



Research article

Can stream and riparian restoration offset climate change impacts to salmon populations?

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ARTICLE INFO

Article history:

Received 12 May 2016

Received in revised form

28 November 2016

Accepted 5 December 2016

Available online 14 December 2016

Keywords:

Riparian

Climate change

Water temperature

Restoration

Chinook Salmon

ABSTRACT

Understanding how stream temperature responds to restoration of riparian vegetation and channel morphology in context of future climate change is critical for prioritizing restoration actions and recovering imperiled salmon populations. We used a deterministic water temperature model to investigate potential thermal benefits of riparian reforestation and channel narrowing to Chinook Salmon populations in the Upper Grande Ronde River and Catherine Creek basins in Northeast Oregon, USA. A legacy of intensive land use practices in these basins has significantly reduced streamside vegetation and increased channel width across most of the stream network, resulting in water temperatures that far exceed the optimal range for salmon growth and survival. By combining restoration scenarios with climate change projections, we were able to evaluate whether future climate impacts could be offset by restoration actions. A combination of riparian restoration and channel narrowing was predicted to reduce peak summer water temperatures by 6.5 °C on average in the Upper Grande Ronde River and 3.0 °C in Catherine Creek in the absence of other perturbations. These results translated to increases in Chinook Salmon parr abundance of 590% and 67% respectively. Although projected climate change impacts on water temperature for the 2080s time period were substantial (i.e., median increase of 2.7 °C in the Upper Grande Ronde and 1.5 °C in Catherine Creek), we predicted that basin-wide restoration of riparian vegetation and channel width could offset these impacts, reducing peak summer water temperatures by about 3.5 °C in the Upper Grande Ronde and 1.8 °C in Catherine Creek. These results underscore the potential for riparian and stream channel restoration to mitigate climate change impacts to threatened salmon populations in the Pacific Northwest.

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1. Introduction

The warming effects of climate change and land use on streams threaten to drastically reduce fish distribution and viability throughout the Pacific Northwest (Beechie et al., 2013) and across the globe (Ficke et al., 2007). Human alterations to the atmosphere and landscape can influence water temperature by changing one or more of the primary factors that regulate stream temperature, including climatic drivers (e.g., air temperature and precipitation), discharge, stream morphology, groundwater interactions, and riparian canopy condition (Poole and Berman, 2001). Human-caused CO₂ emissions have contributed to a significant warming trend in Pacific Northwest streams during summer of approximately 0.22 °C/decade between 1980 and 2009 (Isaak et al., 2012),

and August stream temperatures are projected to increase on average, +2.83 °C by the 2080s (Isaak et al., 2015). In addition to climate impacts, increases in water temperature can result from decreased streamflow, simplification of stream channels (e.g., increased width-to-depth ratio and reduced hyporheic exchange), and reduction of riparian vegetation cover (i.e., increased solar radiation reaching the stream) (Poole and Berman, 2001). These modifications are often the consequence of land use activities such as water diversions for irrigation or urban use, tree harvest in riparian zones (Beschta et al., 1987; Moore et al., 2005), poorly managed livestock grazing (Kauffman and Krueger, 1984; Belsky et al., 1999), and stream channelization associated with construction of roads, levees, and other impediments (e.g., mine tailings) (Simon and Rinaldi, 2006).

Water temperature is widely recognized as one of the most important environmental factors influencing the geographic distribution, growth, and survival of fish and other aquatic organisms (Regier et al., 1990; Armour, 1991; McCullough, 1999). Temperature

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can directly affect physiological processes such as cardiorespiratory performance, food consumption, and osmoregulation (Whitney et al., 2016), as well as migratory behavior, resistance to disease and parasites, and inter- and intra-specific competitive interactions (Armour, 1991; Lynch et al., 2016). In addition, fish will often exhibit thermoregulatory behavior to optimize physiological performance, such as seeking out cold water refuges when ambient temperatures approach stressful levels (Breau et al., 2011; Myrick and Cech, 2004). As stream temperature regimes change in response to land management and climate change, cold-water fishes such as Chinook Salmon (*Oncorhynchus tshawytscha*) and steelhead (*O. mykiss*) may be exposed to temperatures that are outside of their physiologic optimum, resulting in changes to fish communities and potential increased risk of extinction (Poole et al., 2001; Urban, 2015).

Salmon populations are an important cultural, economic and food resource for indigenous tribes and others in the Columbia River basin and throughout the Pacific Northwest. Tributary and estuarine habitat degradation, combined with other factors such as hydroelectric operations in the mainstem Snake and Columbia rivers, predation, and commercial and sport fishing contributed to the decline and subsequent listing of numerous Columbia River basin salmon populations under the Endangered Species Act (ESA) (NOAA, 2008). Recovery of these salmon populations will require a comprehensive management approach that addresses all limiting factors to salmon viability including tributary and estuarine habitat, hydropower impacts, and predation. However, recent emphasis and considerable expense has been directed at restoration of tributary habitat conditions as a means to mitigate for hydropower impacts to threatened salmon populations (BPA, 2008; NOAA, 2008). The extent to which habitat restoration can achieve this goal, particularly in the context of warming stream temperatures due to climate change, has been identified as a critical uncertainty in the Columbia River basin (ISAB/ISRP, 2016).

Given the threat that high water temperature poses on fish populations throughout the Pacific Northwest (Beechie et al., 2013) and across North America (Lynch et al., 2016), it is important to understand the extent to which stream and riparian restoration activities can mitigate future water temperature increases due to climate change (Bernhardt et al., 2005; Ficklin et al., 2014). While similar studies in the Pacific Northwest have examined the potential influence of riparian and channel restoration on water temperature (Chen et al., 1998; Sullivan and Rounds, 2004; Watanabe et al., 2005; Butcher et al., 2010), few have integrated riparian restoration with climate change projections to evaluate whether restoration actions would be sufficient to offset climate change impacts (Battin et al., 2007; Bond et al., 2015). Our research additionally draws on fish-habitat relationships developed from empirical fish and habitat data to evaluate how fish populations would respond to simulated changes in water temperature. Integrating predicted changes in habitat conditions with fish population response provides a critical link needed by natural resource managers to evaluate the potential benefits of restoration actions and to plan and adjust management decisions accordingly.

We used a water temperature simulation model to investigate potential thermal benefits of riparian reforestation and channel narrowing in context of future climate change to Chinook Salmon populations in the Upper Grande Ronde River and Catherine Creek in Northeast Oregon. Our specific objectives were 1) to simulate water temperature changes that may result from restoration actions, 2) to evaluate whether future climate change impacts could be offset by riparian and channel restoration actions, 3) to predict how simulated temperature changes would influence the abundance of Chinook Salmon summer parr, and 4) to develop a tool that can be used by restoration planners and practitioners to investigate alternative land-management strategies and prioritize

restoration actions.

2. Methods

2.1. Study area

The Grande Ronde River is a major tributary of the Snake River, originating in the Blue Mountains of NE Oregon and flowing approximately 340 km north/northwest before joining the Snake River in SE Washington. The study area included select reaches of the Upper Grande Ronde Basin (UGRB), which is located upstream of the Catherine Creek confluence near the city of La Grande, and the Catherine Creek Basin (CCB), a large tributary of the Grande Ronde River (Fig. 1). The UGRB and CCB drain areas of approximately 1896 and 1051 km², respectively. This area is typified by cold winters with ample snow in its headwaters areas, and hot, dry summers. Basin tributaries are primarily fed by snowmelt, with peak flows occurring during the spring, and base flows occurring during the late summer. Due to the relatively lower elevation of headwater peaks in the UGRB compared with CCB, snowmelt generally occurs earlier in the UGRB, often resulting in very low summer base flows and warmer water temperatures.

Habitat for fish and other aquatic life in the Grande Ronde basin has been steadily degraded since the mid-1800s due to land use, with water temperature being arguably one of the most impaired and influential factors for ESA-listed Chinook Salmon, steelhead, and bull trout in the basin. The Environmental Protection Agency (EPA) established a set of temperature water quality standards for the Pacific Northwest region to protect threatened salmonids which include a maximum weekly maximum temperature of 16 °C for juvenile salmon/trout rearing, 18 °C for salmon/trout migration plus non-core rearing, and 20 °C for salmon/trout migration (EPA, 2003). As of 1999, approximately 92% of the Grande Ronde River upstream of the Wallowa River confluence exceeded the 18 °C temperature standard (ODEQ, 2000).

This study focused on two threatened salmon populations within the Snake River spring/summer Chinook Evolutionary Significant Unit (ESU), the Upper Grande Ronde River Spring Chinook and Catherine Creek Spring Chinook. These focal populations were chosen because of the perceived large juvenile life-stage survival gaps due to habitat impairments and because of the existence of high quality fish and habitat monitoring data.

2.2. Temperature model

We used a deterministic water temperature model, Heat Source (Boyd and Kasper, 2003), to simulate water temperature and flow dynamics in major salmon-bearing streams of the UGRB and CCB (Fig. 1). Heat Source uses stream channel geometry, hydrology, climatic conditions, and riparian vegetation cover and height to simulate stream temperature and effective shade at 100 m intervals (termed model nodes) throughout the stream network. The Heat Source model was selected because it has been applied extensively throughout Oregon (ODEQ, 2000; Crown et al., 2008; Watershed Sciences, 2008; Butcher et al., 2010) and elsewhere in the Pacific Northwest to evaluate compliance with water temperature standards, and because it's well suited to simulating the effects of riparian vegetation on stream temperature at a fine spatial resolution—a feature that is useful for restoration prioritization.

Model inputs including channel topography (i.e., stream width and gradient) and riparian vegetation (canopy height and density) were measured using light detection and ranging (LiDAR) data collected in 2009. Climatic data, including air temperature, cloud cover, relative humidity, and wind speed, were recorded by the National Weather Service at the La Grande airport and by the US

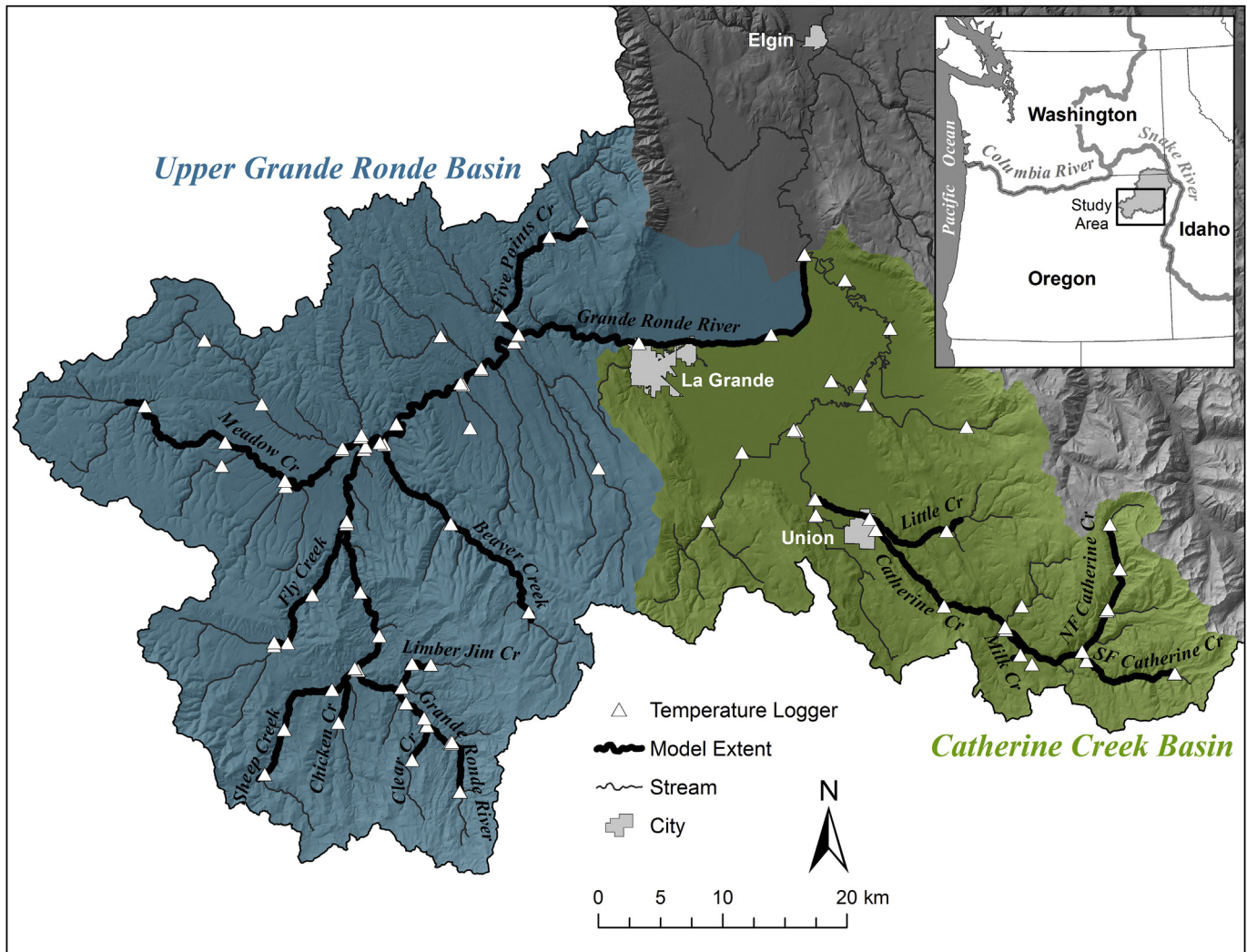


Fig. 1. Map of the study area in the Upper Grande Ronde River and Catherine Creek basins in NE Oregon showing the stream segments that were modeled using Heat Source.

Forest Service at the J Ridge weather station in the UGRB. Streamflow inputs were obtained from nearby gaging stations and manual streamflow measurements collected at discrete locations throughout the stream network. Water temperature was measured using HOBO temperature loggers placed at model boundaries and tributary junctions (Fig. 1). In addition, remotely-sensed forward looking infrared (FLIR) surveys were conducted during summer of 2010 to capture a snapshot of peak daily water temperatures throughout the stream network (Watershed Sciences, 2010). The model was calibrated to conditions observed during 2010 by adjusting model parameters such as channel roughness and hyporheic exchange to minimize the deviation between observed and predicted water temperatures.

The model extent was intended to represent all stream reaches within current and historic Chinook Salmon spawning and rearing areas in the UGRB and CCB. However, some portions of the stream network were excluded from the temperature model because low stream flows precluded accurate temperature modeling with Heat Source (e.g., lower Catherine Creek). Modeling of water temperatures downstream of the Catherine Creek confluence (i.e., migration corridor to the Pacific Ocean) was beyond the scope of this study. A complete description of model development procedures is provided in Watershed Sciences (2012).

Model parameters from the calibrated 2010 model (hereby referred to as current conditions) were used as a baseline for evaluating a suite of model scenarios representing different assumptions about potential riparian restoration, channel morphology restoration, and climate change (Table 1). The model provides hourly temperature predictions for a 10-week period between July 10 and September 20, 2010, a time frame chosen to represent summer base flow conditions when water temperatures are typically highest and salmonids are consequently at risk. For consistency with EPA water quality standards (EPA, 2003), we calculated the maximum 7-day running average of the daily maximum temperature in degrees Celsius, hereby referred to as maximum weekly maximum temperature (MWMT) for each model node. Water temperature predictions were summarized by basin (i.e., UGRB and CCB) to simplify model outputs and aid in interpretation of the results.

Examination of historical air temperature and streamflow data indicated that 2010 was a fairly average year and was therefore a reasonable baseline for comparison with model scenarios investigating future restoration benefits. For example, average summer air temperature (July–September) measured in the city of La Grande between 1965 and 2010 ranged from 13.9 to 21.4 °C (mean = 18.2 °C), compared with an average summer temperature

Table 1

Description of ten model scenarios used to evaluate the influence of riparian restoration, channel narrowing, and climate change on water temperature.

Model number	Model name	Model description
1	Current	Baseline model calibrated using 2010 temperature, climate, vegetation, and hydrologic conditions
2	PNV	Vegetation across the entire model extent set to potential natural vegetation (PNV) cover and height.
3	High Priority	Vegetation in high priority areas set to PNV and other areas set to current conditions.
4	High/Med Priority	Vegetation in high and medium priority areas set to PNV and all other areas set to current conditions.
5	High/Med/Low Priority	Vegetation in high, medium, and low priority areas set to PNV and very low priority areas set to current conditions.
6	Width	Channel width set to historic conditions and vegetation set to current conditions.
7	Width/PNV	Channel width set to historic conditions and vegetation set to PNV.
8	Climate 2080s	Air temperature and streamflow set to 2080s climate projections.
9	Climate 2080s/Veg	2080s climate projections and vegetation set to potential cover and height at 75 years.
10	Climate 2080s/Veg/Width	2080s climate projections, vegetation set to potential cover and height at 75 years, and channel width set to historic conditions.

of 17.4 °C in 2010 (2010 percentile = 27%). Similarly, average summer streamflow (July–September) in the Grande Ronde River near Perry between 1993 and 2014 ranged from 0.6 to 6.8 m³/s (mean = 1.9 m³/s), while summer streamflow in 2010 averaged 1.6 m³/s (2010 percentile = 57%).

2.3. Riparian restoration

To simulate restoration of riparian vegetation, we first needed to estimate the potential height and canopy cover of trees and shrubs in the riparian zone under natural historic conditions (i.e., prior to intensive anthropogenic disturbance). To do this, we assembled a team of riparian ecologists with extensive experience in the Blue Mountains region to develop a detailed map of current vegetation and potential natural vegetation (PNV) for a 100-m wide stream buffer (each side of stream centerline) throughout the Chinook-bearing portions of the Upper Grande Ronde and Catherine Creek watersheds that incorporates physiography, geomorphology, soils, vegetation, and disturbance (Wells et al., 2015).

Potential tree and shrub canopy cover was estimated from extensive field measurements of canopy cover collected within each plant association group (PAG) as documented in local vegetation classifications (Kauffman et al., 1985; Johnson and Simon, 1987; Johnson and Clausnitzer, 1992; Crowe and Clausnitzer, 1997; Crowe et al., 2004; Wells, 2006). Specifically, potential tree and shrub cover were estimated independently by summing the product of constancy (proportion of total vegetation plots in which a species occurred) and average cover (percent canopy cover for a species averaged over all vegetation plots where the species occurred) across all species within each PAG.

Potential tree height was estimated from species-specific dominant tree height growth curves from regional forestry literature (Dahms, 1975; Clendenen, 1977; Barrett, 1978; Herman et al., 1978; Cochran, 1979, 1985; Monserud, 1985; Nussbaum, 1996). Within each PAG, we calculated a weighted-average growth curve by averaging all species-specific growth curves weighted by the average canopy cover value for each species. We used these growth curves to estimate the average tree height under fully restored PNV conditions (model scenario 2). Height at 300 years was assumed to represent the maximum potential tree height due to limitations with extrapolating the growth curves too far beyond the range of the data and because tree growth beyond 300 years is minimal. For scenarios that combined climate change impacts with riparian restoration (i.e., scenarios 9 and 10), we estimated tree height at 75 years from current to correspond approximately with the time frame for which climate projections were available. Tree height and cover for these scenarios represents a snapshot at 75 years along the trajectory toward the PNV condition.

Potential shrub heights were obtained from Steele and Geier-

Hayes (1987, 1989, 1992, 1993, 1994) and from species descriptions in the Fire Effects Information System (USFS, 2015). Shrub heights were weighted by average cover of each species within each PAG to produce an average potential shrub canopy height.

To determine how quickly water temperatures could be reduced by riparian restoration, we ran a series of model scenarios (results not shown) representing maximum vegetation growth at 25, 50, 75, and 100 years from current, assuming that trees were planted at maximum density in year 1. For each of these scenarios, we calculated the percentage of maximum temperature change, assuming that the maximum temperature change was represented by the difference between the current and PNV (i.e., tree height and density at 300 years) scenarios.

2.3.1. Restoration prioritization

We developed four riparian restoration scenarios (model scenarios 2–5) by dividing the stream network into priority areas (high, medium, low, and very low priority). We started by weighting each model node by its distance from current spawning areas. For each model node i , the distance weight (DW_i) was given by the formula:

$$DW_i = e^{-0.075 \times d_i} \quad (1)$$

where d_i is the distance in kilometers from the model node to the boundary of the nearest spawning area. This weighting scheme was based on a “restore from the core” perspective in which the most productive areas are restored first, and additional areas are subsequently restored moving in a downstream (i.e., less productive) direction. Sites within the current spawning area received a weight of 1, and the weight declined exponentially to near-zero as the distance from spawning increased to about 60 km. This formula was developed subjectively to conform to our general assumption that the value of riparian restoration to salmon population viability would decline steeply as the distance from productive spawning and rearing habitats increased, with most of the benefit occurring within a distance of 10–20 km from the current spawning extent and very limited benefit (i.e., $DW_i < 0.05$ wt) beyond a distance of 40 km.

Next we assigned each model node a weight based on the shade deficit (i.e., the difference between current and potential effective shade). Effective shade is a simulated output from the Heat Source model and is defined as the fraction of the total solar radiation that is blocked by streamside vegetation. Sites with the greatest difference between current and potential shade received a weight of 1, and the weight declined to zero as the shade deficit decreased. This approach essentially assigns the greatest weight to areas with the largest shade deficit, and thus the greatest potential to benefit from

riparian restoration. The shade weight (SW_i) at each model node i was given by the formula:

$$SW_i = (SP_i - SC_i) / SDMax, \quad (2)$$

where SP_i is the effective shade under fully restored PNV conditions, SC_i is the effective shade under current conditions, and $SDMax$ is the maximum difference in effective shade between PNV and current conditions across all model nodes. Shade differences were divided by $SDMax$ to ensure that the shade weights ranged from 0 to 1, consistent with the distance weights.

A final integrated weight for each model node was then calculated as the product of the distance and shade weights. Final weights were then averaged over 1 km stream segments and simplified into categories of high, med, low, and very low priority (Fig. 2). Model nodes with final weights that were at or above the 75th percentile of all final weight values were assigned to the high priority category. Model nodes between the 50th and 75th percentile were assigned to the medium category. Model nodes with weights between the 25th and 50th percentile were assigned as low priority, and nodes with weights below the 25th percentile were assigned as very low priority. The resulting four riparian restoration scenarios were implemented in Heat Source by setting the tree and shrub cover and height values to their maximum

potential within selected priority areas including: 1) high priority, 2) high and medium priority, 3) high, medium and low priority, and 4) high, medium, low, and very low priority (i.e., PNV).

2.4. Channel width

We simulated the influence of channel narrowing and deepening—a common restoration goal—on water temperature by adjusting channel width inputs in Heat Source to correspond with observed historical changes in stream width (model scenarios 6, 7 and 10). To define a historical baseline for channel width, we described changes to river channel widths since the late 1800s, with expectations that the magnitude of change would be greater in areas with more intense ranching, logging, agriculture, and other forms of land use. Historical estimates of river width were based on General Land Office (GLO) surveys (*Principle Clerk of Surveys, General Land Office, 1855*) conducted in the mid to late 1800s. Contemporary estimates of river width (1990s and 2000s) were based on Oregon Department of Fish and Wildlife's (ODFW) Aquatic Inventories Project (AIP) (*Moore et al., 2008*), a spatially continuous survey of common fish habitat characteristics across the river network.

Observed changes in channel width since the historical period

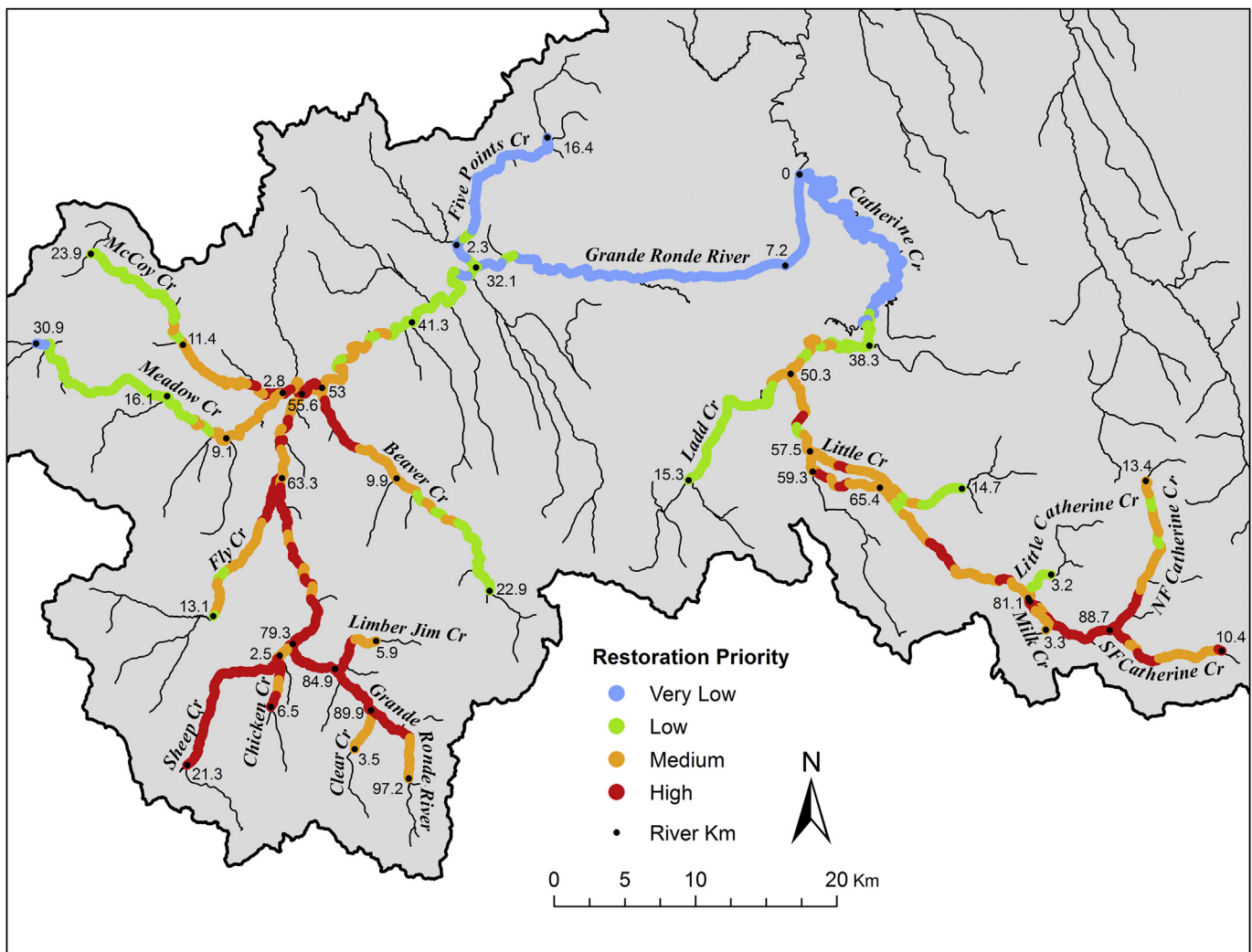


Fig. 2. Map of restoration priority areas in the Upper Grande Ronde and Catherine Creek basins.

were summarized using a geomorphic valley setting classification. The classification system consisted of dividing the stream network into small and large streams using an 8-m bankfull width criterion (Beechie and Imaki, 2014). Next, we divided the stream network into three valley types based on valley confinement (laterally unconfined, partly confined, and confined) following the methodology described in the River Styles Framework (Brierley and Fryirs, 2005). We simplified the classification into three classes for this analysis: (1) large streams, (2) small/partly confined and confined streams, and (3) small/laterally unconfined streams. For each class, we calculated the average percentage change in channel width from historic to current (i.e., large streams = 45.9%, small/partly confined and confined streams = 31.2%, small unconfined streams = 115.8%), and used these average values to simulate restoration of channel width across the stream network. Due to model constraints limiting the amount of channel narrowing possible for a given discharge, we fixed the % increase from historic to current for mainstem Catherine Creek to 25%. A detailed description of the channel width analysis used to support this modeling work is provided in McCullough et al. (2016).

2.5. Climate change

Air temperature and streamflow inputs to the temperature model were modified to predict how future climate change would influence water temperature (model scenarios 8–10). Projected changes to monthly air temperature and stream discharge for the 2080s future time period (2070–2099) were obtained from the University of Washington Climate Impacts Group (UWCIG) (Littell et al., 2011). For this assessment, we used a composite of modeled changes to local climate, which includes an average of the ten global climate models with the best performance at simulating observed historical summer temperature and precipitation trends across the U.S. Pacific Northwest region. We used the A1B scenario family, which lies between the middle and high end of the spectrum of projected anthropogenic greenhouse gas emissions (UWCIG, 2010) and was previously chosen for region-wide stream temperature modeling efforts (Isaak et al., 2011). Projected changes to stream discharge for this scenario were produced with the VIC hydrology model by the UWCIG using the same set of global climate models.

In order to use these data in our assessment, we first needed to adjust for the differences between the baseline period in our study (2010) and the baseline period used by the UWCIG (i.e., 1916–2006). To do this, we used the Pacific Northwest Index (Ebbesmeyer and Strickland, 1995) summarized online by the UWCIG (2012) that includes measurements of air temperature, precipitation, and snowpack at three stations in the Pacific Northwest from 1891 to 2011. Using this index, we calculated the difference between the 2010 and 1916–2006 periods to be as follows: air temperature +0.8 °C; precipitation +1.6%. These differences were then used to modify the future climate change scenario adjustments so that they appropriately pertained to the 2010 baseline period. The final set of monthly projected changes in air temperature and streamflow used in the Heat Source model are provided in Table 2.

2.6. Fish abundance

We developed a statistical model using empirical fish abundance and remotely-sensed habitat data from the Upper Grande Ronde and Catherine Creek basins to predict how simulated water temperatures for each model scenario would influence juvenile Chinook Salmon abundance. Specifically, we used a linear mixed-effects model with Chinook summer parr density (fish/m stream

length) as the dependent variable (*Density*), survey location (*Site*) as a random effect, and cumulative watershed area (km²) upstream of the site (*Area*), site gradient (%) (*Gradient*), site MWMT (°C), and average redd density (redds/100 m stream length) within 2 km upstream of the site (*Redds*) as fixed-effect explanatory variables.

Snorkel counts of juvenile salmonids were conducted by field crews from the Columbia River Inter-Tribal Fish Commission (CRITFC) and ODFW at 77 sites in the Grande Ronde basin between 2011 and 2014 using a spatially balanced random survey design (Stevens and Olsen, 2004). A portion of sites were surveyed every year while others were surveyed every 3 years, resulting in a total sample size of 129. We used a mixed-model approach in order to utilize all of the data (i.e., increase statistical power) without violating the assumption of independence (i.e., samples within a site are not statistically independent). Snorkel counts at each site were expanded using a correction factor developed from paired mark-recapture and snorkel survey data to account for fish that were not observed by snorkelers (Jonasson et al., 2015).

Watershed area and gradient estimates were obtained from NetMap (<http://www.terrainworks.com>). Water temperature data were measured at each site using HOBO water temperature loggers as part of the Columbia Habitat Monitoring Program (CHaMP; <https://www.champmonitoring.org/>). Redd densities were calculated from GPS coordinates of all Chinook Salmon redds surveyed by ODFW and the Confederated Tribes of the Umatilla Indian Reservation (CTUIR) between 2010 and 2013 (McCullough et al., 2015). Redd locations were joined to the NetMap stream layer using ArcGIS and redd densities were calculated for each 100 m stream reach. For predictions of fish abundance by model scenario, we used the average redd density for all years combined (2010–2013). Redd densities during this time period were generally representative of typical redd abundance in the basin. For example, the number of redds observed in the Upper Grande Ronde River between 2010 and 2013 averaged 184 compared with a 10-year mean (2005–2014) of 122 (range 1–360). For portions of the river network that were not surveyed, we assumed redd densities were equivalent to the average value by river class (see classification system description in Channel Width). In addition, we applied a temperature screen of 22 °C MWMT (i.e., redd densities in unsurveyed reaches with MWMT > 22 °C were set to 0) to ensure that redds were not extrapolated into areas with temperatures exceeding the upper tolerance limits for Chinook spawning (Cooney et al., 2007).

Fish density, watershed area, and redd density were log transformed (natural logarithm) prior to analysis to ensure normality and homogeneity of variance. We added a small constant k to parr density ($k = 0.005$) and redd density ($k = 0.02$) values prior to transforming to avoid problems with log transformation of zero values. The relationship between temperature and fish density conformed to a piecewise functional relationship with a threshold temperature of 18 °C based on visual inspection of the raw data and documentation in the literature of 18 °C as a common upper threshold for rearing preference of juvenile salmonids (Welsh et al., 2001; EPA, 2003). Model assumptions were tested using standard diagnostic plots as well as formal tests of normality (Shapiro-Wilks Test). Potential collinearity among predictor variables was assessed by evaluating the variance inflation factors (VIF), where a VIF of <3 was interpreted as low evidence for collinearity (Zuur et al., 2010).

We used the linear mixed-effects model to predict fish density at each NetMap reach for each of the 10 model scenarios. Model predictions were based on fixed-effects only (i.e., random effects for *Site* were set to 0) to allow for predictions at out-of-sample locations. We assumed that fish could distribute into any stream reach within the model extent, which included both current and historic use areas for Chinook Salmon, at densities determined by the

Table 2

Projected changes in monthly air temperature and streamflow resulting from climate change for the 2080s time period.

Climate projections	Month			Average
	July	August	September	
Air temperature increase (°C) from 2010	4.7	5.0	4.3	4.7
Change in streamflow (%) from 2010	−21.2	−20.7	−18.3	−20.1

predictive model. Thus, as temperatures in currently warm areas are improved due to restoration, the predicted distribution and abundance of salmon expands accordingly. Density estimates were converted to abundance for each reach by multiplying the predicted fish density by reach length. Reach-scale abundance predictions were then summed for each basin (UGRB and CCB) to calculate total population abundance.

3. Results

3.1. Water temperature

Simulated water temperatures from the calibrated 2010 model tracked very closely with measured FLIR data. Root mean square error (RMSE) for all mainstem and tributary models ranged from 0.26 °C to 1.16 °C (mean = 0.62 °C), indicating good model accuracy across the model extent. Detailed model calibration results and longitudinal temperature profiles for each modeled stream within the study area were provided in Watershed Sciences (2012).

Water temperatures under current climatic, hydrologic and riparian conditions were substantially higher in the Upper Grande Ronde River basin (UGRB) (median *MWMT* across all model nodes = 24.4 °C) compared with the Catherine Creek basin (CCB) (median = 18.7 °C; Fig. 3). Simulated restoration of riparian vegetation substantially reduced the percentage of the stream network with peak summer water temperatures above 16 °C from 93% to 73% in the UGRB, and from 70% to 48% in the CCB (Fig. 4). Similarly, the proportion of habitat with *MWMT* exceeding 18 °C was predicted to decline from 86% to 61% in the UGRB and from 53% to 21.2% in the CCB following riparian restoration. Finally, the proportion of the stream network exceeding the incipient lethal limit of 25 °C declined from 40% to 6% in the UGRB, and from 6% to 0% in the CCB as a result of riparian reforestation. Despite these substantial potential reductions in water temperature, some areas in the lower mainstem Grande Ronde River and select tributaries were predicted to reach stressful peak temperatures even after riparian restoration (Fig. 4).

The relative temperature change resulting from riparian restoration was substantially higher in the UGRB compared with the CCB (Fig. 3). For example, the predicted change in median *MWMT* from the current condition to the PNV scenario was −5.5 °C in the UGRB, compared with −2.4 °C for the CCB. Similarly, the High, High/Med, and High/Med/Low priority scenarios reduced median *MWMT* by 2.9 °C, 4.1 °C, and 4.8 °C in the UGRB, and by 1.5 °C, 2.4 °C, and 2.4 °C in the CCB respectively.

Comparison of the current mainstem Grande Ronde River temperature profile with the four riparian restoration scenarios (scenarios 2–5) indicated that expected temperature reductions from riparian reforestation would be greatest in the upper to middle portion of the river (river km 86–32), but the influence of riparian restoration on water temperature diminished rapidly as the river entered the lower Grande Ronde Valley near river km 22 (Fig. 5). Simulated riparian restoration in high priority sites alone produced substantial reductions in *MWMT* in the mainstem Grande Ronde River up to 6.9 °C (median = 1.0 °C) compared with current

conditions, but these benefits diminished rapidly downstream of Beaver Creek (river km 53). Similarly, restoration of high and medium priority areas was predicted to reduce water temperatures up to 7.2 °C (median = 3.6 °C) below current. Temperature reductions from this scenario extended further downstream to about river km 40, after which the stream was predicted to warm rapidly to a level similar to current conditions. Complete riparian restoration throughout the UGRB (i.e., PNV scenario) could reduce median *MWMT* in the main stem by approximately 4.4 °C. However, post-restoration water temperatures in the lower portion of the river (i.e., below river km 32) were predicted to be quite high (>22 °C), suggesting that restoration of these very low priority areas would likely provide limited additional thermal benefit for salmonids in the mainstem Grande Ronde River.

Model simulations representing maximum vegetation growth at 25 year increments showed that the most rapid reductions in temperature occurred during the first 25 years, with incremental reductions in temperature gradually leveling off over time, with relatively little additional benefit occurring after year 75. For example, between year 0 and year 25, predicted median water temperature in the Upper Grande Ronde basin declined from 24.4 °C to 22.2 (−2.2 °C). In comparison, the decrease in median water temperature between year 75 and year 300 (PNV) was only −0.7 °C. By year 75, approximately 85% of the maximum temperature benefit was achieved, with relatively little benefit occurring thereafter.

Simulated narrowing of the river channel—assuming that riparian and climatic conditions remained constant—reduced median *MWMT* in the UGRB from 24.4 °C under current conditions to 22.2 °C (difference = −2.2 °C). Similarly, median *MWMT* in the CCB was predicted to decline from 18.3 °C under current conditions to 17.7 °C following channel narrowing (difference = −0.6 °C) (Fig. 3). A combination of basin-wide riparian restoration and channel narrowing (model scenario 7) was predicted to reduce median water temperatures by 6.5 °C in the UGRB compared with 3.0 °C in the CCB.

Projected increases in air temperature and reduced summer streamflow associated with climate change for the 2080s time period increased median *MWMT* by 2.7 °C in the UGRB and 1.5 °C in Catherine Creek (Fig. 3). However, simulated basin-wide restoration of riparian vegetation coincident with climate change impacts was estimated to decrease median *MWMT* by 1.9 °C and 1.7 °C in the UGRB and CCB respectively. Finally, a combination of climate change, vegetation restoration, and channel narrowing produced a net decrease in median water temperature of 3.5 °C in the UGRB and 1.8 °C in the CCB.

Median water temperature in the mainstem Grande Ronde River was projected to increase by approximately 3.4 °C above current conditions as a result of climate change, with maximum summer water temperatures exceeding the lethal limit for Chinook Salmon (i.e., 25 °C) across most of the river length (Fig. 6). However, basin-wide riparian restoration was predicted to offset these impacts and even reduce water temperature by as much as 3.4 °C (median = 0.4°) below the current condition (scenario 9), particularly in the middle and upper portions of the river upstream of

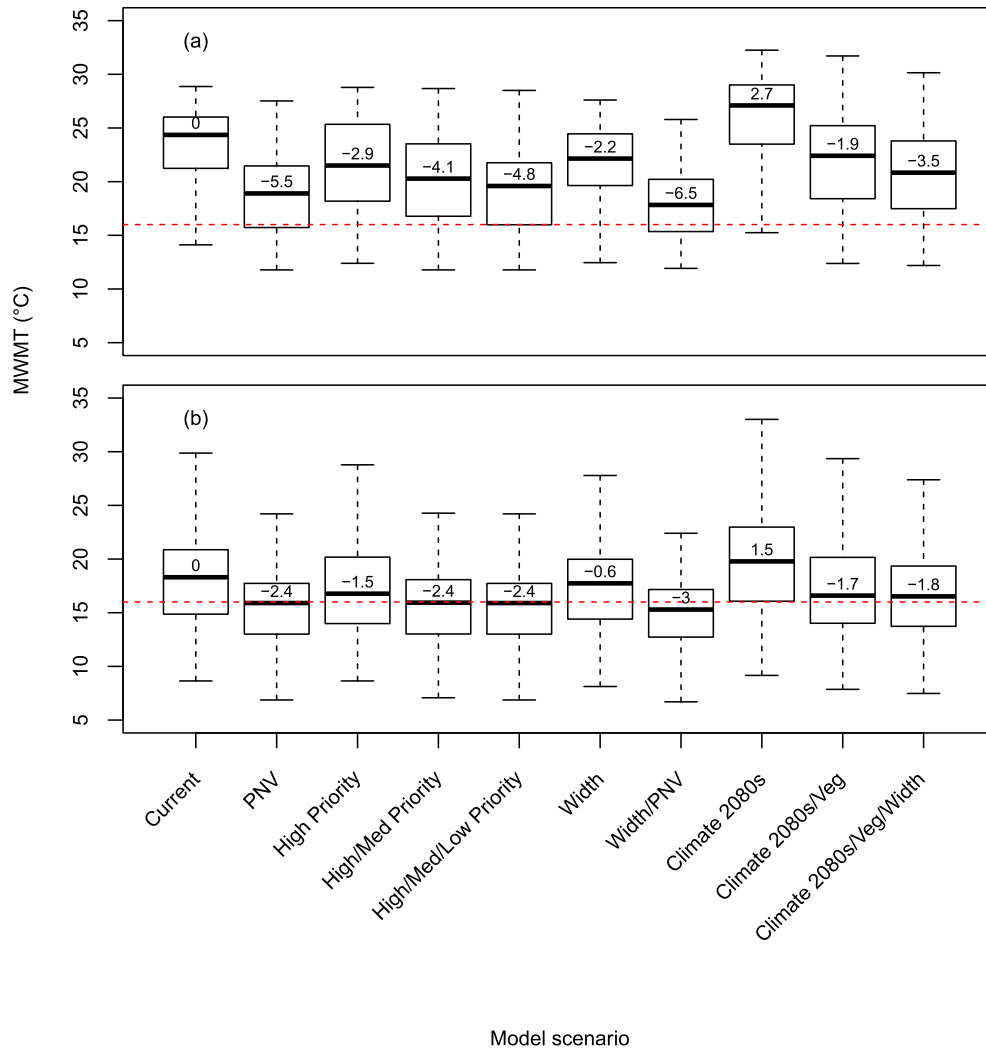


Fig. 3. Simulated maximum weekly maximum water temperatures (MWMt; °C) for each model scenario in (a) the Upper Grande Ronde River, and (b) Catherine Creek basins. Each box plot represents the distribution of MWMt values across all model nodes within the two focal watersheds including main stem and tributary locations. The numbers within each box represent the change in median temperature from current condition. The dashed red line indicates the Environmental Protection Agency's (EPA) temperature standard for core juvenile salmon rearing of 16 °C.

Beaver Creek. A combination of climate change, vegetation restoration, and channel narrowing (scenario 10) was predicted to reduce peak water temperatures in the mainstem Grande Ronde River by as much as 5.1 °C (median = 2.0 °C). These temperature reductions were limited primarily to the area upstream of Five Points Creek. Downstream of Five Points Creek, the predicted temperature profile for scenario 10 tracked closely with current conditions.

3.2. Fish abundance

The linear mixed effects model explained approximately 51% (adjusted R^2) of the variation in Chinook Salmon parr density. Temperature explained the greatest proportion of the model variation (31.5%), followed by redd density (14.7%), gradient (2.8%) and drainage area (1.8%). The temperature-density relationship was a piecewise function that was flat up to 18 °C, and then declined to near 0 at 28 °C (Fig. 7). All fixed-effect model terms were statistically significant at the $\alpha = 0.05$ level ($p < 0.01$). The random grouping factor (site) explained a negligible amount of the total variation in the data (site standard deviation < 0.001, residual

standard deviation = 1.761), but was included in the model to ensure that the model assumption of independence was satisfied. Diagnostic plots as well as formal tests of normality indicated that model assumptions were satisfied (Shapiro-Wilks Test, $p < 0.05$). Variance inflation factors (VIF) for independent variables were less than 3, indicating a low degree of collinearity among model covariates. The fit of the model to the data was only fair as evidenced by the somewhat low overall R^2 for the model and consequently, predictions to out-of-sample locations will be even less precise. However, because model errors were reasonably symmetric across the range of the data, we assumed that relative differences in predicted abundance among the different model scenarios are reasonably accurate. The final model used to predict fish abundance was given by:

$$\log(\text{Density} + 0.005) = \beta_0 + \beta_1 \log(\text{Area}) + \beta_2 \text{Gradient} + \beta_3 (\text{MWMt} - 18) + \beta_4 \log(\text{Redds} + 0.02), \quad (3)$$

where model coefficients (β_i) were given by:

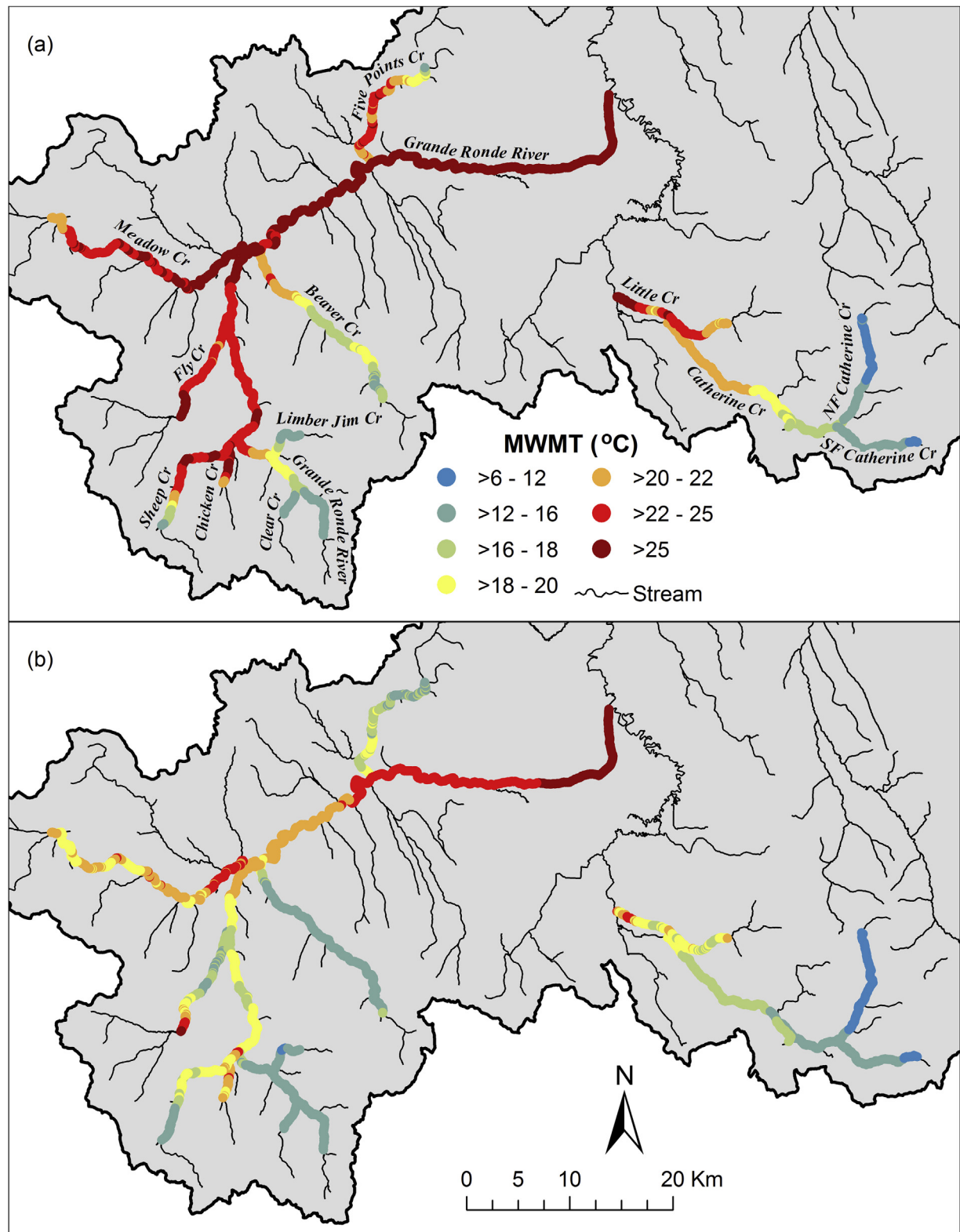


Fig. 4. Map of simulated maximum weekly maximum water temperature (MWMt; °C) for (a) current conditions and (b) potential natural vegetation (PNV) conditions in the Upper Grande Ronde River and Catherine Creek basins.

$$\beta_0 = -2.0194(\text{SE} = 1.1259)$$

$$\beta_1 = 0.7222(\text{SE} = 0.2084),$$

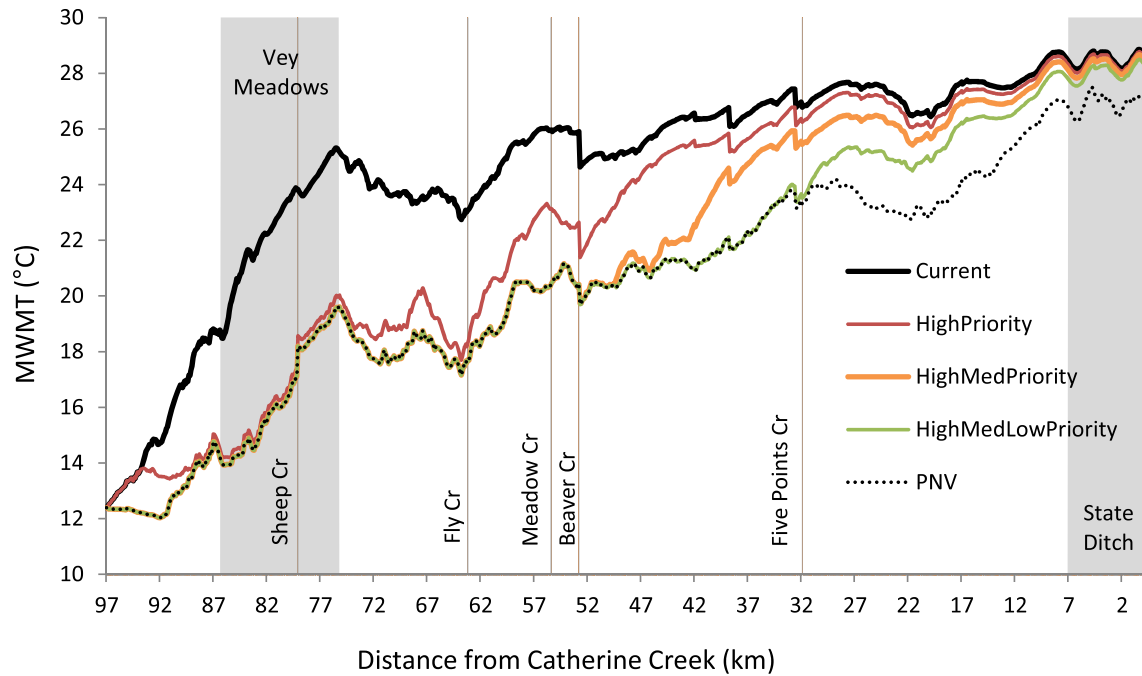


Fig. 5. Simulated maximum weekly maximum water temperature (MWMt; °C) in the mainstem Grande Ronde River from the headwaters to the Catherine Creek confluence under current conditions and four riparian restoration scenarios.

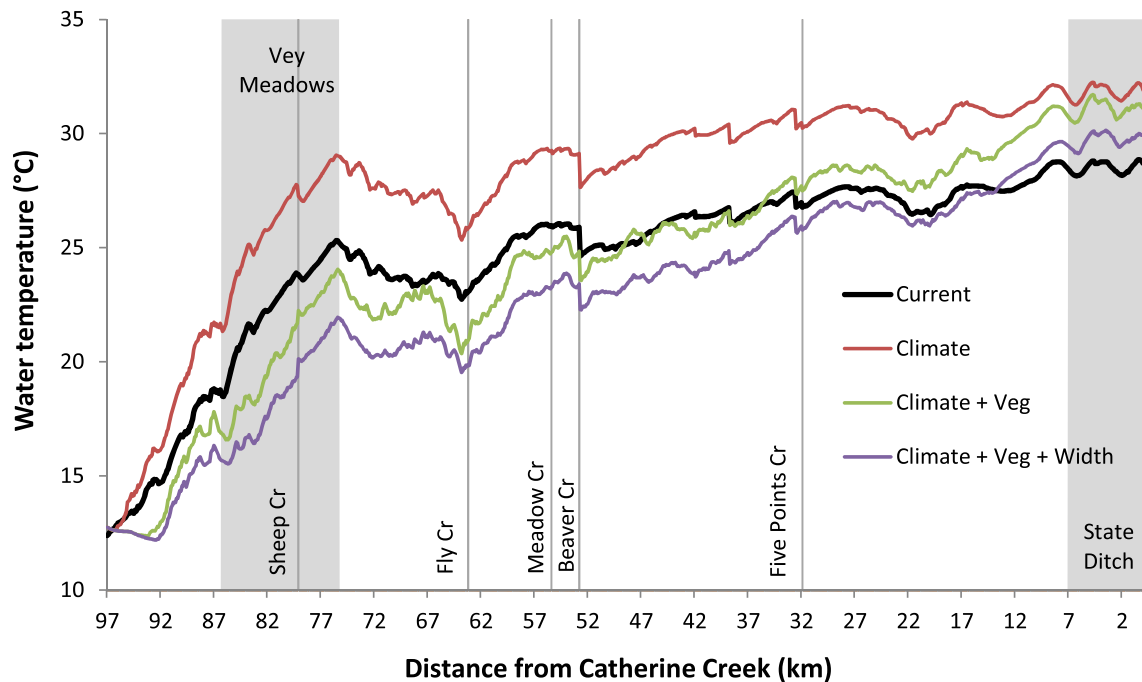


Fig. 6. Simulated maximum weekly maximum water temperature (MWMt; °C) in the mainstem Grande Ronde River from the headwaters to the Catherine Creek confluence for four model scenarios including current conditions, 2080s climate conditions, 2080's climate conditions plus riparian vegetation restoration, and 2080's climate conditions plus riparian and channel width restoration.

$$\beta_2 = -51.8642(\text{SE} = 17.8852),$$

$$\beta_3 = \begin{cases} 0, & \text{MWMt} \leq 18 \\ -0.2859 (\text{SE} = 0.0714), & \text{MWMt} > 18 \end{cases},$$

$$\beta_4 = 0.7127(\text{SE} = 0.1147).$$

Predicted abundance of Chinook Salmon summer parr under current conditions in the Upper Grande Ronde and Catherine Creek basins was approximately 46,000 and 55,000 respectively (Fig. 8). Basin-wide riparian reforestation in the absence of climate change

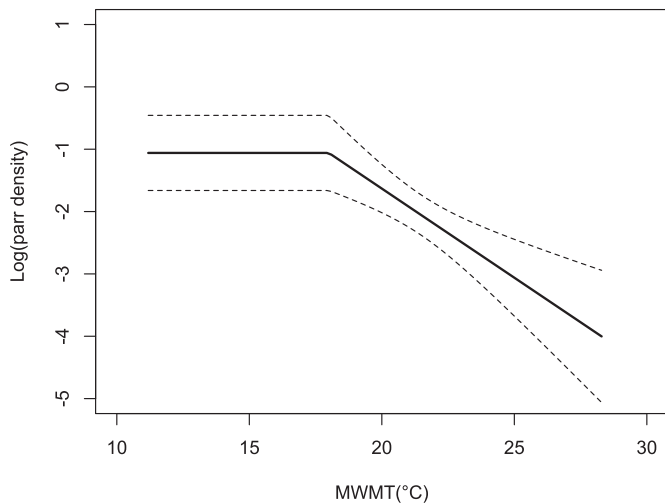


Fig. 7. Relationship between log (parr density (i.e., fish/m)) and maximum weekly maximum water temperature (MWMWT; °C) estimated using a linear mixed-effects model. The dashed line represents the 95% confidence interval.

(scenario 2) could potentially increase abundance to 222,000 in the UGRB (377% increase) and 88,000 in the CCB (61% increase). Restoration of vegetation in high priority areas only was estimated to increase abundance to about 93,000 in the UGRB (100% increase), and 71,000 in the CCB (30% increase). Expanding vegetation restoration to include both high and medium priority areas could increase abundance to 166,000 (257% increase) and 84,000 (54% increase) in the UGRB and CCB respectively. Finally, restoration of high, medium, and low priority areas was estimated to increase abundance to 208,000 (348% increase) in the UGRB and 88,000 (61% increase) in the CCB. Interestingly, restoration of medium priority sites provided the greatest incremental increase in abundance in the UGRB, whereas restoration of high priority sites provided the greatest relative increase in abundance in the CCB. Riparian restoration in low priority areas provided little additional benefit in terms of total fish abundance in the CCB (7% increase over the High/Med scenario), but produced a substantial additional increase in fish abundance in the UGRB (91% increase over the High/Med scenario).

Simulated narrowing of the river channel produced a substantial increase in fish abundance, although the effect was relatively low compared with the expected benefits from riparian restoration. For example, fish abundance in the UGRB was predicted to increase by 57% as a result of channel narrowing, compared with an increase of 377% following riparian reforestation (Fig. 8). Similarly, channel narrowing was estimated to increase fish abundance by 22% in the CCB compared with a predicted increase of 61% resulting from riparian restoration. Combining channel width reductions with riparian reforestation produced the greatest predicted increases in fish abundance, with a potential increase of 590% in the UGRB and 67% in the CCB.

Projected water temperature increases resulting from climate change for the 2080s time period were predicted to reduce fish abundance by approximately 53% in the UGRB and 36% in Catherine Creek, assuming riparian vegetation cover and height remain similar to current conditions (Fig. 8). However, if riparian vegetation was restored basin-wide concurrently with these climate changes, we estimated that fish abundance could increase by 63% in the UGRB and 20% in the CCB. Additionally, if channel width reductions were combined with riparian restoration, we estimated that fish abundance could increase by as much as 114% in the UGRB and 37% in the CCB, despite the warming effects of climate change.

4. Discussion

4.1. Effect of scenarios on water temperature

The relatively large potential reductions in water temperature achievable through riparian restoration as demonstrated by our model results are generally consistent with findings from similar modeling assessments (Theurer et al., 1985; ODEQ, 2000; Sullivan and Rounds, 2004; Watanabe et al., 2005; Butcher et al., 2010) and with numerous field experiments throughout the Pacific Northwest that have demonstrated that forest harvesting can substantially increase stream temperatures, primarily by increasing the amount of solar radiation reaching the stream surface (Beschta et al., 1987; Moore et al., 2005). However, the magnitude of change in water temperature resulting from riparian restoration can vary considerably depending on other factors including streamflow, channel morphology, climate, existing vegetation, and anthropogenic stressors (e.g., grazing, timber harvest, dams) (Penaluna et al., 2015; Lynch et al., 2016). For example, the 4.4 °C median temperature reduction from riparian reforestation we predicted for the mainstem Upper Grande Ronde River was large compared with results from a similar simulation study in the Rogue River basin in Southwest Oregon, in which peak summer stream temperatures were predicted to decline by less than 1 °C following full riparian restoration (Crown et al., 2008). The lower temperature response to riparian restoration in the Rogue River appeared to be largely influenced by the Lost Creek Reservoir, which significantly increases summer base flow and reduces peak summer water temperatures. Similarly, a temperature modeling study in the North Fork Salmon River of Northern California showed that full riparian reforestation could reduce peak summer water temperatures by only 0.26 °C (Bond et al., 2015). In this case, temperature reductions from riparian restoration were likely limited by the existence of large conifer stands throughout much of the riparian zone under current conditions, and thus, a limited potential to increase riparian cover. On the other hand, simulated riparian restoration in the upper John Day River (above rkm 275), a nearby large tributary to the Columbia River with a similar history of habitat degradation and high water temperatures, reduced peak summer water temperatures by approximately 3.8 °C below the current condition (Butcher et al., 2010), a similar magnitude of reduction as predicted for the Grande Ronde River.

The greatest potential reductions in water temperature resulting from riparian restoration occurred in the upper to middle portion of the Upper Grande Ronde River (above Five Points Creek), particularly in areas that were designated as high and medium priority. These results lend credibility to the prioritization framework presented here and provide useful guidance for restoration practitioners seeking to cool water temperatures in the basin. Restoration practitioners have previously relied heavily on expert opinion in prioritizing restoration actions; however empirical results are preferred when available (Booth et al., 2016). However, despite these substantial potential cooling benefits from riparian restoration, we found that large portions of the stream network, particularly in the lower portion of the basin, would continue to exceed EPA temperature standards (i.e., 16 °C for “core” juvenile salmon rearing, and 18 °C for salmon migration and “non-core” juvenile rearing) even after restoration of riparian vegetation to its natural potential (Fig. 3). One reason for this may be that our model did not account for various other ecological factors that can contribute to cooling (e.g., hyporheic exchange, flow restoration). However, it is reasonable to assume that because of its arid climate, water temperatures were historically stressful to salmon during July and August in some portions of the Upper Grande Ronde River, particularly in the lower main stem where the river is wide and

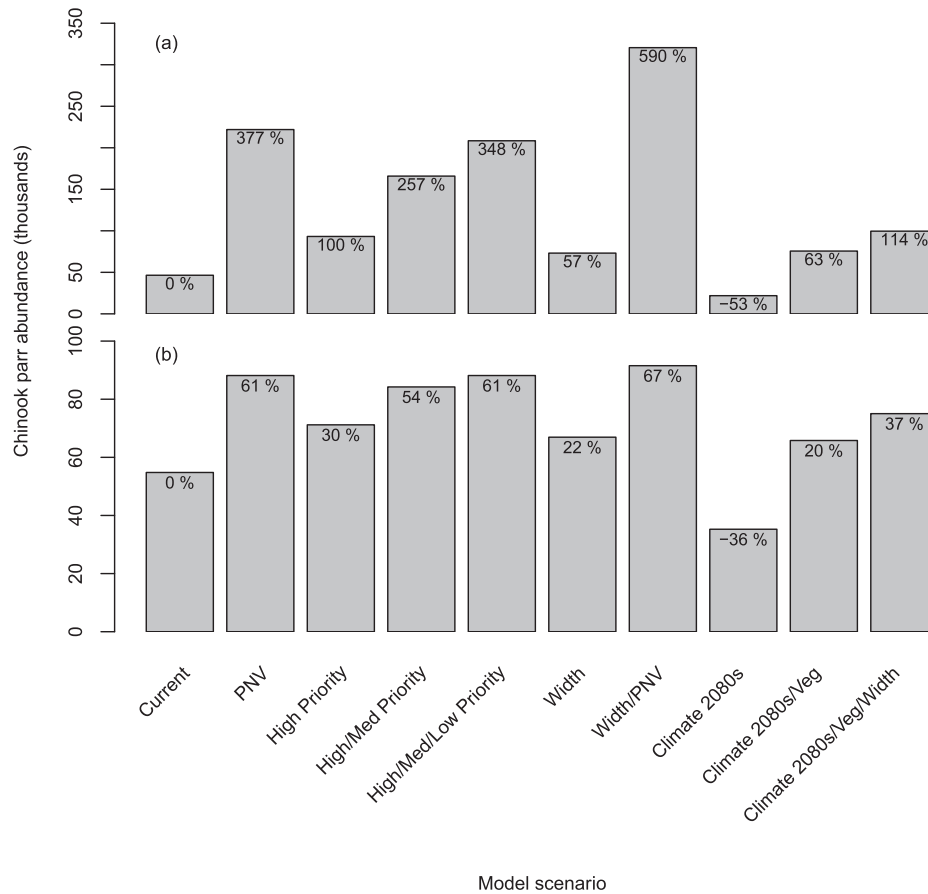


Fig. 8. Predicted abundance of Chinook Salmon summer parr for each model scenario in (a) the Upper Grande Ronde River, and (b) Catherine Creek basins. Numbers at the top of each bar indicate the percentage change in abundance from the current condition.

streamside vegetation is dominated by meadow and shrubland habitats with limited potential tree cover. This finding is congruent with temperature modeling results from the John Day River, which showed that water temperatures following restoration of riparian vegetation, channel morphology and streamflow were still well above biologically based temperature criteria for salmon across a large portion of the stream network (Butcher et al., 2010).

Model simulations demonstrated that channel narrowing (i.e., decreased width-to-depth ratio) can have an important cooling influence on water temperature in the Upper Grande Ronde and Catherine Creek basins, although thermal benefits associated with channel width were relatively small compared with the reductions in temperature achieved through riparian restoration (Fig. 3). Similar thermal responses to channel narrowing have been documented in other modeling analyses investigating land use impacts on water temperature (LeBlanc et al., 1997; ODEQ, 2000; Butcher et al., 2010). In the Upper Grande Ronde TMDL analysis, ODEQ also found that simulated channel width reductions in the main-stem Grande Ronde River provided a relatively small cooling benefit compared with vegetation restoration (ODEQ, 2000). In contrast, Butcher et al. (2010) found that reductions in channel width of 30% provided approximately equivalent reductions in water temperature compared with vegetation restoration in the upper 150 km of the John Day River, and substantially greater temperature reductions (i.e., approximately 2 °C below current) compared with vegetation restoration in the lower river where the channel is very wide and riparian shade potential is limited.

The simulated warming effect of climate change on water

temperature in the Upper Grande Ronde basin (median increase of 2.7 °C by 2080) and Catherine Creek (+1.5 °C) was generally consistent with other recent assessments of future climate change impacts on stream temperatures in the Pacific Northwest (Wu et al., 2012; Isaak et al., 2015) and in North America in general (Lynch et al., 2016), although the magnitude of change in water temperature varies depending on model assumptions, elevation, and geographic location. In a modeling study of climate change impacts on hydrology and temperature of Pacific Northwest rivers, Wu et al. (2012) predicted that climate change for the 2080s future scenario would increase average summer stream temperatures by 2.1 °C. Similarly, Isaak et al. (2015) predicted that mean August water temperatures in Pacific Northwest streams would increase by 2.8 °C on average by the 2080s. These increases correspond with—and even slightly exceed—expected climate-related changes in water temperature for rivers globally, with predicted increases in 95th percentile temperatures ranging from 1.0 to 2.2 °C for the years 2071–2100 (van Vliet et al., 2013).

Our model results demonstrated that restoration actions including riparian reforestation and channel narrowing could offset future climate-related increases in water temperature in the Grande Ronde basin, with net reductions in peak summer water temperature of 3.5 °C and 1.8 °C in the Upper Grande Ronde and Catherine Creek basins respectively. Similarly, Bond et al. (2015) found that climate-related warming in the North Fork Salmon River could be mitigated by full riparian reforestation for climate scenarios with a modest air temperature increase (+2 °C), but for scenarios with severe flow reduction and warming (71% flow

reduction, 6 °C air temperature increase), riparian reforestation could only reduce the predicted stream warming by half. Battin et al. (2007) predicted that full restoration of the Snohomish River basin in Western Washington, which included riparian reforestation, reduction of impervious surfaces, and improvements to in-stream complexity and floodplain connectivity, could offset climate warming and reduce average late summer (August 15–September 15) stream temperatures by 0.13–0.16 °C by the year 2050. However, under a more moderate restoration scenario, they predicted that water temperatures in year 2050 would increase by 0.03–0.04 °C above current temperatures.

Other factors beyond riparian vegetation and channel width can have an important influence on water temperature but were not included in our modeling analysis. For example, restoration actions such as channel reconstruction, levee removal, and construction of large woody debris jams have the potential to cool or buffer water temperatures by aggrading the stream bed, thereby increasing the degree of mixing between surface and subsurface water in the floodplain alluvium (i.e., hyporheic exchange) (Arrigoni et al., 2008; Beechie et al., 2013). In addition, flow restoration, which includes increasing in-stream flow by purchasing water rights, altering dam operations, or improving irrigation efficiency can substantially reduce water temperatures by increasing a stream's ability to withstand a given heat load without substantive change in water temperature (i.e., assimilative capacity) (Poole and Berman, 2001). Inclusion of such factors into future model scenarios could improve our understanding of the relative benefits of alternative restoration strategies.

Our model unrealistically assumes that—for the full riparian restoration scenario—the entire stream network could be restored to its natural potential. Currently, access to some private lands in the basin is restricted because of landowner denial and future access to this land is uncertain. In addition, some level of grazing impacts will likely persist due to its economic importance to landowners. Grazing by native ungulates like deer and elk may continue to be unnaturally high without a threshold abundance of natural predators (Ripple and Beschta, 2004). Survival of riparian plantings may also be impacted by factors such as desiccation, soil conditions, competition, animal damage, or channel incision, which could impact the likelihood that the riparian vegetation reaches its natural potential (Anderson and Graziano, 2002; Wall, 2011). Our model also unrealistically assumes that riparian restoration across the stream network occurs immediately, starting on day one of the model simulation period. In reality, tree planting typically occurs in a staggered fashion with the amount of riparian habitat restored being dictated by annual differences in funding and landowner permissions. These model uncertainties suggest that our full riparian restoration scenarios likely represent a best-case scenario in terms of expected thermal benefits from riparian restoration. As such, riparian restoration scenarios targeting only high and medium priority areas may be more realistic in that they assume only the highest priority portions of the basin would be restored to their full potential.

4.2. Effect of scenarios on fish abundance

The negative relationship we observed between juvenile Chinook Salmon density and water temperature is largely consistent with findings from similar field studies of salmonid populations throughout the Pacific Northwest (Ebersole et al., 2001; Welsh et al., 2001; Madriñán, 2008; Thompson et al., 2012). Potential increases in Chinook Salmon summer parr abundance up to 590% in the Upper Grande Ronde and 67% in Catherine Creek resulting from restoration of riparian vegetation and channel width are encouraging and highlight the importance of riparian shade—and to a

lesser extent channel width—for salmon productivity in these basins. Higher relative abundance increases in the Upper Grande Ronde compared with Catherine Creek are reflective of higher overall water temperatures and lower riparian shade in the Upper Grande Ronde, and consequently, a greater potential for water temperature improvements.

Climate change under current policy is predicted to accelerate extinction risk for endemic flora and fauna across the globe, including but not limited to fish (Urban, 2015). Of the studies that have focused on freshwater fish populations, the vast majority have concluded that climate change has significantly altered the distribution, phenology (e.g., migration timing), abundance, growth, recruitment, genetics and assemblage structure of many fish species, with the greatest impacts occurring in cold-water species like salmon and trout (Lynch et al., 2016). These findings are generally consistent with our results, which demonstrated a potential decline in juvenile salmon abundance of 53% in the Upper Grande Ronde and 36% in Catherine Creek due to climate-related water temperature increases.

Integrating climate change impacts with both riparian restoration and channel narrowing indicated that basin-wide restoration actions, if fully implemented immediately at a system level, could more than offset impacts from climate change, with net predicted increases in juvenile salmon abundance of 114% in the Upper Grande Ronde and 37% in Catherine Creek. In a similar modeling analysis of Chinook Salmon population response to habitat restoration and climate change in the Snohomish River basin in Western Washington, Battin et al. (2007) found that full habitat restoration could help mitigate the negative impacts of climate change and even allow salmon populations to increase in the face of climate change. Specifically, they found that full basin-wide restoration could limit declines in salmon populations to 5%, or could increase salmon populations by 19%, depending on the severity of the climate scenario selected.

We acknowledge that our model predicting juvenile fish abundance was relatively simplistic in that it does not include all important factors to which fish are known to respond (Jackson et al., 2001). We do however maintain that our approach represents a plausible relative fish response due to the well-supported effect of temperature on fish physiology, behavior, and distribution (Regier et al., 1990; Armour, 1991; McCullough, 1999). Our model did not incorporate density dependence explicitly in the form of a standard stock-recruitment function (i.e., Beverton-Holt or Ricker); however density dependence was implicitly accounted for by using redd abundance as an explanatory variable, and fish abundance values corresponded to average values observed during the study period. Although we focused on the juvenile rearing life stage, other salmon life stages (e.g., pre-spawn holding or egg incubation) can also be impacted by water temperature. We assumed that egg incubation, which is the most sensitive to high water temperature (optimal range 6–10 °C; EPA, 2003), is not significantly impacted by water temperature in the Grande Ronde basin because eggs incubate during fall and winter when temperatures are low. However, gamete viability in holding adults as well as survival of pre-spawn adults themselves can also be impaired by high water temperature (EPA, 2003). Interestingly, pre-spawn mortality data estimated from carcass recoveries in the Grande Ronde basin (Joseph Feldhaus, ODFW, pers. comm.), did not show a meaningful relationship with temperature and was therefore not included in our model. Utilizing the temperature results from this analysis within the context of a life-cycle model that properly accounts for density-dependent factors and temperature impacts on multiple life stages is an objective of ongoing research.

5. Conclusions

Results from our water temperature model indicated that intensive basin-wide restoration actions targeting both riparian vegetation and channel width could offset projected climate change impacts to salmon populations in the Upper Grande Ronde and Catherine Creek basins. A combination of riparian reforestation and channel narrowing was predicted to greatly reduce peak summer water temperatures in the study watersheds relative to current conditions, translating to large predicted increases in salmon abundance. However, significant portions of these watersheds—particularly in lower elevation reaches—continued to exceed stressful temperature limits for salmon growth and survival even after full riparian restoration, suggesting that alternative restoration strategies (e.g., increasing floodplain connectivity, streamflow restoration, and enhancement and protection of cold-water refuges) should also be implemented in order to maximize thermal benefits to threatened salmon populations.

These findings likely represent a best case scenario in terms of expected temperature reductions from riparian restoration because they are based on optimistic assumptions regarding the timing and spatial extent of future restoration actions. Given that a more realistic restoration scenario (e.g., high priority areas only) would result in more modest temperature improvements, combined with the rapid projected rise in water temperatures due to climate change, we emphasize the urgent need for a targeted and aggressive restoration strategy which includes riparian restoration as a key component. The temperature modeling and prioritization framework presented here should prove useful for restoration planners and practitioners seeking to improve habitat conditions for fish populations and other aquatic biota in temperature-impaired watersheds.

Acknowledgements

Special thanks to Ted Sedell, Chris Horn, Shelley Tattam and others with the Oregon Department of Fish and Wildlife and Les Naylor and Allen Childs with the Confederated Tribes of the Umatilla Indian Reservation for contributing fish and habitat survey data. We are grateful to Aaron Wells and Rich Blaha with ABR, Inc. and Elizabeth Crowe for their work in developing a riparian vegetation map and to Brian Kasper and others at Quantum Spatial for their work in calibration of the Heat Source model and assistance with running the model. Thanks also to Bob Lessard of CRITFC for his assistance with quantitative analysis and coding in R, and to Denise Kelsey of CRITFC for helping with map development. Funding for this project was provided by the Bonneville Power Administration as part of the Columbia Basin Fish Accords Agreement (Project # 2009-004-00).

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