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Response of Riparian Vegetation, Instream Habitat, and Aquatic Biota to Riparian Grazing Exlosures

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Abstract

Improper riparian grazing can alter riparian vegetation and reduce streambank stability, negatively impacting aquatic habitat and biota. We evaluated differences in riparian and instream habitat, benthic macroinvertebrates, and fish inside versus outside seven riparian exclosures constructed from 1982 to 2005 in Idaho. The normalized difference vegetation index from Landsat imagery (1985 to 2015) showed significant increases in riparian vegetation productivity after some but not all exclosures were constructed. Field data collected in 2015 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, a benthic macroinvertebrate index, fish species richness, and adult salmonid density showed no response. Higher age-0 salmonid 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, such as increased riparian vegetation, reduced streambank alteration, and improved channel stability, but may not always influence aquatic biota that commonly reflect larger watershed-scale processes and conditions. Therefore, well-maintained riparian exclosures should be used to eliminate livestock use in localized areas of stream corridors that are sensitive to disturbance. To improve stream health at the larger watershed scale that is relevant to most aquatic biota, exclosures should only be one of several grazing management tools implemented simultaneously to reduce the impacts of improper livestock grazing.

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 destabilize streambanks, leading to wide, shallow, and incised stream channels with low physical habitat complexity and

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Received June 13, 2018; accepted August 23, 2018

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 (Bayley and Li 2008; Herbst et al. 2012).

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 re-grazing) 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; Goss and Roper 2018). Off-stream water, employing riders to encourage livestock movement, 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, overgrazed 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 can 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 northeastern Oregon 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 fish may respond to exclosures whereas large adults that require larger expanses of habitat beyond that exclosed often show no response (Bayley and Li 2008).

Our goal was to evaluate the response of woody riparian vegetation, instream habitat, and aquatic biota to riparian exclosures in a grazing allotment in south-central Idaho. 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 macroinvertebrate, 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.—Our study occurred in the Goose Creek Group Allotment, which is located in Cassia County, Idaho, 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 main stem to the west. Major streams in the allotment are Goose Creek main stem (1,500 m in elevation) and the tributaries Cold Creek, Emery Creek, and Little Birch Creek (up to 2,000 m in elevation; Figure 1). All streams flow through sagebrush steppe in lower elevations, juniper *Juniperus* spp. woodlands in middle elevations, and mixed forests comprised of Douglas fir *Pseudotsuga menziesii* and quaking aspen *Populus tremuloides* at the 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 nonpoint sources, and total maximum daily loads have been developed for the subbasin (IDEQ 2010a, 2010b).

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

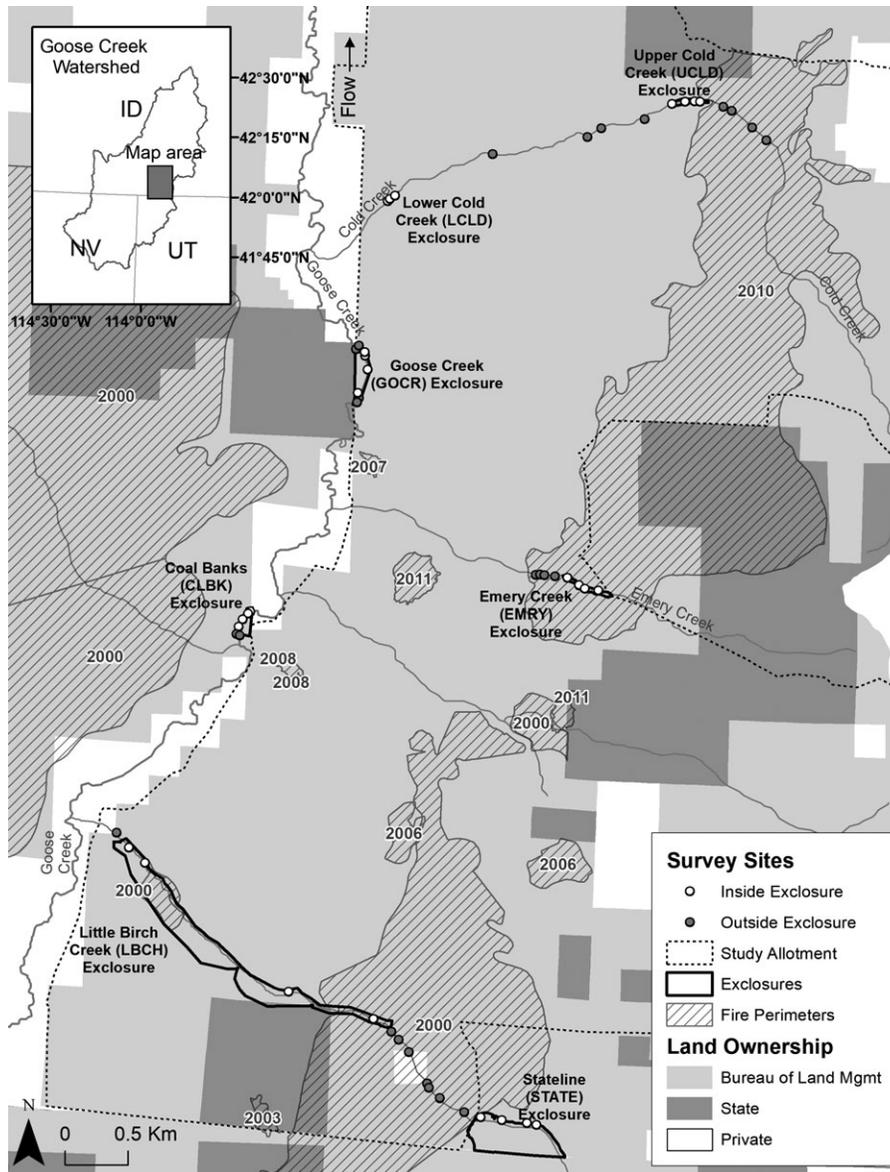


FIGURE 1. Map of Goose Creek Group Allotment, study exclosures, and field survey sites sampled for riparian and instream habitat, aquatic macroinvertebrates, and fish in 2015. Only perimeters from fires since 2000 are shown.

pasture-to-pasture movement of livestock across rough topography and through dense upland vegetation (e.g., *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 implemented, specifically regarding the management of riparian areas since annual changes can be made to the grazing system, based on 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 with crested wheatgrass *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 1; Figure 1).

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. We strove to randomly select four sites on the streams inside each of the seven exclosures in the study allotment and four sites outside each exclosure (eight sites per exclosure). Because hydrography datasets (e.g., National Hydrography Dataset) may not accurately represent stream locations, we digitized streams using National Agriculture Imagery Program imagery in ArcGIS software (ESRI, Redlands, California). Sites were randomly located on streams using the Create Random Points ArcGIS tool (ESRI). Site selection was limited to public lands; when possible, sites were constrained to be no closer than 100 m to each other. However, space limitations precluded reaching our sample size and site spacing goals for our sampling design (Figure 1). The two exclosures on the Goose Creek main stem (GOCR and CLBK) enclosed almost all Bureau of Land Management lands encompassing the Goose Creek main stem. As a result, only two sites outside the Coal Banks 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. Therefore, only 53 sites

were sampled (27 inside, 26 outside). This paired study design with replication facilitated both allotment-wide and exclosure-specific analyses and inferences, and we discuss the limitations of implementing an ideal sampling design in the Discussion.

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 before and after exclosures were constructed (Table 1). The 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, the NDVI is effectively a measure of greenness that represents vegetation productivity (Pet-torelli et al. 2005). Woody riparian vegetation often has higher NDVI values and, therefore, the 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 from 1985 to 2015 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,

TABLE 1. Riparian exclosures evaluated in Goose Creek Group Allotment, Cassia County, Idaho.

Exclosure	Stream	Year built	Function	Stream length enclosed (m)	Area enclosed (ha)	Comments
Coal Banks (CLBK)	Goose Creek	1996	Exclosure	440	4.7	Former recreational site
Goose Creek (GOCR)	Goose Creek	1988	Exclosure	675	8.8	Expanded in 1995
Upper Cold Creek (UCLD)	Cold Creek	1988	Exclosure	525	2.4	Considerable maintenance in 2014
Lower Cold Creek (LCLD)	Cold Creek	1982	Exclosure	180	0.5	
Emery Creek (EMRY)	Emery Creek	2003	Exclosure	700	3.1	Structural integrity an issue, cattle use apparent
Little Birch Creek (LBCH)	Little Birch Creek	2005	Riparian pasture	7,530	89.4	5 d use in October
Stateline (STATE)	Little Birch Creek	1999	Riparian pasture	1,680	45.1	Pasture not used in rotation since 1999

and 8 surface reflectance archive and summarize the 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 from June 22 to August 27, 2015. All sites associated with an exclosure were sampled within a short time period (<1 week) because all analyses (see below) were focused on evaluating relative differences in riparian vegetation, instream habitat, and aquatic biota at sites inside versus outside each exclosure. Each 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. Tributary stream reaches were all <1.9 m in wetted width, which meant that 50 m was at least equivalent to 26 stream widths in length and likely enough to capture 90% of fish species representing >1% of relative abundance (Kanno et al. 2009). Main-stem Goose Creek sites averaged 3.9 to 6.9 m in wetted width, equivalent to 13 and 9 mean stream widths respectively, likely leading to underestimates of true fish species richness within broader stream segments (Kanno et al. 2009).

Benthic macroinvertebrate samples were collected to evaluate the biological condition of the streams and evaluate invertebrate response to exclosures. Four composite benthic macroinvertebrates samples were collected from four separate riffles (when possible) per site using a Surber sampler (0.093 m²; 500- μ m mesh) following the methods of the Bureau of Land Management National Aquatic Monitoring Framework (USBLM 2017). Samples were fixed with 95% ethanol, sorted, and identified to genus (class Insecta) and those that were not insects to a coarser resolution using unambiguous operational taxonomic units (Yuan et al. 2008). Data were used in a multimetric index developed for Idaho's Plains, Plateaus, and Broad Valleys region containing six metrics shown to discriminate between reference (least impacted) and nonreference sites: Simpson's diversity index, percent noninsect taxa, percent filterer taxa, percent clinger taxa, percent tolerant taxa, and semivoltine taxa richness (Jessup 2011). 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 the streams. For the Plains, Plateaus, and Broad Valleys region, an overall score of 68 represents the 50th percentile of reference sites, and a score of 54 represents the 10th 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).

Fish were collected from each 50-m reach using day-time 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 main-stem sites where two electrofishers and three netters were used. Subsequent passes were made if salmonids were caught on pass one to estimate salmonid abundance using removal sampling; three passes were typically completed, unless no salmonids were caught on pass two then sampling ceased, or more than three passes were completed if 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 from pass one only. Abundances of each salmonid species were estimated separately for individuals <100 mm TL and >100 mm TL using a Zippin estimator with data from all passes and the “removal” function in the FSA package (Ogle 2017) of Program R (R Core Team 2015), and abundances of all salmonid species were summed and expressed as a linear density (number per 100 m). Fish species richness was also computed per site using data from the first electrofishing pass only.

Riparian and instream habitat was sampled immediately after fish sampling. Transects perpendicular to the channel were established every 5 m at bank-full height for a total of 10 transects per site. Channel depth from bank-full height, 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 the 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 or fewer points across the 10 transect measurements in the wetted portion of the channel. Stream substratum was classified according to the modified Wentworth scale: bedrock, silt/clay (<0.064 mm in 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 follows: boulder, large wood (>10 cm in diameter, >4 m in length), small wood, aquatic vegetation (macrophytes), overhanging bank vegetation, undercut bank (>10 cm in 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 bank-full as follows: 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 alteration was recorded as the presence of livestock hoof prints or trails within 2.5 m of the transect endpoint. The streambank at each transect endpoint was categorized 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 enclosures on riparian vegetation using the Landsat NDVI as the response variable in generalized linear mixed models (GLMMs) with a Gaussian link function (a linear model). Site location relative to the enclosure (inside = 1, outside = 0) and timeframe (before enclosure = 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 enclosures but only during years after the enclosures were constructed (see Table 1). 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 2 years after a fire (fire = 1, no fire = 0). A global model was fit using data from all sites, years (1985 to 2015), and enclosures to assess the general response of riparian vegetation to the enclosures across the Goose Creek Group Allotment. For this model, all terms mentioned above were included, as well as a random effect for each individual enclosure. We also fit separate general linear models for each enclosure to parse any anomalous responses for individual enclosures; the fire term was omitted from general linear models for enclosures 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.

We also used GLMMs to evaluate the general, allotment-wide effects of enclosures 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, enclosure treatment (inside = 1, outside = 0 [baseline]) was the treatment effect, and each enclosure was treated as a random effect. A linear model (Gaussian link function) was fit for all response variables except for those expressed as a percentage that were fit using a logistic model (binomial link function). This global analysis estimates a common enclosure effect across all enclosures in the allotment for each response variable; the random effect (the intercept) adjusts for the uniqueness at sites associated with each enclosure. The Emery (EMRY) enclosure data were omitted from analysis of fish response metrics because the stream was fishless. The enclosure treatment term was assessed for significance using a one-tailed test at $\alpha = 0.10$ and the predicted response. In addition to the global analyses, we also fit separate models to estimate the effect of each individual enclosure on all response metrics. The same set of response metrics were used in the enclosure-specific models and enclosure treatment was the treatment effect (inside = 1, outside = 0 [baseline]) as before, but these models had no random effect. Again, the enclosure 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 enclosures (effect size) was related to the size of each individual enclosure. The treatment effect size estimated for each response variable was obtained from the general linear model from the previous analysis (except using a linear response function to maintain comparability across response metrics) and then correlated with the length of stream enclosed by each enclosure. A Spearman's rank correlation coefficient (r_s) was used to evaluate the strength of correlation, and significance was evaluated at $\alpha = 0.10$ with $n = 7$ for each correlation.

RESULTS

Satellite-Measured Riparian Vegetation

Riparian vegetation was generally more productive inside the enclosures after they were constructed (Figure 2), and this was reflected in the Landsat NDVI time series (Figure 3). The global GLMM showed there to be a significant interaction between site location and timeframe, owing to higher NDVI values inside enclosures after they were constructed, as predicted ($t_{1,641} = 4.013$, $P < 0.001$; Figure 3). In addition, precipitation had a significant

positive effect and fire (for two subsequent years) had a negative effect on NDVI as expected (precipitation: $t_{1,641} = 19.90$, $P < 0.001$; fire: $t_{1,641} = -6.82$, $P < 0.001$). Despite a global positive effect of exclosures on the NDVI, analysis of the individual exclosures showed a higher NDVI after exclosures were built for five of six exclosures evaluated individually ($|t_{242}| \geq 1.78$, $P \leq 0.038$). The Lower Cold Creek (LCLD) exclosure could not be evaluated by itself because it was built in 1982 and 30-m resolution Landsat NDVI data were not available until 1985. Upper Cold Creek (UCLD) was the only exclosure where the NDVI did not show a significant increase at sites inside the exclosure after it was constructed ($t_{242} = 0.359$, $P = 0.360$). Like the global analysis, precipitation had a positive effect on the NDVI at each individual exclosure ($|t_{242}| \geq 6.77$, $P < 0.001$). Fire only occurred at sites associated with four exclosures but not at all sites per exclosure (UCLD, EMRY, LBCH, STATE; Figures 1, 3), and it had a significant negative effect on the NDVI at sites associated with three of four exclosures ($|t_{242}| \geq 1.78$, $P = 0.038$); the Stateline exclosure (STATE) was the only exclosure where the NDVI did not show a response to fire ($t_{242} = -1.161$, $P = 0.123$).

Field-Measured Riparian Vegetation, Instream Habitat, and Biota

Several riparian vegetation and stream channel characteristics differed inside versus outside the 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 ≥ 1 m tall was higher inside exclosures versus outside ($P < 0.10$; Table 2). Streambanks were less altered (fewer livestock trails or hoof prints), were in better condition (less sloughing and slumping), and were narrower and deeper (smaller channel width : depth ratios) with less wetted width inside versus outside exclosures ($P < 0.10$; Table 2; Figure 4). The benthic macroinvertebrate multimetric index showed no difference inside versus outside exclosures ($P = 0.14$; Table 2; Figure 4), and only one fish metric showed a significant difference due to exclosure treatment. Age-0 salmonid (< 100 mm TL) densities (Rainbow Trout *Oncorhynchus mykiss* and Brook Trout *Salvelinus fontinalis*) were estimated to be 4 fish/100 m higher inside versus outside exclosures ($P = 0.065$; Table 2; Figure 4); all age-0 salmonids sampled were Brook Trout, except one Rainbow Trout from Upper Cold Creek (outside) and one Rainbow Trout from main-stem Goose Creek (outside).

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 2). The estimated exclosure effect size for percent gravel substrate was significantly

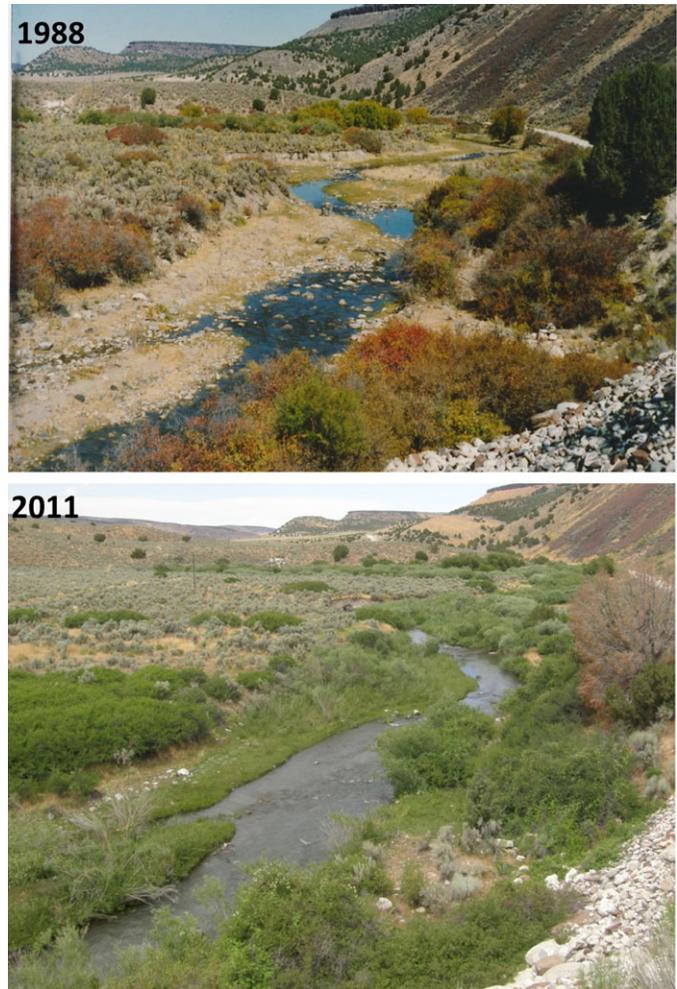


FIGURE 2. Photos from before (top panel, 1988) and after (bottom panel, 2011) installation of a riparian exclosure (GOOCR) on Goose Creek, Cassia County, Idaho. Photo credits: Burley Field Office, Bureau of Land Management.

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 average scores above the 50th percentile of reference conditions (Figure 4). All others sets of sites averaged between the 10th and 50th percentile of reference site scores (fair condition), with the exception of the Goose Creek main stem inside the Coal Banks exclosure (CLBK; mean score = 41.5, SD = 2.9) and Emery Creek outside the

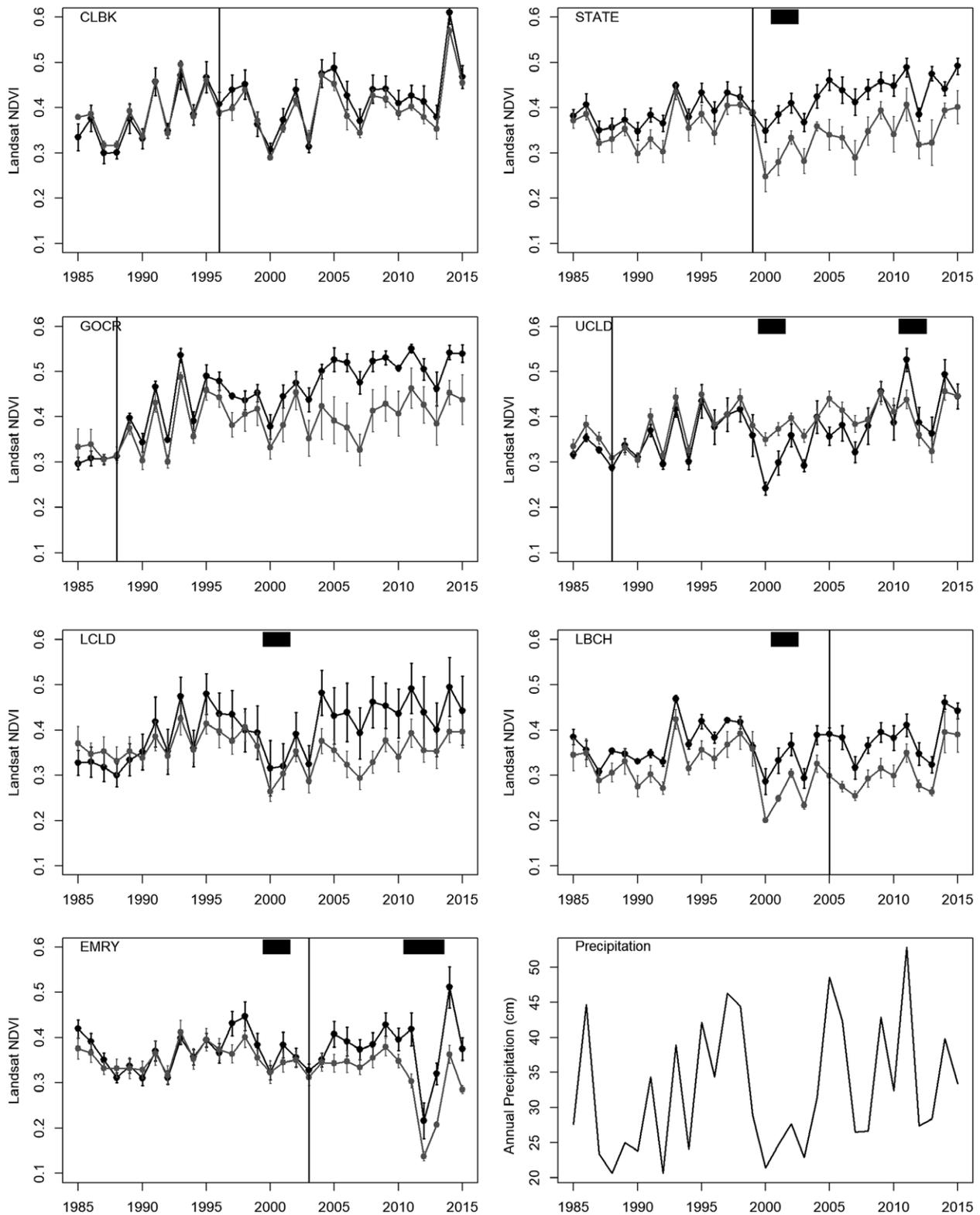


FIGURE 3. Mean (error bars show ± 1 SE; $n = 4$) Landsat NDVI over time by exclusion treatment (black = inside, gray = outside) for each exclusion. The vertical line indicates when the exclusion was built; the exclusion for LCLD was built in 1982. The years that fire was expected to influence NDVI values at some or all sites per exclusion is indicated by a black rectangle at the top of the graph, typically for 2 years after a fire (see Figure 1). Annual precipitation during the water year (October 1 of the prior year through September 31) is shown in the bottom right panel.

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 noninsect metric, as most benthic macroinvertebrates were insects. The lowest scoring metric differed for each exclosure and whether sites were inside versus outside the exclosure (results not shown).

DISCUSSION

We evaluated seven riparian exclosures in a livestock grazing allotment in the northern Great Basin using remote sensing data from 1985 to 2015 and paired field data from 2015, and our data showed that while exclosures contained more woody riparian vegetation and narrower and deeper channels, instream habitat and aquatic biota showed no response, except for wetted width and age-0 salmonids (primarily nonnative Brook Trout). Given the numerous one-tailed statistical tests using a liberal type I error rate ($\alpha = 0.10$) that was uncorrected for multiple tests, any real response by aquatic biota to exclosures should have been detected with high statistical power (Zar 1999; Moran 2003). Our findings do 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 (Wang et al. 1997), and we suspect this is the case on our study area. For example, the transport of sediment and other upstream factors likely offset any benefits to instream habitat and aquatic organisms provided by the improved local riparian and channel conditions we observed with our study exclosures. Since we observed no correlation between exclosure effect size (i.e., the difference in response metrics inside versus outside exclosures) and exclosure length (size) despite numerous statistical tests for significant correlation, the riparian exclosures in the Goose Creek Group Allotment may not have been large enough to have a detectable 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 USA (usually <20 ha; Sarr 2002), but with a caveat that it is used as a riparian pasture for 5 d each year (Table 1).

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; the NDVI showed an immediate response to some exclosures

TABLE 2. Riparian exclosure effect size (b_i) for riparian and instream habitat, aquatic macroinvertebrate, and fish metrics (response variable) in a generalized linear mixed model (GLMM) with exclosure treatment (inside = 1, outside = 0) and exclosure as a random effect. All nonpercentage response variables were modeled as a linear response function and all percentage variables as a logistic response function. Positive estimates (predicted response) indicate higher values inside of exclosures. Spearman's rank correlation coefficients (r_s) and P -values are given for the association between exclosure effect size and exclosure length for each response variable.

Category	Response (predicted)	GLMM			Spearman	
		b_i	SE	P -value	r_s	P -value
Riparian vegetation	Woody vegetation height (+)	0.567	0.205	0.003	-0.464	0.302
	% Woody vegetation (>1 m tall) (+)	2.311	1.290	0.037	-0.464	0.302
Stream bank	% Bank alteration (-)	-0.915	0.679	0.089	0.000	1.000
	% Bank slough/slump (-)	-6.486 ^a	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 (-)	1.064	0.760	0.919	0.429	0.354
	Residual pool depth (m) (+)	-0.027	0.038	0.765	0.286	0.556
	% Cover (+)	-1.109	0.958	0.877	0.018	0.969
	% Gravel (+)	1.511 ^a	3.111	0.314	0.857	0.024
Habitat diversity	Substrate H' (+)	-0.032	0.065	0.690	0.286	0.556
	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 macroinvertebrates	Multimetric index (+)	-0.463	2.522	0.573	0.643	0.139
Fish	Fish species richness (+)	-0.185	0.272	0.751	0.631	0.129
	Salmonids < 100 mm (fish/100 m) (+)	3.949	2.826	0.081	-0.291	0.527
	Salmonids > 100 mm (fish/100 m) (+)	-1.330	2.883	0.678	0.464	0.302

^aModel with a logistic response function would not converge. Parameter estimated using a linear response function.

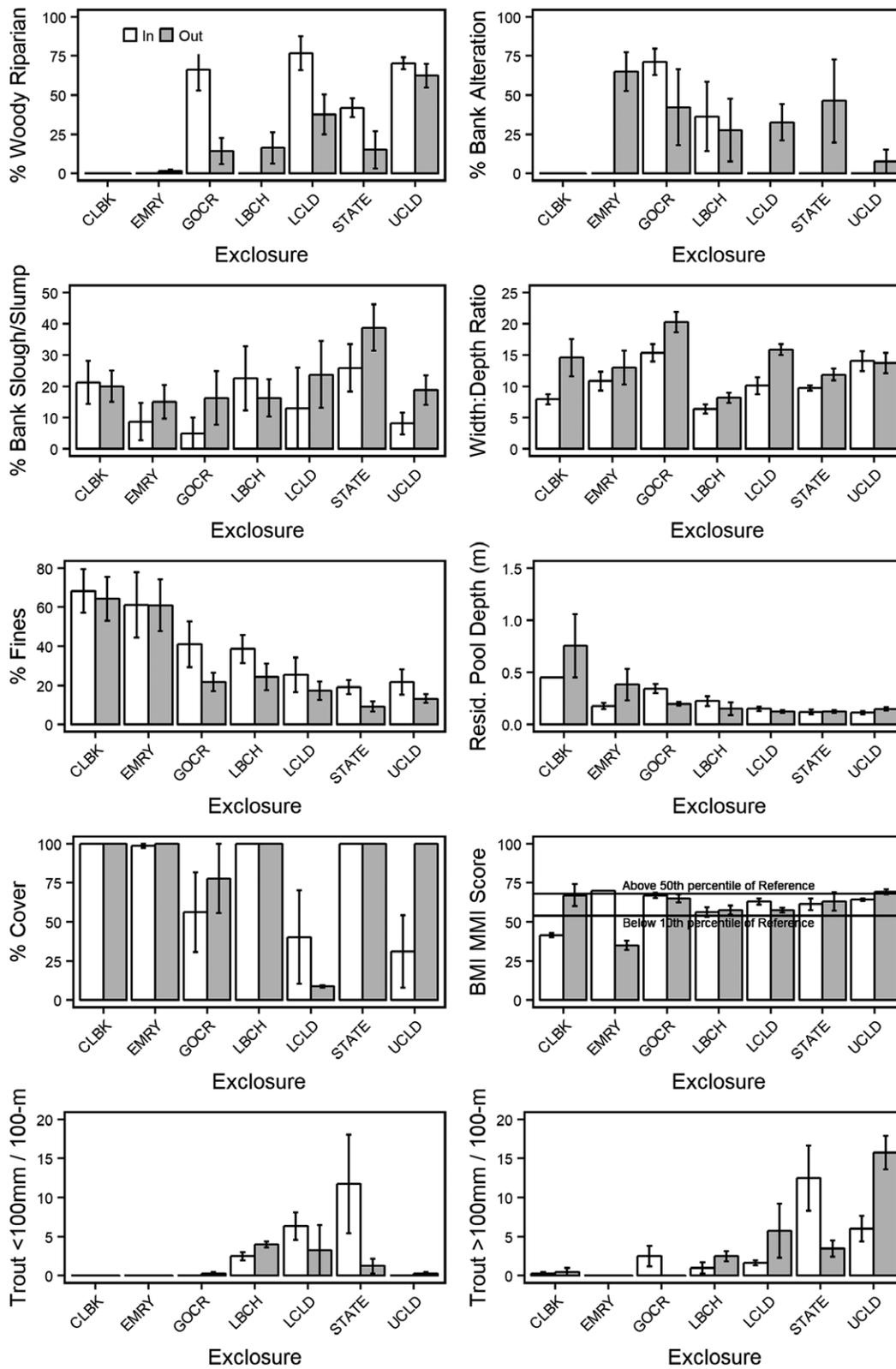


FIGURE 4. Mean (error bars show ± 1 SE) values of select response metrics by exclosure treatment (white = inside, gray = outside) for each exclosure in the Goose Creek Group Allotment. The 50th and 10th percentile scores of reference sites for the benthic macroinvertebrate multimetric index (BMI MMI) in the Plateaus, Plains, and Broad Valleys region are shown for context.

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 2 to 3 years after implementation of a conservation grazing plan. Others have also shown recruitment of aspen *Populus* spp. 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 (Nusslé et al. 2017). This highlights the importance of environmental context and growing conditions on vegetation response. The 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, and exclosure integrity related to fence maintenance (Table 1) 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 the number of age-0 salmonids per 100 m, a biological metric that has shown a positive response to exclosures in previous studies. Age-0 salmonids were, on average, four individuals more abundant per 100 m inside exclosures versus outside, albeit only significantly different from zero with a liberal probability value ($P = 0.081$). In a similar study, Bayley and Li (2008) found age-0 Rainbow Trout 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 salmonid habitat along streambanks, and increased invertebrate production (unmeasured). We did observe 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 nearby 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 exclosure studies containing salmonids.

The observed positive response by age-0 salmonids to exclosures was likely driven by the large difference in age-0 salmonids (nearly all age-0 Brook Trout) inside versus outside the Stateline (STATE) exclosure (Figure 4). If real, it appears that any increase in age-0 salmonids from exclosures does not influence the broader salmonid populations since age-1 and older fish did not show a parallel response. This may be due to the limited capacity of instream habitat to support older fish, especially since instream habitat also did not change in response to improved riparian, streambank, and channel conditions. The Brook Trout and Rainbow Trout densities we observed were at or below average compared with other naturally reproducing salmonid fisheries in the region (e.g., Meyer et al. 2006).

Despite only one aquatic organism response to exclosures, the benthic macroinvertebrate multimetric index suggested stream health overall was often between the 10th and 50th 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 with 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 despite some water quality concerns and the existence of a total maximum daily load (IDEQ 2010a, 2010b; 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 general inferences can only be made across all exclosures in the study, which confounds the power to detect differences at individual exclosures given natural environmental differences among systems. 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 enclosure on response metrics with error. Despite a novel study design, site selection associated with some enclosures, although mostly random, was affected by logistical constraints. The two enclosures on the Goose Creek main stem enclosed nearly all public land (U.S. Bureau of Land Management) encompassing the main stem, and our field sites outside of the enclosures were on livestock water gaps or near road crossings—the reasons they remain unenclosed. The main stem upstream and downstream of these two enclosures was on private land. Thus, while our study design allowed for inferences regarding enclosure effects on the entire allotment, in practice the instream habitat and biological communities in these enclosures were influenced by adjacent private land management. Other sites associated with enclosures (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 enclosures. While some have used remotely sensed data to evaluate riparian vegetation recovery (Booth et al. 2012; Hausner et al. 2018), 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 d 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, as demand for livestock-related products continues to increase with human population increases (Thornton 2010). What role, then, do enclosures play within the broader realm of grazing management since much enclosure research, including that presented herein, suggests they benefit local riparian vegetation but have inconsistent effects on aquatic habitat and biota? First, not all enclosures are constructed to solely benefit aquatic biota; they are often used for broader stream ecosystem benefits with objectives that

include vegetation, birds, and mammals (Sarr 2002). Fesenmyer et al. (2015) used aerial and satellite imagery and grazing history data from a nearby Great Basin watershed to show that changes from conventional to conservation grazing and enclosures lead to increases in woody riparian vegetation that were an important precursor to colonization by American beaver *Castor canadensis*, which can be important to passive restoration of incised stream channels common in the western USA (Pollock et al. 2014). Riparian enclosures are also often used to improve habitat for riparian-dependent birds, which have shown responses in species richness and composition due to changes in riparian vegetation (Dobkin et al. 1998).

Second, when the health of aquatic ecosystems and biota is a primary goal, riparian enclosures should be used strategically and in concert with other complementary grazing management techniques within entire allotments (thousands of hectares) and watersheds. Enclosures are often used to eliminate livestock use in sensitive areas, such as springs or areas prone to erosion and high sediment yields that can be deleterious to aquatic systems (Waters 1995; Sarr 2002). In these cases, enclosures should be used with grazing management techniques, such as planting of upland forage, use of off-stream water systems, and grazing rotations that are high intensity but short duration with long recovery times and that can be managed adaptively (Swanson et al. 2015). Riders on horseback can also be used to continually move livestock and keep them from overutilizing specific areas of streams. Together these management techniques limit the amount of time livestock are concentrated in stream corridors and can lead to broader allotment- or watershed-wide changes that improve habitat quality and benthic macroinvertebrates communities (Herbst et al. 2012; Fesenmyer et al. 2015). Upland forage, off-stream watering, and an altered grazing regime were all implemented in addition to riparian enclosures in our study allotment, which based on benthic macroinvertebrate data now has streams in fair condition and comparable to some regional reference streams without grazing (Jessup 2011).

Third, we suggest that enclosures may still offer benefits not yet revealed by run-of-the-mill enclosure studies—including our study. Enclosures may protect or create unique habitat conditions that may unknowingly facilitate species persistence. For example, in our study area 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 2018), 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 main-stem Goose Creek, and the Goose Creek enclosure (GOCR) is one of

the only areas on the main stem that has mature riparian vegetation (Figure 2). Most of the main-stem floodplain on private land is devoid of woody riparian vegetation because it was removed to maximize hay production and minimize water use 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 the persistence of Northern Leatherside Chub in other nearby areas of Goose Creek, where habitat quality is presumably lower. The increase in age-0 salmonids 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 as they grow, which may be why we observed no enclosure effect in age-1 and older salmonids. 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 sink habitat. Such research would help managers strategically use enclosures to protect known critical habitats and, therefore, continue to find enclosures a useful tool in the grazing management toolbox.

ACKNOWLEDGMENTS

We thank S. Walsh, M. Baker, and R. Lee for field assistance. B. Bohn and S. Sayer provided interesting insights into enclosure effectiveness and provided helpful comments on a draft manuscript. This project was funded by the U.S. Bureau of Land Management, Idaho State Office, Challenge Cost Share agreement L12AC20416, and Trout Unlimited's Coldwater Conservation Fund. There is no conflict of interest declared in this article.

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