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RESEARCH ARTICLE

Natural-Channel-Design Restorations That Changed Geomorphology Have Little Effect on Macroinvertebrate Communities in Headwater Streams

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Abstract

Stream restorations that increase geomorphic stability can improve habitat quality, which should benefit selected species and local aquatic ecosystems. This assumption is often used to define primary restoration goals; yet, biological responses to restoration are rarely monitored or evaluated methodically. Macroinvertebrate communities were inventoried at 6 study reaches within 5 Catskill Mountain streams between 2002 and 2006 to characterize their responses to natural-channel-design (NCD) restoration. Although bank stability increased significantly at most restored reaches, analyses of variation showed that NCD restorations had no significant effect on 15 of 16 macroinvertebrate community metrics. Multidimensional scaling ordination indicated that communities from all reach types within a stream were much more similar to each other within any given year than they were in the same reaches across years or within any type of reach across streams. These findings indicate that source populations and watershed-scale factors were more important to macroinvertebrate community characteristics than were changes in channel geomorphology associated with NCD restoration. Furthermore, the response of macroinvertebrates to restoration cannot always be used to infer the response of other stream biota to restoration. Thus, a broad perspective is needed to characterize and evaluate the full range of effects that restoration can have on stream ecosystems.

Key words: **Catskill Mountains, habitat, multidimensional scaling (MDS) ordination, stream restoration, streambank stability.**

Introduction

Stream restoration has received increasing attention in recent years because of a growing awareness that even mildly degraded streams can impair water quality and reduce biodiversity (Sala et al. 2000; Young 2000; Lepori et al. 2005). Restoration activities to improve degraded streams can range widely in their degree of modification from simple replanting efforts in the riparian zone to the complete redesigning of channel morphology (Bernhardt et al. 2005; Roni et al. 2008; Miller et al. 2009). Stream restorations that involve channel modification typically improve bed and bank stability, habitat heterogeneity, and sediment transport, which can directly and indirectly affect the health of local aquatic communities. Maximizing biodiversity or improving habitat for sport fish or endangered species is often a primary goal in stream restoration efforts (Young 2000), but others target key aspects of stream geomorphology with the assumption that local ecosystems will respond positively when stream conditions improve (Palmer et al. 1997; Doll et al. 2003). One of the most widely applied stream restoration methods used to establish stable stream geomorphology is known as natural-channel-design (NCD) restoration (Rosgen 1996). NCD methods and principles have been used in streams across the United States, yet only a few studies have quantified associated changes in aquatic communities (Nagle 2007). Baldigo et al. (2010) and Ernst et al. (2010) documented significant increases in the abundance and biomass of trout and improvement in many fish community and habitat metrics across several Catskill Region streams following NCD restorations. However, no studies have assessed the response of benthic macroinvertebrate communities explicitly to large-scale NCD stream restoration techniques.

NCD is a common stream improvement and bank stabilization approach that uses bankfull hydraulic-geometry data from nearby stable reference reaches to recreate natural channel dimensions, patterns, and profiles in the stabilized streams (hereafter referred to as "restored" streams) (Rosgen 1994,

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1996; Doll et al. 2003). The NCD approach generally includes removal of disturbances that are causing local channel instability, reshaping of unstable stream reaches into stable streams and floodplains, and installation of in-stream structures and planting of riparian vegetation to stabilize stream banks and enhance fisheries habitat (Rosgen 1994, 1996; Doll et al. 2003). The physical process of restoration using NCD often has a large impact on the stream, including temporarily dewatering the streambed and moving large quantities of earth. Restored channels often have higher pool-riffle ratios (NRCS 2007), and are deeper and narrower (Ernst et al. 2010) than before treatment. Habitat heterogeneity and in-stream refuges can increase considerably after restoration (Klein et al. 2007; Ernst et al. 2010). Improvements can be short-lived, however (Rosgen 2006; NRCS 2007), and failures due to poor designs are not uncommon (Nagle 2007; Lave 2009). Given its emphasis on channel morphology, improvement of faunal communities is generally not a direct consideration in NCD principles. However, biological enhancements that follow geomorphic restoration are often a stated secondary goal (Doll et al. 2003). Stabilized channel geometry and normalized geomorphic processes from NCD restoration should improve the overall structure and function of degraded stream channels.

NCD restoration has been criticized as over-simplifying the complexity of fluvial systems (Doyle et al. 1999; Kondolf 2006; Simon et al. 2007). Critics argue that the form-based classification system underlying NCD restoration ignores the process of alluvial streams adjusting to varying energy and material inputs (Doyle et al. 1999; Simon et al. 2007); that channel instability cannot be predicted nor mitigated based on channel form (Juracek & Fitzpatrick 2003; Kondolf 2006; Simon et al. 2007); that channel form can change naturally over time (Juracek & Fitzpatrick 2003); and that bankfull stage, a key parameter of NCD restoration, is difficult if not impossible to identify in the unstable streams targeted by NCD restoration (Simon et al. 2007). Despite the controversy, NCD restoration principles continue to be widely applied. Few investigations, however, have evaluated their overall effectiveness in creating stable and sustainable stream channels (e.g. Nagle 2007; Lave 2009) and none has assessed the response of macroinvertebrate communities to NCD restoration.

Except for freshwater mussels, benthic macroinvertebrates are rarely a target of stream restoration efforts (NRC 1992; Strayer 2006); however, they are commonly used as surrogates for broader ecosystem responses in stream restoration projects (Doll et al. 2003; Miller et al. 2009; Palmer et al. 2010). Stream macroinvertebrates are generally recognized as good indicators of ecosystem health; they are numerous, easy to collect and identify, extremely varied in their range of tolerance to environmental degradation, and may respond more rapidly to degradation than other taxa such as fish (Merritt et al. 2008). Numerous metrics have been developed using stream invertebrates as bioindicators (Hilsenhoff 1987; Novak & Bode 1992) and these metrics are used in the vast majority of rapid-assessment programs across the United States (Southerland & Stribling 1995). Although the response of stream macroinvertebrate communities to restoration efforts that increase habitat heterogeneity should be positive (O'Connor 1991; Miller et al. 2009), observed changes have been inconsistent or vary with the scale and specific metrics assessed (Brooks et al. 2002; Muotka et al. 2002; Lepori et al. 2005; Palmer et al. 2010).

In this paper, we examine how macroinvertebrate communities respond to changes in habitat associated with NCD restorations. Ernst et al. (2010) found that streambank stability increased, stream-channel dimensions became narrower and deeper and had more pools, and shade coverage decreased in several Catskill Mountain streams following NCD restoration efforts. Although the literature is equivocal, we hypothesized that macroinvertebrate assemblages would change following restoration in these streams, given the magnitude of documented changes in stream habitat for our study reaches. Increased streambank stability was expected to cause a decrease in small, multivoltine taxa characteristic of unstable environments (Gurtz & Wallace 1984). Increased pool depth and frequency should increase species richness through greater habitat heterogeneity (Cowx et al. 1984; Brasher 2003). Decreased shade coverage should lead to higher macroinvertebrate densities (Hawkins et al. 1982; Snyder & Johnson 2006), higher scraper densities (Gurtz & Wallace 1984), and lower shredder densities (Webster et al. 1992). By reestablishing natural ecosystem function in restored streams, we expected invertebrate communities to have greater evenness, be less tolerant of degraded stream conditions, and more closely resemble an idealized macroinvertebrate community.

Methods

Study Scope and Area

In 1999, the U.S. Geological Survey (USGS), in cooperation with the New York City Department of Environmental Protection (NYCDEP) and the Greene County Soil and Water Conservation District (GCSWCD), began an 8-year study to evaluate the effects of NCD restoration projects on habitat and biota in the Catskill Mountains of southeastern New York. Large (0.4 to 3.0 km long) NCD restoration projects were implemented at six unstable reaches in five streams from 2000 to 2005 (Fig. 1). Benthic macroinvertebrate surveys were done annually within six sets of reference, treatment, and control reaches (74 to 243 m long) 0 to 2 years before restoration, and 1 to 5 years after treatment reaches were restored, depending on the stream (Table 1). Treatment reaches were not surveyed at Batavia Kill, Broadstreet Hollow Brook, or East Kill before restoration because these projects were completed before research on the macroinvertebrate assemblages began (Table 1). The reference reaches were unmodified reaches that were more geomorphically stable than treatment reaches and had forested riparian zones, low rates of erosion, and a well-balanced mix of pools and riffles. The degraded control reaches were unstable reaches located either upstream or downstream from the treatment reach with characteristics that were similar to the pre-treatment conditions at the restoration site, but which remained unmodified throughout the restoration process. Two treatment reaches were in Batavia Kill (upper and

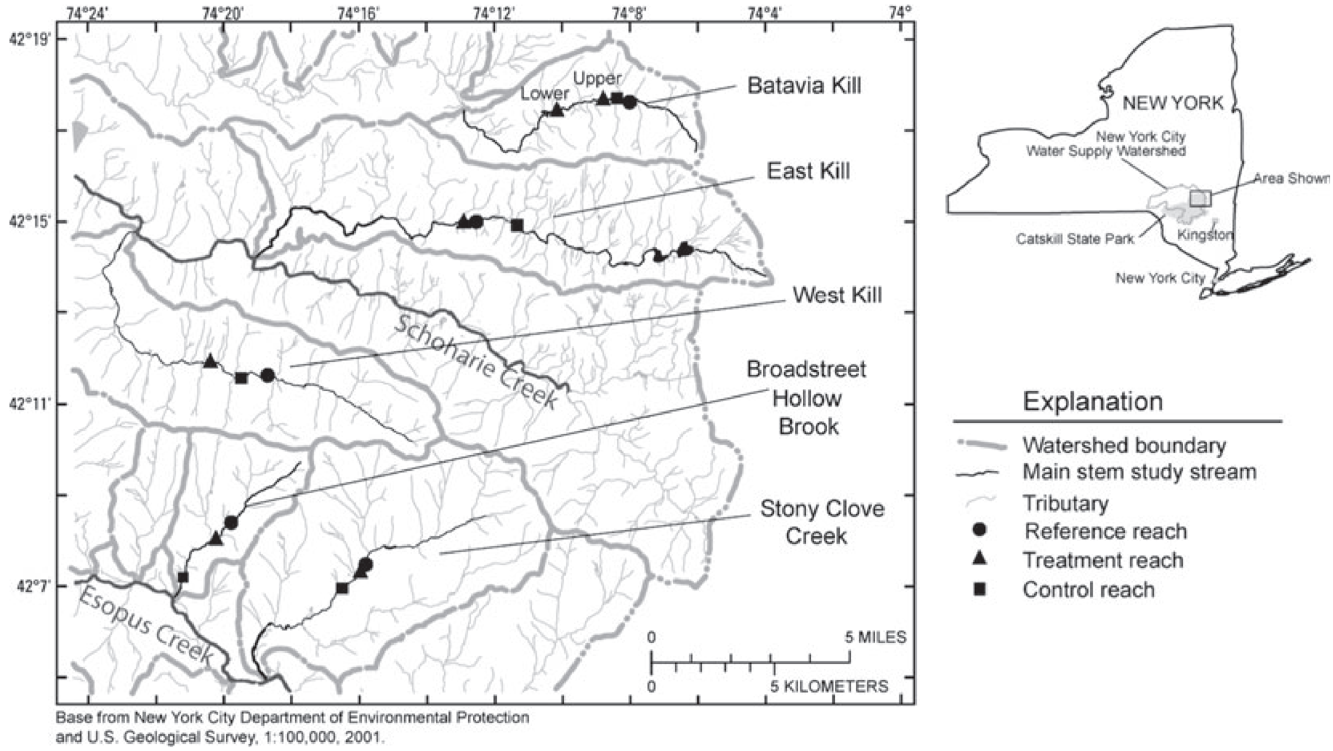

Figure 1. Location of six groups of reference, treatment, and control reaches on five stream restoration projects in the eastern Catskill Mountains, southeastern New York. Map modified from Baldigo et al. (2008).

Table 1. Schedule of restorations (X) and invertebrate samples at each of three reach types (treatment, T; reference, R; and control, C) on five streams in the Catskill Mountains.

	Sample Year												
Stream	*2000*		*2001*		*2002*		*2003*		*2004*		*2005*		*2006*
Broadstreet Hollow Brook		X			T, R, C		T, R, C		T, R				T
East Kill		X			T, R, C								
West Kill					T, R, C		T, R, C			X	T, R		T, R
Stony Clove Creek					T, R, C		T, R, C	X	R			X	T, R
Batavia Kill (upper)						X	T, R, C		T, R		T, R		T, R
Batavia Kill (lower)				X	T, R, C		T, R, C		T, R		T, R		T, R

Two different treatment reaches were restored within the Batavia Kill restoration-project reach; they shared the same reference and control reaches. All invertebrate sampling was conducted in July of each sampling year. All restorations were conducted during the fall, except for East Kill (summer 2000) and Stony Clove Creek (August 2003 and summer 2005).

lower) and one each in Broadstreet Hollow Brook, East Kill, West Kill, and Stony Clove Creek (Fig. 1). Both treatment reaches in Batavia Kill shared the same reference and control reaches. Details on stream characteristics are provided in Table 2. All macroinvertebrate sampling was conducted during July or August of each sample year, and all reaches within a stream were sampled within a week of each other each year. Most restorations were done in the fall; the East Kill treatment reach was restored in summer 2000 and the Stony Clove Creek reach was restored during August 2003 and summer 2005.

Project reaches were targeted for restoration because they had low geomorphic stability as indicated by high rates of bed and bank erosion; over-widened, shallow channels; frequent channel avulsions; homogeneous riffle habitat; and elevated turbidity from high bed-sediment loads and erosion of channel clays. Channel instability and excessive erosion and channel migration rates were the primary issues at all treatment reaches. Many headwater Catskill Mountain streams also have steep slopes, poorly drained soils, and unstable streamside glacial deposits. These conditions cause flashy stream flows which help account for the long history of damaging flood events in the region. The restorations were intended to reestablish channel structure, sediment transport equilibrium, and floodplain dynamics, and thereby improve water quality and reduce the damage to public infrastructure and private property caused by local flood events. The stream restorations used the Rosgen (1994, 1996) stream-classification system, regional hydraulic-geometry models, and NCDs based

Table 2. Characteristics of reference, treatment, and control reaches of five restored streams, including USGS station number, drainage area, mean daily discharge, and elevation.

Stream and Reach Type	*USGS Station Number*	*Drainage Area (km^2)*	*Mean Daily Discharge (m^3/s)*	*Elevation (m)*
Broadstreet Hollow				
Reference	0136223076	10.4	0.39	411
Treatment	0136223079	11.9	0.45	402
Control	0136223099	23.5	0.88	305
Stony Clove Creek				
Reference	0136234191	37.3	1.39	427
Treatment	0136234192	39.4	1.47	421
Control	0136234193	40.7	1.52	360
Batavia Kill				
Reference	0134984494	9.2	0.30	610
Treatment (upper)	0134984498	15.2	0.50	558
Treatment (lower)	01349845	18.7	0.61	549
Control	0134984496	12.0	0.39	567
East Kill				
Reference	01349674	50	1.10	530
Treatment	01349676	50.2	1.05	525
Control	01349672	47.9	1.00	541
West Kill				
Reference	01349752	37.3	1.10	518
Treatment	0134975915	41.4	1.23	497
Control	01349756	38.6	1.14	512

Table 3. Mean habitat metric values in restoration reaches before and after NCD restoration at five streams relative to changes at an upstream reference reach, percent change, and significance, based on a two-factor ANOVA of effects of restoration and stream on habitat variables.

Habitat Parameter	*Mean Initial Value*	*Mean After Restoration*	*Percent Change*	*p Value*
Depth (ft)	0.24	0.48	98	**0.004**
Thalweg depth (ft)	0.81	1.04	28	**0.021**
Width (ft)	26.5	21.7	−18	**0.012**
Pool-riffle ratio	0.92	1.63	77	**0.077**
Particle size (mm)	145	177	22	0.245
Substrate category	7.3	7.5	3	0.368
Fish cover (%)	3.2	5.8	82	0.187

Larger substrate categories indicate larger dominant particle sizes. Fish cover indicates percent of area that provides cover for a 10-in trout. Table modified from results presented in Ernst et al. (2010).
Bold font indicates p values < 0.10.

on bankfull-channel characteristics measured in nearby stable reaches of the same stream type. The restorations generally followed NCD principles, although bank hardening with rocks was sometimes used at the request of landowners to protect selected banks near homes and outbuildings. Restorations increased habitat heterogeneity (Table 3) and channel stability at most treated reaches (Ernst et al. 2010).

Macroinvertebrate Sampling

Invertebrate assemblages were sampled from riffles using a 5-m traveling kick method (Bode et al. 2002). All samples were collected by one of two researchers with comparable effort (5 minutes of active sampling). Samples were rinsed in a 500-μm mesh sieve and preserved in 95% ethanol. In the laboratory, specimens were identified to the lowest possible taxonomic level (generally genus or species, including Chironomidae) and enumerated. Apart from abundance, community metrics were calculated from 300-count subsamples of each kick sample to standardize metrics among reaches and years. Abundance estimates are measures of catch per unit effort. Sixteen commonly evaluated community metrics were chosen a priori as most likely to be affected by stream restoration, based on findings in literature. These metrics encompassed broader aspects of community composition (abundance, richness, % Chironomidae, relative abundance of the dominant three taxa, Shannon diversity, and Percent Model Affinity), sensitive taxa (EPT [Ephemeroptera-Plecoptera-Trichoptera] richness, Hilsenhoff's Biotic Index, Fine Sediment Biotic Index, Temperature Preference Metric, and number of intolerant taxa), and functional feeding groups (relative abundance of shredders, gatherers, filterers, scrapers, and predators).

Data Analyses

Macroinvertebrate responses to restoration were evaluated through analysis of variation (ANOVA) and multidimensional scaling (MDS) ordination analysis to test hypotheses that (1) macroinvertebrate assemblages would respond to habitat changes following restoration and (2) assemblages in treatment reaches would be similar to those in control reaches before restoration and similar to those in reference reaches after restoration. For each of the 16 macroinvertebrate community metrics, we performed a two-factor ANOVA to test for differences among four reach types (Reference, Control, Unrestored Treatment, and Restored Treatment) using data from all five streams, with stream as the second factor. When there was a significant effect of type, we used a Fisher's least significant difference procedure to determine which reach types had significantly different mean values. A significant difference between Unrestored and Restored Treatment reaches was attributed to the NCD restoration.

Differences in macroinvertebrate assemblages among reach types were also evaluated before and after restoration by assessing spatial patterns in macroinvertebrate-community composition and classifications (grouping of reaches with similar assemblages) using Primer-E version 6 software (Clarke & Warwick 2001) to develop a nonmetric MDS ordination of taxa relative abundance data (Shepard 1962; Kruskal 1964). Because of the large spread of abundances, we square root-transformed our data to compress higher values and prevent numerically dominant species from masking less abundant taxa. The MDS ordination generates an arrangement of samples in "species-space" according to the nonparametric ranks of their Bray–Curtis similarities (Clarke & Warwick 2001). Bray–Curtis similarities were estimated from the same data metrics using hierarchical cluster (group-average linking) analysis and permutation tests of similarity profiles ($p < 0.05$; Clarke & Warwick 2001). We expected the MDS ordinations to group control reaches with treatment reaches before restoration and apart from treatment reaches after restoration. Likewise, we expected the ordination to group reference reaches apart from treatment reaches before restoration and with treatment reaches after restoration.

We initially tried both a mixed-effects model and a paired BACI (before-after control-impact) model for analysis. The mixed-effects model gave the same results as our 2-factor ANOVA, and the BACI analysis relied too heavily on "Control" data as a surrogate for "Before" conditions, so we did not include these. Real-world restoration studies typically suffer from small sample sizes and often lack pre-restoration data, which limits the ability to use a full BACI design.

Results

Results of the ANOVA indicate that NCD restorations had little effect on any of the 16 invertebrate community metrics evaluated. Many metrics differed significantly among streams, but few metrics changed significantly following NCD restoration (Table 4). Percent gatherer, percent predator, Hilsenhoff's Biotic Index (HBI), and number of intolerant taxa all differed significantly among reach types, but only percent gatherer had a difference attributable to restoration (Table 4), with 16.7% more gatherers in the treatment reach after restoration than before. There was no significant interaction between reach type and stream for this metric. No other metrics were significantly different among reach types (Table 4). When all values from each reach type are considered together, the metric values among the four reach types are similar (Fig. 2).

Table 4. Summary of results of two-factor ANOVA tests of restoration effects on invertebrate community metrics using raw data, including differences among four reach types (Reference, Control, Unrestored Treatment, and Restored Treatment) and five streams.

Invertebrate Community Metric	*Significant Differences (p Value)*		
	Type	*Stream*	*Interaction*
Abundance	0.250	0.189	0.365
Richness	0.184	**0.035**	0.147
EPT richness	0.197	**0.042**	0.106
% Chironomidae	0.332	**0.063**	0.128
Dominant 3	0.116	**0.093**	0.298
H′	nd	nd	nd
% Shredder	0.250	0.390	0.189
% Gatherer	**0.031***	0.644	0.216
% Filterer	0.612	0.255	0.189
% Scraper	nd	nd	nd
% Predator	**0.014**	**0.029**	**0.085**
HBI	**0.064**	0.414	0.243
Fine sediment biotic index	0.601	**0.058**	0.198
Temperature preference metric	**0.069**	0.396	0.190
Intolerant taxa richness	**0.007**	**0.041**	**0.053**
Percent model affinity	0.232	**0.039**	0.179

nd, p value could not be determined.
Bold font indicates p values < 0.10.
* A significant difference between Unrestored and Restored Treatment reaches.

Results of the MDS ordination indicate that macroinvertebrate communities from all reaches within a stream were much more similar to each other within any given year than they were in the same reaches across years or within any type of reach across streams. Of the 21 statistically indistinguishable individual or groups of species assemblages, 11 comprised macroinvertebrate data from reference, treatment, and/or control reaches in the same stream during the same year, and only one of the groups included data from the same stream reach in different years (Broadstreet Hollow Brook reference reach, 2002–2004). Data from Stony Clove Creek demonstrate this pattern most clearly; reference, treatment, and control reaches were clustered by year based on their species assemblages, and reaches within each of these clusters were not significantly different ($p > 0.05$) from each other (Fig. 3a). Data from West Kill show the same pattern; all reach types within each year were statistically indistinguishable based on their species assemblages, except for the 2003 treatment site (Fig. 3b). The species assemblage in this reach was at least 50% similar to assemblages in the other 2003 reaches (and to all the 2002 reaches), but still differed significantly ($p < 0.05$). Species assemblages at all West Kill reach types sampled before restoration (2002 and 2003) were more similar to each other than to assemblages at all reach types sampled after restoration (2005 and 2006), suggesting greater year-to-year variability and a limited effect of restoration (Fig. 3b). Likewise at Batavia Kill, all reach types within each year were statistically indistinguishable based on their species assemblages, with the exception of the 2002 lower treatment reach, which had a unique assemblage (Fig. 3c). Species assemblages were statistically indistinguishable during 2005 and 2006, both of which are post-restoration years (Fig. 3c). At Broadstreet Hollow Brook, there were no significant differences among species assemblages in the reference reach in 2002, 2003, or 2004, but the assemblage in 2006 was significantly different ($p < 0.05$; Fig. 3d). Assemblages at control and treatment reaches were not significantly different between 2002 and 2003 (Fig. 3d).

Discussion

We found little evidence that NCD restorations affected macroinvertebrate assemblages in temperate, forested streams of the Catskill Mountain Region. Despite significant changes to stream habitat and fish assemblages at most restored reaches (Baldigo et al. 2008, 2010; Ernst et al. 2010), only 1 of the 16 macroinvertebrate metrics changed significantly at any reach following NCD restoration. Rather, macroinvertebrate assemblages from all reach types (reference, treatment, and control) within each stream were generally more similar to each other within a given year than to assemblages from the same reach type in different streams or years.

The strong similarity among reach types on the same stream within the same year indicates that watershed-scale factors are more relevant to macroinvertebrate community characteristics

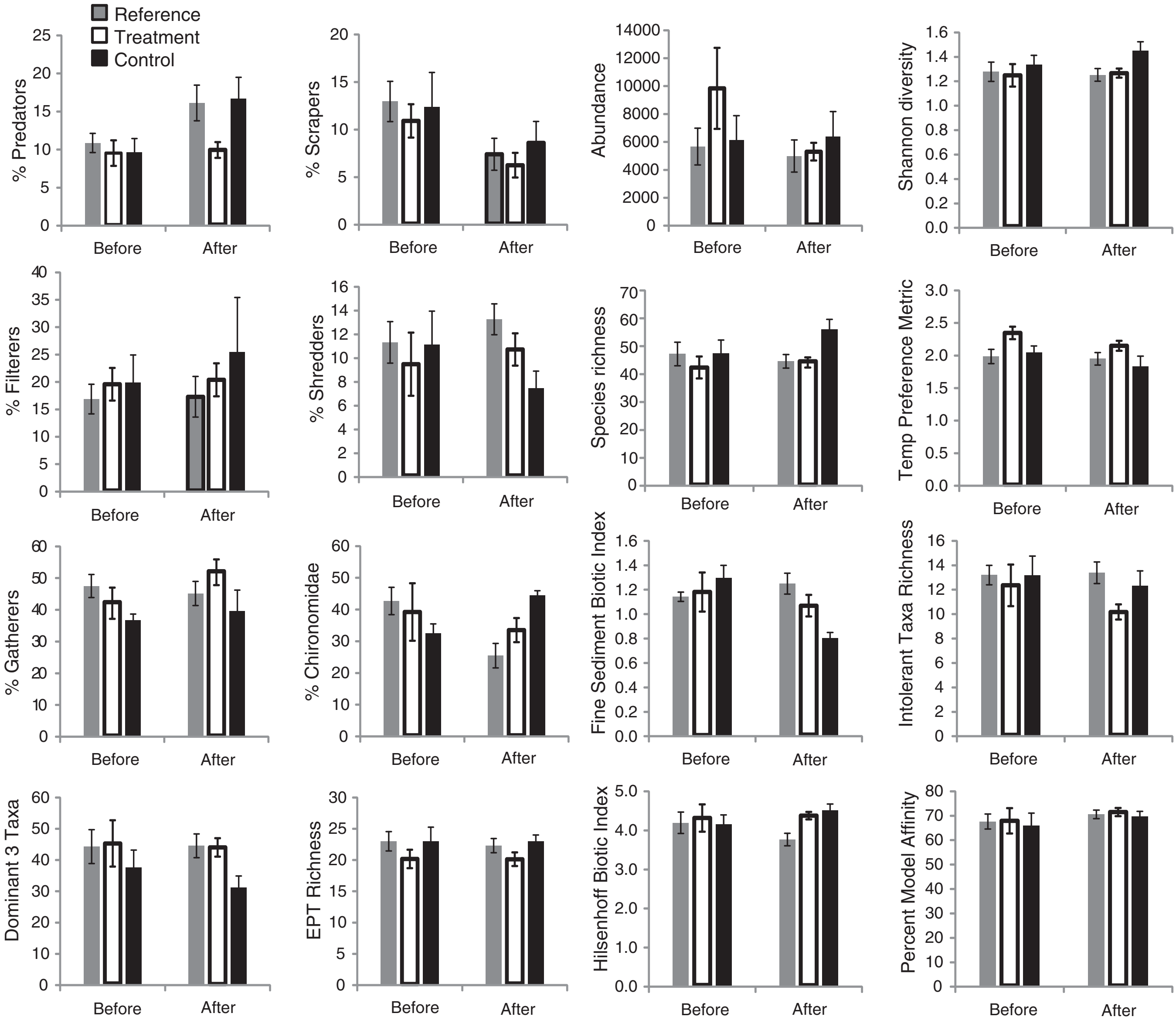

Figure 2. Mean value of 16 metrics measured for each reach type (Reference, Treatment, Control) at five study streams 0–2 years before and 1–5 years after restoration. Bars represent one standard error.

than is restoration, over the time frame of our evaluation. Reach-scale stream restorations such as those in this study may not be sufficient to affect macroinvertebrate communities if larger scale issues, such as land use, disturbance regime, or food and species resources, exist within the watershed. The lack of response contrasts with our initial hypotheses based on the geomorphic responses at these sites, but is consistent with other studies that found little or no response of macroinvertebrate communities to restoration. For example, Tullos et al. (2009) found that restorations in North Carolina Piedmont streams did not affect aquatic assemblages in urban settings compared to those in rural and agricultural settings because the modified hydrological regimes of urbanization constrained the assemblages. Similarly, Lepori et al. (2005) concluded that regional- and watershed-scale factors were stronger drivers of macroinvertebrate assemblages in Swedish streams than were the local-scale restorations. In a summary of published stream restorations from 1975 to 2008, Palmer et al. (2010) found that only 2 of 78 independent restoration projects resulted in increases in macroinvertebrate diversity. Other studies have found that restoration improvements can be short-lived because of basin-wide issues. For example, Moerke and Lamberti (2003) found that restoration of a channelized stream in the Midwest improved habitat quality immediately following restoration, but the improvements declined three years later because of continued high rates of erosion in the watershed. The reach-scale restoration efforts of our study did not address other upstream or basin-wide instability

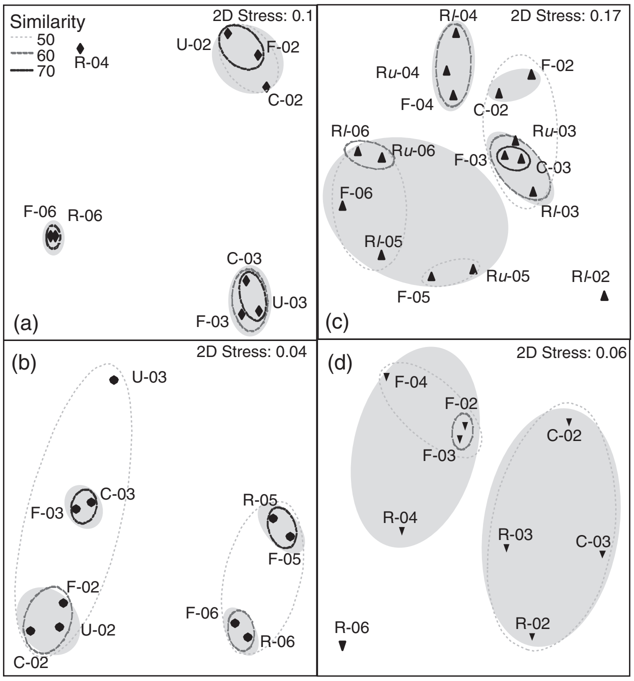

Figure 3. Ordination plot of macroinvertebrate assemblages based on square-root transformed relative abundance data from reference (F), unrestored treatment (U), restored treatment (R), and control (C) reaches in (a) Stony Clove Creek, (b) West Kill, (c) Batavia Kill (upper, *u*, and lower, *l*, treatment reaches), and (d) Broadstreet Hollow Brook. Stream codes indicate reach type and sample year (last 2 digits). The bubbles denote group membership (50, 60, and 70% Bray-Curtis similarity, $p < 0.05$) based on group-averaged cluster analysis. The shaded areas group assemblages that do not differ significantly ($p < 0.05$) from each other.

issues that may continue to adversely affect habitat and fish assemblages within restored reaches over the long term. Thus, local habitat changes, such as those produced by reach-level restoration, may not have an effect on local macroinvertebrate communities if larger scale problems still exist in the watershed.

Despite findings from this study and others, some channel modifications have positively affected responses in aquatic macroinvertebrate communities. For example, Pedersen et al. (2007) noted increased habitat heterogeneity along with community diversity and evenness 3 years after restoring meanders in a lowland stream in Denmark. Similarly, O'Connor (1991) found that woody debris additions increased habitat heterogeneity and macroinvertebrate species richness in an Australian lowland stream. Selvakumar et al. (2010) also found slight increases in several community metrics including HBI and EPT taxa 2 years after restoring an urban stream in Virginia, although the metrics indicated that water quality remained impaired. Other studies noted significant changes in at least some aspect of community health following changes in geomorphology or pool formation (Miller et al. 2009); however, a meta-analysis of 24 studies indicated that invertebrate community responses to stream restoration were weakest in channel-modification restorations (Miller et al. 2009). Although some types of stream restoration may have modest effects on macroinvertebrates, NCD restorations as applied here did not produce measureable changes in their communities. Our results may be limited, however, by low power caused by limited replication and sampling.

The scale of observation and the scale at which geomorphic processes influence macroinvertebrate communities are important considerations when evaluating their responses to restoration. We collected macroinvertebrates only in riffles according to standardized sampling procedures, which may have biased reach-level generalizations. The NCD restorations produced a significant increase in pool depth and availability in restored streams (Ernst et al. 2010), which contributed to changes in fish community metrics and increased salmonid biomass in most study streams (Baldigo et al. 2010). The restorations created new habitats (e.g. deep pools, backwaters) that could have permitted colonization by new macroinvertebrate taxa, but those habitats were not sampled. Focusing on riffle habitats provides a means to standardize the conditions under which samples are collected and allowed us to compare specific assemblages; however, such sampling reduces the range of habitat conditions over which a response is evaluated. Thus, extensive reach-level changes could have occurred, but not been detected by the limited habitat sampling.

A major goal of NCD restoration is to increase streambank stability, but our results show no macroinvertebrate response to stabilized streams. Studies linking specific streambank stability metrics to macroinvertebrate assemblages are rare; however, Sullivan et al. (2004) assessed 18 paired stable and unstable stream reaches in Vermont, and found that geomorphically stable reaches provided better physical habitat than similar, unstable reaches, and higher percentages of EPT taxa than did paired unstable reaches. This pattern did not hold true for reaches that were adjusting to a more stable geomorphology (Sullivan et al. 2004). Overall, the reach-level changes in stability in our study streams may not have occurred at a large enough scale to influence macroinvertebrate assemblages.

We draw two primary conclusions from this study: First, NCD restorations had limited effects on invertebrate communities 1–5 years after restoration in these streams. Overall, watershed-scale factors, stream communities (source populations), and annual variability in community composition seem to have greater influence on stream macroinvertebrate communities at the reach scale than did initial geomorphic condition or geomorphic modifications associated with the NCD restorations. This suggests that stream stability has little or no direct effect on macroinvertebrate communities, although this is speculative because it was not tested directly. Second, macroinvertebrate responses to NCD restorations did not reflect positive changes in geomorphology or fish communities quantified in most of the restored reaches. This is a particularly noteworthy result given the widespread use of macroinvertebrates as indicators of overall stream water quality or system health, often in the absence of fish or geomorphic assessment. Macroinvertebrate communities represent an important functional group in streams; however, their lack of response suggests that a broader perspective is needed to fully evaluate stream ecosystems and their response to NCD and other comparable stream restoration projects.

Implications for Practice

- Resource managers should not gauge the effectiveness of stream-channel restoration efforts on changes in macroinvertebrate assemblages. Macroinvertebrates cannot be used to infer responses among other stream biota, nor can they be used as the sole metric to assess the success of restoration that alters stream geomorphology.
- Evaluating stream responses to NCD restorations and other restorations that modify geomorphology requires a broad ecosystem perspective that includes both the physical characteristics of a system and a range of biota. Macroinvertebrates, fish, and geomorphology do not necessarily respond in unison.

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