBridgeCr	108.82884	2016	26.72	0.6039
MFJD_bClearCr	109.20864	2016	25.13	0.6507
MFJD_aClearCr	111.02589	2016	24.78	0.5686

Table A-7. Maximum 7DADM (°C) water temperatures and proportion of summer days with temperatures greater than 18°C for water temperature locations in Bridge Creek.

Site Name	rKM	Year	Maximum 7DADM	Proportion Days>18°C
BridgeCr_bdam1	0.56611	2009	24.70	0.68033
BridgeCr_apond1	1.21726	2009	23.18	0.46721
BridgeCr_bdam1	0.56611	2010	21.62	0.51639
BridgeCr_bdam2	0.56624	2010	21.81	0.51639
BatesPond_linebottom	0.84801	2010	22.22	0.35246
BatesPond_linemiddle	0.84801	2010	22.73	0.51639
BatesPond_linetop	0.84801	2010	22.55	0.38525
BridgeCr_apond2	1.21705	2010	21.20	0.42623
BridgeCr_apond1	1.21726	2010	21.12	0.42478
BridgeCr_pnt5mileapond	1.80105	2010	22.06	0.40164
BridgeCr_1	2.83522	2010	21.90	0.34118
BridgeCr_2	8.04197	2010	21.28	0.28235
BridgeCr_bdam1	0.56611	2011	18.29	0.40909
BridgeCr_bdam2	0.56624	2011	19.96	0.56557
BatesPond_linebottom	0.84801	2011	19.99	0.00000
BatesPond_linemiddle	0.84801	2011	19.47	0.00000
BatesPond_linetop	0.84801	2011	19.25	0.00000
BridgeCr_apond2	1.21705	2011	19.76	0.38136

BridgeCr_apond1	1.21726	2011	19.79	0.36066
BridgeCr_pnt5mileapond	1.80105	2011	20.29	0.32787
BridgeCr_1	2.83522	2011	19.48	0.13953
BridgeCr_2	8.04197	2011	19.28	0.04651
BridgeCr_bdam2	0.56624	2012	22.30	0.73874
BridgeCr_apond2	1.21705	2012	22.11	0.53153
BridgeCr_apond1	1.21726	2012	22.25	0.54955
BridgeCr_pnt5mileapond	1.80105	2012	22.23	0.53153
BridgeCr_1	2.83522	2012	21.91	0.33607
BridgeCr_2	8.04197	2012	21.79	0.22951
BridgeCr_amouth	0.00076	2013	26.10	0.82301
BridgeCr_bdam2	0.56624	2013	26.50	0.82301
BridgeCr_apond1	1.21726	2013	26.35	0.69027
BridgeCr_amouth	0.00076	2014	26.59	0.83607
BridgeCr_bdam1	0.56611	2014	26.31	0.72951
BridgeCr_bdam2	0.56624	2014	27.78	1.00000
BridgeCr_apond1	1.21726	2014	26.56	0.45082
BridgeCr_pnt5mileapond	1.80105	2014	27.67	0.45082
BridgeCr_amouth	0.00076	2015	26.47	0.72951
BridgeCr_bdam1	0.56611	2015	26.66	0.73770
BridgeCr_apond1	1.21726	2015	26.54	0.68033
BridgeCr_pnt5mileapond	1.80105	2015	26.36	0.62295
BridgeCr_amouth	0.00076	2016	24.50	0.71552
BridgeCr_bdam1	0.56611	2016	24.84	0.71552
BridgeCr_apond1	1.21726	2016	24.63	0.60345
BridgeCr_pnt5mileapond	1.80105	2016	24.57	0.61475

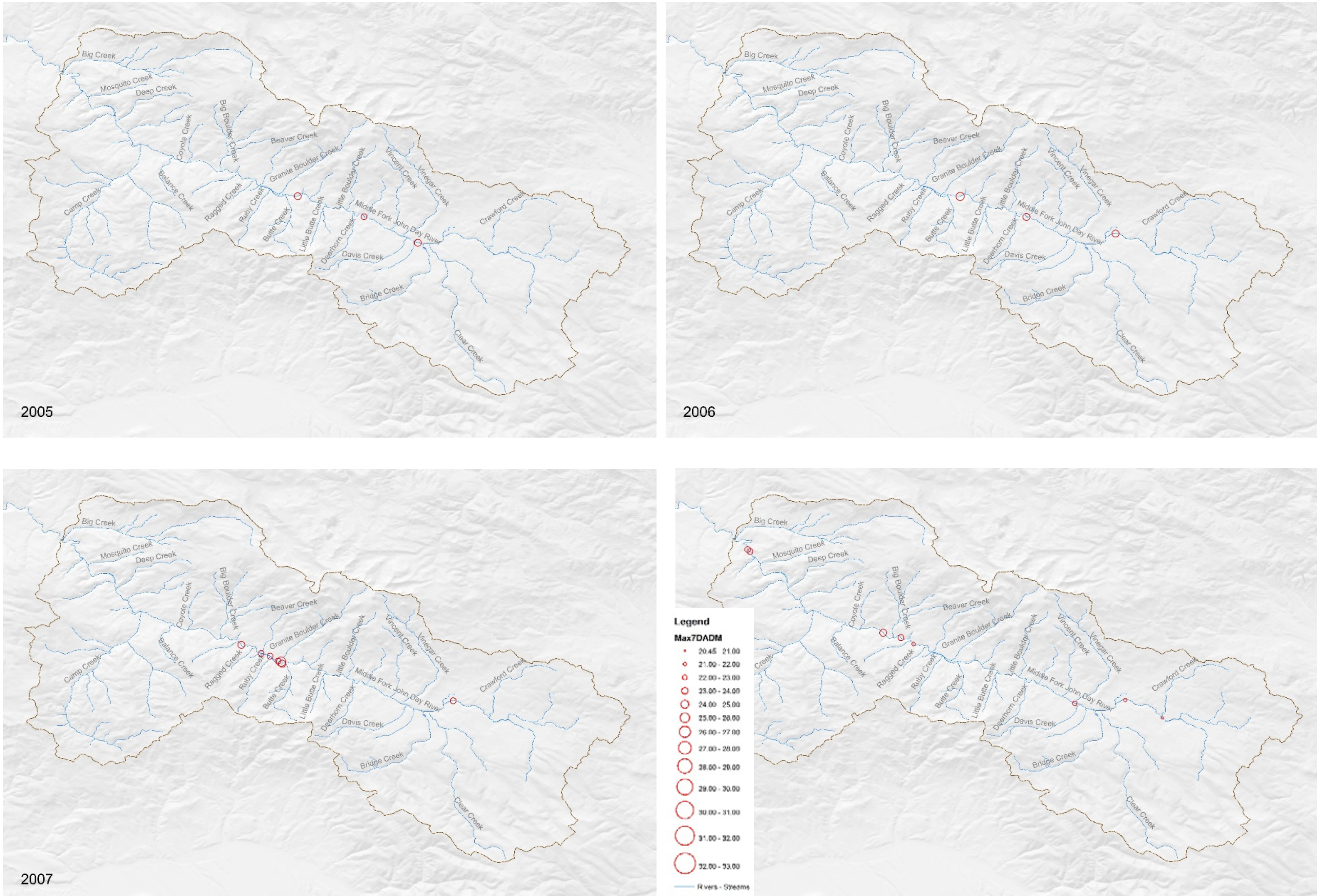


Figure A-1. MFJD maximum 7DADM (°C) water temperatures on the Middle Fork John Day by year.

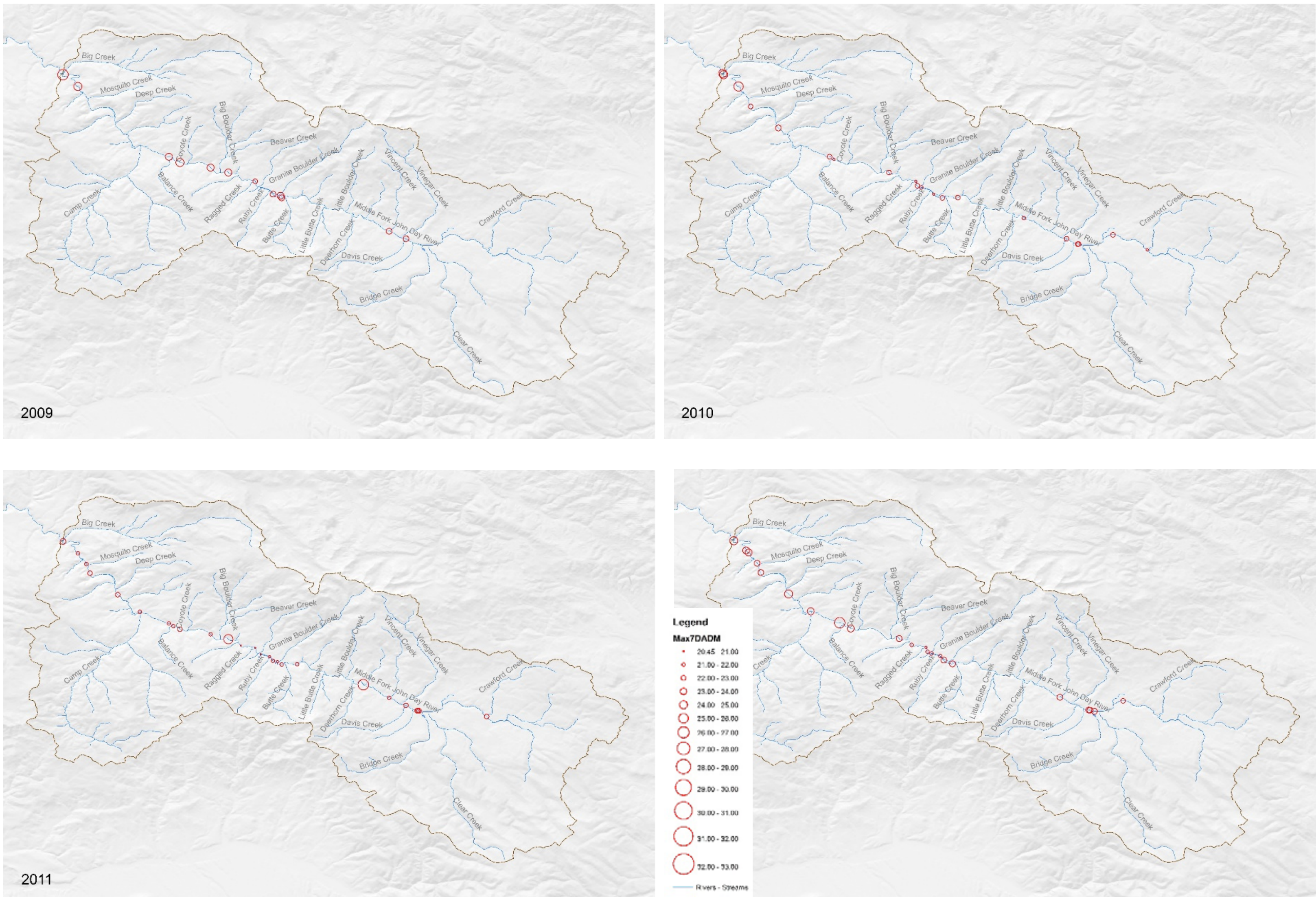


Figure A-1, con't. MFJD maximum 7DADM (°C) water temperatures on the Middle Fork John Day by year.

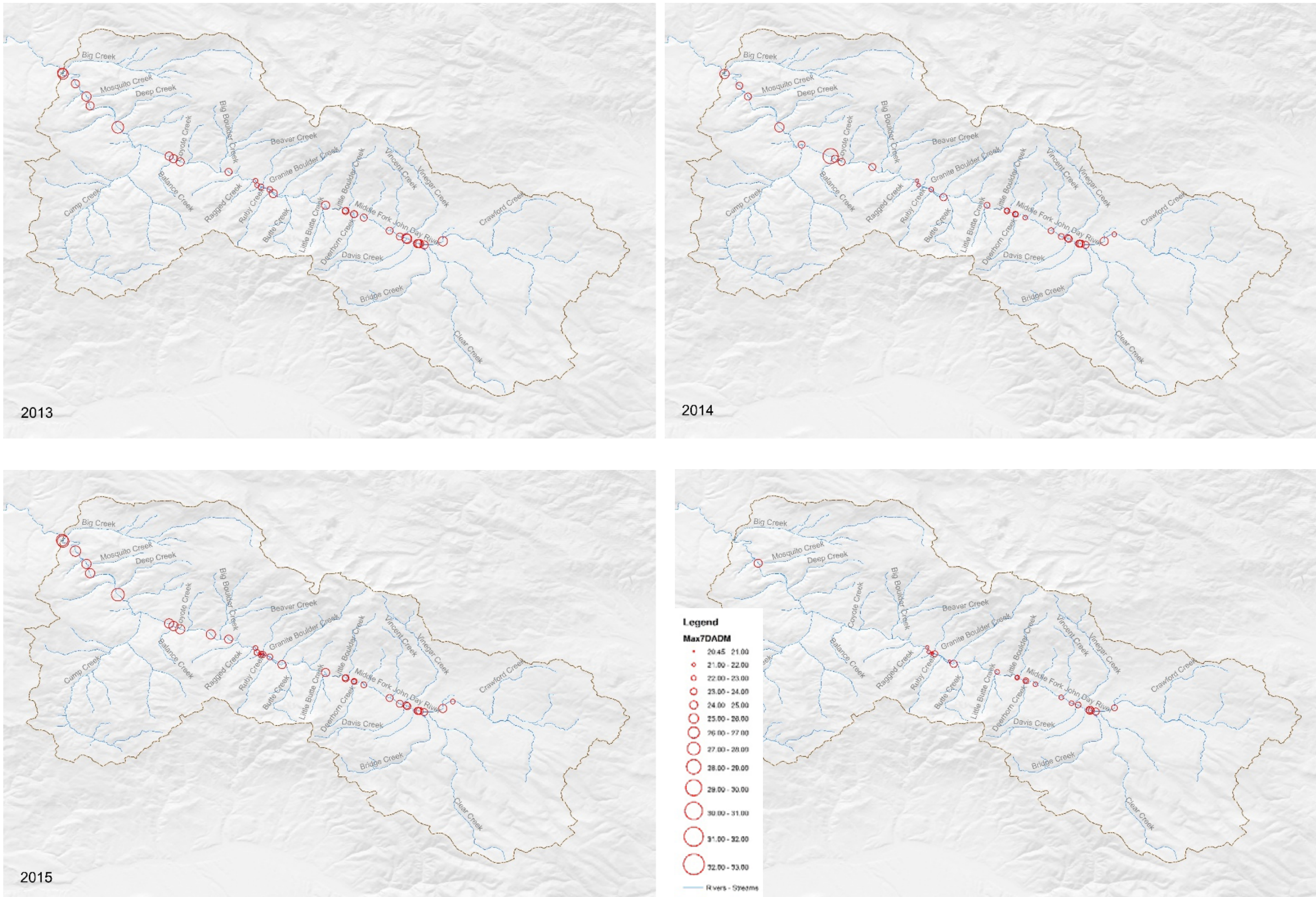


Figure A-1, con't. MFJD maximum 7DADM (°C) water temperatures on the Middle Fork John Day by year.

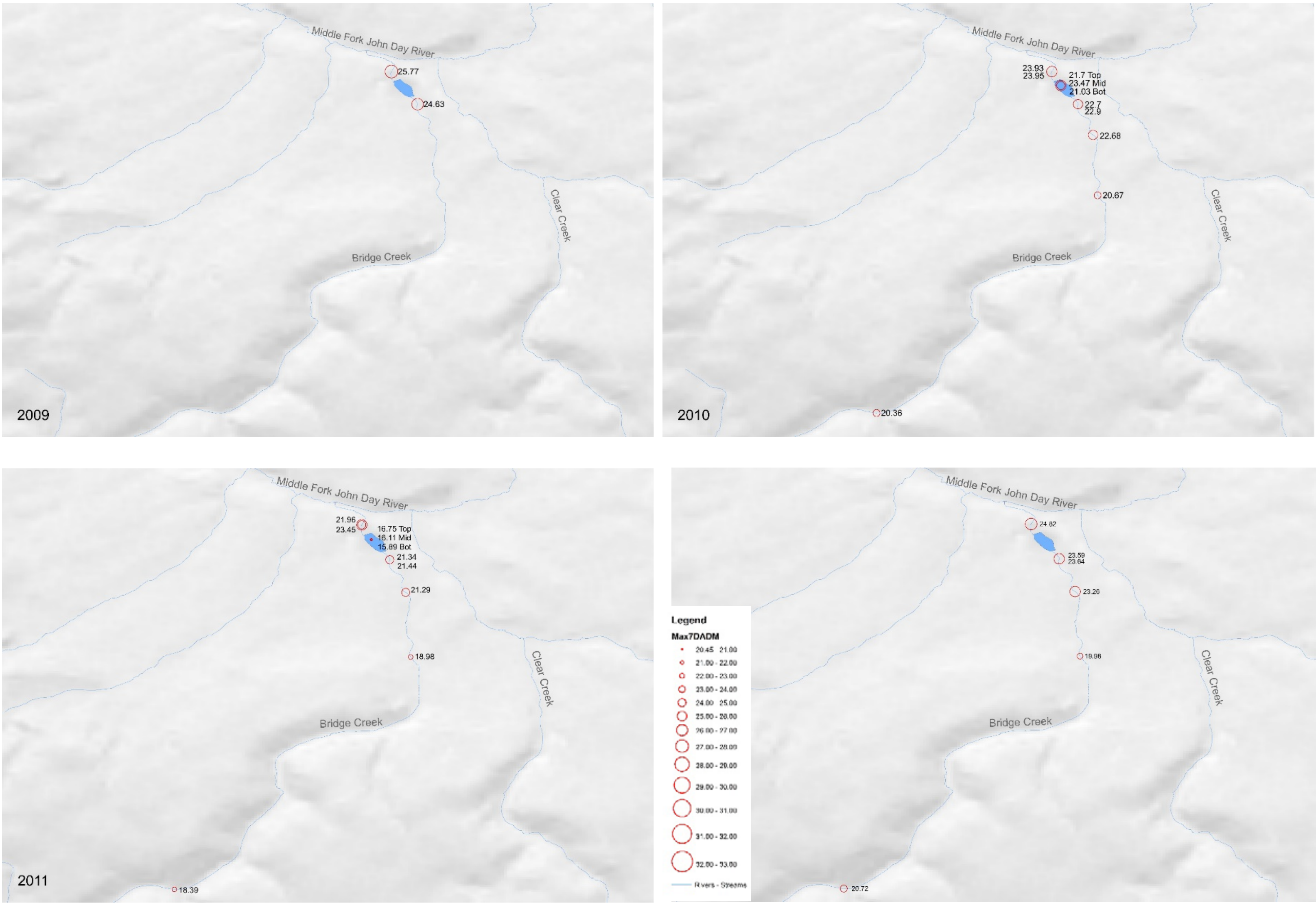


Figure A-2. Bridge Creek maximum 7DADM (°C) water temperatures.

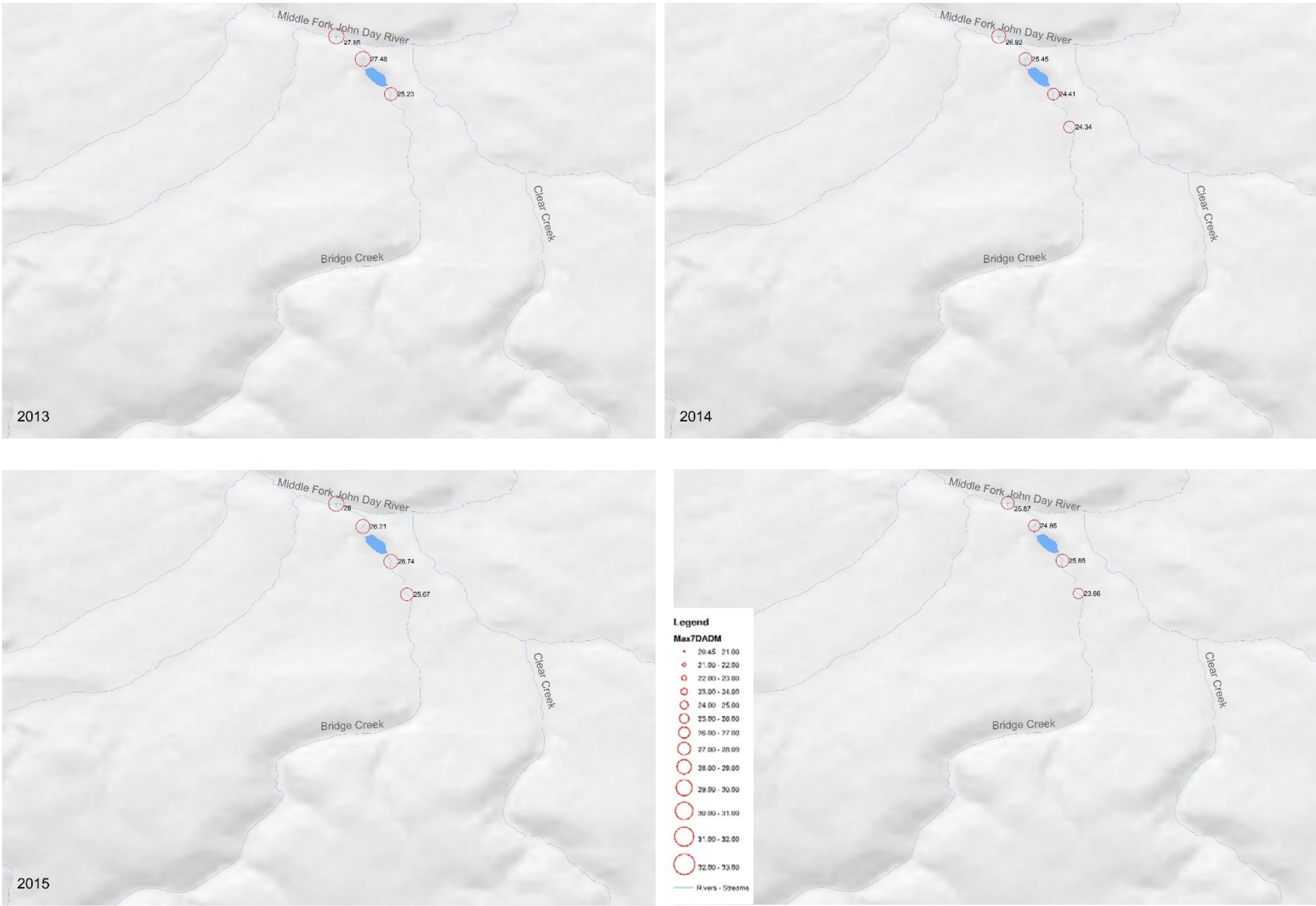


Figure A-2, con't. Bridge Creek maximum 7DADM (°C) water temperatures.

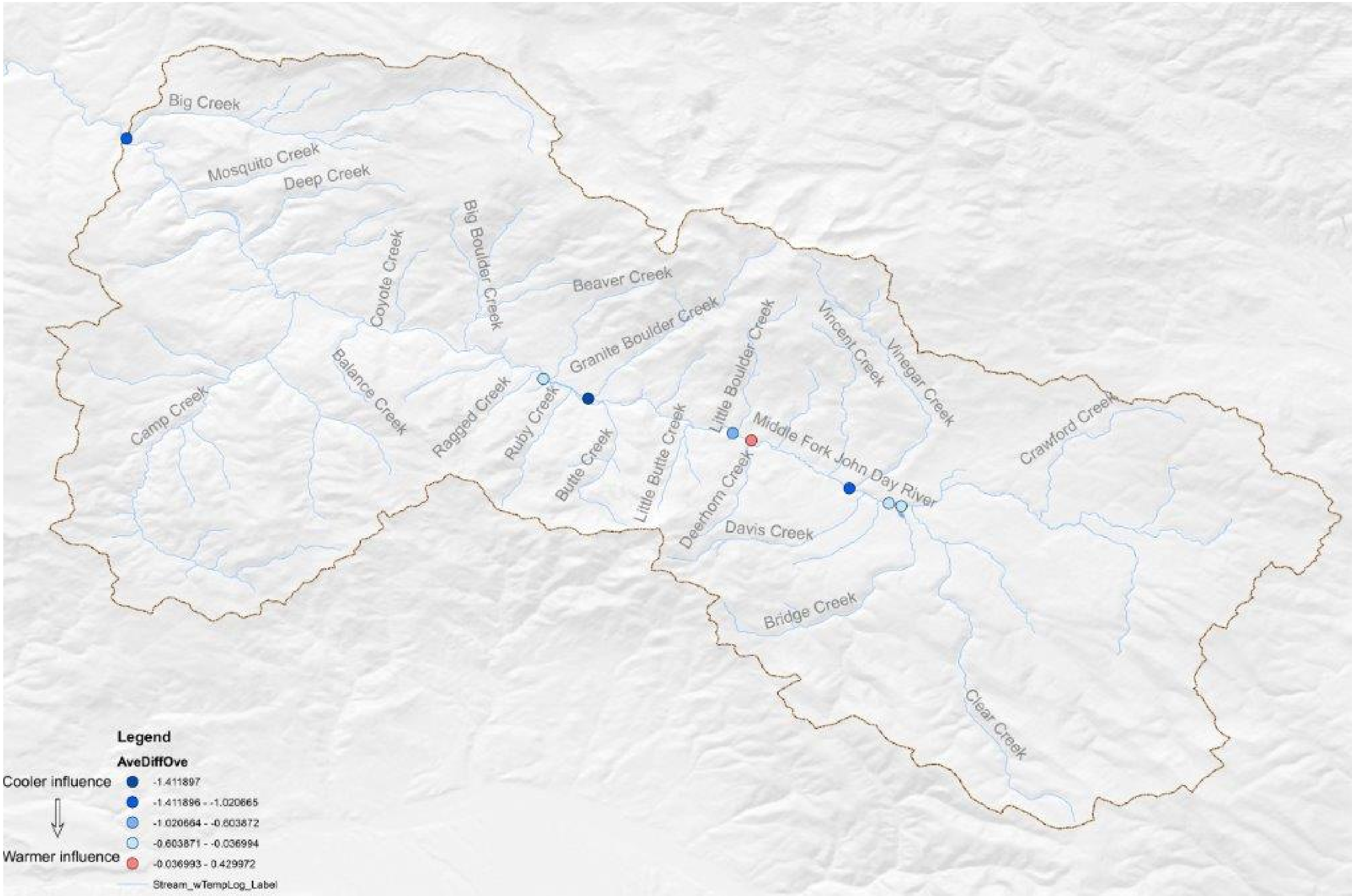


Figure A-3. Average difference between water temperature loggers upstream and downstream of tributaries on the MFJDR.

Appendix B.

Water Temperature Logger Accuracy and Logger Placement Protocols

NFJDWC

Loggers are anchored to a rebar post driven into the creek bed. If water conditions make it unsafe, they are anchored to vegetation on the bank with cable. The logger is attached to a brick and set such that the logger rests below the brick against the stream bed. This avoids solar heating of the logger through the water. Loggers are placed in the deepest part of the stream that can be safely and securely accessed. We are also in the process of developing a site survey for each logger, (NFJDWC) which should give some insight to each loggers' placement.

Data is collected at the end of the season when loggers are pulled.

CTWSRO

Our loggers are placed in a slightly different way than what Justin describes here. I will email you the doc that describes our placement and then if you like you can create kind of a hybrid description of the two methods.

Prior to my arrival, our loggers were placed in spring, checked (maybe) throughout the summer once or more, and downloaded in the late fall. When I arrived we started downloading in the fall and then putting them out again for the winter season, then collecting and downloading again in the spring (so twice a year). But that has only been going on for just over a year now, although I do plan for it to continue.

Appendix H – Monitoring and Assessment of Critical Thermal Dynamics in Upper Middle Fork of the John Day River, 2008-2016

Oregon State University MFIMW Team, Corvallis OR

Abstract

The Oregon State University (OSU) team conducted hydro-thermal stream monitoring on the Middle Fork of the John Day River (MFJDR) at the Oxbow and Forrest Conservation Areas since 2008. Regulation of temperature within these critical habitats is a primary factor in fish survival. Fiber optic distributed temperature sensing (DTS) monitored about 8,000 meters per summer with 1 meter and at most 1 minute resolution to observe peak summer temperatures, supplemented by groundwater contribution, stream discharge, and stream bathymetry across the conservation sites. Diurnal cycles during summer observation ranged from absolute minimum of 9° Celsius (C) to absolute maximum of 26°C. Salmonids are sensitive to stream temperatures above 18°C, resulting in depressed growth and survival, while sustained temperatures above 24°C have directly lethal effects (Bell, 1991). Groundwater inputs did not significantly decrease MFJDR stream temperatures, but did effect tributary temperatures. The primary cooling mechanism of the MFJDR occurred at the confluence of the main stem and its tributaries, where tributaries supplied cooler, groundwater rich water into the main channel. Physically-based thermal modeling indicated that solar radiation was the primary driver for gains in stream temperature in the main stem MFJDR; river surface area change associated with restoration actions of the MFJDR main stem explained 98% of the change in stream temperature. DTS monitoring of the Phase-2 Oxbow Conservation Area (OCA) restoration project was shown to decrease main-stem temperature by over 0.6°C (1°F), which model results indicate is due to reduced water surface area. Model results of shade impact provided by riparian vegetation was shown to be a very slow restoration method, unlikely to provide significant thermal effects within a decade on rivers the size of the MFJDR. Finally, while re-connecting the river with the floodplain has many habitat benefits, model results indicate neither an increase to summer low-flow nor a reduction in summer peak temperatures.

Introduction

Background

Stream temperature is a primary factor of ecosystem function, especially with respect to salmonid survival which requires temperatures below critical thresholds, and thus is employed in regulations supporting the Clean Water and Endangered Species Acts. Thus, Oregon has developed temperature standards which lead to river specific Total Maximum Daily Loads (TMDL) of heat to provide regulatory framework to achieve benchmarks to inform restoration efforts to improve thermally impaired streams (ORDEQ, 2017). Stream temperature varies both spatially and temporally, which fiber optic DTS is capable of measuring 1-m and 1-min resolution with better than 0.1^{o}C accuracy (Selker et al., 2006). Typical habitat analysis relies on characterization based on large longitudinal units, typically over 50 meters (ORDEQ, 2017). Despite mixing processes occurring at these large scales, discrete cold patches have been shown to provide significant habitat for temperature sensitive species (Torgersen et al., 1999; Ebersole et al., 2003; RDG, 2007; BOR, 2008). While measurement of temperature is necessary, prediction of stream temperature under the variability of climate, and due to restoration efforts, can only feasibly be completed using physically-based numerical models of stream thermodynamics. The spatial and temporal temperature data obtained using DTS is essential for accurate calibration of stream temperature models, testing their ability to capture large-scale and fine-scale temperature dynamics, as required to guide science-based restoration planning.

Goals and objectives

The goal of the study was to assess the impact of restoration efforts on stream temperature. Thermal impacts of effective shaded area, groundwater influence, total stream area and depth, were assessed. Specific monitoring objectives were: to determine the occurrence of thermal refugia along the mainstem, evaluate peak temperature in restored and unrestored mainstem sections, determine tributary contribution to mainstem temperature, and determine floodplain groundwater gradients on summer stream flow. To isolate restoration effects from particular weather, restoration impact was evaluated by simulation of pre- and post-restoration using the same meteorological inputs into validated HeatSource models (Huff, 2009; O'Donnell, 2012; Hall, 2015). These objectives were addressed through the efforts of seven graduate students and Dr. John Selker in the water resources engineering graduate program at OSU who invested over 20 person years of research in this project.

Key Findings

1. Decreased stream surface area results in proportional decrease in stream temperature. Corollary: Consolidation of two channels to one decreased seven-day average daily maximum (7DADM) stream temperature.
2. DTS data indicate large tributaries acted as the primary source of cool water during summer peak temperatures. No additional sources of cold water refugia were identified in the main-stem of the MFJDR due to turbulent mixing, but could be created in side-channels located near groundwater upwelling.
3. Increasing shade cover along the mainstem will ultimately reduce stream temperatures, but due to the slow pace of vegetation growth, thermal impacts associated with riparian plantings were not detectable over the 9-year period of this study.
4. Groundwater thermal influence upon headwater tributary channels was significant, but not along the mainstem of the MFJDR.
5. Restoration and re-connection of floodplains to the MFJDR did not contribute to increased late season flow or temperature reduction as shown by model results. Riparian vegetation sufficiently tall to create shade could consume significant portions of current summer low-flow.

Site Selection

The foci of this study were sections of the MFJDR where the restoration efforts would have immediate effect and were appropriate for the evaluation of longer temporal scale impacts of restoration such as the growth of woody plants.

The first section of the MJFDR that was monitored is on the Middle Fork Forrest Conservation Area (FCA) property, river miles 64-67 (Figure 1). This section of river has a disused railroad berm running parallel to the river. DTS monitoring extended a quarter mile (0.4 km) above and below the 1-mile (1.6 km) section of river channel constrained between the road and railroad berm. DTS monitoring was implemented throughout the FCA (Figure 1) (See Appendix G for table of DTS monitoring efforts completed each year). Groundwater monitoring wells were installed along the full 5.6 km (3.5-mile) river reach of the FCA (Figure 1). Past restoration on this site included installation of engineering log jams (ELJ) and removal of bank stabilizing rip-rap along 0.6 miles of stream reach, in addition fencing of riparian area, and vegetation management was implemented throughout the 3.5-mile reach (See Background, Restoration Inventory).

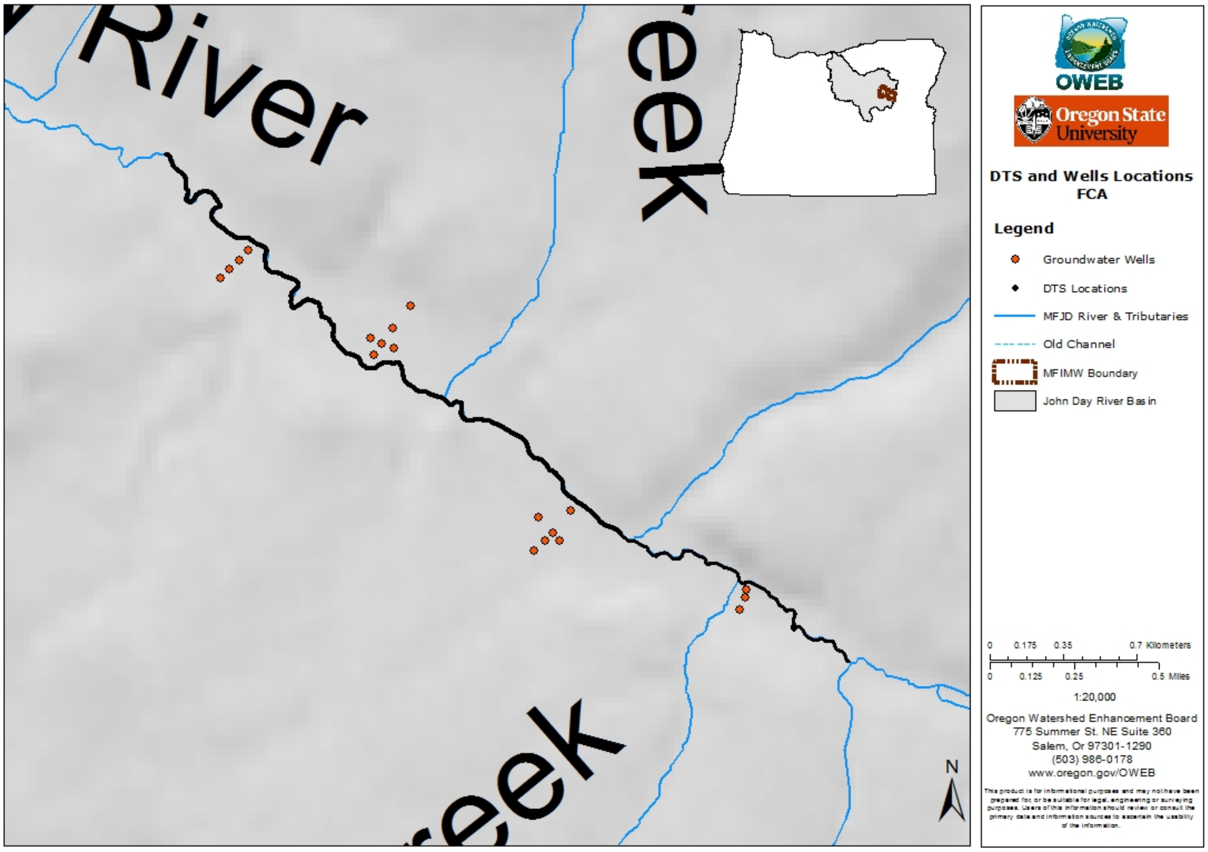


Figure 1. DTS and groundwater monitoring locations on the Middle Fork John Day Forrest Conservation Area (FCA).

In the early to mid-1900's there was severe dredge mining of the Oxbow property's flood plain, which resulted in widespread geomorphic disturbance, splitting the single channel into two parallel channels for about 1 mile (1.6 km). The OCA property was monitored using DTS and groundwater observation wells over the most impacted section (river miles 55-58) (Figure 2). Phase 2 restoration efforts resulted in abandoning the northern stream channel and directing all flow into the southern channel (See Background, Restoration Inventory). DTS monitoring occurred in the northern channel, the southern channel, upstream of the split channels to the Coyote Bluff bridge and downstream of the split channels by approximately 2 miles (3.2km) (Figure 2) (See Appendix G for table of DTS monitoring efforts completed each year). Phases 3, 4 and 5 are planned to restore channel geometry and vegetative covering of the impaired sedge-lined linear channel to a natural meandering stream, thus increasing channel length, which was estimated to increase stream surface area by 70%. Groundwater monitoring was carried out using crossing transects of wells that spanned the OCA with 500m spacing (Figure 2).

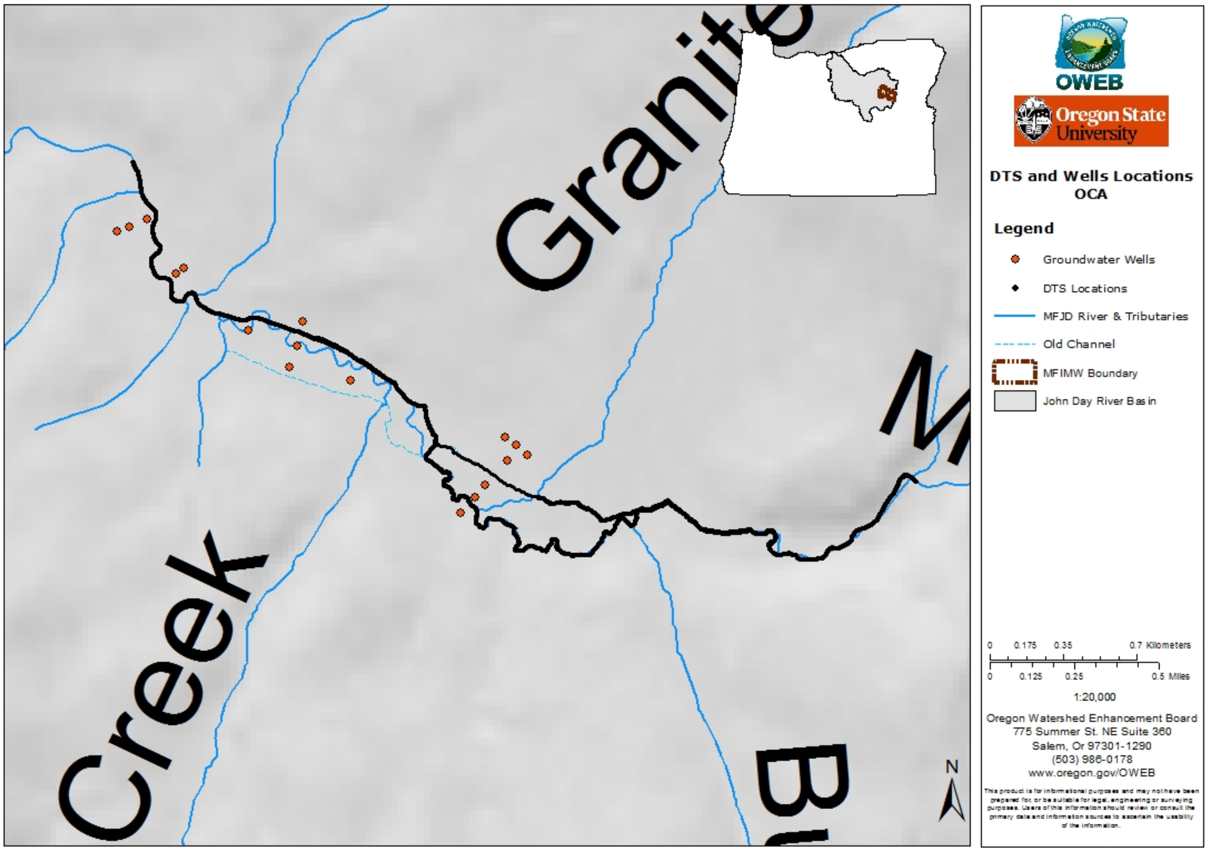


Figure 2. DTS and groundwater monitoring locations on the Middle Fork John Day Oxbow Conservation Area (OCA).

Methods

Temperature Monitoring

The DTS was employed to measure temperature at most each 10 minutes with spatial resolution of 1.5m or less along 2,000m installations upstream and downstream from the instrument. Cables were secured to the streambed every 5 to 10m using alluvial rocks. At the time of each cable placement, a high precision RTK TOPCON GR-3 survey grade GPS (10mm horizontal, 15mm vertical accuracy) was used to map its location and associated stream depths. High precision data-logging thermometers (RBRsolo, +/- 0.002°C) were placed in three controlled temperature baths with fiber optic cable; and temperature loggers with 0.1^{o}C accuracy were placed at 500m intervals along the fiber for calibration and validation (per Hausner et al. 2011). 7 to 14 days of temperature data were collected along each river section in mid-summer of 2008, 2009, 2010, 2011, 2013, 2014 and 2016 (see Appendices A, C, D and G for further monitoring details).

Temperature Modeling (HeatSource)

Stream temperature modeling (HeatSource v8.08, Boyd et al., 2012) was undertaken to assess restoration and baseline thermal conditions of the OCA Phase 2 and Phases 3, 4, and 5 projects. Model evaluation was used to interpret changes to stream area, effective shade and to identify primary drivers of thermal influence upon the monitored MFJD reach (Huff, 2009; O'Donnell, 2012; Hall, 2015; Appendices A, C and D). Boundary conditions from various observational and literature sources were used as model inputs, including: vegetation, topographic angle, stream elevation, stream width, streambed porosity, streambed thermal conductivity, streambed diffusivity, bank angle, Manning's n, boundary temperatures, streamflow, groundwater, tributary temperatures, meteorological data, and wind sheltering (O'Donnell, 2012; Hall, 2015).

TIR and FLIR data anomaly

Thermal Infrared Radiometry (TIR) and Forward-Looking Infrared Radar (FLIR) provide potentially large spatial scale temperature measurements for identification of cold water influences upon a stream reach (Torgersen et al., 2001). Stream temperature maximums are the greatest influence upon fish habitat viability, therefore TIR and FLIR data was collected between 13:50 and 17:00 on the OCA in four studies (Figure 3) (O'Donnell, 2012). These FLIR stream data suggested a persistent and significant cooling trend between Butte and Granite Creeks (Figure 3), which was hypothesized as resulting from groundwater influence, and thus taken to be an area of special interest for restoration. DTS instrumentation was deployed and discharge measurements were collected at the same location where TIR/FLIR data was captured to assist in identification of the cause of this colder water observed at this location.

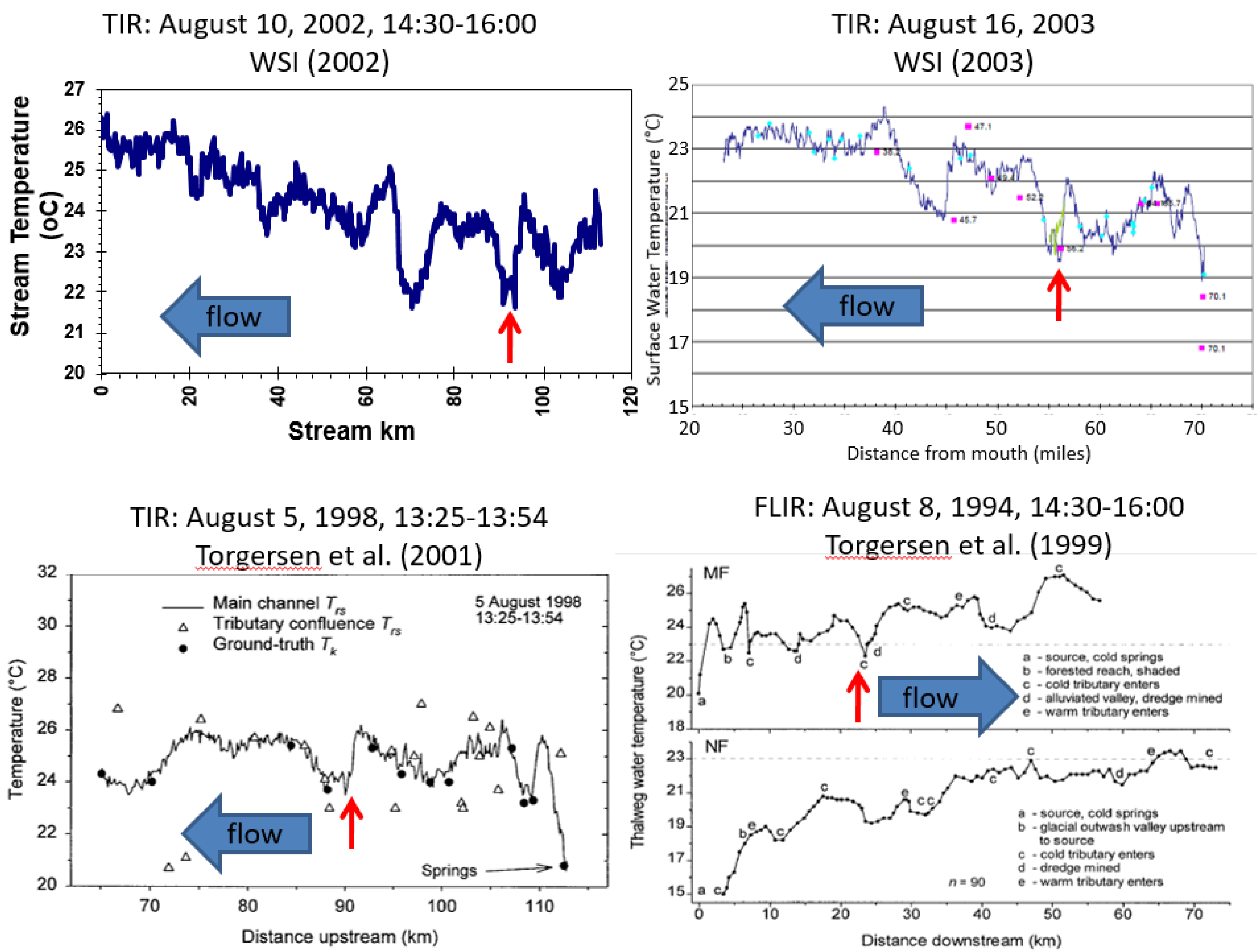


Figure 3. FLIR data taken on four previous studies showing an apparent decrease in temperature between Butte and Granite creeks (red arrows mark these locations) (O'Donnell, 2012).

Cold-water Refugia

The DTS data were interpreted seeking to identify cold-water refugia, hyporheic exchange (HE) and groundwater (GW). Cold-water refugia were sought both in the thalweg (center) and bank edges looking for any locations where consistent temperature decreases were discernable. GW has very stable temporal temperature due to long subsurface residence times, and thus mid-summer locations of GW upwelling are evident as a locations of decreased stream temperature. Stream water can also interact with the local geological setting by entering the subsurface then re-emerging into the stream. This is known as hyporheic exchange (HE), and it is evident in thermal data as an averaging process which reduces daily maximum and increases daily minimum temperatures. Locations identified as GW or HE were quantified based on DTS temperature to provide the magnitude and locations of these processes (Huff, 2009; Arik, 2011; O'Donnell, 2012).

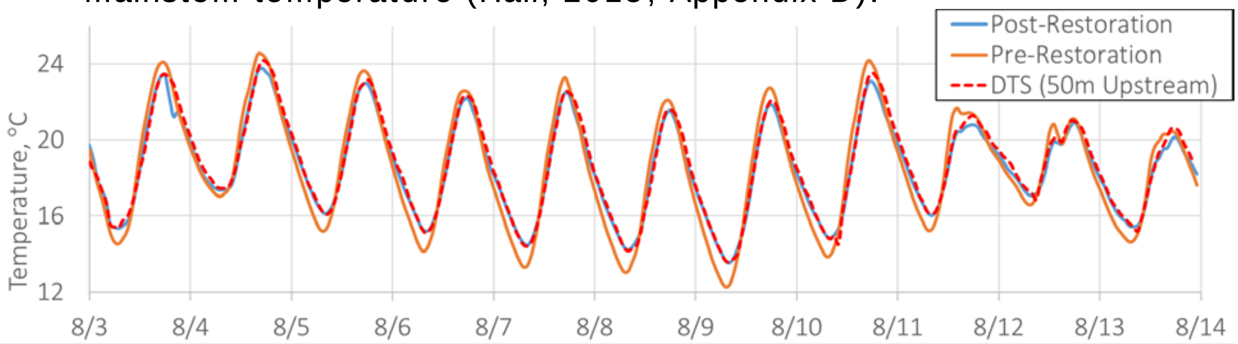
Groundwater

Groundwater temperature and level were recorded on one hour intervals with HOBO U20 and U20L pressure transducers in transects perpendicular and parallel to each reach on approximately 500m spacing (Figures 1 and 2). Data were used to assist in calibration of HeatSource thermal models (Hall, 2015; Appendix D), Floodplain connectivity modeling (Nash et al., 2017; Appendix E), and Evapotranspiration (ET) modeling (Kollen, 2016; Nash et al., 2017; Appendices E and F).

Results

Temperature Monitoring / Modeling (HeatSource)

- DTS monitoring and HeatSource model evaluation produced a key finding from the OCA Phase 2 channel closure of the North channel. A reduction of peak temperature by 0.91°C, an increase in nighttime temperature by 0.86°C and a reduction in the 7 Day Average Daily Maximum (7DADMT) temperature by 0.65°C were observed (Figure 4, Hall, 2015). This field result agreed well with numerical HeatSource simulations (Figure 4), supporting the critical importance of reduction of low-flow stream area to reduce peak stream temperature, and confirming the utility of modeling stream temperature for the purposes of restoration planning. A very practical outcome of this work was the finding that planners can use the pre- and post-restoration stream area as a key metric of expected thermal impact of restoration projects (explaining essentially all the observed change in stream temperature). Employing this result on the MFJDR, we found that the planned restoration of the OCA Phase 3, 4 and 5 projects, which increased stream surface area, should be expected to slightly increase peak stream temperatures of the unvegetated channel, however if shade cover becomes established or exposed area is less than predicted a decrease in temperature is expected (Hall, 2015; Appendix D). Model results do not seek to predict locations of refugia or habitat established due to restoration, but rather indicate overall changes to mainstem temperature (Hall, 2015; Appendix D).



Nighttime, Daytime, and Maximum Changes

Location	Nightly Change, °C (2:00 AM-6:00 AM)	Peak Daytime Change, °C (2:00 PM-5:00 PM)	7DADM Change, °C
Confluence	**0.86**	**-0.91**	**-0.65**
Downstream	0.72	-0.62	-0.54

Figure 4. DTS monitored temperature and HeatSource modeled impact of Phase 2 restoration on stream temperature (From Hall, 2015).

- HeatSource model results showed a linear relationship between the exposed water surface area of a reach and change in stream temperature at the downstream end of the modeled reach (Figure 5). This is as expected since all major thermal exchange processes are proportional to stream area (e.g., solar radiation, evaporation, bed conduction). Model results indicate solar input is the main agent of heating. While modeling of shade cover shows a theoretical reduction in stream temperature over an extended period of time, due to the width of the MFJDR and slow growth of tall plants, no significant impact to mainstem temperature was observed due to introduction of bank shading or ELJ (Hall, 2015; Appendix D).

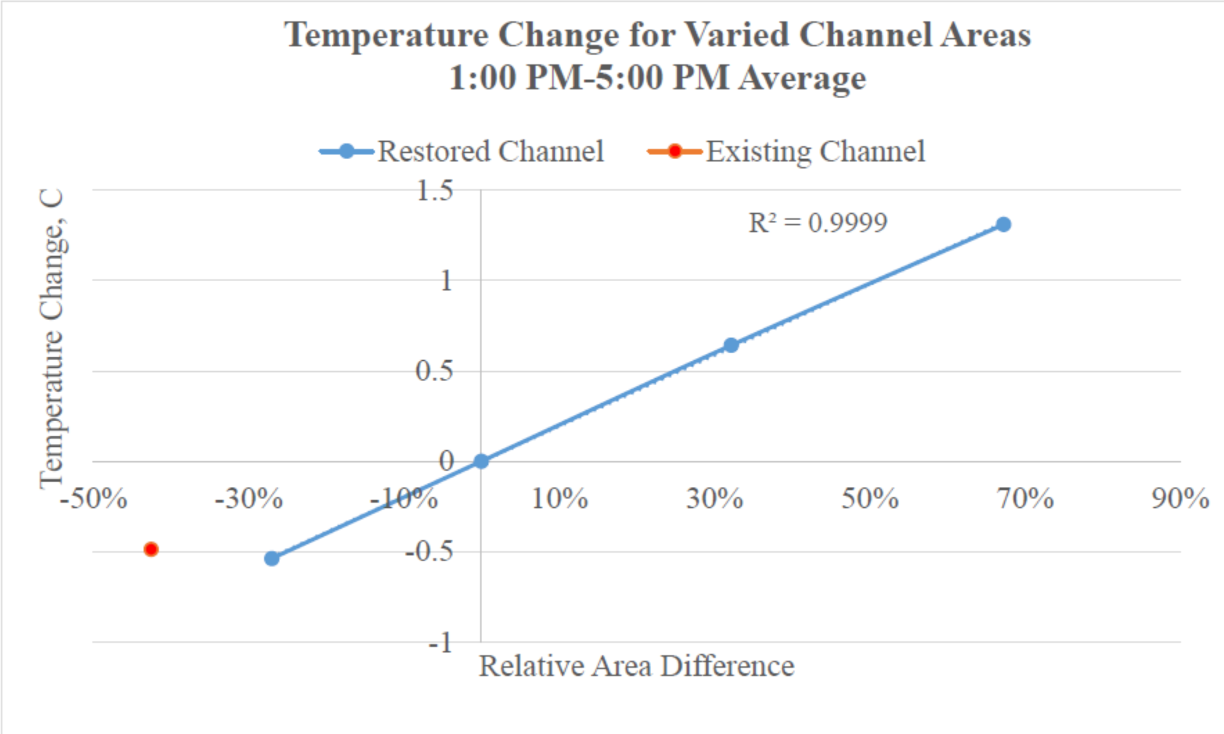


Figure 5. HeatSource model predictions of OCA Phase 3 restoration impacts on stream temperature, showing the strong correlation between stream temperature and changes to effective stream area. Increasing stream area is predicted to causes a direct increase in stream temperature and decreasing stream area would be expected to cause a decrease in stream temperature (Hall, 2015).

- Comparison of TIR/FLIR data with results from DTS dataset analysis and synoptic discharge measurements indicate that the temperature decrease observed was unlikely to be due to groundwater inflows. A temporal thermal lag of 70 minutes in stream temperature was observed caused by slow flow velocities in a deep pools (Figure 6). Thus, the apparent cooling seen in the FLIR data, which was always taken in the mid-afternoon disappeared later in the afternoon. These effects were captured in the DTS data, and confirmed by hydrodynamic modeling of this reach of the river (O'Donnell, 2012). Thus, it must be emphasized that the time of day and collection rate of TIR and FLIR data are important factors when interpreting this type of "snap-shot" data, spatially and temporally limited data likely does not necessarily accurately describe stream temperature dynamics over the course of an entire day. At the same time, regardless of its source, this cold pool was seen to be used as a refuge for salmon during hot days. The key finding was that the cause of the cool pool was due to stream-flow processes rather than groundwater processes. Understanding this is critical to making appropriate restoration decisions.

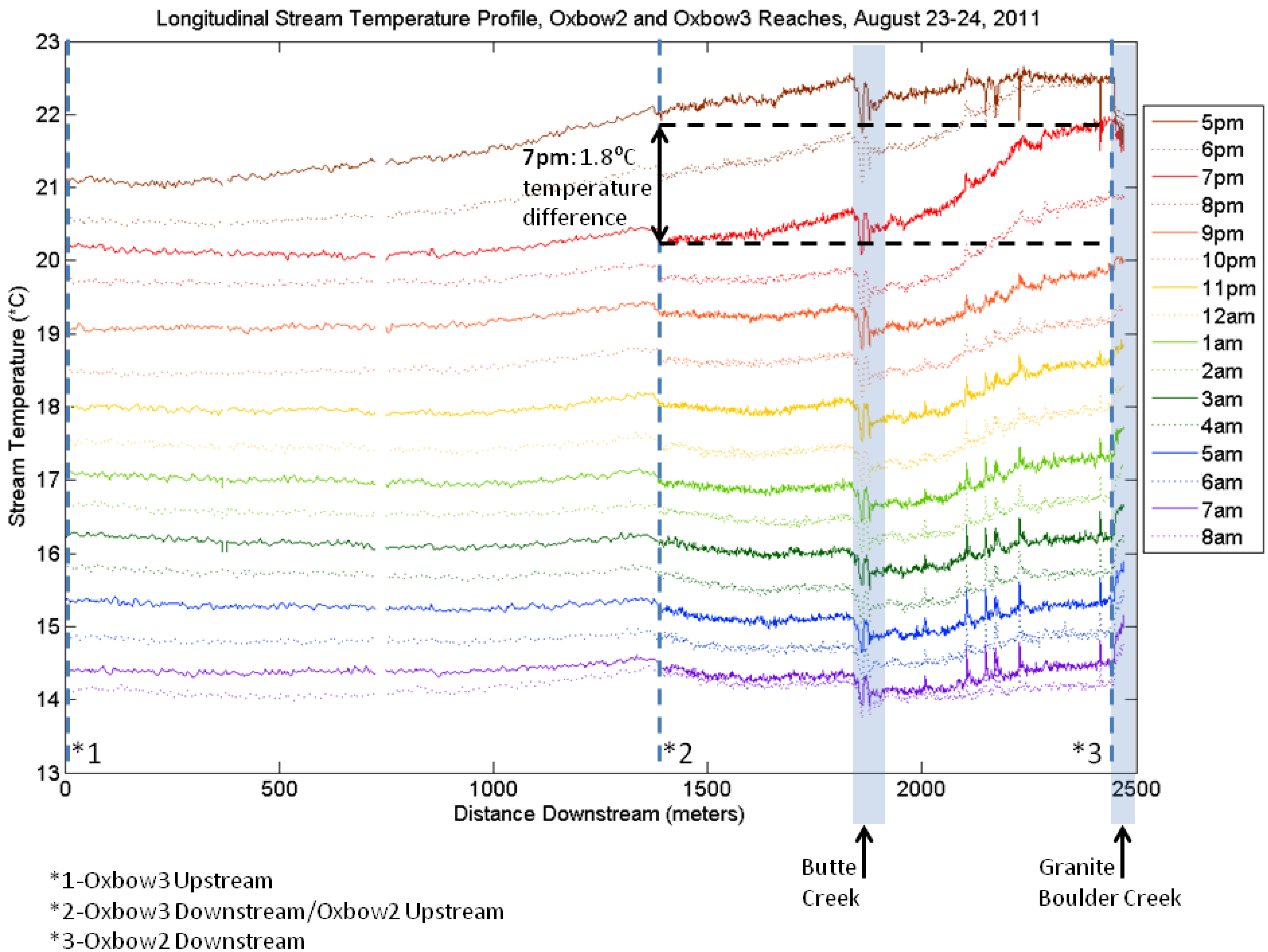


Figure 6. Hourly stream temperature observations from the reach which was identified by TIR/FLIR as an area of potential groundwater cooling (O'Donnell, 2012; August 12-25, 2011). While in the mid-day the temperatures do decrease in the downstream direction, this decrease entirely disappears by 17:00 since it reflected a moving patch of cold water rather than a local addition of groundwater. This illustrates how snapshot data from TIR/FLIR can potentially misinform identification of cold water influences upon the reach due to the transient nature of stream temperature propagation.

Cold-water Refugia

- DTS data revealed no significant localized cold water refugia locations along the monitored sections of the mainstem of the MFJDR due to GW or HE (O'Donnell, 2012, Appendix C). This was consistent with expectations, since the river has turbulence sufficient to maintain full mixing, and the GW and HE fluxes were insignificant in comparison to the streamflow.
- Larger tributaries such as Davis, Vinegar and Granite Boulder creeks show significant influence on mainstem temperatures for an extended length (>100m) due to their significant contribution to discharge (Figure 7), up to 25% from Granite Boulder Creek (Hall, 2015, Appendix D). Creek temperature contribution was shown to change MFJDR temperature between 0.2°C and 2.5°C due to the contribution of the larger tributaries (Huff, 2009, O'Donnell, 2012, Hall, 2015; Appendices A, C and D).
- Temperature impacts of GW and HE were significant in tributaries, for instance reducing peak Big Boulder Creek temperatures by approximately 0.5°C (Figure 8, Arik, 2011). Groundwater fed tributary inflows to the mainstem were found to provide significant thermal refugia, providing locations of step drop in temperature (Appendices A, B and C).

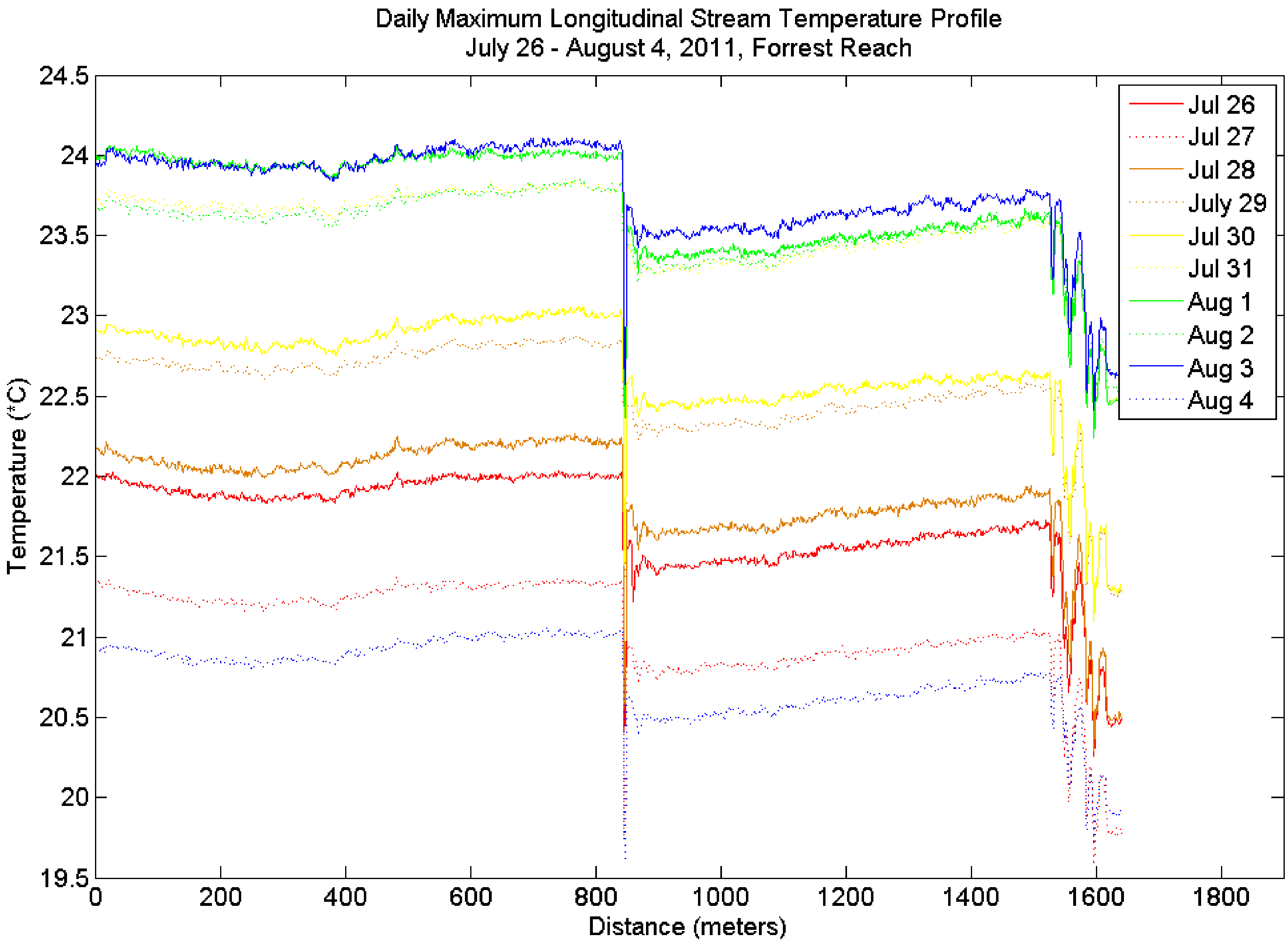


Figure 7. Example DTS trace data during peak summer of 2011, Davis and Vinegar creeks contribution to MFJDR temperature. See Appendices A, C and D for additional DTS trace analyses.

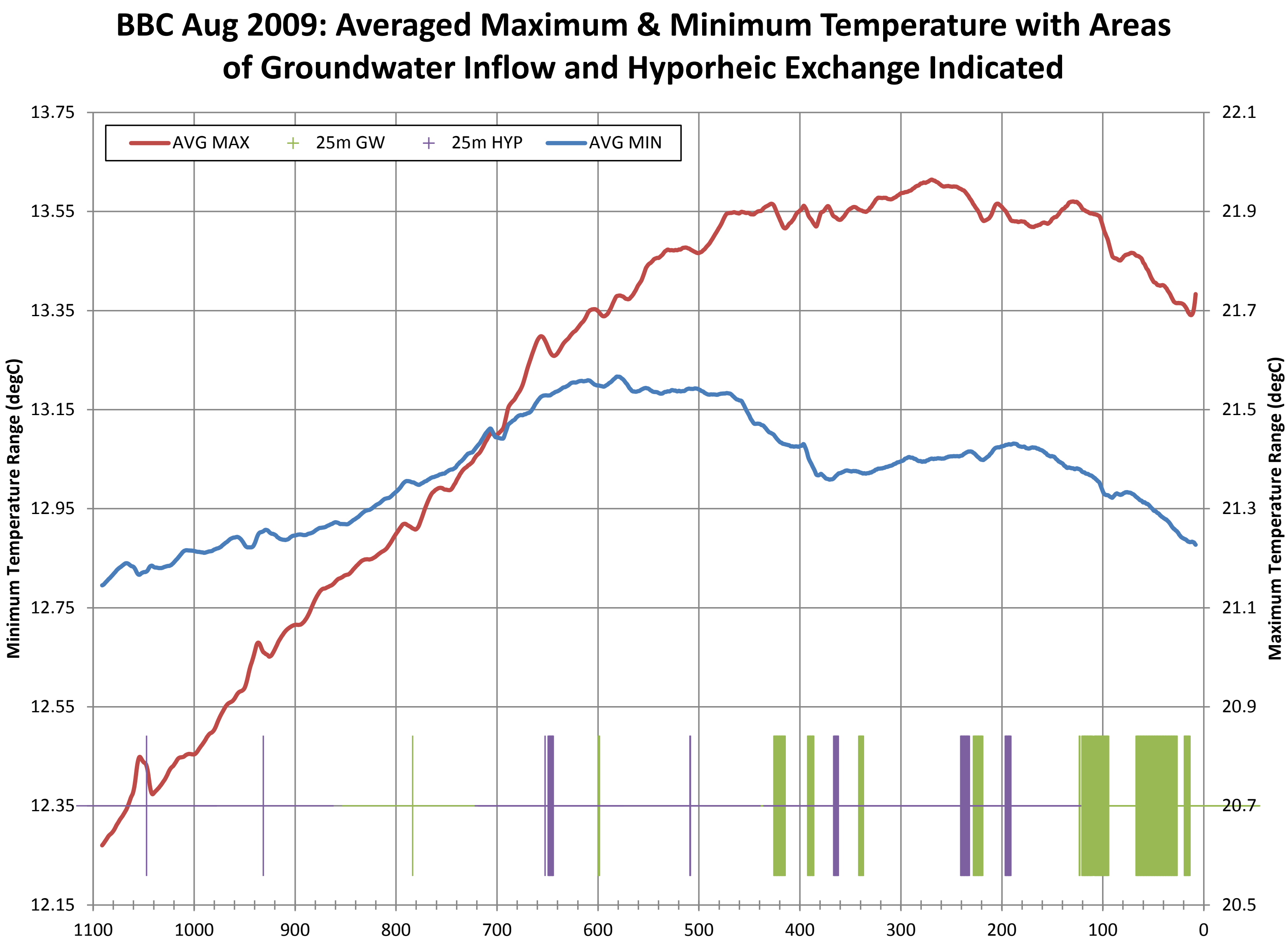


Figure 8. Thermal influences upon Big Boulder Creek through DTS observations (Arik, 2011). The upward trend in temperatures suggests that without GW and HE the stream would have been over 1^{o} C warmer.

Groundwater

- The impact of re-connection of the MFJDR to its floodplains was studied with respect to summer low-flow. By reducing the vertical displacement between the stream bed and the floodplain. Modeled water table elevation (WTE) showed strong agreement with observed data (Figure 9).
- Model results predict increased ET losses due to an increase in the water table elevation, allowing vegetation to access this water, which could lead to significant consumption of water which would otherwise have gone to the stream (Kollen, 2016; Nash et al., 2017; Appendices E and F) at the rate of approximately 3-6 cfs for a 10 mile reach.
- The spring-time storage of water within the floodplain was found to increase, but since the gradient in water level between the floodplain and stream was reduced, lateral drainage to the channel appeared to decrease (Figure 10), though the change was inconsequential relative to temperature control (Nash et al., 2017; Appendices E and F).

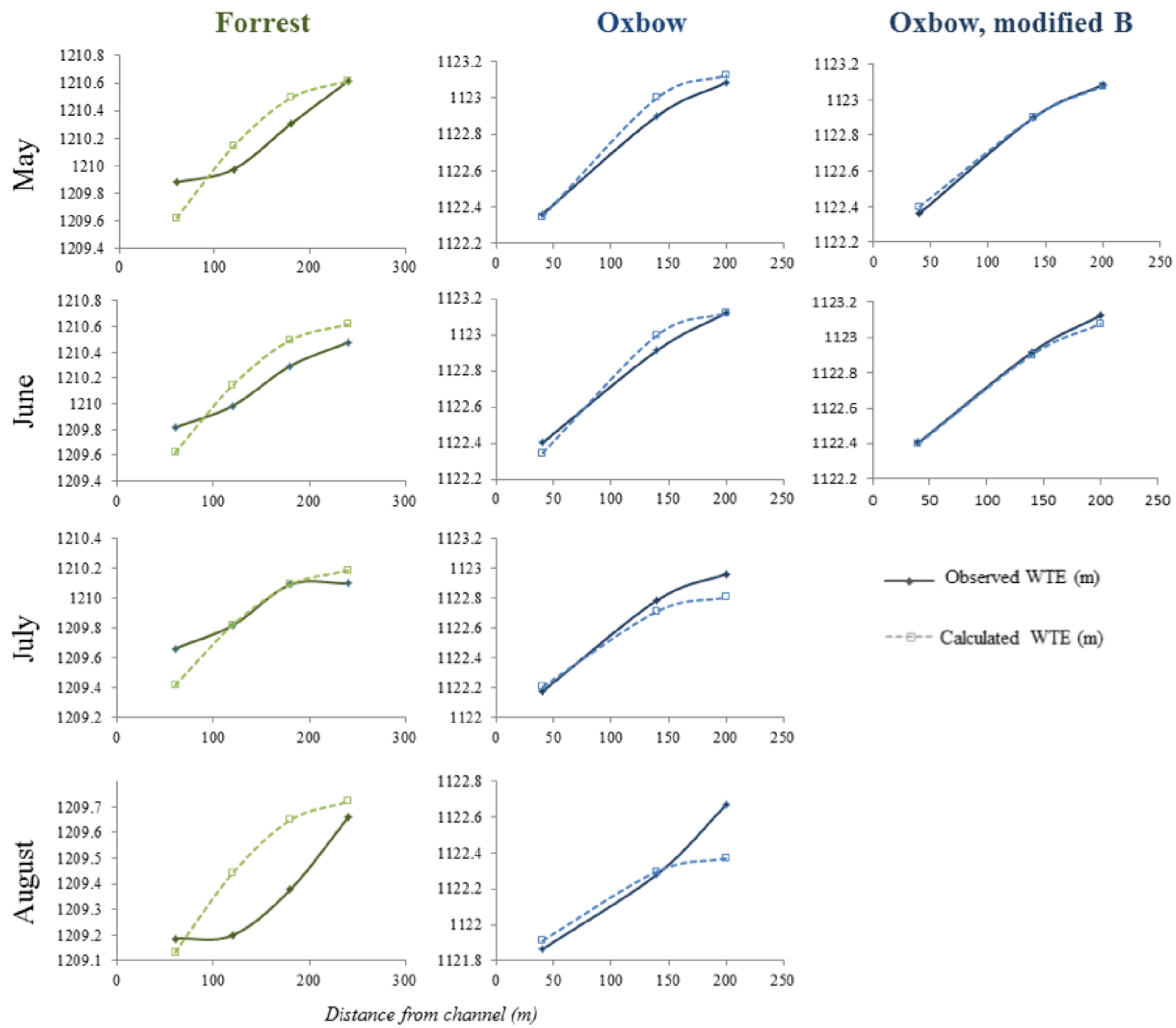


Figure 9. Observed (solid line) and calculated (dashed line) water table surfaces at Forrest and Oxbow sites, May 1 to August 1, 2010. modifying valley width to be slightly larger at the Oxbow site improves the fit of early season calculated WTE surfaces. Agreement between observed and calculated values helps reinforce the strength of model predictions in Figure 7.

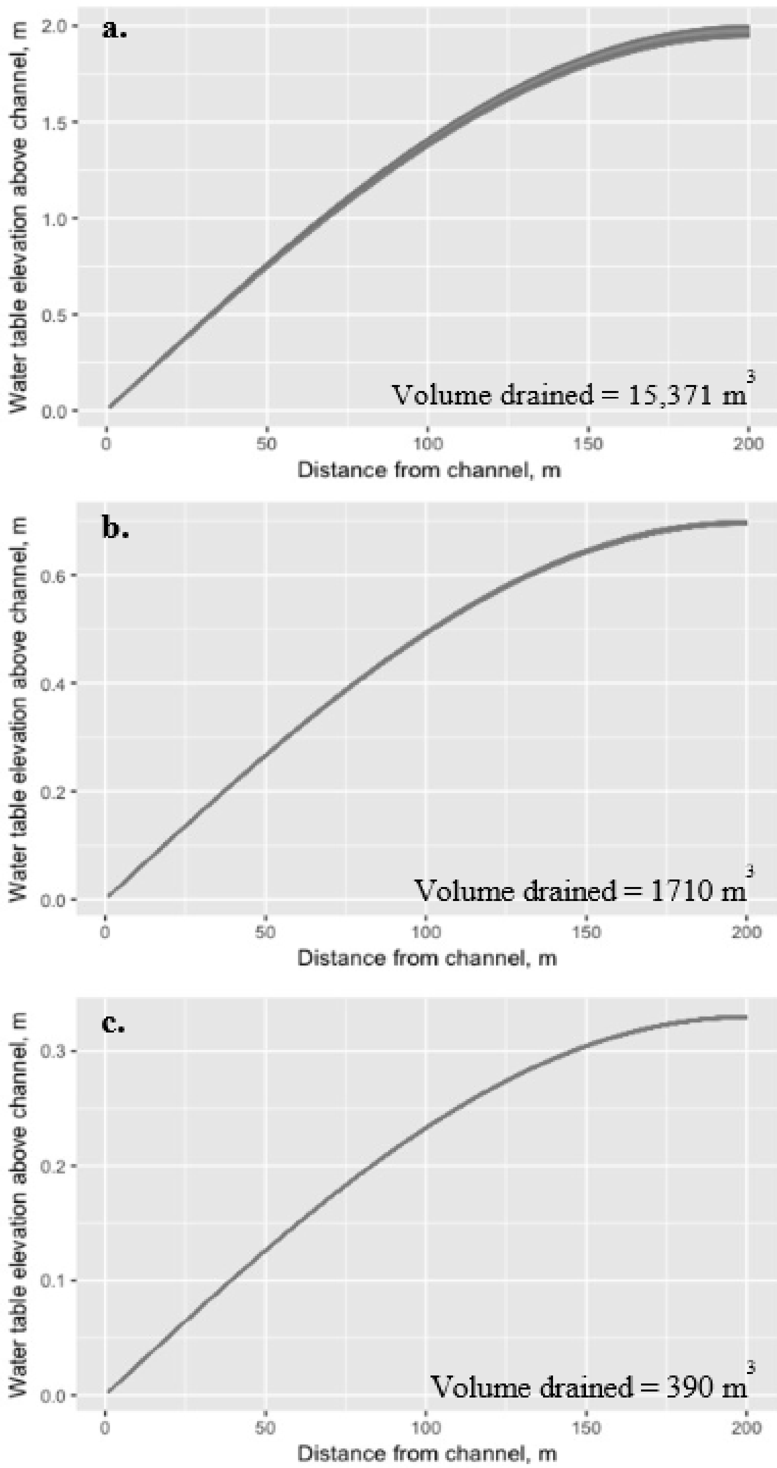


Figure 10. Modeled results of available lateral groundwater drainage for low (3m channel incision, a.), medium (1m channel incision, b.) and high (0.33m channel incision, c.) floodplain connectivity. (Nash et al. 2017). If floodplain restoration actions are implemented the loss of about 15,000 m^3/year/km of stream due to re-connecting the stream to its floodplain is predicted to have less than 0.01 l/s/km impact on stream flow, which would not be of significance to stream temperature regulation.

Discussion

The results of this long-term study on the temperature effects of restoration actions show that restoration can both increase and decrease MFJDR stream temperatures. Many factors can limit salmonid productivity including access to floodplains, sinuosity, and channel complexity. However, these restoration goals must be considered together with their potential to increase stream temperatures if increases in effective stream area are proposed (Figure 11). For example, creating alcove and floodplain habitat, known to benefit juvenile salmonids, may enhance productivity and offset slight negative impacts to salmonids from small increases in temperature. However, it is also possible that if restoration actions increase exposures to temperatures exceeding 18°C, these actions may not achieve desired outcomes, and may even have deleterious consequences. In summary, there is a balance between restoration goals, and future restoration efforts should include temperature analyses in their restoration impacts assessments in order to maximize the benefit to salmonid species.

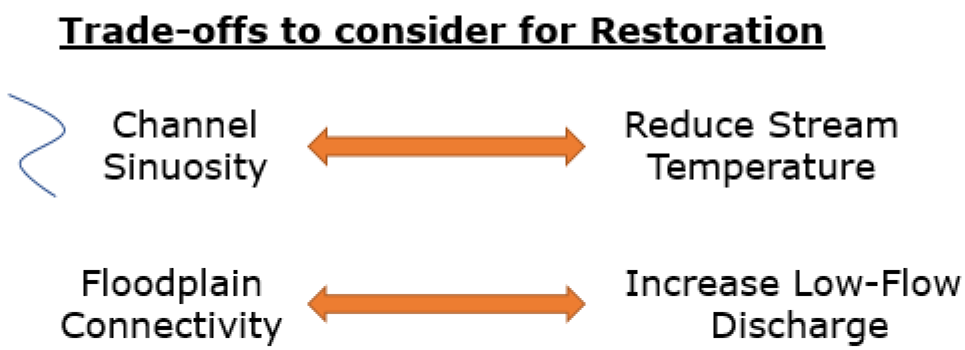


Figure 11. Trade-offs that are important considerations for future restoration planning. Increased sinuosity may improve habitat, but may also lead to greater surface area of water and lower water velocity, leading to increased stream temperature (top arrow); while improved floodplain connectivity may improve winter juvenile survival by giving refuge from being washed downstream during main-stem floods, but might lead to lower later-summer flows due to increased evaporative demand of floodplain plants with roots that can reach the water table when the summer stream water level is increased.

Potential changes in transpiration

Our considerations of the stream flow did not consider the potential impacts of changes in plant transpiration from restored areas. As floodplains are re-connected with streams, the riparian and valley-bottom vegetation generally has greater access to water, which should be expected to increase water consumption, potentially reducing streamflow. It may be wondered what the impact of changes in riparian vegetation might have on stream flow due to plant-water demand. Here we will only consider the band of vegetation immediately adjacent to the river.

It is widely recognized that local bands of well-watered vegetation in otherwise arid areas can have transpiration demands far in excess of those seen in expansive crops. This is known as the “oasis” or “clothesline” effect (Allen et al. FAO, 1998), and is accounted for by application of a “crop coefficient” that expresses the relative water use compared to a well-watered, extensive field of alfalfa. Employing the equation provided by the FAO, the predicted K_c (crop coefficient) of 2.5 is found for a well-watered 20m tall, 15m wide cottonwood stand. For a well-watered reference crop (e.g., alfalfa), the expected transpiration in a sunny area such as found on the MFJDR would be about 1l/s/ha on a 24-hour basis. This then suggests that the rate of water consumption by the riparian buffer is expected to be about 2.5l/s/ha. With this, taking the riparian zone to be about 15m width on each side of the stream, the expected water consumption per river km can be calculated. Assuming a sinuosity of 1.5, there would be 1.5 km of riparian vegetation 30 m wide per km in the direction of river flow, which results in 45,000 m^2 of riparian vegetation per km, or 4.5 ha/km. Thus, the transpiration demand would be expected to be 4.5 ha/km x 2.5 l/s/ha = 11.25 l/s/km (0.64 cfs/mile). Along a 10-mile section of the MFJD with complete coverage of riparian vegetation, this would be 6.4 cfs of consumption. Note that if the riparian vegetation was only 5 m tall and 10 m wide (more typical of woody shrubs), this result would be cut in half. Therefore, 10 miles of woody shrubs should be expected to draw about 3 cfs from the system. Given that 4 of the last 10 years have had low-flows of under 30 cfs at bridge creek (USGS), a loss of 3-6 cfs for 10 river miles of restoration could be of significance under low-flow conditions, representing as much as 20% of the flow.

Future Monitoring

Though this study lasted nearly a decade, the processes and cycles which influence salmonid populations span much longer time scales, and continued monitoring is recommended.

- Additional data is necessary over decadal-scales to accurately assess how changes to vegetative cover (shading) might impact stream temperatures.
- The magnitude and location of cold water inputs into the MFJDR from tributaries and GW upwelling can be leveraged in restoration designs. Targeting cold water inputs locations for habitat improvements (e.g. LWD additions, channel reconfiguration) may have additive, or even multiplicative, effects on salmonid productivity. These strategies can be better understood by continued monitoring of the Oxbow Phase 3, 4 and 5 projects which occurred at the end of the current IMW study.

Key Findings

- Reduction in peak temperatures and dampening of diurnal fluctuations were achieved through the consolidation of two channels into one in the OCA Phase 2 restoration, demonstrating that stream area reduction is a viable means of reducing peak stream temperatures (Hall, 2015).
- OCA Phase 3, 4 and 5 restoration projects, which resulted in an increase in stream area through introduction of meander bends and a slowing of velocities due to increased channel length thus reducing hydraulic gradient. Model results of the unvegetated restored channel indicate an increase in stream temperature at the downstream end of the restored reach, if shade cover becomes established or exposed stream area is less than anticipated a decrease in stream temperature is expected (Hall, 2015).
- DTS was an essential tool for accurate calibration of HeatSource model results due to Phase 2 restoration and for identification of all thermal processes along the mainstem and tributaries, which were then capable of being applied to the Phase 3, 4 and 5 projects (Huff, 2009, Arik, 2011; O'Donnell, 2012; Hall, 2015).
- Understanding that deep, slow-moving sections of stream can exhibit lagged temperature behavior and that temperature patterns can vary temporally is necessary for accurate interpretation of TIR and FLIR river temperature datasets.
- Groundwater and hyporheic contributions were found not to influence the mainstem temperatures (shown through HeatSource modeling and DTS measurements, Hall 2015) or provide detectable cold-water refugia for salmonids (Huff 2009; O'Donnell, 2012).
- Model results provided by groundwater monitoring of the floodplain showed that bank storage along the mainstem river would not positively impact late season flow or temperature, though habitat improvement and winter flood processes may well justify re-connection of streams to their floodplains (Nash et al., 2017).

Recommendations:

- Stream area exposed to sun was found to be the primary factor influencing changes in stream temperature. Along with other habitat improvement goals, it is recommended that restoration incorporate the reduction of exposed stream area to maximize salmonid productivity and restoration effectiveness. It is proposed that future restoration efforts should consider this a critical design criterion.
- Floodplain restoration incentives need to be balanced with expected late summer discharge expectations to not negatively impact stream volume to the extent that a loss of habitat occurs.

Table 1. Summary of Projects, Results and Recommendations

Study Name	***Scale of Study***	***Results***	***Recommendations***	***Appendices***
Hyporheic Influence and Identification in mainstem MFJD	Reach: FCA, OCA	Hyporheic exchange acts as temperature buffer.	DTS with deterministic temperature modeling can be used to predict changes due to hyporheic exchange and inform restoration planning regarding locations of existing flow-paths	A, C
Groundwater Influence and Identification in mainstem MFJD	Reach: FCA, OCA	Groundwater contribution will reduce stream temperature throughout the day.	DTS can be implemented to identify locations and magnitude of groundwater influence.	A, C
Thermal influence on tributaries	Tributary: Big Boulder	Groundwater is primary contributor to tributary temperature.	Tributary restoration to improve groundwater connectivity can assist in reducing tributary temperature	B
Cold Patch Survey	Reach: FCA, OCA focusing on Pools and Refugia: >1m length	Tributaries provide primary source of cooler water to mainstem. Strong thermal mixing in pools.	Focus on tributary health and connection to mainstem can potentially improve cooling mechanism for the mainstem. Additional information is necessary to understand preferential refugia in mixed pools.	C
TIR/FLIR data evaluation	Reach: OCA	Temporal temperature data should complement “snap-shot” TIR/FLIR data.	TIR/FLIR can be useful in evaluating stream temperature at a large spatial scale, but it is important to collect data throughout the day to evaluate the transient nature of the temperature signature.	C
Evaluation of Phase 2 Restoration	Reach: OCA, Phase 2	Phase 2 restoration reduced 7DADM temperature by 0.65°C due to the reduction in overall stream area and consolidation of two channels into one.	Stream area and shade are major contributing factors to stream temperature and need careful consideration when planning restoration. Reducing stream area has a more immediate effect whereas improvements to effective shade should be a long-term goal.	D
Model Prediction of Phase 3, 4 and 5 Restorations	Reach: OCA Phase 3, 4 and 5	Estimated increases in planned restored stream area is predicted to increase overall stream temperature.	Stream area and shade are major contributing factors to stream temperature and need careful consideration when planning restoration. Reducing stream area has a more immediate effect whereas improvements to	D

			effective shade should be a long-term goal.	
Prediction of Floodplain Restoration on Groundwater Contribution	Reach: FCA, OCA	Reconnection of stream and floodplain is predicted to reduce river low flow due to increased transpiration, and increased flood-plain storage.	Floodplain restoration plans need to be balanced with potential decreases in late summer discharge which can result from increased vigor in floodplain vegetation.	E
Evaluation of Riparian Transpiration	Reach: OCA	ET of shade-producing vegetation can consume on the order of 0.6 CFS per mile, which should be factored into projections of future river dynamics.	ET for the restored system should be analyzed based on the changes in the riparian system. Greater shade requires larger plants, which consume water.	F

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Appendices

Appendix A.

Huff, J.A. 2009. Monitoring River Restoration using Fiber Optic Temperature Measurements in a Modeling Framework. M.S. Thesis. Oregon State University, Corvallis, OR. Available at: https://ir.library.oregonstate.edu/xmlui/handle/1957/13847

Appendix B.

Arik, Aida D. 2011. A Study of Stream Temperature Using Distributed Temperature Sensing Fiber Optics Technology in Big Boulder Creek, a Tributary to the Middle Fork John Day River in Eastern Oregon. M.S. Thesis. Oregon State University, Corvallis, OR. Available at: https://ir.library.oregonstate.edu/xmlui/handle/1957/26338

Appendix C.

O'Donnell, Tara. 2012. Evaluation of stream temperature spatial variation using distributed temperature sensing. M.S. Thesis. Oregon State University, Corvallis, OR. Available at: https://ir.library.oregonstate.edu/xmlui/handle/1957/28627

Appendix D.

Hall, Austin. 2015. Drop it like it's Hot: Combining DTS and Temperature Modeling to Evaluate Stream Restoration on the Middle Fork of the John Day River. M.S. Thesis. Oregon State University, Corvallis, OR. Available at: https://ir.library.oregonstate.edu/xmlui/handle/1957/56455

Appendix E.

A physical framework for evaluating net effects of wet meadow restoration on late summer streamflow

Caroline S. Nash, John Selker, Gordon E. Grant, Sarah L. Lewis, Paul Noël

Key words: stream restoration, wet meadow, late summer streamflow, hydrology, incised channels

Abstract

Restoration of degraded wet meadows found on upland valley floors has been proposed to achieve a range of ecological benefits, including augmenting late-season streamflow. There are, however, few field and modelling studies on hydrologic changes following restoration that can be used to validate this expectation, while published changes in groundwater levels and streamflow following restoration are inconclusive. Here, we assess the streamflow benefit that can be obtained by wet meadow restoration. The physically-based quantitative analysis employs a one-dimensional linearized Boussinesq equation with a superimposed solution for changes in storage due to groundwater upwelling and evapotranspiration, calculated explicitly using the White Method. The model and assumptions gave rise to predictions in good agreement with field data from the Middle Fork John Day watershed in Oregon, USA. While raising channel beds can increase total water storage via increases in water table elevation in upland valley bottoms, the contributions of both lateral and longitudinal drainage from restored floodplains to late summer streamflow were found to be undetectably small, while losses in streamflow due to greater transpiration, lower hydraulic gradients, and less drainable pore volume were likely to be substantial. Although late-summer streamflow increases should not be expected as a direct result of wet meadow restoration, these approaches offer benefits for improving the quality and health of riparian and meadow vegetation that would warrant considering such measures, likely to be at the cost of increased water demand and reduced streamflow.

1.0 Introduction and Background

Late summer streamflow in the Western U.S. is critically important for environmental, economic, and recreational demands (NRC, 2002; Poff et al., 2003)). In this region, snowpack can be the largest component of water storage, holding winter precipitation for release as streamflow during dry summer months (Mote et al., 2005). Demonstrated declines in snowpack (Nolin and Daly, 2006; Payne et al., 2004; Safeeq et al., 2013; Safeeq et al., 2015) have been linked to changes in the timing and overall reductions in summer streamflow (Cayan et al., 2001; Knowles and Cayan, 2002; Safeeq et al., 2013; Seager and Vecchi, 2010; Stewart et al., 2004; Tague and Grant, 2009). This has prompted broader consideration of other potential sources of water storage (Barnett et al., 2005; Barnett et al., 2008; Palmer et al., 2009; Poff, 2002). Restoring incised and often ephemeral stream channels back to wet meadows – valley spanning, seasonally inundated wetlands - in montane environments has been proposed as one such source of late season water (Lindquist and Wilcox, 2000; NFWF, 2010; Podolak et al., 2015; USDA, 2013).

Strategies to accomplish this restoration vary widely, but can be broadly classified between two end members: (1) the entire incised channel is filled with sediment and other material and a new, smaller and typically more sinuous channel is constructed on the adjacent valley floor (e.g., plug and pond restoration)(Henery et al., 2011; Lindquist and Wilcox, 2000; Ramstead et al., 2012; Readle, 2014); and (2) the incised channel is partially or completely dammed with multiple low-head structures made of various materials (e.g. willow, logs, rock) as a means of holding back water within the existing incised channel (Abbe and Brooks, 2013; Beechie et al., 2010; Harvey and Watson, 1986; Pollock et al., 2014; Roni et al., 2008; Shields et al., 1995). The latter end member includes structures that mimic or derive inspiration from beaver dams. The theory behind all strategies is that by raising the channel, and thus local water table elevations, the increased amount of water stored in adjacent floodplain aquifers will slowly release through the summer, augmenting late summer streamflow (Liang et al., 2007; Loheide and Gorelick, 2007; NFWF, 2010; Podolak et al., 2015; Tague et al., 2008). We refer to these as “wet meadow” restoration methods.

But *does* wet meadow restoration increase water storage and late summer streamflow? The limited suite of modeling and field studies are inconclusive in their findings on both the magnitude and direction change in streamflow, groundwater, and surface storage following restoration (Hammersmark et al., 2008; Heede, 1979; Klein et al., 2007; Liang et al., 2007; Loheide and Gorelick, 2007; Ponce and Lindquist, 1990; Swanson et al., 1987; Tague et al., 2008). These inconsistencies arise, in part, from differences in the types of data collected, the spatio-temporal resolution of measurements, and site-specific climatic, geologic, and geomorphic controls (Ramstead et al., 2012). The limited number of datasets documenting changes following restoration further constrains definitive conclusions as to whether this is a viable strategy for increasing summer streamflows. Understanding the underlying mechanisms driving water storage and release in both incised and restored valley bottoms is necessary if these projects are to be considered useful adaptation options in a water-challenged world.

Prior work provides a foundation for a general framework for understanding the hydrologic changes following restoration, but has not produced a practicable quantitative formulation of the problem, limiting the ability to assess the underlying opportunity, or extrapolate from the specific sites studied. To directly evaluate pre- and post- restoration conditions under the same range of conditions, Hammersmark et al. (2008) used a fully distributed, physically based model using finite differencing to solve the three-dimensional Boussinesq equation for saturated flow, the one-dimensional Richards' equation for unsaturated flow, and the St. Venant equations for channel flow. Loheide and Gorelick (2007) addressed the effect of varying initial conditions on net change to the magnitude and direction of groundwater using finite differencing to solve the three-dimensional Richards' equation for variably saturated transient groundwater flow. To disentangle the effects of inter-annual climate variability from system response, Tague et al. (2008) used a non-parametric statistical model on paired gauge data for a wet meadow site prior to and following restoration of incised channels.

Though the modelling strategies and field data used in these studies vary, they are fundamentally "grey box" modeling approaches, where a correlationally-fit model of a meadow system is calibrated to replicate historical behaviors (Brutsaert, 2005). The downside of this approach is that critical geographic nuances that may exert control over the results remain undescribed within the grey box, and thus these approaches are generally unsuited to quantitative prediction of the impacts of restoration elsewhere. To design restoration without the costly development of site-specific numerical simulation (and extensive associated site characterization) requires a general, quantitative, and physically based framework to evaluate existing data and build a rigorous understanding of the fundamental mechanisms driving the magnitude, direction, and timing of changes to hydrologic fluxes as a consequence of restoration.

1.1 Objectives

The objective of this paper is to develop a generally-applicable strategy to evaluate whether and how much streamflow is generated by the restoration of incised channels to wet meadows. To understand the upper bounds on potential streamflow contributions due to restoration, we develop a parsimonious model of water storage in stream banks that captures the effects of restoration on late summer stream flow and can represent the significant geographic peculiarities of specific sites using readily available data.

2.0 Methods

To evaluate wet meadow restoration's influence on late-summer streamflow, we employ a water budget framework to identify fluxes directly responsible for changes to water output following restoration. We go on to develop a physically based model of those fluxes, and incorporate hillslope groundwater inputs and evapotranspirative use validated via comparison to the White method (Lautz, 2007; Loheide and Gorelick, 2005; White, 1932). We then briefly discuss the selection of parameters to maximize potential outputs within the constraints of realistic meadow conditions.

2.1 A simple model of water fluxes and storages in wet meadow systems

A primary bound on changes to water storage, and therefore late season streamflow in restored meadows, is at the scale of the catchment. The term "meadow" requires definition here, as it is broadly used to describe a range of ecological or geomorphic features. We therefore refer to the nearly flat surfaces extending between two adjacent hillslopes located in the headwater reaches of mountain catchments as *upland valley floors*. Geomorphically, upland valley floors include the channel, neighboring alluvial surfaces (floodplains and terraces), and the sub-surface fill of the valley bottom extending vertically to unweathered bedrock (Figure 1). This valley fill creates an unconfined alluvial aquifer, where shallow sub-surface flow is tightly coupled with surface flow. We use the term "*meadow*" to refer to the vegetated surface of the upland valley floor and any associated channel-lining wetlands, and "*wet meadow restoration"* to refer to bringing the stream surface to an elevation where the meadow area will experience standing water connected to the stream and thus facilitate an ecological shift from xeric to mesic species.

To determine the potential contribution of meadow restoration to streamflow, we first consider the water budget for an un-restored upland valley floor (superscript *U* specifies an un-restored valley floor; subscripts specify direction of flow in or out of the valley floor system):

$$(1)\ Q_{S,out}^{U} = \left(Q_{S,in}^{U} + GW_{in}^{U} + P * A\right) - \left(GW_{out}^{U} + ET^{U} * A\right) - \Delta S^{U}$$

where Q_S represents surface water (channelized and un-channelized) [L^3/t], *GW* [L^3/t] represents the vertical component of groundwater fluxes through confined and unconfined aquifers, *P* [L/t] is precipitation, *A* [L^2] is valley floor area, *ET (>0)* [L/t] is evapotranspiration from the valley floor vegetation and open water, and *ΔS* [L^3/t] reflects changes in valley water storage. When *ΔS* is positive, the valley is filling with water; when *ΔS* is negative, previously stored volumes of water are diminishing. Streamflow is highest when inputs exceed outputs and the storage has been filled to capacity (*+ΔS)* (e.g. early spring melt in a snow dominated climate). As precipitation dwindles and evapotranspirative use rises, outputs may exceed inputs, causing storage to reverse its sign (*-ΔS)*. The reducing stores of water may be used to support increased evapotranspirative demand or may drain into an open channel as streamflow.

In the context of meadow restoration, it is useful to split groundwater into three component fluxes: water in unconfined, alluvial aquifers ($GW_{VF,}$ [L^3/t]); water in shallow soil and regolith on hillslopes (GW_{SS}, [L^3/t]); and water contained in deeper, confined aquifers (GW_C, [L^3/t]). For the purposes of this analysis of generic streamside meadows, we take there to be no alteration of discharge from or recharge to the confined aquifer in the upland valley due to restoration, and thus will not be further considered in this work. Henceforth, the term groundwater will only refer to sub-surface water either in hillslopes or the valley fill within the depth influenced by the restoration effort.

Surface and groundwater are tightly linked in upland valley systems, and both are of interest for downstream users. It is therefore useful to combine valley surface and groundwater outputs into a single output term: valley discharge ($Q_{VF,out}$)

$$(2)\ Q^{U}_{VF,out} = Q^{U}_{S,out} + GW^{U}_{VF,out} = \left(Q^{U}_{S,in} + GW^{U}_{SS,in} + P * A\right) - (ET^{U} * A) - \Delta S^{U}$$

While the combined term effectively represents all water outputs available for downstream water users, it is important to note that groundwater leaving the restored valley may not be accessible downstream without mechanized extraction.

The water budget for a restored meadow follows similarly, with superscript *R* specifying a restored valley floor:

$$(3)\ Q^{R}_{VF,out} = Q^{R}_{S,out} + GW^{R}_{VF,out} = \left(Q^{R}_{S,in} + GW^{R}_{SS,in} + P * A\right) - (ET^{R} * A) - \Delta S^{R}$$

The difference in outflowing valley water between the restored and un-restored upland valley floor can therefore be represented as the difference between these two budgets.

$$(4)\ \Delta Q_{VF,out} = Q^{R}_{VF,out} - Q^{U}_{VF,out}$$

$$= \left(Q^{R}_{inputs} - ET^{R} * A - \Delta S^{R}\right) - \left(Q^{U}_{inputs} - ET^{U} * A - \Delta S^{U}\right)$$

$$= \left(Q^{R}_{inputs} - Q^{U}_{inputs}\right) + A(ET^{U} - ET^{R}) + (\Delta S^{U} - \Delta S^{R})$$

Meadow restoration will not impact the magnitude of water inputs, so we set the difference between inputs to zero, reducing this relationship to:

$$(5)\ \Delta Q_{VF,out} = A(ET^{U} - ET^{R}) + (\Delta S^{U} - \Delta S^{R})$$

It is apparent that any net positive changes in summer-time outgoing valley water due to restoration would be due to reductions in evapotranspirative use or increases in delivery from valley floor storage to the stream. As meadow restoration often aims to increase vegetative vigor (and, by proxy, evapotranspiration), it is expected that evapotranspiration will increase following restoration, which would tend to decrease streamflow.

We therefore focus the remaining modelling efforts on summer-time valley floor drainage, to which any increased streamflow might be attributed.

2.2 A quantitative model for valley floor contributions to streamflow

To determine the streamflow that can be attributed to changes in valley storage as a result of meadow restoration, we developed a model to calculate: (1) the maximum storage physically available in a given valley, (2) the fraction of that storage available for gravity-driven drainage to a channel, and (3) the temporal pattern of drainage, based on the Boussinesq equation, to calculate volume and timing of discharge to the channel. These drainage results are put into a hydrological context through comparison to estimates of subsurface (GW_{SS}) contributions and evapotranspirative (ET) losses.

The total volume of water that can be stored in a valley is a function of its dimensions and the porosity of its fill material; the volume actually stored (V_{max})is a function of the highest annual elevation the water table reaches (WTE_{max}). This can be represented as:

$$(6)\ V_{max} = 2BLn(D_z - WTE_{max})$$

where B is floodplain width on one side of the stream [L], Dz is depth to bedrock [L], WTE_{max} is the distance beneath the valley surface when the water table reaches its annual maximum elevation [L], L is meadow length [L], and n is porosity [L^3 /L^3]; the dimensions are multiplied by two to account for volume changes on both sides of the stream (Figure 2). As the water table drops to its minimum annual elevation (WTE_{min}), only a fraction of the total stored water will drain – the rest will be held in pores by capillarity and other physical forces. We must then define the drainable porosity, φ, as the volume of water that will drain from an area per unit drop in WTE (Bear, 1972; Brutsaert, 2005). Integrating this over the valley area gives our drainable storage (volumetric yield) (V_W):

$$(7)\ V_W = V_{max}\frac{\varphi}{n} = 2BL\varphi(D_z - WTE_{max})$$

As water can only drain into an open stream via gravity, we note that only water held above the channel surface elevation can drain laterally into the channel, or: that the water in a valley that drains into the local channel is coming from the two blocks of fill immediately adjacent to the channel (Figure 2a). The remaining water is available for longitudinal drainage, and is assessed separately below. The amount of drainable storage available for lateral drainage (V_{lat}) is:

$$(8)\ V_{lat} = 2BL\varphi(D_i - WTE_{max})$$

D_i represents the depth of incision [L].

The total volume that will drain over some time period of interest (t_c) can be well approximated using the conservation of mass and Darcy's law under the Dupuit hydraulic flow assumptions embodied in the Boussinesq equation for the transient drainage of an initially saturated unconfined aquifer to a fully penetrating channel. In this well-established approach (e.g. Brutsaert and Nieber, 1977; Rupp et al., 2004) the changing position of the water table a distance x from the stream in the adjacent floodplain is represented by:

$$(9)\ \frac{\partial h}{\partial t} = \frac{K}{\varphi}\frac{\partial}{\partial x}\left(h\frac{\partial h}{\partial x}\right)$$

where h is the elevation of the water table [L], t is the time since the start of recession [t, days here], and K is hydraulic conductivity [L/t]. Homogenous valley fill extends from an impermeable bedrock layer at z=0 to the valley floor surface at $z=D_z$ (Figure 2). This can be analytically solved either for an early-time solution, where drainage occurs only from fill adjacent to the channel and has not yet reached the valley floor edge (x = B =∞, t_1 in Figure 2), or for late-time, when the water table is dropping all the way to the valley floor edge (x=B) (Rupp and Selker, 2005). Since we are interested in late summer streamflow following spring snowmelt (t_c ~ 100 days), the late-time solution represents the relevant state of the boundary condition. To facilitate the development of an analytical solution while maintaining excellent accuracy through the bulk of the drainage process, we employ the standard linearization of the equation by approximating the thickness through which the water flows as the average water table depth (h =D/2 in EQN 10; Brutsaert, 2005). Rearranging terms with the linearized water table depth produces:

$$(10)\qquad \frac{\partial h}{\partial t} = \frac{KD_i}{2\varphi}\frac{\partial^2 h}{\partial x^2}$$

subject to the boundary and initial conditions:

$$h(x=0,t)=0$$
$$\frac{\partial h}{\partial x}(x=B,t)=0$$
$$h(x=B,t=0)=D$$

Integrating EQN 10 to obtain a solution that satisfies these boundary and initial conditions gives:

$$(11)\qquad h(x,t) = D_i sin\left(\frac{\pi}{2B}x\right)e^{-\frac{KD_i\pi^2}{8\varphi B^2}t}$$

which indicates that the water table maintains a constant sinusoidal shape that decreases in amplitude exponentially with time since start of drainage. If the draining floodplains are not initially saturated, D_i must be adjusted to reflect the expected maximum elevation of the water table above the channel bed (e.g. in an incised channel 3 m deep with a water table 1 m beneath the valley surface, D_i = 2 m).

We can employ the solution given in EQN 11 across the valley width for each time step, and integrate the differences between WTE position at t_1 and t_c to give an improved approximation of the laterally drainable subsurface storage (V^*_{lat}) over time of interest, t_c:

$$V^*_{lat} = 2\varphi L \int_0^B [h(x,t_1) - h(x,t_c)]dx \tag{12}$$

This formulation assumes the only system output is valley fill drainage to a channel and no inputs. Valley floors are, however, typically losing water to evapotranspiration or receiving groundwater from the surrounding landscape (upstream valleys and hillslopes, ref Figure 1). Including the effects of a net loss or gain due to groundwater upwelling and evapotranspirative consumption are essential in estimations of water table position and the water budget. The linearization of Boussinesq (*h = D/2* in EQN 10) is central to this, as it allows us to superimpose an additional solution for a system experiencing changes in storage. Looking to the non-streamflow related inputs and outputs, we compute the change in storage as:

$$\varphi \frac{dh}{dt} = GW_{SS} - ET \tag{13}$$

where GW represents the groundwater inflow (m/day) from surrounding hillslopes and ET the rate of evapotranspiration (m/day). These seasonally averaged values can be calculated by closely analyzing the diurnal fluctuations in WTE. Specifically, as suggested by White (1932), we attribute gains in the elevation of the water table in the night as evidence of groundwater upwelling, and lowering of the water table during the day as indicating plant and evaporative consumption in excess of groundwater contribution. Per the formulation of White's method presented by Lautz (2007):

$$ET_G = \varphi(24 r_{gw} \pm s) \tag{14}$$

$$GW_{SS} = 24\varphi r_{gw} \tag{15}$$

where r_{gw} is the rate of water-table rise between 0:00 and 4:00 (m/hr) and *s* is the net rise or fall of water-table during a 24-h period (m) (Lautz, 2007). This approach approximates groundwater upwelling as constant, that lowering WTE is due to ET, and ET and soil-water hysteresis is small in the pre-dawn period (Loheide and Gorelick, 2005; White, 1932). More specific discussion of the conditions under which diel fluctuations occur can be found in Loheide and Gorelick (2008).

Solving EQN. 13 gives the net rate of change in the overall water table. Substituting the values from EQN. 14 and EQN. 15 provides the contribution to the water table height due to groundwater and ET processes on a daily timescale via the formulation:

$$h(t) = \frac{GW_{SS} - ET_G}{\varphi} t = \pm st \quad (16)$$

This solution can be superimposed onto EQN 11 to model the time evolution of the water-table height to obtain an overall model for the meadow water table dynamics:

$$h(x,t) = D_i sin\left(\frac{\pi}{2B} x\right) e^{-\frac{KD\pi^2}{8\varphi B^2} t} + \frac{GW_{SS} - ET_G}{\varphi} t \quad (17)$$

The addition of the second term refines our estimate of the changing position of the water table and thus the total volume of water drained from t_1 to t_c. To explicitly calculate drainage rate on a given date, we multiply by the drainable volume of water per change in water table height, integrate EQN 17 spatially over the meadow area, and take the time-derivative to obtain the predicted rate of change of stored water. will need to model the effect of changing WTE on discharge to produce a time-variable drainage rate. This results in an exponential decay function:

$$Q(t) = V_{lat}^* \alpha e^{-\alpha t} \quad (18)$$

where V_{lat}^* is total lateral drainage, calculated in EQN 12, and α is a decay constant that describes the physical properties of the system. This is a widely employed result, as discussed, for instance, by Rupp and Selker (2005) and Brutsaert (2005). Discharge values produced by this equation are maximized by setting α equal to one over the square root of t_c, assuming the chosen valley has optimal physical conditions for producing streamflow at t_c.

Having established the lateral drainage volumes and rates, we next address the question of longitudinal drainage through the valley floor. Realistically, drainage will occur in three dimensions, the dominant vector of which will be along the maximum hydraulic gradient. This is a function of down-valley and cross-valley gradient, soil media, and the seasonally variable position of the water table relative to the water surface elevation in the channel (Barry et al., 1993; Loheide and Gorelick, 2007; Richards, 1931). To estimate the relative magnitudes and contributions of these terms, we evaluate the upper bounds on expected flow rates, laterally and longitudinally, to estimate relative contributions in either direction. We assume all water is draining at a representative maximum hydraulic gradient towards a final position at the base of the channel, at the downstream end of the valley, (x, y, z) = (0, 0, 0) (Figure 2a). For lateral drainage, we assume an initial position at (B/2, 0, D); for longitudinal drainage, we assume initial position at (B/2, L, D) (Figure 2a). These initial positions represent the maximum width averaged flow possible, which maximize estimations within the bounds of realistic conditions. We assume flow across the maximum available cross-sectional area: for lateral flow, both banks along the entire reach; for longitudinal flow, the cross-section of the upland valley. Reformulating Darcy's law to include the geometric values for these two scenarios, we predict maximum flow rates as:

$$Q_{lat} = 4KL\frac{D^2}{B} \quad (19)$$

$$Q_{long} = 2KBD\frac{LS+D}{\sqrt{\frac{B^2}{4}+L^2}} \quad (20)$$

Assuming equal saturated hydraulic conductivity and valley dimensions, we solve for the dominant direction of flow in any given scenario. Comparing the solution to EQN 19 with the t_1 solution for EQN 18 provides a scale by which to evaluate the solution to EQN 20 in the context of expected discharge. This gives a first-order estimate of how changing the channel depth through restoration will affect dominant flow paths and places upper bounds on expected streamflow contributions. Secondarily, these formulations – being entirely geometric – allow us to compare geomorphic conditions and thresholds in upland valleys that influence the dominant flow path.

2.4 Model parameterization for streamflow optimization

Selecting model parameters requires balancing the goal of optimizing conditions for the production of late season water and accurately representing the typical conditions encountered in upland valley floors. We recognize that there is considerable variation in upland valley systems, and have adopted parameters that provide an upper limit on the potential changes in storage and contributions to flow based on our geologic understanding of hydraulic parameters in these systems; our calculations are therefore biased in favor of the largest potential hydrologic impact of restoration.

Input values for valley dimensions and gradient were selected based on a number of field sites in Eastern Oregon, including those used for model validation, below (Table 1, Figure 2). Soil media was selected from the range of typical soils found in upland valley fill, both from field investigations and literature reported values (Birkeland and Janda, 1971; Koehler and Anderson, 1994; Walter and Merritts, 2008; Wood, 1975). Silt-loam was specifically selected for balancing the trade-offs between high porosity and high hydraulic conductivity. The values for those properties listed in Table 1 were selected from the upper end of associated ranges.

To assess the sensitivity of flow to depth of incision and the magnitude of change in flow based on initial conditions, we model three scenarios: a channel that has incised 3-meters (**IC-3**), a channel that has incised 1 meter (**IC-1**) and an un-incised channel 0.33 m deep (**UC**). These are common depths of incision in upland valley floors, though certainly small compared to some extreme examples (>20 m)(Antevs, 1952; Bull, 1997; Harvey and Watson, 1986; Peacock, 1994; Simon et al., 2000). We exclude those large depths as they would be unlikely candidates for the styles of restoration being examined here.

The maximum elevation that the water table will reach annually in the valley floor (WTE_{max}) is set to scale with depth of incision based on an upper bounds of observed changes in water table following restoration. The model presented here is parameterized so the maximum water table elevation (WTE_{max}) lies 1 m below the valley floor IC3, 0.3 m below the valley floor in IC1, and that the valley floor is completely saturated (WTE_{max} = 0) in UC. Tague et al. (2008) reported pre-restoration (incised-channel) WTE_{max} levels of 1.1 to 1.4 m, and post restoration (un-incised) WTE_{max} levels of 0.8 to 1.0 m; a change of 0.3 – 0.4 m. Hammersmark et al. (2008) observed winter and spring WTE_{max} levels increased 0.72 m and 1.2 m, respectively, following restoration. Klein et al. (2007) reported no statistically significant changes following restoration. Loheide and Gorelick (2007) observed that WTE_{max} in both restored and un-incised meadows was more than 1m nearer the surface than the WTE_{max} in a separate, incised meadow, demonstrating that water table elevations can be considerably different for channels with similar depths of incision. Their modeled hydrographs show a difference in WTE_{max} between restored and deeply (4m) incised to be nearer to 0.75 m. Our assigning a value of one meter to the expected rise in WTE_{max} between IC-3 and UC is thus a reasonable upper bound of expected change to the water table elevation following restoration.

As we are interested in the change to late-summer streamflow, we selected an index date of September 1st on which to calculate changes in streamflow following restoration. Data from piezometers installed along existing, incised (1m) meadows on the Middle Fork John Day show that the water table begins to draw down on June 10th, (t_c = 82 days). Modelled results from Loheide and Gorelick (2007) indicate that a reduction in depth of incision should result in an extended duration of WTE_{max}, as streamflow levels are kept higher in a smaller channel. Per the diminishing exponential relationship between valley floor discharge and elapsed time (EQN 18), a smaller t_c should result in a larger daily streamflow contribution on the index date. We therefore extend t_c to reflect an earlier start of drainage in the deeply incised scenario (t_c = 92 days), and reduce t_c for the restored scenario (t_c = 77 days). The reduced t_c values in the incised and restored scenario increase the potential contributions of bank storage to streamflow on the selected index date.

3.0 Model Validation

We test the validity of assumptions made in our model by comparing calculated water table elevations against measured data from two un-incised floodplain meadows along the Middle Fork John Day River (MFJD), Oregon (Figure 3). We use the dimensions of the floodplain at each site to calculate change in WTE due to drainage and the diurnal signal in two well transects to estimate net effect of groundwater upwelling and evapotranspiration to produce modelled water table surfaces (EQN 17). This gives an estimate of the total volume drained over the summer and temporally explicit drainage rates, hence meadow contribution to streamflow.

The MFJD flows for 120 km, draining 2100 km^2 of the Blue Mountains in Northeastern Oregon, USA. The watershed receives 380 – 640 mm of precipitation annually, the majority of which occurs between October and June as snow. The channel is, on average, 1 m deep and 4 m wide; average streamflow is 7 m^3/s, with peak streamflow of 22.7 m^3/s occurring in mid-spring. Mean daily streamflow at t_0 (June 10) is 12.1 m^3/s and decreases to 0.9 m^3/s by September 1st. The soils in the floodplains are mostly clay loam (Noël, unpublished data) with a drainable porosity of roughly 0.02 (Loheide and Gorelick, 2005). County-wide soil surveys estimate local hydraulic conductivity values ranging from 0.02 to 20 m/day (Dyksterhuis, 1981). Numeric calibrations done to fit data from this field site suggest that 5 m/day is an appropriate magnitude for soils in the area of the well transects (Noël, unpublished data), in keeping with the expected permeability of the stream deposition processes which deposited these sediments. The Forrest transect is comprised of four wells spaced evenly between 50 and 240 m laterally from the channel. The Oxbow transect is comprised of three wells evenly spaced between 50 and 200 m laterally from the channel. Water table elevation was collected continuously at both sites from 2009 – 2010.

The annual hydrograph of each floodplain meadow shows that the water table was maintained within 0.3 to 0.8 m of the surface throughout the year (Figure 4a). Maximum WTE (0.3 m below the surface) occurred in early June, so we set 10 June 2010 as the start of drainage (t_0) which is consistent with values reported elsewhere (Hammersmark et al., 2008; Klein et al., 2007; Loheide and Gorelick, 2007; Tague et al., 2008). Despite several large summer storms (e.g., 26-07-2010 and 11-08-2010), there was no detectable response in WTE during summer drawdown. The model predicted drainage volumes and rates from both sites, without accounting for changes due to groundwater upwelling or evapotranspiration (Table 2).

Fluctuations in water table elevation reveals a regular pattern of diurnal variation indicative of the combined effects of drainage, groundwater upwelling, and evapotranspiration (ET) (Figure 4b). Peak daily WTE occurs around 0700 and lowers to its minimum daily elevation around 1700. The daily decrease can be attributed to evapotranspirative losses exceeding groundwater upwelling.

Using EQN 14 and EQN 15, we are able to estimate daily averages of GW contributions to the WTE and losses due to ET_G. We calculate values for July, when peak ET is expected to occur in meadow systems (Hammersmark et al., 2008) (Table 3). Predicted values for both ET_g and GW are consistent between sites. The values for ET_g are lower than literature reported values for ET in native meadows, which range from 5 to 7 mm/day (Hammersmark et al., 2008; Loheide and Gorelick, 2005). Since potential ET (PET) at this site in July is over 9 mm/d (BATO Station data available at https://www.usbr.gov/pn/agrimet/monthlyet.html), the low rate of consumption by plants reflects the limitations of the combined stored water and groundwater available. Had more water been available, for instance, if a restoration project had elevated the water table, these rates would be expected to increase towards the upper limit of PET.

The volume of water lost daily through evapotranspiration is consistently larger than the net volumetric contribution from groundwater upwelling (31 cm per unit valley width) (Table 3). We tested the validity of using an averaged rate of evapotranspiration by comparing the total observed change in WTE in July 2010 with predicted change using the average value. Between July 1 and 31, the WTE in Forrest-19 dropped 72 cm. Predicted change in WTE using the averaged ET rate was 77 cm, an error of 5 cm (7%). The uncertainty associated with other parameters used in EQN 11 (e.g. hydraulic conductivity and drainable porosity) can often range over two orders of magnitude for a given site, and thus the much smaller error associated with using averaged ET is considered acceptable for this application.

To assess temporal variation in prediction error, we modelled the water table elevations for both sites on the first day of each month from May through August using parameters given in Table 2. The average error between modeled and observed water table surfaces on all dates was 5.0 cm for Forrest and 4.9 cm for Oxbow (Table 4) – a 5.2% and 5.4% error from mean depth to water table over the period of interest, respectively. The average monthly error was smallest in May (3.3 cm) increasing to maximum error in August (7.4 cm), with errors of 5.5 and 6.1% from mean depth to water table at each site, respectively. The observed results at Oxbow closely fit both to the shape and magnitude of change predicted by the model (Figure 7). The congruence between observed and predicted curves is less as the Forrest site, but the model still predicts the overall trend rather well for all months except August. Model fit was improved by extending the floodplain width 50 meters beyond the well furthest from the channel (B=250 m) (Figure 7). With the exception of August at the Forrest site, both observed and predicted results show declining change in WTE over time resulting in a shift in late summer from a concave to a convex curve. In general, observed elevations are lower than calculated values.

The volume of flow draining from the meadow to the river can be computed using this validated model. For the index date of September 1, the drainage from meadow to the stream at Forrest was 6.0×10^{-5} m^3/s per km of stream, and at Oxbow was 3.3 $\times10^{-5}$ m^3/s per km (an increase in stream flow of between 33 and 60 ml/s per kilometer of meadow adjacent to the stream). Given this river had a mean daily flow on September 1 of about 900 l/s, this represents less than 0.01% flow contribution per km, which is about 500 times lower than typical measurement errors for standard stream gauging techniques (e.g. Sauer and Meyer, 1992) suggesting that any flow increases would be undetectable. To put the contributions of lateral inflow to streamflow in context, even if we were to assume this inflow occurred over the entire length of the MFJD River (80% of which does not have adjacent meadows, so this is at least five times what a realistic value would be), the total contribution would be only 7.2×10^{-3} m^3/s or about 0.8% of the mean daily September 1 flow. The maximum longitudinal flow (EQN 20) predicted for this site was an order of magnitude smaller than the maximum lateral flow (EQN 19), indicating that longitudinal flow should account for an even smaller percentage of mean daily September 1 flow. This result is consistent with the fact that the total change in volume of water stored in the meadow is insignificant compared with seasonal stream flow, as we will show in the next section of this paper.

4.0 Results and Discussion

To explore the upper physical bounds on potential streamflow contributions due to restoration, we modelled three incision-depth scenarios (IC-3, IC-1, and UC) using a set of parameters selected to optimize potential streamflow contributions (Table 1). The results from our model and field tests suggest that wet meadow restoration is not likely to result in significant increases in late summer streamflow and may actually decrease flows given the likelihood of increased ET as the water tables in the meadow is raised into the root zone.

4.1 Changes to valley discharge result from a changing evapotranspiration or drained storage

Re-casting a traditional water budget approach to apply to a wet meadow hydro-system gives insight into the magnitude and direction of change necessary to result in increased late season streamflow. Surface (Q_S) and shallow groundwater (GW_{VF}) are tightly linked in upland valley systems, so the outgoing component of each flux were combined into a single term: valley discharge ($Q_{VF,out}$). Solving for the change in this term between restored and unrestored scenarios, it is clearly demonstrated that changes in valley discharge are a function of evapotranspiration (ET) and storage change (ΔS) (EQN 5). The sign of ΔS indicates the direction of change and varies temporally: ΔS is positive when inputs to the system exceed outputs (e.g.

winter) and storage capacity fills; ΔS is negative when outputs exceed inputs (e.g. summer) and storage capacity is drained. The magnitude of ΔS will dictate the total volume of water that either is available to fill or drain; some of the potential drainage may be used for evapotranspirative use.

Thus, to increase late-summer valley discharge following restoration, evapotranspirative use must decrease, or a larger volume of water must drain out of the valley over the same period of time. If the two terms change in opposite directions, the increase in drainage volume must be greater than the increase in evapotranspirative use or, conversely, the decrease in evapotranspirative use must compensate for the reduced drainage volume.

Wet meadow restoration purports to increase storage and then lengthen the time over which that increased storage is drained. For late-summer valley discharge to increase as a result, this increase in storage must translate into an increase in drained storage and any increases in evapotranspirative use must not exceed the increased volume of drained storage.

4.2 Evapotranspirative use increases in restored meadows

Though not directly modelled by this exercise, empirical data and modelled results demonstrate that increasing the elevation of water table into the root zone increases evapotranspirative use both by reducing water stress in existing vegetation and shifting communities towards denser, more mesic species (Hammersmark et al., 2008; Hammersmark et al., 2009; Loheide, 2008; Loheide and Gorelick, 2007). Potential evapotranspiration (PET) represents the maximum rate of evapotranspiration, which occurs when adequate water is present beneath a uniform vegetative cover throughout the growing season (Brutsaert, 1982). Actual evapotranspiration, particularly in arid and semi-arid regions, typically falls far below PET during the summer as available water dwindles. Sustaining the water table within the root zone further into the summer should act to increase ET nearer to PET, reducing the water-stress of the existing vegetation. By this, alone, ET in a restored meadow should be expected to be larger than ET in an unrestored meadow.

However, in some circumstances, raising the water table causes dense, mesic species to replace previously dominant sparse, xeric species on the valley surface. Published data indicate that this shift from xeric to mesic vegetation typically increases evapotranspirative demand by between 1.5 and 3 mm/day (Hammersmark et al., 2008; Loheide and Gorelick, 2007). Over the surface area of the modelled meadow (Table 2), the minimum predicted ET shift would increase water usage by 600 m^3/day throughout the summer, or by 48,000 m^3 over the drainage period of interest. Whether due to reduced water-stress, or new species with higher evapotranspirative usage, restored meadows should be expected to have higher ET than un-restored meadows, making the solution to $(ET^U - ET^R)$ negative.

It appears that the increases in physical storage of water in the restored scenario are used predominantly to support reestablished mesic vegetation. The permanent wilting point for silt loams ranges from 10-15% of the volumetric soil water content (Israelsen and Hansen, 1962; Selker et al., 1999); the plant available water in each scenario is therefore 1,460,000 m^3 for IC3; 1,570,000 m^3 for IC1, and 1,620,000 m^3 for UC. The total increase in water available to plants between the deeply incised and restored scenarios is 160,000 m^3, while the increase in evapotranspirative use is 60,000 m^3. This, at best, leaves an additional 100,000 m^3 available for drainage in the restored scenario, which if spread over several months is inconsequential to streamflow. Drainable storage, however represents a sub-set of plant-available water, as both rely on pores large enough to drain given certain conditions. Plants will compete with gravity for the water stored in the largest, most easily accessible pores, which will more rapidly diminish the newly available drainable storage. This suggests that wetland vegetation will use much of the increased drainable storage early in the growing season, and only begin using water stored in smaller, gravity-inaccessible pores later in the season. This would explain why restored meadows remain green later into the season, even as streamflow diminishes.

4.3 Total storage increases with restoration, but laterally drainable storage decreases.

Restored meadows (UC) store more total water than incised meadows (IC-1 and -3) due to higher water table elevations, but incised meadows have a larger volume of laterally drainable storage by virtue of their larger drainage faces and hydraulic gradients. Total storage increases from 1.6 $\times 10^6$ m^3 in the deeply incised valley (IC-3) to 1.8 $\times 10^6$ in the restored valley (UC) – an 11% increase. The laterally drainable storage in a restored valley (UC) decreases by an order of magnitude over the drainage period of interest – from 14,150 m^3 to 330 m^3, or roughly 98% $^-$ when compared with a deeply incised channel.

The volumes of total storage (V_{max}) and drainable storage (V_W) scale with depth of incision; deeper channels typically have lower maximum water table elevations (WTE_{max}), and therefore store less water. The increase in storage is therefore a function of the imposed values of WTE_{max}, which were selected from literature values to produce a best-case scenario for changes to storage and drainage. For instance, WTE_{max} selected for the restored scenario (UC) was set at 0 m, despite field data typically indicating that maximum water table elevations fall 0.7-1.3 meters below the surface (Klein et al., 2007; Loheide and Gorelick, 2007; Hammersmark, 2008; Tague et al., 2008). This overestimation increases confidence in the trends demonstrated by the analysis, and while the absolute volumes are a function of imposed valley dimensions and soil media properties, the direction of change should hold for any scenario.

Laterally drainable storage (V_{lat}) is, additionally, a function of the position of the channel bed relative to the valley surface. The deeper the incision, the larger the surface area through which water can drain. Similarly, the greater the difference between the valley surface and the channel bed elevations, the larger the hydraulic gradient over which late-season streamflow can drain. This effectively increases the size of the spigot draining valley fill, maximizing drainage rates across all times. Given that restoration increases total drainable storage and decreases laterally drainable storage, it follows that the volume of storage available for longitudinal drainage (V_{long}) must increase in restored channels.

4.4 Streamflow decreases in restored meadows

Increased ET and reduced drainable storage in restored meadows is predicted to lead to an overall decrease in valley discharge ($\Delta Q_{VF,out}$). The volume of water drained in the un-restored scenario (IC-3) is larger than the volume drained in a restored scenario (UC), making the solution to (ΔS^{U}- ΔS^{R}) negative (EQN 5). This result, coupled an increase in evapotranspirative demand following restoration, gives strong evidence that valley discharge should decline following restoration. Though restoration increases the total amount of water stored in meadows, the new storage does not discharge to streamflow. The majority of this new storage is, instead, used to support a higher evapotranspirative demand from previously water-stressed plant communities or denser, mesic plant communities. That said, even if ET were neglected, the available volume of water would be generally insufficient to augment streamflow to the level of detection.

Modelled results demonstrate both the reduction in streamflow and that valley discharge only contributes very small volumes of water to streamflow in any scenario. By using a mass-balance approach, we were able to compute the potential daily drainage from the meadow into the stream over a time period of interest. The rate at which stored water drains laterally to the channel decreases both in early and late time in the restored scenario (Table 5). On the selected index date, September 1st, the model predicts that a kilometer of deeply incised channel (IC-3) will drain 1.2 $x10^{-6}$ m^3/s to streamflow; a kilometer of restored valley (UC) storage will drain 7 $x10^{-8}$ m^3/s. Both values represent less than 0.01% of an average daily flow rate of 0.1 m^3/s. Moreover, the restored valley will contribute two orders of magnitude less streamflow than the incised valley.

On the first day of drainage (t_0 = June 11), one kilometer of deeply incised valley (IC-3) is predicted to contribute 1330 m^3/km to streamflow; one kilometer of restored valley (UC) contributes a total of 33 m^3. This amounts to average flow rates of 1.5 $x10^{-2}$ and 3.8 $x10^{-4}$ m^3/s, respectively, per kilometer of stream. The restored scenario, again, contributes less to the channel due to the reduced bank height and lower hydraulic gradient in the valley fill. This reduction in flow is consistent with modelled results demonstrating that peak flood flows downstream of a restored channel are reduced (Hammersmark et al., 2010), and suggests the mechanism reducing peak flow into the channel is the same mechanism that lowers overall lateral drainage to the channel: reduced channel depth (hence drainable bank height) and decreased hydraulic gradient.

The dominant flow pathway shifts from lateral to longitudinal in restored meadows (UC), but total volumes of drainage in both directions are still lower than those in the incised meaodws (IC-3 and IC-1). Previous modelled results suggest that restoration projects that reduce incision favor longitudinal discharge over lateral (Hammersmark et al., 2008; Loheide and Gorelick, 2007), which can be a favorable restoration outcome. To account for potential streamflow augmentation downstream of the restoration site, we evaluated the upper bound on expected longitudinal contributions to streamflow using a geometric estimation of the maximum longitudinal hydraulic gradient.

In all the modelled scenarios, as well as the field data, the maximum lateral flow rate calculated by this method (EQN 20) is an order of magnitude greater than the peak lateral flow rate calculated by the more explicit linearized Boussinesq equation (EQN 19). This is to be expected, as the large, near-channel lateral gradients accounted for by the Boussinesq equation rapidly give way to lower overall hydraulic gradients.

In the incised scenario, maximum longitudinal flow rate is predicted to be an order of magnitude less than maximum lateral flow rates. Scaling by the lateral discharge calculated at t_c, we expect longitudinal discharge at t_c to be a fraction of a mL per second. In the restored scenario, maximum lateral flow rate decreases by an order of magnitude, which is consistent with the change in discharge at t_c. Longitudinal flow rates do not appreciably decrease, indicating that a larger percentage of water stored in the restored scenario will drain longitudinally, but the total contribution to flow is still negligible.

The scenarios modelled here optimized parameters towards streamflow generation; that the modelled volumes were so small in late summer for all scenarios indicates that drainage from unconfined alluvial aquifers does not constitute a major source of late season streamflow. That the amount of streamflow sourced from these aquifers drops by orders of magnitude with reducing depths of incisions gives confidence that the pattern of reducing streamflow due to meadow restoration is correct.

4.7 Caveats and limitations

The scenarios modelled in this analysis are of a straight channel through homogenous fill that does not cut down to bedrock. One benefit of a physically-based, linearized model is the ease with which additional terms can be incorporated. In so doing, we can demonstrate that increasing sinuosity, adding a hydraulically conductive gravel lens, and cutting a channel down to bedrock do not appreciably increase the contributions of streamflow, and even if they do, the relative contribution still decreases with reduced incision (or, restoration).

Lengthening the restored channel, either by increasing its sinuosity or adding a second braided channel, serves to increase the area from which the valley fill can drain, and should theoretically increase drainage into the channel. If we increase the sinuosity of the restored channel from straight (sinuosity = 1.0) to meandering (s = 1.5) and add a second meandering channel with the same dimensions, we effectively triple the drainage face. This increases the volume of water drained over the day on September 1st to 990 m^3 from 330 m^3. Though this is an overall increase in the contribution from the restored channel, the total volume is still two orders of magnitude less than the volume of water drained from the deeply incised channel (IC-3) – 14,200 m^3.

This conclusion stands even if the analysis is expanded to consider a valley fill with a large, hydraulically conductive gravel lens (cross-sectional area, A = 1 m^2) continuous along the entire reach that acts as a preferential pathway (K = 90 m/day). Under these conditions, the valley could produce an additional 0.92 m^3/day (1.1 x10^{-5} m^3/s of discharge); for a total discharge of 1.23 x10^{-5} m^3/s. This discharge would similarly diminish over the course of the summer, and would likely be utilized by reestablishing species, particularly woody riparian speices (e.g. *Salix sp.*, *Populus sp.)*

The good fit between modelled and measured data also suggests that assumptions made about the depth of fill, impermeability of bedrock, and dominant flow pathways were reasonable for this particular site. These same assumptions may not hold true in areas with highly fractured bedrock and active aquifer recharge/discharge underlying the upland valley floors, and might require that additional processes be represented in a similarly physically-based model.

4.6 Physically-based model results fit well with field data

The good fit between modelled results and field data from the Middle Fork John Day River, OR gives confidence that the formulation of the model captures the main fluxes of waters, and dominant processes governing the drainage of water out of an upland valley floor into a stream.

The model accurately represented the changes in water table elevations in two meadows over our period of interest (May – August). The goodness of fit decreases over the course of the summer, with modelled results under-predicting drainage from early-season floodplains, and over-predicting drainage from late-season floodplains. We remedied especially poor initial model fit by adjusting floodplain width in the model, which indicates that the geometric parameters exert considerable control over dynamics of the actual water table.

The model's overprediction of late-season drainage serves the overall goal of conservatively modelling the maximum expected contributions of valley drainage to streamflow. This trend in error would correspond to the observed streamflows being higher than modelled results suggest in early season, and lower than modelled results in late-season. This overprediction of late season streamflow serves the general purpose of the analysis by erring towards maximum possible streamflow in late-season, and gives an additional level of confidence in the observed trends.

Moreover, the good fit of modeled results with field data bolsters confidence in the general form and structure of the model and also highlights key parameters that need to be carefully measured in the field. Calculated and modelled water table surfaces at two sites in the MFJD matched closely, particularly in early season, confirming the validity of combining the linearized long-time solution to the Boussinesq equation with estimates of evapotranspiration and groundwater upwelling using the White method. utility of a physically-based modelling effort in representing how changing geomorphic features affects hydrology.

4.7 Potential strategies to maximize late summer streamflow

Although our results demonstrate that even under the best of circumstances, increases in streamflow following restoration are unlikely, the model highlights how restoration practices might be implemented if the goal of meadow restoration is actually to increase late-summer streamflow. A restoration program aimed at increasing late summer streamflow must consider both hydraulic and geomorphic issues. Hydraulically, the restoration strategy should maximize laterally drainable storage and minimize critical time (EQN 18). Laterally drainable storage is optimized in long, fully saturated valleys with large depths of incision (effectively, large drainage face surface area) (EQN 12, 19). Critical time is minimized by extending the duration of maximum water table elevation, which can be accomplished by the presence of surface floodplain storage. Practically, this would be accomplished by saturating long reaches of upland valley, adding surface storage, maintaining a large surface area through which to drain the storage, and introducing a time-variable boundary condition at the channel edge, or x = 0 (Figure 2). A time variable boundary condition is created by a lowering water surface elevation in the open channel that can access more laterally drainable storage as the summer progresses.

Geomorphic considerations include the dimensions and fill material of the valley in question. Restoration programs aiming to augment streamflow within the restored reach, per these modelled results, should target long, gently sloping and narrow valleys; restorations aiming to increase downstream flow contribution should target short, steep, wide valleys. In both cases, valley fill with a high drainable porosity and hydraulic conductivity will increase floodplain contributions to channel discharge. If changes to streamflow are the only objective, maximum impact will be felt in places where vegetation is limited, or the vegetation communities will not likely change as a result of the restoration.

In summary, specific strategies for increasing summer streamflow would include:

- Maintain the incised channel bed to maximize area of drainage face and hydraulic gradients. This is effectively what is done when fields and meadows are ditched or drainage tiles are installed;
- Increase WTE_{max} as close to the valley floor surface as possible and ideally above it so as to extend duration of WTE_{max}, minimize t_c and increase the size of the valley bottom wedge draining laterally to the channel;
- Decrease the water level in the channel slowly over time, thereby accessing additional storage that can drain through the bank face. To maximize streamflow contribution, the water level in the channel would be dropped quickly to immediately access early-time bank drainage into the channel;
- Maximize the reach length (L) over which bank drainage is occurring to increase the surface area through which water stored in valley fill can drain. This might mean increasing sinuosity or reoccupying additional channels;
- Reduce or minimize change in evapotranspirative demand by suppressing vegetation.

5.0 Conclusions

Despite the implicit attraction of using wet meadow restoration as a means of augmenting late-season streamflow, a physically-based conceptual groundwater model employing reasonable assumptions of channel and valley hydraulic parameters severely constrains the amount of late-summer streamflow benefit that can be obtained by channel reconstruction style restoration practices. While raising channel beds through restoration can increase total water storage in upland valley bottoms, and extend duration of peak flows by a few days, the contributions of both lateral and longitudinal drainage from floodplains to late summer streamflow are expected to decrease following wet meadow restoration. Good agreement between modeled results and field data gives confidence that the basic model structure and assumptions are valid.

Although regional late-summer streamflow increases should not be expected following wet meadow restoration using meadow restoration techniques, the demonstrated improvements in the quality and health of wetland vegetation could warrant considering such measures, even at the cost of increased evapotranspirative use. Increased ET usage represents a change in either ecosystem type, or a reduction in water-limitation – either of which might represent a valuable restoration objective. It is critical, however, to separate these achievable goals of improving ecosystem health and sustaining wetland ecosystems from the untenable goal of increasing late-season streamflow. Goal setting is a critical component of restoration projects, and the accurate representation of expected changes as a result of restoration is necessary for sustaining public trust and achieving desired outcomes. Without an understanding of the possible range of eco-hydrologic responses to restoration, it is difficult to set reasonable metrics of success. The model presented here, and its initial results, can be used to set more realistic objectives for restoration programs, and help guide future research on how site-specific factors will affect the desired outcomes.

Future research can work to both refine its calibration in a range of other, geologically distinct environments and expand the application of this modelling approach to other restoration styles. For instance, beaver-inspired forms of restoration (e.g. beaver dam analogs, artificial beaver dams, etc.) often has similar goals as wet meadow restoration writ large, and the evaluation of such structures using this modelling framework would more easily allow for a direct comparison of expected results.

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Table 1. Dimensions of idealized upland valley floor

Parameter		**Value**
Floodplain width		200 m
Channel width	*c*	6 m
Depth to bedrock	*z*	10 m
Reach length		1000 m
Volume of meadow fill		4,000,000 m^3
Longitudinal gradient		0.01 m/m
Depth of incision	*i*	Restored: 0.33 m Incised: 1 m Deeply incised: 3 m
Soil media		Silt-loam
Porosity		0.45
Drainable porosity		0.1
Hydraulic conductivity		0.5 m/day

Table 2. Model inputs and results for Forrest and Oxbow sites at Middle Fork John Day, OR. Note that the index date for these model runs was set to August 1, hence the smaller t_c values.

			Forrest	Oxbow
Channel depth	D_i	m	1.6	1.12
Time of interest	t_c	days	51	51
Max WTE, below valley floor	WTE_{max}	m	0.3	0.3
Valley width	B	m	240	200
Drainable porosity	ϕ	m^3/m^3	0.04	0.04
Hydraulic conductivity	K	m/day	5.87	5.45
Laterally drainable storage per km	V_{lat}	m^3	62,400	32,800
Volume drained per km, t_0 to t_c	V_{lat}^*	m^3	4180	2460
Max Lateral discharge	Q_{lat}	m^3/s	2.9×10^{-3}	1.6×10^{-3}
Max Longitudinal discharge	Q_{long}	m^3/s	9.5×10^{-4}	7.0×10^{-4}
Discharge per km, $t = t_c$	Q_{-t_c}	m^3/s	6.0×10^{-5}	3.3×10^{-5}
Discharge per km, at $t = t_1$	Q_{-t_1}	m^3/s	6.6×10^{-2}	3.6×10^{-2}
Total discharge per km, $t = t_c$	$Q_{tot}-t_c$	m^3/d	5.2	2.8
Total discharge per km, $t = t_1$	$Q_{tot}-t_1$	m^3/d	5700	3100

Table 3. Calculated ER_g and GW rates for all wells at both sites for July 2010.

	Forrest 17	**Forrest 18**	**Forrest 19**	**Forrest 20**	**Oxbow 32**	**Oxbow 33**	**Oxbow 34**
ET_G (mm/day)	2.4	2.7	2.2	1.8	2.1	2.6	2.7
$24*r_{GW}$ (mm/day)	106	115	85	77	94	117	123
GW (mm/day)	2.1	2.3	1.7	1.5	1.9	2.3	2.5

Table 4. Average error between calculated and observed water table surfaces at Forrest and Oxbow sites from May to August, 2010.

	05/01/10	06/01/10	07/01/10	08/01/10	*Site Average (cm)*
Forrest (cm)	2.4	7.7	3.9	7.0	*5.025*
Oxbow (cm)	4.2	0.9	6.7	7.8	*4.9*
Monthly Average (cm)	*3.3*	*4.3*	*4.85*	*7.4*	*4.96*
% difference	*5.5%*				

Table 5. Model results for incised and restored scenarios, with percent change

			Deeply Incised (IC-3)	**Incised (IC-1)**	**% Change (IC-3 to IC-1)**	**Restored (UC)**	**% Change (IC-3 to UC)**
Channel depth	D_i	m	3	1	-66.7	0.33	-89.0
Time of interest	t_c	days	92	82	-10.0	77	-15.0
Max WTE, below valley floor	WTE_{max}	m	1	0.3	-70.0	0	-100
Available Storage	V_{max}	m^3	1,800,000	1,800,000	0.0	1,800,000	0.0
Maximum Storage	V_{max}	m^3	1,620,000	1,746,000	+7.8	1,800,000	+11.1
Drainable Storage	V_W	m^3	360,000	388,000	+7.8	400,000	+11.1
Laterally drainable storage per km	V_{Wlat}	m^3	80,000	28,000	-65.0	13,200	-83.5
Volume drained per km, t_0 to t_c	V_{Wlat}^*	m^3	14,146	1558	-90.0	326	-97.7
Max lateral flow rate	Q_{lat}	m^3/s	$1.0x10^{-3}$	$1.2x10^{-4}$	-88.0	$1.3x10^{-6}$	-98.7
Max longitudinal flow rate	Q_{long}	m^3/s	$3.0x10^{-4}$	$2.5x10^{-4}$	-16.7	$2.4x10^{-4}$	-20.0
Discharge at t = t_c	$Q(t_c)$	m^3/s	$1.2x10^{-6}$	$2.3x10^{-7}$	-80.8	$6.6x10^{-8}$	-94.5
Discharge at t = t_1	$Q(t_1)$	m^3/s	$1.5x10^{-2}$	$1.7x10^{-3}$	-88.7	$3.8x10^{-4}$	-97.5
Total discharge, t = t_c	$Q_{tot}(t_c)$	m^3/d	$1.0x10^{-1}$	$2.0x10^{-2}$	-80.0	$5.7x10^{-3}$	-94.3
Total discharge, t = t_1	$Q_{tot}(t_1)$	m^3/d	1329	154	-88.4	33	-97.5
Daily ET (from: Loheide and Gorelick, 2005)	ET	mm/d	3.0	5.0	+66.7	5.5	+83.3
Total ET usage, t = t_c	ET_{tc}	m^3	$1.0x10^5$	$1.6x10^5$	+49.1	$1.7x10^5$	+54.0

Figure 1. Schematic representing water fluxes into and out of an upland valley floor (meadow). Abbreviations as described in text. Impermeable bedrock based is assumed to extend up into hillslopes along the dotted line. We assume that water held in confined aquifers does not interact with the valley in question.

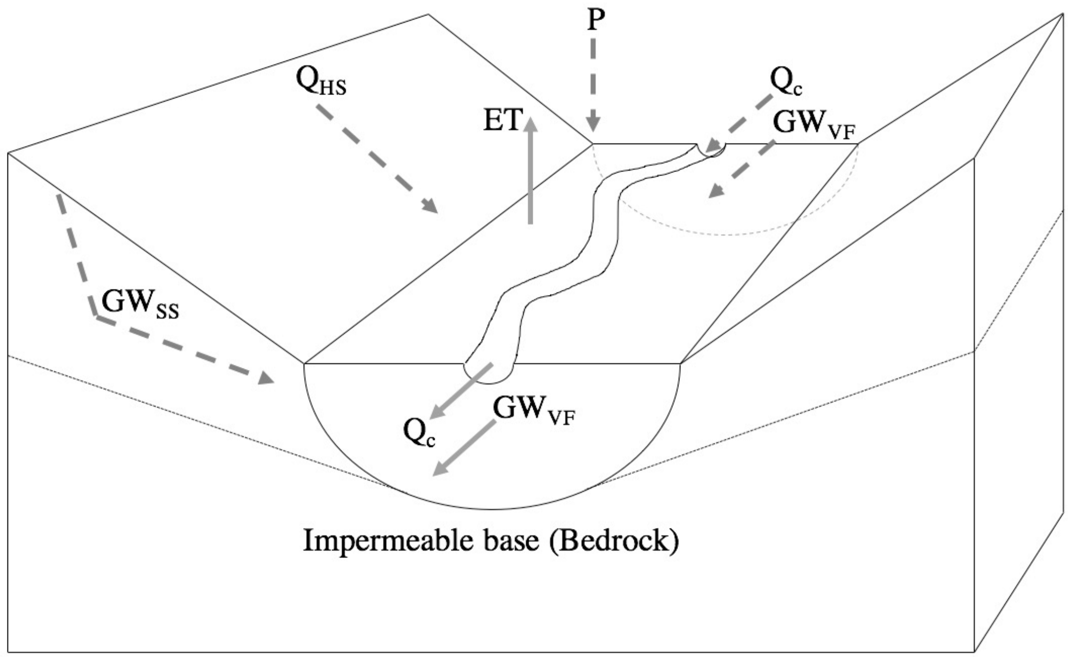


Figure 2. Dimensional sketch of upland valley floor, channel and time-variable water table position for (a) any un-impeded channel cut into valley fill and (b) a channel restored with low-head dams. After Rupp and Selker (2005)

a.

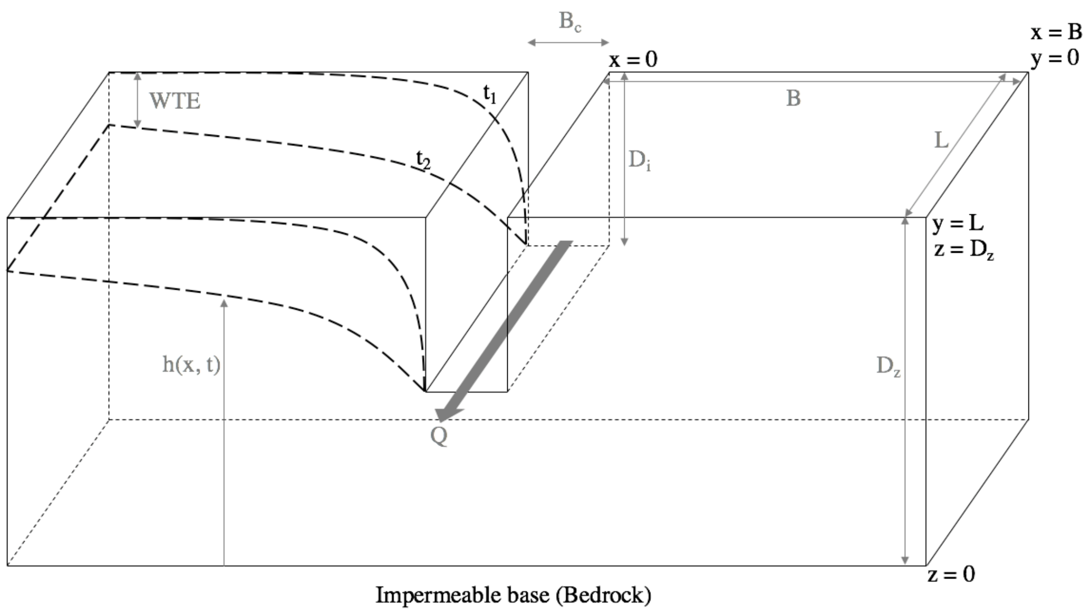


Figure 3. Drainage curves produced from EQN 12 for incised and restored scenarios. Top of the curve shows position of water table at t_1, bottom of curve shows position of water table at t_c. Note differences in scale of Y-axis.

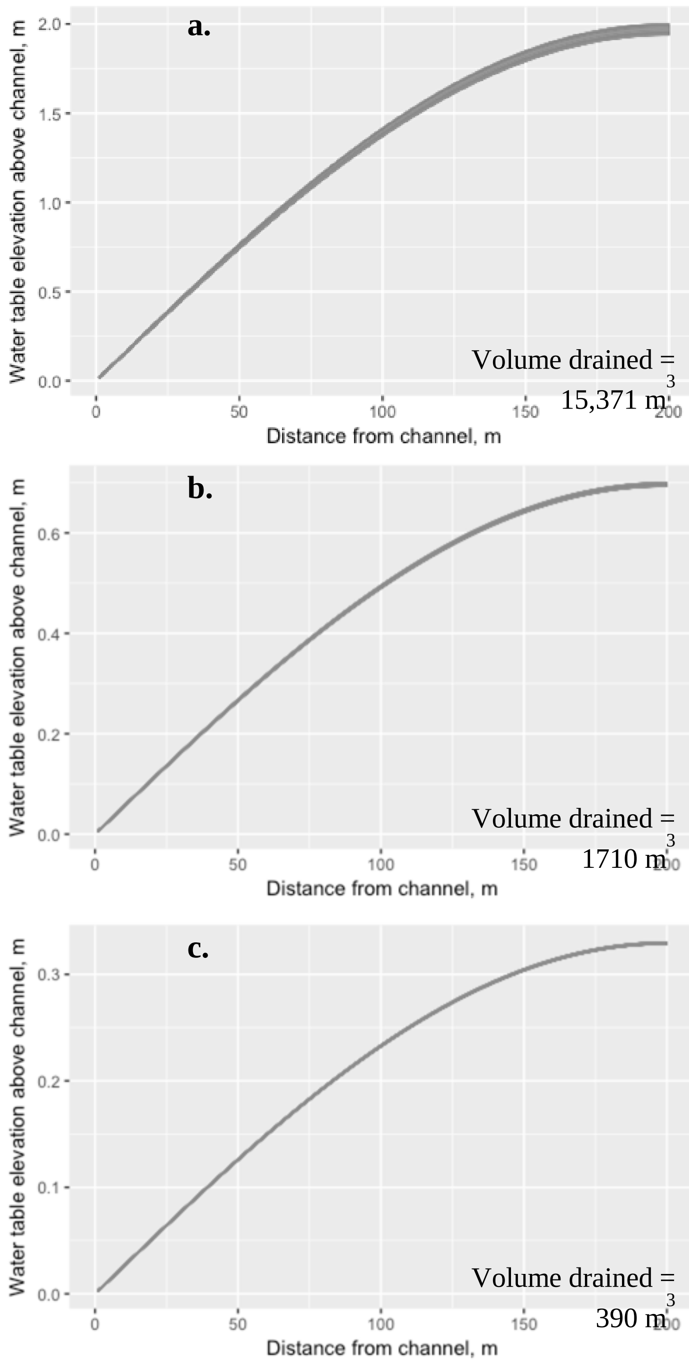


Figure 4. The study site for meadow drainage on the Middle Fork of the John Day River (a) with an example of a 4-well transect in the Forrest property (b) which is immediately downstream of Bates Oregon, and 15 km upstream of the similar 3-well transect in the Oxbow. The site is located in Northeastern Oregon, USA

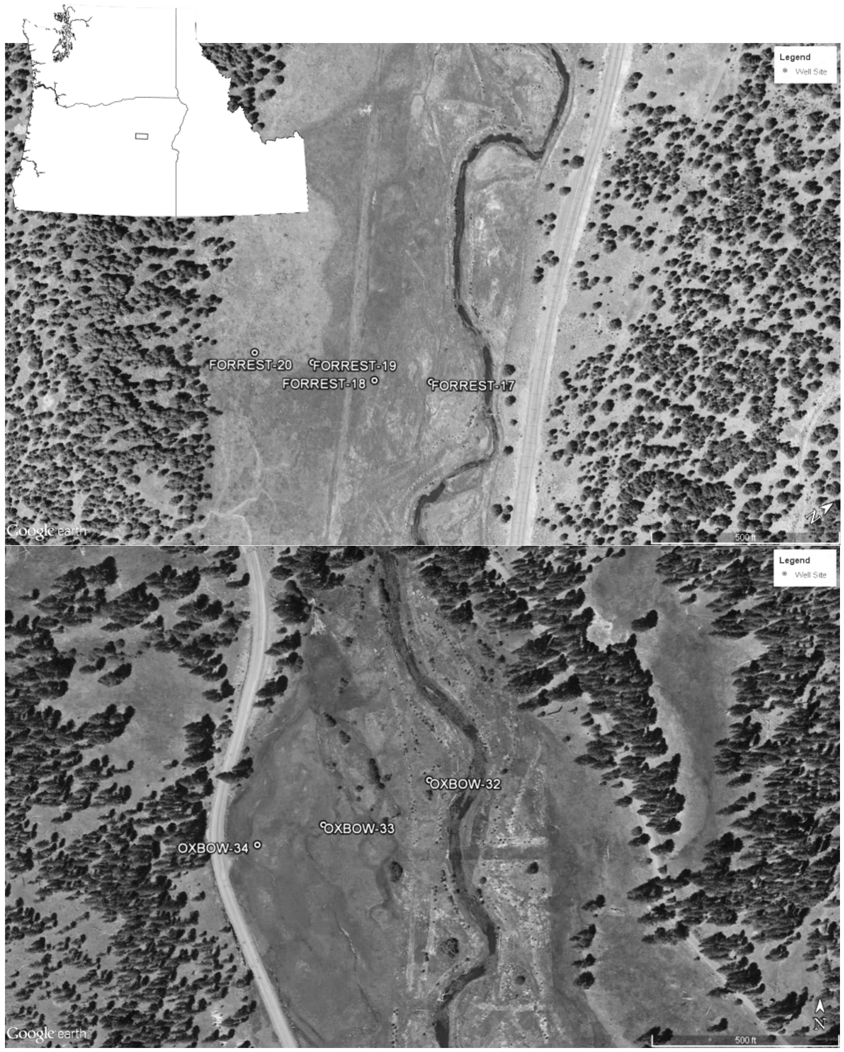
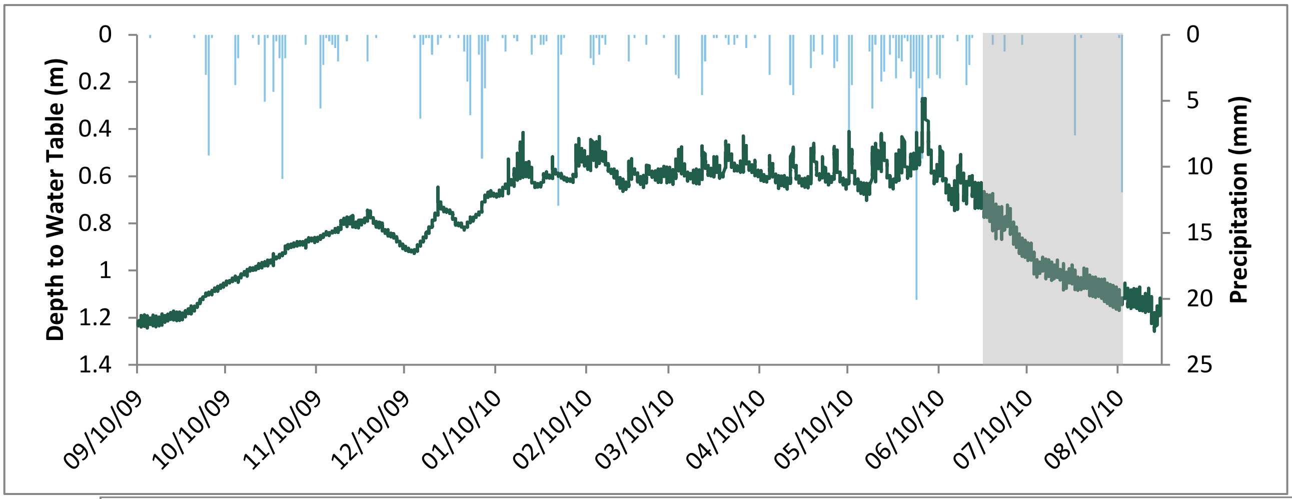


Figure 5. (A) Depth to water table at Oxbow-34 during 2009 water year. Precipitation shown on top of graph is from Agrimet station in Prairie City, 25 km to the south. (B) Daily signal in water table elevation during the decrease of the water-table at Oxbow well 34 during portion of summer of 2010 (shaded in (A))



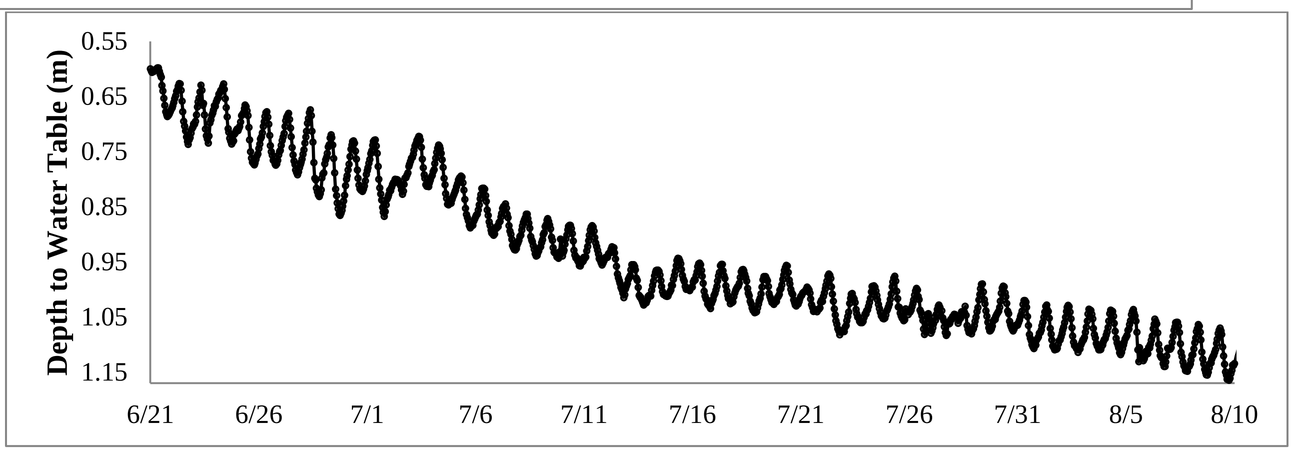
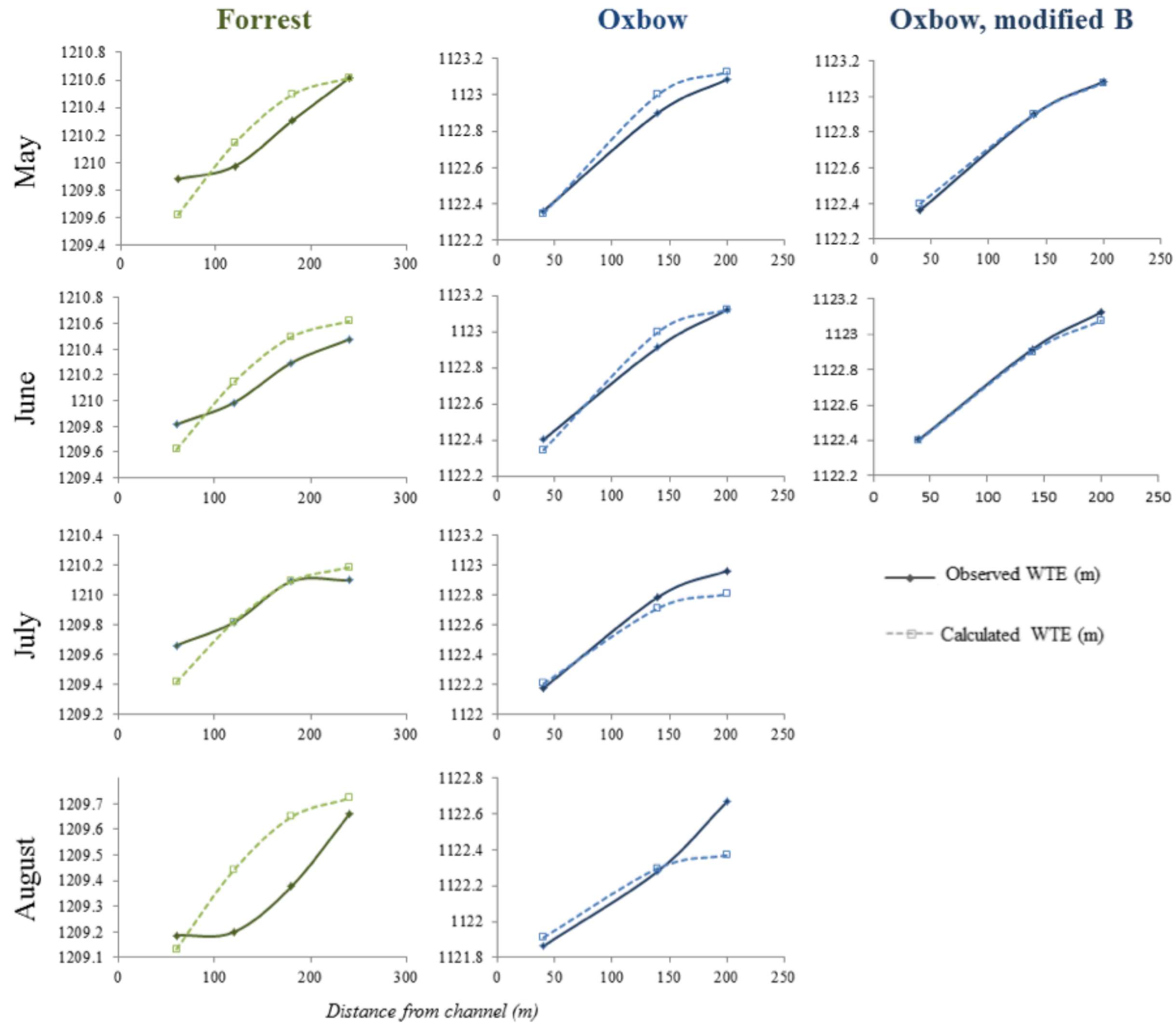


Figure 6. Observed (solid line) and calculated (dashed line) water table surfaces at Forrest and Oxbow sites, May 1 to August 1, 2010. modifying valley width to be slightly larger at the Oxbow site improves the fit of early season calculated WTE surfaces.



Appendix F.

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Appendix I – Future Changes in Mainstem Water Temperatures in the Upper Middle Fork John Day River and the Potential for Riparian Restoration to Mitigate Temperature Increases

Steven M. Wondzell, USDA Forest Service, Pacific Northwest Research Station, Corvallis OR

Mousa Diabat, Water Resources Graduate Program, Oregon State University, Corvallis OR

Roy Haggerty, Oregon State University, Corvallis OR

Abstract

Stream temperature regimes are expected to change in response to changes in air temperature and stream discharge that result from global change. Stream temperatures are also influenced by anthropogenic changes to riparian vegetation that either increase or decrease stream shade. The mechanistic stream temperature model, Heat Source, was used to analyze potential changes in stream temperature along a 37-km study segment of the upper Middle Fork John Day River (MFJDR), located in northeast Oregon, USA. The river currently supports populations of spring Chinook salmon, steelhead, and bull trout, all of which are cold-water dependent species and both steelhead and bull trout are listed as threatened under the USA Endangered Species Act. Because of their population status, the river has been a focal point of restoration. However, maximum stream temperatures already exceed lethal thresholds in some summers and there is concern that future increases in air temperature will further threaten these populations. Heat source was used to examine alternative future scenarios based on down-scaled projections from climate change models and the composition and structure of native riparian forests. The 36 scenarios examined all possible combinations of future increases in air temperature (ΔTair = 0, +2, and +4 ^{o}C), stream discharge (ΔQ = –30%, 0%, and +30%), and riparian vegetation (post-wildfire with 7% effective shade, current vegetation with 19% effective shade, a young-open forest with 34% effective shade, and a mature riparian forest with 79% effective shade).

Simulation results suggested that the upper Middle Fork John Day has a wide range of potential future thermal regimes. Specifically, the future 7-day, daily average maximum stream temperature (7DADM) ranged from ~4 ^{o}C hotter to ~8 ^{o}C colder than current conditions under a future climate in which air temperatures were 4 ^{o}C hotter than today. Shade from riparian vegetation had the largest influence on stream temperatures, with the range in effective shade from 7% to 79% changing the 7DADM from +1 ^{o}C to –7 ^{o}C. In comparison, the 7DADM increased by less than +2 ^{o}C in response to

the ±30% change in discharge or +4 ^{o}C increase in air temperature. Because many streams supporting cold-water dependent species throughout the interior western United States have been anthropogenically altered in ways that have substantially reduced shade, there is great potential to restore shade over long segments of these streams. The effect of such restoration could be so large that future stream temperatures could be colder than today, even under a warmer climate with substantially lower late-summer streamflow.

Introduction

Background

Populations of salmon, steelhead trout, and char have been listed as threatened or endangered throughout much of their native range (Nehlsen et al. 1991), including the Columbia River Basin (Fig. 1A). Many factors have contributed to the decline of these populations, including loss of high quality freshwater habitat for spawning and rearing (Federal Caucus 2000). Habitat factors are multiple and complex; no single factor accounts for population declines. However, water quality, especially summer maximum stream temperature, is clearly implicated in these population declines. Furthermore, high water temperatures have also been identified as a critical barrier to species recovery. Simply put, summer maximum stream temperatures are near thresholds of thermal tolerance for these species in many streams throughout the interior Columbia River Basin and elsewhere throughout much of their native ranges in the conterminous USA. There is great interest in restoring salmon and trout populations within their native range. Combined, restoration projects within the Columbia River Basin constitute one of the single most expensive recovery efforts ever undertaken within the United States (GAO 2002). Climate change, however, raises serious questions about the long-term outcomes of restoration because projected increases in air temperature could make many of these streams and rivers uninhabitable for salmon and trout within a few decades (Battin et al. 2007; Mantua et al. 2010; Isaak et al. 2012).

Studies examining stream energy budgets and the relative influence of different energy terms show that short-wave radiation, especially direct-solar radiation, dominates the stream heat budget and is therefore the single biggest determinant of stream temperature on summer days. Thus, the degree to which a stream is exposed to the sun is likely to have the greatest effect on summer daily maximum temperatures. A number of factors determine this solar exposure. Channel width and sinuosity are the basic determinants of the surface area of the stream for a given length of stream valley. Topographic and vegetation shade (angular height above the stream, and location with respect to the sun and the stream) then determine how much of the surface area is exposed to the sun over the course of the day.

Topography is non-changing, and channel width and sinuosity are usually considered relatively static at the time scales of interest so that most analyses examining potential change in a stream's exposure to direct solar radiation focus on shade from riparian vegetation. We do note, however, that active channel restoration can rapidly change both channel width and sinuosity. Channel width and sinuosity will also change in response to natural processes. And while such changes are slow, they may not be substantially slower than the time needed to grow trees to provide shade.

Here, we explore how direct and indirect effects of climate change might interact with riparian vegetation to influence future stream temperatures. The influence of active channel restoration that changes width and sinuosity, and thus solar exposure, is examined in a separate IMW report (Selker et al., 2017). Climate change projections for mid-latitudes consistently agree that air temperatures will warm in the future and increased air temperatures may lead to increased stream water temperatures. The loss of existing shade could amplify the increase in stream temperatures expected from warming air temperatures. Conversely, increasing shade where it is currently limited or lacking could mitigate changes expected from the increase in air temperatures. Future climate change may also influence stream temperature through indirect effects on stream discharge (Mantua et al. 2010). Ensembles of multi-model and multi-emission scenario simulations project slight decreases in summer precipitation for the Pacific Northwestern United States as well as warmer winter air temperatures which would decrease accumulated winter snow and lead to earlier snowmelt. Analyses of these climatic forcings with Variable Infiltration Capacity (VIC) Macroscale Hydrologic Model (Liang et al. 1994) suggests that climate change will increase the length of summer low flow periods and reduce summer stream discharge.

Goals and objectives

Increased air temperature, decreased stream discharge, and loss of stream shade all have the potential to increase stream temperatures in the future. However, the relative influence of each factor, and their interactions, are poorly understood. Further, the potential for riparian restoration to mitigate potential changes is also poorly documented. The objective of this study was to examine potential changes in stream temperature resulting from increased air temperatures and changes in both riparian shade and stream discharge. Realistic scenarios for changes in air temperature, shade, and discharge were used in a sensitivity analysis using the mechanistic stream temperature model, Heat Source. Interactions among factors were examined to identify potential management decisions that could mitigate expected increases in stream temperatures that are expected to occur over the next 40 to 80 years.

Site Selection

The study segment comprises 37 km of the upper Middle Fork John Day River (MFJDR) in northeastern Oregon, USA (Fig.1), beginning upstream of the confluence with Clear Creek where the contributing watershed area is 167 km^2 and ending downstream of Camp Creek where the contributing watershed area is 827 km^2. The streambed elevation decreases by 210 m over this distance, resulting in an average longitudinal gradient of 0.0058 m/m. The valley is comprised of a series of flatter, unconstrained reaches separated by slightly steeper moderately constrained and narrowly constrained reaches with narrow valley floors (Bureau of Reclamation, 2008).

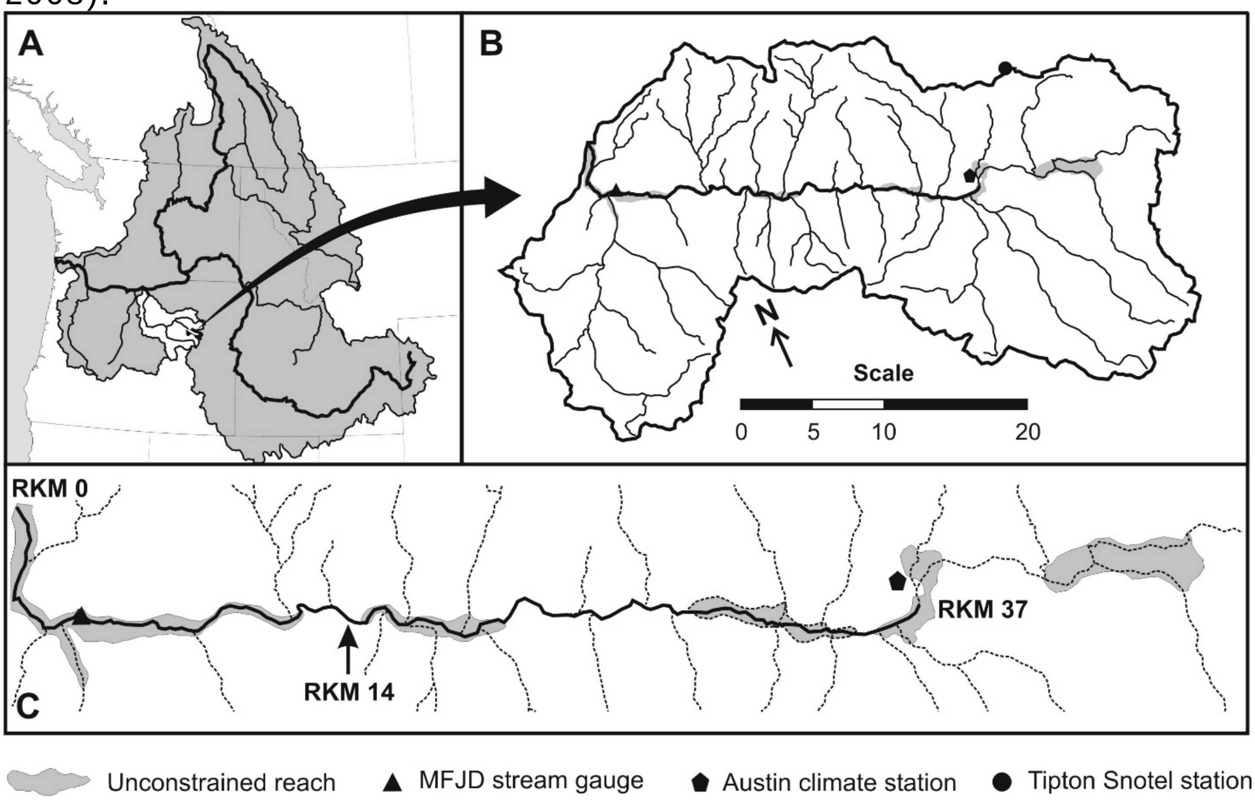

Figure 1. Study site location, showing the location of the John Day River basin within the Columbia River Basin (A), the upper Middle Fork John Day basin, mainstem river and major tributaries (B) and a close up of the 37-km long simulation reach used in this study (C).

Methods

Stream temperature was simulated using the mechanistic model, Heat Source v. 8.04 (Boyd 1996; Boyd and Kasper 2003; http://www.deq.state.or.us/wq/tmdls/tools.htm, accessed 8 July 2014). A version of the Heat Source model had been parameterized and calibrated by Crown and Butcher (2010) to simulate stream temperatures of the MFJDR for 2002 and used for Oregon Dept. of Environmental Quality's Total Maximum Daily Load analysis of stream temperatures in the John Day River

Basin (ODEQ 2010). The 37-km long study segment was extracted from Crown and Butcher's (2010) model, and used in the analyses presented here, with 2002 as the base case for comparison with simulations of future stream temperatures.

Heat Source simulates the stream's heat budget to predict changes in stream temperature. Upstream boundary conditions are specified in the model, describing both discharge and water temperature. Similarly, lateral boundary conditions describing both discharge and temperature are specified for all tributary and groundwater inputs entering the study segment. Heat Source combines the input discharge with data describing the channel conditions (bottom width of channel, bank slopes, longitudinal gradient, and Manning's n) to calculate average velocity, and both the top width and average depth of the wetted channel for each finite-difference element along the entire length of the study segment. Atmospheric data consist of a time-series of air temperature, humidity, cloud cover, and wind speed. The available projections of future climate only project changes in air temperature. Therefore, humidity, cloud cover, and wind speed from the 2002 base case were used in all model simulations.

Effective shade describes the proportion of shortwave radiation blocked from reaching the stream by either topography or vegetation. The potential solar radiation reaching the earth's surface is based on the atmospheric conditions and the position of the sun which is calculated from latitude, day of year, and time of the day. Then, land-cover data consisting of tree height and canopy density, and topographic elevation within a defined distance from the stream, are used to calculate the portion of the shortwave radiation that would be blocked from reaching the stream. Hourly data files from Heat Source, recording both the potential solar radiation and the amount reaching the streams surface (which includes both direct and diffuse solar radiation) were used to calculate the average effective shade between 9:00 and 15:00 for period July 8 through 14, and averaged over the entire 37-km study segment.

Multiple model scenarios were developed by modifying inputs to HeatSource to realistically characterize possible future climatic and riparian vegetation conditions. Air temperature scenarios contrasted the 2002 base case parameterized by Crown and Butcher (2010) with a 2 ^{o}C and a 4 ^{o}C warmer future, by adding either 2 or 4 ^{o}C to each hour of the base-case hourly input files characterizing air temperature. Stream discharge scenarios contrast the 2002 base case with two future conditions representing either increased or decreased stream discharge. To increase (decrease) the discharge, the hourly time series representing the upstream boundary condition was increased (decreased) by 30%. Finally, the range in riparian forest structures in state and transition models (STMs) simulating dynamics of riparian vegetation in the upper MFJDR were examined to determine likely future conditions (Wondzell et al. 2017). The STMs simulate a wide range of potential riparian vegetation conditions, from recent stand replacing wildfire that entirely lack shade, through young forest stands, up to mature forest stands of large trees with closed, multi-storied canopies. This range in conditions was simulated as:

1. a recently burned condition dominated by herbaceous vegetation (grasses and forbs 1-m tall with 10% canopy cover) providing 7% effective shade;
2. the 2002 base case or current condition providing 19% effective shade;
3. a young forest condition (trees 10-m tall with 30% canopy cover) providing 34% effective shade; and
4. a mature forest condition (trees 30-m tall with 50% canopy cover) providing 79% effective shade.

These 3 potential future riparian vegetation conditions were contrasted with the current condition parameterized by Crown and Butcher (2010) and represented by the 2002 base case. Additional details describing these model simulations can be found in Diabat (2014).

This study focused on late summer warm temperatures because the upper MFJDR and many of its tributaries are listed as Water Quality Limited for temperature with EPA-approved TMDLs and Water Quality Management Plans (ODEQ, 2010). Further, die-offs of adult salmon occurred within the study segment in both 2007 and 2013 due to high water temperatures, and in 2013 resulted in an estimated loss of 60% of the adult spawning population (Jim Ruzycki, Oregon Department of Fish and Wildlife, unpublished data). We analyzed our model results for the 7-day running average of daily maximum stream temperature (7DADM) because: 1) the 7DADM integrates maximum temperatures over a biologically relevant length of time, and 2) the 7DADM is a regulatory criteria for water quality in the State of Oregon. Therefore, we compared our simulations using the 7DADM, which we calculated for each finite-difference element along the study segment for each model run.

We examined stream temperatures for each scenario at river kilometer 14 (RKM14). Only a few relatively small tributaries enter the mainstem MFJDR for many kilometers upstream of this site. Also, this location is sufficiently distant from the upstream boundary of our study segment that stream temperatures were not heavily influenced by of the designated boundary conditions. This site is also located within an open meadow (RKM16 to RKM21) where effective shade averages 19% under the base-case conditions so that changes in the energy budget will reasonably reflect changes in effective shade occurring over the length of the study segment.

Results

Summary of Analyses

Changes in shade, air temperature and stream discharge influenced the thermal regime in different ways (Fig. 2, 3, and 4). For example, an analysis of hourly stream temperatures for the 7-d period over which the stream reached its 7DADM at RKM14 showed that increasing effective shade from the base-case to the mature-forest scenario decreased daily maximum stream temperatures by 7.8 ^{o}C, decreased the daily average stream temperatures by 4.5 ^{o}C, but only decreased the daily minimum stream temperatures by 1.6 ^{o}C (Fig. 2). Conversely, simulations showed that loss of shade from a large-scale disturbance (such as a wildfire burning the entire study segment) would increase stream temperatures relative to the base case (Fig. 2). These increases are relatively modest because much of the upper MFJDR currently flows through wide, open meadows where there is little shade. Thus the change from the base case to the post-wildfire vegetation (1-m tall; 10% canopy density) only reduces the average effective shade from 19% to 7%. The simulations also showed relatively little change in stream temperatures under the young-open forest vegetation scenario (10-m tall with 30% canopy cover, 34% effective shade) relative to the base case (Fig. 2).

The influence of increased air temperatures was relatively uniform over the entire 24-h daily cycle in stream temperatures such that the 4 ^{o}C increase in air temperature increased the minimum, average, and maximum daily temperatures by 1.4 ^{o}C (Fig. 3). The influence of changing discharge was quite different. Increasing the discharge from –30% to +30% reduced the daily maximum stream temperatures by 0.7 ^{o}C but increased the nightly minimum stream temperatures by 1.3 ^{o}C. Changing discharge did not influence the daily average stream temperature (Fig. 4). In other words, increasing the discharge narrowed the daily range in stream temperatures.

A sensitivity analysis compared 36 scenarios based on all possible combinations of changes in air temperature, stream discharge, and riparian vegetation (Fig. 5). Those model simulations showed that shade was the single biggest factor influencing the projected 7DADM along the 37-km long study segment, regardless of changes in air temperature or stream discharge. At the very bottom of the study segment (RKM0), the 7DADM ranged over 10 ^{o}C from changing just riparian vegetation (Fig 5, range among vegetation scenarios at RKM0), whereas changes in air temperature and stream discharge led to an ~2 ^{o}C range in the simulated 7DADM (Fig. 5, range within a vegetation scenario at RKM0). These patterns were similar at RKM14, where the 7DADM ranged over 8 ^{o}C from changing just riparian vegetation, whereas changes in air temperature alone resulted in 1.4 ^{o}C range in 7DADM and changes in stream discharge alone resulted in a 0.7 ^{o}C range in the simulated 7DADM (Fig. 5).

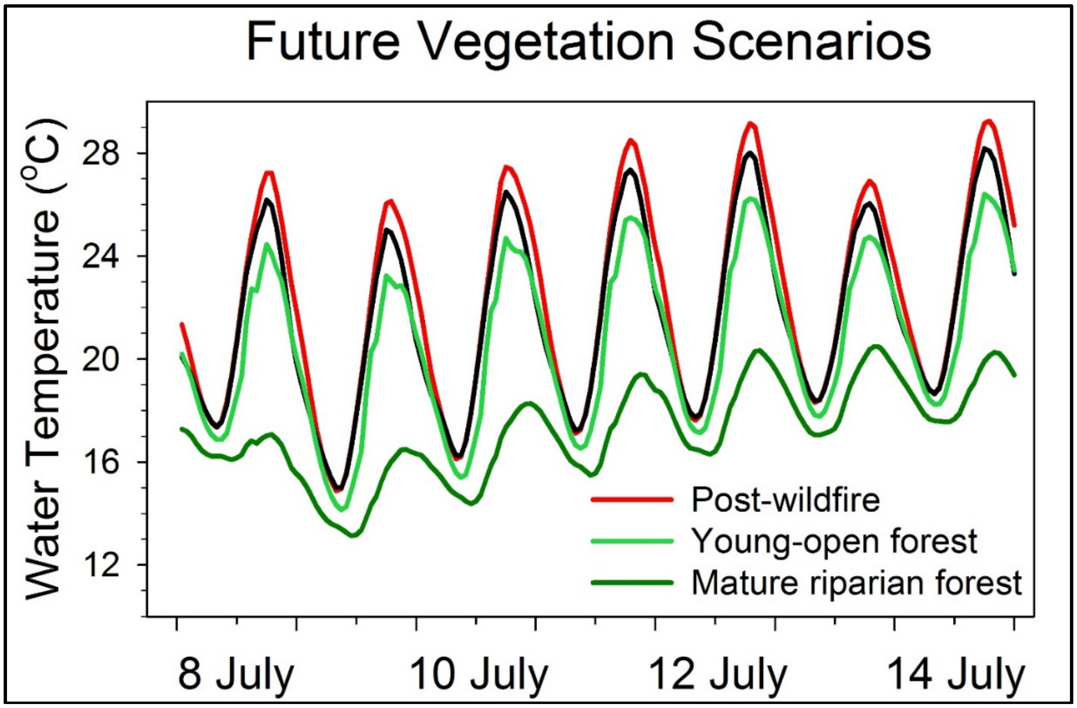

Figure 2. Hourly stream temperature time series contrasting 3 riparian vegetation scenarios with 2002 base-case conditions at RKM14.05 over the 7-d period during which the simulations reached 7DADM. Black line represents the 2002 base case scenario.

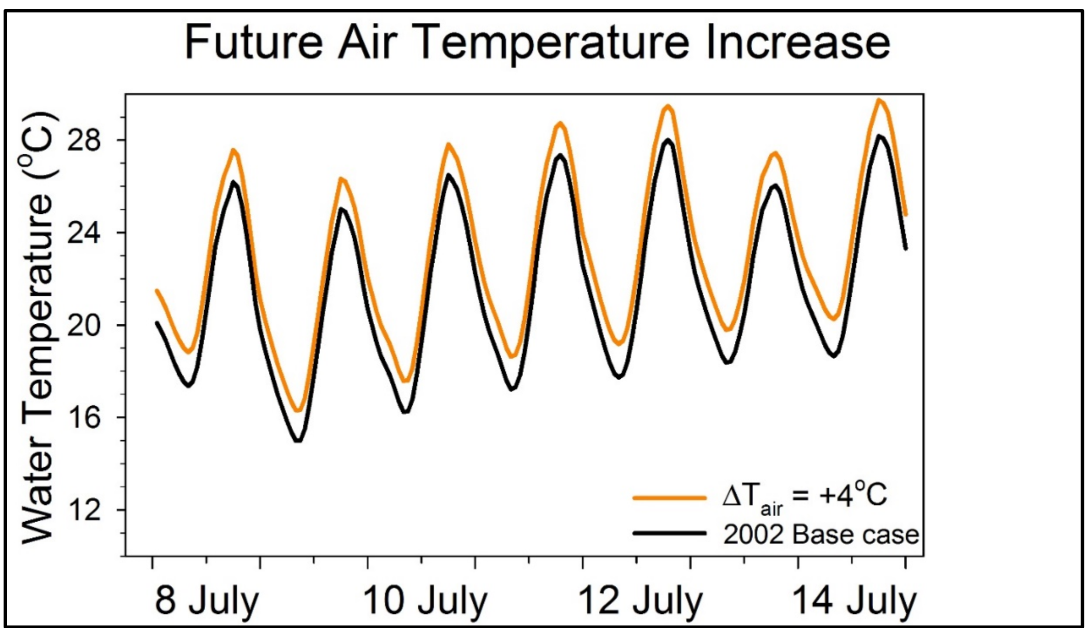

Figure 3. Hourly stream temperature time series contrasting a simulation of future conditions when air temperatures are 4 ^{o}C warmer with the 2002 base-case conditions at RKM14.05 over the 7-d period during which the simulations reached 7DADM.

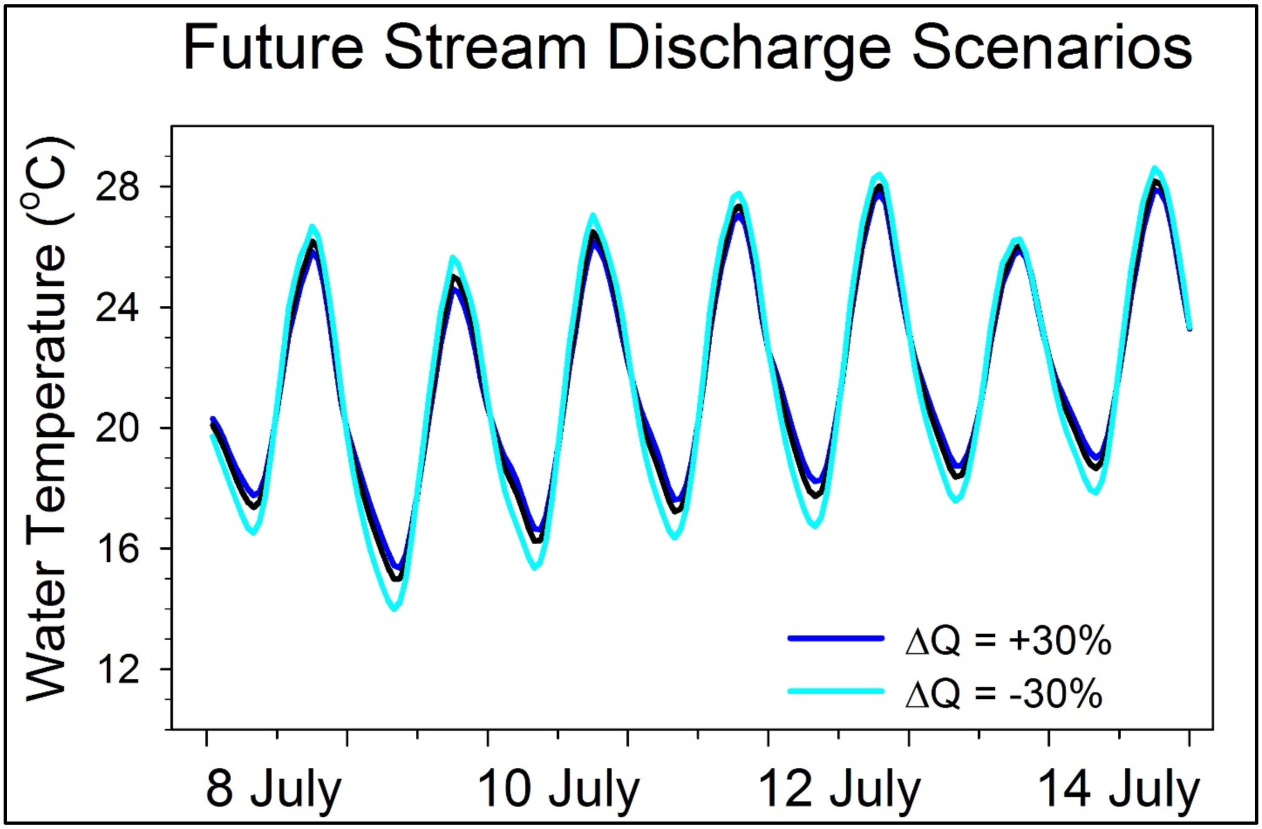

Figure 4. Hourly stream temperature time series contrasting two simulations of future conditions in which discharge either increases (+30%) or decreases (-30%) with the 2002 base-case conditions at RKM14.05 over the 7-d period during which the simulations reached 7DADM. Black line represents the 2002 base case scenario.

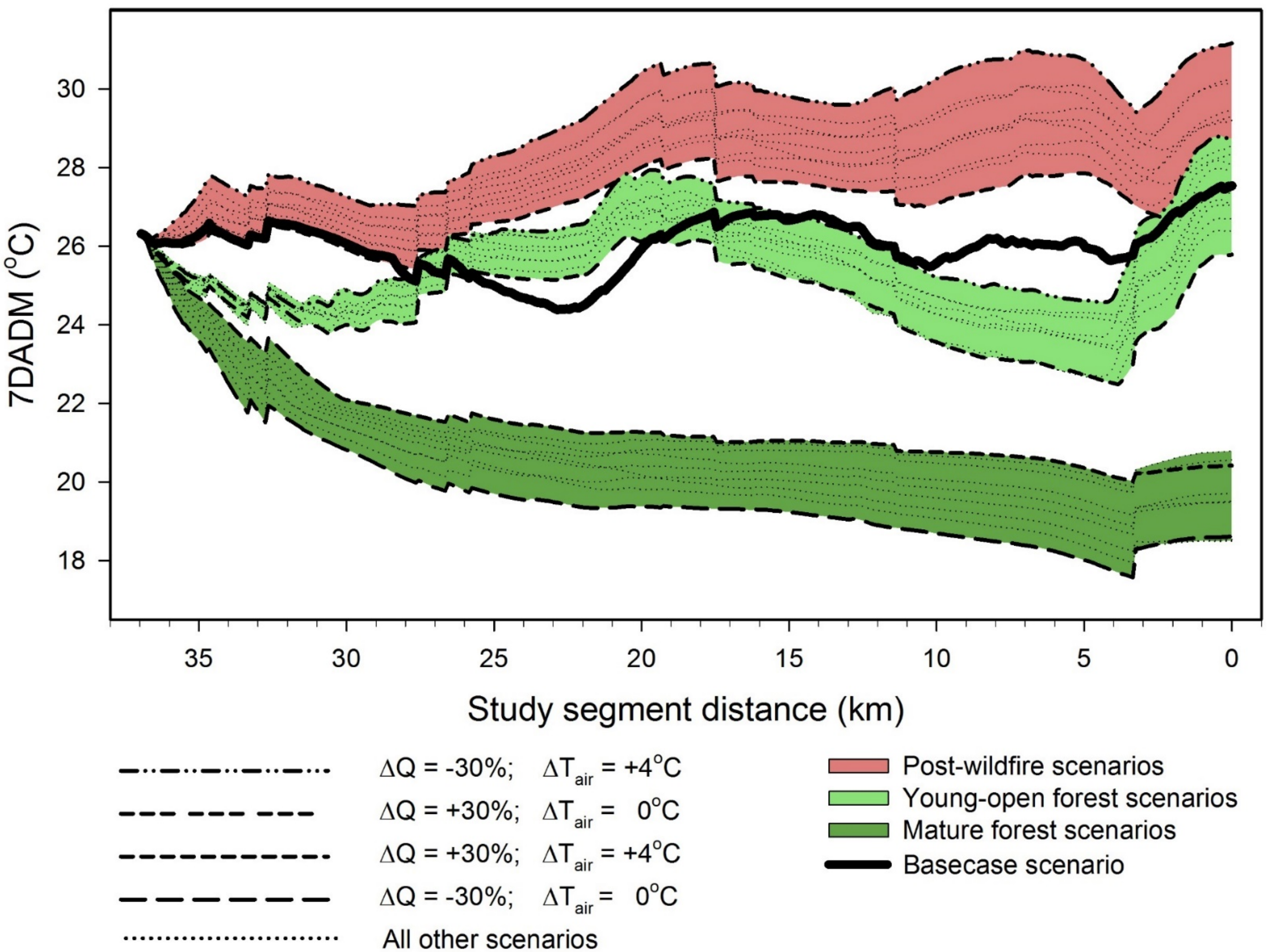

Figure 5: Simulated 7DADM stream temperatures over the length of the study segment. Simulation results are grouped for three riparian vegetation scenarios (pink, light green, and dark green shaded zones) bounded by bold lines representing combinations of T_{air} and Q representing the scenario with the warmest or coldest simulated 7DADM stream temperatures. The remaining simulations for each vegetation scenario are indicated by light dotted lines bracketed by the warmest and coolest simulations within each vegetation scenario. Note that abrupt step changes in temperature result from tributary inputs of warmer or cooler water. Also note that under both the post-wildfire and young-open forest scenarios, the +30% Q simulations result in the coldest stream temperatures. This pattern is reversed under the mature forest scenario where the +30% Q simulation results in the warmest stream temperatures.

The simulations showed that loss of shade from a large-scale disturbance such as a wildfire burning the entire study segment would increase the 7DADM relative to the base case (Fig. 5). The increases are relatively modest because much of the upper MFJDR currently flows through wide, open meadows where there is little shade. Thus the change from the base case to the post-wildfire vegetation (1-m tall; 10% canopy density) only reduces the average effective shade from 19% to 7%. The resulting changes in 7DADM are smallest for the scenario with +30% increase in stream discharge and no change in future air temperature, however, a 4^{o}C increase in air temperature and a -30% decline in stream discharge, coupled with the loss of existing shade increased the 7DADM over most of our study segment by 3^{o}C to 5^{o}C (Fig. 5).

The simulations showed relatively little change in 7DADM under the young-open forest vegetation scenario (10-m tall with 30% canopy cover) relative to the base case (Fig. 5). The results, however, varied with location along the study segment, with the 7DADM actually warmer than the 2002 base case from RKM27 to RKM18. However, the 2002 base-case effective shade is high from RKM30 to RKM24 so that the young forest scenario actually leads to a decrease in effective shade over this portion of the study segment. From RKM20 to RKM5, the young-open forest scenario increases effective shade by 22%, so that the 7DADM decreases relative to the base-case scenario. There is substantially less effective shade in the lowest 5 km of the study segment so that the 7DADM increases rapidly over these 5 km (Fig. 5). Finally, the 7DADM is also sensitive to underlying changes in stream discharge and air temperature over the entire study segment. If air temperatures do not increase, 7DADMs will be lower than the base-case scenario under increased discharge. Under decreased discharge and increased air temperatures, the 7DADM is higher than the base case (Fig. 5).

The scenarios with mature riparian forest, characterized by 30-m tall trees with 50% canopy density, showed large decreases in 7DADM over the entire 37-km study segment (Fig. 5). These decreases ranged from 5.8 ^{o}C to 7.6 ^{o}C at RKM14 and from 7.1 ^{o}C to 8.9 ^{o}C at RKM0. Surprisingly, the 7DADM was warmer under the mature forest scenario at high discharge (+30%) than at low discharge (–30%). The decrease in the 7DADM was persistent over all scenarios examined, even in the face of a 4 ^{o}C increase in air temperature (Fig. 5).

Discussion

Simulation Results and Projected Future Stream Temperatures

The simulation results suggested that the upper Middle Fork John Day has a wide range of potential future stream temperatures. Specifically, estimates of the future 7DADM range from ~4 ^{o}C hotter to ~8 ^{o}C colder than current conditions under a future climate in which air temperatures are 4 ^{o}C hotter than today.

Shade was by far and away the single biggest factor influencing future stream temperatures – as previously demonstrated in similar studies employing mechanistic models (Chen et al. 1985; Cristea and Burges 2010; Lawrence et al. 2014; Justice et al. 2017). For example, considering average conditions in the lower 15 km of the study segment, changing discharge from -30% to +30% changed the 7DADM by 1.1 ^{o}C, about the same amount as would a 8.6% change in shade. Although this 60% change in discharge is very large relative to changes forecast under climate change scenarios (Hamlet et al. 2010 and 2013), it produces only a small change in the 7DADM. In contrast, an 8% or 9% change in shade is small, relative to the current conditions and the potential of riparian forests to shade the stream. Stream temperatures are slightly more sensitive to changes in air temperature, such that changing air temperatures by 4 ^{o}C changed stream temperatures as much as a 13% change in effective shade.

Under current conditions, there is relatively little shade from riparian vegetation, so disturbances that remove shade have small effects. However, loss of shade can interact with increases in air temperature to substantially increase maximum water temperatures. Conversely, if little shade is currently available, then there must be long lengths of stream where growing riparian forests to shade the stream may have a potentially huge influence on future thermal regimes. This was borne out in the Heat Source simulations. Increasing shade by growing riparian forests that were 30 m tall with 50% canopy cover reduced maximum stream temperatures well below current temperatures, even under warmer future climatic conditions.

The interaction between discharge and shade turned out to be surprisingly complex (Fig. 5). Under low shade conditions, simulated maximum stream temperatures were higher at low discharge than at high discharge. This relationship reversed under high shade conditions so that simulated maximum stream temperatures were actually lower at low discharge than at high discharge. This surprising result can be explained by examining the interactions between discharge and shade with changes in the heat budget of a stream.

How rapidly the stream temperature changes in response to a change in the heat budget depends on two factors: 1) the magnitude of the change in the heat budget, and 2) how much water must be heated or cooled. For a fixed amount of heat gained or lost, the observed temperature change will be inversely proportional to the amount of water that will be heated or cooled. Much of the scientific literature and the application of that literature concerns warming of streams when they are exposed to increased heat fluxes. Under these conditions, the general rule of thumb – that smaller streams will warm more than larger streams – generally holds true.

However, streams are not always warming. For example, under the mature forest scenarios, the downstream temperatures were substantially cooler than the upstream temperatures. That is, the stream was losing heat over most of the study segment. Under these conditions, higher discharge at the head of the study segment meant that more heat needed to be dissipated, so the stream cooled more slowly.

Streams do not consistently warm as they flow downstream. Some reaches will be cooling, others will be warming. Further, these relationships will change between night and day, among days with different weather patterns, and among seasons. The conditions that either tend to promote large heat fluxes per unit volume of water, or decrease the volume of water, will make the stream's temperature change more quickly. Thus, not only will shallow, wide streams with slow flow velocities and low discharge warm more quickly when they are heated, they will also cool more quickly when they are chilled. The management implication seems counter intuitive, but it is true. If the heat budget of a specific stream reach is consistently negative (i.e., the stream is cooling), then restoration efforts that deepen and narrow the channel, increase flow velocity, or increase discharge, will actually result in a warmer stream.

Growing Riparian Forests to Provide Shade

Given the potential importance of shade to future stream thermal regimes, a critical question then becomes – "Is it realistic to grow extensive riparian forests to shade this, or similar, stream reaches and thereby substantially reduce future maximum summer temperatures?" This question can be evaluated in light of the historic conditions that occurred along this stream as well as the desired future condition for the channel and riparian forest. This is not to say that the restoration objective is to return the stream to some earlier condition. Rather, that historical condition can be used to inform choices of desired future conditions for a given stream reach and its riparian zone and the potential of maintaining that condition (Wohl 2011; Woelfle-Erskine et al. 2012). Of course, the desired future condition is not a static state, but rather incorporates a range of conditions in response to both natural disturbance and anthropogenic activities. Finally, the desired future condition should also recognize likely changes in natural disturbance regimes resulting from climate change (Bollenbacher et al. 2014; Millar 2014).

The current conditions of the channel and riparian forest along the upper Middle Fork John Day River are far different than their conditions prior to Euro-American settlement (Wondzell et al. 2017). Historic conditions were more complex, especially in stream reaches with wide, or unconstrained, valley floors. These reaches have been converted from sinuous, multi-thread channels to straighter, single-thread channels. Historic vegetation in these reaches included conifer forest, hardwood forest, woody riparian shrubs, and wet meadows whereas today most of these reaches support dry meadows with substantial cover of introduced European pasture grasses. The effect of these changes on stream thermal regimes would be complex. For example, increased sinuosity, multi-thread channels, and the likely presence of beaver dams on some back channels would all increase the stream surface area, increase total channel length and decrease flow velocity so that a greater surface area of water would be exposed to sunlight over a longer period of time and thus potentially leading to warmer summer stream temperatures than occur today. However, multi-thread channels and channel sinuosity would promote hyporheic exchange, narrower multi-thread channels would be more completely shaded by tall riparian shrubs, and channels might be narrower and deeper – all of which would promote cooler water temperatures.

Because restoration efforts are not currently focused on recreating a pre-Euro-American landscape, efforts to identify likely historic thermal regimes might only be of academic interest. Perhaps more immediately important is to recognize that neither the dry meadow vegetation nor the relatively straight single-thread channels that are currently present represent likely, historical conditions. If the current condition is not representative of the historical condition, then restoration efforts have substantial leeway to explore alternative desired future conditions.

Conditions were likely too wet to support riparian trees where wet meadows conditions existed historically. However, not all of the unconstrained stream reaches were in wet-meadow conditions, and where wet meadows were present, much of the valley floor has been drained to increase production of high quality forage for domestic livestock. The resulting dry meadow complexes have site characteristics that are likely to support a variety of riparian woody vegetation, with conifers and hardwood trees dominating drier sites and woody shrubs and wet meadows dominating wetter sites. Cottonwood and/or aspen are also likely to find conditions sufficient to support their growth across a wide range of micro-environments within the unconstrained stream reaches. Further, a variety of tall riparian shrubs can grow either as the dominant vegetation or as a sub-dominant canopy layer in combination with taller over-story trees and several of these shrubs are well adapted to wet sites. The potential for the currently unshaded stream reaches to support taller woody shrubs and trees has been widely recognized. As a consequence, major investments have been made to restore native riparian forests and shrubs throughout the upper Middle Fork John Day, and elsewhere throughout the interior Columbia Basin.

The simulation scenarios specifically examined these restoration treatments. Scenarios changing riparian vegetation were simplistic - simulating uniform vegetation over the entire riparian zone along the full 37-km length of the study segment. It is unrealistic to grow and maintain uniform riparian forests over such a long stream reach – but the intent was to explore the potential of riparian restoration to mitigate future increases in stream temperature due to climate change. The model simulations clearly show that, in streams where shade is currently limited, restoring riparian forest can offset the effect of future increases in air temperature and decreases in stream discharge.

Overall, the simulation results showed that maximum daily stream temperatures (the 7DADM) were not sensitive to even relatively large changes in stream discharge. Thus, projects that are specifically designed to mitigate high stream temperatures are likely to see greater reductions in stream temperature from restoring riparian vegetation to shade stream reaches where shade is currently limiting than from increasing base flow stream discharge.

This result poses an additional critical question: "*Can shade be grown fast enough to mitigate effects from climate change, given the rate at which such changes are projected to occur?*" A number of riparian woody species have been planted along the upper MFJDR in an attempt to restore forested riparian vegetation. One of the most successful of the species planted is ponderosa pine showing relatively low mortality and substantial growth even when exposed to severe browsing (Wondzell and Cochran 2017). However, evidence suggests that it is likely to take more than 100 years for ponderosa pine to grow to 30 m height. For example, the state and transition models (STMs) developed for riparian vegetation in the MFJDR (Wondzell et al. 2017) require 120 years for a ponderosa pine stand to grow from seedling initiation state into a large, closed canopy state. Thresholds for the large, closed canopy state classes require that trees exceed 50 cm DBH and canopy cover exceeds 40%. Data from 30 ponderosa pine trees in 15 riparian vegetation plots in the upper MFJDR used to confirm model parameterization showed that trees 30 m tall ranged from 50 to 70 cm in diameter. Thus the large, closed canopy state class for ponderosa pine closely resembles the mature forest stands simulated in the Heat Source models. The STM model parameterization agrees well with height-age data for ponderosa pine in eastern Oregon. For example, a query of the USFS's Forest Inventory and Analysis database (Donnegan et al. 2008) for the ages of 30-m tall ponderosa pine returned 146 trees with an average breast-height age of 184 years. The first- and third-quartile ages for the distribution were 120 and 225 years, respectively, and the minimum age required to reach 30 m was 62 years. Similarly, height growth curves for ponderosa developed from even-aged, managed stands in eastern Oregon and Washington show a wide range in the time required to reach 30-m height – from as little as 55 years in highly productive stands to more than 130 years in poorly productive stands (Barrett 1978).

In contrast to ponderosa pine, the STMs for cottonwood dominated stands in the MFJDR (Wondzell et al. 2017) require only 45 years to grow from seedling initiation state into a large, closed canopy state. Relatively little information is available on growth rates or height-age relationships for black cottonwood. In western Washington, a planted stand averaged 31 m tall 24 years after planting (Murray and Harrington 1983). Similarly, stands in western British Columbia, ranging in age from 44 to 57 years ranged in average height from 29 to 38 m (Kellogg and Swan 1986). Black cottonwood might grow more slowly in the interior west, although stands along the Coeur D'Alene River, Idaho, reached heights of 20 m in 15 to 20 years and exceed 30 m within 50 to 60 years (Moseley and Bursik 1994).

While planting ponderosa pine seedlings might be an effective long-term strategy to shade streams, it is unlikely to provide sufficient shade to mitigate the effects of climate change over the next several decades. Obviously there is a large difference in the time needed to grow trees to 10 m tall versus 30 m tall (the two riparian forest types simulated with HeatSource). Trees even 10 m tall do help shade the stream and any increase in shade will help mitigate stream temperatures. Compared with ponderosa pine, cottonwood (and aspen) are much faster growing and, if successfully established, could reach the height and canopy cover necessary to shade streams over the same time periods that climate change models suggest stream temperatures would increase.

Taken together, our model simulation results; knowledge about tree growth rates embodied in the riparian state and transition models (Wondzell et al. 2017); and the difficulties establishing new riparian plantings shown in the seedling browse study conducted on the Oxbow and Forrest properties (Wondzell and Cochran 2017) suggest that substantial hurdles remain when attempting to mitigate impacts of future climate changes by growing trees to provide shade. Clearly, planting fast growing species such as cottonwood and aspen are more likely to provide shade at the time scales needed to mitigate climate change impacts. However, mortality appeared to exceed 95% for Cottonwood plantings and growth of both species was severely limited by deer and elk browsing. Exclusion of browsers allowed rapid growth of these species and within a few years they would grow their canopies beyond the reach of deer and elk. Unfortunately, field observations also showed that, wherever browse-exclusion fences were removed, aspen were quickly cut down by beaver. Other management strategies can also help mitigate climate change impacts. For example, active channel restoration narrowed the channel through the Oxbow Conservation Area and reduced maximum stream temperatures (Selker et al. 2017). Further, shorter statured vegetation such as the tall shrubs, alder, willow, and hawthorn, as well as younger trees of a variety of species, can more effectively shade narrower stream channels.

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Appendix J – Analysis of Benthic and Drift Macroinvertebrate Samples

Robin Henderson, Washington State University, Tri-Cities, School of the Environment

Abstract

To assist the MFIMW evaluate physical and biological responses to stream restoration, we compared macroinvertebrate communities between control and treatment (restored) reaches and streams. With both the benthic and drift macroinvertebrate datasets, we detected significant differences in years using analysis of variance (ANOVAs) and multiple comparisons tests ($p < 0.10$). Between controls and treatment reaches, significant differences were only detected with drift taxa richness. As determined using ANOVAs and multiple comparisons tests with years and the control/treatment streams as factors, there were not any significant differences in O/E scores among the control and treatment streams ($p = 0.78$). However, it is interesting to note that the treatment reaches were able to withstand the climate conditions in recent years better than the control reaches. We suggest exploring if functional group analysis and the use of spatial models would assist in providing conclusive evidence supporting the hypothesis that management actions are affecting the biotic integrity of the MFJDR.

Introduction

Background

Annually, a large amount of resources are utilized to restore biodiversity and ecological function in streams and rivers degraded by land use change and other anthropogenic activities. It is estimated that hundreds of thousands of miles of river corridors are degraded throughout the nation, with 42 percent of streams classified as having poor biotic integrity (EPA 2012, 2014). As such, stream restoration is an increasingly common approach utilized to reverse past degradation of freshwater ecosystems and to mitigate anticipated damage from future development and resource-extraction activities.

While there are many definitions of ecological restoration, the National Research Council defines restoration as the reestablishment of the structure and function of ecosystems (National Research Council 1992). One method to measure the trajectory of restoration is by assessing the current ecological condition or integrity of the ecosystem through the use of biotic indices (Palmer et al. 2005). Biotic indices can be used to set protection and restoration goals, identify stresses to the stream and decide how they should be controlled, as well as assess and report on the effectiveness of management actions (Cairns and Pratt 1993).

The underlying assumption of stream restoration programs is that biological integrity will improve following restoration, based on the hypothesis that “if we build it, they [the organisms] will come” (i.e., the “Field of Dreams” hypothesis) (Palmer et al. 2010). Unfortunately, there is little scientific evidence to support this assumption since most projects lack effectiveness monitoring and results have been widely variable (Alexander and Allan 2007; Bernhardt et al. 2005; Bernhardt et al. 2007; Frissell and Nawa 1992; Jähnig et al. 2011; Miller et al. 2010; Palmer et al. 2010; Roni et al. 2008; Rumps et al. 2007; Stewart et al. 2009; Whiteway et al. 2010). Furthermore, in instances where post-monitoring biological studies have occurred, they are limited in scope, employ overly simple methodology, and are not conducted synchronously with studies of the environmental stressors that were responsible for the degraded condition of the stream (Bernhardt et al. 2005). When biological assessments have been conducted, the variability inherent in biological systems has made it difficult to draw conclusions (Bernhardt et al. 2005; Roni et al. 2008; Rumps et al. 2007). As a consequence, scientists and practitioners know less than is necessary for determining whether stream restoration is leading to recovery of biological integrity in degraded streams and, if recovery does happen, when it will be evident (Bernhardt et al. 2005; Roni et al. 2008; Rumps et al. 2007).

Goals and Objectives

The goal of this project was to assist the IMW in gaining an understanding of the causal mechanisms linking stream restoration and salmonid production by comparing benthic and drift macroinvertebrate communities between control and treatment (restored) reaches. Specifically, we asked:

1. if the accuracy/precision of models that produce the Observed/Expected (O/E) index can be improved when using additional reference sites during model development.
2. if changes occurred post-restoration in the MFJDR benthic macroinvertebrate communities, as measured using the O/E index.
3. if changes occurred post-restoration in the MFJDR drift macroinvertebrate community, as measured using dry weight biomass (g) and taxa richness.

Hypotheses

Objective 1

For this analysis, we predicted that the O/E model would experience an increase in accuracy and precision with the addition of new reference sites.

Objective 2

For objective 2, we predicted that the biotic integrity of the MFJDR would improve following restoration.

Objective 3

For this analysis, we predicted that the MFIMW drift macroinvertebrates' biomass (g) and taxa richness would increase following restoration.

Site Selection

To develop the predictive model that generates the O/E index, we used the datasets (Oregon Department of Environmental Quality, Utah State University, and the Washington Department of Ecology) previously aggregated by Hubler (2008) for both reference and test sites (n = 105, n = 414) in Central and Eastern Oregon and Washington (Hubler 2008, 2013) (Figure 1). Additionally, we used sample datasets provided by the Environmental Protection Agency (EPA) and the United States Forest Service (USFS) as sources for new reference sites (n = 83) for O/E model development (Figure 1) (EPA 2010, 2014, 2017; US Forest Service 2015). In total, these datasets provided macroinvertebrate abundance data representing 188 reference sites and 414 test sites for predictive model calibration and development (Figure 2). Spatially, these sites are distributed within three Level 3 ecoregions (the Blue Mountain [BM] ecoregion, the Columbia Plateau [CP] ecoregion, and the Eastern Cascades Slopes and Foothills [ECF] ecoregion), representing approximately 315,942 square kilometers (Figure 2).

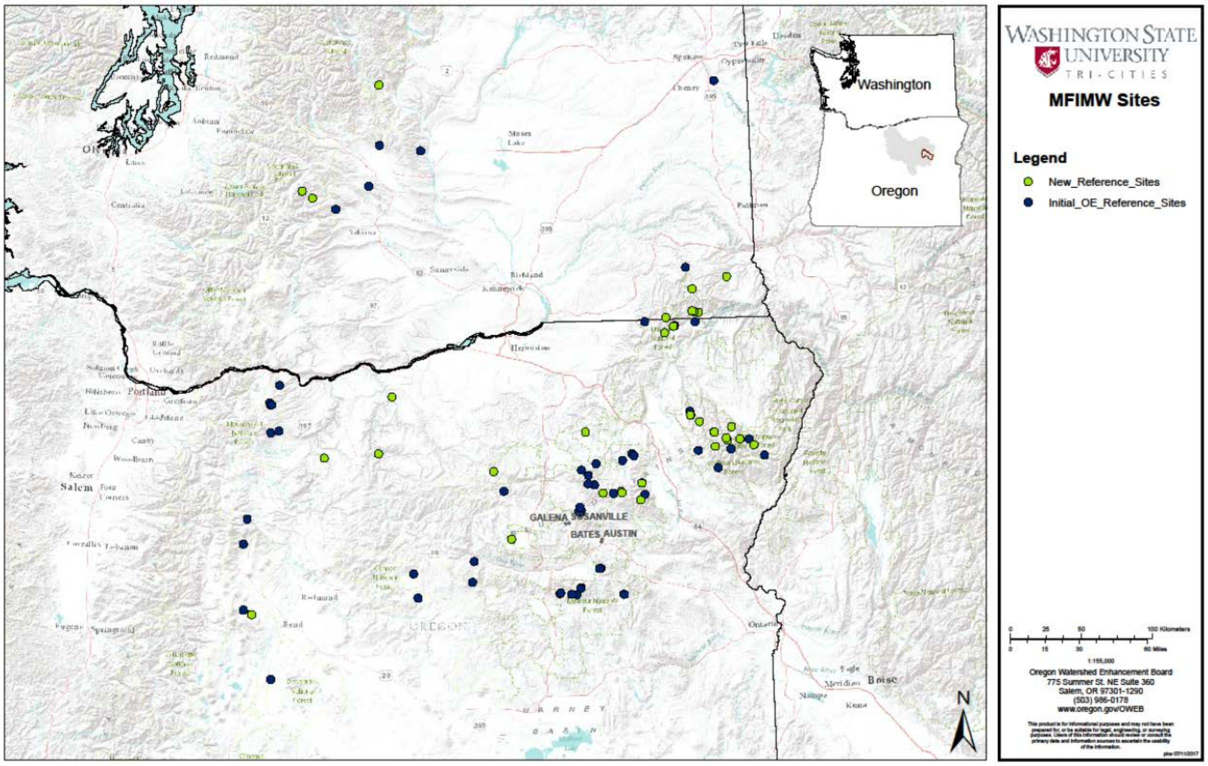

Figure 1. Map showing the new reference sites (n= 83) as well as the reference sites used for the initial O/E model (n=105).

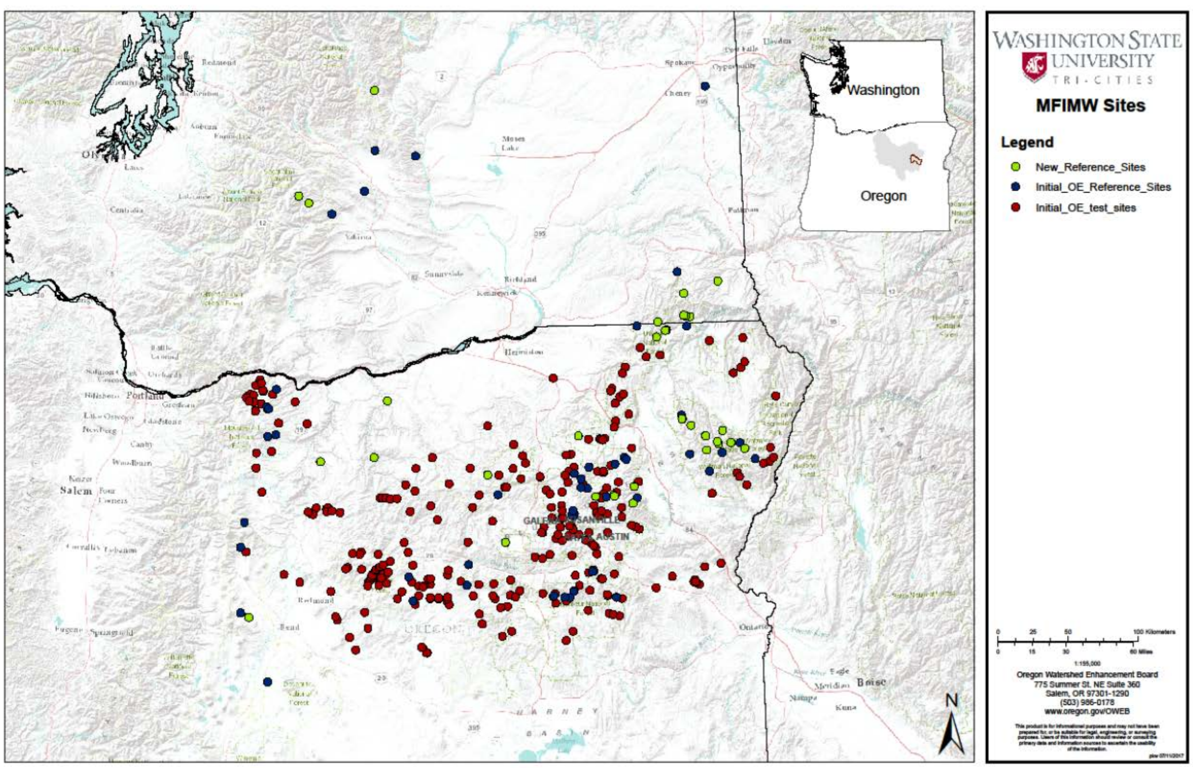

Figure 2. Map showing the reference sites (n= 188) as well as the test sites (n=414) used for model development.

The benthic macroinvertebrate sample dataset used to evaluate the restoration actions in the MFJDR and the associated control stream, the SFJDR, was collected by the NFJDWC from 2010-2015 (Table 1). This dataset was used to calculate the O/E index for the MFJDR & SFJDR post-restoration. The sample sites were randomly selected by the NFJDWC from a set of 15 existing PIBO monitoring stations along the mainstem of the MFJDR. Additionally, 10 sites from the SFJDR were selected by the NFJDWC for a total of 20 sites sampled annually (Figure 3).

Table 1. Description of the data sources and sampling methods. All sources collected samples with equal efforts and consistent taxonomy.

Sampling Source	Sampling Season	Habitat	Sampling Device	Sample Area	Laboratory Subsample	Identification
Model Development Data						
EPA, ODEQ, USFS	Summer low flow	Riffle	D-frame kicknet, Surber sampler; 500 µm mesh	4-8 composited kicks (8 ft.^{2})	500 individuals	Typically Genus/species
Restoration Project Data						
NFJDWC	Summer low flow	Riffle	D-frame kicknet; 500 µm mesh	8 composited kicks (8 ft.^{2})	500 individuals	Typically Genus/species

(EPA 2010, 2014, 2017; Hubler 2013; North Fork John Day Watershed Council 2014; US Forest Service 2015)

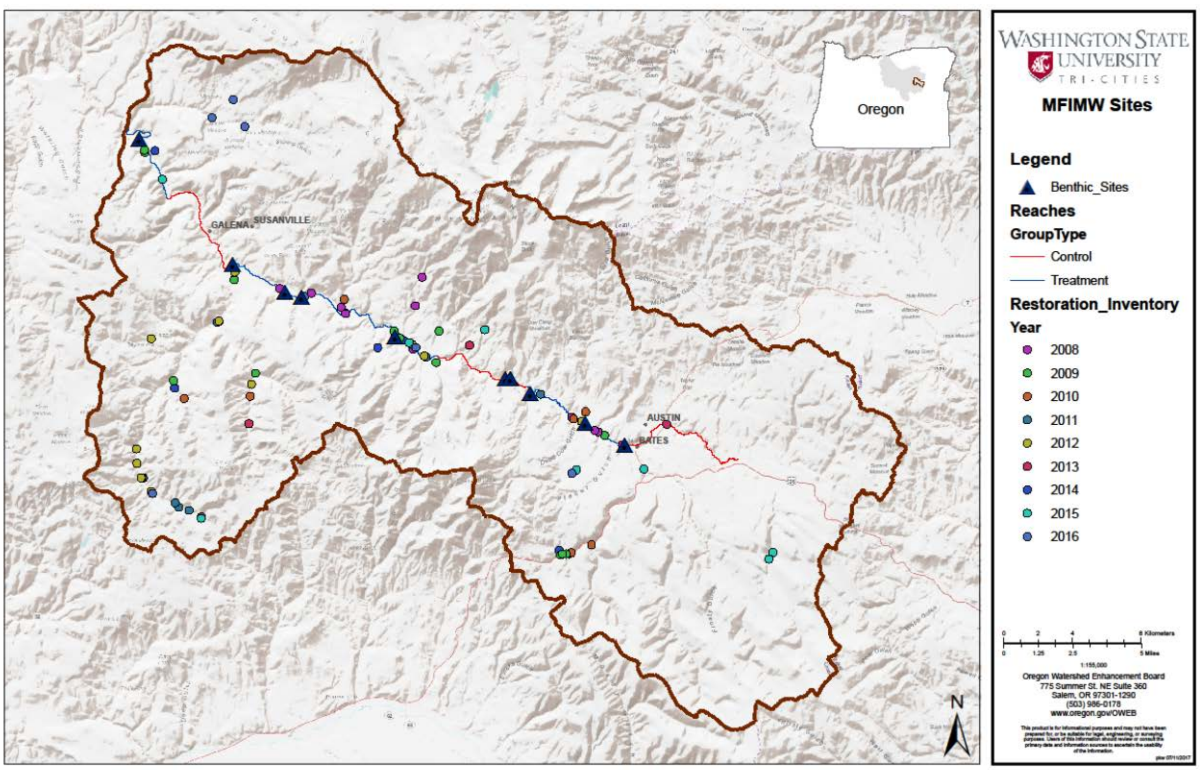

Figure 3. Map of benthic macroinvertebrate sample sites in the Middle Fork John Day River in relation to restoration activities by year.

The drift macroinvertebrate sample dataset of dry weight biomass (g) used in this study was collected by the NFJDWC from 2010-2015 (Table 1). Drift samples were not collected from the SFJDR. The sample sites were randomly selected by the NFJDWC from a set of 15 existing PIBO drift monitoring stations along the mainstem of the MFJDR (Figure 4).

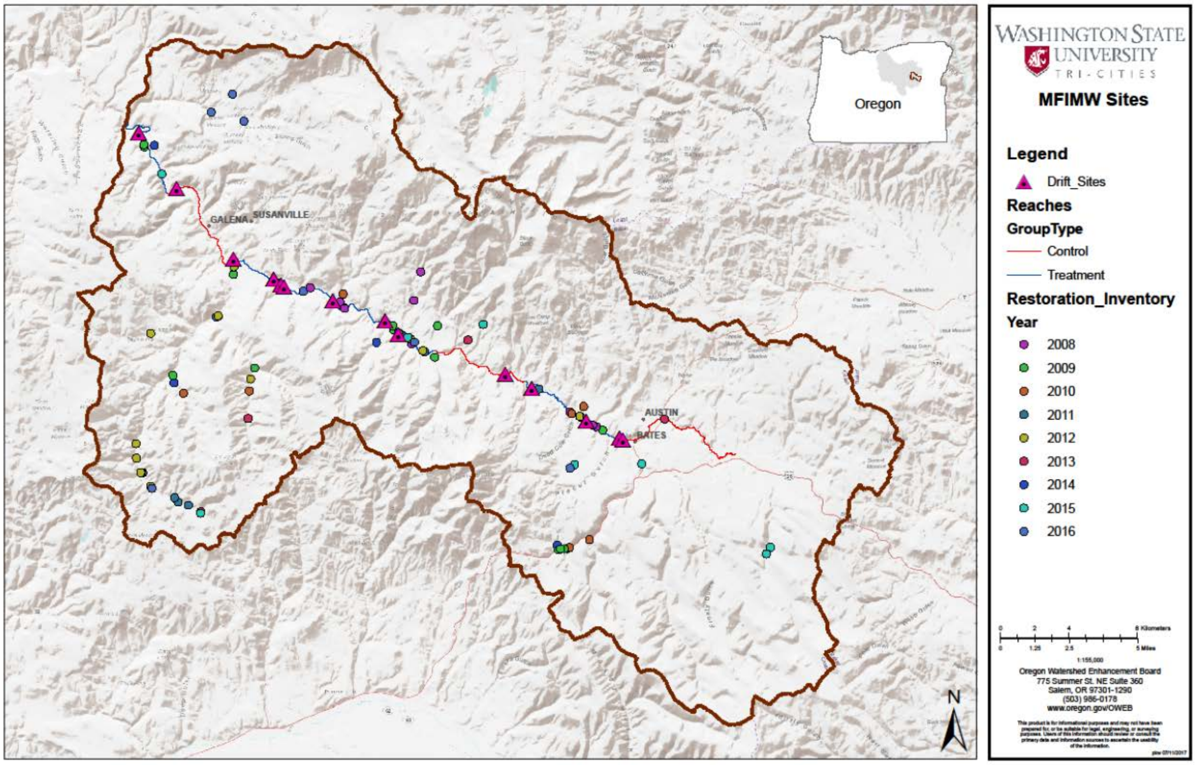

Figure 4. Map of drift macroinvertebrate sample sites in the Middle Fork John Day River in relation to restoration activities by year.

Methods

Sampling Protocols, Collection Periods, & Taxonomy

When using data from multiple sources, a consistent level of sampling effort and taxonomy in terms of sampling protocols (i.e., sampled habitat, number of composite samples and total sampled area) and laboratory procedures (i.e. sample sorting, subsample count level, and taxonomic resolution) is critical, particularly when utilizing inferential statistics (Hubler 2008; Van Sickle et al. 2006). An extensive review of the all data was completed before analysis to make sure the aggregated data from separate sources included the same taxonomic groups, followed the same spelling and abbreviation procedures, and had appropriate taxonomic resolution. Although there are slight differences in sampling methods, benthic macroinvertebrate data assembled for this study were considered to be comparable (Table 1). Although the drift dataset was supplied by a single source, the taxonomy was examined to verify that the same operational taxonomic units (OTUs) were used.

Benthic field sampling was performed at each site using the kick-net sampling protocols as described in Heitke et al. (2008) by the NFJDWC. Each sample consisted of 8 subsamples collected at each site (reach) from 4 riffle habitats using a 500 µm mesh, 0.09 m^2 fixed-area benthic invertebrate sample kick net. The 8 subsamples were composited in the field into sample jars as a single sample and preserved with 80% ethyl alcohol for laboratory identification (Table 1) (Heitke et al. 2008; North Fork John Day Watershed Council 2013).

Drift samples were collected at each site using the sampling protocols as described in Heitke et al. (2008) with a 1000 µm mesh drift net. Drift nets were placed in riffle habitats with a sampling duration of roughly 3 hours and then preserved with 80% ethyl alcohol for laboratory identification (Table 1) (Heitke et al. 2008; North Fork John Day Watershed Council 2013).

Approximately 500 individuals were identified from each benthic and drift macroinvertebrate sample. In general, data for dominant aquatic insect orders were resolved at genus level, less common orders were aggregated to family level, and rare organisms or those with difficult taxonomy were aggregated to order or higher. Samples were resolved to genus using the standard taxonomic effort developed by the Southwest Association of Freshwater Invertebrate Taxonomists (SAFIT) (SAFIT 2016). Resolving the taxonomy was necessary to remove ambiguous taxa and to create a list of OTUs. For the benthic samples utilized in the predictive model to generate O/E scores, the 500 organism count from each sample was randomly sub-sampled to 300 individuals to standardize the sampling effort across samples and even out the effects of differing sample sizes (Ostermiller and Hawkins 2004). Benthic samples collected from reference sites that contained fewer than 200 individuals were excluded from the model building process.

Study Design

We assessed the MFIMW macroinvertebrate communities using a BACI experimental design. The multiple BACI sampling design allowed for monitoring of the short-term (seasonal) variation within the stream while simultaneously tracking longer-term (multi-year) changes likely to be associated with restoration for multiple scales. Concurrent changes at both the control and treatment sites were interpreted as indications of seasonal trends, with the life-cycle of the benthic macroinvertebrates, biotic interactions, or weather as likely causes, while divergent responses were considered to indicate alterations of the stream biota resulting from restoration activities (Meisenbach et al. 2012). The MFJDR is designated as the treatment stream whereas the SFJDR is the control stream, with control stream defined as a stream comparable to the treatment stream with regards to the regional physical, chemical, and biological characteristics (Reynoldson et al. 1997). Additionally, the MFJDR was further divided into treatment reaches (n = 3) and control reaches (n = 3). Each reach contains at least 1 macroinvertebrate sampling site with the exception of the downstream control reach, which has no sample sites (Figures 3 & 4).

It is important to note that while the overall restoration project used a BACI sample design, the initiation of sampling macroinvertebrates in 2010 within the MFIMW began after restoration occurred within the MFJDR, which began as early as 2008. Additionally, some restoration activities occurred in designated control reaches and a few restoration activities occurred upstream of the most upstream control sample sites for macroinvertebrates (Figures 3 & 4). Furthermore, due to the use of inferential statistics and the importance of a balanced sample designs (Ott and Longnecker 2008; Sokal and Rohlf 1995), it is imperative to consider the number of macroinvertebrate sampling sites (n=3)in the three control reaches as compared to the number of macroinvertebrate sampling sites (n=7) in the three treatments reaches.

Objective 1: New Model Development for O/E Index

To test the hypothesis that the accuracy and precision of River InVertebrate Prediction and Classification System (RIVPACS) type models are improved by increasing the number of reference sites used during model development (Bailey et al. 2004; Bailey et al. 2014; Miller et al. 2015), we will utilize benthic macroinvertebrate data from additional reference sites collected by the EPA and USFS to develop new RIVPACS models (n= 84). Once verified that the data from the EPA and USFS was not previously included in the development of the initial RIVPACS model using ArcGIS, the taxonomy of the samples from the new reference sites was resolved into OTUs.

Reference sites were then grouped according to the biological similarities of their sampled macroinvertebrate assemblages. Specifically, reference sites (n=189) were clustered into biologically similar groups using the Sorenson dissimilarity distance measure and flexible beta linkage (β = -0.6) based on the presence and absence of taxa. The resulting reference groups were then evaluated against numerous environmental predictor variables to determine what factors best predicted reference group membership. These predictor variables and associated reference site groups create the basis for predictive models since a test site is assigned a likelihood of belonging to each reference group based on the values of predictor variables. The set of predictor variables that best explained differences in reference groups was determined through random forest (RF) models, shown to produce models that are more robust in predicting biotic integrity compared to the original framework (Clarke et al. 2003; McCune et al. 2002). Predictor variables for the new model development as well as the initial RIVPACS model developed for the MFIMW are provided in Table 2.

Table 2. Predictor variables used for O/E model development.

Variable	**Description**
Site	
Baseflow index	The ratio of base flow to total flow, expressed as a percentage, at the sample site.
Distance to dam, fish barriers, and NPDES discharge points	Linear distance to the nearest dam, fish barriers, and NPDES discharge points.
Elevation	Elevation of the sampling site in meters.
Annual precipitation	Mean annual precipitation at the sampling site in millimeters.
Annual mean temperature	Mean annual air temperature at sampling site in degrees Celsius.
Julian date	The day the sample was collected in the year (Day 1 to Day 365).
Forest fragmentation type	Types of fragmentation (interior versus the exterior of a forest patch).
Land cover diversity	Relative diversity of a point, considering both natural and anthropogenic land cover types.
Stream gradient	Elevation change over the mapped sampling reach length divided by the reach length.
Watershed	
Road density	Road density per watershed.
Percent land use land cover type per HUC12	Percent of watershed area (HUC12) designated as agriculture, forest, shrubland and scrubland, wetlands, and urban.
Percent land use land cover type per 10m riparian buffer per HUC12	Percent of 10 m riparian buffer per HUC12 designated as agriculture, forest, shrubland and scrubland, wetlands, and urban.

O/E was calculated for the probability of capture (P_C) threshold of $P_C >$ 0.5 as these models result in more precise O/E scores which are more sensitive to stress (Hawkins 2006; Hawkins et al. 2010a; Hawkins et al. 2000; Hawkins et al. 2010b; Ostermiller and Hawkins 2004). The RF models were calculated using R software, the code and packages supplied by the ODEQ (Hubler 2013; R Development Core Team 2014). RF models were built by repeatedly applying a randomly selected subset of the predictor variables to a randomly selected subset of the samples with a bootstrapping-type process repeated many times to generate a forest of decision trees. Box and whisker plots were used to determine the optimal number of predictor variables (mtry) that were randomly sampled as candidates at each split and 1,000 trees (ntree) were grown.

The benchmarks for describing the biological condition of a sample were established using the distribution of reference site O/E scores. Specifically, benchmarks were based on the 10th, 25th, and 95th percentiles of reference site O/E distributions. These benchmarks were chosen so as to balance the potential of identifying a sample as disturbed when it isn't (type I error) with failing to recognize biological disturbance when it exists (type II error) (Hawkins et al. 2010a).

For biotic indices to make reliable inferences of ecological condition, predictions of the reference condition must be acceptably accurate, precise, unbiased, responsive, and sensitive to stressors (Hawkins 2006; Hawkins et al. 2010a; Hawkins et al. 2000; Hubler 2008). Therefore, we selected the best performing model (i.e., between the newly developed models and the initial RIVPACS model) based upon these measures (Table 3) using a numerical ranking system, with the best performing model given the score of 1. If a metric consisted of two or more parts (e.g., responsiveness includes Students t statistics produced from comparing reference and test sites O/E scores, the 95% confidence limits from each model, and the magnitude of difference between reference OE and test OE scores for each model; Table 3), then the average was used. If a tie was produced, particularly for the model performance metric sensitivity with only one measure (Table 3), then the average rank for that value was assigned.

Table 3. Metrics used to evaluate model performance.

Metric	Description
Accuracy	1. 10-fold crossvalidation. 2. Slope of O versus E of calibration sites.
Precision	1. Standard deviation of calibration O/E scores. 2. Value of r^2 when regressing O on E for calibration sites.
Bias	1. Comparison of calibration versus validation mean O/E score for unity. 2. Regressing O/E on the predictor variables.
Sensitivity	1. % of test sites below the 10th percentile of reference site O/E scores.
Responsiveness	1. Student's t value estimated from two sample t-tests of reference and test site O/E scores. 2. Magnitude of difference between the reference and test site O/E scores. 3. Upper and lower 95% confidence limits.

(Hawkins 2006; Hawkins et al. 2010a)

Objective 2: Trends in MFJDR Benthic Macroinvertebrate Communities, Post-Restoration

ANOVAs and multiple comparisons tests were used to determine if O/E scores for the MFJDR have changed over time following restoration using a BACI sampling design. The BACI sampling design allowed for monitoring of the short-term variation within/between streams while simultaneously tracking longer-term changes likely to be associated with restoration for multiple scales. Concurrent changes at both the control and treatment sites/reaches/streams were interpreted as indications of seasonal trends, with the life-cycle of the benthic macroinvertebrates, biotic interactions, or weather as likely causes, while divergent responses were considered to indicate alterations of the stream biota resulting from restoration activities (Meisenbach et al. 2012). Additionally, since restoration may increase the variance of O/E scores in the short-term and perhaps minimize the variance in the long-term, the coefficient of variation (CV) for the O/E scores was evaluated.

Objective 3: Trends in MFJDR Drift Macroinvertebrate Communities, Post-Restoration

Drift macroinvertebrates play a role in streams with important management implications for drift-feeding fishes like salmonids (Elliott 1967; Pringle and Ramírez 1998; Rios-Touma et al. 2012; Waters 1972) since restoration has the potential to increase the amount of drift macroinvertebrates in the water column. Consequently, the study of drift macroinvertebrates is a complimentary component of bioassessment when evaluating stream restoration in salmonid habitat. As such, we evaluated how drift macroinvertebrate biomass (g) and taxa richness may have changed post-restoration using ANOVAs and multiple comparisons tests following a BACI sampling design, similar to the benthic macroinvertebrate analysis. Additionally, since restoration may increase the variance of drift macroinvertebrate biomass (g) and taxa richness in the short-term and perhaps minimize the variance in the long-term, the CV for drift biomass and taxa richness was evaluated.

Results

Objective 1: O/E Model Development

Distribution of Reference and Test Sites

Both reference and test sites occurred across a range of environmental settings with differences between reference and test sites in their distribution (Figures 1 & 2). For both the new models and the initial model developed for the MFIMW, over 80% of reference sites were located in the Blue Mountains (BM) ecoregion, with 11% in the Columbia Plateau (CP) ecoregion, and 8% in the Eastern Cascades and Foothills (ECF ecoregion). For test sites, the BM ecoregion contained 87% of sites while the CP ecoregion contained 2% and the ECF contained 10%.

Assessing candidate predictor variables provide insight into the specific niche axes that control macroinvertebrate assemblages, therefore the reference and test site predictor variables of naturally occurring factors were evaluated (reach slope, elevation, temperature, Julian day, precipitation, and baseflow) with significant differences between the test sites and reference sites for both the new reference sites used in model development and the sites used with the initial model development ($t = -17.02$ to 10.08; $p < 0.10$). All reference sites had a greater slope compared to test sites, a higher elevation, and more precipitation using two sample t tests ($t = -1.97$, 6.07, 10.08; $p < 0.10$). Additionally, the reference sites tended to be sampled later in the year than test sites, as measured by the Julian date, and reference sites tended to occur in lower temperatures ($t = 2.31$, -17.02; $p < 0.10$).

Stream Taxonomic Richness and Composition

With the combined data of new reference sites plus the existing reference and test sites from the initial O/E model, 208 OTUs in 82 families were recognized from the study sites, of which the family Chironomidae represented the largest taxonomic group (26% of all OTUs). Sample OTU richness varied approximately six-fold among reference sites (7–44) and approximately nine-fold across test sites (5–47). Only 17 non-insect OTUs were collected and identified, although 8 of these OTUs were only identified to class or higher.

For the data used in the initial model only, 201 OTUs in 81 families were recognized from the study sites. Again, the family Chironomidae represented the largest taxonomic group (26% of all OTUs). Sample OTU richness varied approximately three-fold among reference sites (14–43) and approximately four-fold across test sites (9–43). Also, like the new models, only 17 non- insect OTUs were collected and identified.

Model Development

From the Sorenson cluster analysis that grouped reference sites according to the biological similarities of their sampled macroinvertebrate assemblages, we selected two group sizes (i.e., 2 groups and 4 groups) to use for new RF model development (Figure 5). The number of sites per group ranged from 67 to 121 and 25 to 73 for the 2 and 4 group classifications, respectively, and the initial model containing 12 to 50 sites in each of the 3 groups. When relating the biological data to the predictor variables with random forest models, the relative importance of each predictor variable (how frequently each predictor is selected in individual trees) varied considerably between all models (Table 4). For both the new RF models as well as the initial model developed, one common predictor variable was selected among the top 5 predictor variables, elevation. However, there were several common variables that were grouped in the top 10 predictor variables for all three models (nearest NPDES, elevation of the sample site, % forest in 10 m riparian buffer per HUC12, % shrubland/scrubland in 10 m riparian buffer per HUC12, and mean annual temperature).

Table 4. Predictor variable importance. ● Predictor variable is in the top 5 variables for the model. O Predictor variable is in the top 10 variables for the model. ○ Predictor variable is not in the top 10 variables for the model.

Predictor Variable	RF 2 Groups Model	RF 4 Groups Model	Initial MFIMW O/E Model
Baseflow	O	○	O
Distance_Dam_m	O	●	○
Distance_NPDES_m	O	●	O
ELEV_m	●	●	●
PercForest_10mBuffer	●	O	●
PercForest_HUC12	O	○	O
PercShrub_Scrub_10mBuffer	O	O	●
PercShrubland_HUC12	○	●	O
PercUrban_10mBuffer	○	○	●
PercUrban_HUC12	○	○	●
Precip_mm	●	●	○
Road_density_km_km2	●	O	○
Slope	○	O	○
Temp_C	●	O	O

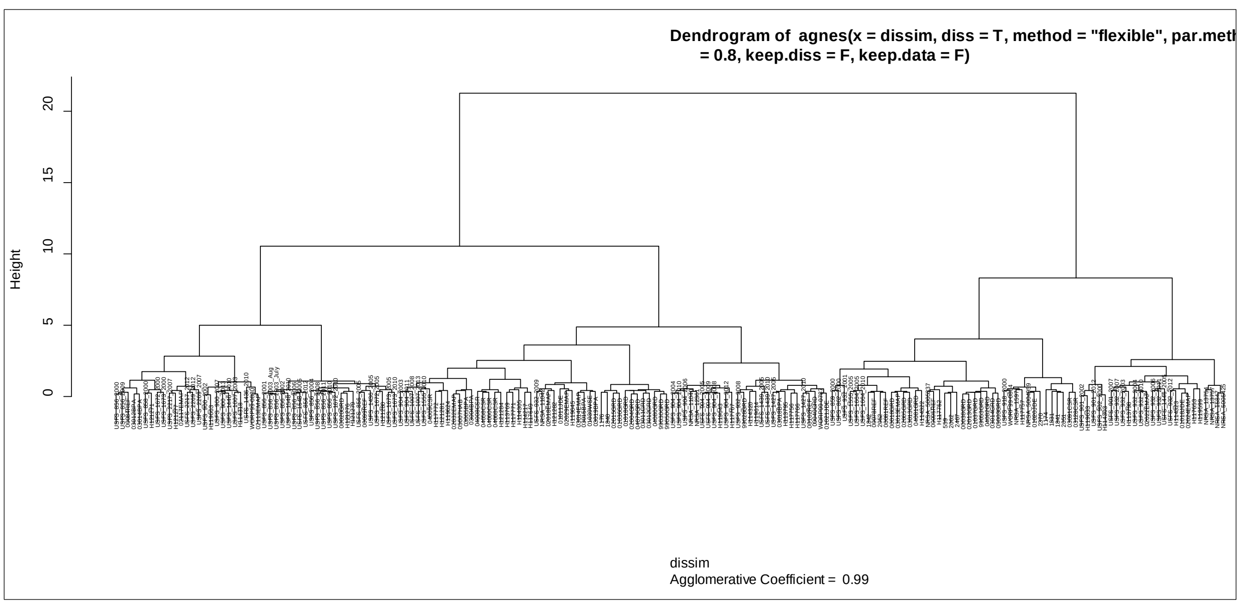

Figure 5. Cluster diagram of reference sites (n = 188) using Sorenson dissimilarity measure and flexible beta linkage (β = -0.6) based on the presence and absence of taxa. The dendrogram was pruned to yield two different group size classifications, comprised of 2 and 4 groups, which produced reference group sizes of ≥ 5 sites and maximized within group fidelity.

Model Comparisons

Accuracy

The accuracy of O/E indices depends in part on how well the mechanics of index calculation account for the effects of these natural gradients on assemblage structure (Hawkins 2006; Van Sickle et al. 2005). Evaluations of group membership predictions using 10-fold cross validation of the new RF models as well as the initial model show that the new 2 group RF model had a greater classification accuracy rate compared to the initial model and the new 4 group model (86%, 83%, and 73%, respectively; Table 5). Comparisons of the 10-fold crossvalidation accuracy rates using two sample t-tests show that the models with the 2 group sizes classification have a statistically higher accuracy rate compared to the initial O/E model and the new 4 group model, which were significant with $p < 0.10$ and $n = 100$. Furthermore, the scatter plot generated for each model of O versus E generally produced a 1:1 regression line; however, the initial model produced less scatter and intercepts closer to zero. Nonetheless, based on the 10-fold cross validation accuracy rates, the new 2 group RF model was more accurate than the initial model and the new 4 group model. The accuracy statistics for each model are shown on Table 5.

Table 5. Summary statistics of the three O/E models. Note: * significant at $p < 0.10$.

Metric	RF 2 Groups Model	RF 4 Groups Model	Initial MFIMW O/E Model
Calibration Sites			
10-fold Crossvalidation	0.86	0.73	0.83
Predictive Model Mean O/E	1.01	1.02	1.02
Predictive Model SD O/E	0.19	0.18	0.15
Null Model Mean O/E	1.00	1.00	1.00
Null Model SD O/E	0.20	0.20	0.17
R^2	0.30	0.51	0.68
Validation Sites			
Predictive Model Mean O/E	1.01	1.02	1.02
Predictive Model SD O/E	0.19	0.18	0.15
Test Sites			
Predictive Model Mean O/E	0.96	0.94	0.91
Predictive Model SD O/E	0.20	0.19	0.17
Predictive Model UCL 95% Mean O/E	0.98	0.96	0.93
Predictive Model LCL 95% Mean O/E	0.94	0.92	0.90
Sensitivity	0.12	0.12	0.24
Student's *t*	2.75*	4.64*	6.51*

Precision

Estimates of index precision varied from SD = 0.15 to 0.19, and the 95% confidence intervals for the SDs for each model were overlapping; however, the initial RF model produced the lowest SD of 0.15 (Table 5). None of the models produced O/E scores which approached or achieved the precision associated with estimated sampling error (i.e., SD = 0.11). The r^2 values produced from a regression of O versus E predicted from each new RF model were less than the r^2 value produced from the initial O/E model; thus, the initial model developed for the MFIMW appears to be more precise than the new models (Table 5). The statistics for precision metrics of each model are shown on Table 5.

Bias

The distribution of O/E values derived from both the calibration and validation reference sites were normally distributed for all models. Likewise, the distributions of O/E values derived from the calibration reference data were statistically indistinguishable compared to the validation data from each model. Moreover, mean O/E values for all models were very close to unity, which implies that all models generated stable, unbiased estimates of error. Despite this, when evaluating the O/E scores for reference sites from each model in comparison to various, non-anthropogenic influenced predictor variables (i.e., reach slope, elevation, temperature, level 3 ecoregion, Julian day, and precipitation), the initial model showed a greater relationship with the variables (r^2 value > 10%) as opposed to the new 2 group and 4 group models (r^2 value < 5%). Consequently, the 2 new models performed better with regards to bias as compared to the initial model developed. The statistics for the bias of each model are shown on Table 5.

Responsiveness

Mean O/E index values estimated at test sites between each model differed by 0.5 standardized units among models (Table 5). The initial O/E model produced lower O/E scores compared to the new 2 and 4 group models, and were thus more responsive based on this measure. The 95 percent confidence intervals for test site mean O/E scores also implied that there were differences in responsiveness between the models, with the 2 group models having the highest upper confidence interval and the initial O/E model possessing the lowest confidence interval. While there was some overlap between the intervals of the 2 new models, the initial model did not overlap at all with the 2 groups model as it possessed the lowest mean and confidence interval (Table 5). Estimates of the Student’s t statistic also implied models differed in their responsiveness ($t = 2.75–6.51$; $p < 0.10$). The new 4 group model and the initial O/E model produced the second highest and the highest Student’s t statistic ($t = 4.64$ and 6.51 respectively; $p < 0.10$), while the new 2 groups RF model produced the lowest Student’s t statistic ($t = 2.75$; $p < 0.10$) thus indicating that this model responded to the differences in predictor variables between the reference sites and test sites better. Since the 3 models we examined varied in their responsiveness, it is clear they characterize the same stressed assemblages in different ways; however, based on the three measures of responsiveness for predictive models provided in Table 5, the new 2 and 4 group models tended to measure responsiveness similarly while the initial model was more responsive (Table 5).

Sensitivity to Stressors

The percent of test sites assessed as most disturbed based on the tenth percentile threshold criterion differed twofold between the new models and the initial model (Table 5). The McNemar tests showed that the new models both were significantly different from the initial model ($p < 0.10$), but not each other ($p = 0.76$). The initial O/E index was most sensitive (24% of test sites) while the new 2 groups and 4 groups models were less sensitive (12% of test sites). The statistics for the sensitivity of each model are shown on Table 5.

Final Model Selection

The final model was selected by using a numerical ranking system from 1-4, with the best performing model given the score of 1 for each metric. After tabulating the results, the new RF 2 group and 4 group models were ranked very similarly compared to the initial O/E model (Table 6); however, the initial O/E model outperformed the new models overall with slight tradeoffs in accuracy and bias. While accuracy is important in regional bioassessments (Ode et al. 2008), the initial O/E model possessed a 10-fold cross validation rate approximating the rate produced from the new 2 groups model (83% and 86%, respectively) while simultaneously achieving the best precision, responsiveness, and sensitivity; therefore, we selected the initial O/E model as the final model (Table 6).

Table 6. The rank of each model, as determined through accuracy, precision, bias, sensitivity, and responsiveness.

Metric	RF 2 Groups Model	RF 4 Groups Model	Initial MFIMW O/E Model
Accuracy	1.8	2.8	2.3
Bias	2.3	1.8	2.8
Precision	3.0	2.0	1.0
Sensitivity	2.8	2.3	1.0
Responsiveness	2.3	2.0	1.7
Rank Sum	**12.1**	**10.8**	**8.7**

Objective 2: Trends in MFJDR Benthic Macroinvertebrate Communities, Post-Restoration

MFJDR v. SFJDR

As determined using ANOVAs and multiple comparisons tests with years and the control/treatment streams as factors, there were not any significant differences in O/E scores among the control and treatment streams ($p = 0.78$). However, significant differences in O/E scores were detected among years ($p < 0.10$). Specifically, multiple comparisons showed that there were differences in 2010 & 2013 ($p < 0.10$). A number of restoration projects occurred in both 2010 and 2013, but more restoration projects were completed in 2012 (Figure 6). The year 2012 also had the smallest 95% confidence interval (0.87-0.96), with the years 2013 and 2015 producing the largest 95% confidence interval (0.83-1.01, 0.80-0.98, respectively). Figure 7 summarizes the distributions of O/E scores for the MFJDR and SFJDR.

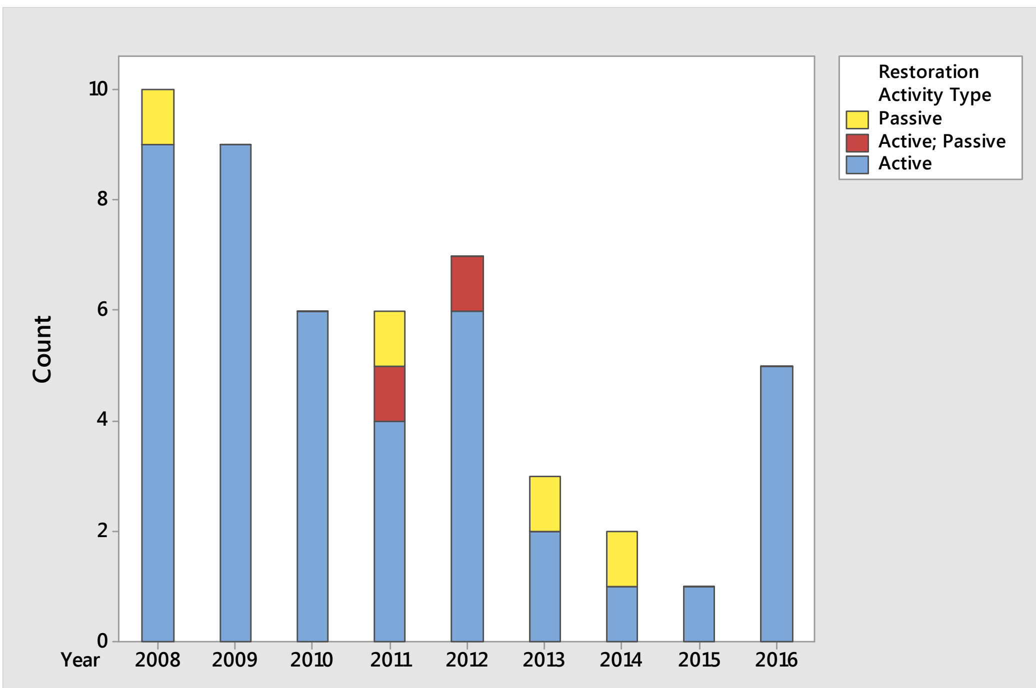

Figure 6. Types and amount of restoration activities in the treatment stream, the Middle Fork John Day River. Restoration activities were typically confined to the treatment sites, although some restoration projects (i.e., fish passage, riparian exclusion fencing) extended between treatment and control reaches.

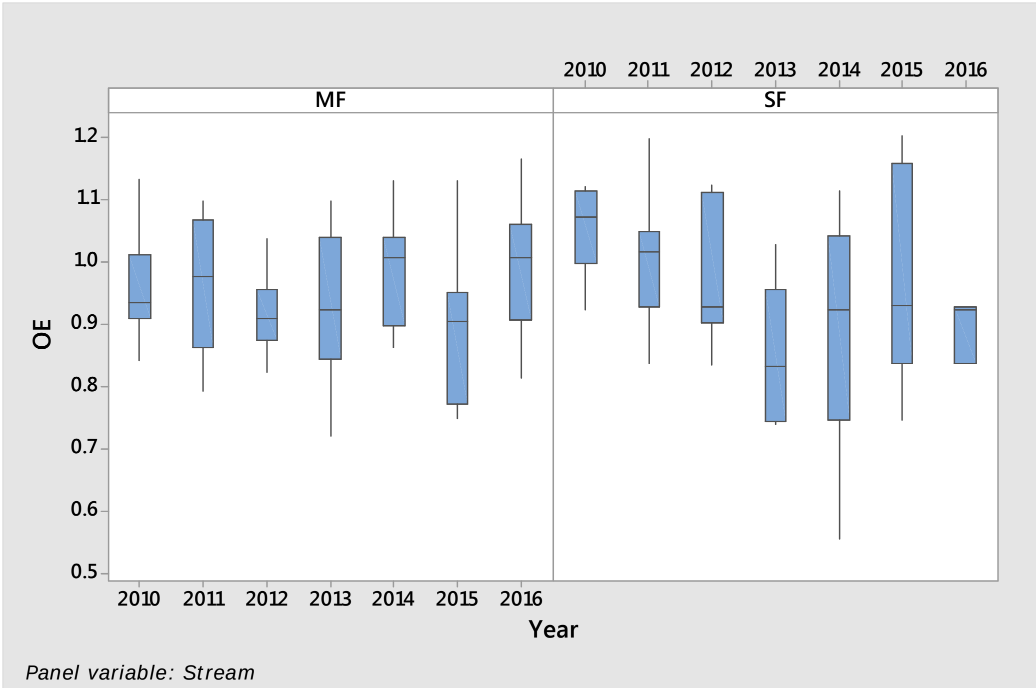

Figure 7. Box and whisker plots of MFJDR and SFJDR O/E scores.

The benchmarks of biological classification (e.g., least disturbed, moderately disturbed, and most disturbed, and enriched, Table 7) varied between streams and years. Generally, the number of MFJDR sites designated least disturbed, moderately disturbed, and most disturbed were similar compared to the number of SFJDR sites for the same benchmark; however, of particular note are the years 2010 and 2013 which were significantly different from all other year based on the ANOVAs and multiple comparisons tests. In 2010, the O/E index benchmark classifications predicted that of the MFJDR sites, 50% were in moderately disturbed and 50% in least disturbed biological condition while the benchmark classifications for the SFJDR were 20% were in moderately disturbed and 80% in least disturbed biological condition. For the year 2013, 20% of the MFJDR sites were in most disturbed biological condition, 30% of the MFJDR sites were in moderately disturbed, and 50% in least disturbed biological condition while the benchmark classifications for the SFJDR were 30% most disturbed, 40% were in moderately disturbed, and 30% in least disturbed biological condition. The benchmarks for each stream by year are shown on Figure 8.

Table 7. The benchmarks of biological condition used to describe the biological condition of a sample (Henderson 2014; Hubler 2008).

Biological Condition Class	**Reference percentile**	**RF 2 Groups Model**	**RF 4 Groups Model**	**Initial MFIMW O/E Model**
Most disturbed	≤ 10th	≤ 0.74	≤ 0.74	≤ 0.81
Moderately disturbed	> 10th to 25th	0.74 to 0.87	0.74 to 0.89	0.81 to 0.93
Least disturbed	> 25th to 95th	0.87 to 1.22	0.89 to 1.24	0.93 to 1.22
Enriched	> 95th	> 1.22	> 1.24	> 1.22

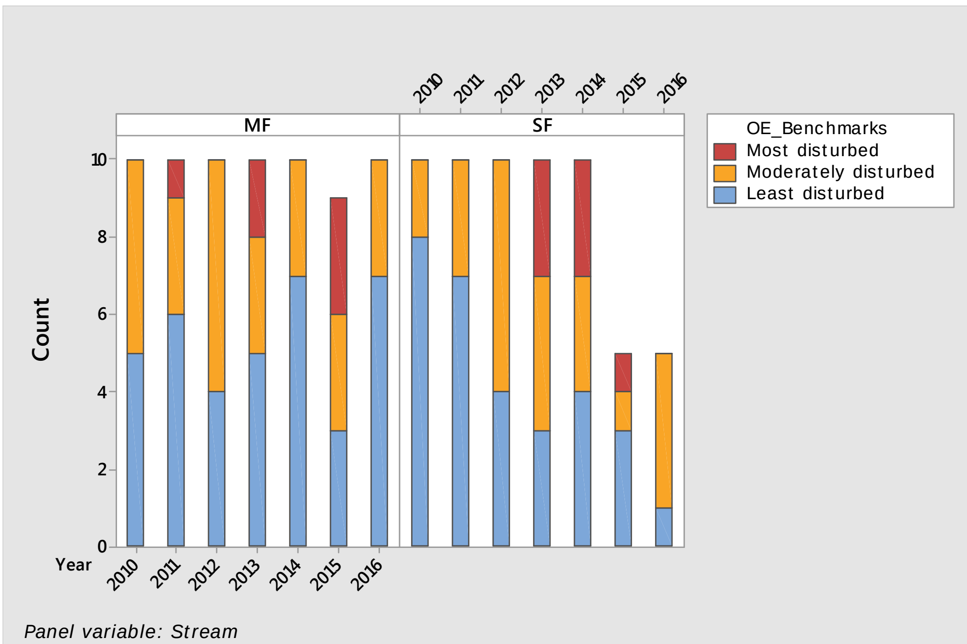

Figure 8. O/E score biological benchmarks for MFJDR & SFJDR.

The MFJDR experienced trends in the coefficient of variation (CVs) of O/E scores that were dissimilar from the trends in the SFJDR CVs, the regional control stream. In fact, the CV for SFJDR O/E scores increased from 2010-2014, but then decreased from 2015-2016. In contrast, the MFJDR CV increased then decreased from year to year, but with an overall increasing trend in the CV. The year 2012 experienced the lowest CV in the MFJDR yet had the most restoration projects occur that year. Figure 9 summarizes the CVs for the MFJDR and SFJDR in relation to O/E scores and the number of restoration projects.

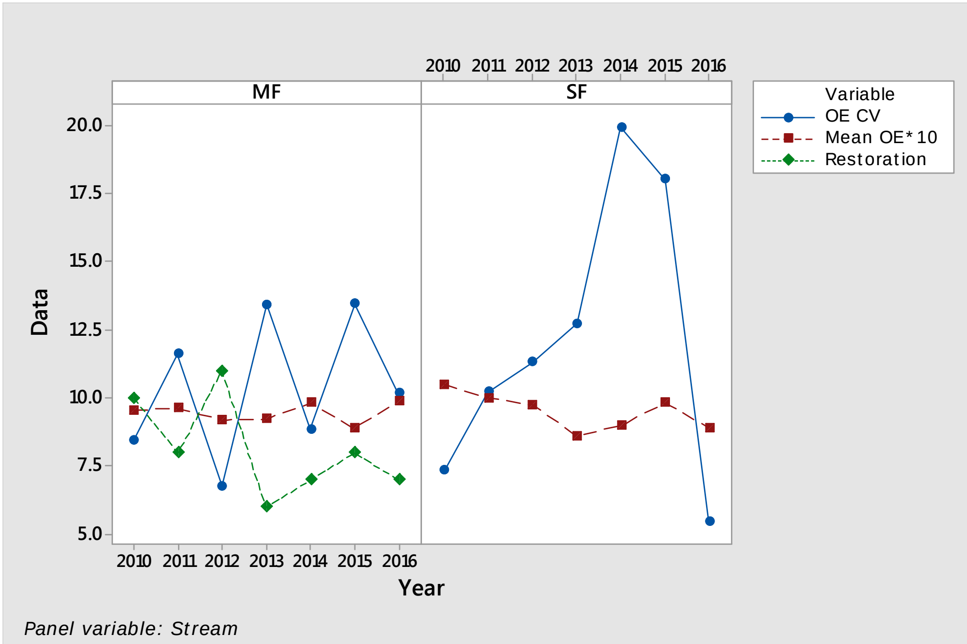

Figure 9. Line plots of mean MFJDR O/E Scores, CV of OE scores, and the number of restoration projects by stream and year. Mean O/E was multiplied by a factor of 10 to allow for comparisons.

MFJDR treatment reaches v. MFJDR control reaches

Comparisons of O/E scores using years and the control/treatment reaches as factors with ANOVAs and multiple comparisons tests do not indicate that restoration events have significantly affected the biotic integrity of the MFJDR. In fact, the ANOVAs and multiple comparisons tests did not detect significant differences in either years or reaches. That being said, the treatment reaches produced relatively stable 95% confidence intervals while they were broader for the control reaches by year. Figure 10 summarizes the O/E score distributions for the MFJDR control and treatment reaches graphically.

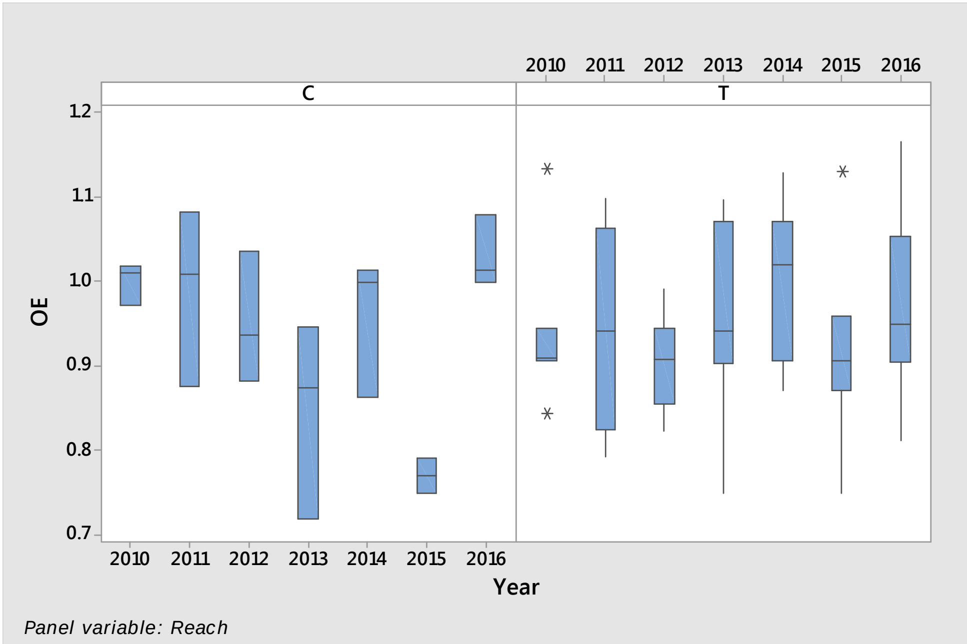

Figure 10. Box and whisker plots of O/E scores for MFJDR control and treatment reaches.

The benchmarks of biological classification fluctuated remarkably between MFJDR control and treatment reaches each year. For instance, in 2010, 100% of the control sites were in least disturbed condition while for the treatment sites, 71% were in moderately disturbed condition and 29% were in least disturbed condition. In 2015, 100% of the control sites were in most disturbed condition while for the treatment sites, 43% were in least disturbed condition, 43% were in moderately disturbed condition, and 14% in most disturbed condition. Since 2015 experience periods of low discharge and high temperature (Henderson 2017; North Fork John Day Watershed Council 2017), it is likely that these benchmarks are a reflection of the climate conditions. However, it is interesting to note that the treatment reaches were able to withstand the climate conditions better than the control reaches. The MFJDR benchmarks for treatment and control reaches by year are shown on Figure 11.

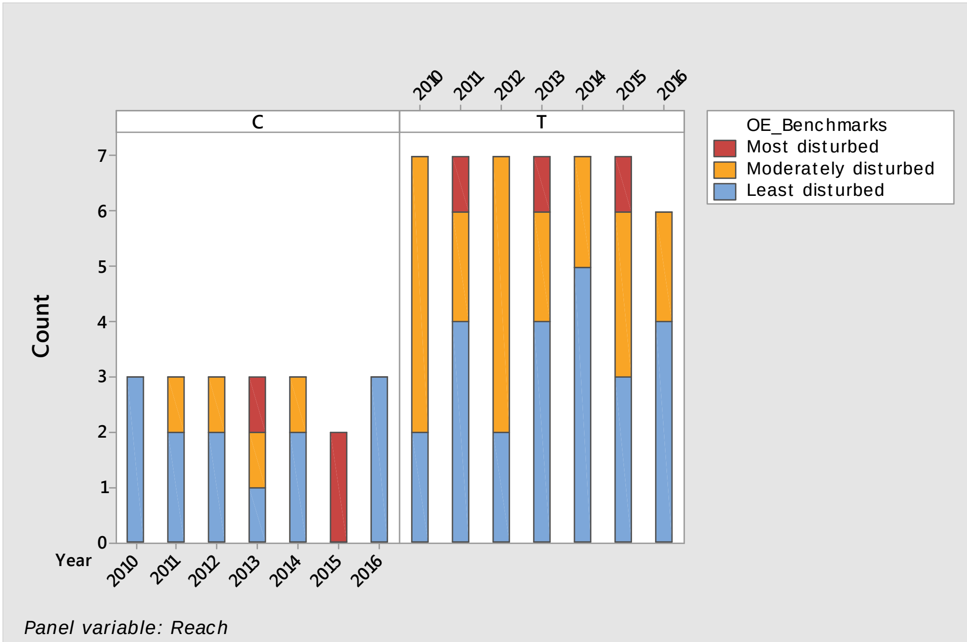

Figure 11. O/E score biological benchmarks for MFJDR control and treatment reaches.

Trends in CVs of O/E scores between the MFJDR control and treatment reaches were very similar, with the prime difference being that the treatment reaches had a higher CV each year except 2012 and 2013 compared to the control reaches. As previously mentioned, the MFJDR had the greatest amount of restoration activities in 2012 while 2013 had the fewest. The greatest CV for the treatment reaches occurred in the year 2011, a year the MFJDR experienced the second highest amount of restoration, including channel reconfigurations in the downstream and middle treatment reaches (Figures 10-12). The highest CV for control reaches was in the year 2013, when 2 restoration activities occurred within the reaches. Figure 12 summarizes the CVs for the MFJDR control and treatment reaches in relation to O/E scores and the number of restoration projects.

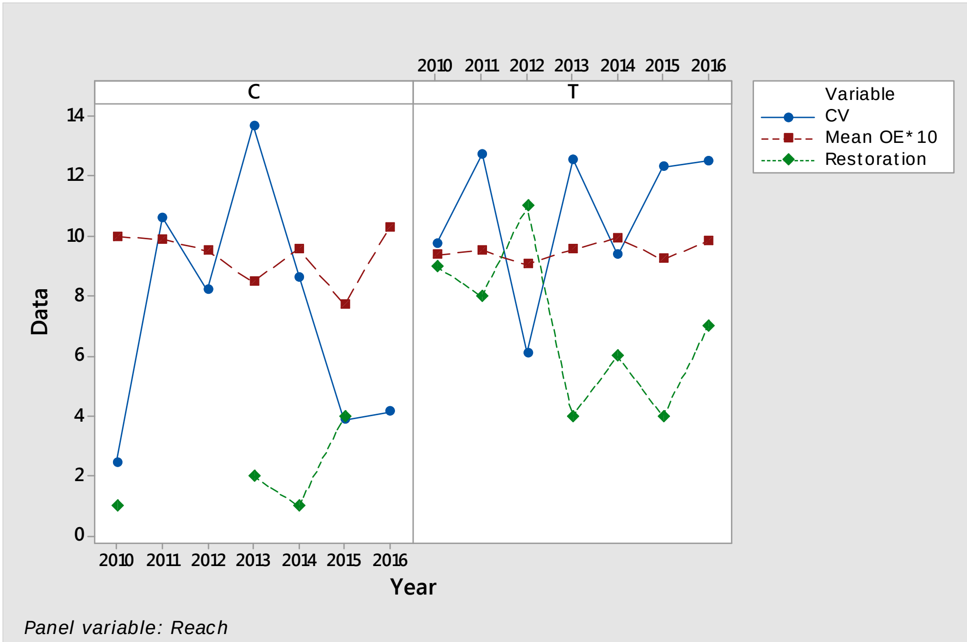

Figure 12. Line plots of mean MFJDR O/E Scores, CV of OE scores, and the number of restoration projects by stream and year. Mean O/E was multiplied by a factor of 10 to allow for comparisons.

Objective 3: Trends in MFJDR Drift Macroinvertebrate Communities, Post-Restoration

Drift Biomass

Restoration events appeared to affect the amount of drift macroinvertebrate biomass within the MFJDR, as determined using ANOVAs and multiple comparisons tests with years and control/treatment reaches as factors. While there were not any significant trends between reaches ($p = 0.74$) for biomass, years were significant in the comparison ($p < 0.10$) with the year 2011 significantly different from all years except 2010. In the year 2011, the MFJDR experienced the second highest amount of restoration. The year 2011 generated the largest 95% confidence interval (0.019-0.122) for the treatment reaches while the year 2014 generated the largest 95% confidence interval (0.019-0.122) for the control reaches. Figure 13 summarizes the drift macroinvertebrate biomass distributions for the MFJDR.

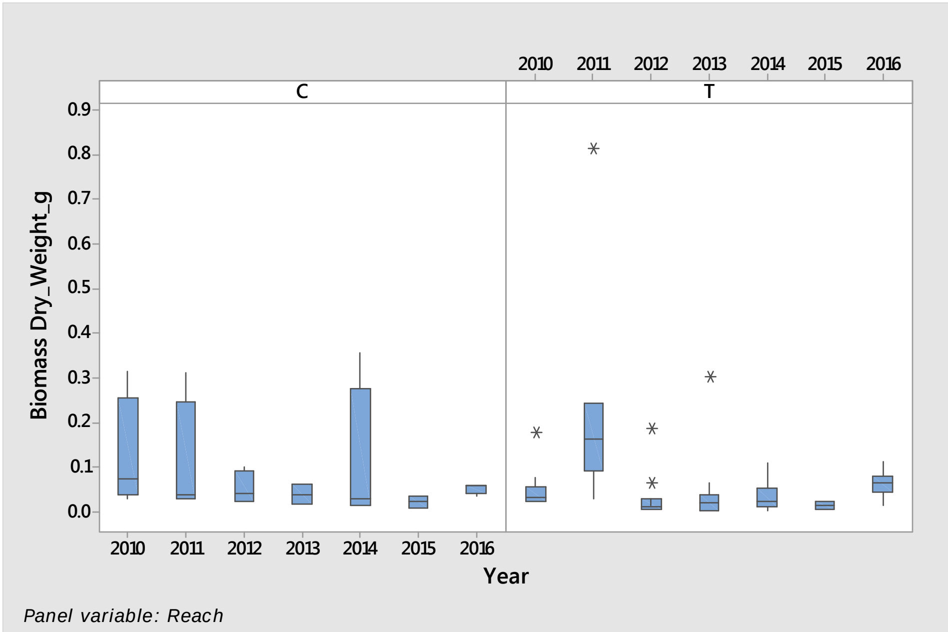

Figure 13. Box and whisker plots of MFJDR biomass dry weight (g) by year.

Trends in the biomass CV between the MFJDR control and treatment reaches were highly variable. In fact, the biomass CVs for both the control and treatment reaches synchronously increased from 2010-2011 but diverged in 2012 when the treatment reached CV continued to increase until 2014. Figure 14 summarizes the CVs for the MFJDR treatment and control reaches.

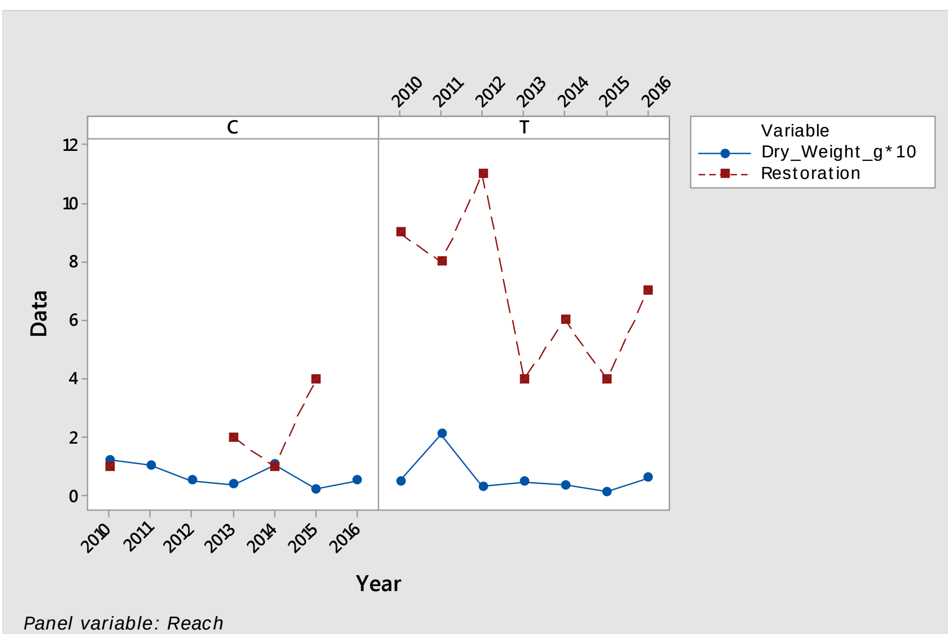

Figure 14. Line plots of MFJDR biomass dry weight (g) CVs and restoration by year and control/treatment reach. Dry weight was multiplied by a factor of 10 to allow for comparisons.

Taxa Richness

In contrast to drift biomass, there were significant differences between years and reaches ($p < 0.10$) for drift taxa richness using ANOVAs and multiple comparisons tests, with treatment reaches experiencing a lower taxa richness ($p < 0.10$). In the comparisons, the years 2013, 2014, and 2016 were significantly different than the years 2010, 2011, 2012, and 2015. During the year 2016, the treatment reaches within the MFJDR experienced the highest mean taxa richness (28) with the lowest mean taxa richness experienced in the year 2013 (12). In the MFJDR control reaches, the highest taxa richness also occurred in 2016 while the lowest was in 2015. Generally, mean taxa richness values in comparison to restoration activities between control and treatment reaches appeared to trend with the number of restoration activities per year. Figure 15 summarizes the drift taxa richness distributions for the MFJDR.

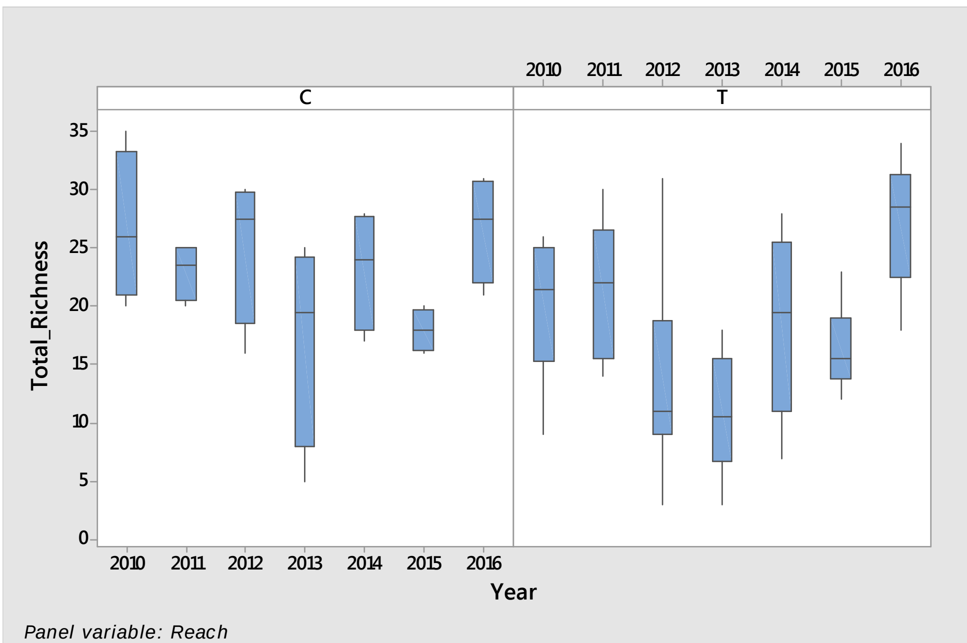

Figure 15. Box and whisker plots of drift taxa richness for MFJDR control and treatment reaches.

Trends in the drift taxa richness CV between the MFJDR control and treatment reaches also appeared to trend with the number of restoration activities per year. The highest CV in the MFJDR treatment reaches was in 2012 while the lowest was in 2016. For the control reaches, the highest CV was in 2013 while the lowest was in 2015. Figure 16 summarizes the CVs of drift taxa richness for the MFJDR treatment and control reaches.

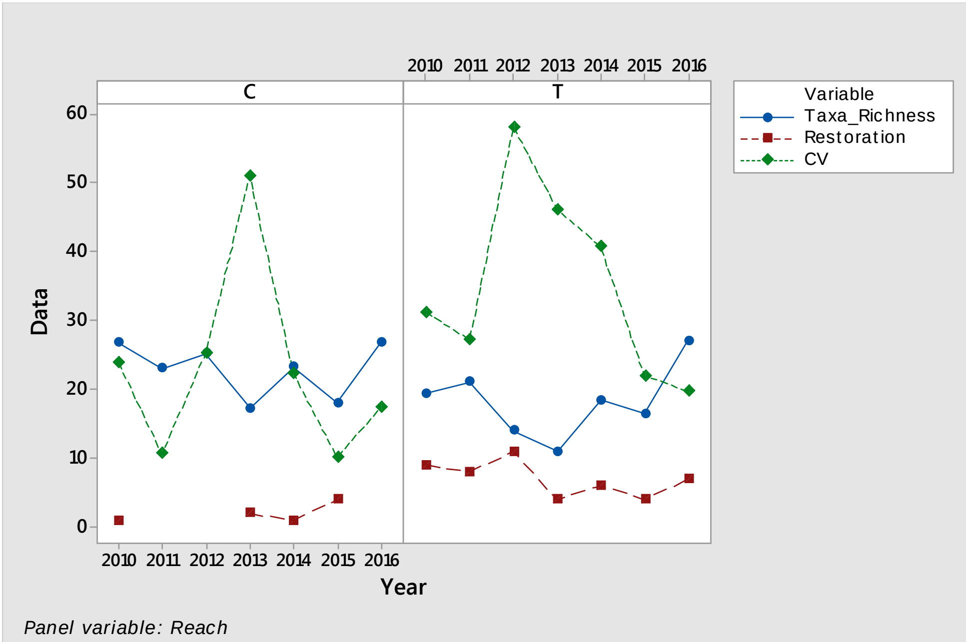

Figure 16. Line plots of mean drift taxa richness, CV of taxa richness, and the number of restoration projects by reach and year.

Discussion

Experimental design and continuity of datasets

Proper experimental design and planning is a critical step in order to ensure that the right type of data and a sufficient sample size and power are available to answer the research questions of interest. Additionally, statistical tests, particularly parametric tests like ANOVAs and multiple comparisons test, perform better if there are equal numbers of measurements for each group. Other reductions in power result from data reliability, selection bias, sampling bias, and random error (Ott and Longnecker 2008; Sokal and Rohlf 1995).

Since restoration occurred before the macroinvertebrate data collection started in the MFJDR, a true baseline of the macroinvertebrate communities never occurred and further, there were restoration activities upstream of the most upstream macroinvertebrate control site. Thus, the comparisons in this study suffer minimally from data reliability, although selection bias, sampling bias, and random error most likely affect these analyses as well. This is due to the selection of new reference sites, different people sampling each site, natural variability, misidentification of macroinvertebrates, and/or potential high autocorrelation. As a result, it was difficult to detect differences between the means of all analyses between control and treatments. Future investigations within the MFIMW would benefit in increasing the number of macroinvertebrate collection sites within the control reaches to assist in providing conclusive evidence supporting the hypothesis that the biotic integrity of the MFJDR has improved with stream restoration.

Final Model Selection

Bioassessment practitioners have largely focused on aspects of accuracy and precision when evaluating the performance of ecological indices. Moreover, the applicability of each index should consider the variability that confounds accuracy as well as the geographic applicability of the index to different regions to minimize both type I and type II errors. Mazor, et al. (2006) found that assessments with higher sensitivity frequently had lower accuracy (Mazor et al. 2006). In regional assessments, accuracy and lack of bias are more important than precision since we can make up for low precision by using large numbers of samples; however, for site-specific assessments, both accuracy and precision are important (Mykrä et al. 2008). As proposed by Levins (1966), an inherent tradeoff exists among 3 desirable predictive model traits: reality (i.e., accuracy, or lack of bias), precision, and generality (Levins 1966; Ode et al. 2008). For this research, where the goal was to create a more accurate and precise predictive model to estimate the biotic integrity of the MFJDR after stream restoration, reality was improved at the expense of precision for the new models while the initial O/E model traded off reality (e.g., accuracy and bias) for precision.

Accuracy and precision are primary considerations so as to detect a significant differences between reference sites and impaired sites in biomonitoring (Resh and Jackson 1993). Nevertheless, neither accuracy nor precision can be directly interpreted in context of index performance that are perhaps ecologically and technically most important to resource managers due to the inability to compare model predictions with the true condition of a site (Hawkins et al. 2010a). However, the responsiveness of an index is directly quantifiable compared to accuracy and precision, which we defined as a measure of how much of an effect a given stress will have on an assemblage. Because the three models varied two-fold (12–24%) in the percent of sites that were predicted to be most disturbed, they clearly varied in their responsiveness to the same stress when applied to the same sites. The fact that the new models underestimated impairment relative to the initial O/E model has at least 2 potential explanations: 1) poorer precision in the new models resulted in lower impairment thresholds and thus fewer impairment decisions, 2) the new models underestimated the probabilities of capture of some of the taxa that contribute to the O/E calculations. These variations in predictions of most disturbed sites would have a profound effect on management decisions, perhaps wrongfully guiding the need for restoration and potentially causing economic and ecological costs associated with those decisions. Therefore, our research shows how predictive model development and subsequent model selection could have important consequences on stream restoration management.

Management Implications for Stream Restoration

Over a billion dollars are spent annually to restore biodiversity and ecosystem services in streams and rivers (Bernhardt et al. 2007). Despite this, little is known about the effectiveness of this effort. Rumps, et al., (2007) found that more than two-thirds of all interviewed restoration managers reported their projects were successful, but 43% either did not have success criteria or were unaware of any criteria for their project (Rumps et al. 2007). Another study found that 89% of project managers reported success, but only 11% due to the response of a specific ecological indicator (Alexander and Allan 2007). Furthermore, less than 10% of stream restoration projects are monitored (Bernhardt et al. 2005) and multiple studies showed that restoration results are highly variable (Miller et al. 2010; Roni et al. 2008; Stewart et al. 2009; Whiteway et al. 2010).

Stream restoration is often practiced unsystematically, with no defined objectives and/or without a rigorous experimental design (Bernhardt et al. 2005; Roni et al. 2002; Rumps et al. 2007); however, restoration can be most effective by providing a target-oriented and integrative approach to measure post-restoration effects. In this manner, Palmer, et al. (2005) proposed five standards for ecologically successful stream restoration. First, the stream restoration design should be based on a specified guiding image of a healthy, more dynamic stream that may exist at the site. Second, successful restoration would reestablish the biological integrity of rivers. Third and fourth, the river system must be self-sustaining and resilient to external perturbations and not inflict any lasting harm on the ecosystem. And lastly, pre- and post-restoration monitoring must be completed and made available to the public (Palmer et al. 2005).

We argue that the restoration actions occurring within the MFIMW have or are in the process of meeting the assessment protocols for ecological restoration, as defined by Palmer, et al. (2005), for the following reasons. First, the MFIMW have defined guidelines and objectives in the restoration plan and thus have a specified guiding image of a healthy, more dynamic stream. Additionally, the MFJDR has not experienced any lasting harm and may even be more resilient as compared to the SFJDR since the CV of the MFJDR was not as strongly affected by climatic variations like streamflow and stream temperature that occurred in 2015 (Henderson 2017). Moreover, the MFIMW has conducted pre-and post-restoration assessment of physical and biological attributes of the stream plus has made the data available to the public. The only criteria not specifically met is the reestablishment of the biological integrity of the stream as well as increased resilience, but there are signals in the data that do suggest that restoration actions in the MFJDR have affected the macroinvertebrate communities.

Based on the available data, we were unable to detect any significant improvement post-restoration to the MFJDR macroinvertebrate communities between control and treatments, although this may reflect the imbalance between the number of control and treatment sample. This unbalanced sample design hinders the analysis, particularly when attempting to infer significant statistical differences in macroinvertebrate communities between treatment and control reaches. Furthermore, spatial autocorrelation is often viewed as problematic with inferential statistics. In fact, when non-spatial models are used to analyze spatially correlated data, it can lead to biased parameter estimates and invalid statistical inferences (Legendre 1993). Additionally, stream restoration often restructures habitat stability, food resources, and the thermal regime, thereby creating conditions that favor species possessing functional traits suited to the restored environment. Thus, in this manner, restoration activities that modify local physicochemical and habitat variables could determine which species from the regional pool would occur locally (Poff et al. 2006; Tullos et al. 2009). Therefore, future investigations within the MFJDR would benefit in improving the experimental sample design as well as exploring if functional group analysis and the use of spatial models would assist in providing conclusive evidence supporting the hypothesis that management actions are affecting the biotic integrity of the MFJDR.

Lessons Learned

The lessons learned from the predictive model development in this research is that there are inherent tradeoffs among desirable predictive model traits and that while a predictive model may improve upon its accuracy, other desirable model qualities like precision, bias, sensitivity, or responsiveness will not necessarily increase with it. Furthermore, to limit noise and variability and subsequently increase power in the ANOVAs and multiple comparisons used to detect differences in means of macroinvertebrate data post-restoration, it is imperative to have a consistent data collection effort across both years and sites. Moroever, the number of collection sites within both treatments and controls should be relatively comparable when using parametric inferential statistics with a low sample size.

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Appendix K – Analysis of the Relationship Between Macroinvertebrates, Streamflow, and Temperature in the Middle Fork John Day River, OR

Robin Henderson, Washington State University, Tri-Cities, School of the Environment

Abstract

We tested how strongly aquatic macroinvertebrates were associated with streamflow and stream temperatures in the Middle Fork John Day River (MFJDR). The strength of the relationships with streamflow, temperature, and the MFJDR benthic and drift macroinvertebrate communities, were measured using taxa composition (the Observed/Expected index), taxa richness, tolerance of taxa, and drift macroinvertebrate biomass (g) as response variables. Benthic macroinvertebrate taxa composition, as measured using the Observed/Expected index, and drift macroinvertebrate biomass were only weakly associated with streamflow and temperature variables, suggesting other factors more strongly influenced these factors. In contrast to the benthic Observed/Expected Index and drift macroinvertebrate biomass (g), taxa richness and percent intolerant taxa exhibited a moderate to strong association with streamflow and temperature. Our results have direct implications for understanding the relative importance of streamflow and temperature in regulating the structure and composition of stream assemblages and for improving management decisions with regards to restoration actions.

Introduction

Background

Annually, a large amount of resources are utilized to restore biodiversity and ecological function in streams and rivers degraded by land use change and other anthropogenic activities. It is estimated that hundreds of thousands of miles of river corridors are degraded throughout the nation, with 42 percent of streams classified as having poor biotic integrity (i.e., the capability to support and maintain an integrated, adaptive community of organisms having a composition and diversity comparable to that of natural habitats of the region) (Karr and Chu 1999; US EPA 2012; 2014). As such, stream restoration is an increasingly common approach utilized to reverse past degradation of freshwater ecosystems and to mitigate anticipated damage from future development and resource-extraction activities.

Stream restoration encompasses many objectives and outcomes (e.g., enhance water quality, manage riparian zones, improve instream habitat, fish passage, bank stabilization) while using varied techniques to accomplish these objectives (Bernhardt et al. 2005). Techniques often utilized by stream restoration practitioners and managers include channel reconfiguration, large woody debris placement, riparian plantings, purchase of land and water rights, fish passage removal or improvement, among others (Bernhardt et al. 2005). For example, to improve thermal refugia and increase net ecosystem productivity (Bilby and Bisson 1992; Bilby and Naiman 1998; MacBroom 1998; Schneider and Winemiller 2008), restoration project managers may place large woody debris (LWD) within a stream as part of a restoration project. Alternatively, restoration managers may choose to plant trees to restore the process of large wood recruitment to the stream (Beechie et al. 2010).

The underlying assumption of stream restoration programs is that biological integrity will improve following restoration, based on the hypothesis that “if we build it, they [the organisms] will come” (i.e., the “Field of Dreams” hypothesis) (Palmer et al. 2010). Unfortunately, there is little scientific evidence to support this assumption since most projects lack effectiveness monitoring and results have been widely variable (Alexander and Allan 2007; Bernhardt et al. 2005; Bernhardt et al. 2007; Frissell and Nawa 1992; Jähnig et al. 2011; Miller et al. 2010; Palmer et al. 2010; Roni et al. 2008; Rumps et al. 2007; Stewart et al. 2009; Whiteway et al. 2010). Furthermore, in instances where post-monitoring biological studies have occurred, they are limited in scope, employ overly simple methodology, and are not conducted synchronously with studies of the environmental stressors that were responsible for the degraded condition of the stream (Bernhardt et al. 2005). When assessments have been conducted, the variability inherent in biological systems has made it difficult to draw conclusions (Bernhardt et al. 2005; Roni et al. 2008; Rumps et al. 2007). As a consequence, scientists and practitioners know less than is necessary for determining whether stream restoration is leading to recovery of biological integrity in degraded streams and, if recovery does happen, when it will be evident (Bernhardt et al. 2005; Roni et al. 2008; Rumps et al. 2007).

Large-scale restoration projects have been implemented along the MFJDR and in over 15 tributaries in the project area since 2007. The John Day River Subbasin Plan identified sediment load, key habitat quantity, and temperature as limiting factors for both summer steelhead *Oncorhynchus mykiss* and Spring Chinook Salmon *Oncorhynchus tshawytscha*, which formed the basis for the type of restoration activities that have occurred. These restoration activities include channel reconfiguration and floodplain reconnection, fish passage, flow increase, grazing/upland management, instream habitat enhancement, and riparian fencing and planting (Abraham and Curry 2012).

Goals and objectives

To assist the MFIMW in gaining an understanding of the causal mechanisms linking stream restoration and salmonid production and to further understand the environmental factors that structure natural communities, we evaluated the strength of the relationship between macroinvertebrate communities, streamflow, and discharge. Specifically, since little is known about the relative or interactive effects of streamflow and discharge on stream macroinvertebrates (Hawkins et al. 1997; Poff and Zimmerman 2010), we evaluated the relationships between streamflow, temperature, and the MFJDR benthic and drift macroinvertebrate communities, as measured using dry weight biomass (g) (drift only), Observed/Expected (O/E) index (benthic only), taxa richness, Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness, % intolerant taxa, and % tolerant taxa.

Hypotheses

For this analysis, we predicted that there was a strong relationship between the MFIMW macroinvertebrates, streamflow, and temperature.

Site Selection

Macroinvertebrate sites

The benthic and drift macroinvertebrate datasets used to evaluate the potential relationship between macroinvertebrate communities, streamflow, and temperature in the MFJDR were collected by the NFJDWC from the year 2010 until 2016 (Table 1) (North Fork John Day Watershed Council 2013a; Rowell et al. 2014). In total, 10 samples were collected in the MFJDR each year using the protocol recommended by the Center for Monitoring and Assessment of Freshwater Ecosystems, Utah State University (Hawkins et al. 2000). The benthic dataset was used to calculate the O/E index for the MFJDR as well as taxa richness, percent intolerant taxa, and percent tolerant taxa while the drift macroinvertebrate dataset was used to calculate the biomass (g), taxa richness, percent intolerant taxa, and percent tolerant taxa (Figures 1 & 2). The designations of tolerance values for each taxon were found in Merritt, et al. (2008); however, in some cases an entry was missing, so the SAFIT or rapid bioassessment protocol tolerance values were used (Barbour et al. 1999; Merritt et al. 2008; SAFIT 2016). If a Pacific Northwest value was missing, then tolerance values from another region were selected. Furthermore, if more than one taxa within a genus or family had a designated tolerance value, we used a majority rule approach and best professional judgment to assign primary tolerance values. We evaluated taxa which had tolerance values ≤3 as sensitive and those which had tolerance values ≥7 as tolerant and then calculated the percentage of tolerant and sensitive taxa per sample. Typically, tolerant taxa can withstand low dissolved oxygen and/or warmer temperatures.

Table 1. Description of the sampling methods for benthic* and drift** macroinvertebrate data provided NFJDWC.

Metric	**NFJDWC**
Sample periods	October 2010; August 2011; October 2012; September 2013; September 2014; October 2015
Sample net	D-frame kick net*; drift net**
Mesh size	500 µm*; 1000 µm**
Net dimensions	0.09 m^2
Field subsamples	8
Laboratory Subsample	500-550*; 600**
Laboratory identification	Typically genus/species; Chironomidae to sub-family

(North Fork John Day Watershed Council 2013a; Rowell et al. 2014)

Streamflow sites

Streamflow data was collected during the years 2013-2016; however, the data from 2013 was inconsistent and thus not used in this analysis. The loggers were installed and data collected from 10 locations, with only 4 sites located in the mainstem MFJDR. Sites located in tributaries of the MFJDR were excluded since there were not any corresponding macroinvertebrate sites for comparisons. Moreover, of the 4 sites within the MFJDR, only 2 of these sites (MFJDR Upstream and MFJDR below Vincent Creek) could be used in this analysis due the reliability of the data as well as inconsistent collection across all years (Table 2, Figure 1).

Table 2. Availability of streamflow data. Only those loggers in the mainstem MFJDR were used in this analysis due to locations of macroinvertebrate data. Furthermore, those loggers in which the data were deemed unreliable* or not comparable ** was excluded from the analysis.

Station	**2013**	**2014**	**2015**	**2016**	**MFJDR or Tributary**
Upstream	x**	x	x	x	MFJDR
Clear	x**	x	x	x	Tributary
bVincent	x**	x	x	x	MFJDR
Butte	x**	x	x	x	Tributary
GraniteBoulder	x**	x	x	x	Tributary
Beaver	x**	x	x*	x	Tributary
Big Boulder	x**	x	x	x	Tributary
Ruby	x**	x	x	x	Tributary
aBeaver	x**	x	x		MFJDR
bBridge	x**	x		x	MFJDR
MFJD aOxbow			x*	x*	MFJDR
Camp				x	Tributary

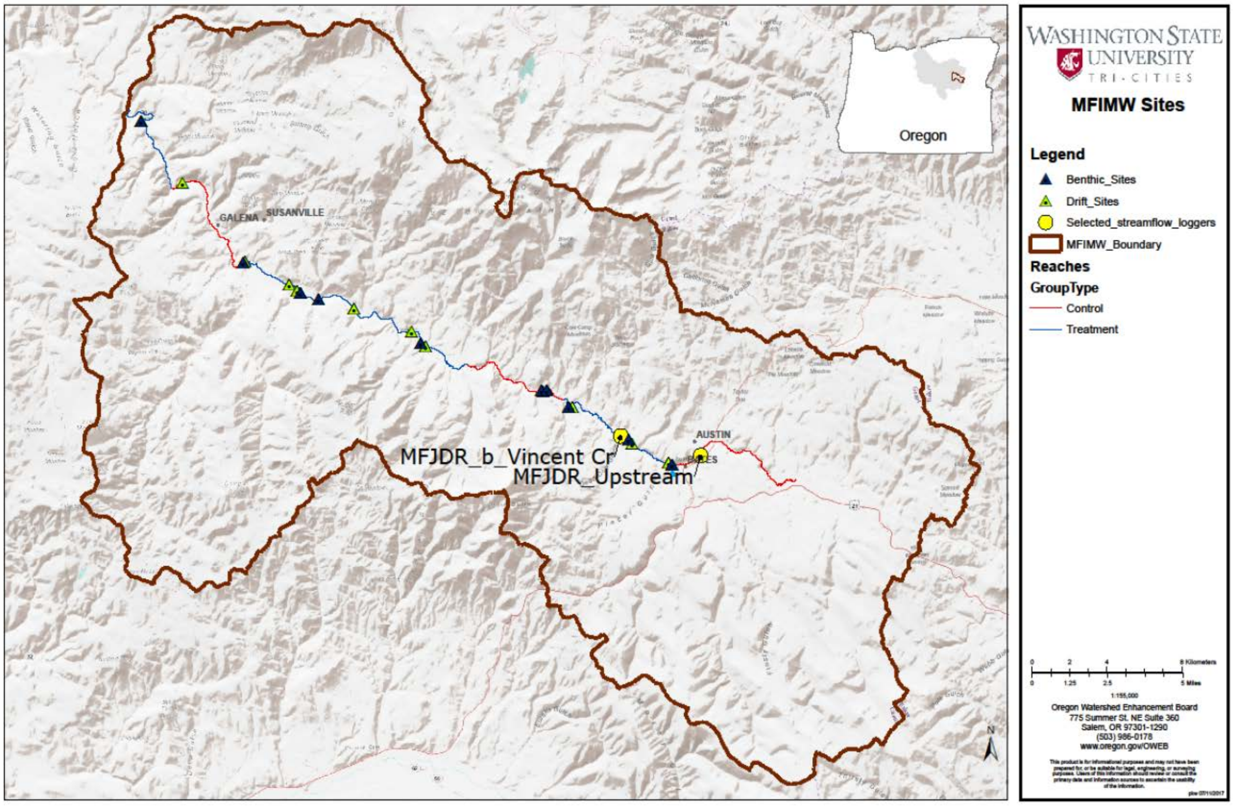

Figure 1. Map of macroinvertebrate sampling sites and selected streamflow loggers in the MFJDR.

Temperature sites

The NFJDWC installed and maintained stream temperature gaging stations in the both the mainstem MFJDR and some of its tributaries (Rowell 2017; Rowell et al. 2014). Stream temperature records were collected from each sampling site at 60 minute intervals. In total, 111 sites were maintained over a period of 11 years from 2005-2016; however, none of the 111 sites had complete records for the entire span and multiple sites had only 1-2 years of record. Furthermore, tributary sites (n=65) were excluded from the analysis since there were not any corresponding macroinvertebrate sites within close proximity (<0.5 km) of a streamflow gaging station. After eliminating sites due to the above reasons, we were left with 12 potential temperature loggers to use for this analysis. Of the 12 remaining sites, only those sites (n=4) with the most complete record were used to evaluate the potential relationship between macroinvertebrates, streamflow, and temperature (Table 3, Figure 2).

Table 3. Availability of selected temperature dataloggers.

Station	2014	2015	2016
Upstream	x	x	x
Below Vincent Creek	x	x	x

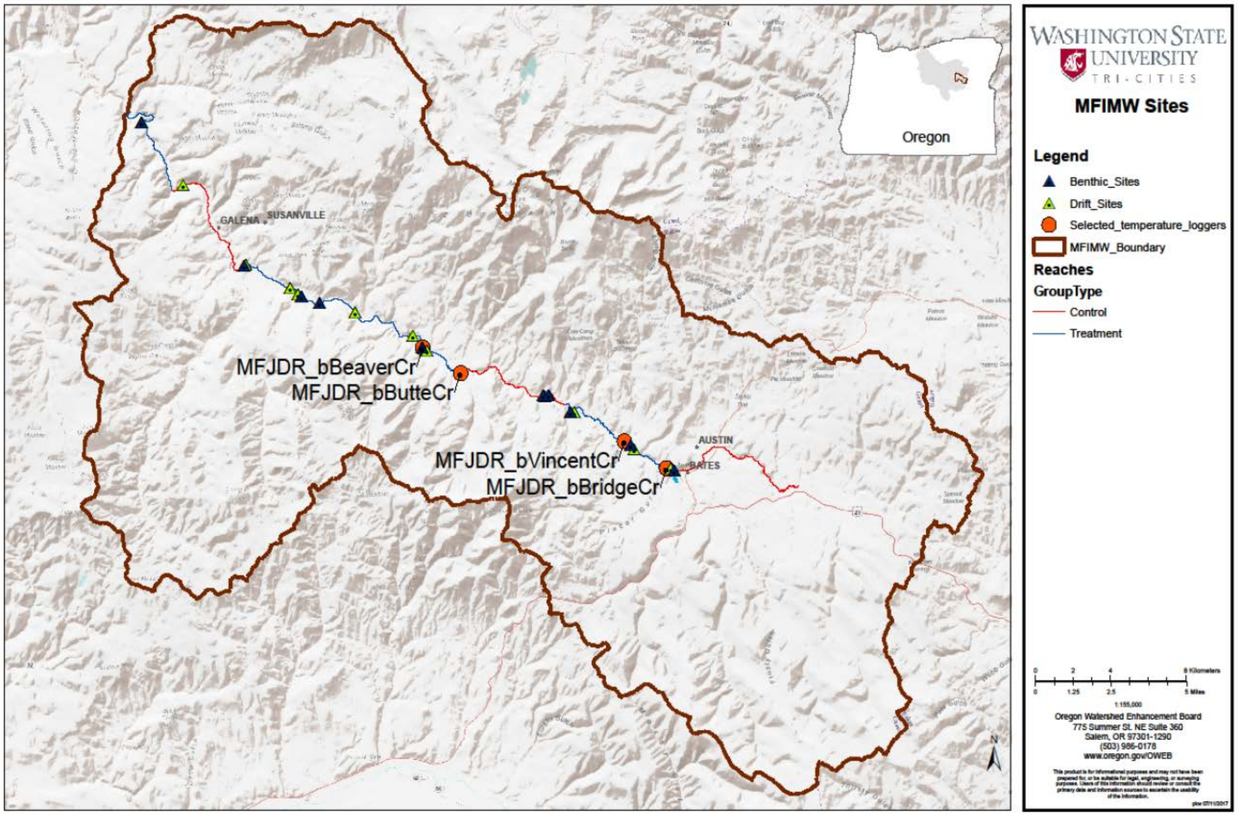

Figure 2. Map of macroinvertebrate sampling sites and selected temperature loggers in the MFJDR.

Methods

Field Methods

Streamflow gaging stations were installed and maintained by the NFJDWC in the MFJDR and some tributaries (North Fork John Day Watershed Council 2013b; Rowell et al. 2014). The streamflow gaging stations were utilized to generate a hydrograph over the period of time the pressure transducer was deployed. Instantaneous discharge was measured at each gaging station at various stage heights using the USGS midsection method. In this method, the stream cross section is divided into rectangular subsections. At the center of each of these subsections (called a vertical), a depth and velocity measurement is made, and the distance from a datum point on the shore is determined. The discharge is calculated for each subsection and the summation of the discharges for all the subsections is the total discharge of the stream at that location and time.

Statistical Analysis

Because aspects of streamflow and temperature may co-vary, we used backward stepwise multiple linear regression to select the set of streamflow and temperature variables that best explained variation in each macroinvertebrate response variable (Chinnayakanahalli et al. 2011). After the removal of nonsignificant variables through the backward stepwise selection, the reduced subset of predictive variables for each attribute was used in multiple linear regressions. The response variables were developed from the NFJDWC macroinvertebrate datasets and summarized into metrics that varied depending on the macroinvertebrate dataset (i.e., benthic or drift macroinvertebrates). For benthic macroinvertebrates, the following metrics were used for response variables: O/E index, taxa richness, EPT richness, percent intolerant taxa, and percent tolerant taxa. Drift macroinvertebrate response variables were as follows: biomass (g), taxa richness, EPT richness, percent intolerant taxa, and percent tolerant taxa.

Predictor variables for the regressions were included based on those aspects of streamflow and temperature thought to influence ecological processes in rivers and streams. For streamflow, the predictor variables were mean annual discharge, coefficient of variation of the annual and summer discharge, mean October discharge, and the mean 7-day maximum discharge while for temperature the variables were mean October temperature, mean annual temperature, mean summer (June, July, August) temperature, and the number of days greater than 18° C as well greater than 22° C. For each macroinvertebrate variable, the best regression model was considered to be the one in which had the highest r^2-adjusted value and the lowest values for Akaike's Information Criterion (AIC). The backward stepwise multiple linear regressions were performed in R using the MASS package (R Development Core Team 2014). The AIC values were calculated in R using the leaps package.

Predictor variable importance in the final models for both benthic and drift macroinvertebrates were measured, with relative importance defined as the proportionate contribution each predictor makes to R^2, considering both its direct effect (i.e., its correlation with the criterion) and its effect when combined with the other variables in the regression equation (Grömping 2006). Predictor variable importance was calculated using the metric lmg from R package relaimpo, where lmg is determined by the average over average contributions in models of different sizes (Grömping 2006; R Development Core Team 2014).

Results

Associations between benthic macroinvertebrates, streamflow, and temperature

Potential relationships between benthic macroinvertebrates, streamflow, and temperature, were analyzed with the O/E index, taxa richness, EPT richness, % intolerant taxa, and % tolerant taxa as the response variables. Based on our results, % intolerant taxa and EPT richness have a stronger relationship with streamflow than the remaining above-listed metrics. In fact, the r^2-adjusted values indicate that both % intolerant taxa and EPT richness have a moderately strong relationship with streamflow and discharge; however, when considering both the r^2-adjusted value and the AIC values, EPT richness is the better model (r^2-adjusted= 70.2% and 64.3%; AIC= 82.3 and 34.5 respectively). Additionally, these models used similar predictor variables, with both models using mean October temperature, number of days greater than 18° C, annual mean temperature, mean 7-day maximum temperature, and mean 7-day maximum discharge (Table 4). The O/E index and % tolerant taxa were only weakly associated with streamflow and temperature variables, suggesting that other factors more strongly influenced these macroinvertebrate variables. All final models produced for each benthic macroinvertebrate variable were significant with the exceptions of the O/E index and % tolerant taxa ($p < 0.10$; Table 4).

Table 4. Summary of backwards stepwise regression statistics for the comparisons between benthic macroinvertebrates, streamflow, and discharge.

Final Model	R^2-adj	AIC	p value
TaxaRichness ~ Oct_Temp + Days>18 + Oct_Discharge + MeanAnnualDischarge + AnnualCVDischarge + SummerCVDischarge + Mean7DayMaxDischarge	41.13%	65.92	**<0.10**
EPTRichness ~ Oct_Temp + Days>18 + AnnualMeanTemp + Mean7DayMaxTemp + Oct_Discharge + MeanAnnualDischarge + SummerCVDischarge + Mean7DayMaxDischarge	64.29%	34.46	**<0.10**
%IntolerantTaxa ~ Oct_Temp + Days>18 + Days>22 + AnnualMeanTemp + SummerMeanTemp + Mean7DayMaxTemp + Mean7DayMaxDischarge	70.20%	82.33	**<0.10**
OE ~ Days>18 + Days>22 + SummerMeanTemp + Mean7DayMaxTemp + Oct_Discharge + AnnualCVDischarge + Mean7DayMaxDischarge	16.74%	-127.08	0.139
%TolerantTaxa ~ Oct_Temp + Days>22 + AnnualMeanTemp + Oct_Discharge +AnnualCVDischarge + SummerCVDischarge + Mean7DayMaxDischarge	0.00%	102.83	0.480

In general, relationships between benthic macroinvertebrates, streamflow, and temperature variables were strongest when a combination of streamflow and temperature variables was used as predictor variables. Specifically, the best performing models had streamflow variables associated with flow magnitude and variation while the temperature variables were associated with frequency, timing, and duration (Tables 4-6). However, when comparing the relative importance of the predictor variables from each final model for benthic macroinvertebrates, all of the top predictor variables were associated with temperature (i.e., the number of days > 22° C, mean October temperature, and mean annual temperature, Table 5). Furthermore, when analyzing the relationship between benthic macroinvertebrates, streamflow, and temperature over time using the top two predictors from the final model, it appears that benthic % intolerant taxa between the years 2010-2016 has generally decreased while the number of days > 22° C and the mean 7-day maximum discharge increased (Figure 3). Additionally, benthic EPT taxa richness between the years 2010-2016 has generally increased while the mean October temperature and the mean 7-day maximum discharge also increased (Figure 4).

Table 5. Predictor variable importance for benthic macroinvertebrate models. Specifically, the count represents the number of times a specific predictor variable was ranked as the top predictor variable.

Predictor variable	**Count**
Days>22	2
Oct_Temp	2
AnnualMeanTemp	1

Table 6. Predictor variable importance for benthic macroinvertebrate models. Specifically, the count represents the number of times a specific predictor variable was ranked in the top 3 variables.

Predictor variable	**Count**
Mean7DayMaxDischarge	3
MeanAnnualDischarge	2
Days>22	2
Oct_Discharge	2
Oct_Temp	1
AnnualMeanTemp	1
Days>18	1
Mean7DayMaxTemp	1
Mean7DayMaxTemp	1
Oct_Temp	1

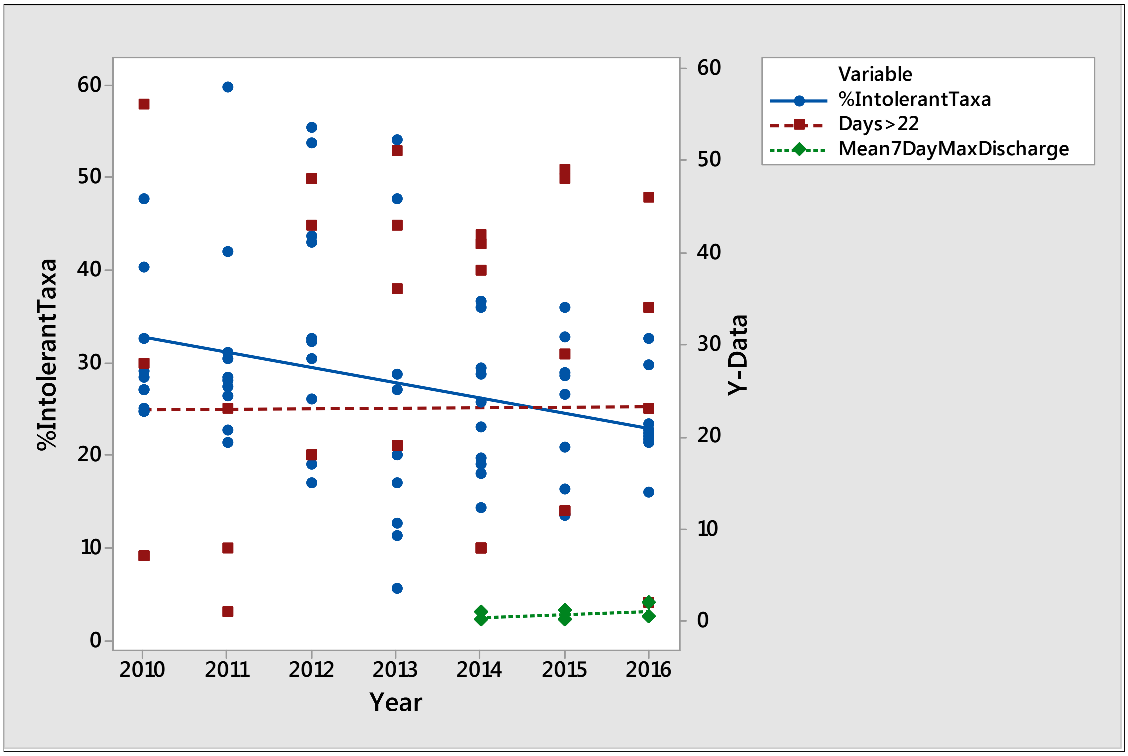

Figure 3. Scatterplot of benthic % intolerant taxa and the top two predictors from the final model, the number of days > 22° C and the mean 7-day maximum discharge, by year.

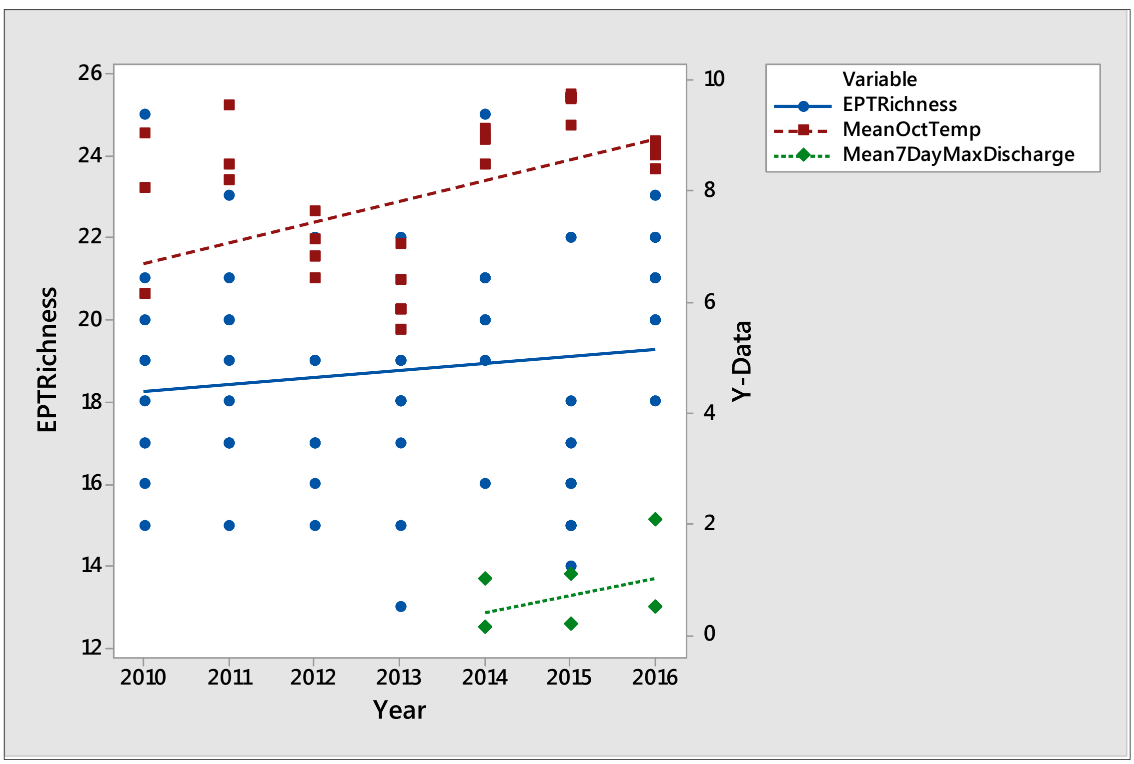

Figure 4. Scatterplot of benthic EPT taxa richness and the top two predictors from the final model, the mean October temperature and the mean 7-day maximum discharge, by year.

Associations between drift macroinvertebrates, streamflow and temperature

For this stepwise regression analysis, we used biomass (g), taxa richness, EPT richness, % intolerant taxa, and % tolerant taxa as the response variables to evaluate potential relationship between drift macroinvertebrates, discharge, and streamflow. Based on the adjusted r^2 and AIC values from the regressions, EPT richness exhibits a stronger relationship with streamflow and discharge than the remaining drift macroinvertebrate metrics (r^2-adjusted = 33.7%, AIC = 96.0; Table 7). Nevertheless, all drift macroinvertebrate final models with the exception of biomass (g) exhibited a weak relationship with streamflow and temperature variables since they had similar adjusted r^2 and AIC values (r^2-adjusted = 25.3-33.7%, AIC = 96.0-240.3; Table 7). Drift macroinvertebrates biomass (g) of did not seem to possess a relationship with streamflow and temperature variables, with the best performing model containing summer mean temperature as the predictor variable (r^2 = 0.5%, AIC = 242.0; Table 7) thereby implying that other factors more strongly influenced drift biomass. All final models with the exception of biomass (g) were significant ($p < 0.10$; Table 7).

Table 7. Summary of backwards stepwise regression statistics for the comparisons between drift macroinvertebrates, streamflow, and discharge.

Final Model	R^2-adj	AIC	p value
TaxaRichness ~ AnnualMeanTemp + Mean7DayMaxTemp + MeanAnnualDischarge + AnnualCVDischarge + SummerCVDischarge + Mean7DayMaxDischarge	33.73%	148.16	**<0.10**
EPTRichness ~ Days>22 + AnnualMeanTemp + SummerMeanTemp + MeanAnnualDischarge + AnnualCVDischarge + SummerCVDischarge + Mean7DayMaxDischarge	25.71%	96.03	**<0.10**
%IntolerantTaxa ~ Days>22 + SummerMeanTemp + Mean7DayMaxTemp + Oct_Discharge + MeanAnnualDischarge + SummerCVDischarge	25.37%	196.77	**<0.10**
%TolerantTaxa ~ Days>18 + Days>22 + SummerMeanTemp + Mean7DayMaxTemp + Oct_Discharge + AnnualCVDischarge + SummerCVDischarge + Mean7DayMaxDischarge	33.57%	240.32	**<0.10**
Biomass_g ~ SummerMeanTemp	0.52%	241.96	0.509

As with benthic macroinvertebrates, relationships between the drift macroinvertebrates, temperature, and streamflow variables were strongest when using streamflow variables associated with flow magnitude and variation and temperature variables associated with frequency, timing, and duration (Tables 7-9). Even when comparing the relative importance of the predictor variables from each final model for drift macroinvertebrates, the most important predictor variable for each final model was evenly divided between streamflow and temperature variables (mean 7-day maximum temperature and the annual variation in discharge, Table 8). Also, when analyzing the relationship between drift macroinvertebrates, streamflow, and temperature over time using the top two predictors from the final model, it appears that drift EPT taxa richness between the years 2010-2016 has generally increased along with the variation in annual discharge while the number of days > 22° C decreased (Figure 5). Moreover, drift taxa richness between the years 2010-2016 has generally increased along with mean annual discharge while the variation in summer discharge has decreased (Figure 6).

Table 8. Predictor variable importance for drift macroinvertebrate models. The count represents the number of times the predictor variable was ranked as the top predictor variable for drift macroinvertebrate models. Note that predictor importance for the biomass (g) final model could not be determined as there was only one predictor variable in the final model.

Predictor variable	Count
Mean7DayMaxTemp	2
AnnualCVDischarge	2

Table 9. Predictor variable importance for drift macroinvertebrate models. The count specifically represents the number of times the predictor variable was ranked in the top 3 predictor variables.

Predictor variable	Count
Days >22 deg C	3
Mean7DayMaxTemp	3
AnnualCVDischarge	2
Mean7DayMaxDischarge	1
MeanAnnualDischarge	1
OctDischarge	1
SummerCVDischarge	1

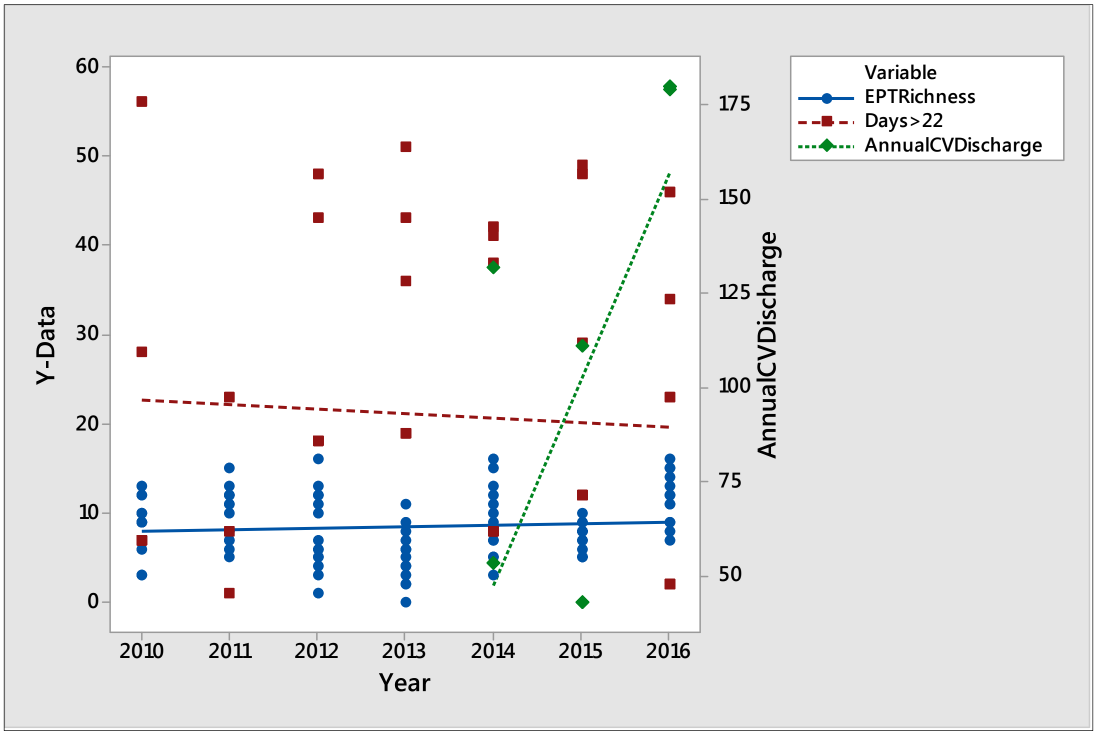

Figure 5. Scatterplot of drift EPT taxa richness along with the two top predictors from the final model, the number of days > 22° C and the annual coefficient of variation for discharge, by year.

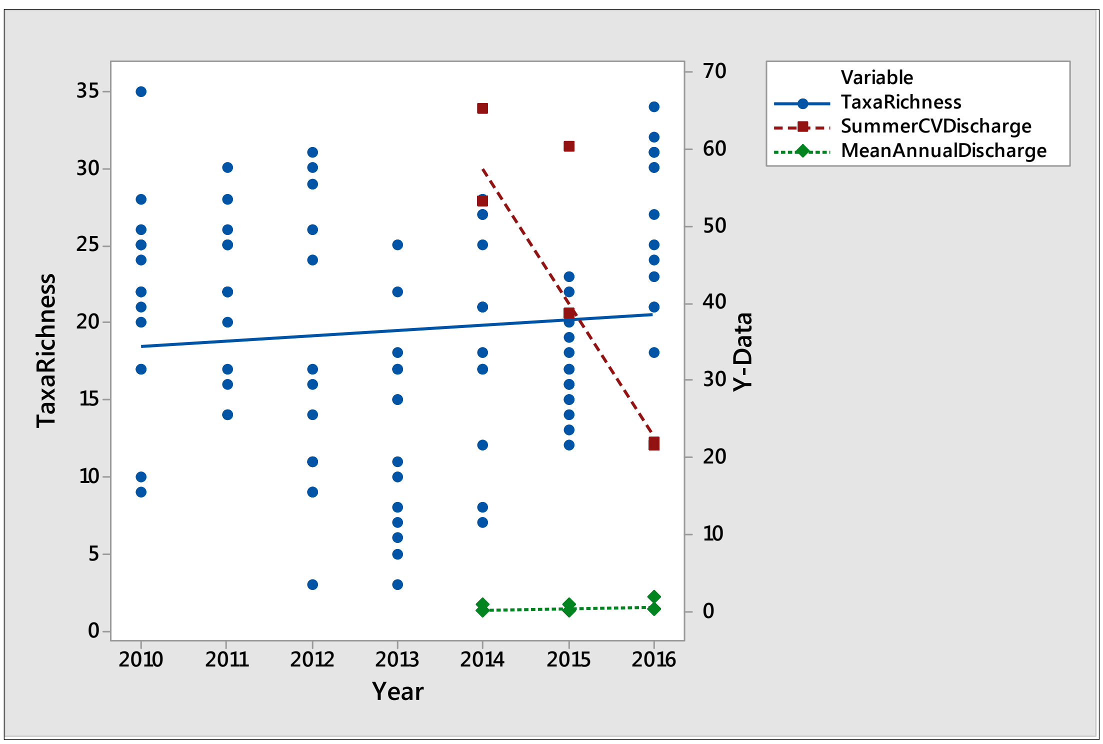

Figure 6. Scatterplot of drift taxa richness with the two top predictors from the final model, summer coefficient of variation for discharge and mean annual discharge, by year.

Discussion

Choice of streamflow variables and temperature variables

To quantify which aspects of streamflow and temperature most strongly affect macroinvertebrates within the MFJDR, it is critical to ascertain the predictor variables that are most useful in understanding ecological patterns and processes from the potential variables available. The streamflow and temperature variables that we utilized influenced the subsequent final models selected to represent the relationship between macroinvertebrates, discharge, and streamflow. Nonetheless, it was not obvious from prior studies which variables should have been selected; therefore, we selected variables based on observations from preceding studies, discussions with colleagues, and our own experience. The number of variables was kept low (n = 10) to facilitate interpretation as it would have been increasingly difficult to interpret and understand the physical characteristics of classifications based on more variables. The use of Pearson's correlation coefficient prior to the regression analysis also reduced collinearity among variables, which helped with both physical and ecological interpretations.

Streamflow and temperature are central components in structuring the local and regional composition of macroinvertebrates (Hawkins et al. 1997; Sweeney and Vannote 1981). Because temperature variables co-varied with some of the streamflow variables we used, it was challenging to segregate the biological effects of one set of variables from the other. However, in our analysis, use of both streamflow and temperature variables resulted in the best performing final models, which implies some degree of independent response of biota to both types of variables (Tables 4-6, 7-9). Furthermore, since the relationships between macroinvertebrates, streamflow, and temperature were nonexistent to weak in some instances to moderately strong in others (Tables 4 & 7), our research implies that other aspects of the streamflow and temperature regime like variation in temperature or baseflow conditions may be more directly associated with the MFJDR macroinvertebrate composition.

Relationships between streamflow, temperature, and macroinvertebrates in MFJDR

A prime goal of stream ecologists is to understand the independent and interactive effects of environmental factors on the structure and function in lotic ecosystems; thus, it is imperative to understand how it affects the communities of stream organisms. To evaluate the relationships of macroinvertebrates with streamflow and temperature, our modeling focused on four aspects of stream invertebrate assemblages: taxa composition (benthic macroinvertebrates only), richness, size (drift macroinvertebrates only) and tolerance. We observed reasonably strong relationships between taxa richness, streamflow, and temperature as well as percent intolerant taxa for benthic macroinvertebrates, but less of a relationship for drift macroinvertebrates (Tables 4 & 7). Although associations do not necessarily imply causation, two factors, mean October temperature and mean annual streamflow, stood out as being important in predicting benthic macroinvertebrates. However, for drift macroinvertebrates the variation in discharge as well as the number of days > 22° C were important in predicting their response. Despite the problems with the underlying streamflow and temperature datasets, our results were encouraging because they have clear implications for understanding the factors that regulate the specific taxa occurring in the MFJDR and thus what restoration actions may be used in the future.

Both streamflow and temperature variables had, for the most part, some measure of relationship with macroinvertebrates in the MFJDR, which is not unexpected considering the repeated reference to these factors in the stream ecology literature (Allan and Castillo 2007); however, it is unclear, that their separate effects can be clearly distinguished from one another. Moreover, assessments of the factors that influence taxa richness may be of limited use in understanding assemblage structure of lotic ecosystems like streams where many of the observed taxa may be accidentals that have drifted into lower reaches from more suitable upstream habitats (Chinnayakanahalli et al. 2011). These issues notwithstanding, our results have direct implication for understanding the relative importance of streamflow and temperature in regulating the structure and composition of macroinvertebrate assemblages in the MFJDR and consequently, the management actions undertaken in an effort to restore the stream.

Lessons Learned

Experimental design and continuity of datasets

It is imperative to have a consistent data collection effort across both years and sites to limit the noise and variability in the analyses and subsequently increase power. Furthermore, care should be taken to have comparable data from year to year. Proper experimental design and planning is a critical step in order to ensure that the right type of data and a sufficient sample size and power are available to answer the research questions of interest. Additionally, statistical tests, particularly parametric tests like regressions, perform best if there are equal numbers of measurements for each group. Other reductions in power result from data reliability, selection bias, sampling bias, and random error (Ott and Longnecker 2008; Sokal and Rohlf 1995).

Due to inconsistent streamflow and temperature data collection, the comparisons in this study suffers from most, if not all, of the above-mentioned problems that result in reductions of power for inferential statistics. These inconsistencies in the data have led to only a few streamflow and temperature loggers from which to derive the regression analyses, which caused difficulties in separating the main effect of the response variable from the main effects of the predictor variables. In fact, for streamflow, only two loggers could be used while temperature had four loggers. The unbalance in the sample design ultimately hindered the analyses and perhaps minimized the strength of the relationship between the predictor variables and response variables. Therefore, future investigations within the IMW would benefit in increasing streamflow and temperature data collection at sites near macroinvertebrate collection sites within the mainstem MFJDR to assist in providing conclusive evidence supporting the hypothesis that the macroinvertebrate communities have a strong relationship with temperature and streamflow.

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Appendix L – Camp Creek Restoration: A BACI Comparative Analysis

Mark Rogers, Oregon State University, Corvallis, OR

Abstract

The MFIMW employed a hierarchical experimental framework to evaluate restoration actions at both watershed and subwatershed scales. While the watershed scale experiment evaluated the response of multiple restoration types over a large time scale, a subwatershed experiment (Camp and Murderer's Creek Restoration Experiment) evaluated a single restoration action type, the removal of log weir fish passage barriers along Camp Creek, in a single restoration event in summer 2011. Removal of these barriers were hypothesized to increase age 1 steelhead density, growth and productivity by lessening interference and exploitative competition within these sites. While the results indicate that age 1 steelhead densities increased following restoration, it was found that discharge, and not restoration actions, was most likely responsible for the observed increases. Furthermore, exploitative competition, estimated by density regulation of summer growth, remained within the system after restoration. Interference completion, estimated by age 1 steelhead survival, also did not change following restoration. Finally, the presence of juvenile chinook in CMP prior to restoration and no detectable increase in chinook migration after restoration suggest that log weirs did not significantly limit steelhead habitat utilization in CMP, and most likely did not increase utilization in the CMP ODFW Sites. In conclusion, the expected beneficial effects of log weir removal appear to have been overestimated since the changes did not lead to statistically significantly improvements in fish passage, competitive effects, nor increases in age 1 salmonid density, growth or productivity. However, high stream temperatures were shown to dramatically suppress growth and productivity. Therefore, high stream temperatures may have suppressed improvements in steelhead population metrics that would have been detected given a lower temperature regime.

Introduction

Background

Over approximately that last 100 years the Camp Creek subwatershed (CMP) underwent extensive habitat degradation due to grazing, riparian logging, and large woody debris (LWD) removal, as well as the installation of culverts which blocked fish passage. Additionally, log weirs were implemented in hundreds of locations along Camp Creek as a restoration effort to ameliorate stream incision and create pools that were thought to enhance salmonid habitat. Although log weirs did create some desirable habitat, they also induced channel widening, decreased complexity and created potential upstream passage barriers to juvenile salmonids (USDA 2008).

To improve salmonid habitat within CMP, the USFS removed 123 log weirs in 2011 and 81 in 2012. The physical effects of removal are shown in Figure 1. Additionally, 63 lodgepole pine logs were placed in Camp and Lick Creeks to simulate LWD.

Figure 1. (a) Example of Camp Creek reach with log weir installation. Photo taken 6/20/2010 (b) Reach after log weir removal. Photo taken 8/24/2011.

Goals and objectives

To evaluate these restoration actions, a Before-After-Control-Impact (BACI) experimental design was implemented, retrospectively. The objective of this experiment was to investigate the response of juvenile steelhead density, growth and production from sub-watershed-level fish passage impediment removals and channel complexity enhancements.

Hypotheses

The removal of log weirs and the addition of in-channel structures in CMP was expected to improve channel complexity, floodplain connectivity, and water quality (USDA 2008). While not part of the original Camp Creek action plan (USDA 2008), we tested the hypothesis that these improvements in habitat quality would lessen intraspecific and interspecific density pressures on the CMP steelhead population. Additionally, we tested the hypothesis that removal of log weirs in CMP would allow for greater juvenile steelhead utilization of the CMP main channel, increasing habitat quantity. These actions would increase density, growth and productivity at the CMP restoration sites.

Hypothesis 1

Age 1 steelhead density, growth and production would increase following restoration actions in 2011.

Hypothesis 2

Interspecific exploitative competition, estimated by density suppression of summer growth, would be eased by habitat quality and quantity improvements following restoration in 2011.

While BACI designs can provide statistical confirmation that observed changes in steelhead metrics are due to restoration actions, this inference has its limitations. Migration to and from CMP are not accounted for by our BACI design. Since CMP is a tributary to the MFJDR, conditions within the MFJFR could induce steelhead to migrate to tributary habitat. Given that temperature is a limiting factor in the MFJDR, it is plausible that changes in density observed in CMP following could be the result of differences in migration to tributary habitat from the MFJDR. This hypothesis was tested as a plausible mechanism explaining increases in CMP density observed following restoration.

Hypothesis 3

Steelhead migration from MFJDR to CMP from high MFJDR stream temperatures is sufficient to explain observed changes in CMP density following restoration

Methods

Site Area

Two sub-watersheds, CMP of the MFJD (Figure 2) and Murderer's Creek (MRC) of the SFJD were selected as the treatment and control locations, respectively. MRC, the control watershed, did not receive restoration, and was employed as a comparator to detect restoration effects in CMP. Both provide steelhead habitat for juvenile rearing and because of their similarities, comparable fisheries responses to climatic and other temporal changes were expected.

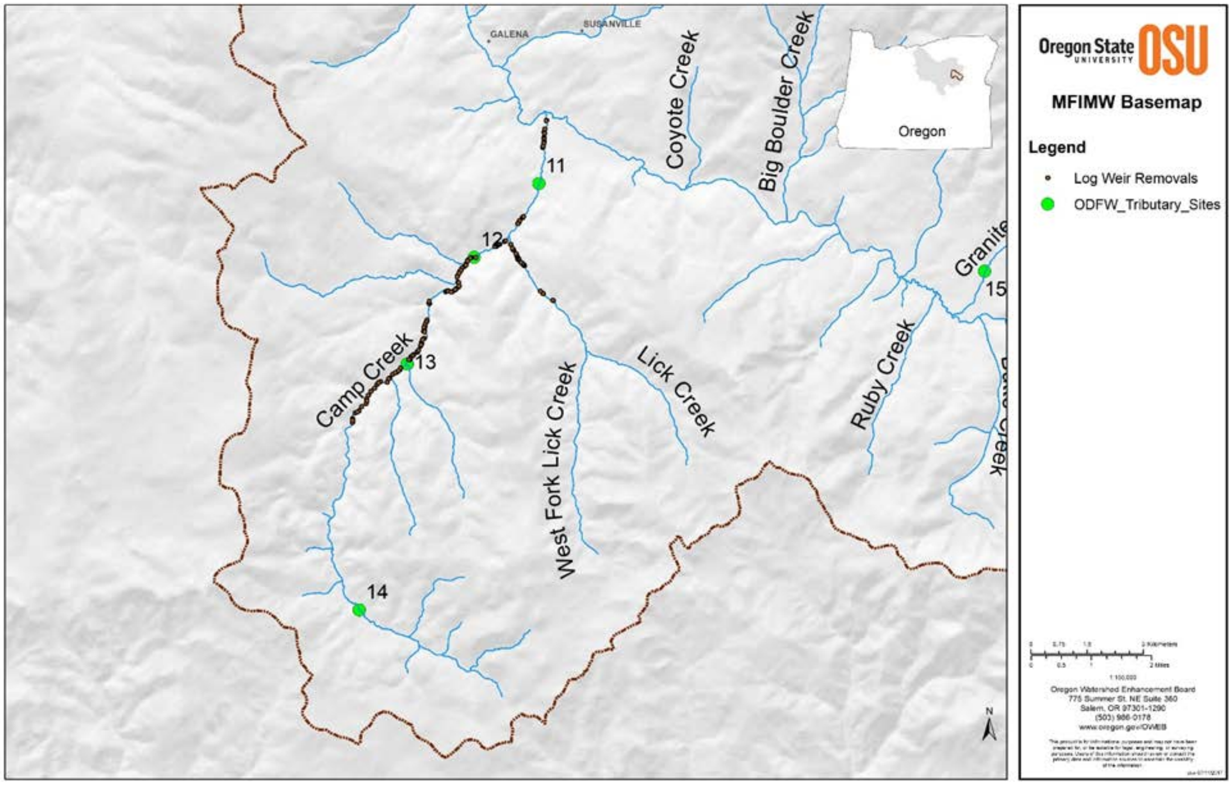

Figure 2. Map of study site area. Restoration actions were monitored at the Camp Creek (CMP) ODFW sites.

Field Methods

Sampling sites (4 in CMP and 3 in MRC) were monitored in the summers between 2008-2015 for steelhead abundance and individual growth. Primary sampling events took place in June, July and October. For further details, see ODFW (2017) and Bouwes et al. (2016).

Additional field variables included stream temperature, measured at the ODFW Sites, steelhead redd densities, and discharge. Steelhead densities were measured at the watershed scales. Discharge was measured differently for MRC and CMP. While a USGS station exists in MRC, the closest gage in relation to CMP is the Camp Creek USGS station on the MFJD main stem, just upstream of the confluence of CMP and MFJD. The Camp Creek USGS station was used as a surrogate for discharge measurements in CMP.

Experimental Design and Statistical Methods

The BACI design is a nested two factor comparative experimental design. The two main effects are Period (Before (2008-2011) and After (2012-2015) restoration) and Treatment (Restoration (CMP) and Control (MRC). The random factor Years are nested within the fixed factor Period and the random factor ODFW Sites are nested within the fixed factor Treatment. Restoration impact is assessed through an ANOVA test for significance of the Treatment x Period interaction term (Underwood 1992, 1994; Downes et al. 2002)

One of the strengths of the BACI experimental design is that temporal fluctuations in response variable covariates (e.g. climate variables, spawner abundances) are accounted for by the design. However, this requires that these covariates must correlate temporally between the treatment and control watersheds. This concept of “parallel trajectories” is a central assumption to the BACI design and must apply to all covariates that may affect the response variable(s).

CMP and MRC have similar temperature regimes (Figures 3) that covary with moderate significance (r=0.70, p = 0.12; Figure 3). While Figure 3 suggests MRC is systematically cooler than CMP, July-August mean daily 7dADM values are nearly identical (CMP = 21.21C, MRC = 21.14C). Respective redd densities also covary (r=0.76, p = 0.08; Figure 4). While discharge covaried with moderate significance during the experiment (r = 0.67, p = 0.06), discharge did not covary significantly in the after period (r = -0.47, p = 0.53; Figure 5). Since CMP and MRC discharges do not covary in the after period, there may have existed differential discharge effects on steelhead metrics that are not accounted for the experimental design, and these potential effects must be further examined.

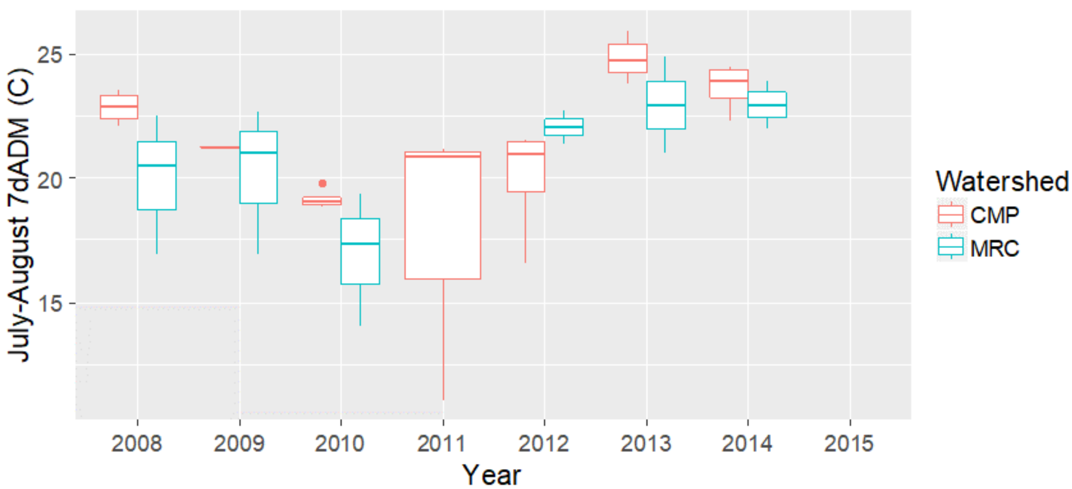

Figure 3. Mean annual temperature (7dADM) of CMP and MRC sampling sites

Figure 4. Steelhead redd densities (spawners/km) observed at CMP and MRC by year.

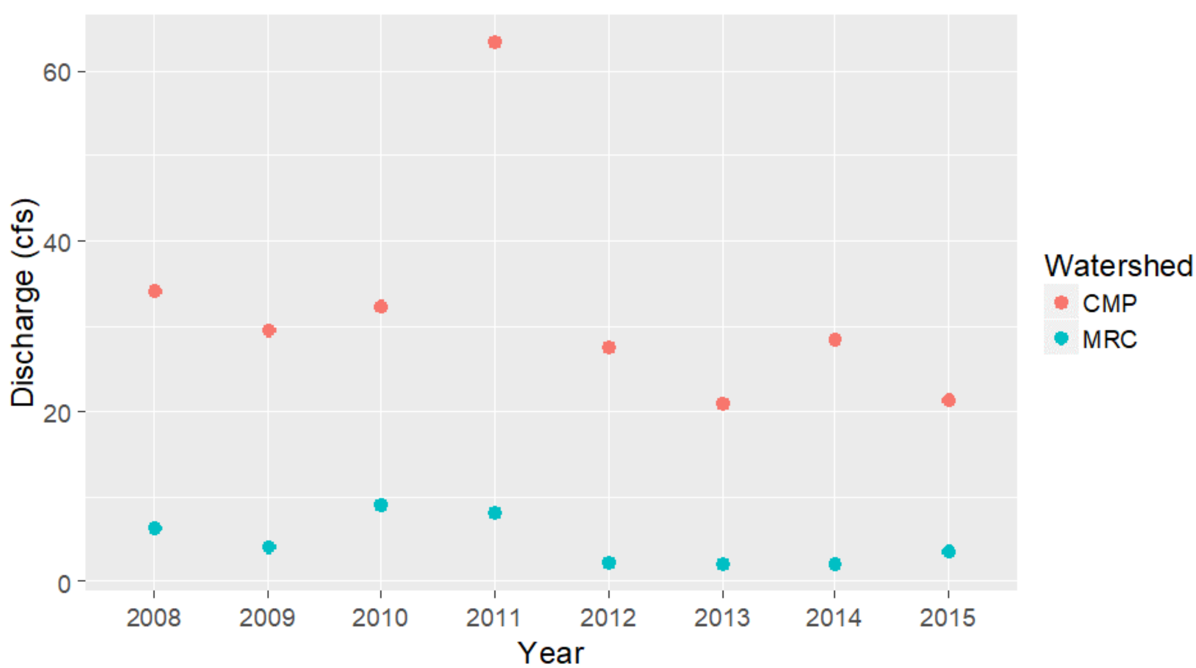

Figure 5. Mean annual discharge of CMP and MRC. Note: The Camp Creek USGS gage (located just upstream of the CMP-MFJD confluence) was applied as a surrogate for discharge measurements in CMP.

Results

Hypothesis 1: Steelhead Response to Camp Creek Restoration

Changes to Steelhead Density

July, Oct and Mean Summer age 1 steelhead density CMP values increase following restoration while those of MRC show a slight downward trajectory (Figure 6a-c). The observed increases in net densities (Net = CMP-MRC) and the ANOVA analysis substantiates the result. The Treatment x Period interaction for Oct and mean summer densities are highly significant, indicating that a change in density, especially in the Fall, transpired following restoration (Table 1).

Table 1. Steelhead (age 1) density responses to CMP restoration actions.

Response	Pr(>F)
July Density (N/100m)	0.159
October Density (N/100m)	0.021
Mean Summer Density	0.039

Changes to Steelhead Growth

There is no observable change in net mean summer age 1 steelhead growth following restoration (Figure 7). The ANOVA test verifies this observation and we are unable to reject the null hypothesis of no change in net growth ($p=0.42$; Table 2).

Changes to Steelhead Productivity

The results do not indicate an increase nor decrease in age 1 steelhead production following restoration in the CMP sites. From visual assessment of Figure 8, there appears to be no change in net production from before to after restoration. This impression is confirmed by evaluating the Treatment x Period interaction in an ANOVA analysis, which showed this was nonsignificant ($p = 0.53$; Table 2); the null hypothesis of no change in production from before to after restoration could not be rejected.

Table 2. Significance tests for steelhead (age 1) responses to CMP restoration actions.

Response	Pr(>F)
Mean Summer Density (N/100m)	0.04
Mean Summer Growth (mm/day)	0.42
Productivity (mm/day/100m)	0.53

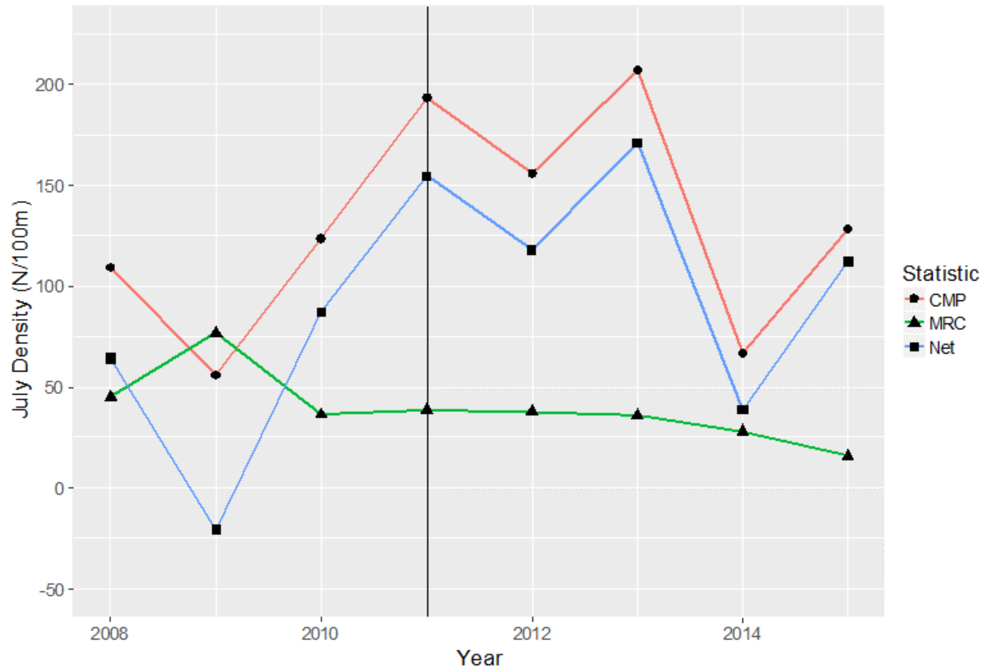

Figure 6a. BACI diagram of July density (N/100m) response to restoration for steelhead juvenile age 1 (Net Density = CMP Density – MRC Density).

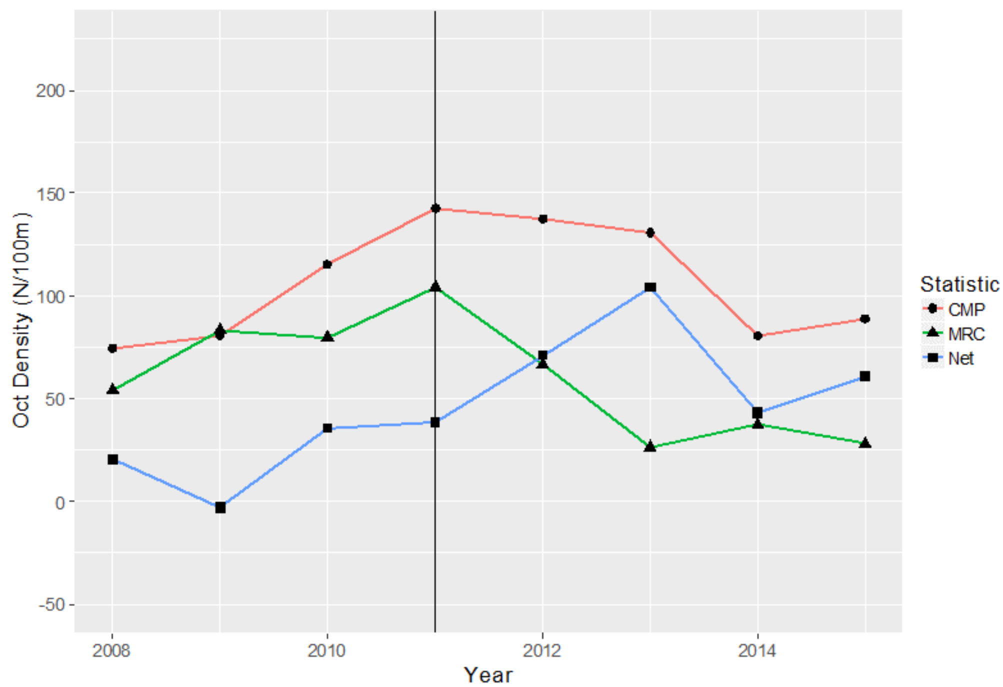

Figure 6b. BACI diagram of October density (N/100m) response to restoration for steelhead juvenile age 1 (Net Density = CMP Density – MRC Density).

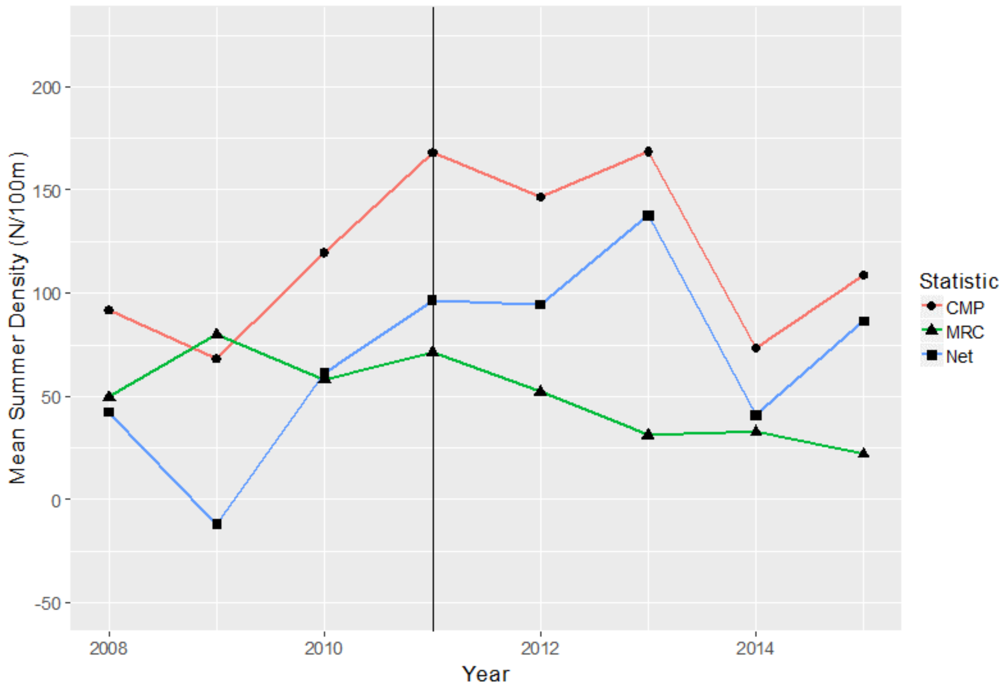

Figure 6c. BACI diagram of mean summer density (N/100m) response to restoration for steelhead juvenile age 1 (Net Density = CMP Density – MRC Density).

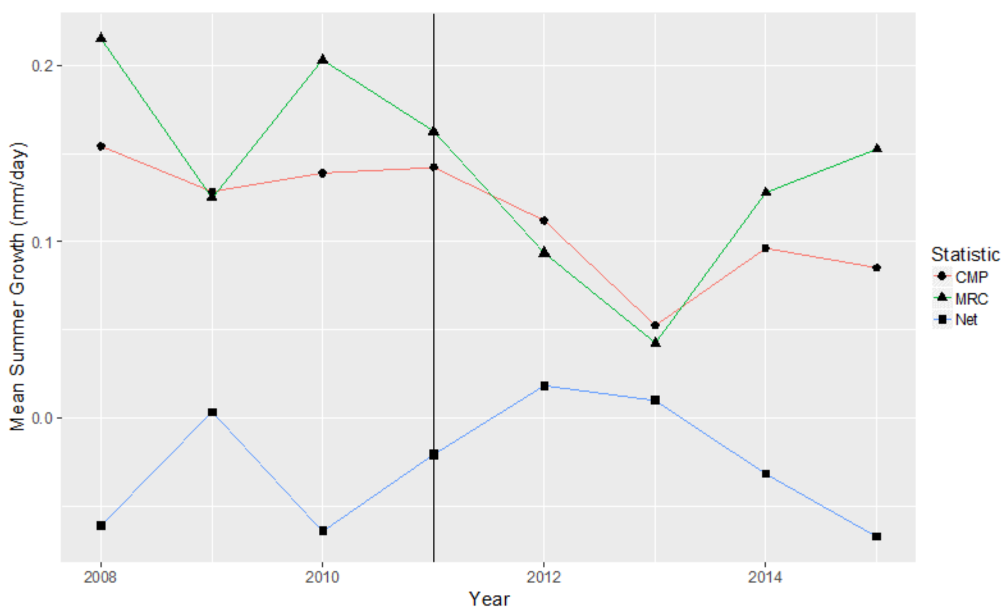

Figure 7. BACI diagram of mean summer growth (mm/day) response to restoration for steelhead juvenile age 1 (Net Growth = CMP Growth – GBR Growth).

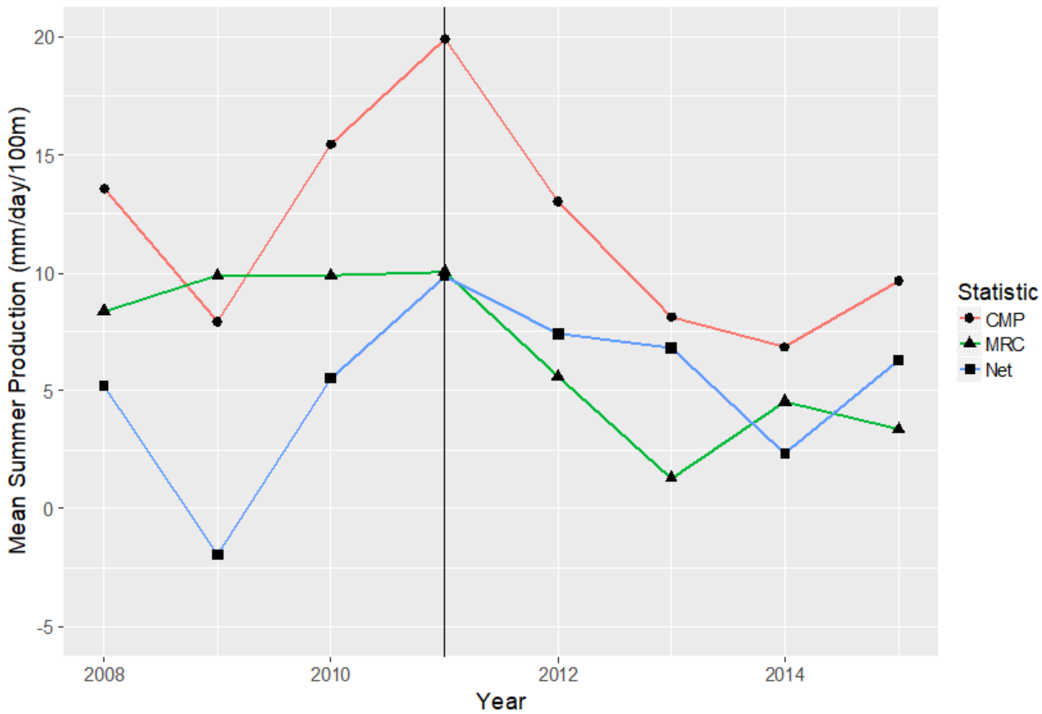

Figure 8. BACI diagram of production (mm/day/100m) response to restoration for steelhead juvenile age 1 (Net Production = CMP Production – GBR Production).

Environmental Factors Influencing Steelhead Population Metrics

Density, temperature, macroinvertebrate drift, hydraulic habitat characteristics, predation, intra-specific competition, and habitat quantity can all affect the steelhead population. This section will describe and quantify some of the factors that limit the CMP population and investigate whether these factors were addressed by restoration. Additionally, this analysis will provide insight on whether discharge, which did not covary between CMP and MRC, was an important factor influencing steelhead metrics.

To gain a global perspective of the factors that may explain growth and density trends in CMP in addition to restoration actions, yearly means of growth, density, temperature and mean summer discharge were tabulated (Table 3) and their relationships plotted (Figure 9-10).

Table 3. Camp Creek yearly averages of steelhead (ages 1 and 2) population metrics and covariates.

Year	Period	Growth (mm/day)	Mean Summer Density (N/100m) Age 1	Mean Summer Density (N/100m) Age 2	Temp (7dADM C)	Discharge (cfs)
2008	Before	0.15	92	22	22.85	34.20
2009	Before	0.13	68	21	21.25	29.59
2010	Before	0.14	119	32	19.18	32.24
2011	Before	0.14	168	52	17.69	63.41
2012	After	0.11	147	68	20.00	27.61
2013	After	0.05	169	44	24.84	20.98
2014	After	0.10	74	36	23.67	28.53
2015	After	0.09	108	49	26.73	21.36

Mean summer discharge did not significantly correlate with summer growth (Age 1: $r=0.61$, $p = 0.1$; Age 2: $r=0.52$, $p = 0.18$). However, removing the outlier 2011 reveals that a significant and positive relationship between growth and discharge existed in the remaining years (Age 1: $r=0.94$, $p < 0.001$; Age 2: $r=0.94$, $p < 0.001$). Also in those remaining years, a moderately negative correlation existed between discharge and mean 7dADM ($r=-0.69$, $p = 0.08$; Figure 9b). No statistically significant relationship was found between discharge and density for either age.

A moderate and negative correlation was observed in all years between temperature and age 1 summer growth ($r=-0.69$, $p = 0.06$; Fig 9a), while age 2 summer growth was weakly correlated with temperature ($r=0.55$, $p = 0.16$; Fig 10a). No correlation was found between temperature and density for either age (Figs 9a and 10a).

Density regulation of summer growth was observed for age 2 steelhead ($r=-0.67$, $p = 0.07$; Fig 10a), but not for age 1 steelhead ($r=-0.26$, $p = 0.54$; Fig 9a) on a yearly mean basis.

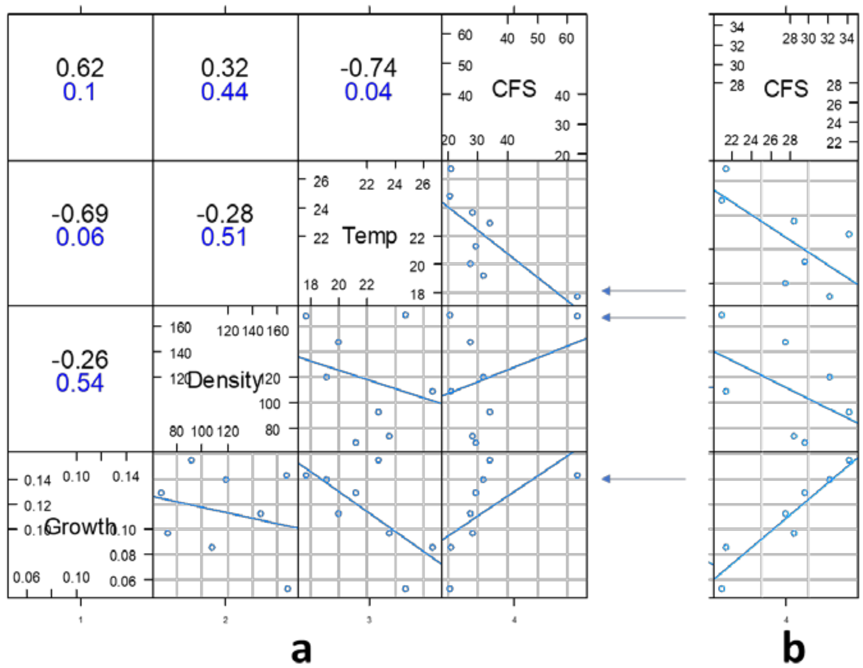

Figure 9. Scatterplot matrix of the yearly averages of steelhead population metrics (age 1) and associated covariates. (a) Years 2008-2015. Arrows denote the year 2011 on discharge plots. (b) 2011 excluded.

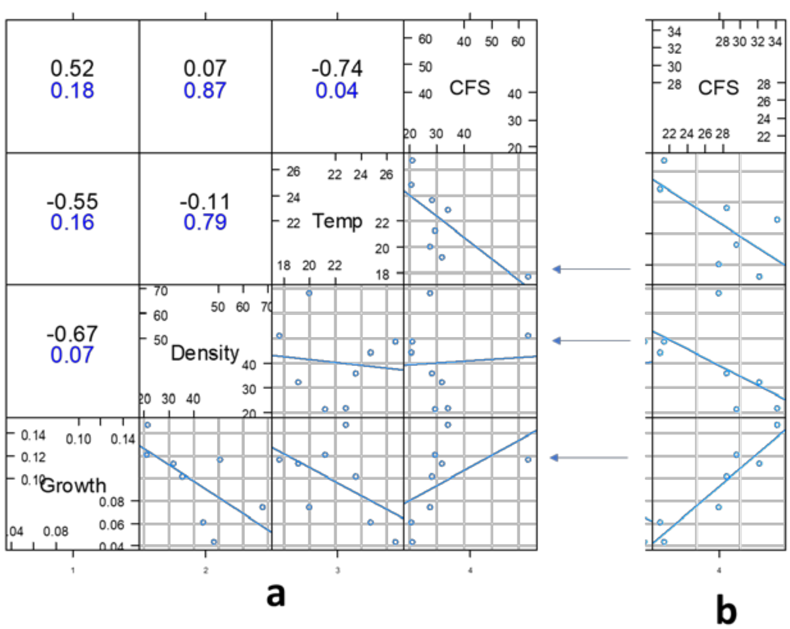

Figure 10. Scatterplot matrix of the yearly averages of steelhead population metrics (age 2) and associated covariates. (a) Years 2008-2015. Arrows denote the year 2011 on discharge plots. (b) 2011 excluded.

Hypothesis 2: Alleviation of Growth Suppression by Density following Restoration

To closely examine density regulation of steelhead age 1 summer growth, data were analyzed at the higher site-year resolution (n=32). A significant, negative relationship between mean summer densities and mean individual summer growth was observed for age 1 and 2 steelhead (Age 1 [Fig 11a]: r=-0.51, p = 0.004; Age 2 [Fig 11b]: r=-0.49, p <0.001), indicating that density regulation of summer growth was indeed present in the CMP sites throughout the course of the experiment.

We hypothesized that restoration may alleviate density regulation on growth; growth-density data were plotted before and after restoration (Figure 12a-b). The slope intervals overlap and are not statistically different (Table 4). Therefore, no discernable change was detected in the growth-density slope following restoration for age 1 and 2 steelhead.

Table 4. Growth-log(density) slope estimates and 95% confidence intervals for before and after restoration.

Age	Period	Slope Est.	p	2.50%	97.50%
1	Before	-0.032	0.019	-0.057	-0.006
1	After	-0.051	0.001	-0.078	0.024
2	Before	-0.032	0.13	-0.076	0.011
2	After	-0.059	0.03	-0.111	-0.007

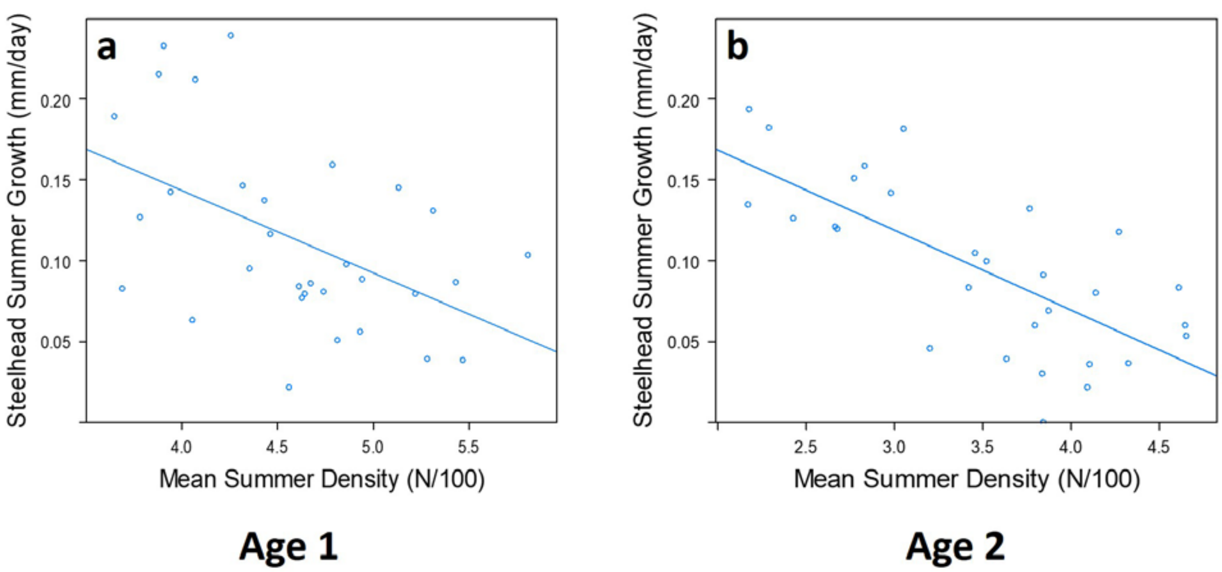

Figure 11. Mean steelhead individual summer growth (mm/day) vs. mean summer density log(N/100m). (a) Age 1 steelhead population (b) Age 2 steelhead population.

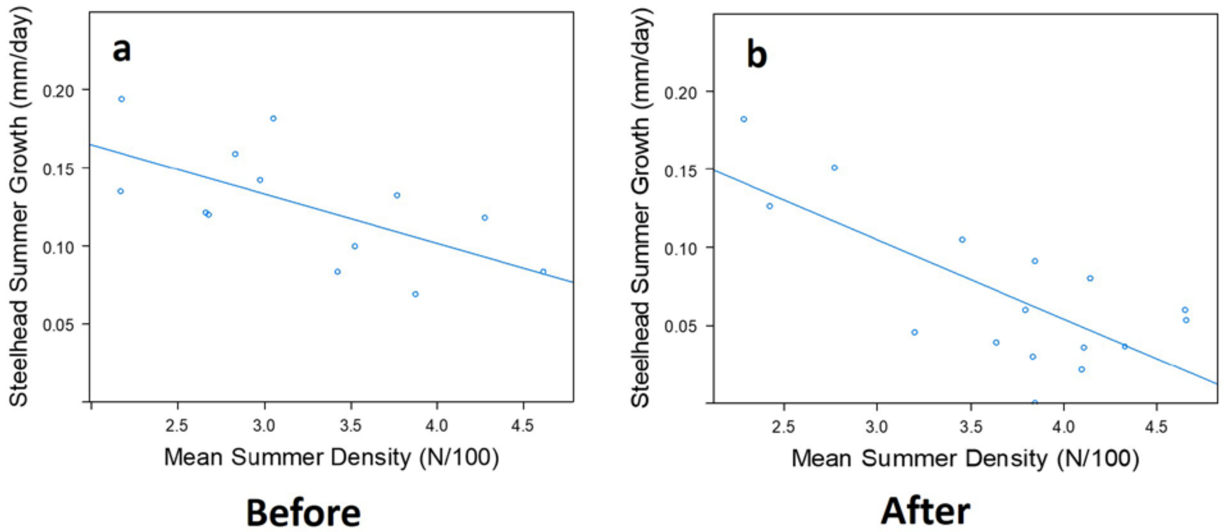

Figure 12. Mean steelhead individual summer growth (mm/day) vs. mean summer density log(N/100m). Population illustrated is age 2 juvenile steelhead, growth and density values plotted are site*year resolution. (a) Before restoration (b) After restoration.

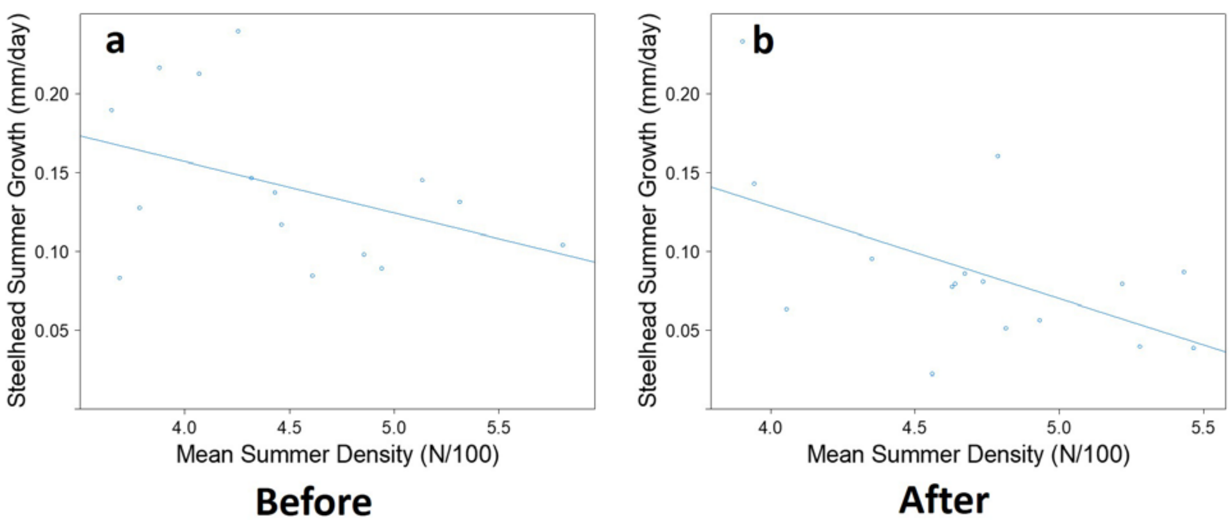

Figure 12a. Mean steelhead individual summer growth (mm/day) vs. mean summer density (N/100m). Population illustrated is age 1 juvenile steelhead, growth and density values plotted are site*year resolution. (a) Before restoration (b) After restoration.

Hypothesis 3: Juvenile Steelhead Migration from MFJDR to CMP

Stream temperatures in the MFJD are elevated due to many factors (e.g., increased channel width, low flow due to water abstraction, climate change, lack of riparian vegetation) and remain a significant limiting factor within the MFJD main channel. During high stream temperature periods, steelhead are observed to migrate to cooler habitat including tributary habitat (Handley and Ruzycki 2017). In the years preceding restoration

(2008-2011), the mean 7dADM for all MFJD sites (see ODFW 2017) was 17.89C while the mean for the years following (2012-2015) was significantly higher, 22.18C (Figure 13). From 2011 to 2015, the mean 7dADM increased by roughly 5C. Comparatively, SFJD mean 7dADM at the screw trap rose less than 1C from before to after periods (20.3C to 21.1C). Because of this large temperature difference between Before and After periods in the MFJD main stem, it is possible that juvenile steelhead migrated from MFJD to CMP habitat during those high temperature years coinciding with the years following restoration.

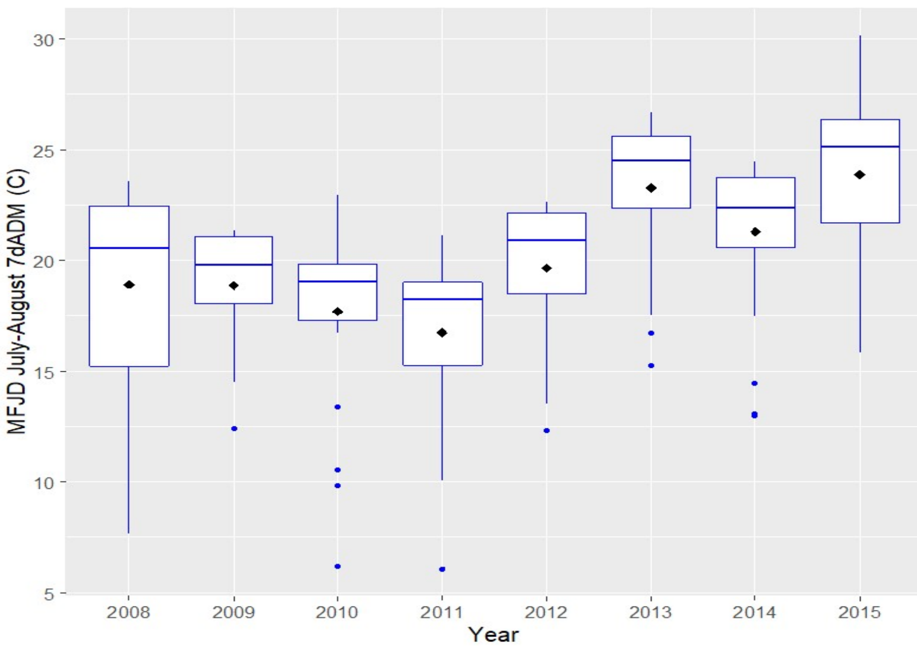

Figure 13. Boxplot of MFJD July-August 7dADM (C) by Year. (◆) denotes mean.

This hypothesis was tested by examining the distribution of juvenile steelhead from downstream to upstream sites in CMP (Figure 14). Clearly, the downstream sites are more abundant than the upstream sites. However, the years 2013 and 2015, which have the highest observed MFJD temperatures in 2008-2015 (Figure 13), do not consistently show higher abundances across all sites in CMP (Figure 14). Furthermore, no correlation was found between yearly MFJD mean summer 7dADM and CMP mean steelhead age 1 density($p=0.94$).

This question was further explored by examining chinook densities in the CMP sites. Chinook presence in CMP is entirely due to migration alone since these salmon do not spawn in the CMP watershed (Handley and Ruzycki 2017). Chinook fall densities did not correlate with that of age 1 steelhead ($p=0.50$), and no difference was found between mean chinook fall densities before and after restoration ($p=0.27$; Figure 15). These results suggest that migration was not a significant contribution to CMP age 1 densities.

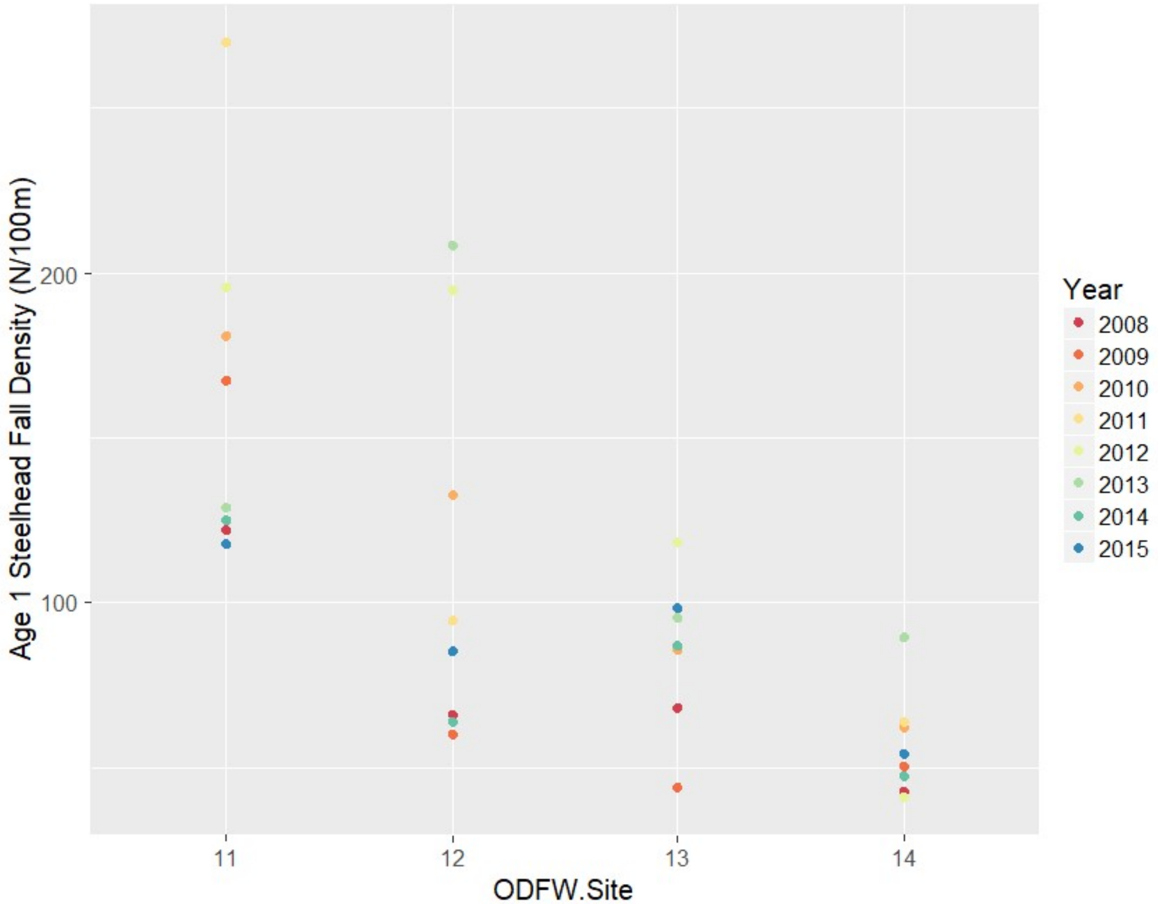

Figure 14. Age 1 steelhead density vs. ODFW Site by Year. Site designations increase from downstream to upstream.

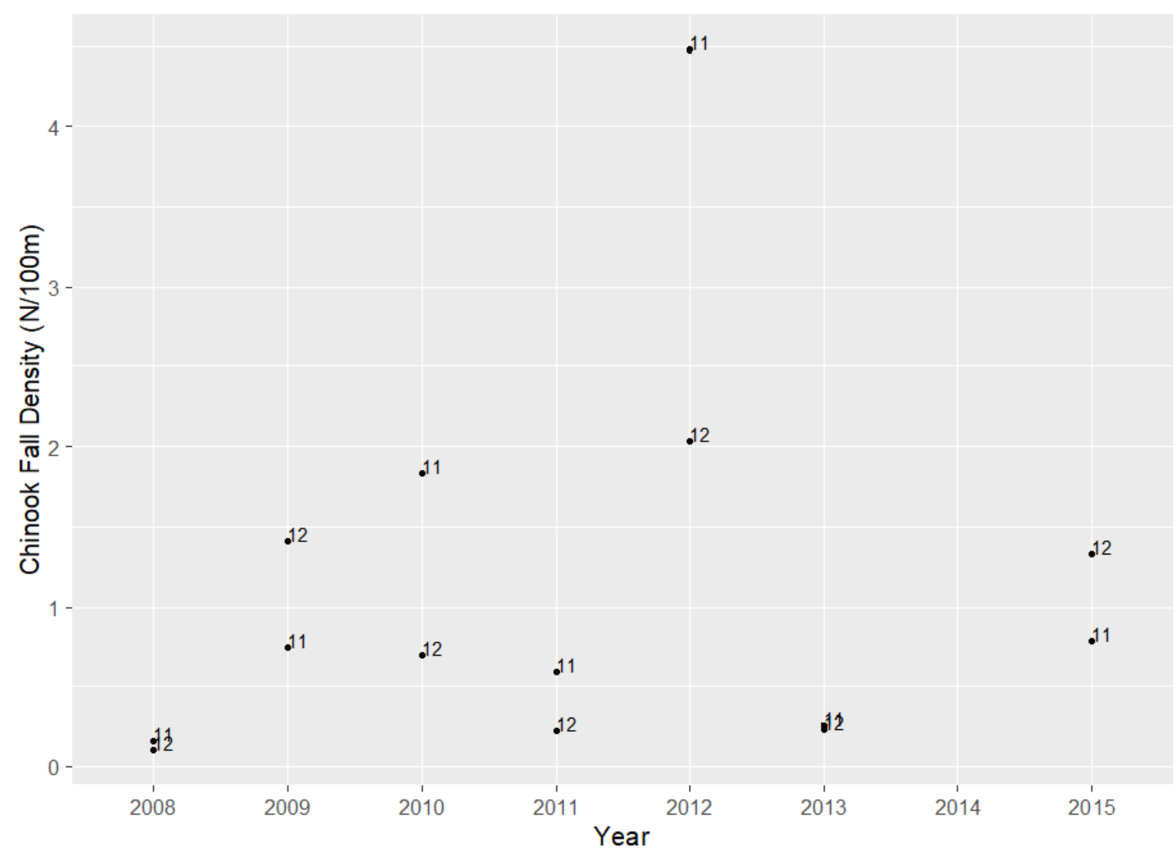

Figure 15. Chinook Fall densities by Year, labels are ODFW Sites. Site designations increase from downstream to upstream.

Discussion

Overview

We present an analysis based on the comparison between a creek that experienced a major restoration effort to a creek that remained in the same condition through the period of analysis. In this way, we seek to illuminate any changes in stream productivity resulting from the restoration activities. It is important to point out certain unavoidable limitations to this study. Underlying this analysis is the assumption that the two streams would have not shown relative differences in steelhead responses in absence of the restoration activity. While the comparator stream sections were selected specifically so that they would satisfy this requirement, it should be noted that the pre-restoration and post-restoration periods experienced significant changes in climate drivers, Specifically, an unexpected and sustained major upward shift in the regional (i.e. MFJD main stem) stream temperatures was not also detected in the SFJD region. These high temperatures were associated with the three lowest seasonal flows all occurring in the post-restoration period. This may have differentially impacted the restored and control stream. Additionally, these two watersheds experienced different hydrologic regimes especially in the post-restoration period. Clearly the pre- and post-restoration periods experienced drastically different climate and hydrologic drivers. Further, we acknowledge that many of the restoration actions are expected to require many years to become fully functional (e.g. woody shrub development), whereas this comparison was made for only the four years following restoration. Such limitations are intrinsic to in-situ evaluations of environmental interventions.

The Effects of Restoration Actions on Age 1 Steelhead Density

In this comparison, restoration actions were associated with an increase in age 1 steelhead density at the ODFW CMP sites by applying a BACI comparison with MRC. Additional factors however, were also investigated to explain the increase. These factors were immigration into CMP from the MFJD main stem, and the effects of redd densities, temperature, and discharge. While stream temperatures in the MFJD were significantly higher in the After period, as compared to the Before period, chinook immigration into CMP did not correlate with age 1 steelhead densities. Therefore, no compelling evidence was found that immigration caused the increase in densities observed in CMP following restoration. While temperature and redd densities of CMP and MRC correlated between the two watersheds, and can be accounted for by the experimental design, discharge was not correlated between CMP and MRC. Discharge therefore, cannot be accounted for by the experimental design, and could explain the observed increase in density.

It was found that discharge of the sampling years did not influence age 1 density in CMP (Figures 9-10). However, discharge of the preceding years may also influence age 1 density. The survival of age 0 steelhead, which is influenced by discharge, will determine to a large extent age 1 density the following year (Handley and Ruzycki 2017). Relative survival rates of age 0 juveniles were estimated by computing the ratio of age 1 abundance to the number of adult spawner from the preceding year. The data show that age 0 survival was indeed strongly correlated to discharge of ($r=0.88$, $p=0.007$; Figure 16), indicating that density of age 1 steelhead was influenced by discharge of the preceding year.

Upon closer inspection of Figure 6c, the highest densities correspond to the years 2010-2013, and not the period following restoration. Visual inspection of Figure 16 reveals that age 0 survival was also higher than average in the years 2010, 2011, and 2012, and these survival rates coincided with high discharge measurements. It is likely, therefore that discharge contributed significantly to the peak age 1 densities in 2010-2013.

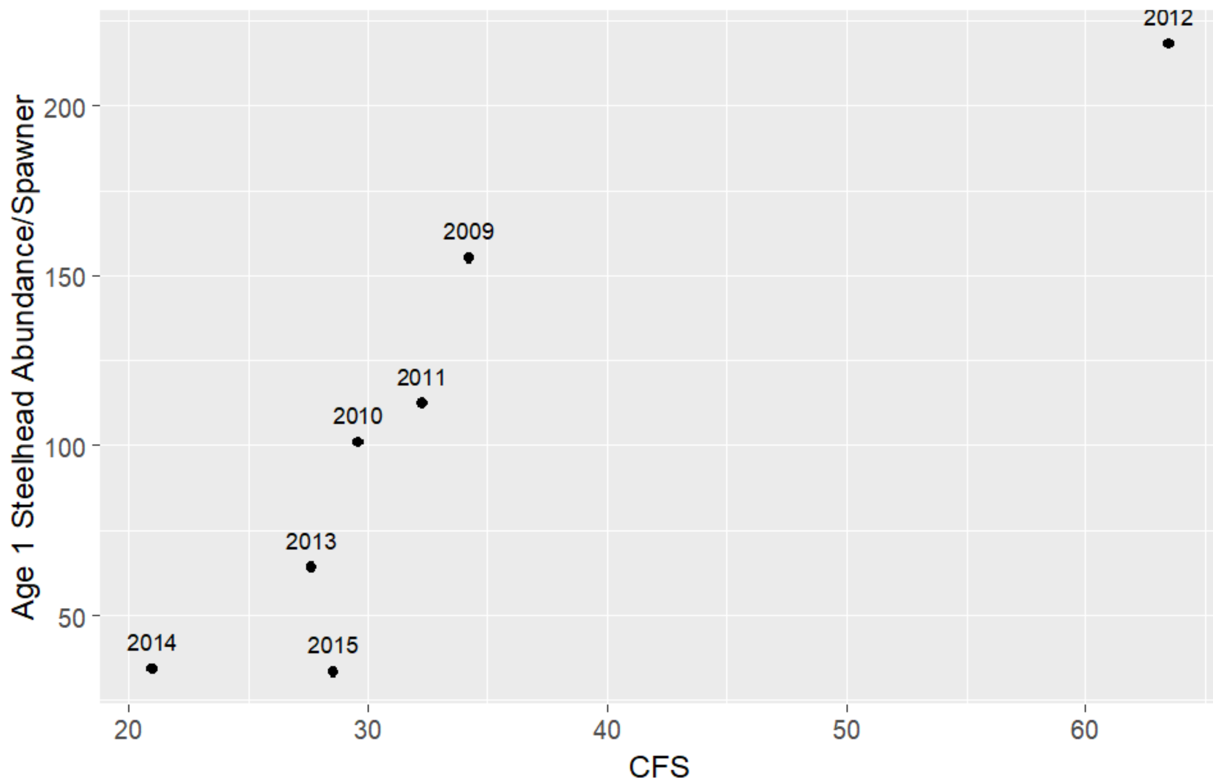

Figure 16. Age 0 survival (estimated by Age 1 Abundance/spawner) vs. mean summer discharge.

Alleviation of Exploitative Competition

Density regulation of age 1 and 2 steelhead summer growth persisted after restoration, and therefore no evidence exists that exploitative competition was lessened by restoration actions. Additionally, CMP experiences high summer stream temperatures (Figure 3), potentially inducing significant energetic strains on individual steelhead metabolism (Beer and Anderson 2013). From 2008-2015, 67% of CMP ODFW site July-August 7dADM temperature measurements exceeded the salmonid threshold of 18C (7dADM) established by USEPA (2003). Maximum daily temperatures exceeded 22C, a threshold associated with decreased foraging and increased aggression (Carter 2005), in 30% of observations. Figure 3 also suggests that 7dADM was greater in the after period as compared to the before period; a one-way ANOVA analysis corroborates this finding ($p = 0.10$).

The data indicate a clear negative relationship between temperature and age 1 and 2 steelhead growth ($p<0.001$) for all CMP 7dADM observations above 18C (Figure 17). Again, 67% of all CMP growth observations were found above 18C, suggesting that temperature suppression of summer growth occurred in CMP throughout the length of the experiment.

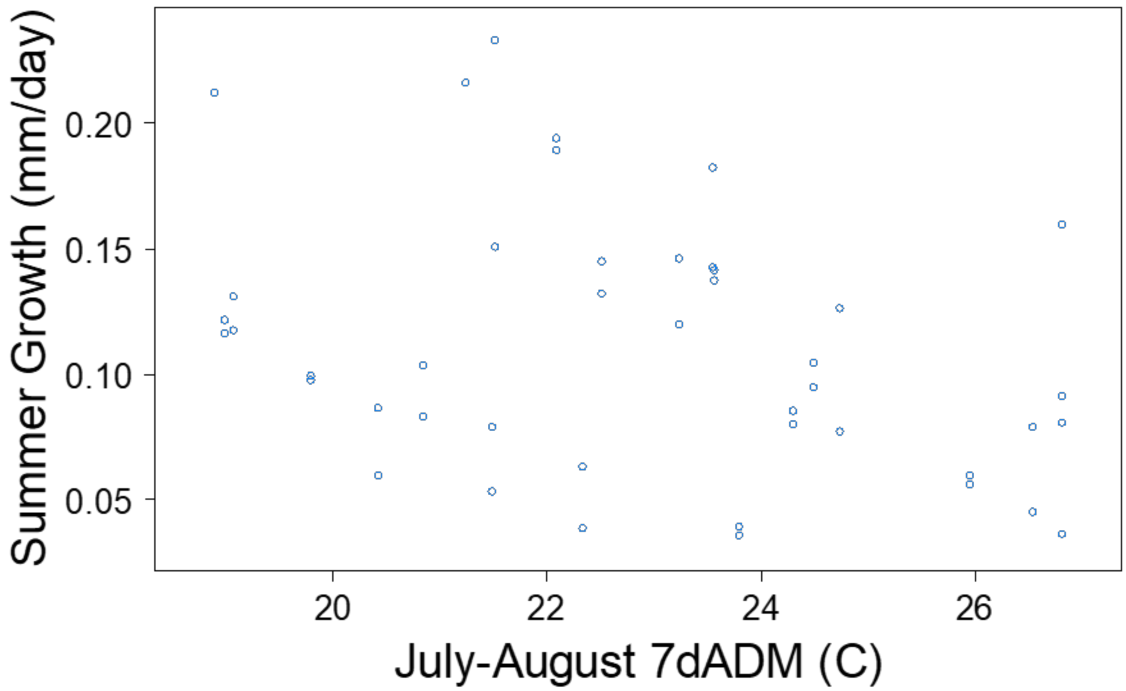

Figure 17. Mean steelhead individual summer growth (mm/day) vs. July-August 7dADM (C). Population illustrated is age 1 and 2 juvenile steelhead observations sampled in sites above 18C (7dADM). Growth and temperature values plotted are site*year resolution (n=64).

The lack of riparian cover may be partially responsible for this continued energetic strain on steelhead. Riparian vegetation was identified as a limiting factor in CMP (USDA 2008), but despite restoration efforts to address this issue, large stretches of stream remained open to direct solar radiation through the period of this study. Additionally, riparian vegetation is a known source of allochthonous macroinvertebrate drift, and if successfully implemented, this action could have augmented the food supply to CMP juvenile steelhead. Therefore, high stream temperatures and low drift fluxes continued to limit the growth environment in the CMP sites. Greater attention to these factors could have resulted in lessened exploitative competition, and improvements in habitat quality from restoration actions may have resulted in observed growth improvements.

Experimental Results Summary

In summary, there does not exist strong evidence that restoration facilitated the observed increases in age 1 steelhead density. While the results of the BACI analysis were significant, the effects of discharge on age 1 steelhead densities were also shown to be significant, and the BACI design could account for the effects of discharge, confounding the results. Furthermore, exploitative competition was not lessened by restoration actions, and other analyses [ODFW 2017] have found no evidence that age 1 survival changed due to restoration actions in CMP either. Additionally, the fact that chinook migration did not significantly increase following restoration indicates that log weirs were not a significant barrier to juvenile migration. In conclusion, the beneficiary effects of log weir removal may have been overestimated, and did not significantly improve fish passage, lessen completive effects or increase salmonid density, growth nor productivity. However, high stream temperatures were shown to dramatically suppress growth and productivity. Therefore, high stream temperatures may have offset improvements in steelhead population metrics that would have been detected given a lower temperature regime.

Experimental Design Assessment

The BACI design requires that the treatment and control watersheds covary with regards to any factor which might influence the response variable(s) (Downes et al. 2002). While MRC satisfied some of the requirements of the BACI design, it did account for discharge, which strongly influenced CMP age 1 densities.[Note: discharge was not directly measured in CMP, and a surrogate measurement was necessary. Given the importance of discharge, this variable should always be directly determined in every large-scale restoration experiment.] The design also did not account for regional temperature differences between the MFJD and SFJD main stems, creating the possibility of differential migration from main stem to tributary habitat. In other words, the design could not account for the influence of discharge on density, resulting in a false positive (type I error) for the effects of restoration on density in CMP.

In practice, finding a suitable control watershed that is correlated with the restoration watershed in every way is very difficult to accomplish, and perhaps even unlikely. Alternative designs should be examined for future watershed scale restoration experiments. An alternative design to the paired-watershed BACI design is the paired-reach BACI design, where restoration and control reaches are paired within a single watershed in close proximity (see Bouwes et al. 2016). Pairing treatments in this way would ensure that explanatory variables (e.g. temperature, redd densities, discharge) will covary with the additional benefit of decreasing between site variance. The challenge of applying this design is to ensure that restoration and control reaches within a pair remain independent. Since these reaches would be in close proximity, fish communication between the restoration and control reaches is very possible. Therefore, paired reaches must be close enough to ensure covariance of explanatory variables while far apart enough to remain independent. An additional alternative would be to randomly disperse both restoration and control reaches within a watershed, with the criteria that reaches remain a specified distance apart.

In terms of the sampling design, it is important to note that the CMP sites were not randomly dispersed within the watershed, but instead randomly allocated along 12km of the Camp Creek main channel (Figure 2). Therefore, responses must be considered a response to restoration in Camp Creek, and not a watershed level response.

Key Findings

- The BACI design, in conjunction with other analyses, is a valuable tool in stream restoration evaluation.
- Log weir removals and LWD additions did not significantly lessen completive effects or increase salmonid density, growth nor productivity.
- High stream temperatures in the post-restoration period may have overcome any potential productivity gains provided by restoration actions.
- Chinook migration did not detectably increase following restoration, indicating that that log weirs were not a significant barrier to migration.
- Discharge, which had a significant effect on age 1 steelhead densities in CMP, did not correlate between CMP and MRC, confounding the results of the BACI analysis.

Lessons Learned

- Channel habitat improvements and monitoring must take place after successful stream temperature reduction.
- Begin plantings early and allow riparian vegetation to recover for several years before attempting restoration experiment.
- Differential climatic factors between restoration and control watersheds limit the ability to confidently attribute observed response changes to restoration actions.
- The paired-reach BACI design may provide a better platform to resolve these issues.
- Alternative BACI designs should be researched through simulation and in the field.

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Appendix M – Middle Fork John Day River Intensively Monitored Watershed Socio-Economic Indicators Follow-Up Study

Michael Hibbard, Susan Lurie, and Rodney Bohner, University of Oregon, Eugene OR

Abstract

Only the Middle Fork John Day River Intensively Monitored Watershed (MFIMW) project includes a socio-economic element that is monitoring the contribution of restoration projects to the socio-economic health of the local community, what is often called the restoration economy.

This study examined the socio-economic effects of restoration work the MFIMW in two ways.

- A set of community indicators assessed the overall socio-economic well-being of Grant County over time and put the restoration economy into context. These measures were collected from existing sources that are sensitive to the effects of restoration work.
- A set of outcome measures estimated the contribution of MFIMW restoration work to the Grant County economy. Outcome measures were based on an inventory and analysis of completed projects.

The indicators show that Grant County was in socio-economic decline over the past 40-50 years but that things are improving recently. In particular, jobs and earnings are both up, and other indicators support that trend. At the same time the Grant County economy is doing better, restoration work is bringing work and money into the economy.

- Restoration related planning, management, and monitoring jobs in Grant County nearly doubled between 2000 and 2016.
- OWEB capacity grants – basic operating funds for watershed councils and SWCDs have brought a total of $1,277,150 to Grant County since 2007. When the multiplier of 5.09 is considered, capacity grants brought about $6.5m to the local economy.
- The 100 restoration projects carried out in the MFIMW area in the period from 7/1/07 to 6/30/17 brought a minimum of $15.6 million dollars into the local economy, along with creating almost 170 jobs and generating additional economic activity in the range of $20-25M.

Introduction

Background

Arguably, one of the most significant developments in natural resource planning and management in the past twenty years has been the emergence of the restoration economy – also referred to as conservation-based development, sustainable livelihood, and the conservation economy, among other terms. As ecological preservation and restoration activities have become more and more important, their potential as a source of local job and wealth creation in rural communities has been recognized (Hibbard and Karle 2002). While the central focus of ecological restoration is healthy, functioning ecosystems, the restoration economy explicitly considers the local economy and community as well. It holds that “ecological integrity, economic opportunity, and community are inextricably linked in the long run” (von Hagen and Fight 1999, 3).

Oregon has been in the vanguard in that effort, through the work of OWEB. The central purpose of OWEB is to support environmental restoration and management. At the same time, however, Oregon law (ORS 541.353) declares that “the long-term protection of the water resources of this state, including sustainable watershed functions, is an essential component of Oregon’s environmental <u>and economic</u> stability and growth” (emphasis added). Moreover, the restoration economy is explicitly acknowledged in OWEB’s mission statement: to “help protect and restore healthy watersheds and natural habitats that support thriving communities and strong economies” http://www.oregon.gov/OWEB/pages/about_us.aspx.

In that spirit, at the inception of the MFIMW ten years ago, OWEB commissioned the development and field-testing of a set of measures to enable socio-economic monitoring to complement bio-physical monitoring (Hibbard and Lurie 2009, 2010). The current study applies those measures to: 1) assess changes in indicators of community socio-economic well-being; and 2) estimate the socioeconomic contribution of MFIMW restoration work to the local community.

Goals and objectives

Overall, the socioeconomic contributions of ecological restoration have been insufficiently examined (Aronson et al. 2010); the restoration economy is more assumed than empirically studied. There is no generally agreed-on set of metrics by which to measure the restoration economy, so there has been no systematic data collection (BenDor et al. 2017, Nielsen-Pincus and Moseley 2013, Aronson et al. 2010). Recent work (e.g., BenDor et al. 2015, Nielsen-Pincus and Moseley 2013, Hibbard and Lurie 2012) advocates for two types of socio-economic measures to more fully understand the local restoration economy.

- A set of **community indicators** that assess overall socio-economic well-being over time and put the restoration economy into context. These are measures collected from existing sources that are sensitive to the effects of restoration work.
- A set of **outcome measures** that estimate the contribution of restoration work to the local economy, based on an inventory and analysis of completed projects.

We used both of these measures in our investigation of the MFIMW and Grant County.

Methods

Community Indicators

To understand the context of the local restoration economy we used a set of indicators of the overall socio-economic well-being of the local community, which we define as Grant County because the MFIMW is totally contained within the county and most available data on socio-economic health are collected at the county level. With respect to the latter point, consistent with most community indicator studies, we depended on existing data for the indicators because it would be both technically difficult and cost-prohibitive to collect original data.

We originally developed the indicators in 2008-09, using a highly participatory four-step process (Hibbard and Lurie 2009, 2010):

1. We organized a small "expert panel" of locally involved people from diverse backgrounds who are known to have a good understanding of how restoration and other watershed activities connect to the socio-economic health of the community.
2. We engaged the expert panel in a workshop process to identify a draft set of measures.
3. We confirmed the technical feasibility of the measures (are the data available and accessible at a reasonable cost in time and money?).
4. We ground-truthed the indicators through a community education/public involvement process.

The present study is based on the previously developed indicators. We reviewed and modified them with counsel from two sources. One is a six member ad hoc expert advisory committee formed specifically for this project. It consists of: Sally Bartlett, Grant County Economic Development Coordinator; Amy Charette, Confdereated Tribes of Warm Springs Watershed Restoration Coordinator; Elaine Eisenbraun, Executive Director, Noeth Fork John Day Watershed Council; Jason Kehrberg, District Manager, Grant Soil and Water Conservation District; Mark Webb, Executive Director, Blue Mountains Forest Partners; and King Williams, Iron Triangle Logging. The other is the UMFWG, an association of agencies, conservation groups, and private landowners who plan, implement, and monitor the effects of restoration efforts in the project area. With their input we settled on a set of measures to assess changes in socio-economic well-being in Grant County and to estimate the socioeconomic contribution of MFIMW restoration work, and then proceeded to collect and analyze the data reported here.

With the expert panel and UMFEG we reviewed the original indicators, dropping one measure and adding two others, as follows:

- Overall population
- School enrollment (added)
- Total employment and job ratio (employment/population)
- Average earnings per job
- Per capita income
- Components of personal income (earned income, property income, transfer payments)
- Employment by major industry
- Housing (added)
- Economic diversification (Hachman Index) (dropped)

The Hachman Index was dropped because it requires a special analysis that is beyond the capabilities of the research team. School enrollments and housing were added at the recommendation of the expert panel as further ways of probing growth, stability, and decline in the local economy. Except where otherwise noted, these indicators are calculated from data taken from the Oregon Regional Economic Analysis Project (OR-REAP), which uses data provided by the U.S. Department of Commerce, Bureau of Economic Analysis.

Outcome Measures

We used four measures to estimate the contributions of the MFIMW to the Grant County economy:

- the change in the number of restoration-related planning, management, and monitoring jobs in Grant County;
- the economic output effects of OWEB capacity grants to organizations in Grant County.
- the employment and economic output effects of OWEB funded restoration projects in the MFIMW; and

- the employment and economic output effects of all restoration projects specifically in the MFIMW area.

As well, we did a sub-analysis of USFS restoration projects in the MFIMW area, since most land in the MFIMW is part of the Malheur National Forest.

Each of these outcome measures has its own data sources and approaches to estimation, so instead of an overall discussion here of the methods we used, we explain them separately as we take them up in the Results section.

Results

Background

The MFIMW is the site of a vigorous effort at environmental restoration, with the aim of returning the sub-basin to a close approximation of its condition prior to the coming of intensive grazing, logging, and mining, re-creating both its structure and function through measures such as stream bank stabilization, revegetation, and restoring meandering channels. The direct intention is to bring back endangered fish species. A hoped-for side effect of the restoration work is socio-economic benefit to the community – the restoration economy. The extent to which that side effect is being realized is the subject of this study.

Community Indicators

Population

In the mid-1980s and again in the last half of the 1990s the overall population of Grant County exceeded 8,000, but it has trended downward over the last fifteen years and is now a little above 7,000 (Table 1). This parallels the trend in rural eastern Oregon overall. The non-metropolitan eastern Oregon[1] share of the state's population has declined from more than eleven percent in 1980 to less than nine percent in recent years.

Table 1. Grant County Population 1970-2015

Year	Grant County Total Population	Grant County Population % of Statewide Total	Nonmetro Eastern Oregon % of Statewide Total
1970	7,095	0.34%	11.56%
1980	8,208	0.31%	11.08%
1990	7,870	0.28%	10.18%
2000	7,906	0.23%	9.82%
2010	7,458	0.19%	9.17%
2015	7,185	0.18%	8.76%

As well, the population is aging. The fraction aged 65 and over increased from 16.8% in 1980 to 28.2% in 2015. And school enrollments have declined across the county, from over 1400 in fall, 2000 to 900 in fall, 2016 (Table 2).

Table 2. Grant County Public School Enrollment, Select Years, 2000-2016

District	2000	2005	2010	2011	2012	2013	2014	2015	2016
Dayville SD 16J	73	68	55	51	60	60	58	52	50
John Day SD 3	982	769	683	671	621	611	617	592	615
Long Creek SD 17	85	60	46	38	37	36	26	28	34
Monument SD 8	72	50	55	45	44	45	48	59	62
Prairie City SD 4	249	151	149	157	149	149	142	148	139
Total	**1461**	**1098**	**988**	**962**	**911**	**901**	**891**	**879**	**900**

Source: Oregon Department of Education, Fall Membership Reports

[1] Non-metropolitan eastern Oregon consists of Baker, Gilliam, Grant, Harney, Malheur, Morrow, Umatilla, Union, Wallowa, and Wheeler counties.

Employment

The decline and aging of the population are counterbalanced by a recent small upturn in employment (see Table 3). The number of jobs in Grant County bottomed out in 2012 at 3,617; by 2015 (the most recent year for which data are available) the job count was up to 3,718. Similarly, the job ratio (the number of people employed/total population) has inched up, from 49% in 2012 to 52% in 2015. Compared to the US as a whole, however, the job ratio has been on a long downward slope, from well above the national average to well below. On the positive side again, the unemployment rate, 7.1% according to the Oregon Employment Department's Research Division, is at its lowest point since comparable records began in 1990.

Table 3. Grant County Employment Change, 1970-2015

Year	Grant County Employment	% of Statewide Total	Grant County Job Ratio	Job Ratio: % of U.S. Average
1970	3,451	0.37%	49%	108.60%
1980	3,760	0.28%	46%	91.32%
1990	4,360	0.27%	55%	99.97%
2000	4,347	0.21%	55%	93.82%
2010	3,780	0.17%	51%	90.61%
2011	3,680	0.17%	50%	87.80%
2012	3,617	0.16%	49%	86.62%
2013	3,655	0.16%	50%	87.20%
2014	3,691	0.16%	51%	87.94%
2015	3,718	0.16%	52%	87.45%

Earnings per job

Incomes are also moving up in Grant County compared to state and national incomes, although they are still relatively low. From 1990 to 2010 average earnings per employed person in Grant County were less than 70% of Oregon as a whole and less than 60% of the U.S. average. By 2015 they were almost 79% of the state average and over 70% of the U.S. average. For comparison, in 2015 eastern Oregon non-metro average earning were 81.9% of the state (see Table 4).

Table 4. Grant County Average Earnings per Job, 1980- 2015

Year	Earnings (Current $)	Grant County % of Statewide Average	Grant County % of U.S. Average	Eastern Oregon Non-Metro % of Oregon Average
1980	$13,823	89.4	88.1	91.9
1990	$17,691	73.8	66.4	82.5
2000	$22,923	62.6	57.3	77.5
2010	$31,082	68.9	59.9	79.7
2011	$33,213	71.8	62.3	80.1
2012	$36,268	74.4	66.0	81.7
2013	$36,466	73.9	65.7	81.3
2014	$39,568	78.1	69.6	81.4
2015	$41,777	78.9	71.8	81.9

Per capita income

Like earnings per job, per capita incomes in Grant County bottomed out in the early years of the new century but have now returned to near 1980-85 levels (see Table 5). In 2015 they were almost 90% of that of Oregon as a whole and slightly over 80% of the U.S. average.

Table 5. Grant County Per Capita Income, 1980-2015

Year	Per Capita Income (Current $)	Grant County % of Oregon Average	Grant County % of U.S Average	Eastern Oregon Non-Metro as % of Oregon Average
1980	$9,046	89.1	89.1	94.0
1990	$15,083	83.5	77.0	83.2
2000	$21,329	74.6	69.7	76.1
2010	$29,270	82.0	72.7	82.7
2011	$31,283	83.7	73.7	82.9
2012	$32,772	83.8	74.0	83.4
2013	$33,546	84.9	75.4	83.5
2014	$36,627	87.9	78.9	83.6
2015	$38,647	88.3	80.3	84.2

Components of Personal Income

The aging of the population noted above is reflected in changes in the sources of personal income – earned income, property income, and transfer payments (see Table 6). Earned income is defined as compensation for labor services, wages and salaries paid for work. Property income represents payments in the form of dividends, interest and rent for the services of capital owned by persons. Transfer payments are payments that are not related to the provision of services. The most important are social security and disability payments. The next largest category is medical payments, programs such as Medicare and Medicaid. Transfer payments, which accrue primarily to the elderly, have more than doubled in Grant County, from about twelve percent of all personal income in 1980 to over 25% in 2015. Over the same period, as the fraction of the population that is working age declined, earned income also declined, from two-thirds of all personal income in 1980 to less than half in 2015. Property income has held steady at 20-25% over that time.

Table 6. Major Components of Personal Income in Grant County, 1980-2015

Year	Earned Income as % of Total	Property Income as % of Total	Transfer Payments as % of Total
1980	64.1	23.5	12.4
1990	57.7	26.9	15.5
2000	53.6	25.4	21.0
2010	47.4	22.2	30.4
2011	47.4	22.4	30.2
2012	49.2	22.5	28.3
2013	48.4	23.1	28.5
2014	49.2	22.2	28.6
2015	49.8	21.8	28.4

Employment by Major Industry

Between the low point in 2012 and 2015, the most recent year for which data are available, Grant County added a total of 101 jobs, an increase of almost three percent (see Table 7). There was growth in both wage and salary employment and “nonfarm proprietors,” people starting their own businesses. Federal employment expanded, the number of state jobs held steady, and local government jobs declined a little. By industry, there was significant change between 2014 and 2015. For example, the number of jobs in “professional, scientific, and technical services” grew quite a bit.

Table 7. Grant County Full-time and Part-time Employment by Major Industry

	Year			
Employment by Place of Work	**2012**	**2013**	**2014**	**2015**
Total Employment	**3,617**	**3,655**	**3,691**	**3,718**
By Type:				
Wage and Salary Employment	2,463	2,465	2,511	2,511
Proprietors Employment	1,154	1,190	1,180	1,207
Farm Proprietors	359	358	353	352
Nonfarm Proprietors	795	832	827	855
By Industry:				
Farm Employment	459	460	453	434
Construction	155	149	S	145
Wholesale Trade	41	45	48	52
Retail Trade	339	338	337	349
Information	88	90	100	66
Finance and Insurance	78	88	80	79
Professional, Scientific, and Technical Services	88	90	100	116
Administrative and Waste Services	146	130	132	97
Other Services (except Public Administration)	167	170	183	186
Federal Civilian	248	266	282	287
Military	20	20	19	18
State Government	124	120	125	129
Local Government	590	565	560	559
Other/Suppressed Industries*	1,065	1,114	1,105	1,201

Housing

Housing data provide valuable insights into socio-economic conditions. First, as with Oregon in general, Grant County households are somewhat more stable than the country as a whole: 94% of those who lived in Grant County in 2015 had lived there the year before (see Table 8). By comparison, about 88% of Americans as a whole live in the same county from year to year.

Table 8. Geographical Mobility in the Past Year for Current Residence, 2015

	Grant County	Oregon
Same House One Year Ago	85%	82%
Moved within Same County	9%	11%
Stable	94%	92%

Source: 2011-2015 American Community Survey 5-Year Estimates, B07003: Geographic Mobility

An important measure of socio-economic well-being is the fraction of household income spent on housing. As general principle, a household should spend no more than one third of their income on housing. Grant County residents measure up pretty well to that rule of thumb, though housing has gotten more expensive in the last five years, especially for renters (see Table 9).

Table 9. Households Spending More Than 35% of Income on Housing[2]

	Oregon 2006-2010	Oregon 2011-2015	Grant County 2006-2010	Grant County 2011-2015
With Mortgage	31%	27%	27%	31%
Without Mortgage	11%	12%	6%	11%
Renters	42%	45%	22%	34%

Source: U.S. Census Bureau, 2011-2015 American Community Survey 5-Year Estimates, B25070 and B25091

Grant County's housing stock is somewhat older than that of Oregon as a whole (see Table 10). This suggests there was a decline in demand for dwellings as the population declined. That may be changing in recent years with the stabilization of the population level.

[2] "With Mortgage" refers to all forms of debt where the property is pledged as security for repayment of the debt. The category "without mortgage" is comprised of housing units owned free and clear of debt.

Table 10. Age of Housing Stock, 2015

	Oregon Number of Units	Oregon Percent of Total	Grant County Number of Units	Grant County Percent of Total
Pre 1970	598,608	35%	2,004	46%
1970 to 1999	814,314	48%	1,772	41%
2000 or later	282,261	17%	544	13%
Total Housing Units	1,695,183	100%	4,320	100%

Source: U.S. Census Bureau, 2011-2015 American Community Survey 5-Year Estimates, B2503

The "profile" of dwellings suggests some interesting points. Although the proportion of single family houses in Grant County (71%) is similar to the state as a whole (68%), there is a much smaller fraction of multi-family units – apartments – in Grant County. This is explained by the much larger fraction of "other" dwellings – mobile homes – in Grant County. Mobile homes are typically much more abundant in small towns and rural areas and Grant County is no exception.

Table 11. Housing Profile, 2015

	Oregon Number of Units	Oregon Percent of Total	Grant County Number of Units	Grant County Percent of Total
Single Family	1,154,878	68%	3,078	71%
Multi-Family	396,724	23%	311	7%
Other	143,581	8%	931	22%
Total Housing Units	1,695,183	100%	4,320	100%

Source: U.S. Census Bureau, 2011-2015 American Community Survey 5-Year Estimates, B25024

Summary

These indicators paint a socio-economic picture of Grant County over the past 40-50 years in which there was a gradual decline, corresponding with the trend for rural eastern Oregon as a whole. However, things have stabilized and even moved up in some ways in recent years. Jobs and earnings are both up and transfer payments to the growing elderly population have also contributed to improved socio-economic well-being. The data on housing stability and costs support this conclusion.

The recovery in jobs and earnings is particularly relevant. The indicators cannot directly capture the effect of restoration work in the MFIMW (that is reported in the Outcome Measures section below), but it suggests a local economic revival of which the restoration economy is a part.

Outcome measures

In this section we estimate the contributions of the restoration economy to Grant County. Building on the approaches developed for the preliminary study in 2009 (Hibbard and Lurie 2010), and drawing on more recent research (Nielsen-Pincus and Moseley 2013, BenDor et al. 2015), we use four measures to assess the socio-economic effects of MFIMW restoration and monitoring activities:

- the change in the number of restoration-related planning, management, and monitoring jobs in Grant County;
- the economic output effects of OWEB capacity grants to organizations in Grant County; and
- the employment and economic output effects of OWEB funded restoration projects in the MFIMW; and
- the employment and economic output effects of all restoration projects specifically in the MFIMW area.

Restoration-Related Planning, Management, and Monitoring Jobs in Grant County

The MFIMW restoration effort embodies a wider effort across Grant County – indeed across the Pacific Northwest. The number of organizations concerned with restoration in Grant County increased from 10 to 13 between 2000 and 2016, and the number of people they employ in restoration planning, management, and monitoring increased from 52 (46.55 full time positions) to 97 (83.5 full time positions). These data (see Table 12) are based on interviews with key personnel in relevant organizations in 2010 and 2017. The numbers include full-time and regularly recurring part-time and seasonal jobs. They do not include contract workers or paid jobs that function as internships or other learning opportunities because we were not able to assemble comprehensive data on such positions. However, there may be a significant number of them. For example, in summer 2017, the NFJDWC expected to employ 61 local students representing 6.25 annual FTE.

Table 12. Number of Restoration Related Planning, Management, and Monitoring Jobs in Grant County: 2000, 2009 and 2016

Organization	2000		2009		2016	
	FTEs	Employees	FTEs	Employees	FTEs	Employees
Grant County Soil and Water Conservation District	4.00	5	7.50	8	8.00	10
North Fork John Day Watershed Council	1.50	2	3.75	4	5.50	6
Confederated Tribes of Warm Springs	2.00	2	6.40	10	18.90	20
Oregon Department of Fish and Wildlife	27.25	29	30.50	33	33.00	37
The Nature Conservancy	1.00	1	2.50	3	0.00	0
USDA Malheur National Forest	6.00	6	7.00	7	10.00	10
USDI Bureau of Reclamation	0.00	0	1.00	1	1.00	1
South Fork John Day Watershed Council	0.00	0	0.50	1	1.20	2
Natural Resources Conservation Service	1.50	2	2.00	2	2.00	2
Monument Soil and Water Conservation District	3.00	3	1.00	1	2.00	2
Oregon Department of Forestry	0.00	0	0.00	0	0.45	3
Oregon Water Resources Department	0.30	2	0.30	2	0.30	2
USDA Farm Service Agency	Unknown	Unknown	Unknown	Unknown	0.10	1
Blue Mountains Forest Partners	0.00	0	1.00	1	1.00	1
Totals	**46.55**	**52**	**62.45**	**73**	**83.5**	**97**

OWEB Capacity Grants

OWEB provides capacity grants – basic operating funds – to watershed councils and soil and water conservation districts (SWCD) around the state. Hibbard and Lurie (2006) calculated a multiplier[3] of 5.09 for OWEB capacity grants. That is, every OWEB capacity grant dollar generates an additional $5.09 for the local community.

Relevant to the MFIMW, from 7/1/07 to 6/30/17, the NFJDWC received $521,575 in capacity grants and the Grant County SWCD $755,575, for a

[3] A multiplier is the factor by which gains in total output are greater than the change in spending that caused it.

total of $1,277,150. Considering the multiplier, the capacity grants added about $6.5m (1.3x5.09) to the local economy.

Estimating the Employment and Economic Output Effects of Restoration Projects

Identifying the employment and economic output effects of restoration work is of great interest to policy makers and natural resource managers. For this study, we collected data from OWEB files on grants they awarded for restoration projects in the MFIMW area during the period of interest (7/1/07 through 6/30/17) and then applied multipliers derived specifically for Oregon's restoration economy (Nielsen-Pincus and Moseley 2013) to produce estimates of the overall effects on employment and economic output. Finally, we extrapolated from that to estimate the overall socio-economic effects of all restoration work in the MFIMW area over the same period.

It is important to keep in mind that we are looking only at the limited geographic area of the MFIMW. Much additional restoration work was performed across the John Day basin in Grant County between 2007 and 2017. This study does not reflect the socio-economic impacts of those investments.

OWEB Grants in the MFIMW Area

OWEB made 21 grants for MFIMW restoration projects in the period of interest, most in partnership with other funders. We were able to obtain detailed data on 19 of those grants, which provided a total of $2,644,919.

- Several important points underlie those numbers.
- The 19 grants for which we have data led to 33 contracts for restoration work (Table 13). Of these, 23 (70%) were with Grant county organizations; 18 of the 23 (78%) were private firms. As well, over half the dollar value of the contracts (54%) was with Grant County organizations. That figure would be 65.9% except for one very large contract with an out-of-county non-profit organization.
- The largest Grant County contract was for $381,446; the smallest was for $225; and the average was $40,883. The largest out-of-county contract was for $422,536; the smallest was for $5,020; and the average was $44,526. Overall, the average contract was $42,363.

Table 13. Summary of MFIMW Contracts from OWEB Grants by Location and Dollar Size

Location by county	# of Contracts	$ total	% of total $
Benton	1	$23,709	1.03%
Crook	1	$219,008	9.6%
Deschutes	1	$112,868	4.9%
Grant	23	$1,231,255	53.7%
Multnomah	3	$433,734	18.9%
Union	3	$188,097	7.98%
Out-of-state (WA)	1	$88,000	3.8%
Totals	**33**	**$2,291,671**	**99.91%**[4]

Table 14. Summary of MFIMW Material/Supply Purchases from OWEB Grants by Location and Dollar Size

Location by county	# of purchases of materials/supplies	$ total	% of total $
Baker	2	$189	0.08%
Benton	1	$100	0.04%
Deschutes	2	$23,828	9.6%
Grant	10	$112,648	45.4%
Multnomah	3	$17,080	6.9%
Umatilla	2	$10,838	4.4%
Union	3	$49,525	19.97%
Wallowa	1	$33,735	13.6%
TOTALS	**24**	**$247,943**	**99.99%**[5]

- Of the $2,644,919, 93% ($2,459,775) was expended directly on restoration activities and 7% ($185,144) was used for project management, fiscal administration, and post-implementation work.
- The $2,644,919 for the 19 OWEB grants was leveraged with $5,457,365 in cash match and $1,876,680 in in-kind match contributed by partners participating in the OWEB funded restoration projects. Summing the grant awards and the cash and in-kind match, total project costs were $9,978,964. The total cost of the two grants for which we do not have detailed data was $285,610 ($132,900+$152,710). Adding this in, the overall project costs of the OWEB grants in the MFIMW area was $10,264,574.

[4] Does not equal 100% because of rounding.
[5] Does not equal 100% because of rounding.

The 19 OWEB grants also entailed the purchase of materials and supplies. Forty percent (10/24) of the vendors were located in Grant County and 45% of the dollar value of the purchases was with them (see Table 14).

Applying the 65.9% of contracting dollars spent in Grant County to the overall project costs ($10,264,574), we calculate a direct economic effect of approximately $6,764,354 in Grant County – $6.8M in round numbers – from the OWEB grants.

Multipliers developed by Nielsen-Pincus and Moseley (2013) allow us to use that number to estimate the employment and economic output effects of the MFIMW restoration projects. They calculated that each million dollars invested in restoration produces an average of 16.3 total jobs (direct, indirect, and induced).[6] Thus, in the ten years of the MFIMW the $6.8M in restoration activities related to the OWEB grants produced an estimated 111 jobs (6.8x16.3).

Further, Nielsen-Pincus and Moseley calculated that each million dollars of restoration work produces additional economic output of $1.9 to $2.4 million. Over the ten years of the MFIMW, the $6.8M in restoration activities related to the OWEB grants produced an estimated additional economic activity of $12.9 to $16.3M ($6.8x$1.9 to $6.8x$2.4).

In summary, a majority of the expenditures from OWEB grants to support MFIMW restoration work went to Grant County contractors and suppliers. Those expenditures rippled through the economy to produce more than 100 jobs and $12-16M in economic output.

Extrapolating to all MFIMW Projects

During the period of interest a total of 100 restoration projects were carried out in the MFIMW area, **including** the 21 OWEB-funded projects. Detailed financial data are not available for any of the non-OWEB grants that supported these projects. A further concern is that the reported project costs are much lower than for the OWEB projects, and varies from project to project. For example, in-kind match is included in some reports but not in others. As well, data are missing for 20 of the 100 MFIMW projects. Thus, our estimates of the socio-economic effects of all MFIMW projects are systematically low, nor are they as precise as for OWEB-funded projects only. With those limitations in mind, we approximated the employment and economic output effects of all MFIMW restoration projects.

The total cost of the 80 projects for which we have data was $15,600,126, $15.6M in round numbers. We would expect this to be much larger. It is only about 40% more than we found for the 19 OWEB-funded

[6] For Nielsen-Pincus and Moseley, a job entails doing the direct, on-the-ground work of forest and watershed restoration projects.

projects, even though it includes four times as many projects (19 vs. 40). Thus, the following analysis is probably very low.

- Applying the 65.9% of contracting dollars spent in Grant County (calculated from the OWEB data) to the estimated overall project cost of $15.6M, we estimate a direct economic effect to Grant County from the MFIMW projects of at least $10,280,400 – $10.3M in round figures.
- Using the Nielsen-Pincus and Moseley calculation that each million dollars invested in restoration produces an average of 16.3 total jobs, we estimate that MFIMW projects produced at least 168 jobs (16.3x10.3).
- Using the Nielsen-Pincus and Moseley calculation that each million dollars of restoration work produces additional economic output of $1.9 to $2.4 million, over the ten years of the MFIMW, the $10.3M in MFIMW restoration activities produced estimated additional economic activity of at least $19.6 to $24.8M ($10.3x$1.9 to $10.3x$2.4).

US Forest Service Projects

Much of the MFIMW area is on the MNF and the USFS is very active in implementing various regional and national restoration programs (CFLRP, Accelerated Restoration). It is therefore of interest to examine separately the USFS projects that are included in the 100 MFIMW projects carried out during the period of interest.

USFS projects vary in contract opportunities for design and implementation but in structure and composition they resemble other MFIMW restoration projects. To take some examples:

- The USFS contracted the design and implementation of the MFJDR Historic Meander Reconnection Project.
- Many of the USFS Aquatic Organism Passage Projects were designed by USFS engineers and implemented by local contractors, until staff turnover occurred and the designs were also contracted out.
- Fencing projects are contracted out and provide opportunities for jobs. These projects likely resulted in maximizing grant dollars to restoration on the ground.

Thus, it is reasonable to apply the Nielsen-Pincus and Moseley multipliers to the USFS projects. The restoration inventory that we used documented a total of 48 projects that the USFS carried out in the MFIMW area during the period of interest, at a total cost of more than $4.5M (cost information was not available for three projects). Again applying the 65.9% of contracting dollars spent in Grant County (calculated from the OWEB data), we estimate that the $4.5M total cost of MNF projects produced about $3M in contracting dollars spent in Grant County ($4.5x.659).

- Using the Nielsen-Pincus and Moseley calculation, the $3M in restoration activities on the MNF produced about 49 jobs (3x16.3).

- The $3M in restoration activities produced estimated additional economic activity of at least $5.7 to $7.2M ($3x$1.9 to $3x$2.4).

In addition to the restoration work in the MFIMW, the USFS has extensive restoration planning and implementation activities across the MNF. Most of the restoration implementation work identified during planning is conducted through the Malheur's 10 Year Stewardship Contract with local contractor, Iron Triangle. While this is beyond the scope of our analysis, it further reflects the importance of the USFS contribution to the restoration economy in Grant County.

Interpretation of findings

Grant County is doing better, socio-economically, in recent years. At the same time, MFIMW projects are making a significant contribution in jobs and dollars. While it is methodologically impossible to demonstrate a direct a cause-and-effect relationship, the MFIMW restoration economy has almost certainly contributed to the increase in jobs and earnings and thus to the socio-economic well-being of Grant County.

Discussion

Summary of Results

We began our work by collecting data on a set of community indicators of the overall socio-economic well-being of Grant County. We used a highly participatory process to identify indicators that measure features of importance to the local community. The purpose of the indicators is to establish a context, a description of the community that helps to interpret the outcome measures.

Based on the indicators they selected, people in Grant County are highly concerned about the trajectory of their economy – where it has been and where it might be heading. Over the past 40-50 years Grant County was in decline. However, things are improving recently. Most relevant to this study, jobs and earnings are both up, and other indicators support that trend.

The indicators cannot specifically capture the role of MFIMW restoration work in advancing the apparent local resurgence, but the two are moving in parallel. As measured by the community indicators, the Grant County economy is doing better at the same time that restoration work is bringing work and money into the economy. With respect to restoration work, we found that:

- The number of restoration related planning, management, and monitoring jobs in Grant County nearly doubled between 2000 and 2016.

- OWEB capacity grants – basic operating funds for watershed councils and SWCDs have brought a total of $1,277,150 to Grant County since 2007. When the multiplier of 5.09 is considered, capacity grants brought about $6.5m to the local economy.
- Many restoration projects have been carried out in the John Day basin in the period from 7/1/07 to 6/30/17. Of interest for this project are the 100 projects in the MFIMW. We collected data from OWEB to give us an empirical base from which to make projections on all the MFIMW projects and separately, on MNF projects in MFIMW area. The results are summed up in Table 15. (Note that these figures are not summative, they overlap in undetermined ways.)

Table 15. Summary of Jobs and Additional Economic Activity in Grant County from MFIMW Projects 7/1/07-6/30/17

Organization	All-projects total cost	Jobs (direct, indirect, and induced)	Additional economic activity generated
OWEB-19 projects	~$10.3M	111	$12.9-$16.3M
All projects-80[7]	at least $15.6M	168	$19.6-$24.8M
MNF-48	~$4.5M	49	$5.7-$7.2M

Exportability

This project reinforces the guiding principles for monitoring the community socio-economic effects of ecosystem restoration discussed above. The measures are context-specific. Both the indicators and outcome measures were developed in consultation with local officials and residents, to gauge metrics that are important to them. The measures can be used to inform the general public about the socio-economic contribution of restoration efforts and as an input to public decision making and action. They also make it possible to consider the effect on the local economy as private landowners contemplate decisions about whether or not to engage in restoration work on their property. Thus, though the measures are not generalizable to other restoration efforts, the process of developing and applying measures is exportable and this project contributes to the small but growing body of literature that is seeking to develop a framework for socio-economic monitoring of restoration efforts.

Lessons Learned

The difficulty of linking economic activity directly to outcomes points to two important issues. First, a significant reason why more socio-economic

[7] As noted on p. 21, we have data on only 80 of the 100 total MFIMW projects.

outcome studies may not be undertaken is that analyzing impacts is not a straightforward, formulaic process. Second, the situation highlights a need for those who want to demonstrate the connection between ecosystem restoration and contributions to local economies to collectively establish guidelines for how to best track and analyze those connections before restoration actions are implemented. Such discussions could provide guidance on what types of data might be common across different contexts and how to extrapolate from unique characteristics in restoration project areas.

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