



# THE IMPORTANCE OF HABITAT COMPLEXITY TO STREAM FISHES

A Multi-Scale Assessment in Goose Creek, Upper Snake River Basin (ID, NV, UT)

## Abstract

The Goose Creek subbasin in the Upper Snake River Basin has a diverse native fish assemblage that reflects the presence of rare non-game species and peripheral populations of Yellowstone cutthroat trout. This assessment examines linkages between native fishes and their habitat in the Goose Creek subbasin with several key findings: 1) fish diversity is linked to habitat diversity, and habitat diversity is linked to stream condition; 2) various elements of habitat complexity are important to several native fishes at different spatial scales; and 3) land management focused on riparian and stream health is important in maintaining the habitat complexity important to fish community diversity and sensitive fish species that are rare in the Upper Snake River Basin.



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## EXECUTIVE SUMMARY

The Upper Snake River Basin has a diverse native fish fauna defined by unique assemblages above and below Shoshone Falls. As highlighted in Chapter 1, a spatially explicit multispecies analysis showed the Goose Creek subbasin to rank very high for native fish conservation because of the occurrence of Yellowstone cutthroat trout, northern leatherside chub, and bluehead sucker. Because of its unique native fish assemblage, we evaluated linkages between Goose Creek fishes and physical habitat at several spatial scales, focusing on habitat complexity, stream condition, and land management. The key findings are:

Chapter 2 shows that fish diversity in Goose Creek is driven by physical habitat diversity, and habitat diversity is linked to stream condition. It is the first study to show that all four elements of habitat diversity – diversity in streamflows, bed morphology, instream cover, and substrates – drive fish diversity. Lower habitat diversity in degraded streams suggests that land management plays a key role in maintaining habitat complexity and conservation of the diverse fish assemblages in the subbasin.

Chapter 3 shows that the distribution of northern leatherside chub in Goose Creek is strongly influenced by streamflow complexity, and that streamflow complexity is typically higher when active or abandoned beaver dams are present. This suggests beaver reintroductions or beaver dam analogs should be evaluated as a stream restoration practice for northern leatherside chub conservation in an adaptive management framework.

Chapter 4 shows that Yellowstone cutthroat trout abundance is limited by both brook trout and habitat complexity at the subspecies' range periphery, and that brook trout negatively influences and instream cover diversity positively influences body condition of age-0 cutthroat trout. The negative interaction between brook trout and cutthroat trout early in life has been shown previously and emphasizes removal of brook trout as a cutthroat trout conservation action. However, this study also suggests that conserving or restoring habitat complexity should be more widely considered and evaluated as a complementary action that could promote coexistence of the two species.

Chapter 5 shows that riparian exclosures, a grazing management tool, in the Goose Creek Group Allotment have a local benefit to riparian vegetation and stream channel morphology. However, they were not shown benefit aquatic habitat and organisms (benthic macroinvertebrates and fish) typically responding to larger watershed-scale conditions; one exception was that age-0 trout were more abundant inside (than outside) exclosures. This study suggests that riparian exclosures should be one of several complementary grazing management actions if the management goal is to benefit aquatic organisms. Small exclosures need to be further evaluated for their role in facilitating persistence of sensitive fishes, such as northern leatherside chub in the Goose Creek mainstem, through source-sink dynamics.

Chapter 6 shows that northern leatherside chub select microhabitats with overhead cover from mature woody and herbaceous riparian vegetation and streamflow complexity influenced by riffle-pool morphology, overhanging bank vegetation, and beaver dams. Maintaining riparian health standards should create northern leatherside chub habitats used at small spatial scales to presumably benefit population dynamics that facilitate persistence.

An overriding theme is that Goose Creek fishes are linked to various elements of habitat complexity at different spatial scales, and habitat complexity is linked to stream condition. Thus, land management focused on riparian health standards and promoting instream habitat complexity will play a key role in conserving the diverse assemblage of native fishes in the Goose Creek subbasin.

## RECOMMENDED CITATION

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## WEB RESOURCES

[www.tu.org/goose-creek-assessment](http://www.tu.org/goose-creek-assessment)

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*Senesced willow branches creating current seam (high flow complexity) where Northern Leatherside Chub were collected in Trapper Creek, 2013. Credit: D. Dauwalter.*

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# CHAPTER 1: PUTTING GOOSE CREEK INTO PERSPECTIVE: A MULTISPECIES ASSESSMENT OF NATIVE FISHES IN THE UPPER SNAKE RIVER BASIN

## Conservation Assessment and Planning

Conservation actions and natural resource management involve myriad decisions across spatial and temporal scales with various goals, objectives, and constraints (Ferrier and Wintle 2009). These actions and decisions include: zoning for various uses such as forest production, livestock production, mineral extraction, recreation; allocation of resources for land owner incentive programs, addressing threats to ecosystems, or restoring habitat; and, acquiring or designating lands for protection. Formal planning processes are often critical to effective implementation of actions intended to achieve conservation management outcomes, and effective planning is collaborative and requires both stakeholders and managers using a transparent process (Carr et al. 1998; Barmuta et al. 2011). Science-based, data driven assessments developed around specific goals are often used to inform these collaborative planning processes (Sarkar et al. 2006).

Conservation and management of freshwater aquatic ecosystems is challenging because of the longitudinal, lateral, and vertical connectivity of riverine systems (Barmuta et al. 2011). Recently, advances have been made on integrating aquatic ecosystem properties into spatial conservation planning and assessment tools that now account for aquatic ecosystem connectivity, species representation, watershed condition, and other factors (Moilanen et al. 2008). This better ensures assessment outputs are both adequate and comprehensive given the planning goals and objectives (Linke et al. 2011) (Figure 1).

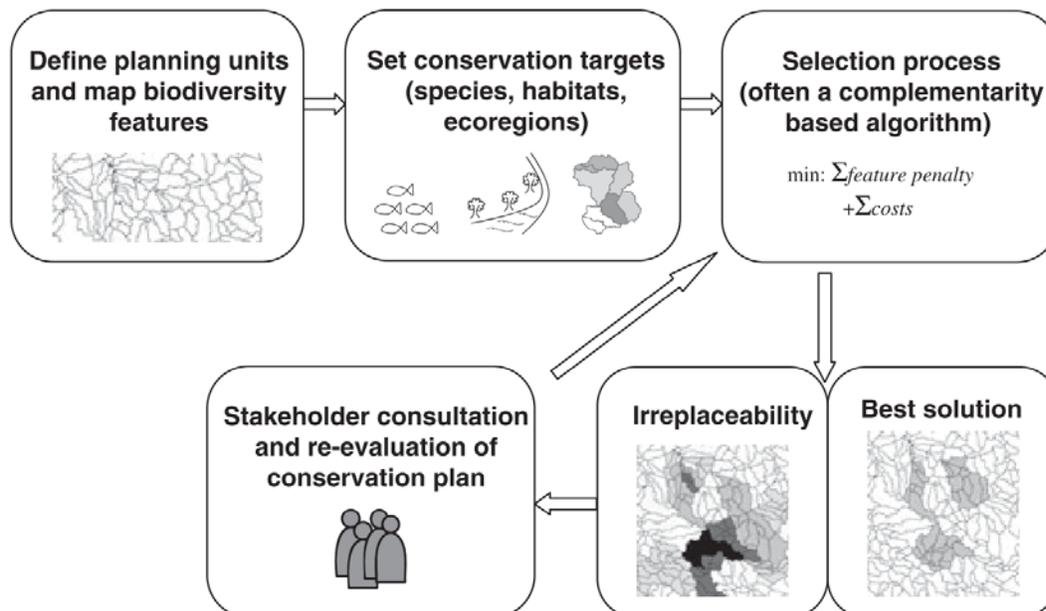


Figure 1. Process of aquatic conservation assessment and planning. From Linke et al. (2011).

## Upper Snake River Basin Native Fish Assessment

Williams et al. (2014) conducted an assessment of the Upper Snake River Basin (upstream of Hells Canyon Dam) with a goal of identifying entire watersheds that had high native fish conservation values where management could emphasize natural watershed function while meeting the life history needs of entire native fish communities (e.g., Native Fish Conservation Areas; Williams et al. 2011). The outcome of the Upper Snake River Basin aquatic assessment was a ranking of all subwatersheds (Hydrologic Unit Code 12, or 6-digit HUC) based on the aforementioned goal. The assessment was data driven and incorporated data on conservation populations of native Redband Trout *Oncorhynchus mykiss gairdneri*, Bull Trout *Salvelinus confluentus*, and Yellowstone Cutthroat Trout *O. clarkii bouvieri*, known and predicted (from species distribution models) distributions of native non-game fishes (excluding spring and large river endemics), subwatershed connectivity, and protected areas. Because the spatial unit of the assessment was subwatersheds, the analysis necessarily accounted for adjacent subwatersheds and their conservation value. The analysis also accounted for whether subwatersheds contained any formal protected lands (National Parks, Wilderness, etc.) as designated in the Protected Areas Database of the U.S. (USGS 2011) because those watersheds are more likely to function naturally and be amenable to management focused on native fishes, especially those that have native trouts that have important native fish conservation and sportfish values (Haak and Williams 2013; Haak and Williams 2015). While subwatershed rankings were based, in part, on proximity to protected areas, a re-analysis removing the influence of protected areas showed only a slight re-balancing of subwatershed rankings (Figure 2A versus B).

The subwatershed rankings can be used in aquatic conservation and land management planning applications that are based on interconnected watersheds containing both native trout and native non-game fishes. Example applications are identifying important areas for watershed restoration efforts, or implementing additional land management or protection efforts to facilitate natural watershed function to meet fish life history needs. Headwaters of the Snake River, Goose Creek, and the South Fork Boise River are examples where subwatersheds ranked high for native fish conservation value.

### The Goose Creek Subbasin

As mentioned, the Goose Creek subbasin has many high ranking subwatershed. This is because of the presence of Yellowstone Cutthroat Trout populations in several tributary streams, and the occurrence of Northern Leatherside Chub *Lepidomeda copei* and Bluehead Sucker *Catostomus discobolus* in several tributaries. Leatherside Chub and Bluehead Sucker only occur in a few other subbasins in Idaho, as well as in the Snake River headwaters in Wyoming (Zafft et al. 2009; Blakney 2012). The high rankings are also solely due to native fishes because the subbasin has no protected areas (Figure 2).

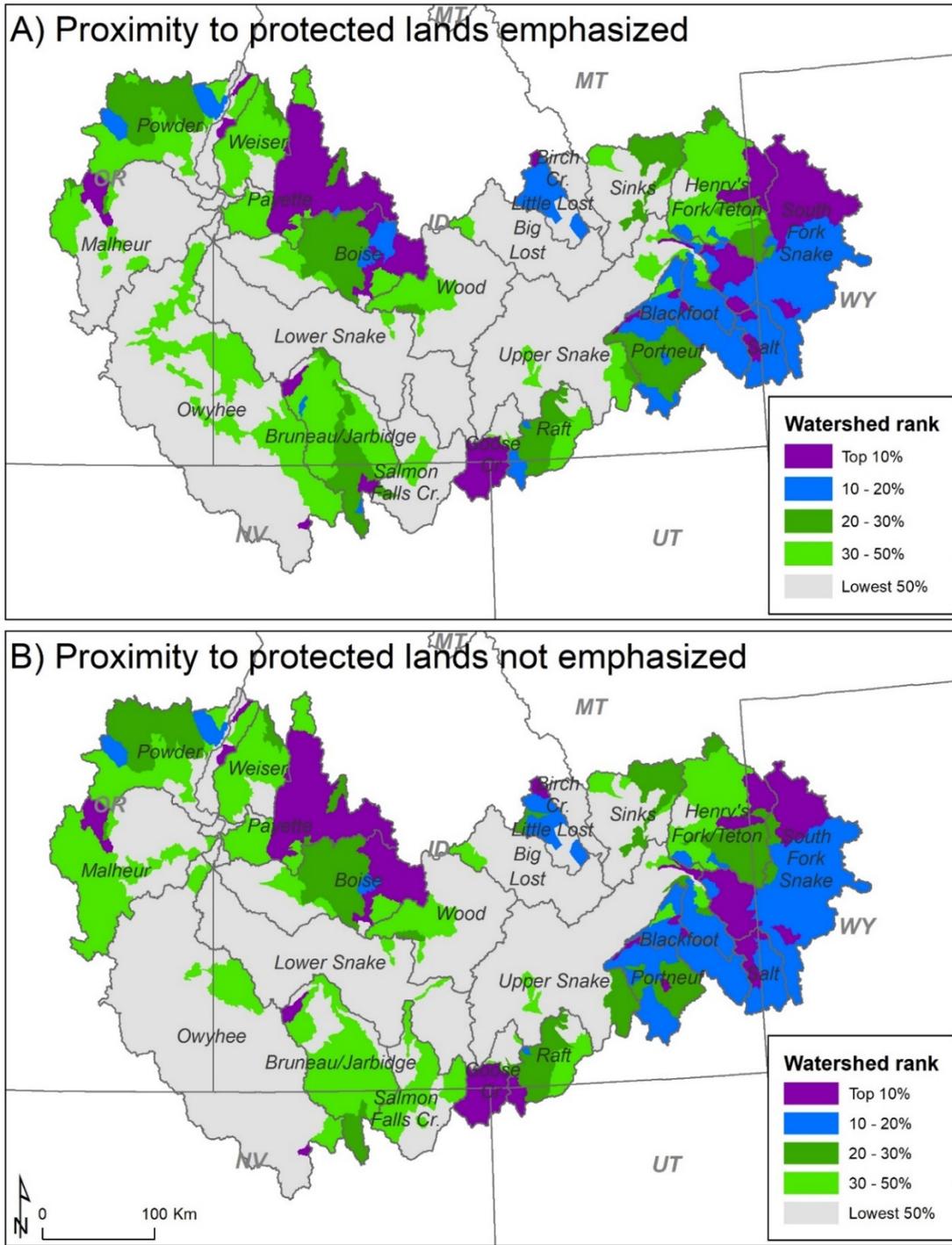


Figure 2. Rankings of Upper Snake River Basin subwatersheds based on native trout and non-game fish known or predicted distributions and watershed connectivity. Scenario 1 emphasized proximity to protected lands (national parks, designated wilderness); Scenario 2 did not emphasize protected lands.

Despite high conservation value for native fishes, the Goose Creek subbasin is not without threats to aquatic ecosystems. The subbasin is included in Idaho and Nevada 303(d) lists for impaired waters (IDEQ 2010a; NDEP 2014), and it contains a Total Maximum Daily Load (TMDL). Major impairments to subbasin streams are: bacteria (fecal coliform and *Escherichia coli*), dissolved oxygen, total phosphorous, sediment and suspended solids, and temperature. Road/trail decommissioning, road management, livestock exclusion, fence repair, riparian management, and streambank protections are listed as TMDL implementation actions (IDEQ 2010a).

Given the high native fish diversity in the Goose Creek Subbasin and that it has formally listed impairments, there is a need to understand linkages between sensitive native fishes and habitat conditions in the subbasin. The remainder of this assessment document then evaluates how fish community diversity and the abundance and distribution of sensitive species are influenced by physical habitat conditions, with a focus on physical habitat complexity, at multiple spatial scales.

### **Supplemental Resources:**

Upper Snake River Basin Assessment Webpage: [www.tu.org/USRB-multisp-assmt](http://www.tu.org/USRB-multisp-assmt)

Upper Snake River Basin Assessment Webmap:

<http://trout.maps.arcgis.com/apps/webappviewer/index.html?id=1d57cf21bd3140a2989d9d4e47dcc360>

Goose Creek/Owyhee Stream Temperature Monitoring Webpage: [www.tu.org/owy-stream-temp](http://www.tu.org/owy-stream-temp)



*Overlooking Little Birch Creek, Cassia County, Idaho. Credit: R. Bjork.*

## CHAPTER 2: INFLUENCE OF STREAM CONDITION ON HABITAT DIVERSITY AND FISH ASSEMBLAGES IN AN IMPAIRED UPPER SNAKE RIVER BASIN WATERSHED

### Abstract

Habitat diversity reflects the range of habitats available used by species with different niche requirements and, therefore, influences species diversity. Land use influences stream condition, with streams in poor condition often wide, shallow, sediment-laden channels with low instream habitat diversity. Our goal was to evaluate the effect of instream habitat diversity on fish species diversity, the effect of stream habitat condition on habitat diversity, and the effect of habitat diversity, stream condition, and other natural stream features on fish assemblage structure (proportional abundance) in an impaired upper Snake River Basin watershed containing a locally diverse but regionally depauperate species pool. We sampled fishes and instream and riparian habitat at 41 sites, focusing on measures of instream habitat diversity and the following stream condition indicators: livestock trails on streambanks, streambank stability, channel width-to-depth ratio, percent fine substrates, and woody riparian vegetation. Multiple regression revealed that fish species diversity was positively associated with all four components of habitat diversity, that is, diversity in substrate, cover, water depth, and water velocity ( $P < 0.09$ ; adjusted- $R^2 = 0.642$ ). All four components of habitat diversity increased with stream size, and each component was negatively but weakly associated with at least one stream condition indicator ( $P < 0.10$ ; adjusted- $R^2 = 0.23$  to  $0.61$ ). Fish assemblage structure was influenced primarily by natural stream features (gradient, temperature) and secondarily by stream condition indicators and streamflow diversity. Our results connect fish species diversity increases in larger streams with concomitant increases in four dimensions of instream habitat diversity, and show how stream condition reflecting land uses, such riparian over-grazing, can negatively impact habitat diversity within that stream-size continuum, thus emphasizing the role land management plays in maintaining fish diversity.



*Aquatic habitat survey on Goose Creek (at Coal Banks). Photo: K. Fesenmyer.*

## Introduction

Structural diversity of habitat - often called habitat heterogeneity or complexity - has been associated with the diversity of many taxonomic groups. This habitat diversity-species diversity relationship is due to increased physical space, refuge, resource availability, and, consequently, niche availability to organisms with varying niche requirements (MacArthur and MacArthur 1961; Tews et al. 2004; St. Pierre and Kovalenko 2014). In streams, fish diversity increases with habitat volume (Schlosser 1982; Angermeier and Schlosser 1989). Sheldon (1968) hypothesized the positive association between fish diversity and water depth in a New York stream was due to increased habitat volume that facilitated vertical niche partitioning by different species. Gorman and Karr (1978) found that the diversity in stream habitat positively influenced fish species diversity across both temperate and tropical streams. Their data showed that the relative contribution of diversity in stream substrates, water velocities, and depths to fish species diversity varied between their two study streams, and they argued that each component of habitat diversity varied in importance to different fish guilds. Subsequent studies have continued to document fish diversity associations with habitat diversity in streams (Schlosser 1982; Smith and Mather 2013).

Streams degraded from anthropogenic activities can often have low instream habitat diversity in addition to other symptoms (Gorman and Karr 1978; Lepori et al. 2005; Laub et al. 2012). Watershed land use influences instream habitat through connections within the aquatic system at multiple spatial scales (Frissell et al. 1986), and many studies have linked watershed and riparian land use to changes in stream habitat (Roth et al. 1996; Nerbonne and Vondracek 2001). Urban watersheds have impervious surfaces and development that encroaches into stream corridors that result in altered flow regimes and riparian vegetation. Urban watersheds, therefore, often have streams with incised channels, eroding stream banks, uniform bed morphologies, and small but sparse wood (Booth et al. 2015). Likewise, agricultural streams often have less riparian vegetation, more streambank erosion, higher levels of fines sediments, and increased nutrients (Vondracek et al. 2005). In the western US, watersheds with higher levels of use (grazing, logging, mining, and roads) have been shown to have more unstable stream banks with fewer undercuts, and shallower pools with more fine sediments embedded in pool tails (Kershner et al. 2004).

Domestic livestock grazing, primarily from cattle, is pervasive in the western US and has had a large impact on native ecosystem health, including stream health (Fleischner 1994; Poff et al. 2011). Grazing impacts to streams result when cattle congregate in riparian areas for easy access to water, lush vegetation, and flatter terrain (Kauffman and Krueger 1984). Intense riparian grazing often alters the community composition of riparian vegetation with a reduction in grasses, forbs, sedges, and woody vegetation (Fleischner 1994; Beschta et al. 2013; Batchelor et al. 2015). Changes in riparian vegetation are negatively correlated with terrestrial invertebrate inputs that are an important prey source for fishes (Saunders and Fausch 2009). Loss of riparian vegetation and stream shading leads to higher stream temperatures that can exceed fish thermal tolerances (Li et al. 1994). Reduction in woody vegetation (e.g., willows, alder, aspen) and streambank trampling cause streambank instability and erosion that, in turn, lead to wider, shallower, and warmer streams with higher concentrations of fine sediments, nutrients, and bacteria (Kauffman and Krueger 1984; Stuber 1985; Armour et al. 1991; Agouridis et al. 2005).

Many studies have shown negative associations between anthropogenic land use and fish occurrence and abundance (Wenger et al. 2008; Dauwalter et al. 2011) and fish assemblage richness, diversity, and integrity (Wang et al. 1997; Dauwalter and Jackson 2004; Perkin et al. 2016). The loss of biological diversity due to stream alteration and degradation from land uses is often attributed to the concomitant reduction in habitat diversity (Stuber 1985; Armour et al. 1991). Despite this common attribution, the link between indicators of stream condition and habitat diversity is rarely quantified (but see Gorman and Karr 1978; Schlosser 1982; Laub et al. 2012) despite habitat diversity (or complexity) often being the goal of stream restoration (Palmer et al. 2010; Laub et al. 2012).

We evaluated how different elements of instream habitat diversity influenced fish diversity and then how stream habitat condition was related to different elements of habitat diversity in the Goose Creek watershed, a tributary to the Snake River near the Idaho-Nevada-Utah border that has been impacted by multiple land uses, with livestock grazing being the predominant use. Specifically, our objectives were to: 1) determine how four dimensions of habitat diversity (cover, substrate, velocity, and depth) influence fish species diversity; 2) determine how woody riparian vegetation, stream channel dimensions, streambank conditions, and fine sediments – all indicators of stream condition reflecting land use and used to monitor grazing impacts – are associated with the four dimensions of habitat diversity; and 3) determine how instream habitat diversity and other instream and riparian habitat features, including stream condition indicators, influence fish assemblage structure (proportional abundance of species). Our study provides insight into how stream condition influences instream habitat diversity, and shows how all four elements of instream habitat diversity are associated with fish species diversity, in small streams in the Columbia River drainage that are characterized by low fish species richness when compared to other regions of the United States (e.g., southeastern U.S.; Abell et al. 2008).

## Methods

*Study area* - Goose Creek originates in southern Idaho on the Sawtooth National Forest at around 2,200-m elevation, and then flows into Nevada, Utah, and then back into Idaho and into Lower Goose Creek (Oakley) Reservoir (1,450-m elevation). Goose Creek below the reservoir is diverted entirely for irrigation purposes and never reaches the Snake River. The Goose Creek basin is a matrix of sage-steppe *Artemisia tridentata* and pine-aspen-juniper forest; higher elevations contain pine-aspen forests *Pinus* spp., *Pseudotsuga* spp., and *Populus tremuloides* whereas lower elevations contain pinyon-juniper-mountain mahogany *Pinus monophylla*, *Juniperus* spp., *Cercocarpus ledifolius*. Riparian areas are comprised of willows *Salix* spp., alders *Alnus* spp., cottonwoods *Populus* spp., and sedges Family: Cyperaceae. Average annual precipitation is 18-cm, and streamflow patterns are dominated by snowmelt runoff. Goose Creek is one of the most fish-species-rich subbasins in the Snake River basin above Hells Canyon with 14 fish species have been documented (Table 1)(Meyer et al. 2013).

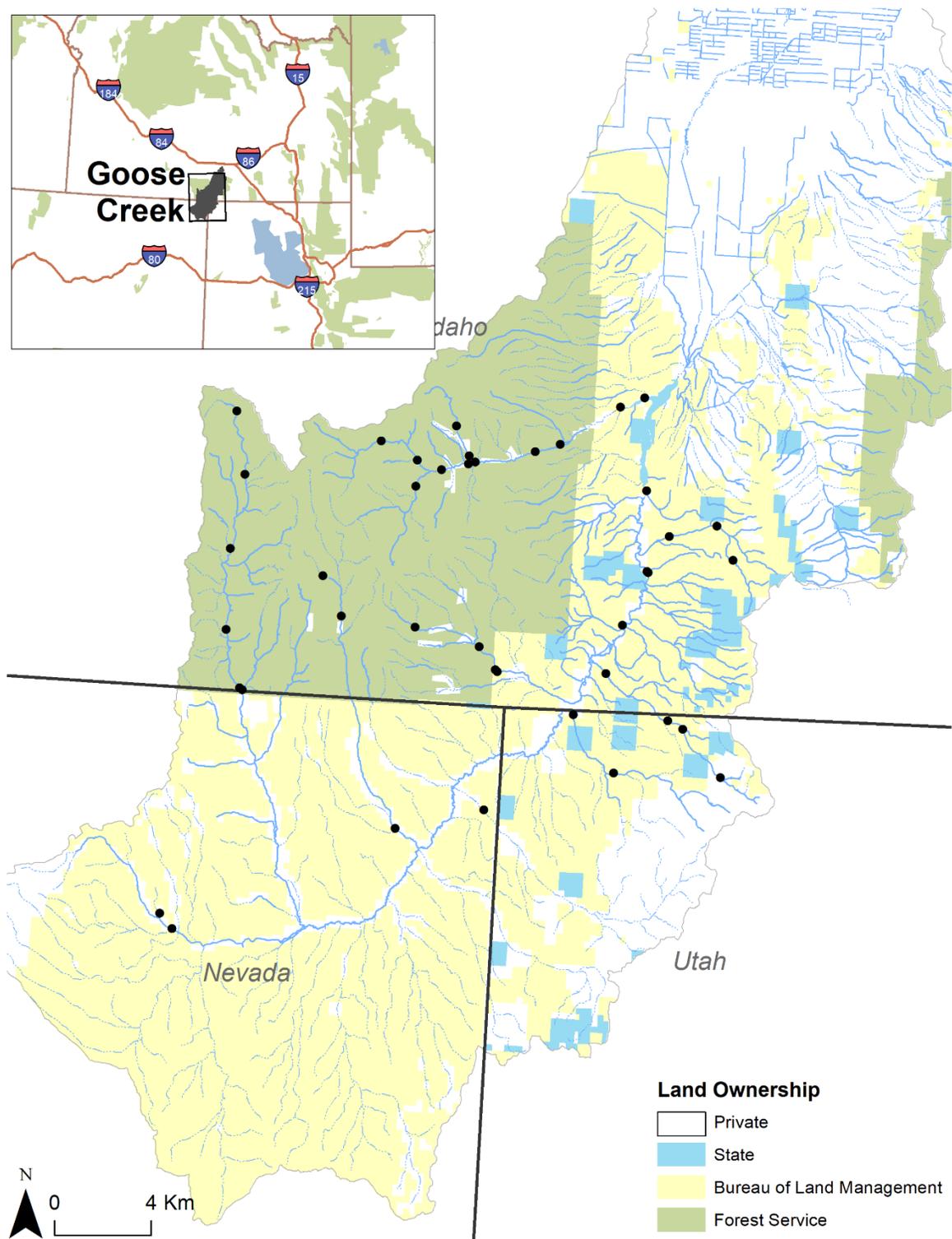


Figure 3. Locations of fish and habitat sample sites ( $n = 41$ ) in the Goose Creek watershed, 2013 to 2015.

The Goose Creek Subbasin was included on the Idaho priority list of impaired water (303(d)) and in 2004 a Total Maximum Daily Load (TMDL) was developed and approved for listed impairments (IDEQ 2010a); portions of the subbasin in Nevada have been or are currently on that state's list of impaired waters (NDEP 2014). Most major tributaries to Goose Creek are impaired, with the major impairments being: bacteria (fecal coliform and *Escherichia coli*), dissolved oxygen, total phosphorous, sediment and suspended solids, and temperature. Two tributaries and portion of the mainstem do not meet beneficial use designations, and road/trail decommissioning, road management, livestock exclusion, fence repair, riparian management, and streambank protections are listed as TMDL implementation actions (IDEQ 2010a). Cattle production is the predominant land use in the region (IDFG 2005), and long-term grazing is cited as having impacted Yellowstone cutthroat trout populations in the drainage (IDFG 2007). Lands adjacent to the Goose Creek mainstem are used for surface-irrigated hay pasture and winter grazing.

*Fish Sampling* - We conducted fish and habitat surveys at 41 sites in the Goose Creek watershed (Figure 3). Our sites were selected based on the stratified-random design used by Meyer et al. (2006) for Goose Creek, while adding additional sites selected to increase spatial coverage, represent the range of stream conditions (including grazing impacts) in the watershed, and increase sample size given access constraints to private land. Each site was sampled one time from 15 July to 6 October during low flow periods from 2013 to 2015 (number of samples in July = 10, August = 19, September = 7, October = 5). Sites ranged in wetted width from 0.4 to 6.2 m, and contributing watersheds ranged in size from 1.7 to 1,480 km<sup>2</sup>. At each site, a stream reach typically 100-m thalweg length (range: 47 to 165-m), similar to Meyer et al. (2006), and was isolated with 6.35-mm bar mesh block nets; nets were not used where impassible beaver dams coincided with upstream reach boundaries. Fishes were sampled with single-pass electrofishing using a Smith-Root LR-24 backpack electrofisher with one or two netters at 37 sites. Mainstem Goose Creek sites wider than 5.4 m wetted width (n = 4) were sampled with 2 LR-24 backpack electrofishers and four netters. Electrofishing was done using pulsed (40 Hz) direct current and 200-450 V. All fishes were identified to species and counted.

*Instream and Riparian Habitat Associations* - After completing electrofishing surveys, we assessed instream habitat, streambank condition, and riparian vegetation using transect-based sampling. At each site, one transect was established every 10-m along the reach beginning at the downstream reach boundary. Transects were placed across the stream channel at bankfull height, which was identified using the following indicators: height of depositional surfaces, perennial vegetation, topographic breaks, bank substrates, undercut banks, and water stain lines (Harrelson et al. 1994; Burton et al. 2011). Channel depth, water depth, water velocity, stream substrate, and cover were measured at 10 equidistant points along each transect (Platts et al. 1983). We measured velocities at 0.6 of water depth using a Hach FH950 velocity meter (HACH Company, Loveland, Colorado). Stream substratum at each point was classified according to the modified Wentworth scale, whereby particles are classified as: bedrock, silt/clay (<0.064-mm diameter on b-axis), sand (0.064-2-mm), gravel (2-15mm), pebble (15-64mm),

cobble (64-256mm), or boulder (>256-mm) (Cummins 1962). Cover was classified as: boulder, large wood (>10-cm diameter, >4-m in length), small wood, aquatic vegetation (macrophytes), overhanging bank vegetation, undercut bank (>10-cm depth), or absent. The surface water elevation difference between upstream and downstream reach boundaries was measured using a survey level and stadia rod, and stream slope was computed as the elevation difference divided by thalweg length (expressed as a percentage). Residual pool depth was calculated as maximum pool depth minus water depth at the downstream riffle crest for all pools identified using the classification of Hawkins et al. (1993). Woody vegetation height was classified above each transect endpoint at bankfull as: 0.0-0.5-m, 0.5-1.0-m, 1.0-2.0-m, 2.0-4.0-m, 4.0-8.0-m, and >8.0-m (Burton et al. 2011), and we computed percent woody vegetation as the percent of transect endpoints with woody vegetation greater than 1-m in height. Streambank stability was classified at each transect endpoint between the water's edge and bankfull height as: fracture, slump, slough, eroding, or absent (Burton et al. 2011). Streambank alteration was defined as the presence of cattle trails within 5-m of transect endpoints, and expressed as the percentage of transect endpoints with cattle trails present. Mean August stream temperature was measured using thermographs (TidbiT v2, Onset Computer Corporation, Bourne, Massachusetts) that recorded temperatures once each hour ( $n = 34$  in the Goose Creek watershed). Temperatures at sites without a thermograph were determined using data from the nearest one or two thermographs and distance interpolation or an elevation correction.

We evaluated associations among instream and riparian habitat variables using a principal components analysis (PCA). The PCA was fitted using the correlation matrix. A scree plot (PCA axis versus % variance explained) was used to determine the number of interpretable PCA axes.

*Fish Species Diversity and Habitat Diversity* - We evaluated how fish species diversity was associated with four dimensions of instream habitat diversity using multiple linear regression and data from sites where at least one fish species was present ( $n = 34$ ). Fish species diversity ( $H'_{spp}$ ) was computed using the Shannon-Wiener index:

$$H'_{spp} = - \sum_{i=1}^S p_i \cdot \log_e p_i$$

where  $p_i$  is the proportion of the total catch at a site comprised of species  $i$ , and  $S$  is species richness (i.e., total number of species) at a site (Legendre and Legendre 2012). Likewise, we computed four dimensions of habitat diversity. Cover diversity and substrate diversity were also computed using the Shannon-Wiener index ( $H'$ ), except that  $p_i$  represented the proportion of all cover (excluding the 'absent' category) or substrates as type  $i$ . Diversity of water velocity and water depth we computed as the standard deviation (SD) of water velocity (m/s) and SD of water depth (m), respectively.

Multiple linear regression was used to evaluate the effect of cover diversity ( $H'_c$ ), substrate diversity ( $H'_s$ ), SD of velocity (m/s), and SD of depth (m) on fish species diversity. Each habitat diversity element was a separate variable in the model, and we evaluated significance of each variable at  $\alpha = 0.10$  (instead of the more traditional  $\alpha = 0.05$ ) because we were more interested in

detection of associations that were real (statistical power; power =  $1 - \beta$ , where  $\beta$  is Type II error rate) along with their strength (i.e., effect size) versus safeguarding against interpreting an association as real when in fact it is not (i.e., Type I error rate,  $\alpha$ , false positive) (Yoccoz 1991). We also used a commonality analysis (aka, element analysis) to partition the variance in species diversity explained by the four dimensions of habitat diversity. Commonality analysis partitions variance in the response variable (species diversity) that is both unique to each explanatory variable as well as common to (or shared with) other explanatory variables, the latter of which cannot be discerned from standardized parameter estimates from multiple linear regression (Nimon et al. 2008). The analysis was done using the *yhat* package (Nimon et al. 2013) in Program R (R Core Team 2015).

*Habitat Diversity and Stream Condition Indicators* - We evaluated the effect of riparian and instream habitat condition on habitat diversity by using multiple linear regression. Five measures of stream condition indicators that reflect impacts to streamside vegetation and streambanks were evaluated: streambank condition (% bank sloughing/slumping), streambank alteration (% cattle trails), percent woody riparian vegetation (woody vegetation >1-m in height), percent fine substrates (sand/silt/clay), and channel width-to-depth ratio (channel width divided by mean channel depth) (Eaglin and Hubert 1993; Kershner et al. 2004; Burton et al. 2011; Swanson et al. 2015). These five stream condition indicators were used as covariates in multiple linear regressions with each of the four dimensions of habitat diversity as response variables: cover diversity ( $H'_C$ ), substrate diversity ( $H'_S$ ), SD of water velocity (m/s), and SD of water depth (m). We included residual pool depth (m) as a covariate for stream size in each of the multiple regressions, as we expected the SD in water velocity and depth to increase in larger streams and wanted to account for this expected variation. Significance of each habitat diversity dimension was evaluated at  $\alpha = 0.10$ . Models were refit with only significant terms to estimate parameters and variance explained (adjusted- $R^2$ ).

*Fish Assemblage Structure* – We evaluated the effect of riparian and instream habitat, including the four dimensions of habitat diversity, on fish assemblage structure (i.e., relative [proportional] abundance of species) using a constrained correspondence analysis (CCA). CCA is a direct gradient (constrained) ordination technique that uses a unimodal model to relate environmental variables to assemblage structure (ter Braak and Verdonschot 1995; Legendre and Legendre 2012). The species matrix used in the CCA was comprised only of sites where fish were present ( $n = 34$ ) and species abundances were un-transformed. The instream and riparian habitat variables evaluated for their influence on fish assemblage structure were placed into three categories: natural stream features, habitat diversity, and stream condition indicators. The suite of variables describing natural stream features were: mean August stream temperature ( $^{\circ}\text{C}$ ), stream slope (%), percent aquatic vegetation (percentage of transect points), percent overhanging vegetation, and percent small wood. Habitat diversity variables were: standard deviation (SD) of depth, SD of velocity, substrate diversity ( $H'_S$ ), and cover diversity ( $H'_C$ ); percent cobble/boulder substrate was omitted from analysis because of its high correlation with substrate diversity ( $r = 0.85$ ). The stream condition variables were: percent streambank sloughing/slumping, percent

streambank alteration (cattle trails), percent woody vegetation, percent fine substrate (sand/silt/clay), and channel width-to-depth ratio. Each variable was evaluated for significance ( $\alpha = 0.10$ ) using a permutation test with 9,999 permutations. A final CCA was refit using only significant variables, and partial CCAs were then used to determine the amount of variation in fish assemblage structure explained by the three variable sets: natural features, habitat diversity, and stream condition indicators (Økland 1999).

## Results

*Fish Sampling* - At least one fish species was collected at 34 of the 41 sites sampled. Among all 12 species collected, Speckled Dace were collected at the highest percentage of sites and were the most abundant across all sites (Table 1). Brook Trout, Rainbow Trout, and Yellow Perch (collected at one site near Lower Goose Creek Reservoir) were the only non-native fish species collected. Mottled Sculpin and Yellow Perch were only collected at one site each (2.4%) and were the least abundant. Species richness ranged from 0 to 8 fish species (mean = 3.1, SD = 2.5).

*Table 1. List of fishes documented in the Goose Creek watershed (Meyer et al. 2006; Blakney 2012; Meyer et al. 2013; Wallace and Zaroban 2013), and their prevalence (percent occurrence) and relative abundance from all 34 (of 41) sites with fish present (3,948 individuals total) sampled from 2013 to 2015. Species codes in parentheses. An asterisk indicates non-native species.*

Common name	Scientific name	Prevalence (%)	Relative abundance (%)
Speckled Dace (SPD)	<i>Rhinichthys osculus</i>	53.7	35.7
Paiute Sculpin (PSC)	<i>Cottus beldingii</i>	46.3	14.9
Brook Trout (BKT)*	<i>Salvelinus fontinalis</i>	36.6	9.2
Rainbow Trout (RBT)*	<i>Oncorhynchus mykiss</i>	34.1	3.7
Longnose Dace (LND)	<i>Rhinichthys cataractae</i>	31.7	8.4
Bluehead Sucker (BHS)#	<i>Catostomus discobolus</i>	31.7	6.6
Redside Shiner (RSS)	<i>Richardsonius balteatus</i>	24.4	17.4
Northern Leatherside Chub (NLC)	<i>Lepidomeda copei</i>	22.0	1.7
Yellowstone Cutthroat Trout (CUT)	<i>Oncorhynchus clarkii bouvieri</i>	17.1	2.1
Utah Sucker (UTS)	<i>Catostomus ardens</i>	7.3	0.18
Yellow Perch (YPC)*	<i>Perca flavescens</i>	2.4	0.05
Mottled Sculpin (MSC)	<i>Cottus bairdii</i>	2.4	0.03
Mountain Sucker (MTS)	<i>Catostomus platyrhynchus</i>	Not collected	--
Utah Chub (UTC)	<i>Gila atraria</i>	Not collected	--

#proposed reclassification as *Pantosteus virescens* (Unmack et al. 2014)

*Instream and Riparian Habitat Associations* - The PCA suggested that variation in habitat among sites was attributable to two main habitat gradients – a longitudinal stream size gradient and an instream cover gradient. Axis 1 explained 25.3% of the variance and Axis 2 explained 21.5%. The scree plot suggested axes 3 (10.7% of variance) and higher did not explain

substantially more variation than axes 1 and 2 and they were, therefore, were not interpreted. A plot of Axis 1 versus 2 showed a stream-size gradient where sites ranged from high gradient and cold with more bank sloughing and slumping, more woody riparian vegetation, and more small wood (low Axis 1 scores in Figure 4) to lower gradient sites with deeper residual pools, warmer temperatures, and more variation in water depths and velocities (high Axis 1 scores but low Axis 2 scores in Figure 4). A second gradient represented sites dominated by high percentages of fines substrates, aquatic vegetation, and overhanging vegetation (high Axis 1 and 2 scores in Figure 4), versus sites with more cobble and boulder substrates, and substrate and cover diversity (low Axis 1 and 2 scores in Figure 4).

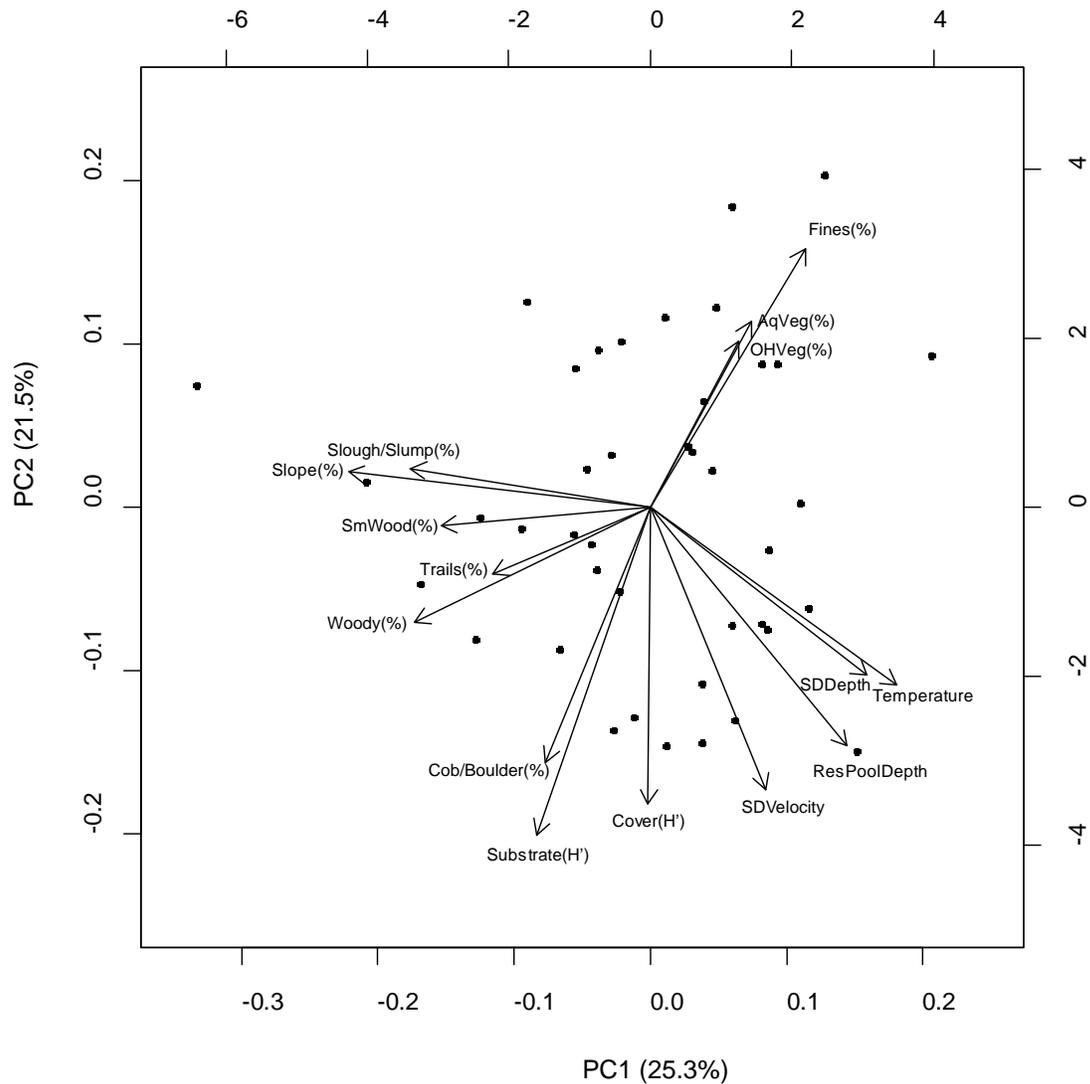


Figure 4. Plot of axis 1 versus 2 (top panel) from a principal component analysis showing interrelationships among instream and riparian habitat variables in Goose Creek, 2013 to 2015.

*Fish Species Diversity and Habitat Diversity* - Fish species diversity ( $H'_{spp}$ ) averaged 0.61 (range: 0 – 1.65) and was positively associated with all four dimensions of habitat diversity (cover, substrate, water velocity, and water depth) in the Goose Creek watershed (Table 2). The overall multiple regression model fit the data well ( $F_{4,29} = 15.8$ ,  $P < 0.001$ ), and explained 64% of the variance in fish species diversity (adjusted- $R^2 = 0.642$ ). All variables were significantly different from zero, and only SD of water velocity would have been unsupported using a more restrictive Type I error rate ( $P = 0.09$ ). Standardized parameter estimates showed SD of depth to have the strongest positive influence on fish species diversity in the watershed, followed by substrate diversity having the second strongest influence; SD of velocity was estimated to have the smallest effect (Table 2; Figure 5). The commonality analysis suggested that SD of depth explained the most variation in species diversity independent of the other habitat diversity dimensions, confirming what was reflected by the standardized parameter estimates from the multiple regression. However, the commonality analysis also showed that although cover diversity and SD of velocity explained little variation in species diversity individually, they both shared a substantial amount of variation with the other dimensions of habitat diversity and, in total, explained almost as much total variance in species diversity as SD of depth (Table 2). This suggests that these different elements of habitat diversity covary with one another and have a shared influence on species diversity.

Table 2. Parameter estimates, standard errors (including standardized estimates), P-values, and proportion of unique, common, and total variance from a multiple linear regression of four habitat diversity variables evaluated for their effect on fish species diversity in Goose Creek.

Variable	Unstandardized	Standardized	P-value	Variance Contribution ( $p$ )		
	$\beta_i$ (1 SE)	$\beta_i$ (1 SE)		Unique	Common	Total
Intercept	-0.993 (0.269)	<0.001 (0.103)	0.001*			
Cover diversity ( $H'_c$ )	0.461 (0.185)	0.294 (0.118)	0.019	0.07	0.22	0.29
Substrate diversity ( $H'_s$ )	0.543 (0.183)	0.334 (0.112)	0.006	0.10	0.09	0.19
SD of velocity (m/s)	1.789 (1.020)	0.200 (0.114)	0.090	0.03	0.18	0.22
SD of water depth (m)	6.786 (1.493)	0.500 (0.110)	<0.001	0.22	0.12	0.34

\*P-value for intercept is for standardized parameter estimate only.

*Habitat Diversity and Stream Condition Indicators* - Variation in each of the four dimensions of habitat diversity was explained, to varying degrees and in different ways, by stream condition indicators after accounting for a strong and persistent stream-size effect (by including the residual pool depth covariate). Residual pool depth had a significant, positive association with each of the four dimensions of habitat diversity, suggesting a stream habitat diversity gradient that increased downstream as streams became larger (Table 3). Cover diversity decreased as streams became wider and shallower with a higher prevalence of fine substrates. Substrate diversity also decreased when fine substrates were more prevalent, but unexpectedly increased with more livestock trails along the streambank. SD of water velocity was lower with more streambank sloughing and slumping and more fine substrates, and SD of water depth decreased with more stream bank sloughing and slumping independent of stream size (Table 3). Only SD of water depth could be predicted with reasonable precision (adjusted- $R^2 = 0.61$ ; Table 3).

Table 3. Parameter estimates ( $\pm 1$  SE),  $P$ -values, and adjusted  $R^2$  for models predicting four dimensions of stream habitat diversity a function of indicators of stream condition. Parameter estimates and adjusted  $R^2$  are for models refit with only significant predictor variables ( $P < 0.10$ ).

Response	Predictors	$\beta_i$ (1 SE)	$P$ -value	Adjusted- $R^2$
Cover diversity ( $H'_c$ )	Woody vegetation (%)		0.602	0.23
	Streambank slough/slump (%)		0.608	
	Streambank trails (%)		0.171	
	Channel width-to-depth ratio	-0.015 (0.008)	0.079	
	Fine substrates (%)	-0.005 (0.002)	0.015	
	Residual pool depth (m)	0.706 (0.288)	0.021	
Substrate diversity ( $H'_s$ )	Woody vegetation (%)		0.739	0.45
	Streambank slough/slump (%)		0.610	
	Streambank trails (%)	0.007 (0.003)	0.095	
	Channel width-to-depth ratio		0.715	
	Fine substrates (%)	-0.011 (0.002)	<0.001	
	Residual pool depth (m)	0.593 (0.286)	0.060	
SD of velocity (m/s)	Woody vegetation (%)		0.210	0.28
	Streambank slough/slump (%)	-0.002 (0.001)	0.012	
	Streambank trails (%)		0.102	
	Channel width-to-depth ratio		0.237	
	Fine substrates (%)	-0.001 (0.0003)	0.015	
	Residual pool depth (m)	0.116 (0.055)	0.042	
SD of depth (m)	Woody vegetation (%)		0.580	0.61
	Streambank slough/slump (%)	-0.001 (0.0003)	0.039	
	Streambank trails (%)		0.856	
	Channel width-to-depth ratio		0.216	
	Fine substrates (%)		0.135	
	Residual pool depth (m)	0.177 (0.026)	<0.001	

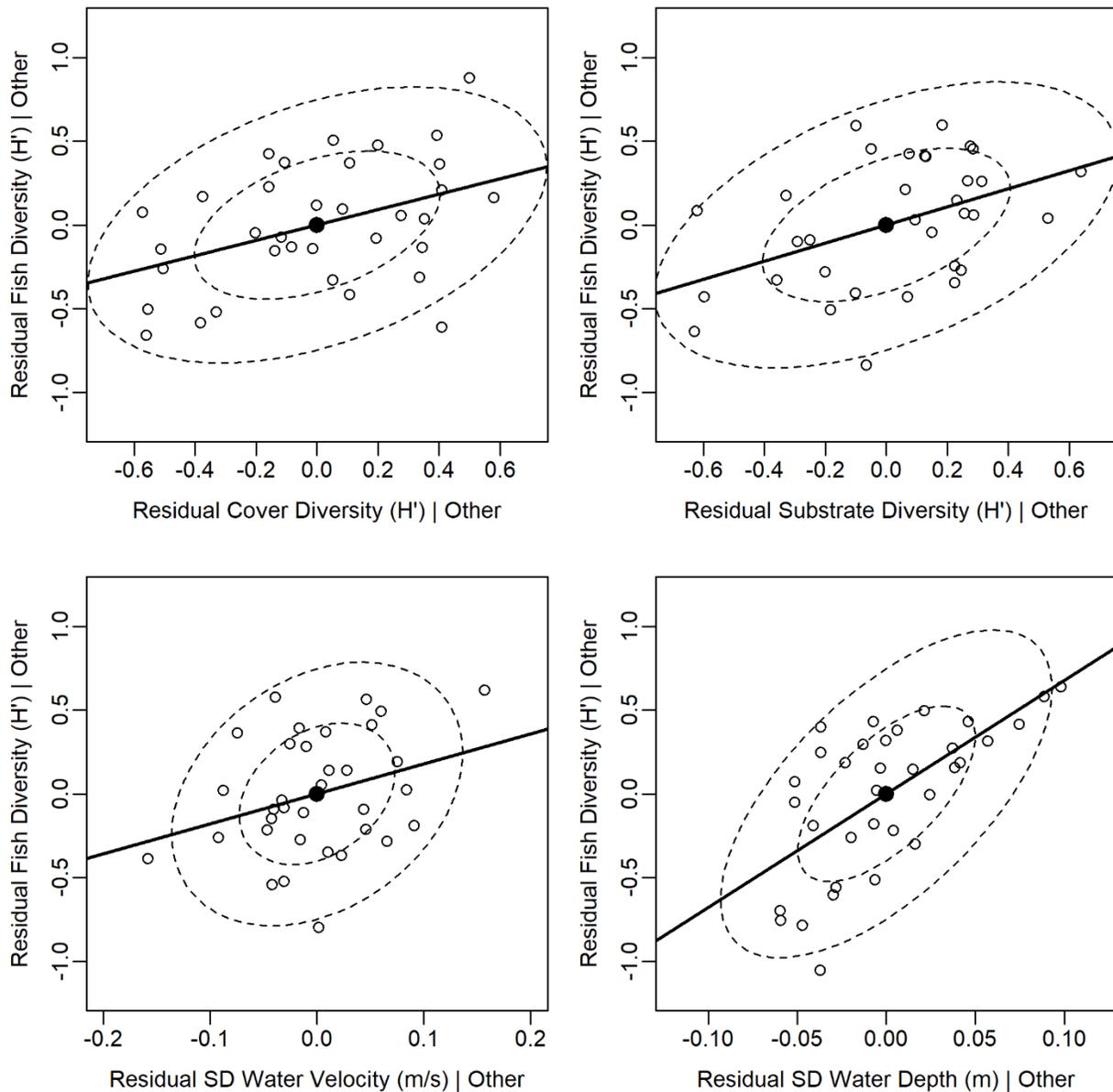


Figure 5. Partial regression plots showing the effect of four dimensions of stream habitat diversity on fish species diversity in Goose Creek, 2013 to 2015. Confidence ellipses represent the 50th and 90th percentiles of residual points.

**Fish Assemblage Structure** - Since Mottled Sculpin and Yellow Perch were collected at fewer than two sites they were omitted from the CCA. The CCA showed only a few instream and riparian habitat variables to be significantly associated with fish assemblage structure: mean August stream temperature ( $P < 0.001$ ), slope ( $P = 0.012$ ), SD of velocity ( $P = 0.011$ ), and percent woody vegetation ( $P = 0.009$ ) (Table 4). Although percent fine substrates (clay/silt/sand) ( $P = 0.160$ ) did not meet our alpha criterion, we retained it in subsequent analyses to explore its association with fish assemblage structure as has been shown in numerous other studies (Waters

1995; Henley et al. 2000); it was the only variable close to our significance threshold. A CCA re-fit with only these variables showed a stream-size gradient where Brook Trout, Paiute Sculpin, Rainbow Trout, and Cutthroat Trout were most abundant in cold, high gradient sites (CCA Axis 1; top left panel of Figure 6). A secondary gradient showed Rainbow Trout and Paiute Sculpin to be more abundant at sites with more woody riparian vegetation and greater variation in water velocity (CCA Axis 2; top left panel of Figure 4).

*Table 4. Significance (P-value) and variance inflation factor (VIF) of instream and riparian habitat variables (permutation test with 9,999 permutations) evaluated in a CCA for their influence on fish assemblage structure. The variable Percent Cobble/Boulder was omitted due to high correlation with Substrate Diversity ( $r = 0.85$ ).*

Variable	P-value	VIF
Temperature (°C)	<0.001	3.58
Slope (%)	0.012	2.07
Residual pool depth (m)	0.983	3.15
SD velocity (m/s)	0.011	1.99
SD water depth (m)	0.436	4.44
Woody vegetation (%)	0.009	3.36
Aquatic vegetation (%)	0.433	6.62
Small wood (%)	0.734	2.04
Overhanging vegetation (%)	0.625	2.41
Bank Trails (%)	0.745	3.02
Bank slough/slump (%)	0.603	2.39
Fine substrate (%)	0.160	6.02
Cover diversity ( $H'_c$ )	0.900	5.18
Substrate diversity ( $H'_s$ )	0.892	2.95

Variance partitioning using partial CCAs showed that the natural stream features – stream temperature and stream slope – explained more variation (51%) in assemblage structure than the variable sets representing stream condition indicators (24%) and habitat complexity (22%). There was little explained variation in fish assemblage structure shared among the three variable sets (Table 5). Partial ordinations showed partitioning among the three trout species across cold streams, where Brook Trout were more abundant in high gradient cold sites, Rainbow Trout and Cutthroat Trout were more abundant at cold sites with intermediate gradients (bottom left panel of Figure 4). The partial ordination emphasizing grazing indicators showed Cutthroat Trout to be more abundant at sites with less fine substrates, and Paiute Sculpin to be more abundant at sites with more woody riparian vegetation (top right panel of Figure 4). The partial ordination of habitat complexity showed Brook Trout to be more abundant at sites with less flow complexity (SD of velocity) and Rainbow Trout to be more abundant at sites with high flow complexity (lower right panel of Figure 4).

Table 5. Variance in fish assemblage structure attributable to natural stream features, indicators of stream condition, and habitat complexity, and shared variances ( $\cap$ ) in the Goose Creek watershed.

Factor	Number of variables	$\sum$ Canonical eigenvalues	% variance	Mean % per variable
Natural features (% slope, temperature [ $^{\circ}$ C])	2	0.536	50.9	25.5
Condition (% woody vegetation, % fines)	2	0.255	24.2	12.1
Habitat complexity (SD velocity [m/s])	1	0.230	21.8	21.8
Natural $\cap$ Condition	4	0.010	1.0	0.25
Natural $\cap$ Complexity	3	0.050	4.8	1.60
Condition $\cap$ Complexity	3	-0.031	-3.0	-1.00
Natural $\cap$ Condition $\cap$ Complexity	5	0.005	0.4	0.08



Conducting a fish community survey on Goose Creek using backpack electrofishing, 2014. Credit: R. Bjork.

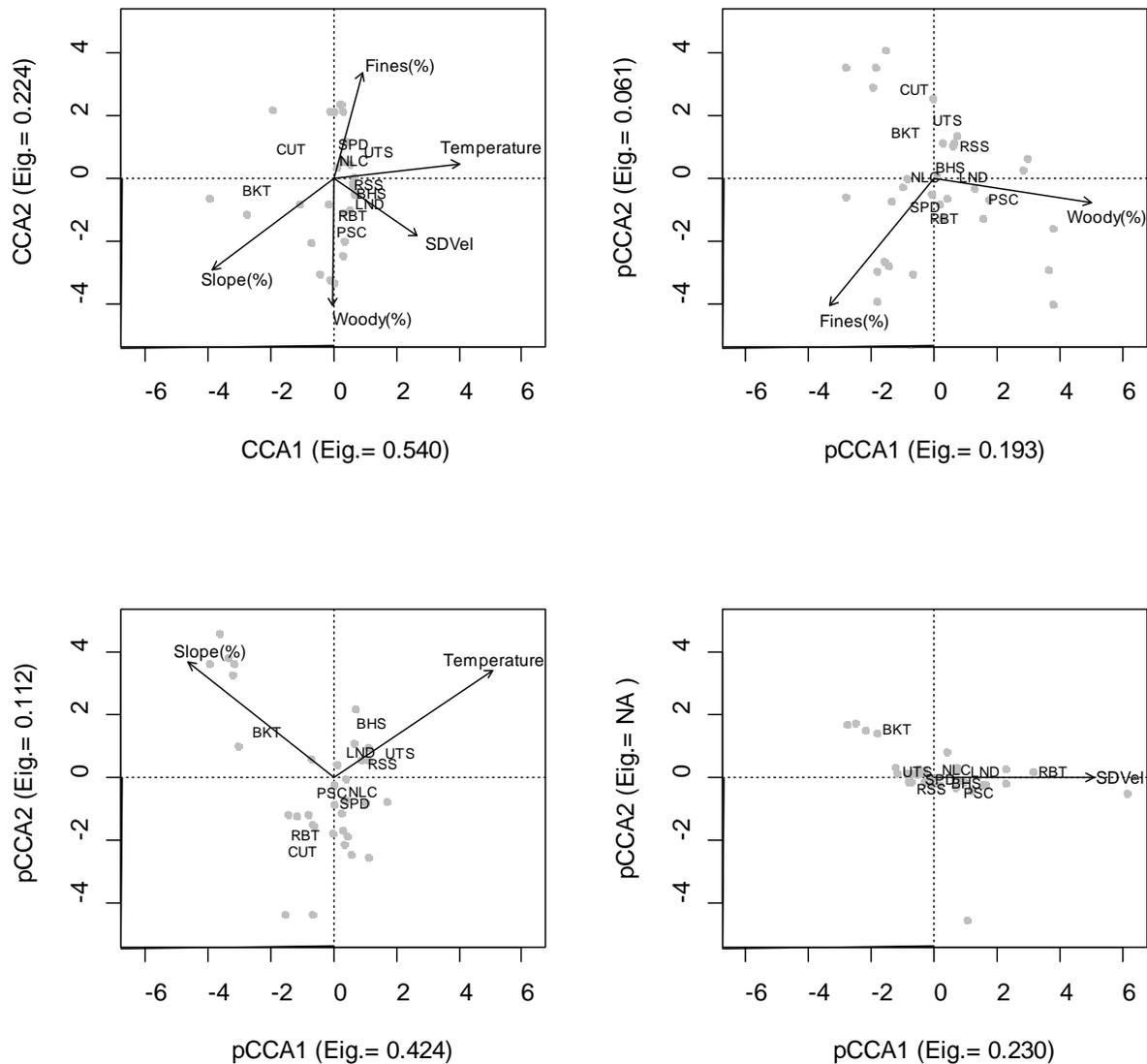


Figure 6. Constrained correspondence analysis (CCA) biplots showing associations of fish species to instream and riparian habitat in Goose Creek, 2013 to 2015. Top left panel shows biplot including all variables, bottom left shows partial CCA biplot with natural stream feature variables, top right panel shows partial CCA with stream condition variables, and bottom right shows partial CCA biplot with a habitat complexity variable. See Table 1 for species codes.

## Discussion

We found fish species diversity to be positively associated with all four dimensions of instream habitat diversity we studied (cover, substrate, water velocity, and water depth), confirming that more diverse habitat is likely to have more of the unique niches available to potential species comprising fish assemblages in our disturbed study watershed. While we expected at least some

dimensions of habitat diversity to influence fish species diversity, we were surprised that all four contributed to fish diversity in some way given the low number of species in our watershed when compared to other regions. Numerous studies have associated habitat diversity with species diversity across taxa (MacArthur 1964; Rosenzweig and Winakur 1969; Allan 1975), but rarely has every element of habitat diversity under study contributed to fish species diversity (Gorman and Karr 1978; Jackson et al. 2001). For example, Gorman and Karr (1978) found that diversity in water velocity and depth influenced fish species diversity in both north temperature and tropical streams, but found no influence of substrate diversity and they didn't evaluate cover diversity. They suggested that the importance of habitat diversity dimensions to species diversity varies stream-by-stream and the guilds represented in the species pool, such as if riffle-dwelling species are present. Previous studies have mostly been conducted in species-rich streams, but we found that all four dimensions of habitat diversity were important to fish diversity even despite our observed species pool being comprised of only 12 species, three of which are non-native and two that were only collected at one site each. While Goose Creek represents a depauperate species pool when compared to those elsewhere in the United States (Abell et al. 2000; Abell et al. 2008), it is one of the more speciose tributaries in the Snake River basin due to occurrence of rare species such as Northern Leatherside Chub and Bluehead Sucker (Meyer et al. 2013).

Our analyses also suggested that habitat diversity persistently increased with stream size. Stream size is often positively associated with fish species diversity due to increased habitat volume and, presumably, a higher diversity of available resources (Vannote et al. 1980; Angermeier and Schlosser 1989), and our study shows this linkage directly with physical habitat availability. Sheldon (1968) found that fish assemblages in a New York watershed changed mainly through species additions (species replacement was minor) that occurred due to increases in water depth and habitat diversity that generally, but not systematically, increased downstream. Rahel and Hubert (1991) found that coldwater species in the headwaters of a Rocky Mountain streams were replaced by warmwater species downstream where additional warmwater species were continually added to the assemblage, that is, they found a strong longitudinal gradient in fish assemblages related to stream size (and the other habitat features associated with stream size). However, when using a coefficient of variation as a measure of habitat diversity they found habitat diversity to be uncorrelated with stream size. The strength of species diversity - habitat diversity associations, if present, likely depends on the range of stream sizes studied and how diversity and thus resource availability is quantified (Tews et al. 2004). We suspect that resource availability is best represented as an unstandardized measure of variability, such as with a standard deviation or diversity index (categorizing continuous data when needed), instead of standardized measure such as a coefficient of variation. Habitat diversity, heterogeneity, and complexity have been defined and quantified myriad ways (Schlosser 1982; Rahel and Hubert 1991; Tews et al. 2004; Taylor et al. 2006; Smith et al. 2016).

Although habitat diversity primarily increased along a stream-size gradient, we determined that stream condition influenced habitat diversity independent of this stream size continuum, a novel aspect of our study. Degraded streams have wide, shallow channels dominated by fine sediments, a lack of riffles and pools, and a lack of instream cover (Kauffman and Krueger 1984; Armour et al. 1991). In almost every case in our study, stream condition indicators had a negative association with the different dimensions of habitat diversity, with a presumably adverse effect on fish diversity as well. What requires further study, however, is whether

streams with low habitat diversity also have habitat conditions that vary more over time. For example, grazing can compact soils and increase overland flow and storm runoff, thus increasing streamflow variability and habitat variability over time (Marston 1994; Trimble and Mendel 1995). Temporal variability in habitat, in turn, has been shown to negatively affect fish assemblage stability (Taylor et al. 2006) and decrease the strength of habitat diversity – fish diversity associations (Schlosser 1982). This phenomenon requires further study in species poor systems such as stream fish communities in the northwestern United States.

While habitat diversity influenced fish species diversity, the constrained ordination showed that habitat diversity explained only a small fraction of fish assemblage structure. Fish assemblages were structured primarily along a longitudinal stream size gradient in Goose Creek; streamflow diversity (SD of velocity) was the only dimension of habitat diversity to significantly influence assemblage structure directly (and was the weakest dimension associated with fish diversity). Therefore, while habitat diversity and species diversity increased in larger streams in our study area, the relative abundances of individual species in the assemblage were largely un-influenced, per se, by habitat diversity. The one exception was that Brook Trout and Rainbow Trout were partitioned along a gradient of streamflow diversity where Brook Trout were more abundant at cold stream sites with simple streamflows and Rainbow Trout were more abundant at cold sites with complex flows (lower right panel of Figure 6). Other studies have determined the structure of the fish assemblages to be primarily a function of stream size and longitudinal gradient (Schlosser 1982; Schultz et al. 2012). For example, Rahel and Hubert (1991) concluded that fish assemblages in a Wyoming stream were structured primarily along a longitudinal thermal gradient, with a trout-dominated assemblage in the headwaters that transitioned to warm-water assemblage that became more species rich downstream, a pattern also reflected in fish assemblages of Goose Creek.

Although salmonids dominated fish assemblages in small, cold streams, we also observed apparent partitioning by salmonids among the coldest headwater streams. Brook Trout were abundant in the high gradient, cold streams, whereas Cutthroat Trout and Rainbow Trout were abundant only in the moderate gradient, cold sites that we sampled, a pattern also observed by Maret et al. (1997) for least-disturbed streams in the Upper Snake River Basin. The mechanisms driving patterns of salmonid relative abundances requires further study, as those patterns could reflect negative competitive interaction (Peterson et al. 2004; Benjamin et al. 2011), hybridization (Meyer et al. 2006), and stocking and invasion history (Benjamin et al. 2007; Neville and Bernatchez 2013). Inconsistent capture of Rainbow Trout suggests this species has invaded only the lower portion of the watershed in streams nearest Lower Goose Creek Reservoir. The diverse fish community in the Goose Creek mainstem may provide some resistance to invasion by Rainbow Trout into tributaries further from the reservoir, as diverse communities are suggested to resist invasion by non-native species (Moyle and Light 1996). Continued monitoring of the fish community would help detect further invasions, species interactions and replacements, and any homogenization of the fish community in the watershed over time (Rahel 2002; Meyer et al. 2014).

In addition to fish assemblages being structured along a natural longitudinal gradient, variance partitioning suggested that the two indicators of stream condition included in the ordination explained 24% of fish assemblage structure independent of the observed longitudinal gradient.

This suggests that while land management can have some influence on habitat diversity (and thus species diversity), it can have a stronger influence on the specific habitat components structuring fish assemblages. It is well documented that logging, road building, grazing and other land uses lead to sedimentation in streams (Eaglin and Hubert 1993; Waters 1995). Regardless of the source, fine sediments smother and embed larger substrates and adversely affect lithophilic spawning and insectivorous fishes (Berkman and Rabeni 1987; Dauwalter et al. 2003). These land uses can also result in reductions in woody riparian vegetation, streambank stability, and terrestrial invertebrate prey inputs and, in turn, negatively impact stream fishes (Bayley and Li 2008; Saunders and Fausch 2012). Mature riparian vegetation results in more overhanging vegetation other habitats used by fishes in the Upper Snake River Basin (Meyer et al. 2013; Dauwalter et al. 2014; Dauwalter et al. 2015). Thus, strategic management of timber harvest, roads, grazing and other land uses can aid the protection and enhancement of stream habitats and fishes; for example, implementation of timber harvest and road construction best management practices (e.g., stream buffers) can minimize impacts to stream ecosystems (Angermeier et al. 2004). Likewise, grazing management such as riparian exclosures, off-stream watering facilities, rotational grazing, and reduced cattle stocking densities can reduce the impacts of grazing on stream ecosystems (DelCurto et al. 2005; Saunders and Fausch 2009; Tufekcioglu et al. 2013; Swanson et al. 2015). Active restoration in heavily impacted stream reaches can also improve riparian vegetation, instream habitat, and overall habitat diversity (Laub et al. 2012). Thus, land management and active restoration are both will likely play a role in the conservation of fish assemblages, such as those in Goose Creek, that represent a diversity hotspot within a larger river basin impacted by land and water uses and show a close linkage to instream habitat diversity (Hauer and Lorang 2004; Meyer et al. 2013).

### **Supplemental Resources:**

Webpage: [www.tu.org/habdivers-fishdivers-links](http://www.tu.org/habdivers-fishdivers-links)



*Bluehead sucker collected from Trapper Creek, 2013. Credit: D. Dauwalter.*

## CHAPTER 3: BEAVER DAMS, STREAMFLOW COMPLEXITY, AND THE DISTRIBUTION OF A RARE MINNOW, *LEPIDOMEDA COPEI*

### Abstract

Freshwater fishes are threatened globally, and often too little is known about threatened species to effectively guide their conservation. Habitat complexity is linked to fish species diversity and persistence, and degraded streams often lack habitat complexity. Beaver *Castor spp.*, in turn, have been used to restore streams and increase habitat complexity. The Northern Leatherside Chub *Lepidomeda copei* is a rare, small-bodied, drift-feeding minnow that has anecdotally been observed to use complex habitats associated with beaver dams in the western United States. To investigate this anecdote, we conducted fish and habitat surveys, the latter focusing on quantifying habitat complexity, in a subbasin of the Upper Snake River Basin in the U.S.A. Complementary generalized linear model and path analyses revealed that Northern Leatherside Chub occurred more often at sites with complex streamflows, and streamflows were more complex when beaver dams were present and pools were deeper. Northern Leatherside Chubs were also more likely to occur when temperatures were warmer, aquatic macrophytes were abundant, and stream channels were narrow and deep. The linkage between chubs, complex streamflows, and beaver dams needs to be evaluated more broadly to completely understand its role in the rangewide status of the species. However, it does suggest that increased use of beaver reintroductions and dam analogs for stream restoration could be a boon for the Northern Leatherside Chub, but such efforts should be monitored to determine their effectiveness to help adapt beaver-based restoration approaches to best benefit the species.



*Northern Leatherside Chub collected from Trapper Creek, 2013. Credit: D. Dauwalter.*

## Introduction

Freshwater fishes are threatened globally. Nearly 70% of over 15,000 known freshwater fish species have gone extinct and 31% for which reasonable data exist are threatened with extinction (Darwall and Freyhof 2016). The most common threats include pollution, human use of water resources, harvest, non-natives species, and habitat degradation among others (Helfman 2007). While some species are well studied, the International Union for Conservation of Nature (IUCN) Red List of Threatened Species – the only globally consistent threat assessment of species – shows there to be not enough information (i.e., data deficient) to even determine a status for over 1,500 freshwater fish species (Darwall and Freyhof 2016). Even when the status is known, the biology of rare species may not be understood well enough to determine what is driving their status and what might be needed to improve it (Marcot and Flather 2007).

Habitat complexity is linked to fish community diversity in stream systems. Several studies have shown this linkage, purporting that higher diversity in habitat types results in more unique niches available to be occupied by more species (Gorman and Karr 1978; Walrath et al. 2016). Habitat complexity is also thought to be correlated with fish species persistence because complex habitats are more likely to have all habitats needed for a species to meet its life history requirements (Horan et al. 2000). Degraded streams often have lower habitat diversity (Walrath et al. 2016), and increasing habitat complexity is often a goal of stream and river restoration (Palmer et al. 2010). For example, Billman et al. (2013) determined that increasing habitat complexity through restoration of side channels facilitated coexistence of native fishes in the presence of a non-native predator in the Provo River, Utah, U.S.A.

The Eurasian beaver *Castor fiber* and American beaver *C. canadensis* are semi-aquatic rodents that require water deep enough to support a winter food cache in cold climates, ensure that their burrow entrance remains submerged, provide predator refugia, and aid in collection and transport of woody materials (Novak 1987; Collen and Gibson 2001). As a result, beaver often build dams on small streams to create impoundments of sufficient depth. Beaver dams, which often occur in multiples, can be built across a range of stream gradients (Beck et al. 2010; Macfarlane et al. 2017b) and can drastically alter stream ecosystems, thus leading beaver to be considered a keystone species (Collen and Gibson 2001). In brief, beaver dams and impoundments can alter stream hydraulics and channel morphology, stream hydrology, water temperatures, and water quality (Hammerson 1994). These changes, in turn, have been shown to alter macroinvertebrate and fish communities, with responses dependent on impoundment age and location within the stream network, among other factors (Collen and Gibson 2001). One notable influence beaver have on stream ecosystems is that they increase stream channel and in-stream habitat complexity (Polvi and Wohl 2012). For example, habitat complexity (heterogeneity) and fish diversity and abundance was shown to be greater around beaver dams in the Ipswich River, a low-gradient catchment in Massachusetts, U.S.A (Smith and Mather 2013).

The Northern Leatherside Chub *Lepidomeda copei* is a rare small-bodied cyprinid residing in the Upper Snake River Basin and portions of the Bonneville Basin in the western United States (Johnson et al. 2004). Despite a broad geographic range where recent genetic studies suggest historical connectivity among populations, the species currently has a rare and patchy distribution within these basins (Johnson et al. 2004; Blakney et al. 2014; Schultz et al. 2016).

The species is generally known to inhabit low velocity areas (pools) of small streams with depth variability and to occasionally use intermittent streams (Wilson and Belk 2001; Schultz 2014; Schultz et al. 2016). Laboratory studies have shown the species to initiate spawning above 19°C and select spawning substrates 21 to 48 mm diameter in areas with higher water velocities (19 cm/s) (Billman et al. 2008a). Optimal growth of age-0 Northern Leatherside Chub is achieved at 23°C, and the upper incipient lethal temperature (UILT) ranges from 26.6 to 30.2°C depending on acclimation temperature (Billman et al. 2008b). A genetic study showed Northern Leatherside Chub to have low allelic diversity and that populations are genetically unique even within subbasins (~3,000 km<sup>2</sup>) owing to habitat fragmentation from land use, use of piscicides for fish management, and non-native predators (Walser et al. 1999; Blakney et al. 2014). In addition, some have purported that Northern Leatherside Chub reside in areas with complex habitat and that the species' reduced distribution reflects a loss of habitat complexity rangewide due to declines in beaver, loss of riparian vegetation, and impacts from over grazing (Blakney 2012). In response to this assertion, Dauwalter et al. (2014) evaluated microhabitat use by Northern Leatherside Chub with a focus on habitat complexity, riparian vegetation, and beaver dams in Trapper Creek (Goose Creek subbasin), a stream where Northern Leatherside Chub are locally abundant. They determined that Northern Leatherside Chub selected deep areas with heterogeneous depths and velocities, overhanging vegetation that was often branches from mature riparian shrubs, and wood associated with beaver dams. While the aforementioned study explained the small-scale distribution of the Northern Leatherside Chub in Trapper Creek, a stream where the species is abundant, it still remained unclear whether habitat complexity, riparian vegetation, and beaver dams influence the species' distribution within watersheds and rangewide as observed anecdotally by others (Blakney 2012). Thus, our objective was to understand the role of habitat complexity and beaver dams in determining the reach-scale distribution of Northern Leatherside Chub within the Goose Creek subbasin in the Upper Snake River Basin at the western edge of the species' range. This additional understanding will help inform conservation strategies for the Northern Leatherside Chub.

## Study area

The Goose Creek subbasin straddles the borders of Idaho, Nevada, and Utah in the western United States (Figure 7). Goose Creek heads in Idaho on the Sawtooth National Forest around 2,200 m elevation. It then flows south into Nevada, east into Utah, and then north back into Idaho where it is impounded by Oakley Dam to form Lower Goose Creek Reservoir (1,400 m elevation). Below the dam all water is used for agriculture and, thus, Goose Creek no longer connects to the Snake River. Higher elevations are a matrix of sage steppe and mixed pine-aspen-juniper (*Pinus* spp.-*Populus* spp.-*Juniperus* spp.) forest; lower elevations are pine-juniper-mountain mahogany (*Cercocarpus* spp.). Riparian areas are primarily comprised of willows (*Salix* spp.), alders (*Alnus* spp.), cottonwoods (*Populus* spp.) and sedges (Cyperaceae). Goose Creek streamflow patterns are influenced by snowmelt run-off and summer thunderstorms. Portions of the subbasin have been listed on state lists for impaired waters (IDEQ 2010a; NDEP 2014). The major impairments of different Goose Creek tributaries result from fecal coliform and *Escherichia coli*, dissolved oxygen, phosphorous, sediment, suspended solids, and temperature; these impairments result from roads, trails, and livestock production (IDFG 2005;

IDFG 2007). The Goose Creek mainstem is mainly surrounded by surface-irrigated hay and winter grazing pastures. The subbasin has some of the highest fish diversity in the Upper Snake River Basin (Meyer et al. 2013). In addition to the Northern Leatherside Chub, the following species have been collected from Goose Creek in the recent past: Yellowstone cutthroat trout *Oncorhynchus clarkii bouvieri*, non-native brook trout *Salvelinus fontinalis*, non-native rainbow trout *O. mykiss*, bluehead sucker *Catostomus discobolus*, mountain sucker *C. platyrhynchus*, Utah sucker *C. ardens*, longnose dace *Rhinichthys cataractae*, specked dace *R. osculus*, redbelt shiner *Richardsonius balteatus*, Utah chub *Gila atraria*, mottled sculpin *Cottus bairdii*, and Paiute sculpin *C. beldingii* (Blakney 2012; Meyer et al. 2013). Goose Creek represents a genetically unique Northern Leatherside Chub population (or populations) isolated from others (Blakney et al. 2014).

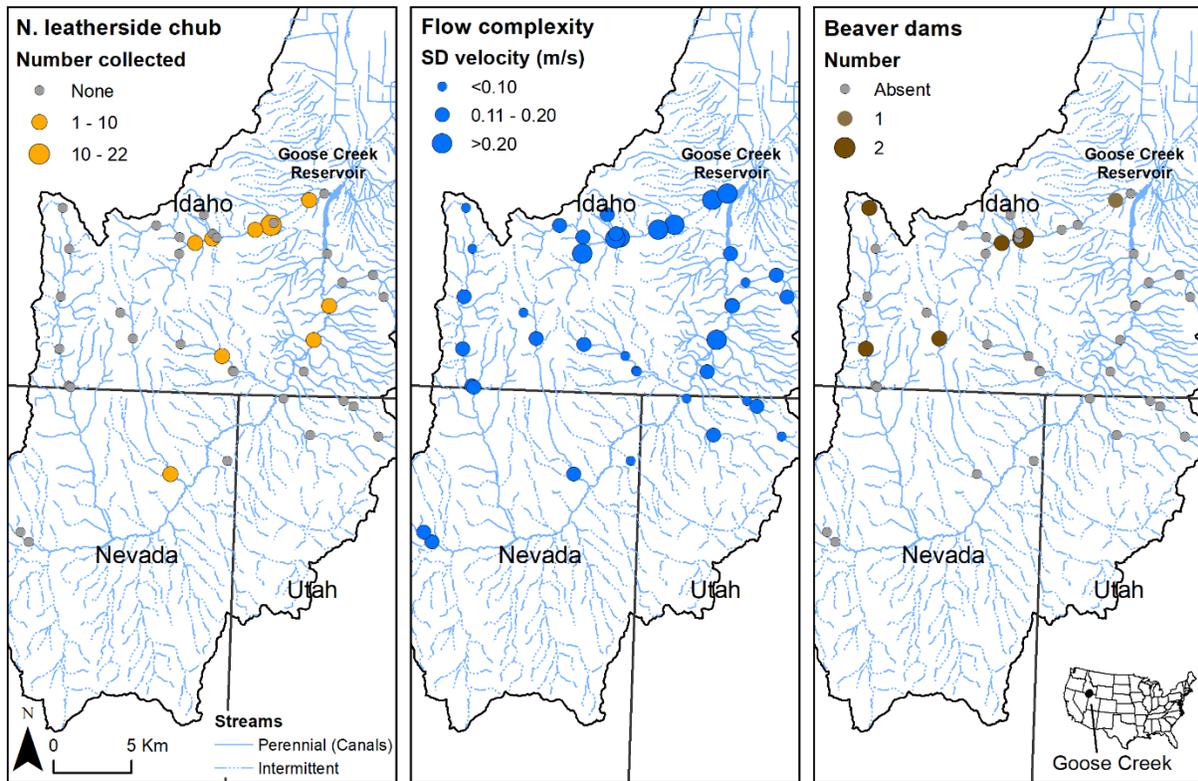


Figure 7. Distribution of sample sites and location of Northern Leatherside Chub occurrences (left panel), streamflow complexity (middle panel), and the presence of beaver dams (right panel).

## Methods

Forty-one sites were sampled from 2013 to 2015 in the Goose Creek watershed to determine the distribution of the Northern Leatherside Chub and how it is associated with instream and riparian habitat with a focus on habitat complexity and beaver dams. Sites were selected based on a stratified (by stream order) random sampling frame used by Meyer et al. (2006) where first order streams were under-sampled but the remaining were sampled in proportion to availability. The

sampling frame was adjusted due to denied access to private land and, in some cases, to increase spatial coverage and the range of habitat conditions sampled. These sites also cover the general stream segments in the subbasin sampled by Blakney (2012) to collect tissue samples for Northern Leatherside Chub genetic analysis, and are a subset of the streams where he anecdotally observed the association between Northern Leatherside Chub, habitat complexity, and beaver dams as described earlier. Sites were typically 100 m in length and isolated by 6.35 mm bar block nets, although site length was sometimes adjusted so that site boundaries coincided with habitat features to ensure secure block net sets. Sites were sampled using a single pass with one Smith-Root LR-24 backpack electrofisher and one or two netters at 37 sites, and two backpack electrofishers and four netters on mainstem Goose Creek where sites were greater than 5.4 m in wetted width. Up to three passes were conducted when salmonids were present. Electrofishing was conducted using direct current at 200-450 V and 40 Hz. All Northern Leatherside Chub were counted.

Instream and riparian habitat was also measured at each site after electrofishing surveys. A transect was established every 10 m along each site perpendicular to the channel at bankfull height. Channel depth, water depth, water velocity, stream substrate, and cover type were measured at 10 equally spaced points along each transect. Water velocity was measured at 0.6 of water depth using a Hach FH950 velocity meter (HACH Company, Loveland, Colorado). Complexity in water velocity and depth were both calculated as a standard deviation. Stream substratum at each point was classified according to the modified Wentworth scale: bedrock, silt/clay (<0.064 mm diameter on b-axis), sand (0.064-2 mm), gravel (2-15 mm), pebble (15-64 mm), cobble (64-256 mm), or boulder (>256 mm) (Cummins 1962). Cover was classified as: boulder, large wood (>10 cm diameter, >4 m in length), small wood, aquatic vegetation (macrophytes), overhanging bank vegetation, undercut bank (>10 cm depth), or absent. Substrate and cover diversity were computed using the Shannon-Wiener index ( $H' = -\sum_{i=1}^n p_i \cdot \log_e p_i$ ) where  $p_i$  represented the proportion of substrate or cover type  $i$  and  $n$  was the number of different types (Legendre and Legendre 2012). Pools were identified as slow water habitat according to Hawkins et al. (1993), and residual pool depth was measured for all pools as maximum pool depth minus water depth at the downstream riffle crest. Woody riparian vegetation height was classified above each transect endpoint at bankfull as: 0.0-0.5 m, 0.5-1.0 m, 1.0-2.0 m, 2.0-4.0 m, 4.0-8.0 m, and >8.0 m (Burton et al. 2011). We computed percent woody riparian vegetation as the percent of transect endpoints with woody vegetation greater than 1 m in height. Streambank stability was classified at each transect endpoint as: fracture, slump, slough, eroding, or absent (Burton et al. 2011). Reach slope was measured as the difference in elevation between reach boundaries divided by reach length and multiplied by 100 (expressed as a percentage). Mean August stream temperature was measured using thermographs (TidbiT v2, Onset Computer Corporation, Bourne, Massachusetts) that recorded temperatures once each hour at 34 locations in the Goose Creek watershed; temperatures at sites without a thermograph were determined using data from the nearest one (with an elevation correction) or two thermographs (distance interpolation). Beaver dams (both active and abandoned) within the sample reach were also counted.

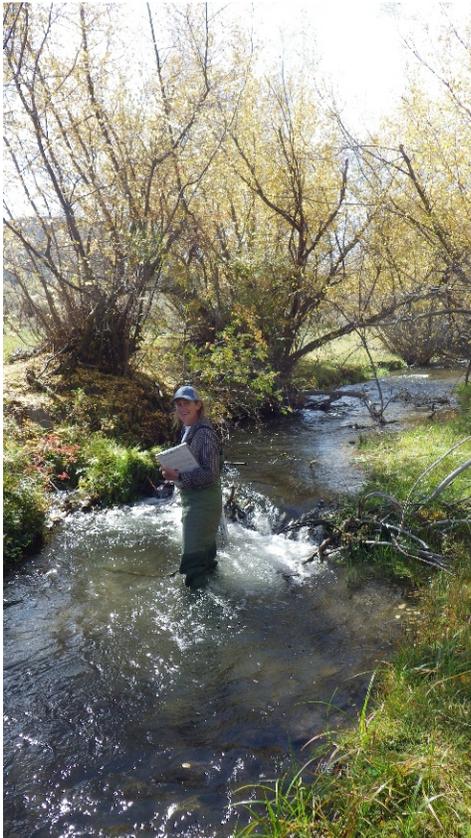
We evaluated the influence of instream and riparian habitat, including habitat complexity and diversity, on Northern Leatherside Chub occurrence using two complementary analyses: multiple logistic regression and path analysis. The multiple logistic regression analysis was done under a

model selection framework. The response variable for occurrence was binary (presence = 1, absence = 0). We evaluated several riparian and instream habitat variables as predictor variables in the multiple logistic regression that are proximal to Northern Leatherside Chub life history needs (as currently understood) or that otherwise influence those proximal habitats. Proximal variables were: SD of velocity as a measure of flow complexity important to drift feeding (Grossman et al. 2002), residual pool depth as a measure pool quality (Wallace and Zaroban 2013), percent overhanging vegetation as a measure of cover and refuge from predation (Wallace and Zaroban 2013), percent aquatic vegetation (macrophytes) as it can provide cover for small fishes, percent pebble substrate that is important for spawning (Billman et al. 2008a), and mean August temperature because of temperatures known influence on growth and thermal tolerance (Billman et al. 2008b). Other covariates were based on factors that influence those proximal habitat variables. A Spearman rank correlation of  $r_s > 0.7$  was used to identify correlated variables potentially causing variance inflation and, if necessary, one of the correlated pair was removed from consideration. Candidate models were constructed using all combinations of variables with a limit of four predictor variables per model to keep variable to sample size ratios near 10:1. Candidate models were fit, and model plausibility was determined using Akaike's Information Criterion for small sample size (Burnham and Anderson 2002). Any candidate models within 4 AIC<sub>c</sub> units of the best model (minimum AIC<sub>c</sub>) were considered plausible. Fit of the most plausible occurrence model was evaluated using a Hosmer-Lemeshow test with 5 bins, and predictive ability was evaluated using a 5-fold cross-validated Area Under the Curve (AUC) of a receiver operating characteristic plot (Hosmer and Lemeshow 2000).

Because we hypothesized that beaver dams influence streamflow complexity and that streamflow complexity influences Northern Leatherside Chub distribution, we did not evaluate beaver dams as a covariate in the logistic regression models because of the potential for variance inflation that would then lead to imprecise parameter estimates perceived to be unimportant. Instead, we explored the influence of beaver dams on streamflow complexity in two ways. First, differences in SD of velocity were compared between sites with and without beaver dams present using a *t*-test ( $\alpha = 0.10$ ). A Bartlett's test was used to assess homogeneity of variances, an assumption for parametric *t*-tests, in SD of velocity between sites with and without beaver dams. Second, we used multiple regression and model selection to evaluate other habitat variables that might influence streamflow complexity. SD of velocity was the response variable, and candidate models were constructed using all combinations of the following predictor variables: number of beaver dams, residual pool depth, percent woody riparian vegetation, percent streambank with sloughing or slumping, channel width:depth ratio, reach slope, percent aquatic vegetation, and percent large wood. As in the logistic regression analysis above, the maximum number of variables allowed in a model was four, and candidate models within 4 AIC<sub>c</sub> units of the best model (minimum AIC<sub>c</sub>) were considered plausible. Model fit of the best model was evaluated using the adjusted  $R^2$ . The sum of Akaike weights ( $w_i$ ) was computed for each model *i* having each predictor variable as a measure of relative variable importance.

In addition to the more traditional generalized linear modeling and model selection approach(es), we used path analysis to evaluate the association of Northern Leatherside Chub occurrence with proximal habitat variables (as above), as well as test for explicit linkages among other instream and riparian habitat features potentially influencing those proximal habitats, including the connection between beaver dams and streamflow complexity. Path analysis is a multivariate

modeling approach that is an extension of multiple regression with the intention of estimating the magnitude and significance of direct and indirect relationships between sets of variables while accounting for their covariance (Shipley 2000). It has an advantage of producing a directed graph (path diagram or model) that shows the direction and magnitude of these interrelationships (Shipley 2000). Therefore, it is a powerful approach for the analysis of complex multivariate relationships and is often thought of as more confirmatory of causal relationships than other statistical modeling approaches (Grace and Pugsek 1998). We developed an initial path diagram that represented a conceptual model of the aforementioned relationships and informed our initial path analysis. It included the hypothesized link between beaver dams and flow complexity, and the link between flow complexity and Northern Leatherside Chub occurrence, while incorporating other linkages between stream morphology, riparian habitat, instream habitat, and Northern Leatherside Chubs and restraining the number of variables due to our sample size ( $n = 41$ ). We used the presence of Northern Leatherside Chub as the response variable (i.e., presence = 1, absence = 0). The initial path model was fit using the lavaan package in R (Rossee 2012; R Core Team 2015); the semPaths function was used to display the directed graph (Epskamp 2015). The directed graph was considered the full model and its fit was evaluated using maximum likelihood. We used standardized coefficients to examine the significance of each pathway using  $P < 0.15$ . We then fit a reduced model with only significant terms. We compared the full and reduced models using AIC.



*Abandoned beaver dam on Trapper Creek (Right), and Northern Leatherside Chub collected from Trapper Creek, 2013 (Above). Credit: D. Dauwalter.*

## Results

The Northern Leatherside Chub was collected at 22% of sites (9 of 41) surveyed by electrofishing in the Goose Creek drainage, where from 1 to 22 individuals were collected in streams from 0.7 to 5.8 m wetted width. When multiple electrofishing passes were conducted, chubs were never collected during a later pass when absent from the first pass. Across all sites, SD of velocity ranged from 0.016 to 0.313 m/s, and Leatherside Chubs only occurred when SD of velocity was 0.09 m/s or greater. Beaver dams were observed at six sites (14.6%), and only one site had more than one dam (two dams total). Northern Leatherside Chub were collected at 2 of 6 sites (33%) with beaver dams present; they were observed at 7 of 35 sites (20%) without a beaver dam.

Table 6. Number of parameters ( $K$ ), Log-likelihood,  $AIC_c$ ,  $\Delta AIC_c$ , and Akaike weights ( $w_i$ ) of the top multiple logistic regression models predicting occurrence of Northern Leatherside Chub at stream sites in the Goose Creek watershed. Only candidate models with  $\Delta AIC_c < 6$  are shown.

Candidate model	$K$	Log-Likelihood	$AIC_c$	$\Delta AIC_c$	$w_i$
SD Velocity + Temperature + WD Ratio + Aq. Vegetation	5	-9.20	30.12	0.00	0.759
Temperature + WD Ratio + Aq. Veg + Overhanging Veg.	5	-11.32	34.35	4.23	0.092
SD Velocity + Temperature + Aquatic Vegetation	4	-12.90	34.91	4.78	0.069
SD Depth + Temperature + WD Ratio + Overhanging Veg.	5	-12.13	35.98	5.86	0.041
SD Velocity + Temperature	3	-14.70	36.04	5.92	0.039

Streamflow complexity was important in explaining the distribution of Northern Leatherside Chub in the Goose Creek watershed. The SD of velocity and SD of depth were the only highly correlated habitat variables ( $r_s = 0.77$ ) and so we restricted candidate multiple logistic regression models to only have one of the two variables but not both. Only one candidate model was plausible given the data (i.e.,  $\Delta AIC_c < 4$ ; Table 6). In addition to containing SD of velocity as a covariate, this top model also contained mean August temperature, channel width:depth ratio, and percent aquatic vegetation as covariates. The Hosmer-Lemeshow test showed the model fit the data ( $\chi^2 = 5.06$ ,  $P = 0.751$ ), and the model showed good 5-fold cross-validated predictive ability (AUC = 0.896). Standardized parameter estimates showed all four habitat variables to have similar influence on Northern Leatherside Chub occurrence, and they showed that chubs were more likely to occur at sites with higher flow complexity (SD of velocity), warmer temperatures, narrower and deeper channels (lower width:depth ratio), and more aquatic vegetation (Table 7; Figure 8). Standardized parameter estimates suggested that an increase in the SD of velocity of 0.07 (i.e., 1 SD) would increase the odds of Northern Leatherside Chub being present at a site by approximately an order of magnitude ( $e^{2.347} = 10.5$ ).

Table 7. Standardized parameter estimates, standard errors, and 90% confidence limits for best multiple logistic regression model predicting occurrence of Northern Leatherside Chub (presence = 1, absence = 0) or multiple regression predicting SD of velocity (m/s) at streams sites in the Goose Creek watershed. Parameter estimates from the multiple regression model are averaged across plausible models with shrinkage (43 candidate models  $\Delta AIC_c < 4$ ).

Parameter	$b_i$	SE( $b_i$ )	Lower 90% CL	Upper 90% CL	$\sum w_i$
Response: N. Leatherside Chub (P/A)					
Intercept	-4.085	1.616	-7.585	-2.050	1.00
% Aquatic Vegetation	2.295	1.016	0.908	4.404	1.00
Mean August Temperature (C)	2.899	1.301	1.190	5.627	1.00
Width:Depth Ratio	-2.552	1.257	-5.085	-0.790	1.00
SD Velocity (m/s)	2.347	1.004	1.007	4.408	1.00
Response: SD of velocity (m/s)					
Intercept	0.000	0.139	-0.234	0.234	1.00
Beaver dams (#)	0.288	0.175	-0.005	0.581	0.88
% Streambank Slough/Slump	-0.178	0.179	-0.476	0.120	0.63
Residual Pool Depth (m)	0.201	0.184	-0.106	0.508	0.68
% Aquatic Vegetation	-0.087	0.153	-0.341	0.167	0.35
Width:Depth Ratio	0.055	0.122	-0.148	0.258	0.26
% Woody Riparian Vegetation	-0.032	0.104	-0.205	0.140	0.16
% Boulder	0.019	0.073	-0.102	0.140	0.13
% Large Wood (>4-m length)	0.014	0.062	-0.090	0.118	0.10
% Slope	-0.005	0.053	-0.095	0.084	0.07

Streamflow complexity was 1.5 times higher on average when beaver dams were present within the stream reach (Figure 9). The SD of velocity was significantly higher when one or more beaver dams were present (mean = 0.20 m/s; 1 SD = 0.09 m/s) than when they were absent (mean = 0.13 m/s; 1 SD = 0.06 m/s). A  $t$ -test showed this difference to be significant ( $t = 2.09$ ,  $df = 39$ ,  $P = 0.043$ ); a Bartlett's test showed variances in SD of velocity to be homogenous (i.e., not significantly different), thus meeting the equal variance assumption of the  $t$ -test ( $K^2 = 1.31$ ,  $df = 1$ ,  $P = 0.253$ ). Multiple regression models also showed beaver dams to influence SD of velocity. There were 43 plausible candidate models with  $\Delta AIC_c < 4$  suggesting much uncertainty in identifying a correct model. However, beaver dam presence was a predictor in 36 of the 43 plausible models (84%), and it was the most important variable explaining SD of velocity as evidenced by the magnitude of standardized parameter estimates and sum of Akaike weights as a measure of relative variable importance ( $\sum w_i = 0.88$ ; Table 2). Other habitat variables influencing SD of velocity most were residual pool depth ( $\sum w_i = 0.68$ ) and percent streambank sloughing and slumping ( $\sum w_i = 0.63$ ). Interestingly, there was little evidence that instream features such as boulders and large wood influenced flow complexity; large wood was only found at 2 of the 44 sites. Model-averaged parameter estimates of all variables had 90% confidence intervals that included zero, with beaver dams just barely so (Table 7). The most plausible model containing beaver dams, percent streambank sloughing and slumping, and residual pool depth had an adjusted  $R^2 = 0.24$ .

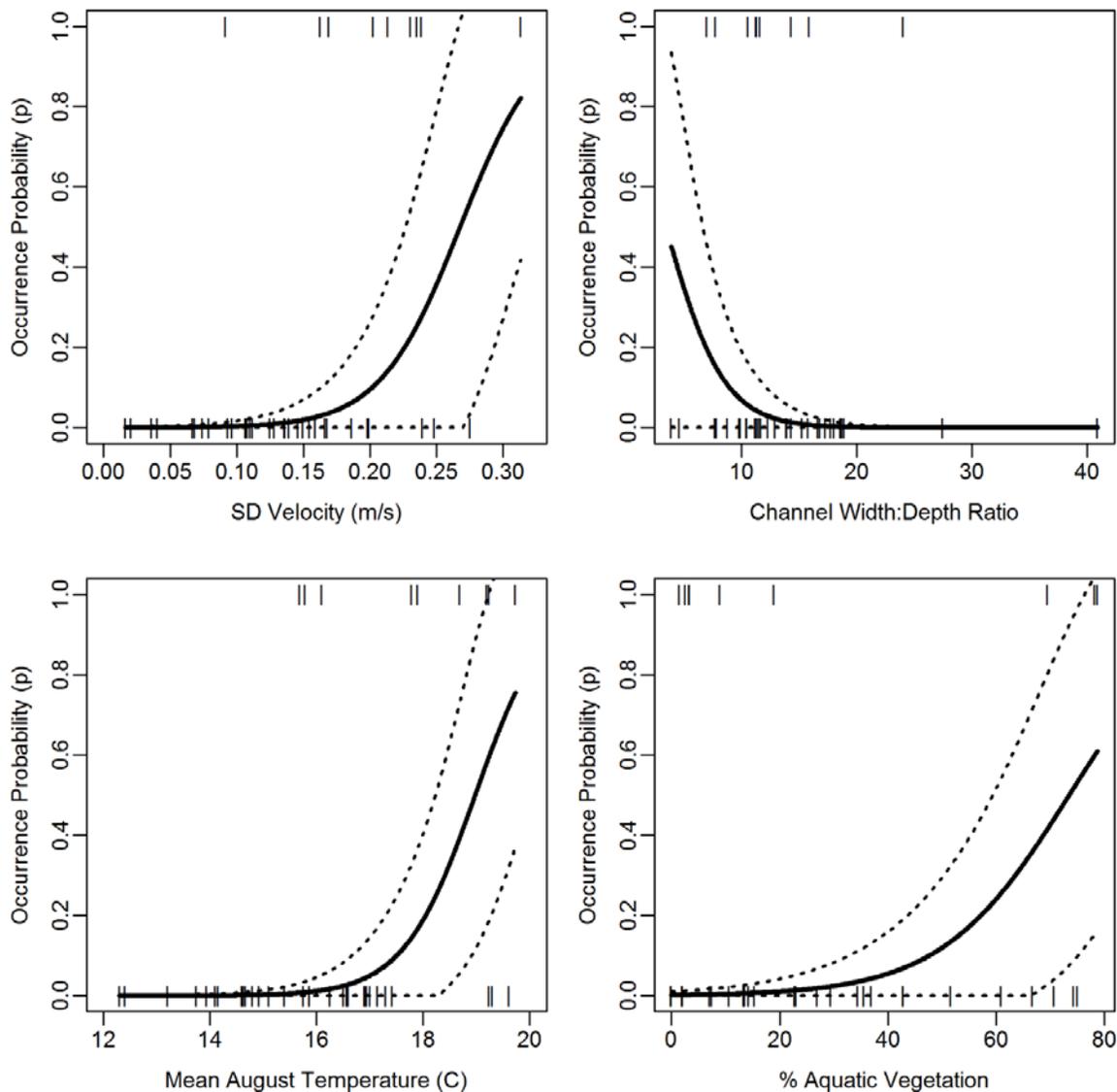


Figure 8. Northern Leatherside Chub occurrence probability as function of SD of velocity, channel width:depth ratio, mean August temperature, and % aquatic vegetation from the only plausible multiple logistic regression model.

Path analysis showed a linkage between Northern Leatherside Chub occurrence, flow complexity, and beaver dams more directly than the generalized linear modeling analyses, in addition to revealing other important relationships (Figure 10). After fitting the full conceptual model, non-significant variables ( $P > 0.15$ ) were removed and a reduced model was fit; the only variable connection retained at  $P > 0.05$  was the beaver dam effect on residual pool depth, which also had a low standardized coefficient suggesting a weak association

Table 8). The reduced model was a more parsimonious model than the initial model ( $\Delta AIC = 1,674$ ) and it showed acceptable fit (comparative fit index = 0.853; root mean squared error approximation = 0.158). The final reduced model showed Northern Leatherside Chub to occur more frequently when flow complexity was higher (Figure 10). In turn, flow complexity was higher when active or abandoned beaver dams were present, but it was also higher when pools were deeper. The reduced model also showed chubs to occur more frequently with higher mean August stream temperatures, with the strength of this association similar to that of flow complexity as revealed by the standardized path coefficients (Table 8). Pools were deeper and temperatures were warmer in lower gradient reaches typical of the lower Goose Creek mainstem – the largest stream we studied (5.4 to 6.2 m wetted width).

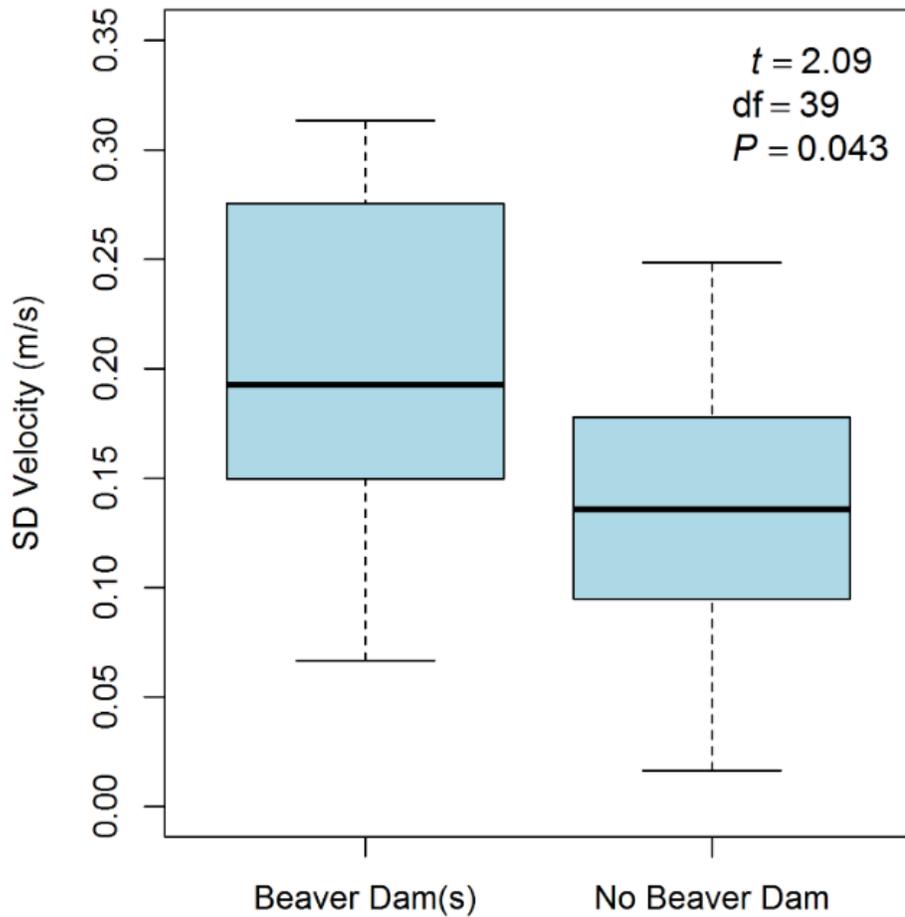


Figure 9. Box plots showing significant differences in streamflow complexity – measured as the SD of velocity – at sample sites with and without active or abandoned beaver dams present.



*Beaver dam on Goose Creek, 2014. Credit: D. Dauwalter*

*Table 8. P-values for predictors of full path model, and standardized parameter estimates (1 SE) from a reduced model refit with parameters  $P < 0.15$  from the full model.*

Response	Predictor	P-value	$b_i$	1 SE( $b_i$ )
Leatherside Chub (P/A)	Residual Pool Depth (m)	0.752		
	% Pebble	0.910		
	SD Velocity (m/s)	0.015	2.097	0.828
	% Aquatic Vegetation	0.192		
	% Overhang. Vegetation	0.712		
SD Velocity (m/s)	Temperature (C)	0.029	0.062	0.030
	Beaver Dams (#)	0.035	0.048	0.034
Residual Pool Depth (m)	Residual Pool Depth (m)	0.024	0.126	0.057
	Width:Depth Ratio	0.934		
% Overhanging Veg.	% Slope	0.005	-0.029	0.011
	Beaver Dams	0.147	0.059	0.062
% Aquatic Veg.	% Woody Riparian	0.177		
% Pebble	Temperature (C)	0.154		
	% Pebble	0.392		
	% Slough/Slump	0.271		
Temperature (C)	% Slope	0.692		
	Beaver Dams (#)	0.003		
	% Slope	0.019	-0.444	0.166
	% Woody Riparian	0.371		
% Slough/Slump	Beaver Dams (#)	0.238		
	% Woody Riparian	0.565		
	% Slope	0.190		
Width:Depth Ratio	% Slough/Slump	0.496		
Beaver Dams (#)	% Slope	0.263		
	% Woody Riparian	0.438		

## Discussion

We found that Northern Leatherside Chub occurred more often at sites with complex streamflows, and complex streamflows were more prevalent when active or abandoned beaver dams were present. Thus, despite small samples sizes (only nine chub occurrences) our data and complimentary analyses support the general field observations made by others and on which our study was founded (Blakney 2012). The working hypothesis for this association is that higher flow complexity increases the chance that flowing and standing water are juxtaposed (aka, current seams) in a way that can be used by drift feeding to maximize consumption of prey with minimum energy expenditure (Grossman et al. 2002). However, additional research is needed to document the behavioral feeding ecology of Northern Leatherside Chub (Hughes and Dill 1990; Rincón et al. 2007). Furthermore, manipulation of streamflow complexity and current seams used for drift feeding (*sensu* Fausch 1993) and quantification of drifting prey could lead to a more mechanistic explanation of why the distribution of Northern Leatherside Chub is linked to streamflow complexity.

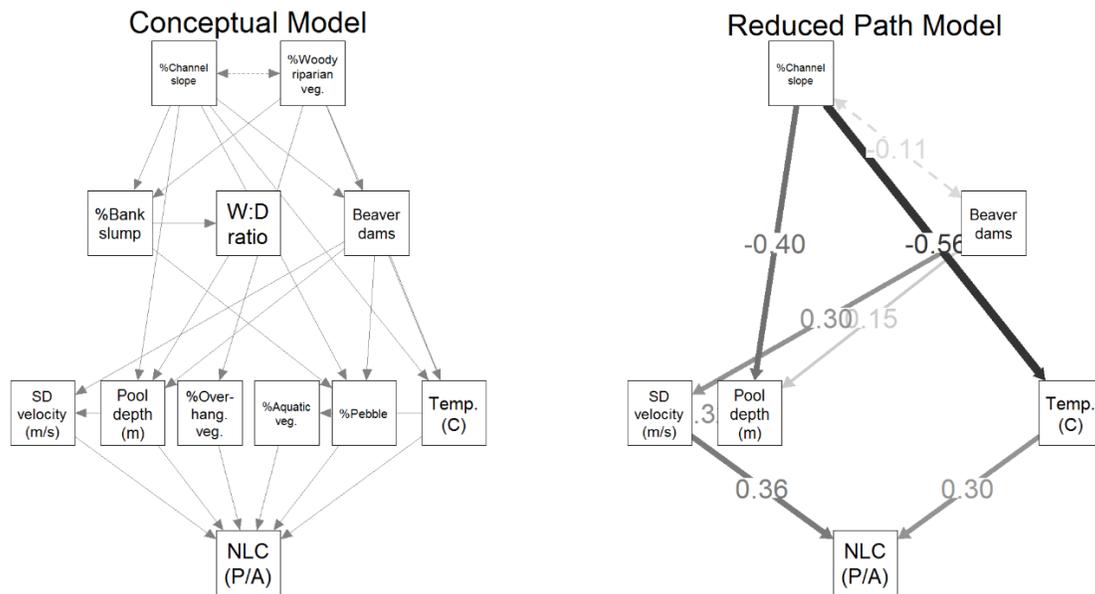


Figure 10. Figure: Left panel shows a conceptual model with hypothesized linkages between measured riparian and instream habitat variables, including beaver dams, and the occurrence of Northern Leatherside Chub, and a reduced path diagram showing only significant ( $P < 0.15$ ) linkages between habitat variables, including beaver dams, and occurrence of Northern Leatherside Chub.

Beaver dams influenced flow complexity directly, but reaches with dams also had deeper pools where flows were also more complex; no other habitat variables influenced flow complexity more in our dataset. While Northern Leatherside Chubs are not likely obligate users of beaver dams themselves, those dams appear to create conditions favorable to chubs when they exist and other stream conditions, such as temperature, are suitable. Other streams where Leatherside Chubs are abundant also have large beaver dam complexes (L. Mabey, Caribou-Targhee National Forest, pers. comm.) and Blakney (2012) observed this pattern while collecting Northern Leatherside Chubs from streams across the species' range, both suggesting the pattern

we observed extends beyond just the Goose Creek subbasin. More research is needed to determine whether the connection between Northern Leatherside Chub and streamflow complexity extends across the species' range, and what role beaver dams play in creating streamflow complexity when it would otherwise have not existed.

Other factors also influenced Northern Leatherside Chub occurrence. Chubs occurred where stream temperatures were warmer. Goose Creek flows into Lower Goose Creek Reservoir before temperatures reach the upper thermal tolerance for Northern Leatherside Chub (UILT = 27 to 30°C; Billman et al. 2008b); however, little is known about whether temperatures in small headwater streams can be too cold for Northern Leatherside Chub populations to be viable. Optimal growth by juvenile Northern Leatherside Chub is achieved at 23°C, but they still grow, albeit slowly, at temperatures as low as 13°C (Billman et al. 2008b). What is unknown is whether slow growth in cold headwater streams prohibits Northern Leatherside Chubs from becoming large enough, or obtaining enough energy reserves, to ensure overwinter survival; larger individuals with more energy reserves commonly have higher overwinter survival in other fish species (Oliver et al. 1979; Biro et al. 2004). In addition, headwater streams may also not be deep enough nor have the streamflow complexity to be suitable for chubs, as suggested by our path model showing flow complexity was higher in reaches with deeper pools (a weak effect independent from beaver dams). Chubs were also less likely to occur in streams with wide, shallow channels that can be symptomatic of impaired streams. Removal of woody riparian vegetation to minimize water loss from evapotranspiration and overgrazing in riparian areas can result in wide, shallow channels with shallow pools (Kauffman and Krueger 1984). This suggests that land management could also be influential in improving habitat complexity for the Northern Leatherside Chub (Swanson et al. 2015).

The positive association between chubs and aquatic vegetation is less clear; the association was revealed by the logistic regression analysis only. Aquatic macrophyte biomass in small streams of the Upper Snake River Basin has been shown to be higher in unshaded streams with stable streamflows (and less streambed substrate mobility) and higher nutrient concentrations (Mebane et al. 2014). Aquatic macrophytes (mostly *Elodea* spp.) in Goose Creek, when abundant, were in large patches where velocities were negligible within patches but high between them. This condition could create current seams used for drift feeding (Sand-Jensen and Mebus 1996), but lack of correlation between SD of velocity and percent aquatic vegetation ( $r_s = -0.16$ ) and weak negative effects in the multiple regression models (Table 7) suggests our data did not capture this phenomenon if present. The association with aquatic vegetation could also reflect an indirect effect of stable streamflow regimes (spring-fed or groundwater dominated) on Northern Leatherside Chub habitat that we did not anticipate or evaluate. Finally, it could also reflect differences in prey densities, as macrophytes stands have been shown to have higher invertebrate taxa richness and abundance than adjacent benthos (Gregg and Rose 1985), although as previously mentioned the feeding ecology of the Northern Leatherside Chub is not well understood.

The distribution of Northern Leatherside Chub is patchy throughout its range (Blakney et al. 2014). Individuals also have a clustered distribution in the streams where they occur (Dauwalter et al. 2014). This patchiness at both the landscape and stream scales makes it difficult to detect Northern Leatherside Chub occurrence and precisely define the species' distribution at a stream

scale during large-scale, spatially distributed fishery surveys using standard stream surveying techniques such as electrofishing in 100 to 200-m stream reaches (e.g., Meyer et al. 2013). Since beaver dams increase streamflow complexity, beaver dams and complexes could be targeted to identify new populations because they are an easily identifiable element of the landscape. Beaver ponds and complexes can be located across large areas using aerial flights (Beck et al. 2010), aerial imagery (Pearl et al. 2015), or habitat suitability modeling (Macfarlane et al. 2017b). Employing stratified or adaptive sampling at beaver dams or complexes may potentially improve the efficiency of large-scale surveys targeted at identifying new populations or precisely defining distributions in some streams (Thompson 2004). Beaver complexes can, however, be difficult to sample efficiently using electrofishing or passive gears (Thompson and Rahel 1996; Hubert et al. 2012), potentially making it difficult to document the presence of Northern Leatherside Chubs when they occur in low abundances. eDNA is a new sampling technique that, if coupled with an efficient sampling design, shows promise as a technique for documenting new populations of Leatherside Chub in difficult to sample beaver complexes (Baldigo et al. 2017).

Interest in using beaver as a stream restoration tool could be a boon for Northern Leatherside Chub. Habitat restoration is one conservation action described in a multi-agency conservation strategy developed for the species (UDWR 2011), and beaver reintroduction and the use of beaver dam analogs are increasingly being used in stream restoration, especially in incised stream channels in the interior western U.S. that typically have low habitat complexity (Marston 1994; Cluer and Thorne 2014; Pollock et al. 2014; Bouwes et al. 2016). In fact, new tools are available to identify the capacity for streams to support beaver (Macfarlane et al. 2017b), and these tools have been combined with information on native fisheries to prioritize stream restoration using beaver dam analogues to optimally benefit rare native species (Macfarlane et al. 2017a). Such an approach could help prioritize streams near extant Northern Leatherside Chub populations for beaver-based restoration with a goal of improving habitat complexity to expand populations or improve their population dynamics.

Although use of beaver for restoration could benefit Northern Leatherside Chub by increasing streamflow complexity in streams where little exists, any such practices should be evaluated in an adaptive management context. Beaver-based restoration projects should be closely monitored so that new data can help refine conceptual models of the Northern Leatherside Chub habitat needs, understand project effectiveness on populations, and inform restoration approaches so they can be adapted to maximize effectiveness, as has been useful for other rare fishes (Roberts et al. 2016). This is especially important since stream restoration is often focused on increasing habitat complexity and heterogeneity under the premise that it will benefit aquatic organisms, including freshwater fishes in peril (Palmer et al. 2010), but most of what is known about how beaver dams and complexes influence fishes is from North American studies on salmonids (Family: Salmonidae) and what little that is known about their influence on non-salmonids is often anecdotal or speculative (Kemp et al. 2012). What is learned through an adaptive management approach, then, may help in the recovery of other species with similar traits that are imperiled globally in areas where beaver are native.

### **Supplemental Resources:**

Webpage: [www.tu.org/beaver-dams-complexity](http://www.tu.org/beaver-dams-complexity)

## CHAPTER 4: NON-NATIVE BROOK TROUT AND HABITAT COMPLEXITY INFLUENCE AGE-0 YELLOWSTONE CUTTHROAT TROUT AT THE RANGE PERIPHERY

### Abstract

Non-native Brook Trout have been widely introduced in the western United States where they have had a disproportional negative impact on native Cutthroat Trout populations. Negative interactions in early life stages have often been hypothesized to result in low juvenile survival of Cutthroat Trout. We used a coupled fish and habitat survey at 41 sites to evaluate the impact of Brook Trout on Yellowstone Cutthroat Trout in the Goose Creek subbasin, which represents the western-most subbasin in the subspecies range periphery, while also considering the role of physical habitat in that interaction. Yellowstone Cutthroat Trout were collected at 8 sites, 4 of which were also occupied by Brook Trout. Cutthroat Trout were never abundant in the presence of Brook Trout. Both physical habitat complexity and age-0 Brook Trout abundance were shown by quantile regression to limit age-1 and older Cutthroat Trout densities. Both age-0 Brook Trout density and diversity of instream cover were shown to negatively affect and positively affect, respectively, the relative condition ( $K_n$ ) of age-0 Cutthroat Trout. Our study suggests that Brook Trout negatively affect Yellowstone Cutthroat Trout at early life stages at the subspecies' range periphery, as shown elsewhere for other subspecies, but it also shows that physical habitat complexity plays a role in regulating abundance and condition. This suggests that protection and restoration of habitat complexity, which is inextricably linked to stream health, should be given consideration as a complementary conservation action as it may help facilitate persistence of Cutthroat Trout populations that have been invaded by Brook Trout.



Yellowstone cutthroat trout collected from Thoroughbred Creek, 2014. Credit: D. Dauwalter

## Introduction

The Brook Trout *Salvelinus fontinalis* is native to eastern North America (Benke 2002). However, beginning in the 1870s Brook Trout have been intentionally introduced into many western U.S. streams for recreational purposes where they have displaced native Cutthroat Trout *Oncorhynchus clarkii* populations and played a key role in extinction of two Cutthroat Trout subspecies (Benke 1992; Dunham et al. 2002; Peterson et al. 2004). Brook Trout invasion success has been linked to connectivity to source populations more so than biotic resistance by the invaded assemblage and habitat quality (Benjamin et al. 2007), but Brook Trout do not relentlessly invade all habitats (Adams et al. 2000; Adams et al. 2002). Repeated introductions from multiple source stocks has also been suggested to have increased genetic diversity, adaptive potential, and invasion success of Brook Trout where they have been introduced (Neville and Bernatchez 2013).

Two species from the same subfamily, such as Salmoninae, would be expected to have some niche overlap and potentially strong interspecific interactions. This is especially true among salmonids that develop size-based dominance hierarchies for drift feeding (Fausch 1988). A majority of authors have suggested that Brook Trout displace rather than replace Cutthroat Trout, especially at warmer temperatures, despite weak data-based evidence (reviewed by Dunham et al. 2002; McGrath and Lewis 2007). Adult Brook Trout often exist in higher densities and exhibit higher production than adult Cutthroat Trout (Benjamin and Baxter 2010; Benjamin and Baxter 2012), but both species show similar use of prey resources and stomach fullness suggesting a lack of interference or exploitative competition for food between sympatric adults (Dunham et al. 2000; Novinger and Rahel 2003; McGrath and Lewis 2007).

Most evidence shows interactions between Cutthroat Trout and Brook Trout to occur in the early stages of life. Juvenile Cutthroat Trout abundance is usually lower in the presence of Brook Trout even when adults are healthy, suggesting poor recruitment or juvenile survival (McGrath and Lewis 2007). Peterson et al. (2004) showed juvenile Colorado River Cutthroat Trout survival but not adult survival to decrease in the presence of Brook Trout, and low age-0 and juvenile survival strongly affects Cutthroat Trout population growth rates (Hilderbrand 2003). Experimental studies have shown juvenile Brook Trout to be more agnostic and occupy more profitable foraging positions than sympatric juvenile Colorado River Cutthroat Trout *O. c. pleuriticus* (De Stato III and Rahel 1994). Outstanding hypotheses on the Cutthroat Trout recruitment bottleneck are age-0 prey resource competition, behavioral interactions that cause age-0 Cutthroat Trout to occupy marginal habitats or emigrate, predation of age-0 Cutthroat Trout in winter, and predation on Cutthroat Trout eggs (McGrath and Lewis 2007).

The Yellowstone Cutthroat Trout *O. c. bouvieri* subspecies is native to the Yellowstone River and Upper Snake River drainages in the northern Rocky Mountains (Benke 2002). Anthropogenic activities have resulted in the subspecies occupying 42% of its historical range, and only 28% of historical range is occupied by genetically unaltered populations (Gresswell 2011). Threats to persistence include habitat degradation from many sources, climate change, and non-native species, including Brook Trout (Meyer et al. 2006; Gresswell 2011). However, Meyer et al. (2014) found that of 14 sites with sympatric populations of Yellowstone Cutthroat Trout and Brook Trout in the 1980s, Yellowstone Cutthroat Trout were only absent from one site

when resurveyed in 2010-11; eight sites still contained sympatric populations, suggesting that there are some situations where the species can co-occur. They noted that stream gradient was positively related to population growth of Brook Trout but negatively related to population growth of Yellowstone Cutthroat Trout, further suggesting habitat conditions mediate the interaction between the two species.

Peripheral populations are those occupying species range peripheries, and are therefore considered to occupy habitat that is less suitable than populations at the core of a species range. For native Cutthroat Trout, peripheral populations often occupy lower elevation, desert streams that have warmer temperatures, are prone to drying, and have otherwise generally harsh conditions relative to the range core. Thus, peripheral populations may be uniquely adapted to those conditions and harbor unique within species genetic and life history diversity and adaptive potential (Haak et al. 2010).

The Goose Creek subbasin contains peripheral populations at the western edge of the range of Yellowstone Cutthroat Trout (Haak et al. 2010). Here it is unclear whether Brook Trout interactions with Cutthroat Trout occur at the early life history stage as has been shown elsewhere for Cutthroat Trout at higher elevation core habitats with a subspecies range (e.g., Peterson et al. 2004). Our goal was to evaluate size-based evidence for negative interactions between the two species. Specifically, our objectives were to quantify any negative association in density and condition between the two species and identify at what life stages any competitive interactions may occur, and then compare how habitat conditions are related to any observed interactions.

## **Methods**

### *Fish Sampling*

We conducted fish and habitat surveys at 41 sites in the Goose Creek watershed from June to October of 2013 to 2015 (Figure 11). Our sites were selected based on the stratified-random design used by Meyer et al. (2006) for Goose Creek, while adding additional sites selected to increase spatial coverage, represent the range of stream conditions, and increase sample size given access constraints to private land. At each site, a stream reach typically 100-m thalweg length (range: 50 – 165m) was isolated with 6.35-mm bar mesh block nets or by impassable barriers such as beaver dams. Fishes were sampled with multiple-pass electrofishing using one Smith-Root LR-24 electrofishers with one or two netters at 37 sites. Larger mainstem Goose Creek sites (n = 4) were sampled with 2 LR-24 backpack electrofishers and three or four netters. At least three electrofishing passes were completed unless no trout were collected on passes 1 or 2 then sampling was terminated. Electrofishing was conducted using pulsed (40 Hz) direct current and 200-450 V. All trout were measured to the nearest mm and weighed to the nearest g.

To estimate the abundance of each trout species, first the abundance of all trout was estimated using the Zippin removal method (Zippin 1958) as implemented in the FSA package in Program R (R Core Team 2015; Ogle 2017); to reduce potential bias due to size-dependent capture efficiency abundances were estimated separately for trout <100 mm TL and  $\geq$ 100 mm TL. If more than one trout species was present, then species abundance in each size class was estimated

based on the proportion of catch representing each species in each size class. Trout densities were expressed as the number of individuals per 100-m<sup>2</sup>, which was computed by dividing the abundance estimate by the wetted reach area multiplied by 100. Wetted reach areas were computed by multiplying reach length by mean wetted width as measured at 10 transects spaced 10-m apart (see below).

### *Habitat Surveys*

Habitat surveys were completed immediately after electrofishing surveys at each site. A transect was established every 10 m along each site perpendicular to the channel at bankfull height. Channel depth, water depth, water velocity, stream substrate, and cover type were measured at 10 equally spaced points along each transect. Water velocity was measured at 0.6 of water depth using a Hach FH950 velocity meter (HACH Company, Loveland, Colorado). Complexity in water velocity and depth were both calculated as a standard deviation. Stream substratum at each point was classified according to the modified Wentworth scale: bedrock, silt/clay (<0.064 mm diameter on b-axis), sand (0.064-2 mm), gravel (2-15 mm), pebble (15-64 mm), cobble (64-256 mm), or boulder (>256 mm) (Cummins 1962). Cover was classified as: boulder, large wood (>10 cm diameter, >4 m in length), small wood, aquatic vegetation (macrophytes), overhanging bank vegetation, undercut bank (>10 cm depth), or absent. Substrate and cover diversity were computed using the Shannon-Wiener index ( $H' = -\sum_{i=1}^n p_i \cdot \log_e p_i$ ) where  $p_i$  represented the proportion of substrate or cover type  $i$  and  $n$  was the number of different types (Legendre and Legendre 2012). Streambank stability was classified at each transect endpoint as: fracture, slump, slough, eroding, or absent (Burton et al. 2011). Reach slope was measured as the difference in elevation between reach boundaries divided by reach length and multiplied by 100 (expressed as a percentage). Mean August stream temperature was measured using thermographs (TidbiT v2, Onset Computer Corporation, Bourne, Massachusetts) that recorded temperatures once each hour at 34 locations in the Goose Creek watershed; temperatures at sites without a thermograph were determined using data from the nearest one (with an elevation correction) or two thermographs (distance interpolation).

### *Cutthroat Trout Density*

We evaluated the influence of Brook Trout and habitat conditions on Yellowstone Cutthroat Trout densities using quantile regression. Quantile regression models the quantiles of a response variable distribution, as opposed to the mean response typically modeled in regression analysis, and can be more informative for understanding ecological processes. When upper quantiles are modeled they effectively represent the potential maximum response to one or more variables at different levels of those variables, and thus can be viewed as an evaluation of limiting factors (Cade and Noon 2003). Single-variable quantile regression models were fit, with age 1 and older (>100-mm TL) Cutthroat Trout density as the response variable (# / 100 m<sup>2</sup>). We focused on age 1 and older Cutthroat Trout because the negative interaction with age-0 Cutthroat Trout would not manifest itself in densities until one year later in our dataset. Predictors were comprised of physical habitat and Brook Trout variables: CV of Velocity, SD Depth, Cover Diversity, and Substrate Diversity as measures of habitat complexity; Percent Fines, Channel Width:Depth Ratio, Percent Streambank Slough/Slump as measures of stream condition; Mean August Temperature; and age-0 (<100-mm) Brook Trout. All quantile regression models were fit to the

90<sup>th</sup> percentile (0.9 quantile) of the response variable, which was the log<sub>e</sub>-transformed Age 1 and older Cutthroat Trout density (# / 100 m<sup>2</sup>). Models were fit using the quantreg package (Koenker 2013) in Program R (R Core Team 2015), parameter standard errors were estimated using the xy-bootstrap method, and models were evaluated using the AIC<sub>c</sub> statistic developed for quantile regression: rqAIC<sub>c</sub> (Cade et al. 2005). The model with the minimum rqAIC<sub>c</sub> was considered the most plausible, and models within <4 rqAIC<sub>c</sub> units were considered plausible as well. If needed, model averaging was done using shrinkage and Akaike weights. Model fit was evaluated using quantile coefficient of determination (R<sup>1</sup>) that represents the proportional reduction in objective function by a model when compared to an intercept-only model (Cade et al. 2005).

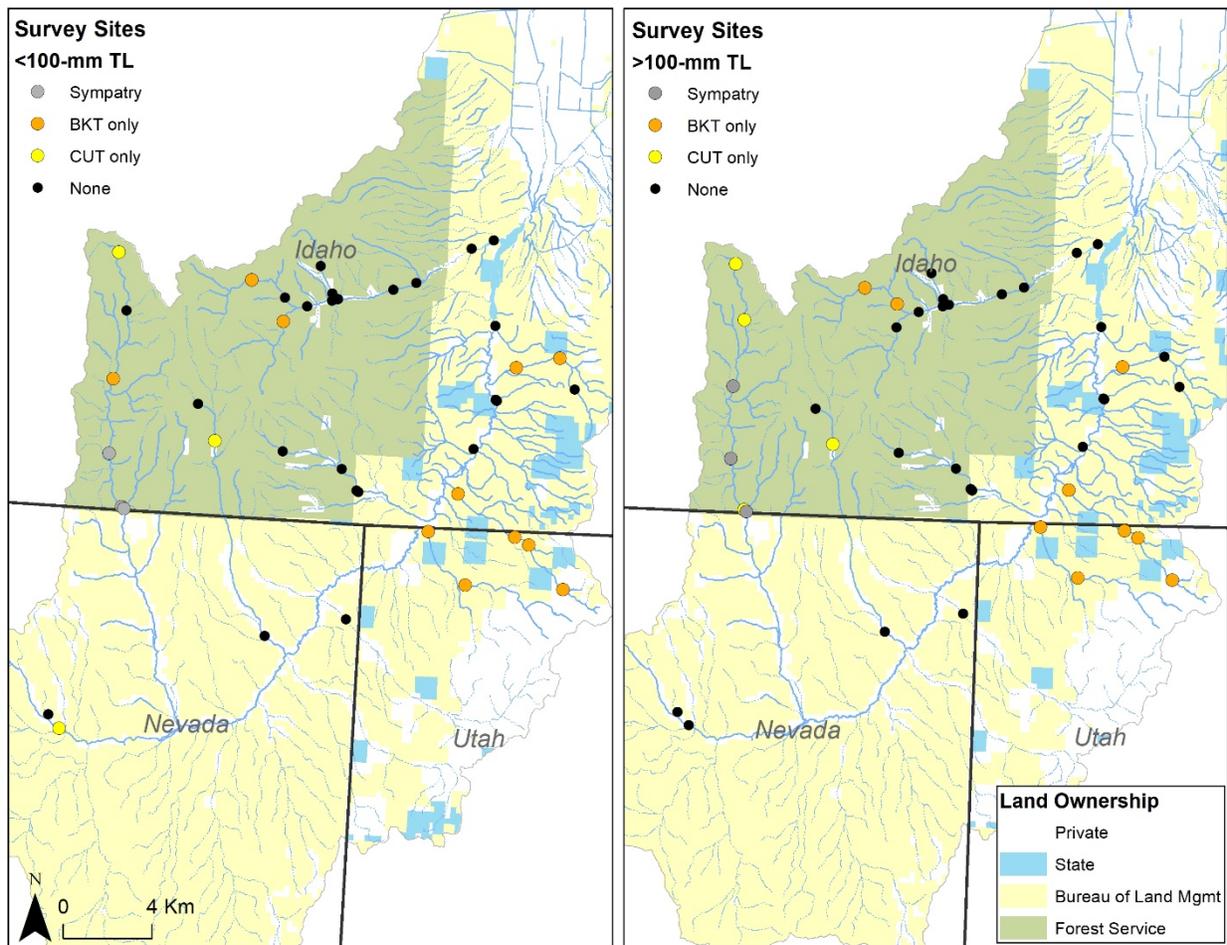


Figure 11. Distribution of surveys sites with sympatric Brook Trout and Cutthroat Trout, Brook Trout (BKT) only, Cutthroat Trout (CUT) only, or neither species within two size classes (<100-mm and >100-mm TL). Sites were surveyed from 2013 to 2015.

### *Cutthroat Trout Body Condition*

We evaluated the effect of Brook Trout on age-0 Cutthroat Trout body condition using two complementary analyses. First, we used a t-test to compare the relative condition of Cutthroat Trout at sites with Brook Trout and at sites without Brook Trout. Relative condition ( $K_n$ ) was computed for each individual fish as the observed weight ( $W$ ) divided by the length-specific mean weight of all Cutthroat Trout sampled across all sites ( $W'$ ) as:  $K_n = 100 * (W'/W)$ .  $W'$  was predicted from a weight-length equation developed from all cutthroat sampled, where  $W' = aL^b$ , where  $L$  is total length and  $a$  and  $b$  are constants (Neumann et al. 2012). The constants were estimated by regressing  $\log_{10}(W)$  versus  $\log_{10}(L)$ . Separate t-tests were computed for Cutthroat Trout  $<100$  mm TL and  $\geq 100$  mm TL, as well as both size classes combined. Significance of each t-test was evaluated at  $\alpha = 0.05$ .

We also estimated the effect of Brook Trout presence on Yellowstone Cutthroat Trout weight using an Analysis of Covariance (ANCOVA) that was an extension of a length:weight curve (explained above). For the ANCOVA,  $\log_{10}$  Cutthroat Trout weight was the response variable,  $\log_{10}$  total length was the covariate, and the presence of Brook Trout at a site (present = 1; absent = 0) was the main effect. Because we hypothesized Brook Trout would impact condition of age-0 Cutthroat Trout only, we included an interaction between the main effect and covariate that, if significant, would show only weight of the smallest Cutthroat Trout to be influenced by Brook Trout presence. The significance of the interaction was evaluated at  $\alpha = 0.05$ .

Since results suggested a Brook Trout influence on body condition of age-0 Cutthroat Trout only (see below). Thus, we used multiple linear regression in a model selection framework to evaluate whether Brook Trout had a stronger effect on age-0 Cutthroat Trout than did physical habitat condition. To keep sample size to variable ratios near 10:1 we limited candidate models to having 3 or fewer predictors: SD Velocity, Cover Diversity, and Substrate Diversity as measures of habitat complexity; Channel Width:Depth Ratio and Percent Fines as measures of habitat degradation; Temperature; and Brook Trout. Akaike's information criterion adjusted for small sample size ( $AIC_c$ ) was used to evaluate the plausibility of all candidate models; the model with the lowest  $AIC_c$  value was considered the most plausible. Akaike weights ( $w_i$ ) were computed as a measure of the probability that the model is the correct model for models within 4  $AIC_c$  units ( $\Delta AIC_c$ ) of the best model (i.e., plausible models; Burnham and Anderson 2002). If multiple models were plausible ( $\Delta AIC_c < 4$ ) then model averaging with shrinkage (Lukacs et al. 2010) and Akaike weights was performed to estimate parameters and standard errors based on model selection and parameter uncertainty. For this analysis, we only used data from sites where age-0 Cutthroat Trout were present, and regression models were fit using the `lm` function in Program R (R Core Team 2015).

## **Results**

Of the 41 sites sampled, Yellowstone Cutthroat Trout were collected at eight sites, Brook Trout were collected at 15 sites, and the two species were sympatric at four sites (Figure 11). Sympatry occurred only at sites on the mainstem Goose Creek in the upper watershed upstream

of the Nevada state line. Rainbow trout were the only other salmonid collected, all from sites near Goose Creek Reservoir.

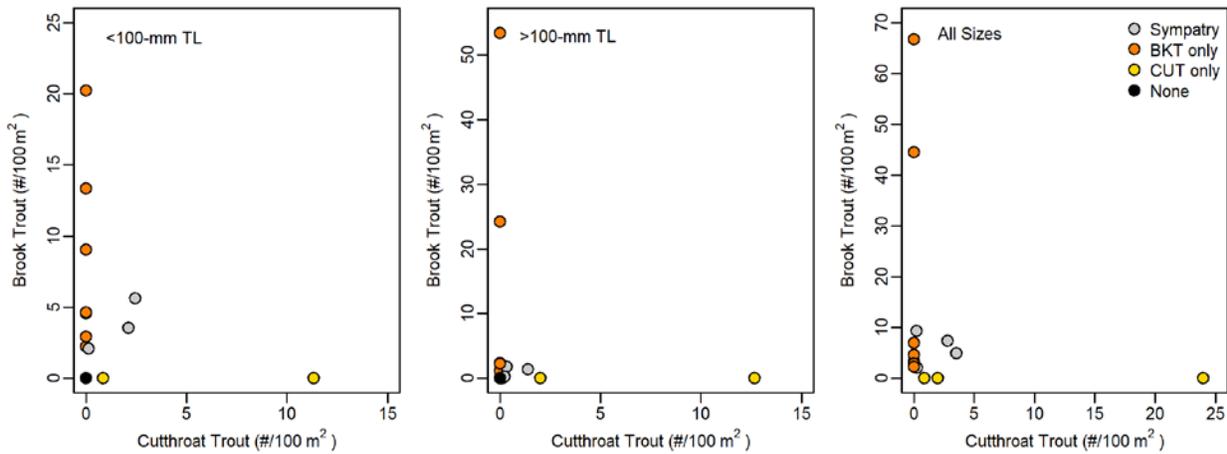


Figure 12. Plots showing the density of Yellowstone Cutthroat Trout versus the density of Brook Trout less than 100 mm TL, greater than 100 mm TL, and all size classes.

Yellowstone Cutthroat Trout densities, when the species was present, ranged from 0.15 to 11.33 fish / 100-m<sup>2</sup> for individuals <100-mm TL, and from 0.15 to 12.68 fish / 100-m<sup>2</sup> for individuals ≥100-mm TL. Despite some observed sympatry, sympatry only occurred at low abundances of both species, for both small (<100-mm) and large (≥100-mm) individuals (Figure 12).

### Cutthroat Trout Density

Age-0 Brook Trout densities and variation in water depth (SD Depth) best explained the upper limit of age-1 and older Cutthroat Trout densities and did so equally. The model with SD of Water Depth as a predictor was most plausible, but that the model with age-0 Brook Trout density just as plausible (Table 9). No other candidate models were within eight rqAIC units of the best model. Quantile coefficients of determination showed both SD of Depth and Age-0 Brook Trout density to predict the upper quantile of age-1 and older Cutthroat Trout density reasonable well ( $R^2_{SD\ Depth} = 0.42$ ;  $R^2_{Brook\ Trout} = 0.41$ ; Table 9). SD of Depth has a positive effect on the upper limits of Age 1 and older Cutthroat Trout densities, and age-0 Brook Trout has a negative effect (Figure 13A and B). Mean August temperature and streamflow complexity (CV Velocity) were the next best models, but had little support (Table 9; Figure 13C and D).

Table 9. Parameter estimates, model fit, and model selection statistics for candidate quantile regression models predicting Age 1 and older Cutthroat Trout density (#/100 m<sup>2</sup>) as a function of physical habitat variables or age-0 Brook Trout density (log(#/100 m<sup>2</sup>))

Variable	$b_i$	Lower 95%	Upper 95%	LogLik	rqAIC	$\Delta$ rqAIC	$w_i$	$R^1$
SD Depth (m)	30.44	-10.954	+inf	-26.13	59.86	0.00	0.621	0.42
Ln Age-0 BKT (#/100m <sup>2</sup> )	-0.522	-0.577	+inf	-26.66	60.92	1.06	0.366	0.41
Temperature (C)	-0.374	-0.963	-0.151	-30.47	68.54	8.68	0.008	0.27
CV Velocity (%)	0.026	-0.021	0.062	-30.9	69.40	9.54	0.005	0.26

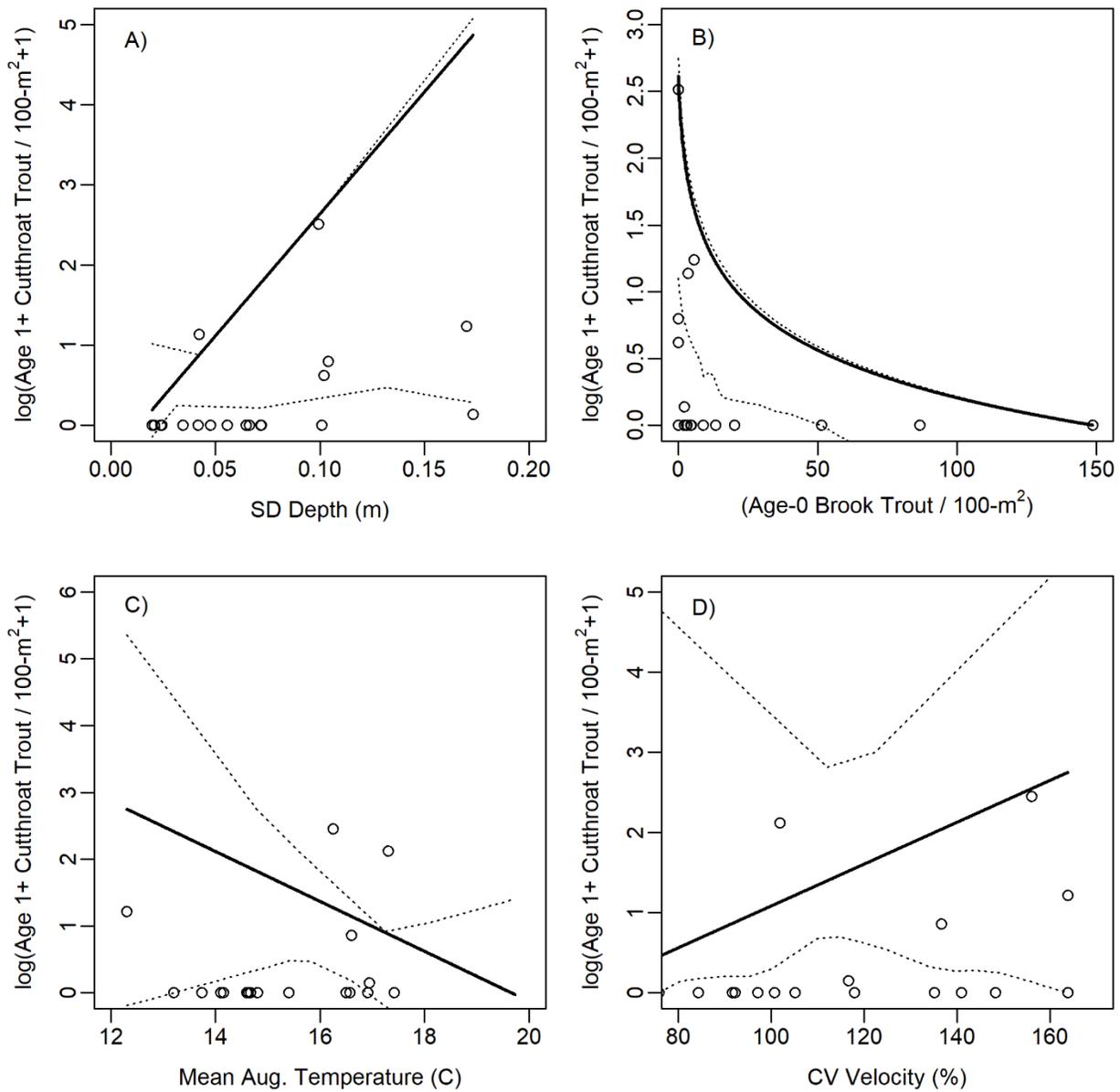


Figure 13. Plots from quantile regression ( $\tau = 0.9$ ) models predicting age-1 and older Cutthroat Trout densities as a function of physical habitat attributes and age-0 Brook Trout density. Solid lines are best fit of the 0.9 quantile, and dashed lines are bootstrapped 90% confidence intervals. Model selection statistics suggested that only SD of depth and Brook Trout density were plausible models (panels A and B).

### Cutthroat Trout Body Condition

One hundred thirty-three Yellowstone Cutthroat Trout were collected across the eight sites in which they occurred; 55 individuals were  $<100$ -mm TL, and 78 were  $\geq 100$ -mm TL. The length:weight equation was:  $W = 0.000005 \cdot L^{3.1337}$ ,  $df = 130$ ,  $r^2 = 0.991$ . Yellowstone Cutthroat

Trout relative condition,  $K_n$ , ranged from 38 to 138 (mean = 101.0, SD =12.6), and was significantly higher in the absence of Brook Trout for individuals <100-mm TL but was not different for individuals  $\geq$ 100-mm TL (Figure 14).

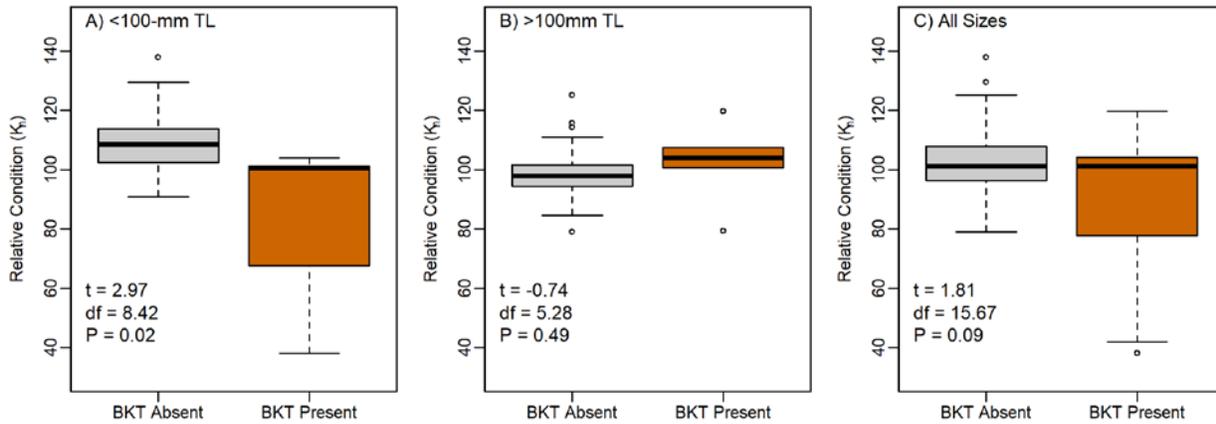


Figure 14. Boxplots of relative condition ( $K_n$ ) of Yellowstone Cutthroat Trout at sites with Brook Trout present versus Brook Trout absent.

Only the smallest Yellowstone Cutthroat Trout were shown to be of lower weight per unit length as shown by the significant Brook Trout x total length interaction term ( $t = 9.22$ ;  $P < 0.001$ ) in the ANCOVA. Confidence intervals on the length:weight plot for Cutthroat Trout with Brook Trout present versus absent showed 95% confidence intervals on the slope to not overlap for the smallest individuals (Figure 15).

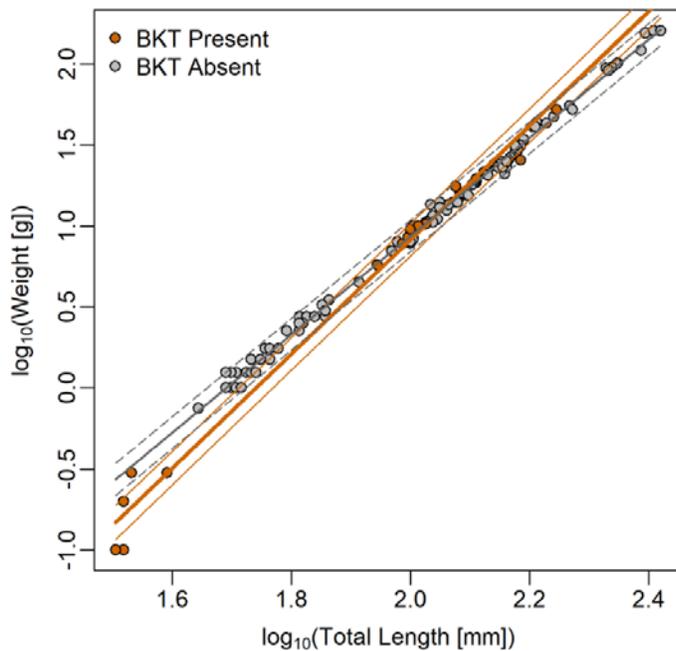


Figure 15. Cutthroat Trout length-weight plots at sites with and without Brook Trout. Thick lines represent best fit lines, and thinner lines represent 95% prediction intervals.

Multiple linear regression also revealed that age-0 Brook Trout density negatively influenced the relative condition of age-0 Cutthroat Trout, but it also showed that the diversity of instream cover (Cover Diversity) had a positive effect on relative condition. Model selection statistics showed multiple candidate models to be the most plausible, but Brook Trout was in all top models except one and Cover Diversity in all but two ( $\Delta AICc < 4$ ; Table 10). Model averaged parameter estimates showed that age-0 Brook Trout to have a negative effect and cover diversity a positive effect on the condition of age-0 Cutthroat Trout (Table 11).

Table 10. Model selection statistics for plausible ( $\Delta AICc < 4$ ) candidate linear regression models predicting relative condition ( $K_n$ ) as a function of physical habitat variables and age-0 Brook Trout density ( $\log(\#/100\text{ m}^2)$ ).

Model	df	logLik	AICc	$\Delta AICc$	$w_i$	$R^2$
Cover Diversity + Age-0 Brook Trout	4	38.48	-68.15	0.00	0.261	0.51
Cover Diversity + Age-0 Brook Trout + Temperature	5	39.51	-67.76	0.39	0.215	0.52
Cover Diversity + Age-0 Brook Trout + SD Velocity	5	38.94	-66.64	1.51	0.123	0.51
Cover Diversity + Age-0 Brook Trout + Pct Fines	5	38.80	-66.35	1.80	0.106	0.51
Age-0 Brook Trout + SD Velocity	5	38.64	-66.03	2.12	0.091	0.51
Cover Diversity + Age-0 Brook Trout + Width:Depth Ratio	5	38.51	-65.78	2.37	0.080	0.51
Cover Diversity + Age-0 Brook Trout + Substrate Diversity	5	38.50	-65.76	2.39	0.079	0.51
Pct Fines + SD Velocity + Width:Depth Ratio	5	37.97	-64.70	3.46	0.046	0.49

Table 11. Model averaged parameter estimates for models predicting relative condition ( $K_n$ ) of age-0 Cutthroat Trout as a function of physical habitat variables and age-0 Brook Trout density ( $\log(\#/100\text{ m}^2)$ ).

Variable	$b_i$	1 SE	Importance
Age-0 Brook Trout	-0.270	0.047	0.86
Cover Diversity	0.778	0.409	0.95
Temperature	0.056	0.042	0.31
SD Velocity (m/s)	0.173	3.090	0.26
Pct Fines	0.003	0.008	0.15
Width:Depth Ratio	-0.010	0.013	0.13
Substrate Diversity	0.023	0.126	0.08

## Discussion

We found that Yellowstone Cutthroat Trout rarely occurred, and were never abundant, in the presence of Brook Trout in the Goose Creek subbasin. This negative interaction appeared to be strongest at the age-0 life stage as shown in other studies despite our study being conducted at the species' range periphery. This suggests that the negative interactions among age-0 Cutthroat Trout and age-0 Brook Trout transcend any differences between the core of a subspecies range and the range periphery (Haak et al. 2010). Our study also suggested, however, that habitat does also play a role in the distribution and well-being of age-0 Cutthroat Trout.

The lower body condition of age-0 Cutthroat Trout in the presence of Brook Trout suggests a negative interaction at that life stage, but our study does not elucidate the nature of that interaction. Dunham et al. (2002) reviewed studies evaluating Cutthroat Trout – Brook Trout interactions and found that most of them concluded that interspecific competition to be the mechanism driving negative impacts despite a lack of clear data-driven evidence. Peterson et al. (2004) found that juvenile Colorado River Cutthroat Trout had lower survival when juvenile

Brook Trout were more abundant, and McGrath and Lewis (2007) hypothesized the mechanisms of low juvenile Cutthroat Trout survival in the presence of Brook Trout result from competition for food resources, behavioral aggression causing juvenile Cutthroat Trout to occupy marginal habitats or emigrate, predation of age-0 Cutthroat Trout during winter, and/or predation of Cutthroat Trout eggs. They suggested studies on feeding, growth, and lipid levels of age-0 Cutthroat Trout would provide insights into the mechanisms of the negative impacts of Brook Trout. We agree that future studies with such focus would help reveal the mechanisms underlying the negative impact of Brook Trout on Cutthroat Trout in the Goose Creek subbasin and elsewhere on the species' range periphery.

Body condition of adults was not affected by the presence of Brook Trout, which suggests little direct or indirect interaction between adults of the two species. Body condition indices would be expected to be lower for adult Cutthroat Trout if Brook Trout competed with them for limited prey resources or Cutthroat Trout expended more energy defending territories. Lower body condition would then negatively affect fecundity and spawning success, which appears to not occur. The two species also spawn at different times of the year, so competition for spawning habitat also does not occur (Benke 2002). Several studies have shown low levels of predation by adult Brook Trout but concluded that it is likely has a minor impact on Cutthroat Trout populations (Dunham et al. 2000; Dunham et al. 2002; McGrath and Lewis 2007). In contrast, Brook Trout can mature 1-year earlier than Cutthroat Trout (Kennedy et al. 2003), and that is one hypothesis as to why Brook Trout exhibit greater production than and eventually replace Cutthroat Trout when in sympatry (Benjamin and Baxter 2012).

We expected to observe a negative impact of Brook Trout on Cutthroat Trout in our study. However, quantile regression model results suggest that age-1 and older Cutthroat Trout abundance is also limited by habitat complexity. Given that age-0 Brook Trout reduce the survival of age-0 Cutthroat Trout, it is not surprising that we observed them to limit the abundance of older fish that reflects reduced survival in the first year of life. What was surprising, is that variation in water depth also appears to limit cutthroat abundance. Variation in water depth as we measured it reflects both lateral and longitudinal variation. Lateral variation is expected to increase in deeper, larger streams as well as in streams with lower width-to-depth ratios. Longitudinal variation is expected to be higher in streams with well-defined riffle pool morphology. Both width:depth ratio and riffle-pool structure reflect stream conditions. Streams in poor condition from land use (e.g., agriculture, livestock production) often have wide, shallow stream channels that lack physical habitat complexity (Kauffman and Krueger 1984; Walrath et al. 2016). Thus, degraded streams with Brook Trout have little capacity to support Cutthroat Trout.

Habitat complexity, in addition to Brook Trout, also appears to play a role in the body condition of age-0 Cutthroat Trout. The diversity of instream cover was positively associated with age-0 Cutthroat Trout body condition. Habitat complexity is thought to provide more niche space and increase the likelihood that all required habitats exist in a smaller unit area for Cutthroat Trout to meet all life history requirements and, thus, facilitate their persistence (Horan et al. 2000). Our data suggest that complexity in cover of various forms improves conditions for age-0 Cutthroat Trout although the mechanisms remain elusive. Diverse forms of cover may create refuge from predators or provide more space for coexistence with congeners such as Brook Trout. Thus, with

Brook Trout are present, diverse forms of cover may allow sufficient habitat availability to limit behavioral interactions that have been suggested to cause stress, low survival, and/or emigration of age-0 Cutthroat Trout and allow the two species to coexist (McGrath and Lewis 2007). For example, Billman et al. (2013) showed that habitat enhancement increased habitat complexity in the Provo River, Utah that then allowed native fishes to coexist with non-native and piscivorous Brown Trout *Salmo trutta*. The role of diverse instream cover in facilitating coexistence of Cutthroat Trout and Brook Trout needs further research.

Our study suggests that Brook Trout negatively influence the distribution, abundance, and condition of Yellowstone Cutthroat Trout at early life stages in the periphery the subspecies' range, suggesting a similar interaction that has been repeatedly observed in core habitats of other Cutthroat Trout subspecies. Removal of Brook Trout and intentional isolation of populations above barriers is a common management action implemented for conservation purposes (Thompson and Rahel 1996; Dunham et al. 2002). However, the viability of Cutthroat Trout populations is often low when they are isolated from non-native invaders in small headwater streams (Peterson et al. 2008; Roberts et al. 2013). Our study suggests that promoting healthy stream conditions and physical habitat complexity through stream protection or restoration may help facilitate persistence of Cutthroat Trout populations in the presence of Brook Trout in peripheral habitats, although this pattern should be evaluated more broadly. Habitat complexity may explain why some populations of Cutthroat Trout and Brook Trout have coexisted for decades (Meyer et al. 2014), and should receive increased consideration as an alternative, or at least complementary, action for conservation of Cutthroat Trout populations invaded by Brook Trout in the western United States.

### Supplemental Resources:

Webpage: [www.tu.org/bkt-yct-interactions](http://www.tu.org/bkt-yct-interactions)



Brook trout collected from Thoroughbred Creek, 2014. Credit: D. Dauwalter

## CHAPTER 5: EFFECTIVENESS OF RIPARIAN EXCLOSURES FOR RESTORING RIPARIAN VEGETATION, INSTREAM HABITAT, AND AQUATIC BIOTA

### Abstract

Improper riparian grazing can alter riparian vegetation and reduce streambank stability, therefore negatively impacting aquatic habitat and biota. We evaluated differences in riparian and instream habitat, benthic macroinvertebrates, and fish inside versus outside seven riparian exclosures in Idaho, U.S.A. The Normalized Difference Vegetation Index (NDVI) from Landsat imagery (1985 to 2015) showed significant increases in riparian vegetation productivity after some but not all exclosures were constructed. Field data showed woody riparian vegetation to be more abundant, streambanks less altered, and stream channels narrower inside versus outside exclosures as we expected. However, instream habitat, benthic macroinvertebrate indices, fish species richness, and adult trout density showed no response. Higher age-0 trout densities were the only aquatic organism response observed. Broader watershed-scale factors likely limited strong and consistent responses by aquatic biota to exclosures. We conclude that riparian exclosures have localized effects on stream systems and should be one of several grazing management tools implemented simultaneously to promote and improve stream health at larger scales relevant to most aquatic biota in the presence of livestock grazing.



*Water gap in  
Little Birch  
Creek exclosure  
fence. Credit:  
D. Dauwalter.*

## Introduction

Livestock grazing is a predominant land use in North America, and increasing demand for livestock products associated with human population growth will increase grazing needs and competition for natural resources in the future (Thornton 2010). This is especially true in the western United States where grazing is widespread, and when done improperly can impact biodiversity and ecosystem function (Fleischner 1994). Livestock often concentrate in riparian areas due to water and forage availability, and this concentration can impact aquatic ecosystems (Kauffman and Krueger 1984). Improper riparian grazing has been shown to reduce riparian vegetation and de-stabilize streambanks, leading to wide, shallow, and incised stream channels with low physical habitat complexity and poor water quality (Kauffman and Krueger 1984; Agouridis et al. 2005; Walrath et al. 2016). These degraded stream conditions typically reduce fish populations and aquatic community diversity (Kauffman and Krueger 1984; Armour et al. 1991).

There are a variety of rangeland management techniques used to maintain riparian plant communities, promote streambank stability, and minimize stream degradation (Swanson et al. 2015). Effective grazing management plans balance grazing (including regrazing) with plant growth. This is done by scheduling grazing with plant recovery through strategic use of timing, duration, and intensity (number of livestock) of grazing. Monitoring plant usage can be done concurrently to achieve management goals and trigger grazing regime changes through adaptive management (DelCurto et al. 2005). Off-stream water, employing riders to encourage livestock movement, and feeding, planting preferred forage, and providing shade away from streams are all techniques used to change the spatial distribution of livestock and reduce their concentration in riparian areas (DelCurto et al. 2005; Swanson et al. 2015). When successful, rangeland management can facilitate adequate riparian function in addition to livestock production and other uses. When unsuccessful, over-grazed riparian vegetation can take over 25 years to recover (Nusslé et al. 2017).

Riparian exclosures are one management tool used to minimize livestock use of riparian areas (Sarr 2002). Riparian exclosures commonly result in increased woody and herbaceous vegetation growth, greater bank stability, narrower and deeper stream channels, and improved fish habitat (McDowell and Magilligan 1997). These changes to physical habitat along with changes in prey abundance, in turn, have been linked to increased abundance of juvenile salmonids in northeast Oregon, U.S.A streams (Bayley and Li 2008). Others have shown terrestrial invertebrate inputs to streams not grazed by livestock to be greater than those in intensively grazed riparian pastures (Saunders and Fausch 2012). Exclosure effectiveness has led to proposals to exclose 20% of public land parcels (both riparian areas and uplands) in the United States to provide a benchmark for understanding grazing impacts and provide a refuge to plants and animals sensitive to livestock grazing (Bock et al. 1993). While many exclosures, including riparian exclosures, already exist, many are small and collectively they represent a very small fraction of all lands. Unfortunately, not all exclosures have the desired influence on riparian vegetation and channel morphology (McDowell and Magilligan 1997). For example, small exclosure size has been cited as one reason why juvenile fishes may respond to exclosures whereas large adults that require larger expanses of habitat beyond that exclosed do not respond (Bayley and Li 2008).

Our goal was to evaluate the response of woody riparian vegetation, instream habitat, and aquatic biota to riparian grazing exclosures in south-central Idaho, U.S.A. Our specific objectives were to: 1) evaluate riparian vegetation response to exclosures over time using remote-sensing data; 2) compare field-measured riparian vegetation, instream habitat, benthic macro-invertebrate, and fish metrics inside and outside of riparian exclosures; 3) evaluate if any observed comparative differences in metrics (effect sizes) were associated with exclosure size; and 4) assess the general health of streams in the allotment to place any comparative differences in the context of overall stream health. As discussed below, our sampling design allowed for inference at both the allotment and individual exclosure scales, and our study will be useful for understanding the effectiveness of riparian grazing exclosures in the context of broader grazing management approaches and stream health.

## **Methods:**

### *Study Area*

The Goose Creek Group Allotment is located in Cassia County, Idaho, U.S.A. in the Northern Basin and Range level III ecoregion (Omernik 1987). It is bounded by the Utah state line to the south and private land along the Goose Creek mainstem to the west. Major streams in the allotment are Goose Creek mainstem (1,500-m elevation), and the major tributaries Cold Creek, Emery Creek, and Little Birch Creek (up to 2,000-m elevation; Figure 16). All streams flow through sagebrush-steppe in lower elevations, juniper (*Juniperus* spp.) woodlands in mid-elevations, and mixed forests comprised of Douglas fir and aspen (*Pseudotsuga menziesii* and *Populus tremuloides*) at highest elevations. Riparian areas primarily are comprised of willows (*Salix* spp.), alders (*Alnus* spp.), cottonwoods (*Populus* spp.), rushes (*Juncus* spp.), sedges (*Carex* spp.) and grasses, with some rose (*Rosa* spp.), dogwood (*Cornus* spp.), and serviceberry (*Amelanchier* spp.). Streamflows are dominated by snowmelt runoff and summer thunderstorms. Goose Creek tributaries have been listed as impaired waters due to fecal coliform, *E. coli*, dissolved oxygen, nutrients, sediment, and temperature from non-point sources, and Total Maximum Daily Loads (TMDLs) have been developed for the subbasin (IDEQ 2010b; IDEQ 2010a).

Beginning in the early 1990s, grazing management changed on the Goose Creek Group Allotment to improve upland and riparian health. Starting in 1991, the allotment was grazed under an informal, year-to-year grazing plan with a primary focus on improving rangeland health within the uplands and riparian areas. In 2005, the Goose Creek Group Allotment was split into two management use areas used by two smaller herds of cattle. The change from one large herd to two smaller herds allowed easier pasture-to-pasture movement of livestock across rough topography and through dense upland vegetation (*Juniperus* spp). Several range improvements were also constructed that allowed for improved livestock control and lessened the duration of grazing on several streams. In return, this allowed for livestock grazing adaptive management to be exercised, specifically regarding the management of riparian areas since annual changes can be made to the grazing system, due to weather or unforeseen circumstances, to ensure resource conditions continue to improve. The management system also allows for riparian management

triggers if needed. For example, once a riparian monitoring threshold had been met, such as reaching certain stubble heights on hydric species or browse utilization limits on preferred woody riparian species, then cattle would be moved to the next pasture in the grazing rotation for that year. Other grazing management actions were implemented to encourage livestock use of uplands. Juniper encroached plateaus lacking herbaceous understory were treated and then seeded to crested wheat grass (*Agropyron cristatum*) to provide preferred forage away from riparian areas and where wind also reduces stress from nuisance insects. Three water systems were installed in upland areas away from riparian areas to deliver water from springs to a series of watering troughs. Lastly, multiple riparian exclosures were constructed from 1982 to 2005 to exclude livestock from lotic riparian areas and springs, some of which are used as riparian pastures for a few days each year (Table 12; Figure 16).

Table 12. Riparian exclosures evaluated in Goose Creek Allotment, Cassia County, Idaho.

Exclosure	Stream	Year built	Function	Stream length (m) and area (ha) enclosed	Comments
Coal Banks (CLBK)	Goose Cr.	1996	Exclosure	440 m (4.7 ha)	Former recreational site
Goose Cr. (GOCR)	Goose Cr.	1988	Exclosure	675 m (8.8 ha)	Expanded in 1995
Upper Cold Cr. (UCLD)	Cold Cr.	1988	Exclosure	525 m (2.4 ha)	Considerable maintenance in 2014
Lower Cold Cr. (LCLD)	Cold Cr.	1982	Exclosure	180 m (0.5 ha)	
Emery Creek (EMRY)	Emery C.	2003	Exclosure	700 m (3.1 ha)	Structural issues. Cattle use apparent
Little Birch Cr. (LBCH)	L. Birch Cr.	2005	Riparian Pasture	7530 m (89.4 ha)	5 days use in October
Stateline (STATE)	L. Birch Cr.	1999	Riparian Pasture	1680 m (45.1 ha)	Pasture not used in rotation since 1999

### Sampling Design

We evaluated differences in riparian vegetation, instream habitat, and aquatic biota inside and outside of the exclosures using a paired study design with replication, while also taking advantage of the Landsat archive to assess riparian vegetation changes over time. Four sites were randomly selected on the streams inside each of the seven exclosures in the study allotment, and four sites were randomly selected on streams outside each exclosure (eight sites per exclosure) for a total of 56 potential sample sites (Figure 16). Because hydrography datasets (e.g., National Hydrography Dataset) may not accurately represent stream locations, we digitized streams using National Agriculture Imagery Program (NAIP) imagery in ArcGIS software (ESRI, Redlands, CA). Four sites were randomly selected inside each exclosure, and four sites were selected outside each exclosure, using the Create Random Points ArcGIS tool (ESRI, Redlands, California). Site selection was limited to public lands; when possible, sites were constrained to be no closer than 100-m to each other. This paired study design with replication facilitated both allotment-wide and exclosure-specific analyses and inferences.

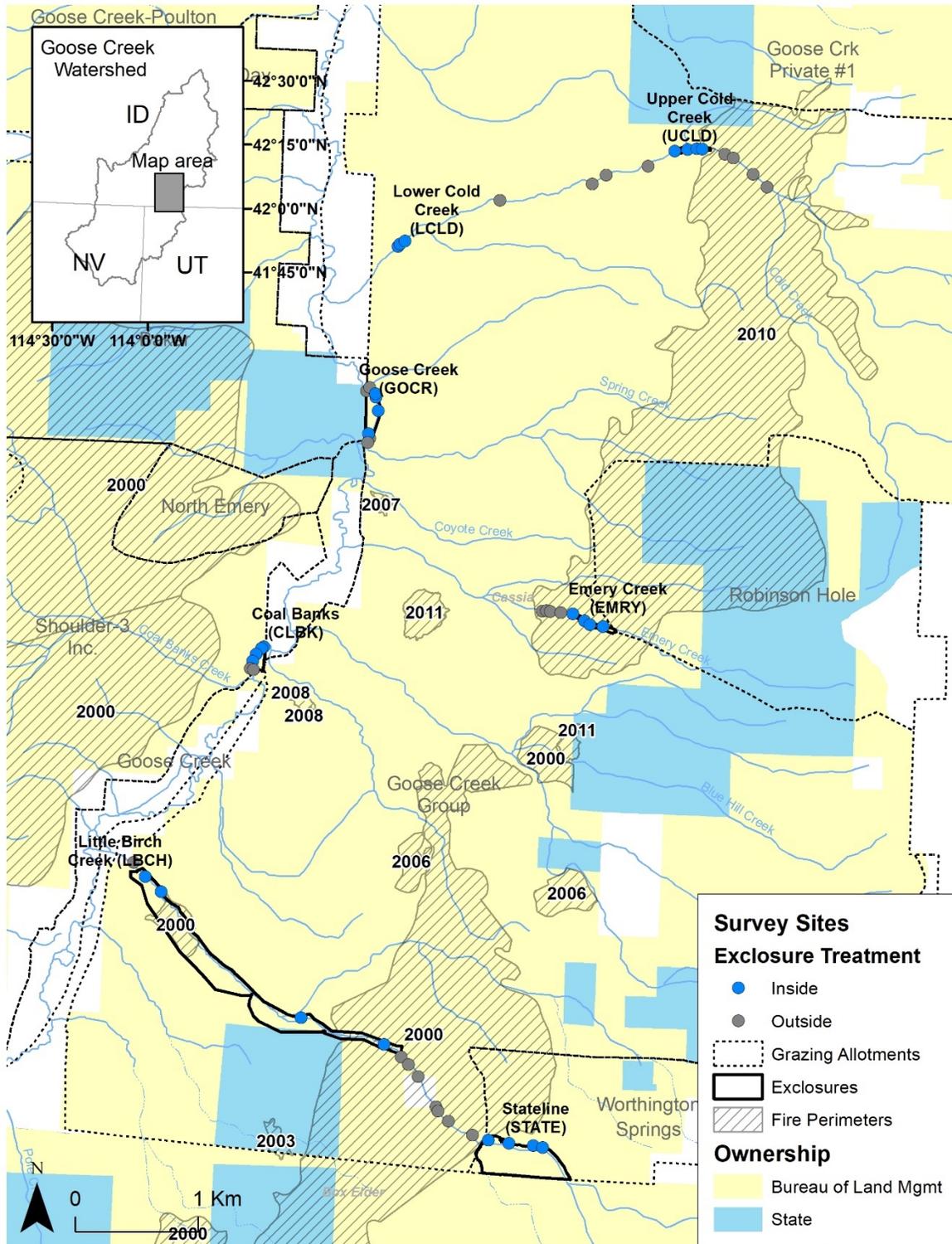


Figure 16. Map of Goose Creek Group Allotment, study enclosures, and field survey sites sampled for riparian and instream habitat, aquatic macroinvertebrates, and fish in 2015. Only perimeters from fires since 2000 shown.

### *Satellite-Measured Riparian Vegetation*

We used 30-m Landsat Normalized Difference Vegetation Index (NDVI) data from the available Landsat archive (1985 to 2015) to evaluate temporal changes in riparian vegetation inside and outside of exclosures. NDVI is computed as a ratio of visible red and infrared electromagnetic band data in multispectral imagery. Since plants absorb red and reflect infrared wavelengths, NDVI is effectively a measure of greenness that represents vegetation productivity (Pettoelli et al. 2005). Woody riparian vegetation often has higher NDVI values and, therefore, NDVI has been used in various applications to characterize terrestrial vegetation productivity and quantify woody vegetation in riparian areas (Dauwalter et al. 2017). We summarized average peak late growing season NDVI within a 25-m buffer around each sample site, although the buffer was sometimes narrower to ensure it remained inside a riparian exclosure. We used the maximum NDVI observed over the late growing season from July 25 to September 17 of each year in order to directly measure riparian zone productivity and minimize the greenness signal from upland vegetation, such as annual grasses (e.g., Cheatgrass, *Bromus tectorum*), which senesce earlier in the growing season (Bradley and Mustard 2008). Google Earth Engine was used to access the Landsat 5, 7, and 8 surface reflectance archive and summarize NDVI for each site (Gorelick et al. 2017).

### *Field-Measured Riparian Vegetation, Instream Habitat, and Biota*

Riparian vegetation, instream habitat, benthic macroinvertebrates, and fish were sampled within a 50-m reach associated with each randomly selected site. The random point served as the downstream reach boundary. Macroinvertebrates were typically sampled 1-d before fish and habitat sampling, and habitat sampling was completed immediately following fish sampling.

Four composite benthic macroinvertebrates samples were collected from four separate riffles (when possible) per site using a Surber sampler (0.093-m<sup>2</sup>; 500 µm mesh) following the methods of the BLM National Aquatic Monitoring Framework (USBLM 2017). For each targeted riffle sample, the Surber sampler was placed in a random location and substrate and organic debris agitated to a depth of 10-cm allowing benthic macroinvertebrates to drift downstream into the net. Samples were fixed with 95% ethanol. The samples from each site were sorted to a split count of 600 individuals, and insects (Class: Insecta) were identified to genus and non-insects to a coarser resolution. Prior to analysis, we standardized the taxonomic resolution among all samples by assigning identified macroinvertebrates to unambiguous operational taxonomic units (OTUs) (Yuan et al. 2008). Individuals identified to a coarser taxonomic resolution that precluded placement into unambiguous OTUs were excluded from all analyses.

Macroinvertebrate counts were subsequently standardized to a fixed-count of 300 randomly selected individuals per sample for richness-based metric computation to minimize bias associated with disparate counts among samples (Vinson and Hawkins 1996; Gotelli and Colwell 2001).

Benthic macroinvertebrate data were used in a multimetric index developed for Idaho's Plains, Plateaus, and Broad Valleys region (Jessup 2011) to assess the biological condition of streams

and evaluate macroinvertebrate responses to riparian exclosures. Metrics included in the index were selected based on their ability to discriminate between reference (least-impacted) and non-reference sites. The six metrics in the index are: Simpsons Diversity Index; % non-insect taxa; % filterer taxa; % clinger taxa; % tolerant taxa; and semi-voltine taxa richness. Jessup (2011) provides the taxa classifications that determined the functional feeding guild, habit, tolerance values, and voltine status. Metrics were scored 0 (most-impacted) to 100 (least-impacted), and metric scores were averaged for a composite site score to provide an overall understanding of the biological condition of streams. For the PPBV region, an overall score of 68 represents the 50<sup>th</sup> percentile of reference sites, and a score of 54 represents the 10<sup>th</sup> percentile. Sites with combined scores above 68 are given a condition rating of 3 (Good; full support of aquatic life) by the Idaho Department of Environmental Quality, sites scoring between 54 and 68 are given a condition rating of 2 (Fair), and those below a score of 54 are given a rating of 1 (Poor)(IDEQ 2016). Raw metric values (e.g., % clingers) were evaluated as response variables in exclosure analyses (see below).

Fish were collected from each 50-m reach using daytime backpack electrofishing (Dunham et al. 2009). Prior to sampling, each reach was enclosed with 6.35-mm mesh block nets. Electrofishing was completed with one Smith-Root LR-24 backpack electrofisher and one to three netters at all sites except on Goose Creek mainstem sites where two electrofishers and three netters were used. All fish species were collected during the first electrofishing pass. Subsequent passes were made if salmonids were caught on pass one; three passes were typically completed, unless no salmonids were caught on pass two then sampling ceased, or more than three passes were completed if a sampling efficiency was perceived to be poor or salmonids were not depleted (i.e., catch declined) during the first three passes. Electrofishing was conducted with direct current at 250-270 V, 30-40 Hz, and a duty cycle of 12-25. Salmonids were measured for total length (TL) and weighed; non-game species were counted. Abundances of each trout species was estimated separately for individuals <100-mm TL and >100-mm TL using a Zippin estimator using the 'removal' function in the FSA package (Ogle 2017) of Program R (R Core Team 2015), and abundances of all trout species were summed and expressed as a linear density (number per 100-m). Fish species richness was also computed per site.

Riparian and instream habitat was sampled immediately after fish sampling. Transects perpendicular to the channel were established every 5-m at bankfull height for a total of ten transects per site. Channel depth, water depth, water velocity, stream substrate, and cover type were measured at 10 equally spaced points along each transect. Water velocity was measured at 0.6 of water depth using a Hach FH950 velocity meter (HACH Company, Loveland, Colorado). Complexity in water velocity and depth were both calculated as a standard deviation of the 100 points across the 10 transect measurements. Stream substratum was classified according to the modified Wentworth scale: bedrock, silt/clay (<0.064 mm diameter on b-axis), sand (0.064-2 mm), gravel (2-15 mm), pebble (15-64 mm), cobble (64-256 mm), or boulder (>256 mm) (Cummins 1962). Cover was classified as: boulder, large wood (>10 cm diameter, >4 m in length), small wood, aquatic vegetation (macrophytes), overhanging bank vegetation, undercut bank (>10 cm depth), or absent. Substrate and cover diversity were computed using the Shannon-Wiener index ( $H' = -\sum_{i=1}^n p_i \cdot \log_e p_i$ ) where  $p_i$  represented the proportion of substrate or cover type  $i$  and  $n$  was the number of different types (Legendre and Legendre 2012). Pools were identified as slow water habitat according to Hawkins et al. (1993), and residual pool depth

was measured for all pools as maximum pool depth minus water depth at the downstream riffle crest. Woody riparian vegetation height was classified above each transect endpoint at bankfull as: 0.0-0.5 m, 0.5-1.0 m, 1.0-2.0 m, 2.0-4.0 m, 4.0-8.0 m, and >8.0 m (Burton et al. 2011). We computed percent woody riparian vegetation as the percent of transect endpoints with woody vegetation greater than 1 m in height. Streambank alternation was recorded as the presence of livestock hoof prints or trails within 2.5 m of the transect endpoint. Streambanks was classified at each transect endpoint using an ordinal scale representing a continuum of bank erosion: absent, fracture, slump, slough, or eroding (Burton et al. 2011); we used the percent of streambank classified as slump or slough as a measure of streambank stability. Reach slope was measured as the difference in elevation between reach boundaries divided by reach length and multiplied by 100 (expressed as a percentage).

### *Data Analyses*

We assessed the effect of riparian exclosures on riparian vegetation using Landsat NDVI as the response variable in generalized linear mixed models (GLMMs). Site location relative to the exclosure (inside = 1, outside = 0) and timeframe (before exclosure = 0, after = 1) were included as main effects. The interaction of these two factors was the parameter of interest, as it was expected that riparian vegetation, especially woody vegetation, would experience increased growth and therefore NDVI values would be higher inside exclosures but only during years after the exclosures were constructed (see Table 12). Because vegetation greenness and productivity are influenced by precipitation and fire, we also included main effect terms to control for these two covariates. We summarized annual precipitation (cm) within the water year (October 1 of the prior year through September 31) for each site using data from Daymet Version 2 (Thornton et al. 2014) accessed using Google Earth Engine. We also determined whether each site fell within a wildfire perimeter from the Monitoring Trends in Burn Severity data (Eidenshink et al. 2007), and attributed each site as being affected by fire for two years post fire (Fire = 1, No Fire = 0). A global model was fit using data from all sites, years (1985 to 2015), and exclosures to assess the general response of riparian vegetation to the exclosures across the Goose Creek Allotment. For this model, all terms mentioned above were included, as well as a random effect for each individual exclosure. We also fit separate general linear models (GLMs) for each exclosure to parse any anomalous responses for individual exclosures; the fire term was omitted from GLMs for exclosures where no sites were located within a wildfire perimeter during the time period of interest, and no random effect term was used. Parameter estimates were evaluated for significance using a one-tailed test at  $\alpha = 0.10$ , with the tail being applied the direction of the predicted response (Table 13).

We also used GLMMs to evaluate the general, allotment-wide effects of exclosures on riparian vegetation, instream habitat, and aquatic biota. Each riparian vegetation, instream habitat, macroinvertebrate, and fish response metric (e.g., %Bank Slough/Slump) was the response variable, exclosure treatment (inside = 1, outside = 0 [baseline]) was the treatment effect, and each exclosure was treated as a random effect. This global analysis estimates a common exclosure effect across all exclosures in the allotment for each response variable; the random effect (the intercept) adjusts for the uniqueness at sites associated with each exclosure. The

exclosure treatment term was assessed for significance using a one-tailed test at  $\alpha = 0.10$ . In addition to the global analyses, we also fit separate GLMs to estimate the effect of each individual exclosure on all response metrics. The same set of response metrics were used in the exclosure-specific linear models, and exclosure treatment was the treatment effect (inside = 1, outside = 0 [baseline]) as before. Again, the exclosure treatment was assessed for significance using a one-tailed test at  $\alpha = 0.10$ .

Next, we assessed whether the strength of any riparian vegetation, instream habitat, or biological responses to exclosures (effect size) was related to the size of each individual exclosure. The treatment effect size estimated for each response variable was obtained from the GLM from the previous analysis and then correlated with the length of stream enclosed by each exclosure. A Spearman rank correlation coefficient ( $r_s$ ) was used to evaluate the strength of correlation and significance was evaluated at  $\alpha = 0.10$ .  $n = 7$  for each correlation.

## Results

### *Sampling Design*

We strove to implement a paired sampling design with replicated sampling both inside ( $n = 4$ ) and outside ( $n = 4$ ) of each of the seven riparian exclosures in the allotment, while also attempting to ensure sites were at least 100-m apart. However, space limitations precluded reaching our sample size and site spacing goals for our sampling design (Figure 16). The two exclosures on the Goose Creek mainstem (GOCR and CLBK) exclosed almost all Bureau of Land Management lands encompassing the Goose Creek mainstem. As a result, only two sites outside the Coalbanks exclosure (CLBK) could be sampled, and sites outside the Goose Creek exclosure (GOCR) were closer than 100-m apart. The Lower Cold Creek exclosure was also only 180-m in length and could only accommodate three sites inside the exclosure and these sites were effectively located back-to-back. We discuss the limitations of implementing an ideal sampling design in the Discussion.

### *Satellite-Measured Riparian Vegetation*

Riparian vegetation was generally more productive inside the exclosures after they were constructed (Figure 17), and this was reflected in the Landsat NDVI time series (Figure 18). The global GLMM showed there to be a significant interaction between site location and timeframe, owing to higher NDVI values inside exclosures after they were constructed as predicted (Figure 18; Table 2). In addition, precipitation had a significant positive effect and fire (for two subsequent years) had a negative effect on NDVI as expected (Table 13). Despite a global positive effect of exclosures on NDVI, analysis of the individual exclosures showed higher NDVI after exclosures were built for 5 of 6 exclosures evaluated individually (Table 13). The Lower Cold Creek (LCLD) exclosure could not be evaluated by itself because it was built in 1982, whereas 30-m resolution Landsat NDVI data were not available until 1985. Upper Cold

Creek (UCLD) was the only exclosure where NDVI did not show a significant increase at sites inside the exclosure after it was constructed. Like the global analysis, precipitation had a positive effect on NDVI at each individual exclosure. Fire only occurred at sites associated with four exclosures but not all sites per exclosure (UCLD, EMRY, LBCH, STATE; Figure 16, Figure 18), and it had a significant negative effect on NDVI at sites associated with 3 of 4 exclosures; the Stateline exclosure (STATE) was the only exclosure where NDVI did not show a response to fire (Table 13).



*Figure 17. Photos from before (Top; 1988) and after (Bottom; 2011) installation of a riparian exclosure (GOCR) on Goose Creek, Cassia County, Idaho. Photo credits: Burley Field Office, BLM.*

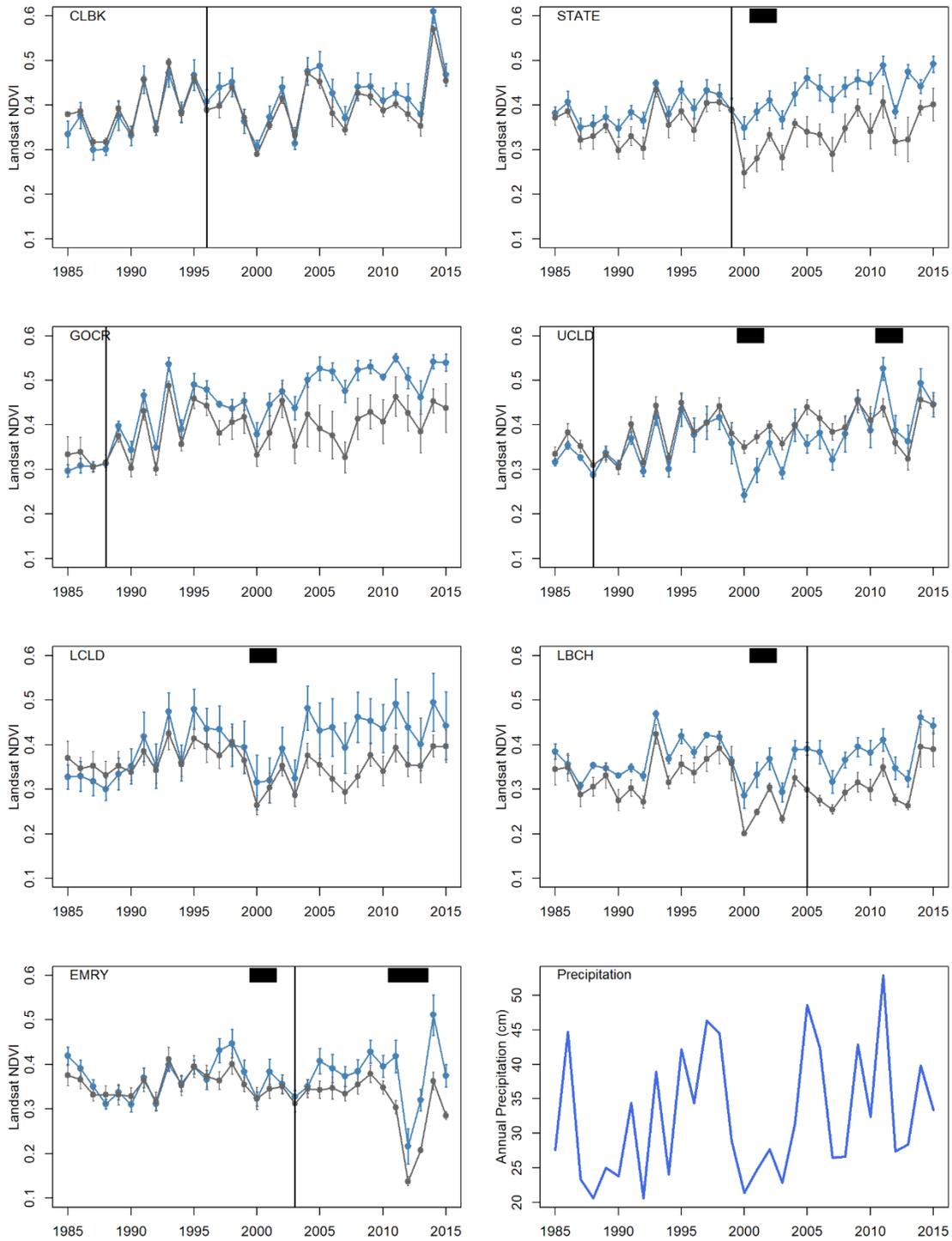


Figure 18. Mean ( $\pm 1$  SE;  $n=4$ ) Landsat NDVI over time by enclosure treatment (inside  $\bullet$ , outside  $\bullet$ ) for each enclosure. Vertical line indicates when enclosure was built; LCLD enclosure was built in 1982. Years fire expected to influence NDVI values at some or all sites per enclosure indicated at top (■), typically two years post fire (see Figure 1). Annual precipitation during water year shown in bottom right panel.

Table 13. Parameter estimates from a linear model with maximum annual NDVI as a response variable as a function of water-year precipitation, exclosure treatment (inside=0, outside=1), and timeframe (pre- or post-exclosure) from 1985 to 2015. *t* and *P*-values are not shown for main effects of terms included in the interaction. The model including all exclosures was a general linear mixed model with a random effect for exclosure. Exclosure LCLD was not evaluated individually because NDVI data were not available pre-exclosure (1982).

Exclosure	Parameter	$b_i$	1 SE	<i>t</i> -value	<i>P</i> -value
All	Intercept	0.253	0.011	23.8730	<0.0001
	Precipitation	0.003	0.001	19.9048	<0.0001
	Fire	-0.052	0.008	-6.8193	<0.0001
	Post Exclosure	-0.001	0.005		
	Inside Exclosure	0.019	0.005		
	Post x Inside	0.026	0.007	4.0127	<0.0001
CLBK	Intercept	0.262	0.020	12.8855	<0.0001
	Precipitation	0.004	0.001	8.0254	<0.0001
	Post Exclosure	-0.003	0.017		
	Inside Exclosure	-0.008	0.016		
	Post x Inside	0.029	0.020	1.4341	0.0766
EMRY	Intercept	0.263	0.013	20.0151	<0.0001
	Precipitation	0.003	0.001	8.4023	<0.0001
	Fire	-0.064	0.011	-5.8038	<0.0001
	Post Exclosure	-0.056	0.010		
	Inside Exclosure	0.007	0.009		
GOCR	Post x Inside	0.064	0.014	4.6798	<0.0001
	Intercept	0.232	0.021	10.8201	<0.0001
	Precipitation	0.003	0.001	6.7660	<0.0001
	Post Exclosure	0.065	0.018		
	Inside Exclosure	-0.017	0.023		
LBCH	Post x Inside	0.087	0.025	3.4594	0.0003
	Intercept	0.215	0.013	16.4044	<0.0001
	Precipitation	0.003	0.001	8.8702	<0.0001
	Fire	-0.034	0.019	-1.7812	0.0381
	Post Exclosure	-0.020	0.010		
STATE	Inside Exclosure	0.047	0.008		
	Post x Inside	0.024	0.014	1.7125	0.0440
	Intercept	0.259	0.013	20.0530	<0.0001
	Precipitation	0.003	0.001	9.1725	<0.0001
	Fire	-0.025	0.022	-1.1612	0.1234
UCLD	Post Exclosure	-0.025	0.009		
	Inside Exclosure	0.030	0.009		
	Post x Inside	0.061	0.013	4.7444	<0.0001
	Intercept	0.216	0.017	12.9237	<0.0001
	Precipitation	0.005	0.001	12.2379	<0.0001
	Fire	-0.035	0.013	-2.7608	0.0031
	Post Exclosure	0.032	0.014		
	Inside Exclosure	-0.024	0.018		
	Post x Inside	0.007	0.020	0.3589	0.3600

*Field-Measured Riparian Vegetation, Instream Habitat, and Biota*

Several riparian vegetation and stream channel characteristics differed inside versus outside riparian exclosures, but most instream habitat and biological metrics, with two exceptions, did not show any difference. Riparian vegetation was taller, and the percentage of stream reach with woody vegetation  $\geq 1$ -m tall was higher inside exclosures versus outside ( $P < 0.10$ ; Table 14). Streambanks were less altered (no livestock trails or hoofprints), were in better condition (less sloughing and slumping), and were narrower and deeper (smaller channel width:depth ratios) inside versus outside exclosures ( $P < 0.10$ ; Table 3). No benthic macroinvertebrate metrics differed inside versus outside exclosures ( $P > 0.10$ ; Table 14), and only one fish metric showed a significant difference due to exclosure treatment. Age-0 trout ( $< 100$ -mm TL) densities (rainbow trout *Oncorhynchus mykiss* and brook trout *Salvelinus fontinalis*) were estimated to be 6 fish per 100-m higher inside versus outside exclosures ( $P = 0.065$ ; Table 14).

*Table 14. Riparian exclosure effect size ( $b_i$ ) for riparian and instream habitat, aquatic macroinvertebrate, and fish metrics (response variable) in a generalized linear mixed model with exclosure treatment (inside = 1, outside = 0) and exclosure as a random effect. Positive estimates indicate higher values inside of exclosures. Spearman rank correlation coefficient ( $r_s$ ) and  $P$ -values for association between exclosure effect size and exclosure length for each response variable.*

Category	Response variable	GLMM			Spearman	correlation
		$b_i$	1 SE	$P$ -value	$r_s$	$P$ -value
Riparian vegetation	Woody Vegetation Height	0.567	0.205	0.003	-0.464	0.302
	% Woody Veg. ( $> 1$ -m tall)	15.537	5.330	0.002	-0.464	0.302
Stream bank	% Bank Alteration	-17.293	8.657	0.023	0.000	1.000
	% Bank Slough/Slump	-6.486	3.894	0.048	0.107	0.840
Channel and instream habitat	Width:Depth Ratio	-3.069	0.794	$< 0.001$	0.595	0.159
	Wetted Width (m)	-0.199	0.144	0.084	0.643	0.139
	% Fines	9.685	4.548	0.983	0.429	0.354
	Residual Pool Depth (m)	-0.027	0.038	0.765	0.286	0.556
	% Cover	-9.387	7.857	0.884	0.018	0.969
	% Gravel	1.511	3.111	0.314	0.857	0.024
	Habitat diversity	Substrate H'	-0.032	0.065	0.690	0.286
	SD Water Depth (m)	0.006	0.006	0.170	0.357	0.444
	SD Velocity (m/s)	-0.019	0.010	0.963	0.429	0.354
Benthic macroinvert.	Simpsons Diversity	0.018	0.034	0.302	0.250	0.595
	% Non-Insects	-0.328	0.990	0.370	0.464	0.302
	% Tolerants	3.320	4.185	0.786	0.286	0.556
	% Filterers	-3.665	4.333	0.801	0.286	0.556
	% Clingers	-0.838	5.512	0.560	0.536	0.236
	Semivoltine Taxa Richness	0.037	0.384	0.461	0.607	0.167
Fish	Multimetric Index	-0.463	2.522	0.573	0.643	0.139
	Fish Species Richness	-0.185	0.272	0.751	0.631	0.129
	Juvenile Trout (#/100-m)	6.103	4.034	0.065	-0.291	0.527
	Adult Trout (#/100-m)	-1.330	2.883	0.678	0.464	0.302

Exclosure effect sizes for each response metric, as estimated from the exclosure-specific general linear models, showed no significant meaningful correlations with length of stream exclosed (Table 14). The estimated exclosure effect size for % gravel substrate was significantly correlated with exclosure size ( $r_s = 0.857$ ;  $P = 0.024$ ). However, the effect sizes for the three smallest exclosures were estimated to be negative (results not shown), which is counterintuitive and made this result difficult to interpret as it is unclear why an exclosure would result in less gravel in smaller exclosures but more gravel in larger exclosures. Thus, we view this as a spurious result.

Composite scores for the benthic macroinvertebrate multimetric index averaged 29 to 74 across all sites, and only Emery Creek inside the exclosure showed scores above the 50<sup>th</sup> percentile of reference conditions (Figure 19). All others sets of sites averaged between the 10<sup>th</sup> and 50<sup>th</sup> percentile of reference site scores (Fair condition) with the exception of the Goose Creek mainstem inside the Coal Banks exclosure (CLBK; mean score 41.5, SD = 2.9) and Emery Creek outside the exclosure (mean score 35.0, SD = 5.9) that scored lower than 54 indicating Poor conditions and impairment (IDEQ 2016). Most sites had scores near 100 for the non-insect metric, as most benthic macroinvertebrates were insects. The lowest scoring metric differed for each exclosure and whether sites were inside versus outside the exclosure (Figure 19).



*Identifying fish collected during an electrofishing survey on Goose Creek, 2015. Credit: R. Bjork.*

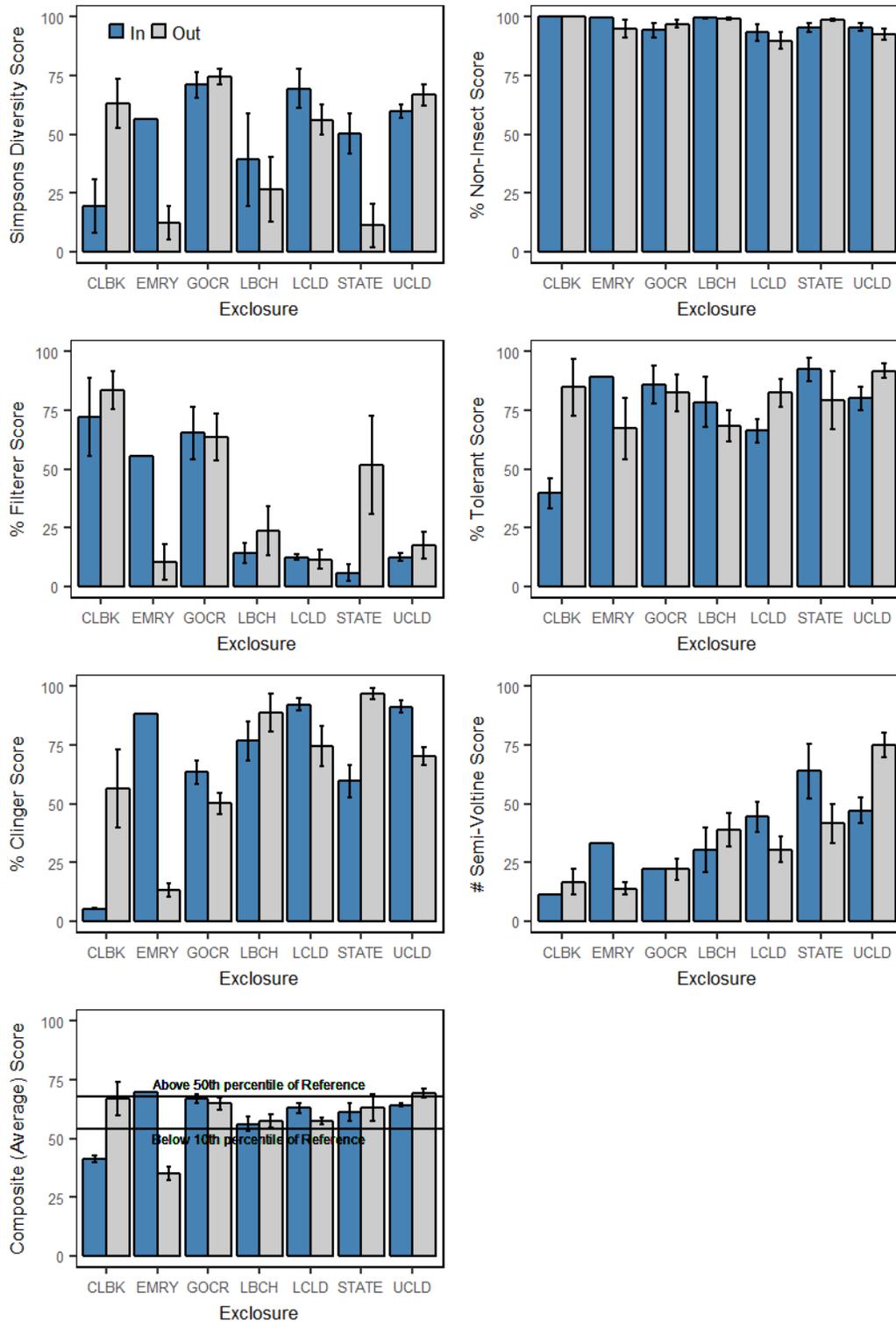


Figure 19. Mean (1 SE) scores from a benthic macroinvertebrate multimetric index by exclosure treatment (inside, outside) for each exclosure in the Goose Creek allotment. 50<sup>th</sup> and 10<sup>th</sup> percentile scores of reference sites in the Plateaus, Plains, and Broad Valleys region are shown for context.

## Discussion

We evaluated seven riparian exclosures in a livestock grazing allotment in the northern Great Basin and found that while exclosures contained more woody riparian vegetation and narrower and deeper channels, instream habitat and aquatic biota showed no response, except for age-0 trout. These findings align with much previous research that shows riparian exclosures often have a localized effect on riparian vegetation and channel morphology but only occasionally are shown to benefit aquatic organisms (McDowell and Magilligan 1997; Herbst et al. 2012). This is intuitive because instream habitat and biota reflect larger-scale watershed processes and disturbances, and we think that the benthic macroinvertebrates and fish assemblages in our study are reflecting aquatic ecosystem properties at a larger, watershed scale as is commonly shown (Wang et al. 1997). Since we observed no correlation between exclosure effect size (i.e., the difference in response metrics inside versus outside exclosures) and exclosure length, the riparian exclosures in the Goose Creek Allotment may not have been large enough to have a strong effect on aquatic communities. However, exclosures did vary in size from 0.5 to 90-ha, the upper range of which is much larger than a typical exclosure on public lands in the western U.S. (<20-ha; Sarr 2002) but with a caveat that it is used as a riparian pasture for 5-d each year (Table 12).

The most immediate and apparent response to grazing exclosures was an increase in woody riparian vegetation as measured both by satellite imagery and field data; NDVI showed an immediate response to some exclosures during wet years in the absence of fire (e.g., STATE). Multiple studies have shown that woody riparian vegetation communities in poor condition from season-long, hot-season grazing can recover under conservation grazing regimes implemented to promote riparian and stream health (Swanson et al. 2015). Booth et al. (2012) showed that riparian willow communities increased two- to three-fold on Great Basin streams two to three years after implementation of a conservation grazing plan. Others have also shown aspen (*Populus* spp) recruitment to increase substantially in the Great Basin after cessation of livestock herbivory (Beschta et al. 2014). Riparian exclosures, when functioning, exclude livestock herbivory and naturally allow for rapid growth and recovery of woody riparian vegetation (Sarr 2002). Vegetation recovery often takes only a few years after cessation of grazing (Sarr 2002), but there are exceptions. Complete willow recovery, especially when planted as part of a restoration program, can take 25 years even when livestock are excluded, as was shown for high elevation meadow streams in the Sierra Nevada mountains in California, U.S.A. This highlights the importance of environmental context and growing conditions on vegetation response (Nusslé et al. 2017). NDVI was still marginally higher inside the CLBK exclosure when woody riparian vegetation was sparse, and is an example from our study that highlights the unique response of riparian vegetation to each exclosure due to exclosure position in the watershed, when the exclosure was built, exclosure integrity related to fence maintenance (Table 12) and, ultimately, that not all riparian exclosures have the same or desired effect on riparian plant communities (Sarr 2002; Swanson et al. 2015).

The one aquatic organism response metric showing an exclosure effect was number of age-0 trout per 100-m, a biological metric that has shown a positive response to exclosures in previous studies. Age-0 trout were, on average, six individuals more abundant per 100-m inside exclosures versus outside, albeit only significantly different from zero with a liberal probability

value ( $P = 0.065$ ). Bayley and Li (2008) also found age-0 rainbow trout *O. mykiss* to be 2.5 times more abundant inside than outside riparian exclosures in northeastern Oregon. They attributed higher age-0 densities inside riparian exclosures to lower avian predation due to dense riparian canopies (unmeasured), better age-0 trout habitat along streambanks, and increased invertebrate production (unmeasured). We also observed better age-0 habitat in terms of narrow and deep channels with more riparian vegetation, habitats shown to be selected by rainbow trout in another Goose Creek tributary (Dauwalter et al. 2014), and dense riparian canopy could have limited predatory opportunities from piscivorous birds (e.g., Belted Kingfisher *Megaceryle alcyon*; Penaluna et al. 2016) although we did not collect bird occurrence or abundance data. And while we did not observe any differences in benthic macroinvertebrate metrics inside versus outside our exclosures, we did not evaluate terrestrial invertebrate subsidies that can comprise a majority of juvenile salmonid diet contents in summer (Allan et al. 2003; Saunders and Fausch 2009). Avian predator abundance and terrestrial prey subsidies should be included in future exclosures studies containing salmonids.

Despite only one aquatic organism response to exclosures, the benthic macroinvertebrate multimetric index suggested stream health overall was often between the 10<sup>th</sup> and 50<sup>th</sup> percentile of least-disturbed (reference) stream scores and should be considered neither in the best condition nor overly stressed relative to other regional streams (IDEQ 2016). Aquatic macroinvertebrates do reflect livestock grazing impacts (Strand and Merritt 1999; Herbst et al. 2012). Thus, despite the lack of a strong biological response to exclosures stream health overall is fair and comparable to some regional reference streams that lack substantial human stressors, including localized grazing (Jessup 2011; IDEQ 2016). This indicates that the larger grazing management changes in the Goose Creek Group Allotment appear to have helped maintain stream health at a level exhibited by other regional, ungrazed stream sites (Jessup 2011).

Our study design was novel, but it was also influenced by practical constraints. Most riparian exclosure studies use a paired design where each exclosure has one field site inside and one site outside of the exclosure. This allows an exclosure effect for one or more response metrics to be estimated, but inferences can only be made across all exclosures in the study. In contrast, we used a paired design with replication per exclosure. This not only allowed us to estimate the exclosure effect across all exclosures and make inferences about exclosure effects across the allotment, but replication also allowed us to assess statistically the effect of each individual exclosure on response metrics with error. Despite a novel study design, site selection associated with some exclosures, although mostly random, was affected by logistical constraints. The two exclosures on the Goose Creek mainstem exclosed nearly all public land (U.S. Bureau of Land Management) encompassing the mainstem, and our field sites outside of the exclosures were on livestock water gaps or near road crossings – the reasons they remain unexclosed. The mainstem upstream and downstream of these two exclosures was on private land. Thus, while our study design allowed for inferences regarding exclosure effects on the entire allotment, in practice the instream habitat and biological communities in these exclosures were influenced by adjacent private land management. Other sites associated with exclosures (inside or outside) were located back-to-back because of space constraints, potentially influencing statistical independence.

A second novel aspect of our study was the use of remote sensing imagery to evaluate temporal changes to riparian vegetation in response to the exclosures. While some have used remotely-

sensed data to evaluate riparian vegetation recovery (Booth et al. 2012), many riparian enclosure studies lack this important temporal dimension that can be achieved with satellite or aerial imagery (Sarr 2002). Studies that do have a temporal dimension usually only have field data from a few years over a longer timeframe (Nusslé et al. 2017). In contrast, much satellite imagery is collected every few days, and some satellite programs have been in place now for decades. Landsat, for example, covers most points on Earth every 16 days dating back to 1985 (60-m Landsat MSS data are available to 1972), and the entire archive is available for free (Woodcock et al. 2008). Many other remote sensing datasets are freely and easily accessible through several delivery platforms to users with a computer and internet connection (Turner et al. 2015). Accessibility of satellite archives will greatly enhance our understanding of the temporal dynamics of aquatic ecosystems (Dauwalter et al. 2017), as historically aquatic studies, and ecological studies in general, have been largely spatial in nature (Franklin 1989).

Livestock grazing has and will continue to be a predominant land use globally. Demand for livestock-related products continues to increase in developing countries in conjunction with human population increases (Thornton 2010). Riparian enclosure research, including that presented herein, suggests that most of the benefits of riparian enclosures to aquatic organisms are localized and influence riparian vegetation but not aquatic habitat and biota (Sarr 2002). Thus, to effectively benefit all elements of stream ecosystems, including biological assemblages, riparian enclosures should be used in concert with other complementary grazing management techniques within entire allotments (1000's of hectares) such as use of upland forage, off-stream water systems, and grazing rotations that use multiple pastures where grazing intensity, timing, and duration can be managed adaptively. Allotment-scale changes to grazing practices have been shown improve habitat quality and benthic macroinvertebrates communities within four years (Herbst et al. 2012).

What role, then, do enclosures play within the broader realm of grazing management? We suggest that enclosures may still offer benefits not yet revealed by run-of-the-mill enclosure studies. Enclosures may protect or create unique habitat conditions that may unknowingly help species persist. For example, the Northern Leatherside Chub *Lepidomeda copei* is a rare minnow with a patchy distribution driven by streamflow complexity in the Goose Creek watershed (Dauwalter and Walrath 2017), and the species selects microhabitats with overhead cover and complex local hydraulics (current seams) that are associated with senesced branches of mature woody riparian vegetation (Dauwalter et al. 2014). Northern Leatherside Chub occupy mainstem Goose Creek, and the Goose Creek Enclosure (GOCR) is one of the only areas on the mainstem that has mature riparian vegetation (Figure 2). Most of the mainstem floodplain on private land is devoid of woody riparian vegetation because it was removed to maximize hay production during summer and hay meadows are used as pasture in winter. Thus, it is not known whether improved riparian conditions in the Goose Creek Enclosure facilitate persistence of Northern Leatherside Chub in other nearby areas of Goose Creek where habitat quality is lower. The increase in age-0 trout inside enclosures we observed also suggests that enclosures may protect or improve spawning habitats (not detectable from our habitat surveys) that produce fish that then emigrate to less-suitable habitats outside of enclosures. Future research should focus on whether enclosures play an important role in source-sink dynamics of sensitive fish species and help them to persist within broader areas of low quality habitat. Such research would help

managers strategically use exclosures to protect critical habitats and, therefore, continue to find them a useful tool in the grazing management toolbox.

**Supplemental Resources:**

Webpage: [www.tu.org/efficiency-exclosures](http://www.tu.org/efficiency-exclosures)



*Habitat survey on Little Birch Creek (inside LBCH exclosure), 2015. Credit: M. Baker.*



*Habitat Survey on Little Birch Creek (outside STATE exclosure), 2015. Credit: M. Baker.*

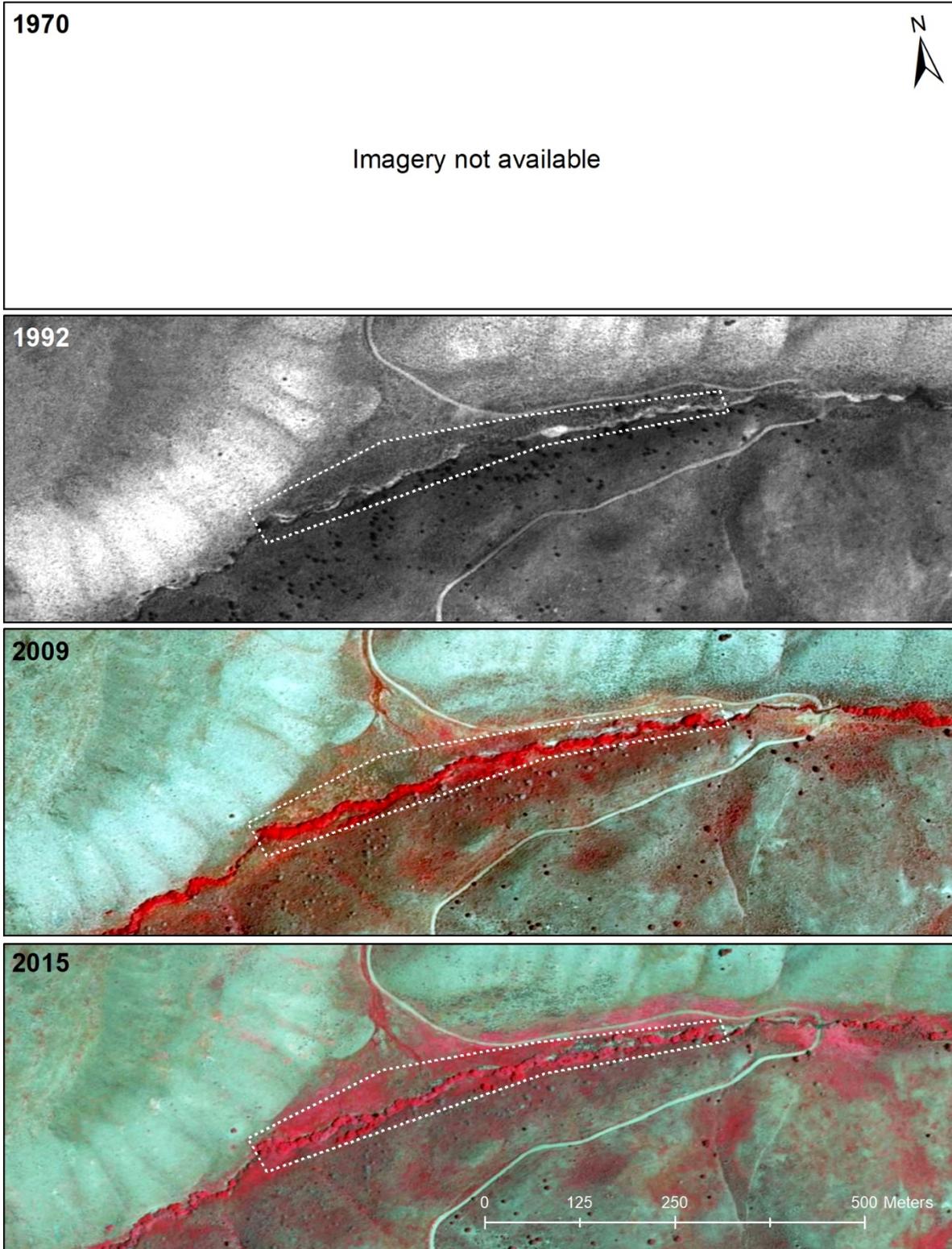


Figure 20S. Aerial imagery of Upper Cold Creek Exclosure from 1970 to 2015 ([www.insideidaho.org](http://www.insideidaho.org)).

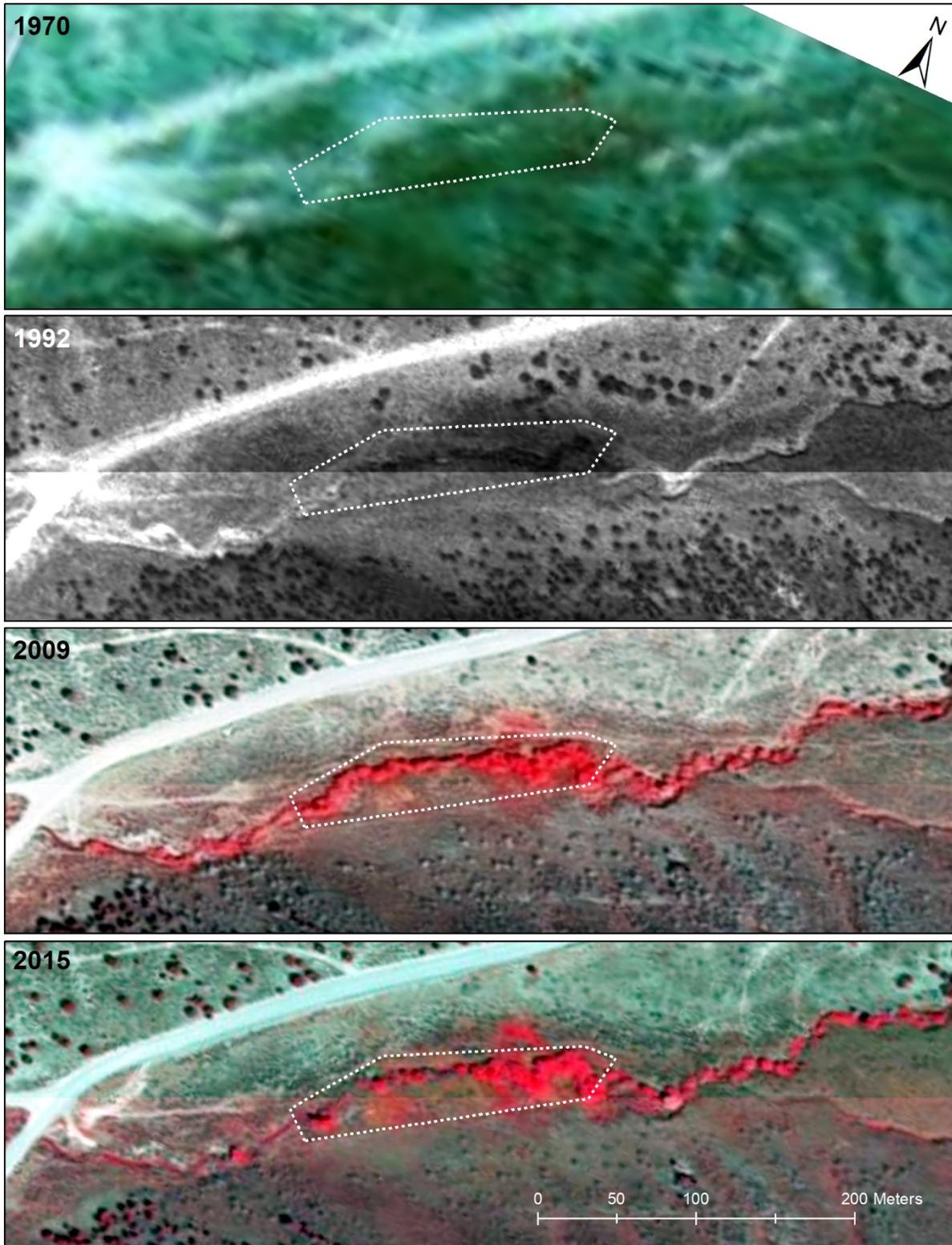


Figure 21S. Aerial imagery of Lower Cold Creek enclosure from 1970 to 2015.

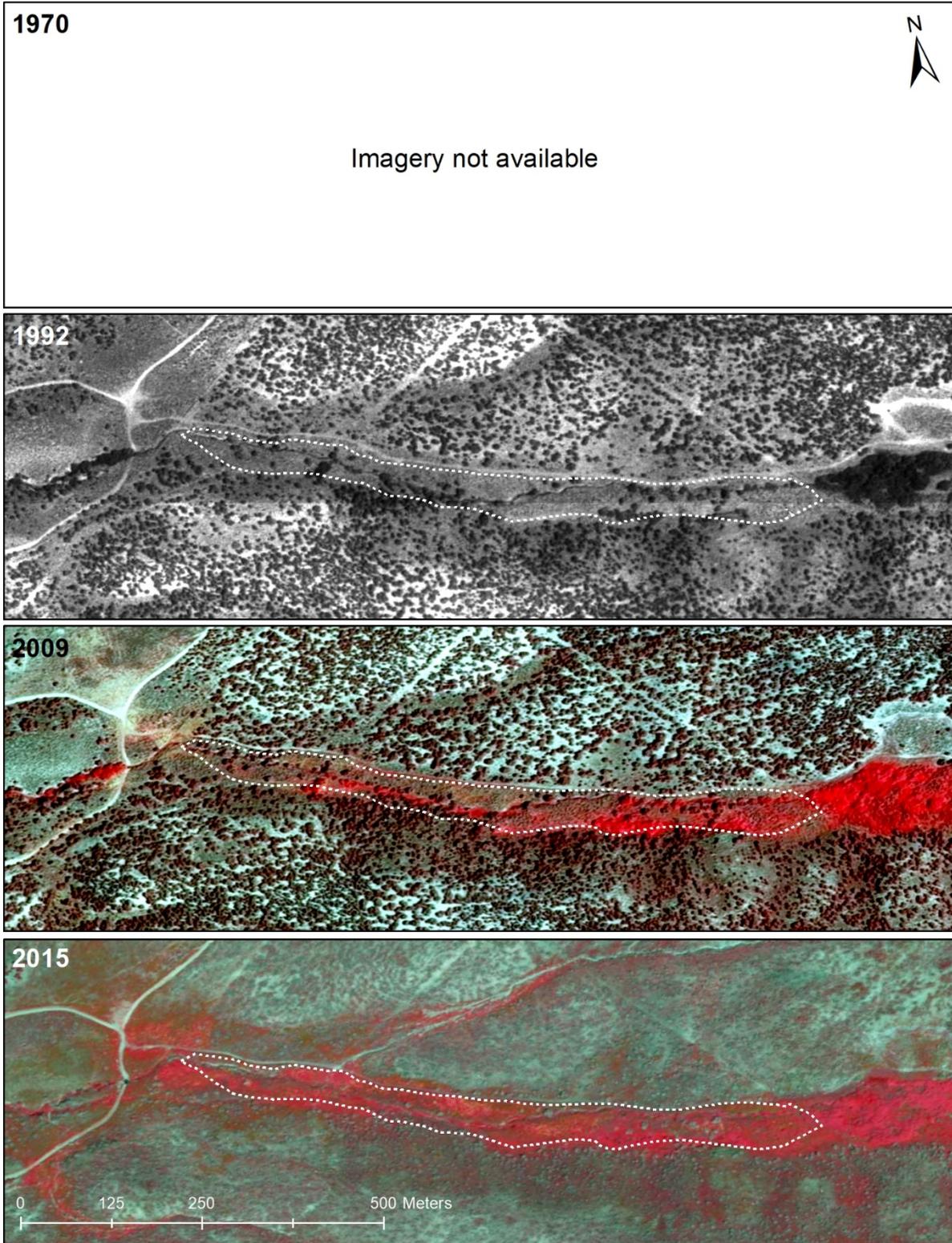


Figure 22S. Aerial imagery of Emery Creek exclosure from 1970 to 2015.

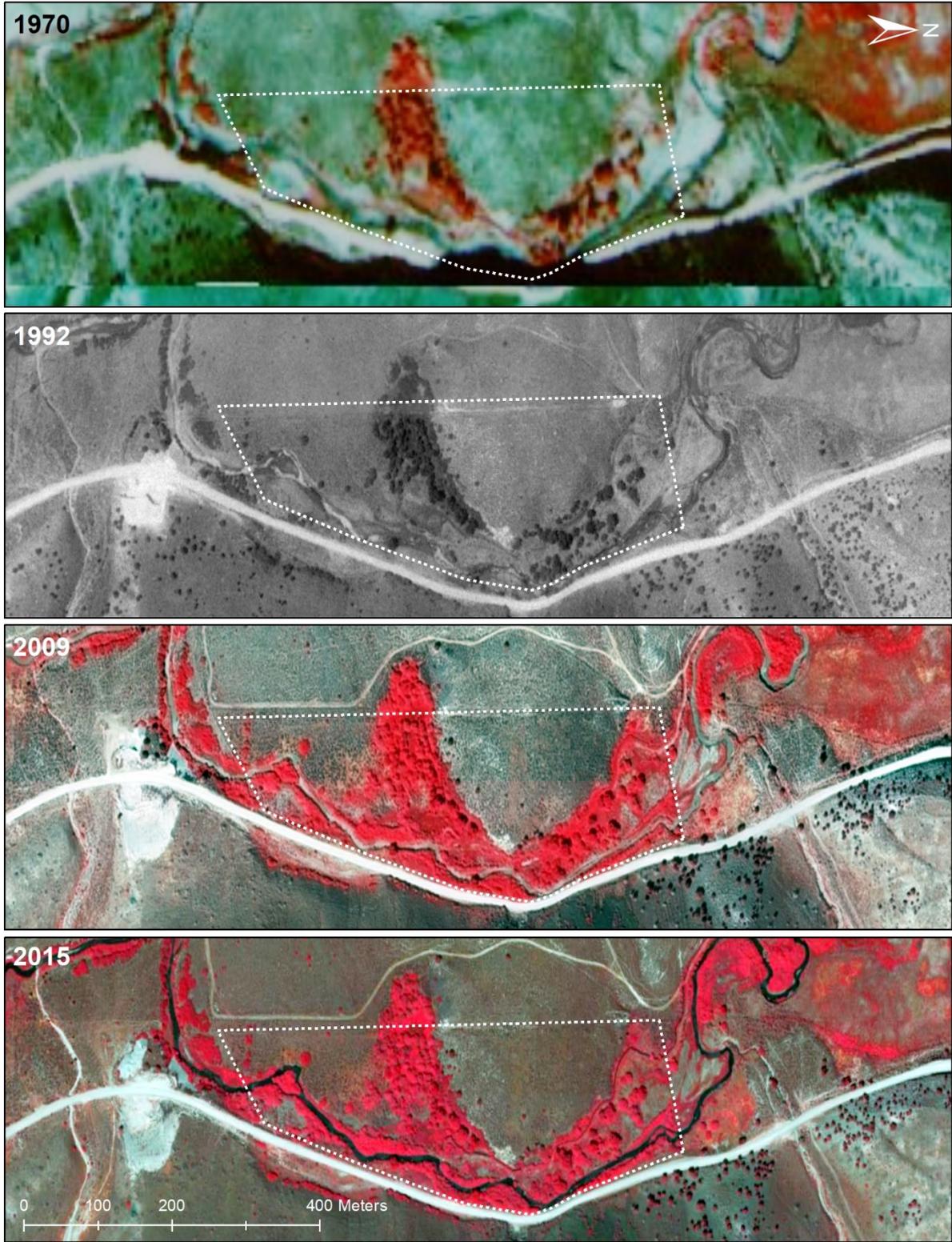


Figure 23S. Aerial imagery of Goose Creek enclosure from 1970 to 2015.

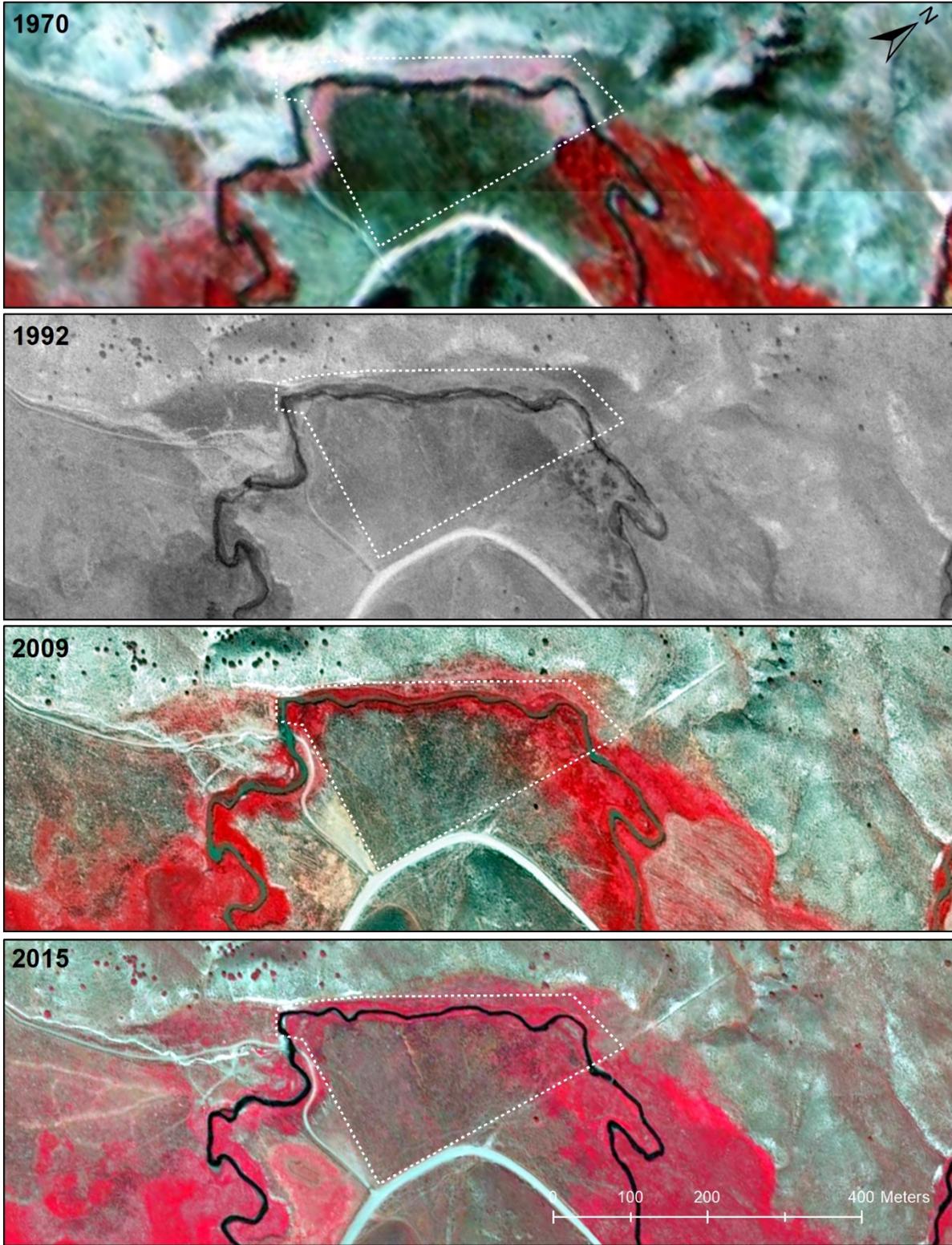


Figure 24S. Aerial imagery of Coal Banks enclosure from 1970 to 2015.

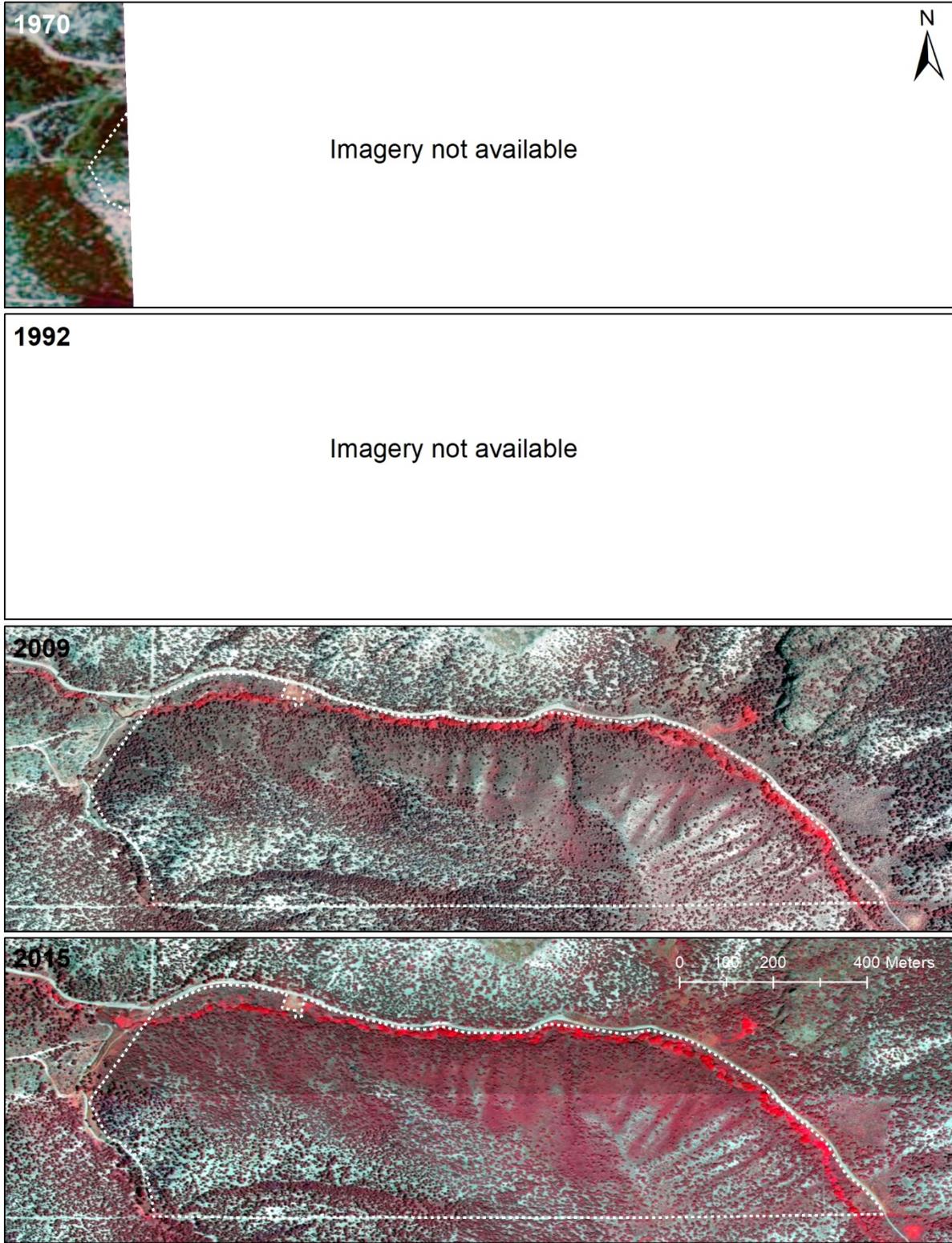


Figure 25S. Aerial imagery of Stateline enclosure from 1970 to 2015.

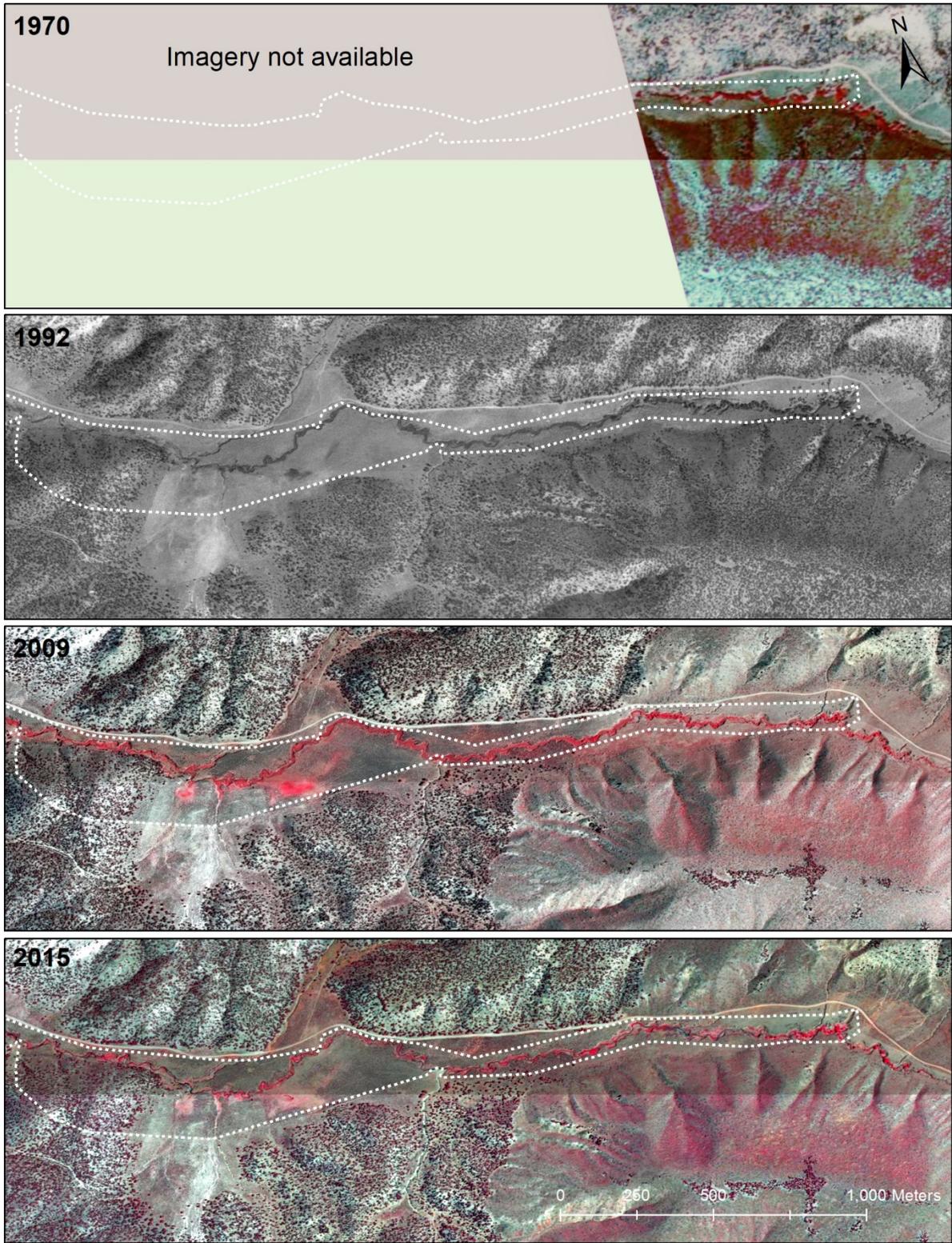


Figure 26S. Aerial imagery of Little Birch Creek (east) enclosure from 1970 to 2015.

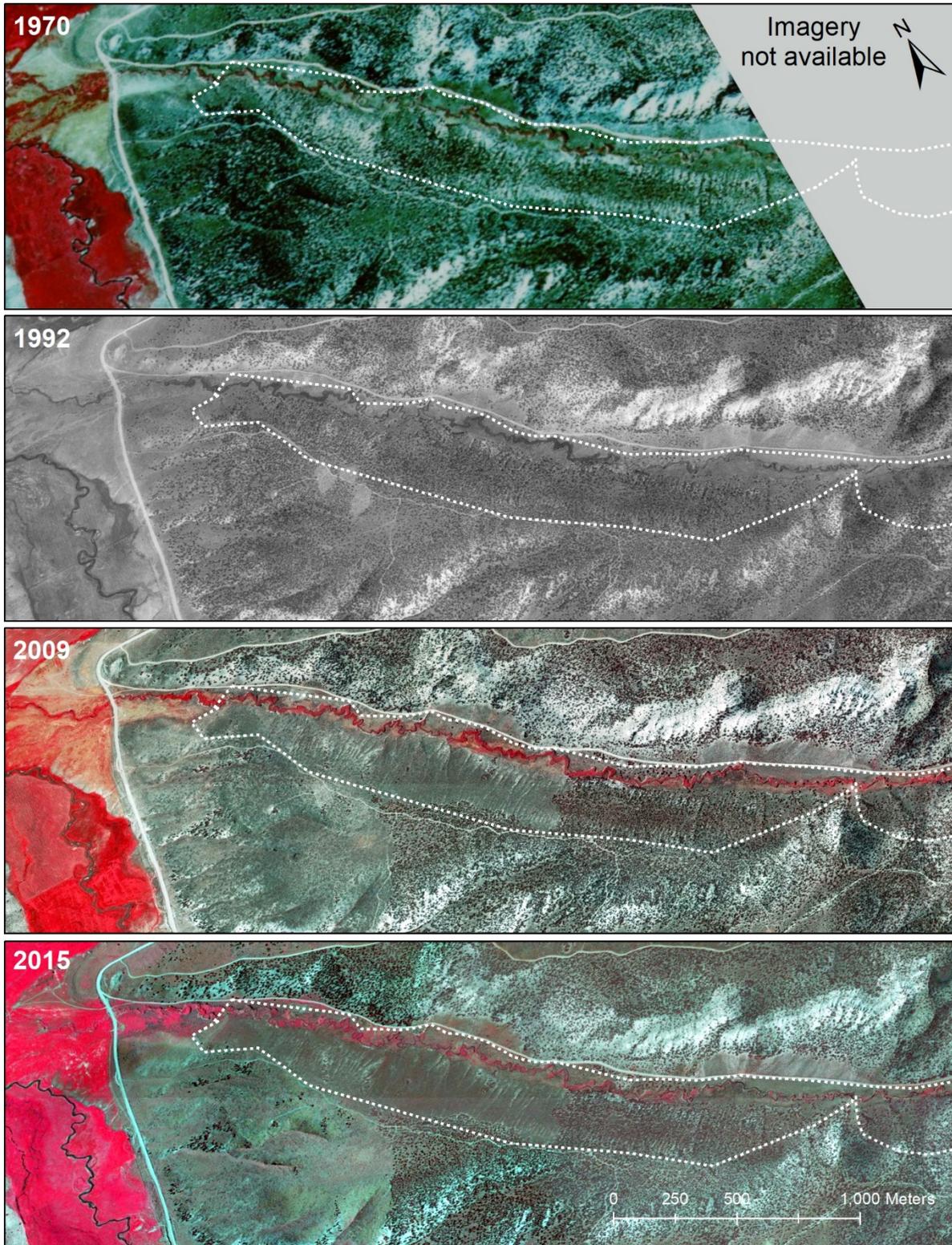


Figure 27S. Aerial imagery of Little Birch Creek (west) enclosure from 1970 to 2015.

## CHAPTER 6: THE ROLE OF COMPLEXITY IN HABITAT USE AND SELECTION BY STREAM FISHES IN A SNAKE RIVER BASIN TRIBUTARY

### Abstract

Impacts from grazing, agriculture, and other anthropogenic land uses can decrease stream habitat complexity that is important to stream biota and often is the goal of stream habitat restoration. We evaluated how microhabitat complexity structured a fish assemblage and influenced habitat selection by the Northern Leatherside Chub *Lepidomeda copei*, a recent candidate for listing under the Endangered Species Act, in Trapper Creek, a tributary to the Snake River in Idaho. Fishes were sampled using pre-positioned areal electrofishing ( $\sim 1 \text{ m}^2$ ), and microhabitat conditions were measured within a 1-m diameter circle centered on the electrofishing anode. Constrained correspondence analysis showed complexity in water depths and velocity to structure the fish assemblage and partition habitat use by Northern Leatherside Chub, rainbow trout *Oncorhynchus mykiss*, and redbreast sunfish *Richardsonius balteatus*. Habitat selection models showed that the Northern Leatherside Chub used areas of heterogeneous depths and flows in addition to the low velocity, deep habitats often considered to be the species' habitat. Additionally, chubs were almost certain to occur in deep-water habitats when overhead cover – often from mature riparian shrubs – was present. The complex depths and flows structuring the fish assemblage, and selected by the Northern Leatherside Chub, were often directly tied to other structural stream features such as boulders, mature riparian vegetation, and beaver dams – stream features that have direct ties to active and passive instream habitat restoration techniques. Our study suggests that habitat complexity should be routinely incorporated into studies evaluating fish habitat use, occupancy, and abundance. Doing so will result in models that are more informative to practitioners conducting stream restoration with a goal of improving habitat complexity.



*Pre-positioned aerial electrofishing setup with wire-loop anode in Trapper Creek. Credit: D. Dauwalter.*

## Introduction

Physical habitat provides a template for ecological strategies (Southwood 1977). Adequate quantities of usable habitat are needed to sustain viable populations, and when habitat features selected by a species are more abundant that area is considered to have higher quality habitat (Manly et al. 2002). The importance of aquatic habitat to fish populations has been widely accepted in fisheries management, and much of recent fisheries research has focused on quantifying habitat quality, understanding causes of degradation and factors limiting to populations, and developing models useful for habitat restoration and enhancement (Fisher et al. 2012).

Complex habitat is often considered to be higher quality habitat for stream fishes (Kovalenko et al. 2012), but habitat complexity has been defined differently across studies and the term has been used synonymously with habitat heterogeneity, diversity, and arrangement of habitat elements (Tokeshi and Arakaki 2012). Habitat complexity has been used to describe the number and diversity of habitat elements (cover types or substrate classes)(Gorman and Karr 1978), increased variance in habitat variables such as water depths or channel widths (Grossman et al. 1998), and the unique combinations of different habitat elements (Fore et al. 2007). For example, complex stream reaches have more niche space likely to meet the life history needs of multiple species in a fish community (Gorman and Karr 1978; Schlosser 1991). In turn, stream habitat complexity has been shown to be positively correlated with cutthroat trout *Oncorhynchus clarkii* abundance (Fausch and Northcote 1992; Horan et al. 2000), and Horan et al. (2000) suggested that Colorado River cutthroat trout *O. c. pleuriticus* populations require more habitat complexity to persist in small patches of isolated stream habitat. Increased complexity has also been shown to be positively associated with the diversity of stream (but see Gorman and Karr 1978; Grossman et al. 1998). Thus, complexity is often considered to be an important determinant of fish habitat use, species diversity, and even ecosystem stability, in streams (Kovalenko et al. 2012; Laub et al. 2012).

Land uses such as agriculture and grazing often impact riparian areas and degrade instream habitat quality (Rinne 1999; Allan 2004). Degraded streams often have low bank stability and wide, shallow stream channels with little or no wood, that is, instream physical habitat for fishes that lacks complexity (Platts and Nelson 1985; Lau et al. 2006). Stream and watershed restoration can focus on reestablishing watershed-scale processes such as hydrology, sediment transport, and wood recruitment that influence channel morphology, physical habitat, and water quality in streams (Roni et al. 2002; Schwartz and Herricks 2007; Palmer et al. 2010). Or, management may circumvent restoration of natural fluvial processes and, instead, create habitat complexity directly through manipulation and enhancement of stream habitat (Roni et al. 2008; Dauwalter et al. 2010). For example, creation of side channels increased habitat complexity in the Provo River, Utah and facilitated habitat partitioning between and the co-existence of native and non-native fish species (Billman et al. 2013). The addition of alcoves and wood in two Oregon streams improved winter habitat conditions and increased abundance and survival of anadromous salmonids (Solazzi et al. 2000). However, restoration does not always create habitat complexity (Laub et al. 2012). Even when it does, increased complexity may not always result in a detectable biological response because additional factors, such as proximity to source

populations or upstream sediment inputs, may prohibit a fish population response (Platts and Nelson 1985; Lau et al. 2006; Roni et al. 2008).

The goal of our study was to understand how stream habitat complexity influenced microhabitat use by the fish assemblage and habitat selection by the Northern Leatherside Chub *Lepidomeda copei* in Trapper Creek, Idaho. For our study we defined microhabitat complexity as the heterogeneity (i.e., variance) in water depths and velocities as well as interactions among different habitat components (i.e., variables), and we show how habitat complexity is created, in part, by secondary instream and riparian structural features: beaver dams and mature riparian shrubs. Our study offers insights into the current distribution of the Northern Leatherside Chub, a species of special concern (e.g., IDFG 2005) that is the focus of a multi-agency range-wide conservation agreement (UDWR 2011) and that was a recent candidate for listing under the Endangered Species Act (Federal Register 76:63444-63478). We then discuss the role of habitat complexity, riparian vegetation, and beavers in restoration efforts targeting the broader fish assemblage in the current range of the Northern Leatherside Chub.

## Study Area

Trapper Creek is a tributary to Goose Creek, which lies in the Basin and Range physiographic province on the Idaho, Nevada, and Utah borders. Goose Creek heads in the Sawtooth National Forest in southern Idaho (2,200-m elevation) and flows south into northeast Nevada and east into northwest Utah before flowing north into Lower Goose Creek Reservoir (1,450-m elevation) in Idaho near the Snake River plain, below which the creek is then diverted entirely for agricultural uses. Trapper Creek flows east from the Sawtooth National Forest and also flows directly into Lower Goose Creek Reservoir. The creek ranges from 2 to 5 m in width and has a riffle-run-pool morphology. The Goose Creek basin is a matrix of sage-steppe *Artemisia tridentata* and pine-aspen-juniper forest; higher elevations contain pine-aspen forests *Pinus* spp., *Pseudotsuga* spp, and *Populus tremuloides* whereas lower elevations contain pinyon-juniper-mountain mahogany *Pinus monophylla*, *Juniperus* spp., *Cercocarpus ledifolius*. Riparian areas are comprised of willows *Salix* spp., alders *Alnus* spp., cottonwoods *Populus* spp., and sedges Family: Cyperaceae. Annual precipitation averages 18-cm, and streamflow patterns are dominated by snowmelt runoff.

Trapper Creek is a species rich tributary to Goose Creek, which is one of the most species-rich subbasins in the Upper Snake River basin above Shoshone Falls (near Twin Falls, Idaho). The following species have been collected from Trapper Creek in the recent past: bluehead sucker *Catostomus discobolus*, mountain sucker *Catostomus platyrhynchus*, Utah sucker *Catostomus ardens*, longnose dace *Rhinichthys cataractae*, specked dace *Rhinichthys osculus*, redbelt shiner *Richardsonius balteatus*, Northern Leatherside Chub, Utah chub *Gila atraria*, mottled sculpin *Cottus bairdii*, and Paiute sculpin *Cottus beldingii* (Meyer et al. 2006; Blakney 2012; Meyer et al. 2013). Introduced brook trout *Salvelinus fontinalis* and rainbow trout *O. mykiss*, including rainbow trout×cutthroat trout hybrids have also been collected (Meyer et al. 2006). Yellowstone cutthroat trout *O. c. bouvieri* occur in other tributaries to Goose Creek, but no longer occur in Trapper Creek; however, Trapper Creek is one of only a few streams in Idaho where Northern

Leatherside Chub are regularly collected (Blakney 2012). Long-term grazing has impacted fish populations in the Goose Creek drainage (IDFG 2007).

## Methods

Fish assemblage and Northern Leatherside Chub associations with microhabitat complexity were assessed using pre-positioned areal electrofishing (Fisher and Brown 1993) in Trapper Creek where Northern Leatherside Chubs have recently been sampled (Blakney 2012). Fishes were collected using a Smith-Root, Inc. LR-24 backpack electrofisher positioned on the streambank and outfitted with an anode consisting of a nylon-coated 0.4-cm diameter braided stainless steel wire (~6-m in length) with a 28-cm diameter loop (uncoated). The anode for each sample was pre-positioned in the stream 15 minutes prior to electrofishing to allow fishes to exhibit normal behavior and habitat use. Pulsed, direct current (40 Hz, 250-350 V) was applied for approximately 10 seconds and a single person netted all immobilized fishes; care was taken to not frighten additional fish into the electrical field. Fishes were identified to species, salmonids and Northern Leatherside Chubs were measured for total length, and all fishes were released. One hundred fifty prepositioned electrofishing samples were collected among seven different areas (from 7 to 68 samples in each area) of Trapper Creek in August 2013.

Instream habitat was measured immediately after electrofishing in a 1-m diameter circle centered on the pre-positioned electrofishing anode. A 1-m diameter circle approximated the distance at which we observed fish to be immobilized and is near the immobilization range measured for other pre-positioned electrofishing arrays (e.g., Schwartz and Herricks 2004). Water depth and velocity were measured at five locations, once at the array center and at four equidistant points along the circle perimeter. Water velocity was measured twice at each point; once at the stream substratum and again at 0.6 of water depth (10 total measurements). Dominant cover was noted as: large wood (>4-m in length, >10-cm in diameter); small woody debris (<4-m in length, or <10-cm in diameter), boulder (>256-mm diameter), emergent vegetation, submerged vegetation, overhanging vegetation, undercut bank (>10-cm deep), or none. Dominant substratum was classified according to the modified Wentworth scale as: bedrock, silt/clay (<0.064-mm diameter on b-axis), sand (0.064-2-mm), gravel (2-15mm), pebble (15-64mm), cobble (64-256mm), and boulder (>256-mm) (Cummins 1962). The density of overhead cover from riparian vegetation or instream wood above the water surface was ranked from 1 (none) to 5 (dense). Channel unit type was classified as riffle, run, or pool based on water depth and velocity (Hawkins et al. 1993). The presence of secondary habitat features influencing microhabitat conditions at electrofishing locations was also noted (e.g., beaver dams, beaver ponds, riparian shrubs). Water temperature and conductivity were measured with an Oakton PCSTestr 35 multi-parameter probe (Oakton Instruments, Vernon Hills, Illinois).

Fish assemblage associations with microhabitats and microhabitat complexity were evaluated using constrained correspondence analysis (CCA; aka canonical correspondence analysis). CCA is a direct gradient analysis that uses a unimodal species model to explain variation in assemblage structure (relative abundance) using environmental variables (ter Braak 1995). Exploratory detrended correspondence analysis showed axis 1 gradient length (a measure of beta

diversity) to be 3.45, suggesting that a unimodal distribution is likely more appropriate than a linear model (e.g., redundancy analysis) (Lepš and Šmilauer 2003); CCA is also robust to the unimodal model assumptions (Palmer 1993). The species matrix used in the CCA was comprised of only sites where at least one individual of one fish species was present ( $n = 106$ ), and species abundances were un-transformed. The habitat variables included in the CCA were: mean water velocity (m/s), mean water depth (m), coefficient of variation ( $CV = 100 \cdot SD / \text{mean}$ ) in water velocity (%), SD water depth (m), substrate rank, overhead cover rank, and wood cover and boulder cover categorical variables (presence = 1, absence = 0). Water velocity and depth variables were summarized using the 10 (velocity) or 5 (depth) measurements. Water velocity and depth complexity were estimated as the CV of water velocity and SD of water depth. We wanted to use un-standardized measures of variation in velocity and depth (i.e., standard deviations) as measures of complexity, but the standard deviation of water velocity was highly correlated with mean velocity (Spearman rank  $r_s = 0.957$ ) so we used the CV instead ( $r_s = -0.551$ ). Substrate size was an ordinal ranking of the modified Wentworth substratum classes (from 1 (bedrock) to 7 (boulder) (Fore et al. 2007). Cover type was grouped into wood (wood = 1, other = 0) and boulder (boulder = 1, other = 0) classes. Significance of CCA axes (comparing the full model to reduced model) and individual habitat variables were assessed using permutation tests with 9999 permutations at  $\alpha = 0.10$ . A final model was refit using only significant variables. The CCAs were fit using the vegan package in Program R (R Core Team 2015).

Selection of microhabitats and microhabitat complexity by Northern Leatherside Chub was evaluated using a resource selection function (Manly et al. 2002). The selection function was based on the presence or absence (non-detection) of chubs at the 150 pre-positioned electrofishing samples (i.e., sampling protocol A and Design I in Manly et al. 2002). Model selection and inference was used to identify the best model, or a plausible set of models, of microhabitat selection from a candidate set of models. Candidate models were comprised of subsets of variables included in a global model containing the following predictor variables: mean velocity, mean depth, CV velocity, SD depth, overhead cover rank, substrate rank, wood cover, and boulder cover. A mean depth $\times$ overhead cover interaction term was evaluated because of the potential synergistic effect of both variables on habitat selection and as another measure of habitat complexity; Northern Leatherside Chubs have been described to inhabit pool habitats and brushy areas near streambanks (Wallace and Zaroban 2013). Additional candidate models were constructed using all subsets of variables in the global model, except that mean velocity and mean depth were included in all candidate models to estimate effect size since the Northern Leatherside Chub does not use the highest water velocities or shallowest depths we sampled (Wilson and Belk 2001; Wesner and Belk 2012). Fit of the global model to the data was evaluated using a Hosmer-Lemeshow goodness-of-fit, and predictive performance of the most plausible model was assessed using the area under the curve (AUC) of a receiver operating characteristic (ROC) curve (Hosmer and Lemeshow 2000). Candidate models were compared using Akaike's Information Criterion for small sample size (AICc), with the model having the smallest AICc value being the best, most plausible model. Candidate models within 4 AICc units of the best model were also considered plausible (Burnham and Anderson 2002). If multiple candidate models were plausible, then parameter estimates (and variances) were averaged using Akaike weights and shrinkage (Burnham and Anderson 2002; Lukacs et al.

2010). In addition to unstandardized parameter estimates, we report standardized parameter estimates (from models fit with data that were standardized with mean = 0, and standard deviation = 1) for comparison of relative effect sizes for the microhabitat variables included in plausible models. Resource selection functions were fit using the glm function with a logit link in Program R (R Core Team 2015).

Associations among microhabitat variables, and between microhabitat variables and secondary habitat features, measured at the 150 electrofishing samples were assessed independent from the species data using a principal components analysis (PCA). Two secondary features associated with electrofishing sites, beaver dams and woody riparian shrubs, were included in the PCA because we observed them to directly influence microhabitat conditions. For example, beaver dams were the source of small woody debris at some sites. The branches of woody riparian vegetation directly provided overhead cover, and, in some cases, entered the water column to contribute small wood and create complex patterns in water velocity. The associations of these secondary habitat features with microhabitat variables were simultaneously assessed in the PCA by including them as binary variables; we omitted them as candidate variables in the CCA and resource selection analyses because they were directly associated (and presumably correlated) with microhabitat features used directly by fishes (e.g., small woody, overhead cover, variance in water velocity). A scree plot was used to determine the number of meaningful principal components for interpretation. The PCA was fit using scaled and centered data (mean = 0, SD = 1) and the correlation matrix in princomp in Program R (R Core Team 2015).

## Results

We collected seven fish species among 150 pre-positioned electrofishing samples in Trapper Creek (Table 15). Water conductivities (corrected for water temperature) ranged from 250-280  $\mu\text{S}/\text{cm}$ , and water temperatures ranged from 15 to 23°C. At least one fish was collected at 106 of the samples. Speckled dace were most often collected and occurred at the most electrofishing sites (Table 15); rainbow trout were collected least often and had the lowest relative abundance across species. Northern Leatherside Chub were collected at 16% of sites, with a relative abundance of 8.6% among all fishes collected; the 47 chubs collected ranged in total length from 40 to 125 mm.

Table 15. Fishes collected during pre-positioned areal electrofishing surveys (n=150) in Trapper Creek, August 2013.

Scientific name	Common name	Percent occurrence	Relative abundance
<i>Catostomus discobolus</i>	Bluehead Sucker	22.6	16.7
<i>Cottus beldingii</i>	Paiute Sculpin	15.3	5.1
<i>Lepidomeda copei</i>	Northern Leatherside Chub	16.0	8.6
<i>Oncorhynchus mykiss</i>	Rainbow Trout	3.3	0.9
<i>Rhinichthys cataractae</i>	Longnose Dace	22.7	10.5
<i>Rhinichthys osculus</i>	Speckled Dace	48.0	40.1
<i>Richardsonius balteatus</i>	Redside Shiner	16.0	18.0

Fish assemblage structure was influenced by water depths and velocities, including complexity in both, as well as overhead cover and instream wood. The first four axes of the CCA were significant ( $P \leq 0.079$ ), and mean velocity, mean depth, SD depth, CV velocity, overhead cover, and wood were significantly related to assemblage structure, whereas substrate and boulder cover were not (Table 16). Variance inflation factor scores were 2.02 or less for each variable (Table 16), indicating a lack of correlation among constraining variables. The CCA refit using only significant ( $P < 0.10$ ) habitat variables showed longnose dace and Paiute sculpin to occur in shallow, high velocity habitats, and Northern Leatherside Chub, rainbow trout, and redbside shiner used deeper habitats (Figure 28, top panel). However, Northern Leatherside Chub, and to a lesser extent, rainbow trout, used deeper habitats with more complex water depths and velocities than did the redbside shiner (Figure 28, top panel). The Northern Leatherside Chub and bluehead sucker were two species most associated with overhead cover (Figure 28, bottom panel). The speckled dace was not strongly associated with any microhabitat feature that was measured.

Table 16. Significance ( $P$ -value) and variance inflation factor (VIF) for microhabitat variables (permutation test with 9999 permutations) included in a constrained correspondence analysis to evaluate their potential effects on fish assemblage structure.

Variable	$P$ -value	VIF
Mean velocity (m/s)	0.030	2.02
CV velocity (%)	0.023	1.78
Mean depth (m)	0.064	1.19
SD depth (m)	0.027	1.37
Substrate (rank: 1 to 7)	0.433	1.44
Overhead cover (rank: 1 to 5)	0.015	1.15
Wood (pres. = 1, abs. = 0)	0.051	1.36
Boulder (pres. = 1, abs. = 0)	0.980	1.26



Microhabitat measurements on a current seam (high flow complexity) where Northern Leatherside Chub were collected. Credit: D. Dauwalter.

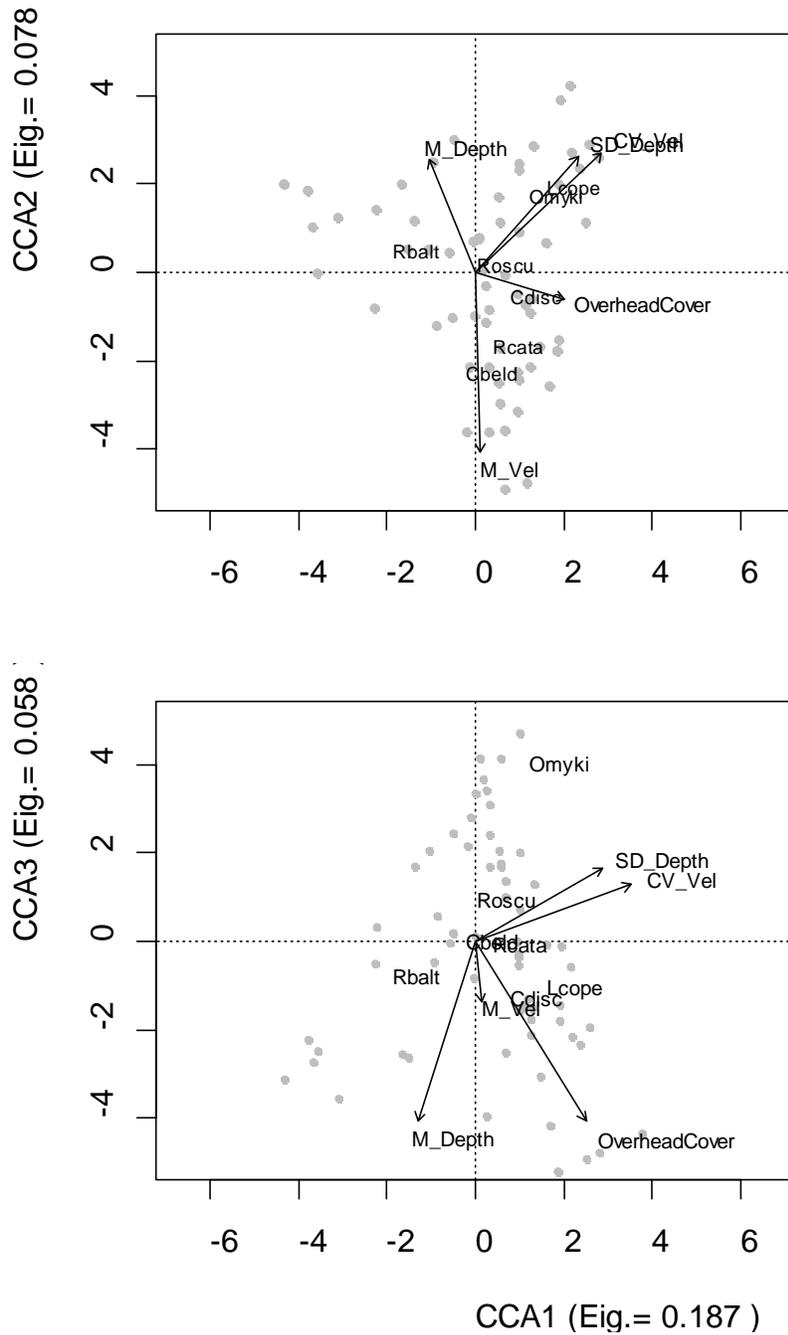


Figure 28. Biplots of CCA axes (1 versus 2, top panel; 1 versus 3, bottom panel) showing associations between fish species, continuous (arrows) and categorical (squares) microhabitat variables, and prepositioned areal electrofishing sites (gray points). Species are abbreviated as first letter of genus, and first four letters of species: Cbeld = Paiute sculpin; Cdisc = bluehead sucker; Lcope = Northern Leatherside Chub; Omyki = rainbow trout; Rbalt = redbelt shiner; Rcata = longnose dace; Roscu = speckled dace.

Northern Leatherside Chub selected deeper, slower, and more complex habitats (Table 17), which was shown by the resource selection models. The global model fit the data (Hosmer-Lemeshow test,  $\chi^2=1.66$ ,  $df = 8$ ,  $P = 0.990$ ). In all, 16 candidate models were plausible (i.e.,  $\Delta AICc \leq 4$ ;

Table 18). The most plausible model showed acceptable discrimination between used and unused sites (10-fold cross validated AUC = 0.76; sensitivity = 0.67; specificity = 0.76; Hosmer and Lemeshow 2000). Because model averaging cannot be done across models with and without interaction terms, we averaged parameter estimates over the 13 (of 16) plausible models that contained a depth $\times$ overhead cover interaction term. A model with averaged parameter estimates showed very good in-sample predictive performance ((insample AUC = 0.82; sensitivity = 0.67; specificity = 0.76; Hosmer and Lemeshow 2000). Importantly, model-averaged parameter estimates showed a complex interaction between depth and overhead cover. Selection probability increased slightly with depth when overhead cover was absent, but Northern Leatherside Chubs are almost certain to be present in deep water when overhead cover is dense (Figure 29, top panels). Chubs also selected areas with slower but more complex water velocities (Figure 29). There was some evidence of a positive effect of water depth complexity on habitat selection, and a positive effect of boulders and a negative effect of wood; however, unconditional standard errors on those parameters suggested that the magnitude, and even direction, of effect was highly uncertain (Table 19).

*Table 17. Summary statistics for microhabitat variables measured at pre-positioned electrofishing samples where Northern Leatherside Chub were present (n=24) versus absent (n=126).*

Variable	Presence	Mean	SD	Minimum	Maximum
Mean velocity (m/s)	Present	0.22	0.12	0.04	0.42
	Absent	0.29	0.17	0.01	0.74
CV velocity (%)	Present	131.4	26.4	99.0	195.7
	Absent	120.3	21.7	94.9	190.0
Mean depth (m)	Present	0.43	0.13	0.21	0.73
	Absent	0.32	0.13	0.06	0.76
SD depth (m)	Present	0.07	0.03	0.02	0.14
	Absent	0.06	0.04	0.01	0.21
Substrate (rank: 1-9)	Present	4.0	1.5	2.0	6.0
	Absent	4.0	1.5	1.0	7.0
Overhead cover (rank: 1-5)	Present	2.2	1.0	1.0	4.0
	Absent	1.6	0.9	1.0	4.0
Wood cover (pres. = 1, abs. = 0)	Present	0.5	0.5	0	1
	Absent	0.4	0.5	0	1

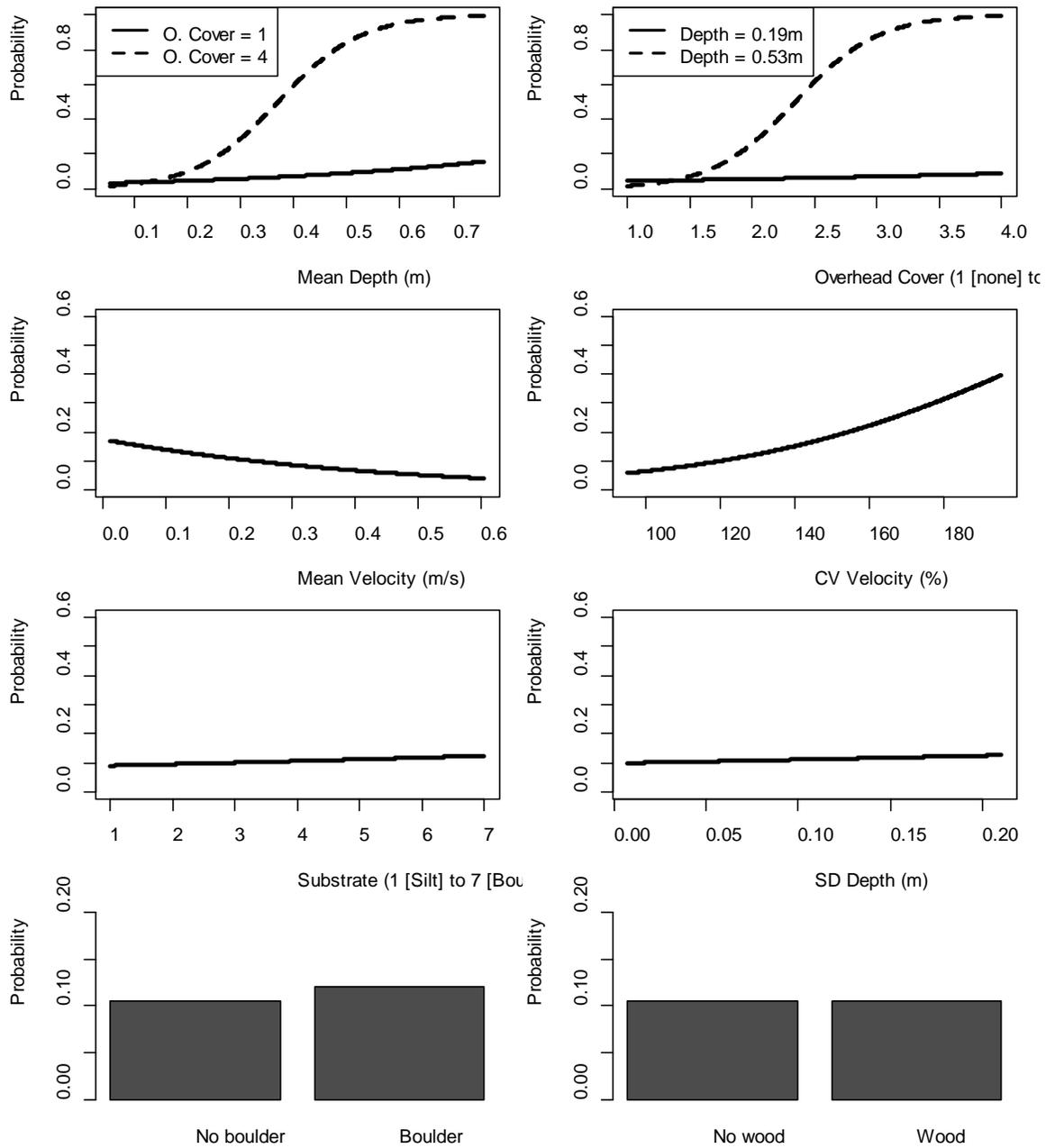


Figure 29. Habitat selection probabilities of Northern Leatherside Chub for different microhabitat variables in Trapper Creek, Idaho. All values of habitat variables were held at their mean value for prediction unless specifically varied for a panel.

Table 18. Number of parameters ( $K$ ), log-likelihood, Akaike's Information Criterion for small samples ( $AICc$ ), delta  $AIC$ , and Akaike weights ( $w_i$ ) for candidate resource selection probability function models describing habitat selection by Northern Leatherside Chub in Trapper Creek. Models with interaction terms also include main effect terms.

Candidate models	K	Log-Likelihood	AICc	$\Delta AICc$	$w_i$
Mean Depth x OverheadCover + Mean Velocity + CV Velocity	6	-50.846	114.28	0.00	0.213
Mean Depth x OverheadCover + Mean Velocity + CV Velocity + Substrate	7	-50.428	115.64	1.37	0.108
Mean Depth x OverheadCover + Mean Velocity + CV Velocity + Boulder	7	-50.568	115.92	1.65	0.094
Mean Depth x OverheadCover + Mean Velocity + CV Velocity + SD Depth	7	-50.715	116.22	1.94	0.081
Mean Depth x OverheadCover + Mean Velocity + CV Velocity + Wood	7	-50.822	116.43	2.15	0.073
Mean Depth + OverheadCover + Mean Velocity + CV Velocity	5	-53.092	116.60	2.32	0.067
Mean Depth x OverheadCover + Mean Velocity + CV Velocity + Substrate	8	-50.089	117.20	2.92	0.050
Mean Depth x OverheadCover + Mean Velocity + CV Velocity + Substrate + Boulder	8	-50.244	117.51	3.23	0.042
Mean Depth x OverheadCover + Mean Velocity	5	-53.603	117.62	3.34	0.040
Mean Depth + OverheadCover + Mean Velocity + CV Velocity + Substrate	6	-52.528	117.64	3.37	0.040
Mean Depth x OverheadCover + Mean Velocity + CV Velocity	8	-50.428	117.88	3.60	0.035
Mean Depth + OverheadCover + Mean Velocity + CV Velocity + Boulder	6	-52.658	117.90	3.62	0.035
Mean Depth x OverheadCover + Mean Velocity + CV Velocity + SD Depth + Bldr	8	-50.442	117.91	3.63	0.035
Mean Depth x OverheadCover + Mean Velocity + CV Velocity + Wood + Boulder	8	-50.568	118.16	3.88	0.031
Mean Depth x OverheadCover + Mean Velocity + SD Depth	6	-52.837	118.26	3.98	0.029
Mean Depth x OverheadCover + Mean Velocity + SD Depth + Substrate	7	-51.742	118.27	3.99	0.029



Microhabitat on current seam with high flow complexity below beaver dam (left), and microhabitat with overhead cover from bank vegetation (right) on Trapper Creek, 2013. Credit: D. Dauwalter.

The PCA showed most of the variation in microhabitats surveyed was associated with a riffle-run-pool gradient, as indicated by axis 1 (Figure 30, top left panel). Mean water depth and SD of water depth were positively related to axis 1 that explained 24.4% of the variation in habitat variables, and mean water velocity, SD velocity, and substrate size were negatively related to axis 1. The second PCA axis (18.7% of variation) represented a gradient of instream (wood and boulder) and overhead cover and showed that overhead cover was tightly coupled with riparian shrubs (Figure 30, top left panel). Shrubs and beaver dams were highly correlated with PCA axis 3 (12.1%), and showed that these features often, but not always, were associated with wood cover and complex water velocities and depths (Figure 30, bottom left panel). When Northern Leatherside Chub abundance was plotted for each in the PCA plots, it showed chubs to occur more frequently and be more abundant in deeper habitats with variable depths and more overhead cover, conditions that were often created by mature riparian shrubs and beaver dams (Figure 30, right panels). The scree plot suggested axes 1 through 3 explained the most variation among the variables included; therefore we did not interpret PCA axes 4 or higher.

*Table 19. Unstandardized and standardized unconditional parameter estimates (model averaged using shrinkage) and standard errors for a resource selection probability function for Northern Leatherside Chub in Trapper Creek, Idaho. Parameter averages were done excluding three candidate models without a depth x overhead cover interaction term. Akaike weights ( $w_i$ ) were summed across all models as a measure of variable importance.*

Variable	Unstandardized		Standardized		Sum( $w_i$ )
	$\tilde{\beta}_i$	$SE_{\tilde{\beta}_i}$	$\tilde{\beta}_i$	$SE_{\tilde{\beta}_i}$	
Intercept	-5.677	1.868	-2.119	0.222	1.00
Mean depth (m)	-1.072	3.290	0.665	0.211	1.00
Overhead cover (rank: 1 to 5)	-0.424	0.626	0.710	0.193	1.00
Mean depth x Overhead cover	3.496	1.592	0.434	0.197	0.86
Mean velocity (m/s)	-2.654	2.226	-0.354	0.297	1.00
SD depth (m)	1.331	2.375	0.048	0.086	0.17
CV velocity (%)	0.023	0.008	0.531	0.193	0.24
Substrate (rank: 1 to 7)	0.060	0.084	0.089	0.124	0.27
Wood (pres. = 1, abs. = 0)	-0.009	0.006	-0.004	0.003	0.10
Boulder (pres. = 1, abs. = 0)	0.146	0.298	0.049	0.100	0.22

## Discussion

Our study showed fish assemblage structure and habitat selection by the Northern Leatherside Chub to be influenced, in part, by habitat complexity at the microhabitat scale in our study system. Our multivariate analysis showed complex water velocities and depths to structure fish assemblages in deep-water habitats, with Northern Leatherside Chub and rainbow trout being more common where water velocities and depths were heterogeneous whereas reidside shiner occurred in deep, homogenous habitats (e.g., beaver ponds). Our habitat selection models also showed the Northern Leatherside Chub to select deep areas more often, and the species was almost certain to use deep areas when overhead cover was dense. The selection models also showed, to a lesser extent, that chubs occurred more frequently in habitats with complex water velocities and depths. As shown by the PCA, overhead cover is tightly coupled with woody riparian shrubs, and complex water depths and velocities are commonly created by both mature riparian vegetation at the water surface and beaver dams.

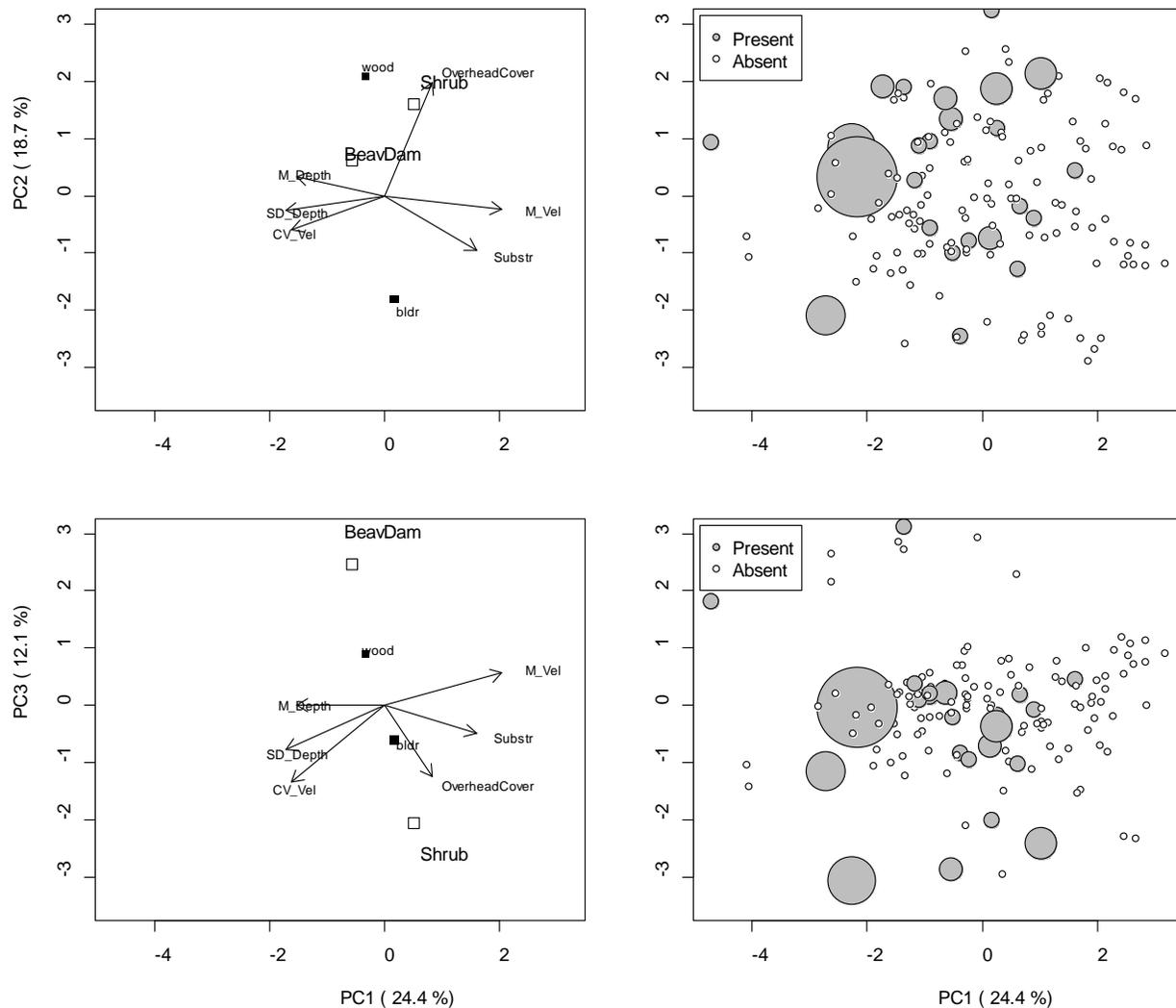


Figure 30. Biplots of PCA axes 1 versus 2 (top panels) and 1 versus 3 (bottom panels). Left panels show arrows for continuous habitat variables (scores multiplied by 4), centroids for wood and boulder cover (black) and beaver dams and woody shrubs (white). Right panels show site scores with symbol size scaled by number of Northern Leatherside Chub collected (Gray = 1 to 9 individuals; white = absent).

The selection of microhabitats by Northern Leatherside Chub in our study confirmed the general habitat use patterns shown by other studies, but our study also highlights the role of complexity in microhabitat selection. Wilson and Belk (2001) found that Northern Leatherside Chubs in Trapper Creek selected slower water velocities but found no effect of water depth, substrate, instream cover (aquatic vegetation, wood, undercut banks), overhead cover, or surface turbulence on selection. In experimental tanks, the Northern Leatherside Chub selected small cobbles for reproduction (Billman et al. 2008a). While no other studies of microhabitat selection for Northern Leatherside Chub have been conducted, the closely related Southern Leatherside Chub *L. aliciae* (Johnson et al. 2004) has been found to select slow water velocities and coarse substrates (Wilson and Belk 2001) and deep, silty habitats without brown trout *Salmo trutta* (Walser et al. 1999). At the reach scale, the presence of Northern Leatherside Chubs in a stream

reach was positively correlated with more pool, backwater, and side-channel habitats, and more understory woody and non-woody vegetation in the Bear River basin, and abundance was positively associated with pool density and amount of coarse substrate (Wesner and Belk 2012). However, these studies did not evaluate the influence of habitat complexity, either measured as the variation in a habitat variable or the interaction between two or more variables, on microhabitat selection, occurrence, or abundance of Leatherside Chubs at the reach scale. The fish assemblage in our study system showed clear associations with complex water velocities and depths, and Northern Leatherside Chub showed a synergistic response to overhead cover and water depth. Habitat complexity has been a major theme in stream ecology and restoration for over a decade and has been shown to influence microhabitat selection of stream fishes (Dauwalter and Fisher 2007; Ayllón et al. 2009) and fish occurrence, abundance, and diversity at the reach scale (Gorman and Karr 1978). However, complexity is still not routinely evaluated in studies of fish habitat selection, occupancy, and abundance.

The extent to which our results can be generalized to other streams and seasons requires further evaluation because we only conducted our study in one stream during one summer. However, Trapper Creek is one of a few remaining streams in the Upper Snake River basin where the Northern Leatherside Chub is consistently collected, and the general patterns of habitat use we observed match other qualitative descriptions (Wallace and Zaroban 2013; J. Wesner, University of South Dakota, personal communication). Ayllón et al. (2010) showed that brown trout routinely selected deep, slow habitats across several stream systems but that selection of a specific cover type varied by stream. The authors suggested that selection of specific habitats can vary based on habitat availability that is driven by environmental conditions set at larger spatial scales (e.g., watershed). Kanno et al. (2012) also found that microhabitat selection by the spotfin chub *Erimonax monachus* varied spatially as habitat availability changed with stream size and changed by season due to water temperatures. Future studies of the Northern Leatherside Chub and co-occurring species should focus on understanding the influence of micro- and macro-habitats, including the role of complexity, across space and time to assess the generality of our results as they relate to the species' life history requirements and those of other sympatric species. If our findings do apply beyond Trapper Creek, they suggest that degraded riparian conditions from livestock grazing and beaver-trapping and eradication activities may have reduced the availability of complex habitats and negatively influenced the distribution of the Northern Leatherside Chub rangewide, as purported in other studies (Zafft et al. 2009; Blakney 2012).

The complex microhabitats in our study system were also associated with other components of stream systems – mature riparian vegetation and beaver dams – that are often a focus of stream restoration. Riparian vegetation is often negatively impacted by cattle grazing and can lead to wide, shallow, and simple stream channels with high sedimentation levels, little recruitment of wood, and low terrestrial invertebrate inputs that are consumed by fishes (Platts and Nelson 1985; Saunders and Fausch 2009). Many restoration projects focus on excluding cattle from riparian areas through fencing or limit their impact by use of rotational, short-term, or seasonal grazing regimes (Roni et al. 2008), which can increase instream physical habitat complexity, terrestrial food subsidies, and fish population biomass (Saunders and Fausch 2009). The Goose Creek watershed, one of the few remaining watersheds where the Northern Leatherside Chub occurs in Idaho (Meyer et al. 2013; Wallace and Zaroban 2013), has extensive grazing impacts to

riparian areas (IDFG 2007), and our study suggests that restoration of mature riparian vegetation in the watershed may provide important habitat for the Northern Leatherside Chub in tributaries other than Trapper Creek.

Active and abandoned beaver dams sometimes, but not always, provided the microhabitat complexity used by fishes in our study system. Beaver dams can create discontinuities in riverine systems that alter physical habitat and biogeochemical processes over decadal time scales (Burchsted et al. 2010). Beaver dams can impede riverine connectivity important to the viability of fish populations (Collen and Gibson 2001). However, beaver-influenced stream systems can act as refuge areas during extreme drought conditions because of their ability to retain water (White and Rahel 2008), and beaver have been used to restore incised stream channels (Pollock et al. 2014). Beaver dams are common in Trapper Creek where the Northern Leatherside Chub is most abundant in Idaho, and we collected individuals at some dams during our study. The species has also been collected at beaver dams, including abandoned dams, in tributaries to the Salt River near the Idaho-Wyoming border (J. Blakney, Idaho State University, personal communication). Reduced abundance of beaver and beaver dams, and subsequent loss of habitat complexity associated with beaver dams, throughout the range of Northern Leatherside Chub has been cited as a potential reason for the species' decline (Blakney 2012). This suggests that beaver reintroduction may be a simple and cost effective restoration strategy for Northern Leatherside Chub as long as suitable food resources for beaver are available at reintroduction sites (Beck et al. 2010).

Habitat complexity has been documented as an important determinant of habitat use, abundance, and diversity in stream biota for over 30 years (e.g., Gorman and Karr 1978), but it is still not routinely evaluated in studies of fish habitat quality. We argue that habitat complexity should routinely be assessed in studies evaluating fish-habitat relationships and developing models useful to practitioners managing and restoring streams and watersheds. For example, transect-based stream habitat survey methods have been used for over three decades, and the data collected in that way are amenable to quantifying habitat complexity (Arend and Bain 1999; Fisher et al. 2012). Instead, data are routinely summarized using measures of central tendency (means and median). We argue that additional measures of complexity (e.g., variance, diversity) should also be quantified using those same data. This would allow formal evaluation of hypotheses regarding the effects of stream habitat complexity on stream fishes (including re-analysis of data from past studies) and would provide more informative models for use by managers in an era of unprecedented habitat alteration and expenditures toward stream restoration (Bernhardt et al. 2005; Kovalenko et al. 2012).

### **Supplemental Resources:**

Webpage: [www.tu.org/microhabitat-selection](http://www.tu.org/microhabitat-selection)

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# APPENDIX A. SUPPLEMENTAL INFORMATION FOR GOOSE CREEK FISH SURVEYS

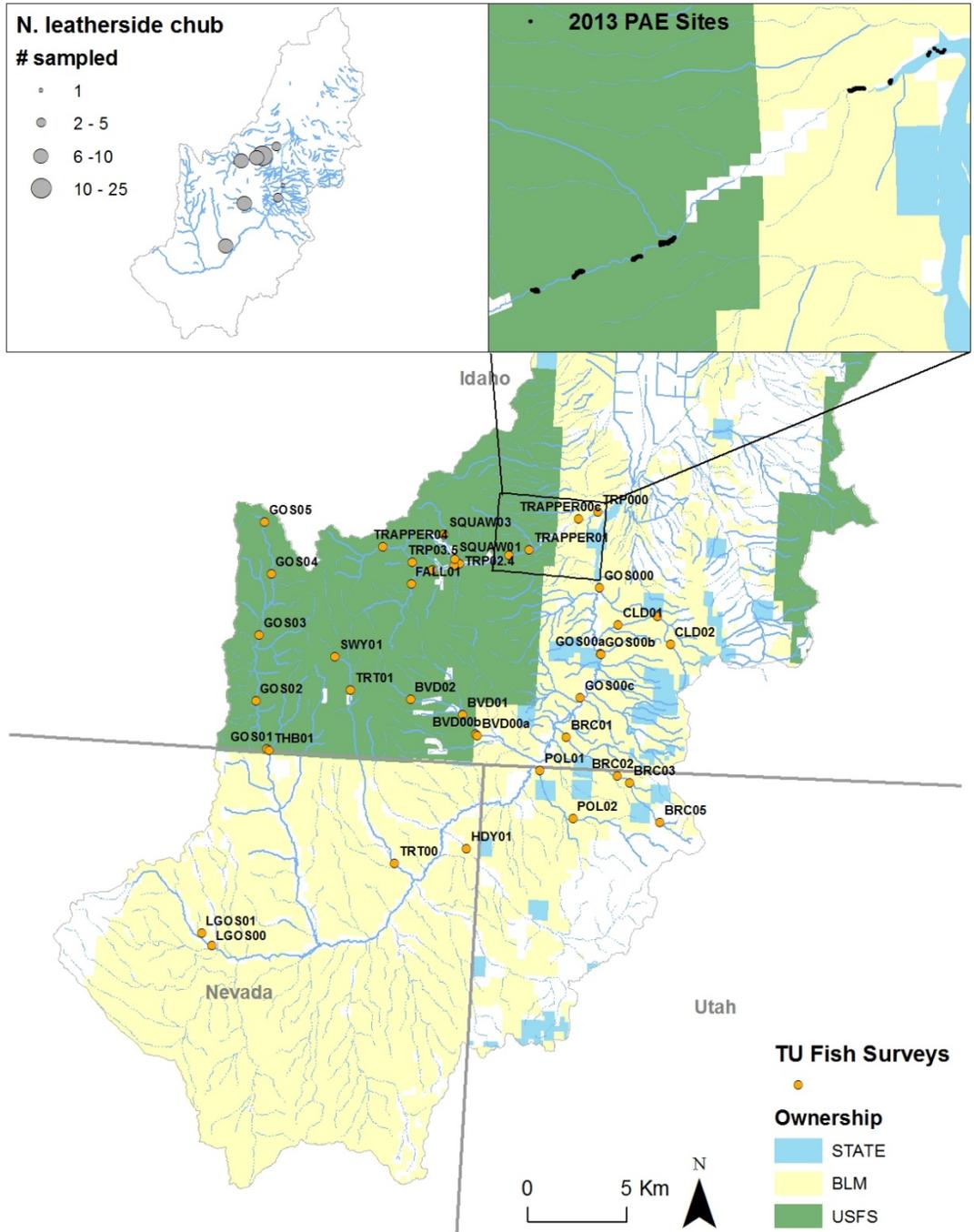


Figure 31A. Fish sampling locations (n = 41) surveyed from 2013 to 2015 in the Goose Creek watershed upstream of Goose Creek Reservoir. Left inset map shows locations where northern leatherside chub were collected, 2013 to 2015. Site labels correspond to SiteID in Table 6. Right inset shows location of pre-positioned areal electrofishing (PAE, n = 150) samples in Trapper Creek, August 2013.

Table 20A. Descriptions of electrofishing sampling locations (n = 41) in the Goose Creek watershed, 2013 to 2015. Datum for UTM coordinates is North American Datum 83.

SiteID	Stream	Date	Easting	Northing	UTM Zone	Reach length (m)
BRC01	Birch Creek	7/27/2014	254978	4656097	12	100
BRC02	Birch Creek	7/24/2014	259131	4653071	12	75
BRC03	Birch Creek	9/9/2014	260136	4652525	12	100
BRC05	Birch Creek	9/10/2014	262687	4649395	12	100
BVD00a	Beaverdam Creek	10/6/2015	744577	4655802	11	47
BVD00b	Beaverdam Creek	10/6/2015	744441	4655906	11	100
BVD01	Beaverdam Creek	8/21/2014	743242	4657348	11	100
BVD02	Beaverdam Creek	8/21/2014	738899	4658225	11	62
CLD01	Cold Creek	7/23/2014	258966	4665271	12	100
CLD01.5	Cold Creek	8/13/2014	262102	4666024	12	100
CLD02	Cold Creek	7/23/2014	263205	4663780	12	97
FALL01	Fall Creek	8/7/2014	738091	4667519	11	70
GOS000	Goose Creek	10/5/2015	257389	4668247	12	106
GOS00a	Goose Creek	9/3/2014	257540	4662920	12	100
GOS00b	Goose Creek	9/3/2014	257606	4662854	12	100
GOS00c	Goose Creek	9/4/2014	255997	4659332	12	100
GOS01	Goose Creek	7/15/2014	727714	4653177	11	133
GOS02	Goose Creek	7/16/2014	726466	4656931	11	70
GOS03	Goose Creek	7/16/2014	726253	4662287	11	113
GOS04	Goose Creek	7/18/2014	726755	4667272	11	58
GOS05	Goose Creek	8/28/2015	725828	4671391	11	100
HDY01	Hardesty Creek	10/8/2014	744549	4646606	11	100
LGOS00	Little Goose Creek	8/14/2014	724723	4636884	11	100
LGOS01	Little Goose Creek	8/11/2014	723821	4637819	11	100
POL01	Pole Creek	9/11/2014	252879	4653337	12	100
POL02	Pole Creek	9/11/2014	255633	4649550	12	100
SQUAW01	Squaw Creek	8/13/2013	741414	4669856	11	100
SQUAW03	Squaw Creek	8/13/2013	740395	4671738	11	100
SWY01	Swanty Creek	8/20/2014	732500	4661060	11	100
THB01	Thoroughbred Creek	7/15/2014	727898	4653058	11	75
TRAPPER00c	Trapper Creek	8/14/2013	255554	4673760	12	85
TRAPPER01	Trapper Creek	8/14/2013	747354	4671162	11	100
TRAPPER02	Trapper Creek	8/13/2013	745747	4670541	11	100
TRAPPER03a	Trapper Creek	8/18/2013	739674	4668780	11	100
TRAPPER04	Trapper Creek	8/12/2013	735538	4670293	11	50
TRP000	Trapper Creek	8/13/2014	257124	4674401	12	100
TRP02.4	Trapper Creek	10/9/2014	741847	4669479	11	94.5
TRP02.5	Trapper Creek	8/12/2014	741413	4669325	11	100
TRP03.5	Trapper Creek	8/12/2014	738028	4669242	11	100
TRT00	Trout Creek	8/14/2014	738817	4644867	11	100
TRT01	Trout Creek	7/17/2014	733967	4658522	11	120

Table 21A. Number of individuals collected during electrofishing in the Goose Creek drainage, 2013 to 2015. All fishes were released after capture.

Site ID	<i>C. ardens</i>	<i>C. discobolus</i>	<i>C. beldingii</i>	<i>C. bairdii</i>	<i>L. copei</i>	<i>O. clarkii</i>	<i>O. mykiss</i>	<i>P. flavescens</i>	<i>R. cataractae</i>	<i>R. osculus</i>	<i>R. balteatus</i>	<i>S. fontinalis</i>
BRC01#										55		5
BRC02#												83
BRC03#												69
BRC05#												92
BVD00a*												
BVD00b*												
BVD01			2		10							
BVD02*												
CLD01#							11					3
CLD01.5							2					5
CLD02#*												
FALL01			57				2					2
GOS000		69	24				5	2	80	56	151	
GOS00a	4	13	34				1		18	60	96	
GOS00b	2	35	46		1		1		23	74	103	
GOS00c		25	13		4				17	54	100	
GOS01			23			1			19	77	26	11
GOS02#			56			3				41	12	10
GOS03#			23									4
GOS04#			15			2				21	74	
GOS05			19			17						
HDY01*												
LGOS00						1				244		
LGOS01#										56		
POL01		1								48		3
POL02#												37
SQUAW01#										11		
SQUAW03#*												
SWY01*												
THB01			9	1		4			6	52	56	5
TRP000		1	25				1		5	4		
TRAPPER00c	1	46	40		2		1		32	53	69	
TRAPPER01#		24	1		22		7		35	57		
TRAPPER02#		15	30		10		2		35	22		
TRP02.4#		1	35				21		17	12		
TRP02.5		8			10		12		29	120		
TRAPPER03a		9	57		1		81	9	16	71		
TRP03.5			80				1					6
TRAPPER04#												29
TRT00#		13			7					207	1	
TRT01#						53				14		

\*no fishes were collected.; #IDFG Native Salmonid Assessment site.

Table 22A. Trout density estimates (#/100-m<sup>2</sup> ± 1 SE) for two size classes (<100mm, and ≥100mm TL) from multiple-pass electrofishing at sites in the Goose Creek drainage, 2013 to 2015.

Site ID	Brook Trout		Rainbow Trout		Cutthroat Trout	
	<100mm	≥100mm	<100mm	≥100mm	<100mm	≥100mm
BRC01	3.16 (0.00)	2.11 (0.00)				
BRC02	148.77 (6.30)	29.47 (0.06)				
BRC03	86.67 (0.90)	10.00 (0.29)				
BRC05	51.46 (0.42)	4.85 (0.54)				
BVD00a						
BVD00b						
BVD01						
BVD02						
CLD01	2.22 (0.00)	1.11 (0.01)		13.33 (0.07)		
CLD01.5	4.58 (0.29)		0.65 (0.04)	1.31 (0.00)		
CLD02						
FALL01	2.92 (0.39)			1.46 (0.00)		
GOS000			0.92 (0.15)	1.22 (0.16)		
GOS00a				0.37 (0.07)		
GOS00b				0.17 (0.00)		
GOS00c						
GOS01	2.09 (0.08)				0.15 (0.01)	0.15 (0.17)
GOS02	5.60 (1.41)	1.75 (0.04)			2.45 (0.65)	0.35 (0.01)
GOS03	9.04 (2.82)	0.24 (0.13)				0.24 (0.13)
GOS04						2.01 (0.00)
GOS05					1.22 (0.23)	14.63 (0.46)
HDY01						
LGOS00					0.86 (0.00)	
LGOS01						
POL01	4.62 (0.00)	2.31 (0.00)				
POL02	20.20 (0.34)	24.24 (0.39)				
SQUAW01						
SQUAW03						
SWY01						
THB01	3.53 (0.13)	1.41 (0.07)			2.12 (0.08)	1.41 (0.07)
TRAPPER00c				0.24 (0.00)		
TRAPPER01			2.80 (0.14)	0.31 (0.00)		
TRAPPER02			0.28 (0.00)	0.28 (0.00)		
TRAPPER03a			15.60 (1.81)	13.89 (0.52)		
TRAPPER04	13.33 (12.51)	53.33 (3.61)				
TRP000				0.31 (0.00)		
TRP02.4			0.60 (0.11)	8.34 (0.27)		
TRP02.5				4.59 (0.11)		
TRP03.5		2.25 (0.00)		0.37 (0.00)		
TRT00						
TRT01					11.33 (0.30)	12.68 (0.49)

Table 23A. Descriptions of sites sampled for fish, macroinvertebrates, and instream and riparian habitat as part of a riparian exclosure evaluation in the Goose Creek Allotment, 2015. Treatment indicates whether site was inside or outside of the riparian exclosure. An x in the Inverts column indicates whether macroinvertebrates were collected

Site ID	Exclosure/Treatment	Stream	Date	Easting	Northing	UTM Zone	Length of reach (m)	Inverts
CLBK_In1	Coal Banks - Inside	Goose Creek	7/13/2015	256153	4659521	12	50	
CLBK_In2	Coal Banks - Inside	Goose Creek	7/14/2015	256123	4659500	12	50	x
CLBK_In3	Coal Banks - Inside	Goose Creek	7/14/2015	256053	4659426	12	50	
CLBK_In4	Coal Banks - Inside	Goose Creek	7/14/2015	256004	4659330	12	50	x
CLBK_Out1	Coal Banks - Outside	Goose Creek	7/14/2015	255976	4659236	12	50	x
CLBK_Out2	Coal Banks - Outside	Goose Creek	7/14/2015	256015	4659219	12	50	
EMRY_In1	Emery - Inside	Emery Creek	8/5/2015	260202	4659950	12	50	x
EMRY_In2	Emery - Inside	Emery Creek	8/5/2015	260353	4659859	12	50	
EMRY_In3	Emery - Inside	Emery Creek	8/5/2015	260424	4659809	12	50	
EMRY_In4	Emery - Inside	Emery Creek	8/6/2015	260599	4659785	12	50	
EMRY_Out1	Emery - Outside	Emery Creek	8/4/2015	259798	4659986	12	50	x
EMRY_Out2	Emery - Outside	Emery Creek	8/4/2015	259854	4659992	12	50	x
EMRY_Out3	Emery - Outside	Emery Creek	8/4/2015	259907	4659982	12	50	x
EMRY_Out4	Emery - Outside	Emery Creek	8/5/2015	260045	4659969	12	50	x
GOCR_In1	Goose Creek - Inside	Goose Creek	8/11/2015	257613	4662837	12	50	x
GOCR_In2	Goose Creek - Inside	Goose Creek	8/11/2015	257618	4662790	12	50	x
GOCR_In3	Goose Creek - Inside	Goose Creek	8/27/2015	257650	4662615	12	50	x
GOCR_In4	Goose Creek - Inside	Goose Creek	8/12/2015	257526	4662316	12	50	x
GOCR_Out1	Goose Creek - Outside	Goose Creek	8/10/2015	257500	4662875	12	50	x
GOCR_Out2	Goose Creek - Outside	Goose Creek	8/11/2015	257540	4662920	12	50	x
GOCR_Out3	Goose Creek - Outside	Goose Creek	8/12/2015	257537	4662251	12	50	x
GOCR_Out4	Goose Creek - Outside	Goose Creek	8/26/2015	257514	4662196	12	50	x
LBCH_In1	Little Birch - Inside	Little Birch Creek	6/30/2015	254600	4656506	12	50	x
LBCH_In2	Little Birch - Inside	Little Birch Creek	6/30/2015	254807	4656307	12	50	x
LBCH_In3	Little Birch - Inside	Little Birch Creek	6/30/2015	256638	4654658	12	50	x
LBCH_In4	Little Birch - Inside	Little Birch Creek	7/8/2015	257725	4654310	12	50	x
LBCH_Out1	Little Birch - Outside	Little Birch Creek	6/30/2015	254444	4656694	12	50	x
LBCH_Out2	Little Birch - Outside	Little Birch Creek	7/8/2015	257952	4654147	12	50	x
LBCH_Out3	Little Birch - Outside	Little Birch Creek	7/8/2015	258046	4654044	12	50	x
LBCH_Out4	Little Birch - Outside	Little Birch Creek	7/15/2015	258176	4653886	12	50	x
LCLD_In0	Lower Cold - Inside	Cold Creek	7/15/2015	257908	4664768	12	39	x
LCLD_In1	Lower Cold - Inside	Cold Creek	6/29/2015	257935	4664799	12	50	x
LCLD_In2	Lower Cold - Inside	Cold Creek	6/29/2015	257999	4664839	12	50	x
LCLD_Out1	Lower Cold - Outside	Cold Creek	6/25/2015	259247	4665371	12	50	x
LCLD_Out2	Lower Cold - Outside	Cold Creek	6/25/2015	260456	4665587	12	50	x
LCLD_Out3	Lower Cold - Outside	Cold Creek	6/25/2015	260637	4665701	12	50	x
LCLD_Out4	Lower Cold - Outside	Cold Creek	6/25/2015	261188	4665819	12	50	x
STATE_In1	Stateline - Inside	Little Birch Creek	7/10/2015	259094	4653053	12	50	x
STATE_In2	Stateline - Inside	Little Birch Creek	7/10/2015	259364	4653014	12	50	x
STATE_In3	Stateline - Inside	Little Birch Creek	7/10/2015	259683	4652980	12	50	x
STATE_In4	Stateline - Inside	Little Birch Creek	7/10/2015	259806	4652955	12	50	x
STATE_Out1	Stateline - Outside	Little Birch Creek	7/15/2015	258410	4653488	12	50	x
STATE_Out2	Stateline - Outside	Little Birch Creek	7/9/2015	258433	4653432	12	50	x
STATE_Out3	Stateline - Outside	Little Birch Creek	7/9/2015	258570	4653300	12	50	x
STATE_Out4	Stateline - Outside	Little Birch Creek	7/9/2015	258885	4653119	12	50	x
UCLD_In1	Upper Cold - Inside	Cold Creek	6/24/2015	261538	4666015	12	50	x
UCLD_In2	Upper Cold - Inside	Cold Creek	6/24/2015	261705	4666039	12	45	x
UCLD_In3	Upper Cold - Inside	Cold Creek	6/24/2015	261826	4666046	12	50	x
UCLD_In4	Upper Cold - Inside	Cold Creek	6/24/2015	261896	4666041	12	50	x
UCLD_Out1	Upper Cold - Outside	Cold Creek	6/22/2015	262194	4665978	12	50	x
UCLD_Out2	Upper Cold - Outside	Cold Creek	6/22/2015	262303	4665925	12	50	x
UCLD_Out3	Upper Cold - Outside	Cold Creek	6/23/2015	262562	4665712	12	50	x
UCLD_Out4	Upper Cold - Outside	Cold Creek	6/23/2015	262747	4665547	12	50	x

Table 24A. Number of individuals by species collected during electrofishing in the Goose Creek enclosure evaluation sites sampled in 2015. All fishes were released after capture.

Site ID	<i>C. ardens</i>	<i>C. discobolus</i>	<i>C. beldingii</i>	<i>L. copei</i>	<i>O. clarkii</i>	<i>O. mykiss</i>	<i>R. cataractae</i>	<i>R. osculus</i>	<i>R. balteatus</i>	<i>S. fontinalis</i>
CLBK_In1		2					1	8	12	
CLBK_In2		12	5	3			3	30	28	
CLBK_In3		6	4	1			1	21	26	
CLBK_In4		2	1				4	26	20	
CLBK_Out1		1	2	1			26	11	12	
CLBK_Out2		9	3			1		73	34	
EMRY_In1										
EMRY_In2										
EMRY_In3										
EMRY_In4										
EMRY_Out1										
EMRY_Out2										
EMRY_Out3										
EMRY_Out4										
GOCR_In1		1	10		1		7	38	23	
GOCR_In2	3	5	3					25	32	
GOCR_In3		7	32		1		15	33	51	
GOCR_In4	2	3	15				12	36	32	
GOCR_Out1		9	13				11	23	17	
GOCR_Out2		4	4					16	34	
GOCR_Out3		2	35		1		32	136	22	
GOCR_Out4		3	15		1		40	79	27	
LBCH_In1								75		2
LBCH_In2								50		2
LBCH_In3										2
LBCH_In4										5
LBCH_Out1					3			88		4
LBCH_Out2										6
LBCH_Out3										4
LBCH_Out4										4
LCLD_In0										2
LCLD_In1					2					6
LCLD_In2					1					3
LCLD_Out1					14					9
LCLD_Out2					3					1
LCLD_Out3										1
LCLD_Out4					1					1
STATE_In1										18
STATE_In2										3
STATE_In3										6
STATE_In4										22
STATE_Out1										2
STATE_Out2										8
STATE_Out3										4
STATE_Out4										2
UCLD_In1					3					1
UCLD_In2					1					
UCLD_In3					6					
UCLD_In4					8					
UCLD_Out1					12					1
UCLD_Out2					9					2
UCLD_Out3					12					3
UCLD_Out4					8					5

Table 25A. Trout density estimates (#/100-m<sup>2</sup> ± 1SE) for two size classes (<100mm, and ≥100mm TL) from multiple-pass electrofishing at site inside and outside of riparian exclosures in the Goose Creek Allotment, 2015.

Site ID	Brook trout		Rainbow trout		Cutthroat trout	
	<100mm	>100mm	<100mm	>100mm	<100mm	>100mm
CLBK_In1						
CLBK_In2						
CLBK_In3				0.51 (0.38)		
CLBK_In4						
CLBK_Out1						
CLBK_Out2				0.35 (0.00)		
EMRY_In1						
EMRY_In2						
EMRY_In3						
EMRY_In4						
EMRY_Out1						
EMRY_Out2						
EMRY_Out3						
EMRY_Out4						
GOCR_In1						0.30 (0.00)
GOCR_In2				0.92 (0.21)		
GOCR_In3				2.68 (0.43)		
GOCR_In4						
GOCR_Out1						
GOCR_Out2						
GOCR_Out3			0.29 (0.00)			
GOCR_Out4						
LBCH_In1	6.96 (2.55)	1.74 (0.00)				
LBCH_In2	3.72 (0.00)					
LBCH_In3	3.67 (0.00)					
LBCH_In4	4.65 (0.00)	6.98 (0.00)				
LBCH_Out1	5.48 (0.00)			4.11 (0.00)		
LBCH_Out2	10.81 (0.96)	8.65 (0.44)				
LBCH_Out3	6.45 (0.57)	4.30 (0.00)				
LBCH_Out4	8.51 (0.44)	2.13 (0.00)				
LCLD_In0	14.83 (1.73)	2.12 (2.46)				
LCLD_In1	15.58 (0.80)			3.46 (0.00)		
LCLD_In2	5.17 (0.00)	1.72 (0.33)		1.72 (0.33)		
LCLD_Out1	17.81 (0.93)	1.37 (0.01)		20.55 (0.20)		
LCLD_Out2		1.12 (0.00)		3.37 (0.00)		
LCLD_Out3		1.27 (0.00)				
LCLD_Out4		1.23 (0.00)		1.23 (0.00)		
STATE_In1	36.23 (2.85)	13.04 (0.38)				
STATE_In2		12.97 (2.22)				
STATE_In3	3.45 (0.66)	13.79 (1.33)				
STATE_In4	28.99 (4.76)	36.23 (2.70)				
STATE_Out1		3.53 (0.31)				
STATE_Out2	7.41 (1.79)	11.11 (0.00)				
STATE_Out3		5.76 (0.00)				
STATE_Out4	1.32 (0.00)	1.32 (0.00)				
UCLD_In1		1.36 (0.15)		6.80 (0.75)		
UCLD_In2				2.37 (0.00)		
UCLD_In3				8.22 (0.00)		
UCLD_In4				15.71 (0.66)		
UCLD_Out1		1.06 (0.02)		14.80 (0.26)		
UCLD_Out2		2.53 (0.06)	1.26 (0.00)	13.91 (0.35)		
UCLD_Out3		5.41 (0.17)		18.38 (0.59)		
UCLD_Out4		6.45 (0.00)		10.32 (0.00)		

