On the efficacy of restoration in stream networks: comments and critiques

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Abstract

Swan and Brown (2017) recently addressed the effects of restoration on stream communities under the meta-community framework. Using a combination of headwater and mainstem streams, Swan and Brown (2017) evaluated how position within a stream network affected the outcome of restoration on invertebrate communities. Ostensibly, their hypotheses were partially supported as restoration had stronger effects in headwater streams: invertebrate taxonomic richness was increased and temporal variability decreased in restored reaches; however, these results were not consistent upon closer scrutiny for both the original paper (Swan and Brown 2017) and the later erratum (Swan and Brown 2018). Here, I provide a secondary analysis of the data, with hypotheses and interpretations in the context of stream, restoration, and metacommunity ecology. I did not find any effects of restoration on local diversity, spatial dissimilarity, or temporal variability, let alone differential effects of restoration between headwaters and mainstems; these results are contrary Swan and Brown (2017, 2018), who reported that restoration increased taxonomic richness, increased spatial dissimilarity, and decreased temporal variability in restored headwater streams. I demonstrate further that the statistical tests conducted by Swan and Brown (2017, 2018) were invalid and, therefore, recommend use of the results presented here. More broadly, I suggest that river and stream restoration will likely have greater success if a regional approach is taken to designing and implementing restoration projects.

Keywords: biodiversity, community ecology, freshwater ecology, metacommunity theory, open science, restoration ecology

Introduction

In a recent study, Swan and Brown (2017) evaluated how restoration affected community diversity in streams through the use of metacommunity theory. Under this framework, local effects are associated with species' niches while regional effects are more associated with dispersal (Leibold et al. 2004). In the context of stream networks, headwaters are isolated patches more likely to be impacted by niches and environmental characteristics and mainstems are well-connected more likely to be affected by dispersal (Heino et al. 2003, Leibold et al. 2004, Grant et al. 2007, Altermatt 2013, Heino 2013). Restoration of stream habitats was therefore expected to have a greater impact on communities in headwaters relative to mainstems (Swan and Brown 2017).

Although Swan and Brown (2017, 2018) noted that restoration techniques can vary in intrusiveness on stream ecosystems, they did not account for this in their experimental design and statistical analyses. Restored streams in their study received various combinations of bank stabilization, in-channel manipulation, and riparian reforestation (i.e. tree planting) treatments, and these treatments were not applied in a consistent or systematic manner (Swan and Brown 2017: Table 2). Swan and Brown (2017) did not set a restoration criterion for site inclusion in their study, instead including all sites regardless of the combination of applied restoration treatments. I suggest that this oversight leads to unnecessary assumptions about the efficacy of restoration by assuming the effects of all treatments combinations are equivalent, and this issue could have been partially resolved a priori by hypothesizing how each restoration reach would affect headwater and mainstem streams and then setting requirements for site inclusion in the analyses.

55 I contend that the various restoration treatments differ not only in their overall effects but
56 also if the treatment is applied in headwater or mainstem streams, and, for these reasons, criteria
57 for site selection could be set. I suggest that bank stabilization and in-channel manipulation
58 treatments are more likely to have consistent effects in both headwaters and mainstems, while
59 riparian reforestation would likely have stronger effects in headwater compared to mainstem
60 streams. A similar argument was made by Swan and Brown, though it was not explicitly noted
61 until the erratum (Swan and Brown 2018). Bank stabilization and in-channel manipulation can
62 increase bed stability and substrate availability and diversity in both headwater and mainstem
63 streams; however, the effects of riparian reforestation could act on a gradient from headwaters to
64 mainstems. For example, leaf litter is an important source of habitat and nutrients in headwaters
65 but less so in mainstems (Vannote et al. 1980, Rosi-Marshall and Wallace 2002). Additionally,
66 the utility of riparian reforestation on reducing nutrient inputs notwithstanding (Collins et al.
67 2013), the effects of riparian reforestation could be stronger in headwater streams because they
68 are isolated systems, whereas mainstem streams receive flows of water, nutrients, and organisms
69 from many tributaries (Vannote et al. 1980). Effectively, mainstems are dependent on other
70 tributaries and any local restoration effects via riparian reforestation could be overwhelmed by
71 incoming flows from unrestored streams (Wahl et al. 2013).
72 Here, I present a re-analysis of the data provided by Swan and Brown (2017, 2018). I
73 hypothesized that stream-channel manipulations would have a more consistent effect between
74 headwaters and mainstems relative to the effect of riparian reforestation, with stronger effects of
75 restoration in headwaters relative to mainstems; I used this hypothesis to guide and inform site
76 selection in my re-analysis. I required sites in the re-analysis to have received both the bank
77 stabilization and in-channel manipulations treatments (hereafter "revised" sites), although sites

receiving riparian reforestation were also included if they received both the bank stabilization and in-channel manipulations treatments. I also re-analyzed the full data (hereafter "full" sites) to determine if any differences, or lack thereof, between the full and revised sites analyses could be attributed to increased variation in the revised sites due to decreased sample size. Finally, I compare the interpretation and conclusions from my re-analysis to those in Swan and Brown (2017) and the erratum (Swan and Brown 2018).

Methods

I generally followed the analyses as written by Swan and Brown (2017), with modifications made when necessary. The three community response variables were local diversity, spatial dissimilarity, and temporal variation. Local diversity was calculated as taxonomic richness (i.e. number of different taxa present) and taxonomic diversity (i.e. Shannon's diversity) and compared using an analysis of variance (ANOVA). The model was constructed to examine the individual effects of reach (restored or adjacent), order (headwater or mainstem), and season (spring, summer, fall, and winter) and all two- and three-way interactions, with individual ANOVAs for richness and diversity; I also fit the full and reduced taxonomic richness models proposed in the erratum (Swan and Brown 2018) as a separate set of ANOVAs. Spatial dissimilarity between communities in restored and adjacent reaches for each order-by-season combination was quantified using the modified Gower index (Anderson et al. 2006) with a logarithm with a base of 5 on an untransformed abundance matrix. Modified Gower dissimilarities were then compared using an ANOVA with the individual effects of season and order as well as their interaction. Temporal variability was measured as the multivariate dispersion (i.e. mean distance to the centroid) of repeated samples for each stream-by-reach-by-order combination (Anderson et al. 2006). Distances were calculated in principal coordinates

space after Bray-Curtis dissimilarity was performed on the untransformed abundance matrix. Temporal variability was then compared using an ANOVA with the individual effects of order and reach and their interaction. All ANOVAs were performed for both the full and revised sites, with stream identity fitted as a random blocking factor in each ANOVA; all ANOVAs were fitted by restricted maximum likelihood.

Exploratory data analysis was conducted prior to any model fitting to determine if the data met test assumptions (Zuur et al. 2010). For the full sites analyses, numerical summaries demonstrated an unbalanced design, with equal representation of restored and unrestored reaches but a large disparity in the number of samples between headwaters and mainstems for each of the taxonomic richness and diversity (headwater n = 38, mainstem n = 62), spatial dissimilarity (headwater n = 19, mainstem = 31), and temporal variation (headwater n = 10, mainstem n = 16) analyses. Additionally, the assumption of homogeneity of variance was violated for the taxonomic richness and diversity and the spatial dissimilarity analyses. The unbalanced design was greatly reduced for the revised sites analyses: taxonomic richness and diversity (headwater n = 30, mainstem n = 40), spatial dissimilarity (headwater n = 15, mainstem = 20), and temporal variation (headwater n = 8, mainstem n = 10); however, the assumption of homogeneity of variance was still violated. To better meet the assumption of equal variance, taxonomic richness was *ln*-transformed, taxonomic diversity was square root-transformed, and spatial dissimilarity was *ln*-transformed for all analyses. Along with using transformations to response variables to meet model assumptions, I used Type III sums of squares for evaluating main and interactive effects of factors included in the ANOVA. Swan and Brown (2017, 2018) used Type I sums of squares, which are inadequate for unbalanced and multi-factor designs with interactions between or among factors (Shaw and Mitchell-Olds 1993, Quinn and Keough 2002).

124 Model assumptions were inspected graphically, and significance was considered at $P <$
125 0.050. I removed the spring sample from the restored reach of site 227 from analyses because
126 there was no corresponding sample from the adjacent reach, which would have precluded paired
127 comparisons of restored-adjacent sites; however, I did not remove any sites prior to fitting the
128 full and reduced model ANOVAs set by Swan and Brown (2018). All analyses were conducted
129 using R version 3.5.3 (R Core Team 2019) with the nlme (version 3.1-139, Pinheiro et al. 2019)
130 and vegan (version 2.5-4, Oksanen et al. 2019) packages; data and R code are deposited in the
131 figshare repository (10.6084/m9.figshare.6448010). Given I made necessary modifications to the
132 analyses written by Swan and Brown (2017, 2018), later comparisons between the re-analysis
133 presented here and the results presented by Swan and Brown (2017, 2018) will only be in terms
134 of statistical and ecological interpretation and not exact values of statistical tests.

135 **Results & Discussion**

136 There were no main or interactive effects of season, order, or reach on taxonomic
137 richness for either the full or revised sites analyses (Table 1, Figure 1). The full model of
138 taxonomic richness proposed in the erratum (Swan and Brown 2018) did not show any main or
139 interactive effects of season, order, or reach (Table 2); however, the reduced model of taxonomic
140 richness demonstrated an interaction between order and season ($F_{3, 80} = 4.105$, $P = 0.009$) and
141 significant main effects of season ($F_{3, 80} = 4.358$, $P = 0.007$) and treatment ($F_{1, 80} = 4.844$, $P =$
142 0.031). In contrast to taxonomic richness, taxonomic diversity varied by season for the full ($F_{3, 80}$
143 $= 12.267$, $P < 0.001$) and revised ($F_{3, 80} = 10.999$, $P < 0.001$) sites (Table 1, Figure 2). There
144 were no further main or interactive effects of season, order, or reach on taxonomic diversity for
145 either the full or revised sites (Table 1, Figure 2). Spatial dissimilarity did not vary by any of the
146 main or interactive effects of season and order for both the full and revised sites (Table 1, Figure

3). Additionally, temporal variation did not vary by the main effects of or interaction between reach and order for the full and revised sites (Table 1, Figure 4).

Differences in significant main effects or interactions within the full and revised sites in the re-analysis did not seem to be the result of increased variation in the revised sites. In fact, variance, as measured by 95% confidence intervals, was either similar or even reduced for each of local diversity, spatial dissimilarity, and temporal variability for the revised sites compared to the full sites (Figures 1-4). It is therefore unlikely that that revised sites analysis was unable to detect effects due to increased variation and more likely due to reduced statistical power associated with a smaller sample size or the true lack of an effect of restoration in this system.

Effectiveness of Local Restoration

I hypothesized that stream-channel manipulations would have a more consistent effect relative to the effect of riparian reforestation between headwaters and mainstems, with stronger effects of restoration in headwaters relative to mainstems. As there were no significant effects of restoration on any of the community metrics between headwaters and mainstems, this hypothesis could be invalid or, at a minimum, revised and re-tested in restoration experiments. I should note that I was unable to directly test this hypothesis because I was re-analyzing data from a previous study and the experimental design precluded any robust test to isolate the effects; however, the hypothesis was intended to guide criteria for site selection and reduce variation in restoration treatments among sites and not to necessarily or strictly compare the effects in-channel manipulations and riparian reforestation treatments on biodiversity in restored streams. Regarding evidence for stream-channel manipulations and other treatments for effective restoration, previous research suggests local habitat manipulations are ineffective for structuring communities and increasing biodiversity (Palmer et al. 2010). An emerging hypothesis is that

local factors, such as habitat complexity and water quality, are overwhelmed by regional factors, such as dispersal and position within the larger network (Heino 2013, Tonkin et al. 2014). Given restoration did not have an effect on any diversity measure of communities in either headwaters or mainstems, this could suggest either restoration was wholly inadequate for both headwaters and mainstems or that the larger network and regional species pool were already degraded (Sundermann et al. 2011), overwhelming any mitigating effects of restoration.

Restoration Ecology & Experimental Design

Restoration of the streams was done in isolation of the study design and prior to data collection, resulting in variation in the types of treatments applied to and the time since restoration of the streams (Swan and Brown 2017). Although Swan and Brown (2017) noted this limitation of their study, they failed to acknowledge they could have better controlled for this variation by setting strict criteria for site selection and inclusion, which informed my hypothesis and was the foundation for my re-analysis. This concern was briefly acknowledged in the erratum (Swan and Brown 2018), where the data quality control process removed sites if they only received riparian reforestation treatments without at least one of either the bank stabilization or in-channel manipulation treatments; however, Swan and Brown (2017, 2018) proceeded to analyze data from sites receiving any combination of restoration treatments, despite suggesting that in-stream modification treatments would have stronger effects on communities relative to riparian reforestation (Swan and Brown 2018). Setting a more stringent criterion for site inclusion, as was done in this re-analysis of the revised sites, would have reduced the variation in the applied restoration treatments and provided a more balanced and adequate experimental design.

192 The inconsistent application of restoration treatments prohibited a robust evaluation that
193 could have been possible with a factorial experiment; therefore, the singular and interactive
194 effects of the restoration treatments in the study system remain untested. This further complicates
195 the indiscriminate usage of "restoration" by Swan and Brown (2017, 2018) and in the full sites
196 analysis presented here, as the underlying mechanism of restoration on the stream invertebrate
197 communities remains an unknown quantity. Identifying how individual and combinations of
198 restoration treatments affect stream communities would provide valuable insight for maximizing
199 the effectiveness of restoration efforts. In the absence of this knowledge, reducing the variation
200 in which restoration treatments were applied to the streams, as done with revised sites analysis,
201 arguably would have been a better avenue.

202 *Contradicting Results*

203 Restoration was not found to have a significant effect on local diversity, spatial
204 dissimilarity, or temporal variability of stream invertebrate communities between paired restored
205 and unrestored reaches in headwaters and mainstems. These results presented here, not exact
206 values of test statistics but in terms of interpretation, directly contradict the results presented in
207 the original paper (Swan and Brown 2017) and in the erratum (Swan and Brown 2018; Table 3).
208 This is concerning, as any data management and analytical errors in the original paper were
209 supposedly resolved in the erratum (Swan and Brown 2018); however, the discrepancies can be
210 partially explained by the erroneous reporting and implementation of statistical analyses. First, as
211 was noted above, Swan and Brown (2017, 2018) analyzed an unbalanced design with unequal
212 variance using an ANOVA with Type I sums of squares, when transformations to response
213 variables were necessary to better meet test assumptions and Type III sums of squares were more
214 appropriate for investigating the main and interactive effects (Shaw and Mitchell-Olds 1993,

Quinn and Keough 2002). This renders the results presented by Swan and Brown (2017, 2018) baseless, as violation of test assumptions makes any results derived from that test invalid. Second, the fitting of the random effects in the ANOVAs was incorrect. Swan and Brown (2018) reported fitting stream identity as a random effect. With the R code provided with the erratum (Swan and Brown 2018: Supporting Information), each site-by-reach combination was fitted as a random effect, despite adjacent and restored reaches being separated by < 10 m (Swan and Brown 2017). Fitting stream identity alone would have been a more appropriate fitting of a random effect that reflected the non-independence of the reaches at such small spatial scales, and, importantly, an accurate implementation of the written methods. Third, removal of the restored site without a paired observation is necessary for the fundamental goal of the study: comparing community diversity between paired restored and adjacent reaches across headwater and mainstem streams. Without removing the site, comparisons would be made to an unpaired reach, violating the experimental design and central goal of the study.

Finally, and of greatest concern, is the wholesale disagreement between the reported analytical procedure and what was actually conducted when analyzing temporal variability. Temporal variability was reportedly quantified as the mean distance to the group centroid (Anderson et al. 2006) after applying a Bray-Curtis dissimilarity index on an untransformed abundance matrix (Swan and Brown 2017). Results presented in the erratum were actually derived from the spatial median after a Jaccard index was applied to a presence-absence matrix (Swan and Brown 2018: Supporting Information). No random effect of stream identity was fitted for this ANOVA, although it would have been appropriate given the study design (Quinn and Keough 2002). None of the changes to the analytical procedure were reported in the erratum, and these alterations were only found upon scrupulous evaluation of the provided R code (Swan and

238 Brown 2018: Supporting Information). Without consulting the supporting information or if no R
239 code was provided, it would have been assumed the results presented in the erratum (Swan and
240 Brown 2018) were derived from the analysis described in the original study (Swan and Brown
241 2017), just with the corrected dataset; this assumption would have been incorrect.

242 *Ecological Implications & Prospective Suggestions*

243 Based on my re-analysis, I have concluded that, given restoration had no effect on any
244 community diversity metric, local restoration of streams can be ineffective if (1) both dispersal
245 and habitat quality are structuring biodiversity (Heino et al. 2015, Smith et al. 2015, Downes et
246 al. 2017) and (2) the larger regional community is already degraded (Sundermann et al. 2011).
247 Future research projects should experimentally manipulate different environmental factors to
248 evaluate the relative impact and effectiveness of restoration techniques, which could allow for
249 the identification of how in-stream and riparian reforestation treatments affect stream
250 biodiversity. More broadly, restoration of individual reaches of rivers and streams might be
251 insufficient to reach objectives of restoration projects (Bernhardt and Palmer 2011, Sundermann
252 et al. 2011). Streams and rivers will likely have great restoration success if a regional approach is
253 taken (Bernhardt and Palmer 2011), whereby heterogeneity among rivers and streams in
254 ecological, geographic, and spatial context is incorporated (Palmer et al. 2010, 2014, Booth et al.
255 2016). In this respect, I agree with Swan and Brown (2017, 2018), along with previous research,
256 that local manipulations are insufficient and a regional perspective is needed for more effective
257 restoration in rivers and streams.

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Tables

Table 1: ANOVA results for taxonomic richness, taxonomic diversity, spatial dissimilarity, and temporal variability. Models were fitted using restricted maximum likelihood and with Type III sums of squares for estimating main and interactive effects of factors.

Source of Variation	Full Sites				Revised Sites			
	numDF	**denDF**	**F**	***P***	**numDF**	**denDF**	**F**	***P***
Taxonomic Richness								
Season	3	73	1.991	0.123	3	47	1.411	0.251
Order	1	11	1.419	0.259	1	7	0.004	0.951
Reach	1	73	0.730	0.396	1	47	0.398	0.531
Season x Order	3	73	1.641	0.187	3	47	0.553	0.649
Season x Reach	3	73	0.045	0.987	3	47	0.143	0.934
Order x Reach	1	73	0.308	0.581	1	47	1.375	0.247
Season x Order x Reach	3	73	0.083	0.969	3	47	0.149	0.930
Taxonomic Diversity								
Season	3	73	12.267	< 0.001	3	47	10.999	< 0.001
Order	1	11	2.073	0.178	1	7	0.253	0.630
Reach	1	73	0.088	0.768	1	47	0.085	0.772
Season x Order	3	73	2.477	0.068	3	47	1.272	0.295
Season x Reach	3	73	0.323	0.809	3	47	0.284	0.837
Order x Reach	1	73	0.139	0.710	1	47	0.917	0.343
Season x Order x Reach	3	73	0.313	0.816	3	47	0.149	0.930
Spatial Dissimilarity								
Season	3	31	1.146	0.346	3	20	0.425	0.737
Order	1	11	0.062	0.808	1	7	0.007	0.934
Season x Order	3	31	1.356	0.275	3	20	0.306	0.821
Temporal Variability								
Order	1	11	0.030	0.866	1	7	0.957	0.361
Reach	1	11	1.906	0.195	1	7	2.693	0.145
Order x Reach	1	11	2.054	0.180	1	7	4.718	0.066

Table 2: ANOVA results for the full and reduced models of taxonomic richness proposed by Swan and Brown (2017, 2018). Models were fitted using restricted maximum likelihood and with Type III sums of squares for estimating main and interactive effects of factors. The unpaired sample was not removed from the data prior to fitting the full and reduced model ANOVAs.

Source of Variation	**Full Model**				**Reduced Model**			
	numDF	**denDF**	**F**	***P***	**numDF**	**denDF**	**F**	***P***
Season	3	74	2.478	0.068	3	80	4.360	0.007
Order	1	11	0.925	0.357	1	11	1.579	0.235
Reach	1	74	1.409	0.239	1	80	4.844	0.031
Season x Order	3	74	2.359	0.078	3	80	4.105	0.009
Season x Reach	3	74	0.152	0.928	N/A	N/A	N/A	N/A
Order x Reach	1	74	0.979	0.326	1	80	3.530	0.064
Season x Order x Reach	3	74	0.071	0.976	N/A	N/A	N/A	N/A

Note: N/A indicates a factor or interaction that was removed in the reduced model.

Table 3: Comparison of ANOVA results between the initial study (Swan and Brown 2017), the erratum (Swan and Brown 2018), and if results are consistent upon re-analysis in the full (Full) or revised (Revised) sites analyses. Support is denoted as: Yes = consistent with both the initial study and the erratum; No = inconsistent with both the initial study and erratum; 2017 = only consistent with the initial study; 2018 = only consistent with the erratum. Spatial dissimilarity was not re-analyzed in the erratum, so results are only provided for the initial study. The three-way season-by-order-by-reach interaction (indicated by *) was considered marginally significant by Swan and Brown (2017). Bold values in the Supported column indicate differences in statistical significance between the initial study and/or erratum and the re-analysis.

Source of Variation	**Swan and Brown 2017**		**Swan and Brown 2018**		**Supported**	
	F	**P**	**F**	**P**	**Full**	**Revised**
Local Diversity						
Season	8.09	< 0.001	16.47	< 0.001	**No**	**No**
Order	0.01	0.919	2.00	0.18	Yes	Yes
Reach	0.07	0.791	1.43	0.22	Yes	Yes
Season x Order	7.34	< 0.001	3.87	0.042	**No**	**No**
Season x Reach	0.44	0.728	N/A	N/A	2017	2017
Order x Reach	0.15	0.697	3.47	0.064	Yes	Yes
Season x Order x Reach	2.70	0.051*	N/A	N/A	**2017**	**2017**
Spatial Dissimilarity						
Season	3.85	0.015	N/A	N/A	**No**	**No**
Order	5.60	0.022	N/A	N/A	**No**	**No**
Season x Order	1.50	0.220	N/A	N/A	Yes	Yes
Temporal Variability						
Order	6.71	0.017	0.88	0.36	**2018**	**2018**
Reach	1.67	0.210	3.04	0.095	Yes	Yes
Order x Reach	4.62	0.043	6.33	0.020	**No**	**No**

Note: N/A indicates a factor or interaction that was removed in the reduced model or not analyzed by Swan and Brown (2018)

Figures

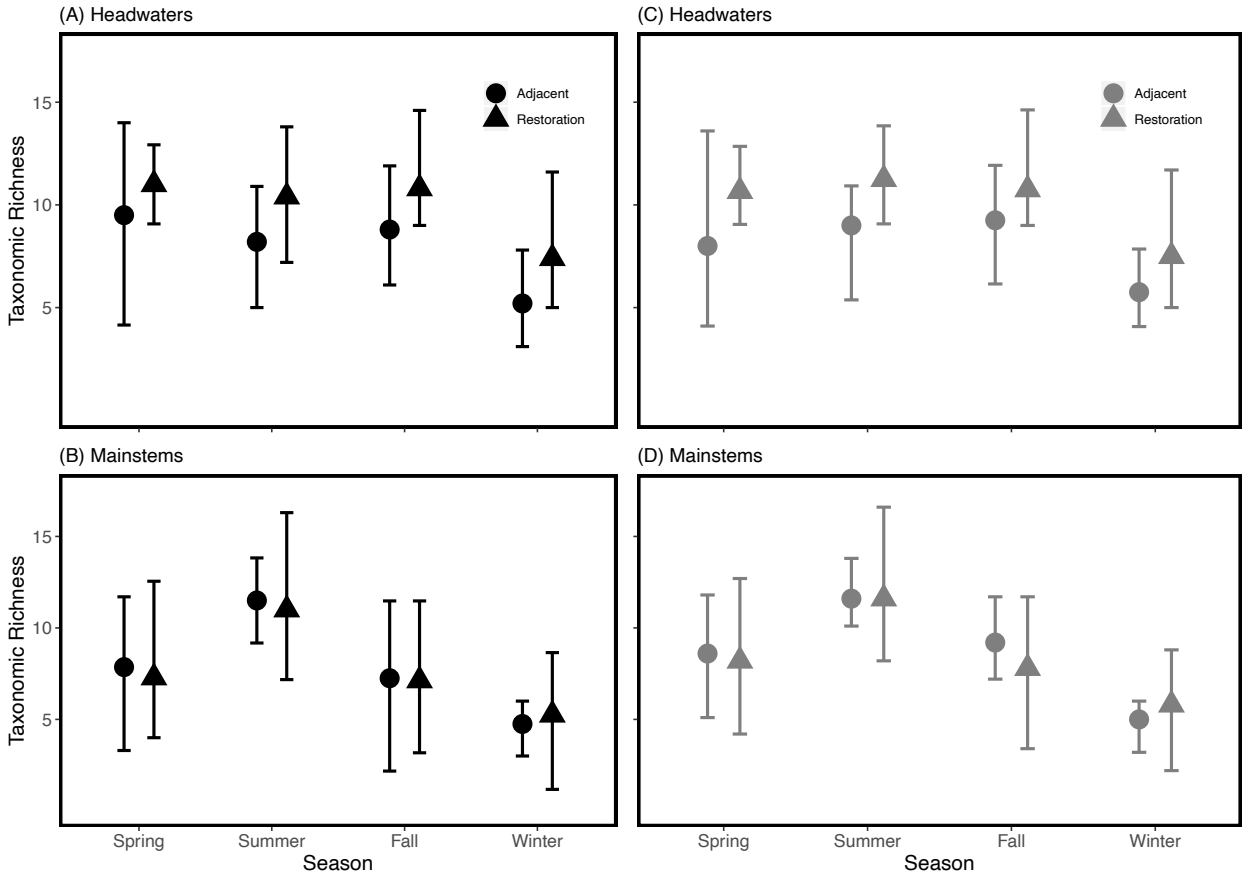

Figure 1: Plots of taxonomic richness in headwaters (A and C) and mainstems (B and D). Taxonomic richness is reported for restored (triangles) and adjacent (circles) sites; values from the full sites are reported in black (A and B), while those from the revised sites are reported in grey (C and D). Points represent mean ± 95% CI.

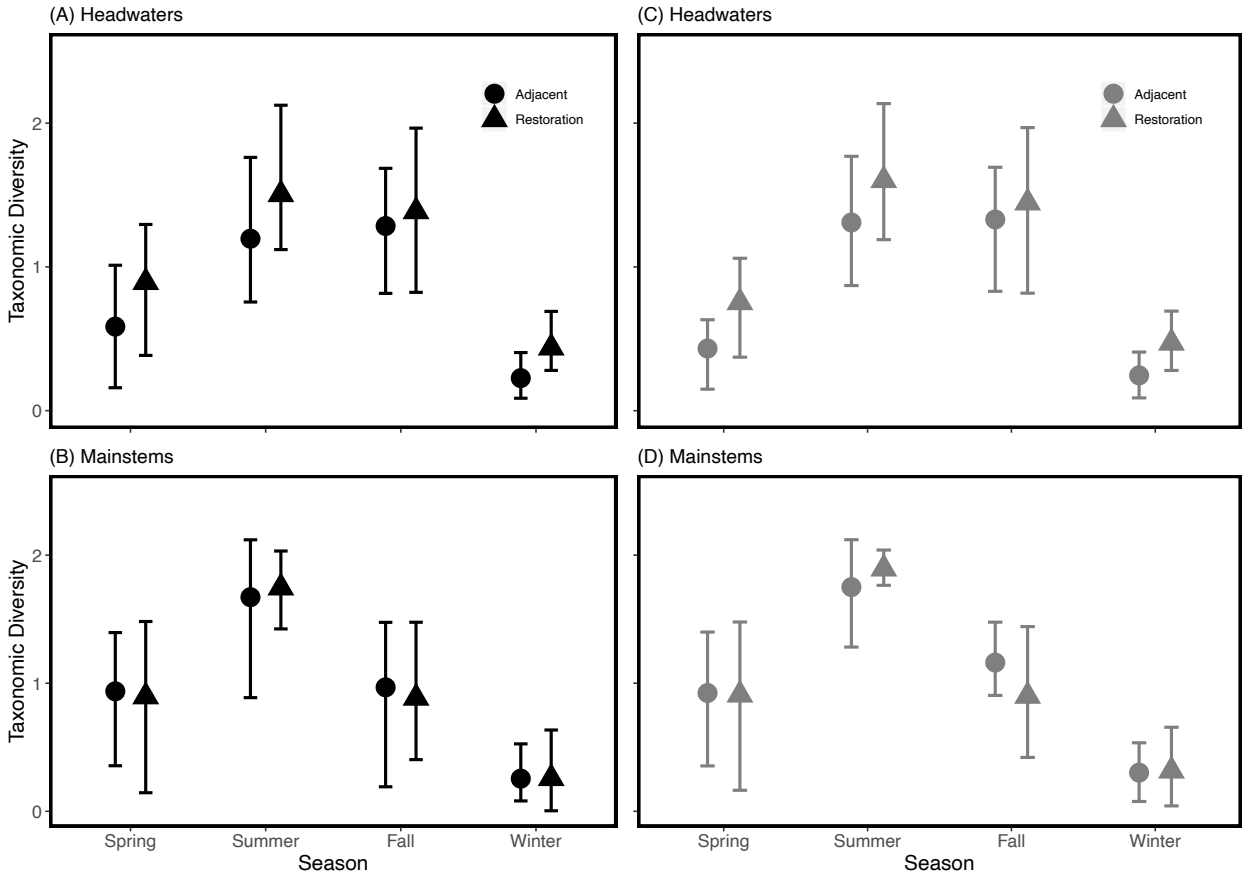

Figure 2: Plots of taxonomic diversity, calculated as Shannon's diversity, in headwaters (A and C) and mainstems (B and D). Taxonomic diversity is reported for restored (triangles) and adjacent (circles) sites; values from the full sites are reported in black (A and B), while those from the revised sites are reported in grey (C and D). Points represent mean ± 95% CI.

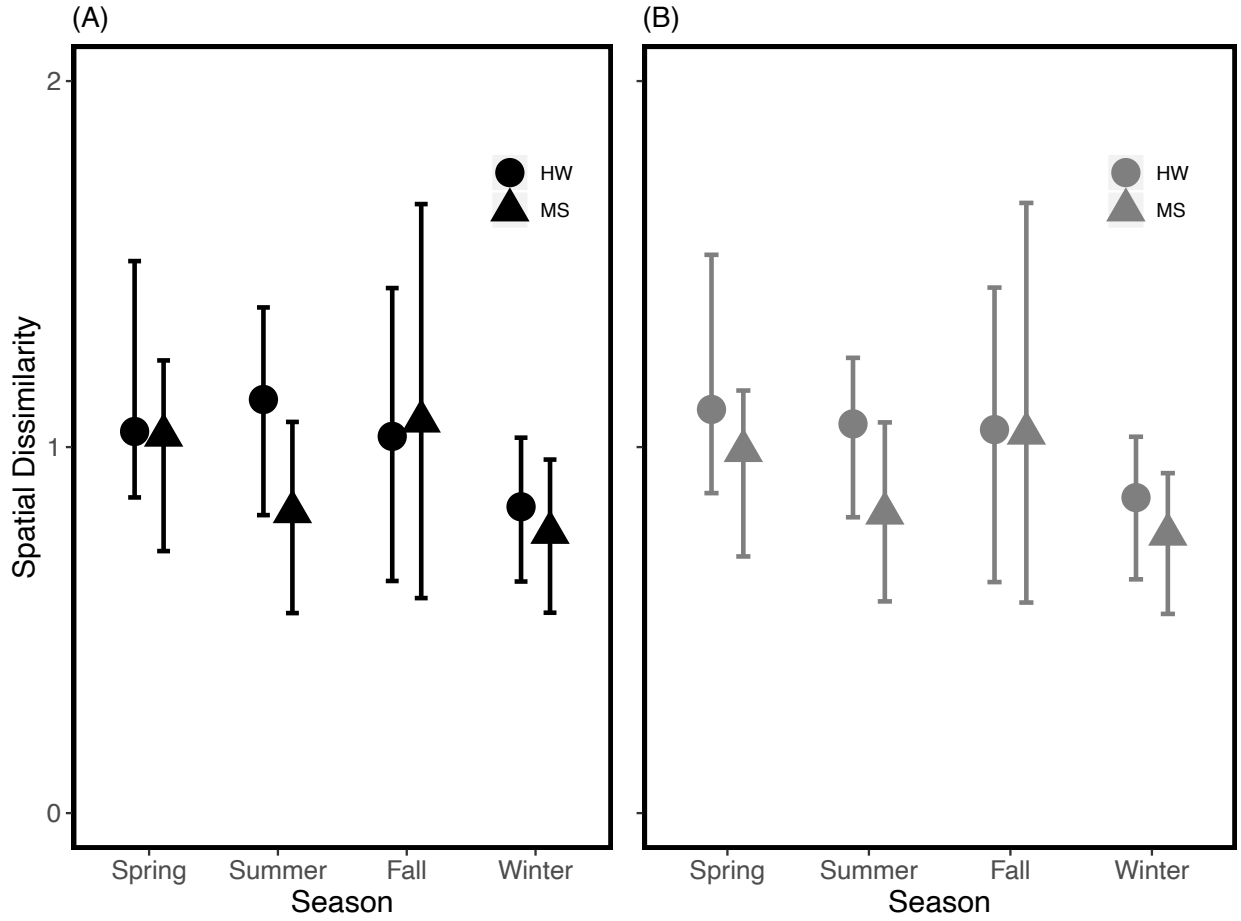

Figure 3: Plots of spatial dissimilarity. Estimates are reported for headwaters (HW, circles) and mainstems (MS, triangles); values from the full sites are reported in black (A), while those from the revised sites are reported in grey (B). Points represent mean ± 95% CI.

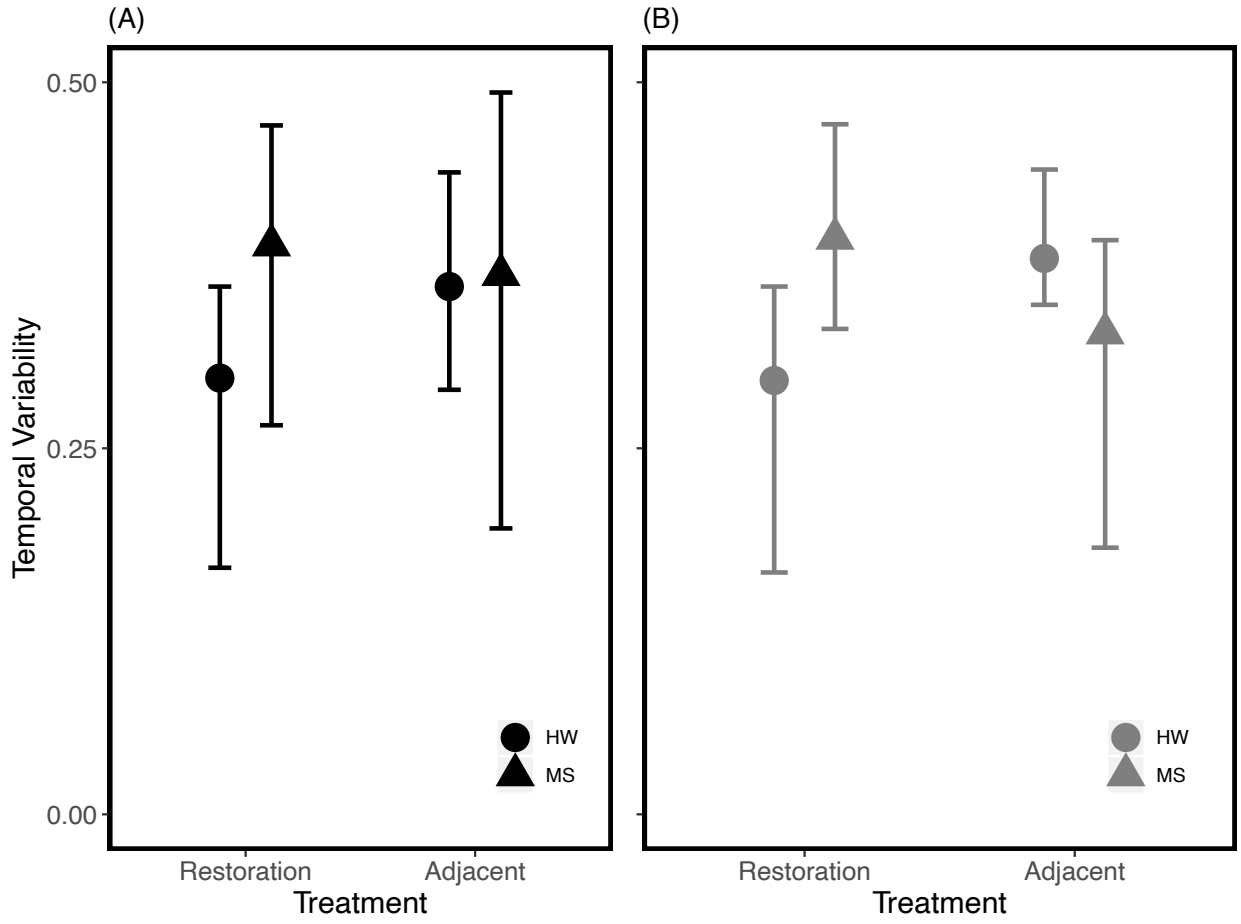

Figure 4: Plots of temporal variability. Estimates are reported for headwaters (HW, circles) and mainstems (MS, triangles); values from the full sites are reported in black (A), while those from the revised sites are reported in grey (B). Points represent mean ± 95% CI.