Macroinvertebrate Responses to Riparian and Uplands Fuels Treatments in the Applegate and Middle Rogue River Watersheds, Oregon

Scott Miller
National Aquatic Monitoring Center, Bureau of Land Management/Utah State University, 5210 Old Main Hill, Logan, UT, 84321
scott.miller@usu.edu

Abstract

Contemporary forest management strategies (e.g., Healthy Forest Initiative, National Fire Plan) seek to reduce the threat of catastrophic wildfire, while maintaining natural fire regimes to support healthy upland, riparian and aquatic ecosystems. Successful implementation of these sometimes conflicting management goals requires an understanding of the risks associated with both wildfire and fire related management actions. Specifically, land management agencies seek to implement fuels reduction treatments in riparian areas to reduce fire intensity, severity and spatial extent; however, the effects of such treatments on riparian and aquatic ecosystems are uncertain. To better understand the effects of fuels reduction treatments on aquatic ecosystems, a replicated, paired-watershed study was implemented to quantify macroinvertebrate responses to riparian and upland thinning and burning. Both upland and riparian thinning and burning had little to no short-term (< 3 months) effects on macroinvertebrate communities relative to control sites. This result was consistent among measures of macroinvertebrate richness and relative abundance, in addition to assemblage composition. Low fire severity likely interacted with unseasonably dry conditions to minimize the occurrence of direct and indirect effects on macroinvertebrate assemblages. Although macroinvertebrate assemblages were not adversely impacted, these findings need to be interpreted in the context of the geographic and temporal scale of the study, fire intensity and severity and post-treatment weather conditions.

Introduction

Located at the interface between terrestrial and aquatic ecosystems, riparian zones harbor high biological diversity (Naiman et al., 1997; Sabo et al., 2005) and perform critical ecological functions (Gregory et al., 1991; Naiman et al., 1997). Specific to aquatic ecosystems, riparian zones are a primary determinant of instream physical and biological patterns and processes. Biologically, inputs of riparian vegetation drive secondary production (Cummins, 1974; Allan, 1995) and inputs of terrestrial arthropods provide critical subsidies to fish and other top predators (reviewed in Baxter et al., 2004). For example, in small forested tributaries, 90% of the organic matter supporting higher trophic levels can be of terrestrial origin (Cummins and Spengler, 1978). Physically, riparian vegetation stabilizes stream banks, dissipates flood energy, structures in stream habitat and moderates stream temperature (Gregory et al., 1991; Grunnell, 1997; Tabacchi et al., 2000). Consequently, activities that alter the type and cover of riparian vegetation can affect the structure and function of aquatic ecosystems.

Given the importance of riparian zones in maintaining functioning aquatic ecosystems, state and federal agencies have sought their protection from anthropogenic activities such as logging, fire, development and grazing. Specific to the Ashland Resource Area where this study was conducted, perennial streams flowing through federal land are buffered by 15 m and intermittent streams by 4.6 m Riparian Reserves. While the intent of this prescription is to protect riparian and aquatic ecosystems, such restrictions can inhibit and even counter act other management actions. Primary among these are fuels reduction treatments aimed at reducing fire frequency and severity. Portions of the riparian buffers are frequently excluded from fuels reduction treatments, yet these areas can act as corridors for the spread of wildfire. Consequently, land management agencies would like to implement fuels reduction treatments in riparian zones to reduce fire severity; however, the ability to implement such treatments while maintaining intact, functioning riparian zones remains uncertain.

Prescribed fires are currently excluded from riparian zones to maintain natural structure and function and avoid the negative effects typically associated with wildfires (e.g., reduced overhanging vegetation, fine sediment loading, altered hydrographs). However, natural disturbance regimes, including wildfire, are thought to play an integral role in sustaining diverse and productive riparian and stream ecosystems (Resh et al., 1988; Bisson et al., 2003; Minshall, 2003), with many organisms adapted to and even dependent on reoccurring disturbances (Lytle and Poff, 2004). Furthermore, low to moderate intensity fires are not predicted to adversely affect stream and riparian ecosystems because of low mortality rates for mature trees. Minshall's (2003) review of the effects of wildfire on aquatic macroinvertebrates revealed minimal short term and no adverse long-term effects within watersheds that have not been severely degraded by anthropogenic activities. However, only a handful of studies have quantified the effects of prescribed fire in riparian zones and these studies have produced equivocal results (Britton, 1991; Chan, 1998; Beche et al., 2005). Variability in post-fire responses of aquatic ecosystems are often attributed to regional climatic variability, fire intensity and severity, physiographic conditions and the degree of anthropogenic alteration (Gresswell et al., 1999; Minshall, 2003).

In this study, a before-after-control-impact (BACI) study design was implemented to quantify the effects of fuels reduction treatments on stream ecosystems. Specifically, the short-term (< 3 months) responses of aquatic macroinvertebrates to upland and riparian fuels reduction treatments (hand thinning and understory burning) were quantified. Aquatic macroinvertebrates were chosen as the primary instream biologic response variable because they are ubiquitous, exhibit graded responses to increasing stress levels and are thus capable of integrating the cumulative direct (e.g., increased temperature, ash, nutrient loadings) and indirect (e.g., increased fine sediment loading, altered hydrologic regimes, reduced organic matter inputs) effects of fire on aquatic ecosystems. This study is one component of a larger, interdisciplinary effort designed to understand how riparian and stream ecosystems respond to fuels reduction treatments. Specifically, researchers seek to understand if riparian fuels reduction treatments provide additional protection from the threat of catastrophic wildfire and if this can be accomplished without threatening the ecological integrity of riparian and stream ecosystems.

Methods

Study Area

This study was conducted in the Applegate and Middle Rogue Rivers, both 4th field subbasins of the Rogue River located in southwestern Oregon. The Applegate and Middle Rogue subbasins originate in the Siskiyou Mountains of the Klamath Mountain geologic providence and are characterized by mild, wet winters and very hot, dry summers. Precipitation (range 66 – 118 cm) largely falls as rain from October to April, while shallow snowpacks can accumulate above 1000 m from December to March. Historically, the region had a frequent, but low intensity fire regime; however, decades of fire suppression, logging and re-seeding has led to high fuel loadings and subsequent fire intensity. In addition to the high risk of fire, the Applegate and Middle Rogue subbasins were chosen for study by a multidisciplinary panel because of: 1. the high percentage of federal land; 2. ongoing and planned fuels treatment projects; and 3. high abundance of 'replicate' drainage basins suitable for treatment.

Study design

12 1st and 2nd order tributaries were selected for study; eight within the Applegate and four within the Middle Rogue subbasins (Table 1). Basins were chosen in groups of three to minimize variability in aspect, slope, watershed area and vegetative community composition among treatment and control basins. Grouped basins were generally located adjacent to one another and randomly assigned one of three treatments: control, upland fuels reduction or riparian and upland fuels reduction. Study systems were 5th and 6th field subbasins ranging in size from 33 to 330 ha and 520 to 1500 m in elevation. All basins experience perennial stream flow with discharge ranging from 0.3 to 22 L/s; however, areas of subsurface flow are common.

Control basins experienced no intervention and were used to quantify background changes in macroinvertebrate assemblages pre- and post-treatment. The uplands treatment consisted of cutting and hand piling small diameter (< 20 cm) woody vegetation followed by low to moderate severity (average CBI = 1.18) understory burning. A 15 m buffer was left adjacent to both sides of the stream channel. For the 'riparian' treatment, small diameter woody vegetation in both riparian and upland regions was cut and hand piled followed by low severity (average CBI = 0.47) understory burning. Overstory shade producing vegetation and riparian species (e.g., maple, alder, dogwood) were not directly treated; however, they were exposed and vulnerable to fire during underburning.

A BACI sampling design was implemented to quantify changes in macroinvertebrate assemblages pre- and post-treatment among control, uplands and riparian treatments. Pre-treatment sampling occurred in June and July of 2006, hand piling and pile burning in 2007, understory burning during throughout the spring of 2008 and post-treatment sampling in June and July of 2008. Through sampling paired treatment and control sites before and after the intervention, natural temporal variation could be separated from treatment effects.

Macroinvertebrate communities

Aquatic macroinvertebrates were sampled once pre- (June – July, 2006) and post- treatment (June – July, 2008) at all sites (Table 1). Qualitative samples were collected at all sites during both time periods, while low discharge rates in 2008 limited quantitative sampling at all sites. Consequently, analyses within this report focus exclusively on qualitative macroinvertebrate collections.

The objective of qualitative sampling was to collect as many different kinds of invertebrates living at a site as possible. Samples were collected with a kicknet (457 x 229 mm) fitted with a 500 micron mesh net and by hand picking invertebrates from woody debris, large boulders and vegetation. All major habitat types (e.g., riffles, pools, back waters, macrophyte beds) were sampled and composited to form a single sample from each site on each sampling date. In the laboratory, all composite samples were processed in their entirety, no subsampling procedures were used. When possible, we identified macroinvertebrates to genus (Merritt and Cummins, 1996 and references therein). Chironomidae midges, however, were identified to tribe and all non-insect taxa were identified to either order or family (Thorp and Covich, 1991 and references therein).

In addition to assemblage level analyses, several macroinvertebrate metrics were compared among treatments because of their hypothesized susceptibility to the direct and indirect effects of fire. Specifically, we compared macroinvertebrate richness, diversity, relative abundance of the numerically dominant family, as well as richness and relative abundance of the scraper functional feeding group, clinger taxa and Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa among treatments pre- and post-intervention. We predicted that if the aforementioned treatments had adverse, short-term effects on aquatic ecosystems, all macroinvertebrate metrics would decrease except for the relative abundance of the numerically dominant family, which would increase and potentially change families.

Statistical Analysis

Prior to conducting statistical analyses, richness estimates were standardized to the density of the least abundant reach using rarefaction. Rarefaction corrects for differences in sampling effort and macroinvertebrate abundances (range 60 – 216) and is recommended because richness estimates generally increase with sampling effort and the number of individuals processed (Gotelli and Colwell, 2001). Rarefaction standardizations were performed by randomly subsampling 60 individuals, density of the least abundant site (BVR: B - 2008) (Appendix 1 – Table 1), from each of the 23 other samples using Ecosim simulation software (Gotelli and Entsminger, 2006); average results from 100 randomizations were utilized.

Differences in macroinvertebrate metrics between years, among treatments and between years within individual treatments were quantified using two-way analysis of variance. The interaction of the two factors, year and treatment, was used to quantify responses to riparian and upland fuels treatments. An alpha-level of 0.05 was used to evaluate statistical significance.

In addition to analyzing macroinvertebrate metrics, responses of macroinvertebrate assemblages were quantified using multivariate statistical procedures. Prior to running multivariate analyses, rare species (i.e., taxa found at less than 2 sites) were deleted because they appeared to obscure patterns in the data by decreasing the signal to noise ratio. Next, macroinvertebrate abundances were relativized by row totals to obtain the relative abundance of individual taxa at a site. Relativizations were necessary since qualitative sampling did not result in equal sampling effort among sites.

Gradients in macroinvertebrate assemblages among sites and time periods were characterized using nonmetric multidimensional scaling (NMDS) ordination. NMDS is an indirect gradient analysis technique that uses rank community dissimilarities to iteratively search for the optimal arrangement of sample objects in as few dimensions as possible (McCune and Grace, 2002). NMDS was run with Sørensen's distance in PC-ORD version 5. Dimensionality was assessed by evaluating the relationship of final stress versus the number of dimensions; in addition, a Monte Carlo test with 250 runs of the randomized data quantified the probability of observing a stress as low as or lower than that observed through chance alone. Ordination solutions were rigidly rotated to maximize loadings of macroinvertebrate metrics with individual ordination axes, which were overlain onto ordinations as joint plots. A value of 0.40 was chosen as a cutoff for interpreting ecologically meaningful joint plot correlations, which is more conservative than the statistically significant value ($r = 0.33$, $P = 0.05$, $df = 23$).

Lastly, differences in the magnitude and direction of macroinvertebrate assemblages pre- and post-treatment among treatments were quantified using compositional vectors computed from the NMDS ordination (McCune, 1992). The magnitude and direction of macroinvertebrate changes pre- and post-treatment was computed from ordination axes scores for each pair of sites using Euclidean distance and compared among treatments with a multiple response permutation procedure (MRPP) (Mielke and Berry, 2001). MRPP is a nonparametric permutation procedure that tests for differences among two or more groups. A *p*-value assesses the probability of observed group differences under the null hypothesis, while an *A*-statistic quantifies effect size and within group homogeneity (McCune and Grace, 2002). Both the magnitude and direction of change were compared among treatments.

Results

A total of 3,121 individual invertebrates were collected and identified from the six control and six treatment sites sampled in 2006 and 2008. Invertebrates were identified from 16 orders, 72 families and 112 genera (Appendix 1 – Table 11). Site assemblages had an average taxa richness and diversity of 34 and 2.9 respectively (Appendix 1 – Table 3). Diversity was highest among insects, particularly Trichoptera and Diptera; Diptera was also the numerically dominant order. Chironomidae, Nemouridae and Heptageniidae were the numerically dominant families, comprising 25 percent of total site abundance on average (Appendix 1 – Table 2). Site assemblages had an average Hilsenhoff Biotic Index (HBI) of 4.6 and an average United States

Forest Service (USFS) community tolerance quotient (CTQ_d) of 61 indicating a macroinvertebrate assemblage moderately tolerant of organic pollution and fine sediment, good water quality and relatively stable substrate conditions (Appendix 1 – Table 6).

No significant short-term effects of upland and/or riparian fuels reduction treatments on macroinvertebrate assemblages were detected. Average values for the nine macroinvertebrate metrics remained fairly consistent pre- and post-treatment and no significant differences in the magnitude or direction of change were found among treatments (Table 2; Figs. 1 and 2). Furthermore, macroinvertebrate assemblages did not exhibit consistent directional changes ($A = 0.03$; $P = 0.71$) or differences in the magnitude of change among treatments ($A = 0.04$; $P = 0.62$) (Figs. 3 and 4). Rather, background variation to other unmeasured environmental factors represented the dominant gradients in macroinvertebrate assemblages.

Differences in macroinvertebrate assemblages were most pronounced among sampled geographic regions ($A = 0.09$; $P < 0.001$) (Fig. 5) and treatment types ($A = 0.03$; $P = 0.05$) (Figs. 1 and 2), although all differences were of marginal ecological significance. Such differences are illustrated by the NMDS ordination, which identified three significant gradients that accounted for 88% of the variance in macroinvertebrate assemblages. Axis 2 (15%) and 3 (66%) accounted for the greatest amount of variability, while axis 1 only retained seven percent of the variation. Axis 3 separated sites according to the richness and relative abundance of cold water, pollution intolerant taxa. Sites located in the Foots and Upper Star subbasins (top of ordination along axis 3) had greater EPT, scraper and clinger richness and relative abundances than sites located in Lower Star or Beaver Creek (bottom of ordination along axis 3). The ordination had a stress of 9.9 and instability of 0.0001, corresponding to a stable solution with little risk of false interpretation (McCune and Grace, 2002).

Control sites also significantly differed from both riparian and upland treatment sites (Table 2; Figures 1 and 2). In general, control sites had greater overall richness and evenness including higher numbers of grazing and clinger taxa and EPT taxa such as Heptageniidae mayflies. Differences between control and treatment sites were consistent pre- and post-treatment.

Discussion

Contemporary forest management strategies (e.g., Healthy Forest Initiative, National Fire Plan) seek to reduce the threat of catastrophic wildfire, while maintaining natural fire regimes to support healthy upland, riparian and aquatic ecosystems (Bisson et al., 2003). Successful implementation of these sometimes conflicting management goals requires an understanding of the risks associated with both wildfire and fire related management actions. To better understand the effects of fuels reduction treatments on aquatic ecosystems, a replicated, paired-watershed study was implemented to quantify the effects of riparian and upland thinning and burning on aquatic macroinvertebrates. Results herein show that both upland and riparian thinning and burning had little to no short-term effects on macroinvertebrate communities relative to control sites. This result was consistent among measures of macroinvertebrate richness and relative abundance, in addition to assemblage composition. However, it is imperative that these findings be interpreted in the context of the geographic and temporal scale of the study, fire intensity and severity and observed weather conditions.

Fire can adversely impact macroinvertebrate assemblages through either direct or indirect pathways (Gresswell et al., 1999; Minshall, 2003). Direct effects include increased temperature, nutrients and ash and occur during or immediately post-fire. These impacts typically have little to no effect on macroinvertebrates except during high intensity fires (Minshall, 2003). Rather, indirect effects such as increased turbidity, fine sediment loading and channel alteration can elicit the greatest macroinvertebrate responses (Minshall, 2003).

The paucity of significant fire effects (direct or indirect) on the physical stream template likely explains why macroinvertebrates assemblages were not adversely impacted in this study. Parallel studies monitoring chemical and physical changes pre- and post-treatment observed only marginal conductivity and temperature responses, while pH, discharge, substrate and percent shade did not exhibit differential responses among treatments (Volpe, 2009). Although statistically significant, observed conductivity decreases (average change was from 519.4 to 488.2 μS/cm within the riparian treatment) were likely too small to be of significance to macroinvertebrate assemblages (Yuan and Norton, 2003). In contrast, observed temperature increases for riparian treatments (seven day average of the maximum daily temperature increased from 16.8 to 18.35^{o}C on average) could reduce macroinvertebrate populations found at the edge of their thermal optimal (Vannote and Sweeney, 1980). Regardless, riparian treatment sites did not exhibit systematic changes in taxa richness or relative abundance as compared to control sites. Our findings parallel experimental studies by Beche et al., (2005) and Britton (1991) and studies of natural wildfires by Minshall et al. (1989, 1997), Royer and Robison (2001) and Rhine (1996) who failed to find significant macroinvertebrate responses in the absence of chemical or physical changes.

Low fire severity likely interacted with unseasonably dry conditions to minimize the occurrence of direct and indirect effects on macroinvertebrate assemblages. Within riparian treatments, burn severity ranged from low to moderate; burned areas were discontinuous, with large patches of unburned vegetation and minimal impact mortality of mature trees (Martin and DeJuilio, 2009). In contrast, burn severity was greater in upland treatments, with burned areas being larger and more contiguous than in riparian zones. However, no significant runoff events occurred post-treatment, minimizing the opportunity for hillslope erosion and instream fine sediment loading. The 2008 water year experienced 15% less precipitation than the ten year average (Volpe, 2009). Given the short duration of post-treatment monitoring, additional monitoring of chemical, physical and biological response variables should occur to document potential longer-term responses to the riparian and uplands fuels treatments.

Responses of aquatic ecosystems to fire can be highly variable, with a litany of factors (e.g., fire severity, watershed size, climate, vegetation type and cover) influencing physical and biological responses (Gresswell et al., 1999; Minshall, 2003). Small, degraded watersheds experiencing high severity fires appear most vulnerable to fire (Minshall, 2003; Bisson et al., 2003). For example, Mellon et al. (2008) found significant, adverse impacts to macroinvertebrate assemblages following a high intensity wildfire within small, intensively managed watersheds, while fires of similar severity in Yellowstone National Park resulted in only minor macroinvertebrate responses (Minshall et al., 1989). Based solely on macroinvertebrate assemblages, the studied subbasins suggest that the resistance of the aquatic ecosystems has not been compromised by excessive anthropogenic disturbances; macroinvertebrate assemblages were not dominated by disturbance tolerant taxa (Appendix 1 - Table 8), but rather a diverse array of life-history strategies and functional feeding groups (Appendix 1 -Table 9). In contrast, fuels reduction treatments implemented in heavily degraded watershed or treatments proceeded by high intensity rain events might elicit considerably different outcome.

Results from this study, support a small, but growing body of literature suggesting low to moderate intensity fires, both natural and prescribed, have little to no adverse impacts on macroinvertebrate assemblages (Britton, 1991; Minshall, 2003; Beche et al., 2005). However, given the individualistic responses of many systems and the myriad of factors influencing post-fire outcomes, the extrapolation of results from one geographic region to another is tenuous. Consequently, the use of fire in riparian areas as a management tool will need to be implemented with extreme caution and evaluated on a case-by-case basis.

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Table 1. Location, assigned treatments and types of macroinvertebrate samples collected at the 12 sites sampled pre-(2006) and post-treatment (2008).

				Sample type 2006		Sample type 2008	
Site	Treatment	Latitude	Longitude	Qualitative	Quantitative	Qualitative	Quantitative
BVR:A	Riparian	42.096111	-122.985	June	NA*	June	NA
BVR:B	Upland	42.103611	-122.995	June	June	July	NA
BVR:C	Control	42.103889	-122.995	June	June	July	July
FTS:A	Upland	42.375278	-123.115	July	July	July	July
FTS:B	Riparian	42.368056	-123.098889	July	July	June	June
FTS:C	Control	42.359722	-123.098333	July	July	July	July
LSTR:A	Riparian	42.154167	-123.068056	June	June	June	June
LSTR:B	Upland	42.154167	-123.071667	June	June	June	June
LSTR:C	Control	42.153333	-123.075278	June	June	June	NA
USTR:A	Upland	42.173611	-123.141667	June	June	June	NA
USTR:B	Riparian	42.1675	-123.131111	June	June	June	June
USTR:C	Control	42.166389	-123.160833	June	June	June	June

*NA indicates that macroinvertebrate samples were not collected at site for a given date.

Table 2. Results (*F* -statistic) of two-way ANOVA comparing macroinvertebrate metrics between periods, among treatmeatments and between periods within treatments. The interaction of these two factors, period and treatment quantifies the significance of riparian and uplands fuels reduction treatments.

	Factor (degrees of freedom)		
Response variable	Period (1)	Treatment (2)	Interaction (2)
Richness	33.0*	249.1**	0.14
EPT richness	0.01	27.4*	0.18
Shannon's diversity index	5.1	6.7	0.29
Scraper richness	6.9	25.2*	0.07
Clinger richness	0.19	43.0*	0.08
Dominant family realtive abundance	3.2	11.8*	0.17
EPT relative abundance	4.7	14.7*	0.24
Scraper relative abundance	12.3*	57.2*	0.02
Clinger relative abundace	0.83	2.5	0.5

* significant at the 0.05 alpha level

** significant at the 0.001 alpha level

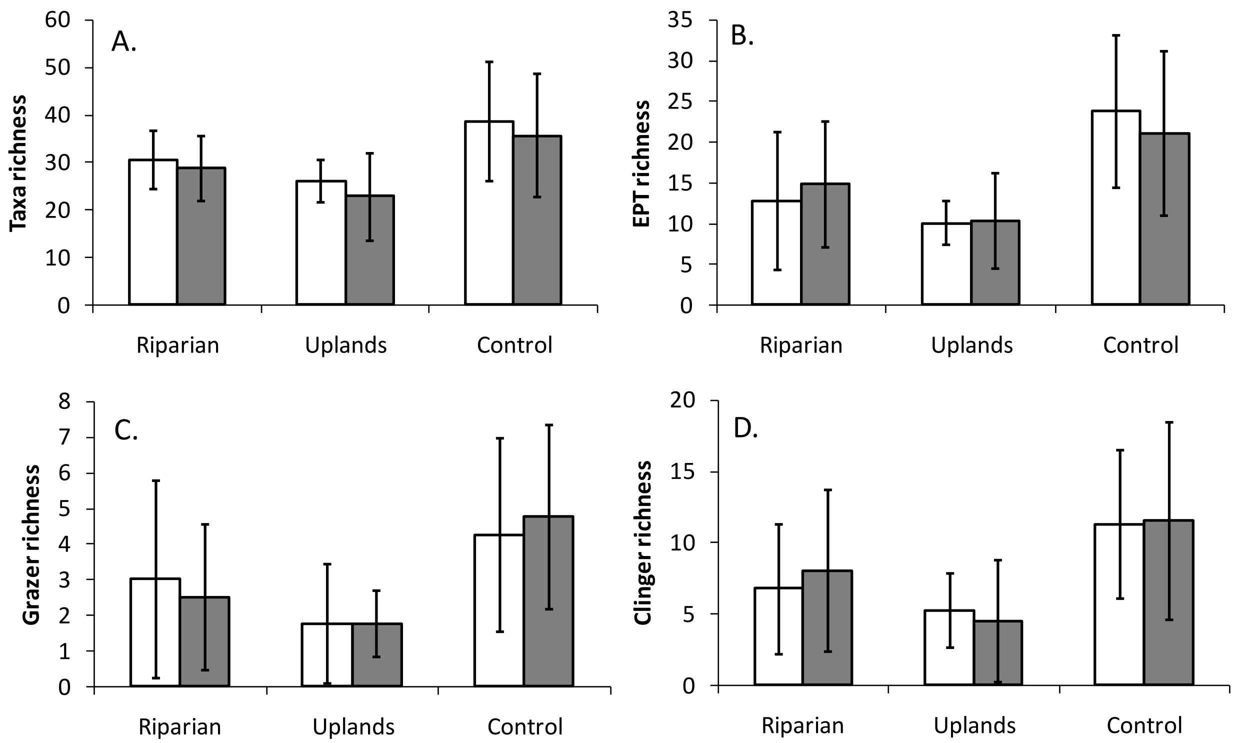

Figure 1. Macroinvertebrate richness measures compared pre- (hollow bars) and post- (shaded bars) treatment among riparian, upland and control treatments. Average values (± 95% confidence intervals) for the four replicates within each treatment and time period are presented. Two-way ANOVA results are presented in table 2.

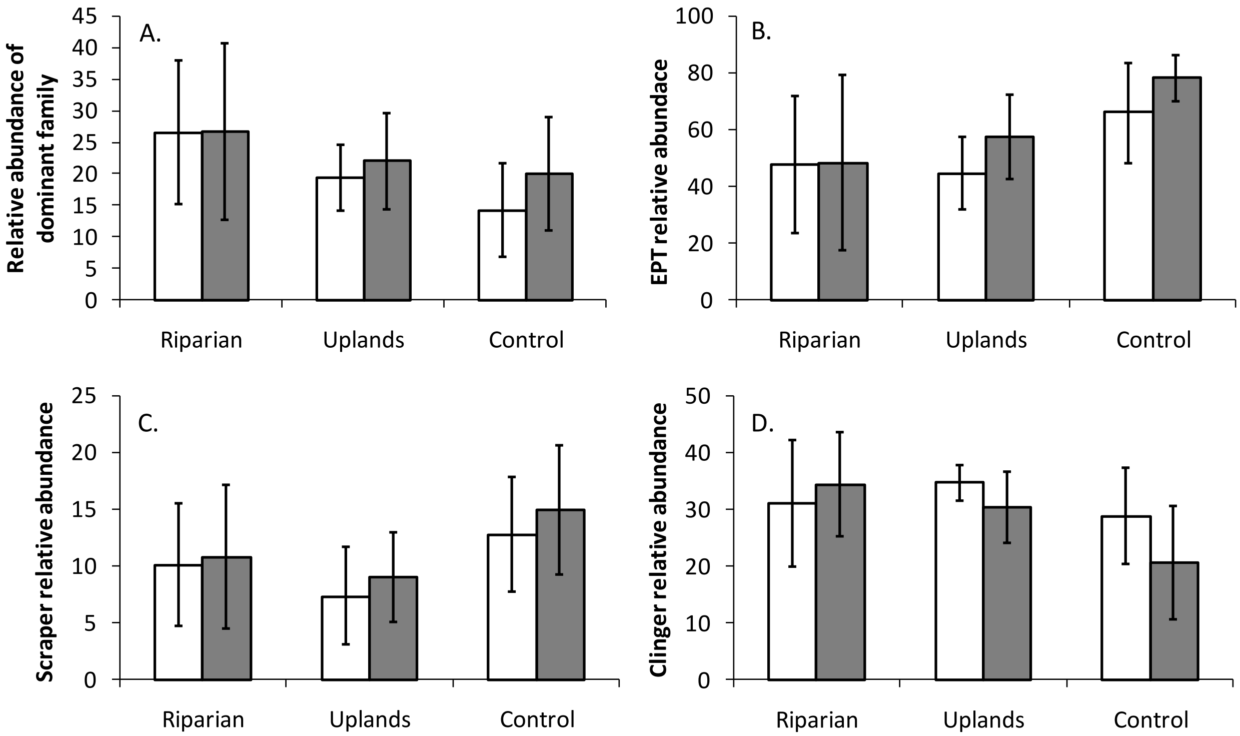

Figure 2. Relative abundance of the dominant macroinvertebrate family (A), EPT abundance (B), scraper abundance (C) and clinger abundance (D) compared pre- (hollow bars) and post- (shaded bars) treatment among riparian, upland and control treatments. Average values (± 95% confidence intervals) for the four replicates within each treatment and time period are presented. Two-way ANOVA results are presented in table 2.

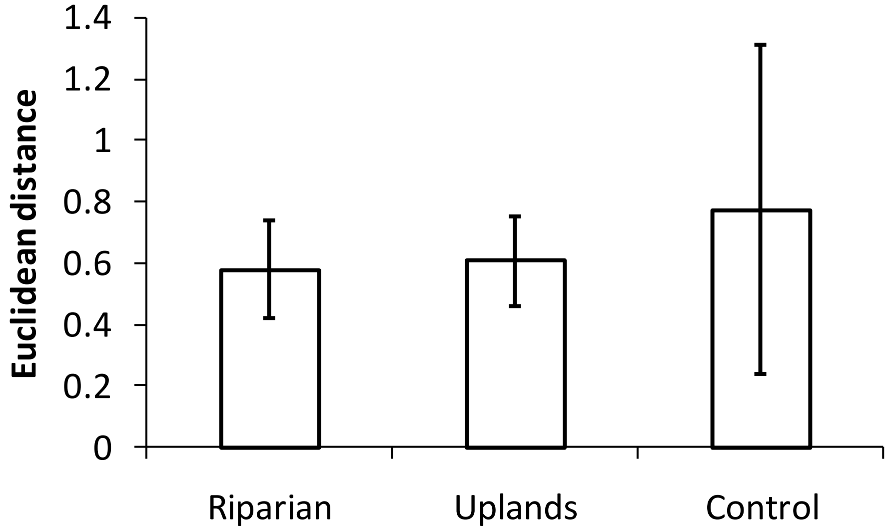

Figure 3. Average Euclidean distance (± 95% confidence interval),computed from the NMDS ordination (Fig. 4), measuring the amount of change in macroinvertebrate assemblage composition among riparian, uplands and control treatments.

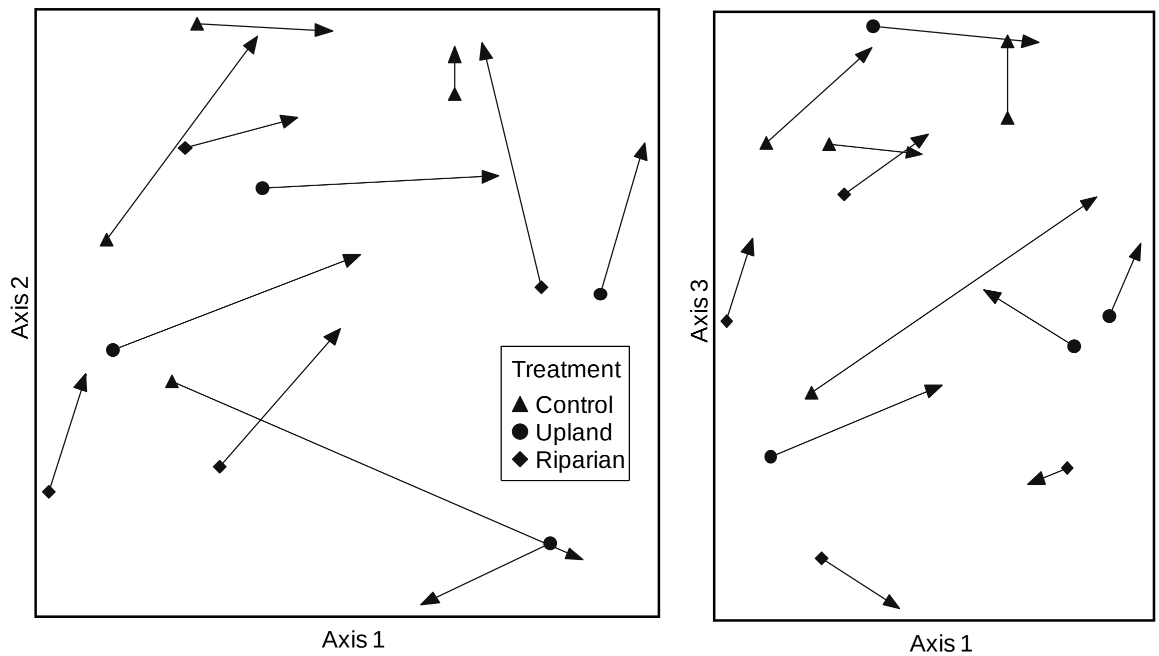

Figure 4. Nonmetric multidimensional ordination of macroinvertebrate relative abundances for the 12 sites sampled pre- and post-treatment. Views of axes 1 and 2 and 1 and 3 are depicted from the three dimensional solution. Compositional vectors are overlaid with the vector tail (pre-treatment) coded by treatment type and the vector tip (post-treatment) a generic arrow indicating the direction of change.

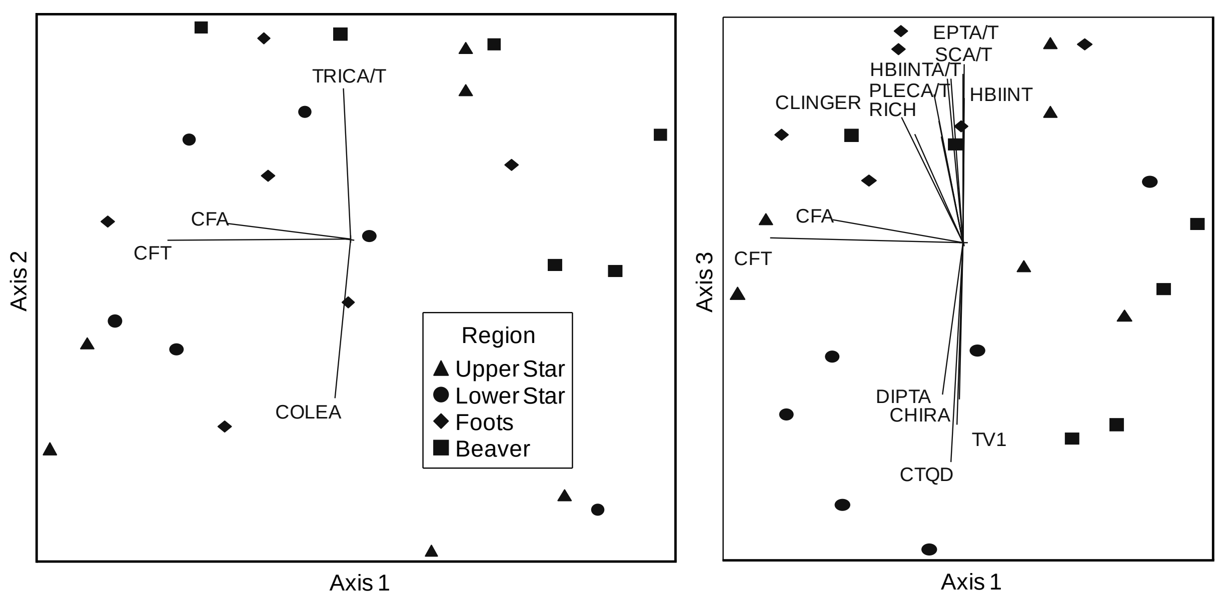

Figure 5. Nonmetric multidimensional ordination of macroinvertebrate relative abundances for the 12 sites sampled pre- and post-treatment. Views of axes 1 and 2 and 1 and 3 are depicted from the three dimensional solution. Sites are grouped according to geographic region, the strongest observed gradient. Overlaid are joints plots illustrate correlations(r > 0.4) of

macroinvertebrate metrics with assemblage composition of individual axes. Joint plot labels are as follows: Trichoptera abundance/richness (TRIA/T); Collector-filterer abundance (CFA) and richness (CFT); Coleoptera abundance (COLEA); Ephemeroptera, Plecoptera, and Trichoptera abundance (EPTA) and richness (EPTT); Scraper abundance (SCA) and richness (SCT); Richness (Rich); Hilsenhoff biotic index intolerant taxa abundance (HBIINTA) and richness (HBIINTT); Plecoptera taxa abundance (PLECA) and richness (PLECT); Clinger richness (CLINGER); Diptera taxa abundance (DIPTA); Chironomidae taxa abundance (CHIRA); USFS community tolerance quotient (CTQD) and Tolerance values (TV1).