

Invited Feature

Evaluating River Restoration¹

This Invited Feature comprises a collection of papers evaluating the ecological outcomes of channel reconfiguration restoration projects in the eastern United States and western Europe. To provide background and context, the first paper (Bernhardt and Palmer) begins by providing a brief summary of recent publications that have reviewed the scientific basis or ecological outcomes of multiple river restoration projects. The types of projects discussed in these previous publications are diverse, but we focus primarily on publications that have evaluated channel reconfiguration projects.

The other six papers in this collection present new findings and insight into the effectiveness of channel-based restoration projects. Unfortunately, the conclusions of these papers are sobering. Empirical evaluation of a variety of channel-based restoration projects discovered little evidence of ecologically successful outcomes. Violin et al. find that urban stream restoration efforts in the southeastern United States had no demonstrable effect on habitat diversity or on macroinvertebrate communities. More disheartening, Louhi et al. find that several restored streams in Finland have stream invertebrate communities that are depauperate relative to unrestored upstream reference reaches even 15 years following restoration. Sundermann et al. find that most of the restored streams they sampled in Germany show no measurable improvement in macroinvertebrate communities, and the few that do are close to intact, forested catchments.

Sudduth et al. report that restored urban streams in North Carolina have significantly higher temperatures than unrestored urban streams as a result of removing riparian trees to facilitate restoration projects. Filoso and Palmer show that efforts to reduce the flux of nitrogen to coastal waters through hydrogeomorphic stream restoration approaches are rarely successful. Jähnig et al. document the existence of different perceptions of restoration success and show that, according to data from river restoration projects in Germany, water managers tend to be overly positive in their self-evaluation of restoration projects.

On the bright side, the papers in this Invited Feature provide insightful perspectives for learning from past efforts and implementing adaptive restoration. The last part of the paper by Bernhardt and Palmer discusses four factors that should inform future restoration efforts and considers the social context within which such efforts must be made.

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River restoration: the fuzzy logic of repairing reaches to reverse catchment scale degradation

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Abstract. River restoration is an increasingly common approach utilized to reverse past degradation of freshwater ecosystems and to mitigate the anticipated damage to freshwaters from future development and resource-extraction activities. While the practice of river restoration has grown exponentially over the last several decades, there has been little empirical evaluation of whether restoration projects individually or cumulatively achieve the legally mandated goals of improving the structure and function of streams and rivers. New efforts to evaluate river restoration projects that use channel reconfiguration as a methodology for improving stream ecosystem structure and function are finding little evidence for measurable ecological improvement. While designed channels may have less-incised banks and greater sinuosity than the degraded streams they replace, these reach-scale efforts do not appear to be effectively mitigating the physical, hydrological, or chemical alterations that are responsible for the loss of sensitive taxa and the declines in water quality that typically motivate restoration efforts. Here we briefly summarize this new literature, including the collection of papers within this Invited Feature, and provide our perspective on the limitations of current restoration.

Key words: catchment land use; evaluation; natural channel design; river restoration; streams; water quality.

INTRODUCTION

The water quality and biodiversity of freshwaters are declining globally. While high-profile examples like the historic “flaming Cuyahoga River of Ohio” or the desperately polluted Ganges may grab the headlines, the bulk of the damage to our global rivers falls into two categories: degradation resulting from major alterations to the landscape or from excessive water withdrawals and construction of dams (Nilsson et al. 2005, Dudgeon et al. 2006). While both types of degradation occur worldwide, the former is a common motivation for river restoration in wet climates such as in the eastern United States and much of Europe where high flows and polluted runoff may accompany land use change. The latter drives restoration efforts particularly in arid or semiarid climates including much of the western United States, the Middle East, Australia, and parts of Africa. Just as the source of degradation differs between these two types, so too do the restoration approaches.

Efforts to restore biodiversity and ecological function in rivers that have been dammed or subjected to flow diversions have included reestablishing part of the historic flow regime, removing levees to recover flood-plain functionality, scheduling water releases from reservoirs to restore native vegetation and riparian functions, and in some cases even removing flow blockages or reconnecting river reaches that have been fragmented (Bednarek 2001, Stromberg 2001, Doyle et al. 2005, Richter and Thomas 2007). While reducing diversions and removing dams have not uniformly been successful in restoring native flora and fauna, there are certainly success stories (e.g., Rood et al. 2003, Hall et al. 2010). Many of these involved restoring flows to river reaches that were “water starved” or had blockages that significantly limited upstream fish dispersal (Rood et al. 2005, Catalano et al. 2007) but were otherwise not seriously stressed by other factors such as uncontrolled runoff or non-point-source pollution.

Efforts to restore biodiversity and ecological function in streams and rivers degraded by substantial land use change or by human activities such as agriculture, development, and channelization have proved to be much more difficult, and reports of effectiveness monitoring (pre- and post-restoration quantitative sampling) of restoration outcome are rare (Bernhardt et al. 2005, Palmer et al. 2005). Yet, human activities

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leading to non-point-source pollution and channel degradation are among the most common motivations for undertaking stream restoration (Walsh et al. 2005, Bernhardt and Palmer 2007) often involving significant channel reconfiguration efforts. Typically these restoration projects focus on increasing channel stability and in-stream habitat by altering channel form and structure along a river reach (Niezgoda and Johnson 2005) with the hopes of restoring stream ecosystems (Rosgen 1996, Hey 2006). These channel-based or “hydromorphological” restoration projects are extremely common worldwide (e.g., Jähnig et al. 2009), and new research efforts to evaluate their ecological effectiveness are just beginning to be published (e.g., Tullis et al. 2009, Baldigo et al. 2010, Miller and Kochel 2010, Miller et al. 2010).

This Invited Feature presents a series of papers that evaluate the ecological outcomes of channel reconfiguration restoration projects in the eastern United States and western Europe. To provide background and context, we first provide a very brief summary of recent publications that have reviewed the scientific basis or ecological outcomes of multiple river restoration projects. The types of projects discussed in these publications are diverse, but we focus primarily on publications that have evaluated channel reconfiguration projects. Overviews of projects directed at restoring environmental flows or riparian zones are available elsewhere (e.g., see special journal issues associated with Arthington et al. [2010] and Mayer et al. [2010]).

Background

The literature on restoration effectiveness is growing. Several recent papers have attempted to synthesize reports of restoration evaluations (Roni et al. 2008, Miller and Kochel 2010, Miller et al. 2010, Palmer et al. 2010, Whiteway et al. 2010). After reviewing a wide variety of stream restoration projects (345 studies), Roni et al. (2008) concluded that few firm conclusions could be reached “because of the limited information provided on physical habitat, water quality, and biota and because of the short duration and limited scope of most published evaluations.” Whiteway et al. (2010) offer a more optimistic perspective, finding that, of 211 river restoration projects aimed at restoring salmonid fish, the majority of projects led to an increase in the abundance or biomass of fish although, we note, that most projects do not distinguish aggregation effects from population-level increases. Baldigo et al. (2010) used a thorough before–after–control–impact (BACI) assessment design and found that in four of six “natural channel design” projects, increases in salmonid biomass were also accompanied by small increases in young-of-the-year trout, suggesting the potential for population-level effects. Because such changes were negligible or absent in two of the projects they studied, Baldigo et al. (2010) emphasize that positive ecological responses from channel design projects should not be presumed and that increases in channel stability do not necessarily lead

to improved fish habitat. It is important to recognize that most projects analyzed in these syntheses were less than a decade post-construction. The lack of demonstrable ecological improvements may thus be due either to a lack of effective physical, chemical, or hydrologic remediation or merely to a lack of time for recovery. Even if restoration efforts could be reliably expected to generate ecological improvements within subsequent decades, such time lags between implementation and ecological recovery must temper expectations that restoration efforts can effectively and quickly mitigate river degradation elsewhere.

Many evaluations of stream restoration projects rely on diversity and species composition of macroinvertebrates as an indicator of ecological outcome and while a growing number of publications report on the outcome of one or a few projects (e.g., Yu et al. 2010), there are also some recent syntheses. Palmer et al. (2010) found that, of 78 channel restoration projects, only two documented a significant improvement in macroinvertebrate species richness or compositional shift toward reference conditions. While they did not measure invertebrate response, the work by Miller and Kochel (2010) was sufficiently comprehensive (monitored 26 restoration projects), that it is worth noting here. Their geomorphic monitoring programs led them to conclude that attempts to restore rivers through channel reconfiguration are extremely difficult because they can lead to major channel adjustments and failures of in-stream structures: “there is little evidence from the examined projects in North Carolina that reconfiguring straightened and/or incised channels along highly dynamic rivers will speed the rate of recovery” (Miller and Kochel 2010:1690).

Contributing papers

Unfortunately, the conclusions of papers comprising this Invited Feature are sobering. Empirical evaluation of a variety of channel-based restoration projects documented little evidence of ecologically successful outcomes. Violin et al. (2011) found that urban stream restoration efforts in the southeastern United States had no demonstrable effect on habitat diversity or on macroinvertebrate communities. More disheartening, several restored streams in Finland were found to have stream invertebrate communities that were depauperate relative to unrestored upstream reference reaches even 15 years following restoration (Louhi et al. 2011). Restored urban streams in North Carolina were found to have significantly higher temperatures than unrestored urban streams as a result of removing riparian trees to facilitate restoration projects (Sudduth et al. 2011). Filoso and Palmer (2011) show that efforts to reduce the flux of nitrogen to coastal waters through hydrogeomorphic stream restoration approaches are rarely successful. Instead, N-removal capacity may depend more on N concentrations than restoration treatment, and projects are unlikely to provide signifi-

cant N-reducing benefits unless the “restoration” project involved converting the stream to a very different type of ecosystem such as a wetland–stream complex (i.e., the end result is a system that may not even have the ecological characteristics of a stream anymore). Despite a lack of measurable ecological improvement (*sensu* Palmer et al. 2005), most restoration practitioners consider their projects to be successful (Bernhardt et al. 2007, Jähnig et al. 2011).

ADAPTIVE RESTORATION

With a rapidly expanding body of literature evaluating river restoration effectiveness, we can move from asking “Why don’t we know more about river restoration success?” (Bernhardt et al. 2005, Palmer et al. 2005) to asking “Why aren’t river restoration projects more effective?” We offer some thoughts that were in part informed by the papers in this Invited Feature, but we want to emphasize the value of past monitoring efforts. The published assessments of restored streams and river reaches have provided a great deal of information that can inform future project designs. Documented failures with good data are fodder for understanding what went wrong and why—indeed, as we emphasized in Palmer et al. (2005), learning from past efforts can be considered one of at least three forms of restoration success (i.e., in addition to ecological improvements and meeting stakeholder needs). As countries around the world move forward in developing restoration plans, we suggest that future projects should be informed by results from past efforts and ecological theory (O’Donnell and Galat 2007, Woolsey et al. 2007). In calling for “restoration reform,” Palmer (2009) discusses in depth factors that should drive river future restoration efforts, and Beechie et al. (2010) provide insightful perspectives on moving toward more process-based restoration. Here we mention four of those factors and perspectives that papers in this Invited Feature help illuminate.

First, much more emphasis needs to be placed on site selection for restoration projects: spatial context may be one of the most important factors controlling stream restoration outcome. Sundermann et al. (2011) found that most of the restored streams they sampled in Germany showed no measurable improvement in macroinvertebrate communities; the few that did were close to intact, forested catchments. They suggest that the proximity of a restored stream reach to potential sources of colonists is an important determinant of restoration success, or, in other words, that the provision of habitat is insufficient if target taxa must disperse long distances within degraded channels or across altered landscapes to colonize a restored site. Another interpretation of these same data might be that river reaches in catchments with less total disturbance have a higher potential to recover from targeted efforts. Additionally, Filoso and Palmer (2011) show that position within a catchment (upland vs. lowland restoration project) may have a major impact on the

nitrogen removal capacity of restored Coastal Plain reaches in urban areas. They found that, during high flows when most of the N is exported, lowland reaches with gentle slopes and wider valleys have higher hydraulic retention and, therefore, capacity to retain N.

Second, given that a number of studies have now found no ecological improvement from channel reconfiguration projects and, in some cases, even found evidence of increased degradation (e.g., Tullos et al. 2009), future restoration approaches should keep earth-moving activities to a minimum, particularly if they include the removal of trees. The studies by Louhi et al. (2011) and Sudduth et al. (2011) both present evidence that restoration activities themselves actually lead to degradation that is not necessarily short lived. In a series of timber-floated streams in Finland, Louhi et al. (2011) found that a critical habitat, native bryophyte patches, did not recover, and thus limited the recovery of stream invertebrates that typically inhabit these patches. Potential colonists were present above the restored reaches, and thus this was not a case of dispersal limitation. Sudduth et al. (2011) document a very different type of habitat degradation post-restoration. They show that the urban streams they studied were significantly warmer with more sunlight penetrating the water post-restoration. The most extensive and expensive types of restoration projects (natural channel design, floodplain reconnection) require, at a minimum, significant earth-moving and temporary piping or rerouting of stream flow. Empirical evaluation of 24 restoration sites led Tullos et al. (2009) to conclude that channel reconfiguration acts as a “disturbance filter” such that taxa sensitive to disturbance were characteristic of their control reaches, whereas insensitive taxa were characteristic of restored reaches. They attributed this to the disturbance effect on food resources and channel stability. Acknowledging that this could be a temporary impact (Tullos et al. [2009] projects were 1–4 years old), the results of Sudduth et al. (2011; projects 1–6 years old), Louhi et al. (2011; some projects 12 years old), and Palmer et al. (2010; some projects 16 years old) suggest that the unintended consequences of restoration may persist for some time.

Third, restoration of streams and rivers should not be expected to alleviate problems generated throughout a catchment. The very problems that lead to stream degradation typically are catchment-scale problems (e.g., large amounts of impervious cover or land in agriculture). Projects that are small in scope simply cannot handle the level of impacts, and yet the vast majority of restoration projects are small and isolated. Even for projects that are large, streams simply cannot always “repair” problems created at broad scales. For example, Filoso and Palmer (2011) found that, across the restored sites they studied, nitrogen (N) export to downstream waters was reduced in restored reaches by a maximum of 11% during base-flow conditions, and 24% during high flows. However, this amounted to only ~5%

of the total N input into the catchment. Further, these “high” N removal values are for a stream that was converted to a stream–wetland complex. Some of the more traditionally restored stream reaches actually contributed more N to downstream waters than control reaches. As we have previously argued, these results highlight the need to reduce N sources on the land and improve land-based “best management practices” (BMPs; Bernhardt and Palmer 2007, Bernhardt et al. 2008).

Fourth, future efforts must shift from a focus on geomorphic structure and form to a focus on restoration of the hydrologic, geomorphic, and biological processes that maintain healthy stream ecosystems. There is a widespread practice of applying structural approaches to restore ecological communities rather than process-based approaches (Roni et al. 2002, 2008), and one of the most widely applied and controversial restoration approaches, natural channel design, focuses exclusively on matching the pattern, profile, and dimension of stream channels to pre-degradation or reference conditions (Rosgen 1994, Lave 2009). There are a few types of restoration activities that are explicitly designed to restore river processes. Most notably the dam and levee removal projects we mentioned at the outset of our article are designed to reinstate disrupted hydrological and sediment-flux processes along river networks or between rivers and their floodplains (Bednarek 2001, Hart et al. 2002, Stanley and Doyle 2003). Although not currently classified as river restoration, efforts to retain and detain storm waters in urban or agricultural catchments can similarly help restore a more natural flow regime. A variety of catchment BMPs and infrastructure improvements aimed at reducing contaminant loading to streams can also “restore” more natural chemical fluxes to degraded stream ecosystems. The current overemphasis on structural improvements to channels in restoration is likely a result of river restoration approaches emerging out of classic hydraulic engineering (see discussion in Lave [2009]).

CONCLUSIONS AND SOCIAL CONTEXT

In the same way that national and international plans for adaptive fisheries management have been difficult to implement (Francis et al. 2007, Walters 2007), adaptive restoration that allows for changes in the design and conceptualization of river and catchment projects will be very difficult to implement. Just as scientists are hesitant to break free of accepted theories, practitioners and managers are hesitant to experiment with new approaches. Additionally, regulatory agencies often incorporate restoration method requirements into policy instruments (Lave 2009), and changes to such policies often lag a decade or more behind the science. Efforts to streamline the regulatory process can also severely constrain opportunities for developing, implementing, and testing innovative approaches to improving the conditions of degraded streams. For example, require-

ments that stream restoration be “in kind” currently prevent catchment managers from spending restoration dollars for out-of-channel improvements in water management that might ultimately better protect water and habitat quality. If regulatory agencies insisted upon evidence of desirable outcomes (e.g., improved water quality, restored environmental flows or dampened floods, improvements in the diversity or abundance of target taxa) rather than requiring particular approaches or evidence of structural changes to the channel, this would provide opportunities for experimentation and the development of a broader array of tools and approaches for reversing and preventing degradation of river ecosystems.

We are optimistic that the recent spate of publications on restoration outcomes is making a difference. Both of us are aware of new efforts in the United States, Europe, and Australia to prioritize and plan restoration efforts at catchment scales, and we believe this reflects not only new scientific findings, but perhaps more importantly an increase in social pressure and environmental awareness. Streams and rivers reflect what we do on the land, and rather than working at the end of the pipe (the stream), community groups and managers may need to be focusing their energies on protecting streams and rivers from continued degradation by keeping storm waters, nutrients, and contaminants contained within catchments. Recognizing the extreme fragility of healthy streams and the nearly insurmountable challenges to stream restoration in the catchments where most people live and work must give us pause—and force us to raise the bar when deciding whether to fill, pipe, or otherwise further degrade stream ecosystems. Finally, we have to recognize that, in many catchments, river restoration (i.e., channel projects) will never be “good enough.” In highly altered agricultural, urban, and suburban landscapes, elevated supplies of storm waters, nutrients, and contaminants to stream ecosystems impose serious constraints on what can be achieved both structurally and functionally through stream engineering. Instead, effective “restoration” in catchments with agricultural or urban development must first focus on slowing and interrupting the heightened hydrologic connectivity between heavily altered catchments and draining streams, thereby improving water quality and reducing peak flows. Protecting and promoting healthy riparian vegetation and reinstating the reduced hydrologic connectivity between streams and their floodplains are important and necessary next steps. Such measures may be sufficient to promote biological recovery in impacted streams; however, structural enhancements within the stream channel will likely be necessary to create and reinstate appropriate habitat diversity.

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Effects of urbanization and urban stream restoration on the physical and biological structure of stream ecosystems

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Abstract. Streams, as low-lying points in the landscape, are strongly influenced by the stormwaters, pollutants, and warming that characterize catchment urbanization. River restoration projects are an increasingly popular method for mitigating urban insults. Despite the growing frequency and high expense of urban stream restoration projects, very few projects have been evaluated to determine whether they can successfully enhance habitat structure or support the stream biota characteristic of reference sites. We compared the physical and biological structure of four urban degraded, four urban restored, and four forested streams in the Piedmont region of North Carolina to quantify the ability of reach-scale stream restoration to restore physical and biological structure to urban streams and to examine the assumption that providing habitat is sufficient for biological recovery. To be successful at mitigating urban impacts, the habitat structure and biological communities found in restored streams should be more similar to forested reference sites than to their urban degraded counterparts. For every measured reach- and patch-scale attribute, we found that restored streams were indistinguishable from their degraded urban stream counterparts. Forested streams were shallower, had greater habitat complexity and median sediment size, and contained less-tolerant communities with higher sensitive taxa richness than streams in either urban category. Because heavy machinery is used to regrade and reconfigure restored channels, restored streams had less canopy cover than either forested or urban streams. Channel habitat complexity and watershed impervious surface cover (ISC) were the best predictors of sensitive taxa richness and biotic index at the reach and catchment scale, respectively. Macroinvertebrate communities in restored channels were compositionally similar to the communities in urban degraded channels, and both were dissimilar to communities in forested streams. The macroinvertebrate communities of both restored and urban degraded streams were correlated with environmental variables characteristic of degraded urban systems. Our study suggests that reach-scale restoration is not successfully mitigating for the factors causing physical and biological degradation.

Key words: *benthic macroinvertebrate; biotic recovery; habitat restoration; species composition; stream restoration; urbanization.*

INTRODUCTION

The world's human population is primarily urban, and future population growth will occur predominantly in urban centers (United Nations 2008). Thus, an increasing proportion of our freshwater ecosystems will become impacted by urbanization, and a larger fraction of humanity will rely on waterways degraded by a

common set of urban impacts. The physical, biogeochemical, and biological stream impairments that occur specifically in urbanized watersheds have been labeled the “urban stream syndrome” (Walsh et al. 2005b). Physical and hydrological consequences of watershed urbanization are well documented and include altered base flow and unstable hydrology with frequent, short-duration, high-peak floods (Booth and Jackson 1997, Paul and Meyer 2001, Meyer et al. 2005, Walsh et al. 2005b). These changes typically lead to channel incision and simplification (Shields et al. 2003, Niezgoda and Johnson 2005, Sudduth and Meyer 2006), and homogenization of benthic habitats (Federal Interagency Stream Restoration Working Group 1998, Malmqvist and Rundle 2002, Walsh et al. 2005b).

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Coincident with hydrological and geomorphological modification, urban streams have elevated nutrient and contaminant concentrations. Hyperconnectivity with the surrounding landscape through roads, storm drains, and leaky and overflowing sanitary sewers efficiently routes watershed contaminants into urban channels (Bernhardt et al. 2008). Pollutant concentrations increase not only due to increased inputs from point and non-point sources but also as a result of decreased nutrient removal efficiency in hydrologically disconnected riparian zones and streambeds (Groffman and Crawford 2003, Grimm et al. 2005, Meyer et al. 2005).

The inverse relationship between urbanization and native biodiversity and species composition is well established and persists across a range of taxa (Blair 1996, Germaine and Wakeling 2001, Clark et al. 2007, Grimm et al. 2008, McKinney 2008). Macroinvertebrate communities are strongly affected by land use patterns (Lenat and Crawford 1994, Sponseller et al. 2001, Allan 2004). Watershed impervious surface cover is generally associated with a decrease in invertebrate species richness and increasing dominance of highly tolerant taxa (Morse et al. 2003, Roy et al. 2003, Moore and Palmer 2005, Collier et al. 2009, Cuffney et al. 2010). Development that is within riparian areas or that is directly hydrologically connected to stream channels (e.g., road crossings and pipes) can be particularly detrimental to stream communities (Wang and Kanehl 2003, Moore and Palmer 2005, Walsh and Kunapo 2009), and there is thus great interest in riparian reforestation and management for urban stream ecosystem protection (Bernhardt and Palmer 2007). While the impacts of watershed urbanization on stream biota are well documented, it is far from clear what combination of reach and watershed scale management is necessary and sufficient to promote community recovery in urban streams.

In the face of channel incision and bank erosion, water quality degradation, and habitat and biodiversity loss, degraded urban waterways are often targeted for restoration. Stream restoration or rehabilitation encompasses a variety of strategies by which human impacts are mitigated and previous damage is addressed, with the overarching goal of returning the stream to as close to pre-impacted conditions as possible (National Research Council 1992). Urban stream restoration presents unique problems: there is minimal space for rehabilitation, and land acquisition is both expensive and complicated because it generally involves multiple landowners. These challenges typically lead to fewer linear feet being restored and higher per-project costs compared to rural and agricultural stream restoration projects (Bernhardt and Palmer 2007). In fact, for many regions of the United States, the majority of restoration dollars are invested in a small number of urban stream projects (Hassett et al. 2005, Sudduth et al. 2007). Stream restoration projects are customarily implemented with the specific goals of water quality improvement and provision of aquatic habitat (Bernhardt et al.

2007, Sudduth et al. 2007), yet few projects have been adequately evaluated to determine whether these goals are met (Charbonneau and Resh 1992, Palmer et al. 1997, 2005, Moerke et al. 2004, Moerke and Lamberti 2004, Bernhardt et al. 2005). Given the frequency with which urban stream restoration is employed to mitigate habitat and water quality degradation and the expenses and challenges involved, it is worth understanding whether these efforts are measurably improving habitat and community structure.

The underlying assumption of stream restoration is that altering channel geomorphology to resemble pre-degradation conditions will lead to the recovery of native aquatic organisms. This assumption is based on prior work demonstrating that fish or macroinvertebrate taxonomic richness and spatial heterogeneity are positively correlated (Gorman and Karr 1978, Angermeier and Winston 1998, Vinson and Hawkins 1998, Brown 2003). Although experimental manipulations have demonstrated that high substrate variability does not per se lead to higher species richness or faster recovery (Brooks et al. 2002, Spanhoff et al. 2006), stream restoration design employs habitat provision, or increased habitat heterogeneity as the primary mechanism for restoring biotic communities (Brooks et al. 2002). Evidence to support the assumption that successfully restoring physical structure is sufficient for community restoration (the “field of dreams” hypothesis) is lacking (Palmer et al. 1997, 2010, Moerke et al. 2004).

We set out to evaluate the effectiveness of four natural channel design (NCD) projects, a common urban stream restoration approach (*sensu* Rosgen 1994, 1996) in mitigating urban stream degradation. NCD reconfigures the pattern, profile, and dimensions of a degraded channel to emulate an unimpacted ideal (Rosgen 2007). This method utilizes heavy machinery to regrade and reshape a degraded channel and employs hard structures such as log vanes or cross vanes to control grade, installs root wads to stabilize banks, adds coarse bed material to create riffles, and revegetates reconfigured or newly created riparian areas.

Effective restoration should recapture the habitat structure and biological communities of forested streams, ideally approaching a stable approximation of “reference” conditions. We tested whether a series of urban restoration projects were achieving or moving toward this goal by examining whether habitat structure and macroinvertebrate community composition in the restored reaches of urban streams were different from similarly situated urban degraded stream reaches and whether the habitat and community structure of these restored reaches more closely matched conditions in nearby forested streams than their unrestored urban counterparts.

METHODS

Site selection

Through consultation with staff of the North Carolina Ecosystem Enhancement Program (EEP) and

the North Carolina Stream Restoration Institute (SRI) we selected four urban natural channel design restoration projects that practitioners and regulators felt were particularly well-designed and implemented. Our goal in selecting restoration projects was not to select a random sample, but rather to choose a set of projects that represented the best-case scenario for urban restoration based on expert practitioners' opinions. Each restored stream reach was then matched with a similarly situated unrestored urban stream and a forested stream in the Raleigh-Durham area in the Piedmont region of North Carolina. The full comparison thus included 12 study sites: four forested (F) sites, within small streams draining forested catchments; four urban restored (R) sites, within recently implemented natural channel design restoration projects; and four urban degraded (U) sites located in urban parks where future restoration activities are likely (Fig. 1; Appendix A). This suite of sites was selected to determine the potential for ecological restoration to restore the physical and biological structure and ecosystem function of stream ecosystems.

Site descriptions

Four sampling blocks were created from the group of 12, each containing one urban degraded, one urban restored, and one forested stream of similar catchment sizes and underlying geology (Table 1). The study area spans the Northern Outer Piedmont, Slate Belt, and Triassic basin ecoregions, and many sites drain multiple ecoregions (Table 1). Soil characteristics affect baseflows and consequently stream size and permanence. Triassic Basin and Slate Belt streams have low summer baseflows due to low clay permeability and low water yield from slate substrate (Griffith et al. 2002). Reduced summer baseflows are not seen in Northern Outer Piedmont streams where streams tend to be larger and less prone to drying. For physical and functional metrics, all streams within a sampling block were sampled within one week with no intervening major storm events. In this way, the blocking factor accounts for both differences in watershed size, and staged timing of field analyses.

Our study included four restored stream reaches, each of which was restored using NCD between 1999 and 2005. The Abbott stream restoration project was implemented in 1999 on a tributary to Walnut Creek, in Raleigh, North Carolina. The goal of this restoration project was "to restore the stream to the stable dimension, pattern, and profile for a C4 stream type as classified using Rosgen's stream classification methodology (Rosgen 1996). ... This type of restoration will reestablish the channel on a previous floodplain, or in this case, the basin of an old pond. Appropriate channel dimensions (width and depth), pattern (sinuosity, belt width, riffle-pool spacing), and profile (bed slope) of the new channel will be determined from reference reaches" (North Carolina Department of Transportation 1999). Rocky Branch is a stream located on the urban North

Carolina State University campus in Raleigh and was restored in 2001. The goals of this restoration project included, "Restore a stable self-maintaining morphological pattern in the stream channel; Stabilize stream banks using vegetation; Create and improve habitat for fish and aquatic invertebrates; Improve the quality of stormwater entering the creek through restoring and enhancing riparian buffers and establishing stormwater control within the creek's watershed; Provide safe and enjoyable access to the stream and passage through the campus by completing the greenway path adjacent to the creek." (Doll 2003). Restored in 2004, Sandy Creek flows through the urban Duke University campus in Durham. The Sandy Creek project goals were to, "Re-contour and restore more than 600 meters of degraded stream to hydrologically reconnect the stream with the adjacent floodplain to improve biogeochemical transformations and stream water quality" (Richardson and Pahl 2005). Third Fork Creek is a stream flowing through an urban park near downtown Durham and was restored in 2005. The goals of this project were to, "Restore stable channel morphology that is capable of moving the flows and sediment provided by its watershed; reduce sediment-related poor water quality impacts resulting from lateral bank erosion and bed degradation; improve aquatic habitat diversity through the reestablishment of riffle-pool bed variability and the use of in-stream structures; restore vegetative riparian buffers utilizing native plant species; and improve natural aesthetics in an urban park setting." (KCI Associates 2003).

All of our urban stream reaches were located in urban parks or protected areas to facilitate access, and are similar to the pre-restoration conditions of our restored study sites. Two of our urban stream sites (reaches of Goose Creek and Ellerbe Creek) were chosen because the North Carolina Ecosystem Enhancement Program listed them as priority stream restoration sites (both were restored after this research effort). Our study reach on Upper Mud Creek is located within the protected Duke Forest, immediately downstream of a 1980s-era subdivision. Cemetery Creek is located on city property in Raleigh and drains an older, high-density, urban neighborhood.

Forested sites were selected from "reference" sites previously used for stream restoration projects as well as sites within Duke Forest. Lower Mud Creek and the Tributary to Mud Creek are located in Duke Forest, in Durham. Stony Creek is located in Duke Forest near Hillsborough. Pot's Branch is located in Umstead State Park near Raleigh. Because of the land use history of the North Carolina Piedmont, these are not pristine reference sites, but rather post-agriculture reforested streams with primarily forested watersheds; thus there may be legacy effects of prior agricultural land use on geomorphology, vegetation, sediment, and biota (Maloney et al. 2008). There are no primary growth forests of sufficient size to have a permanent stream,

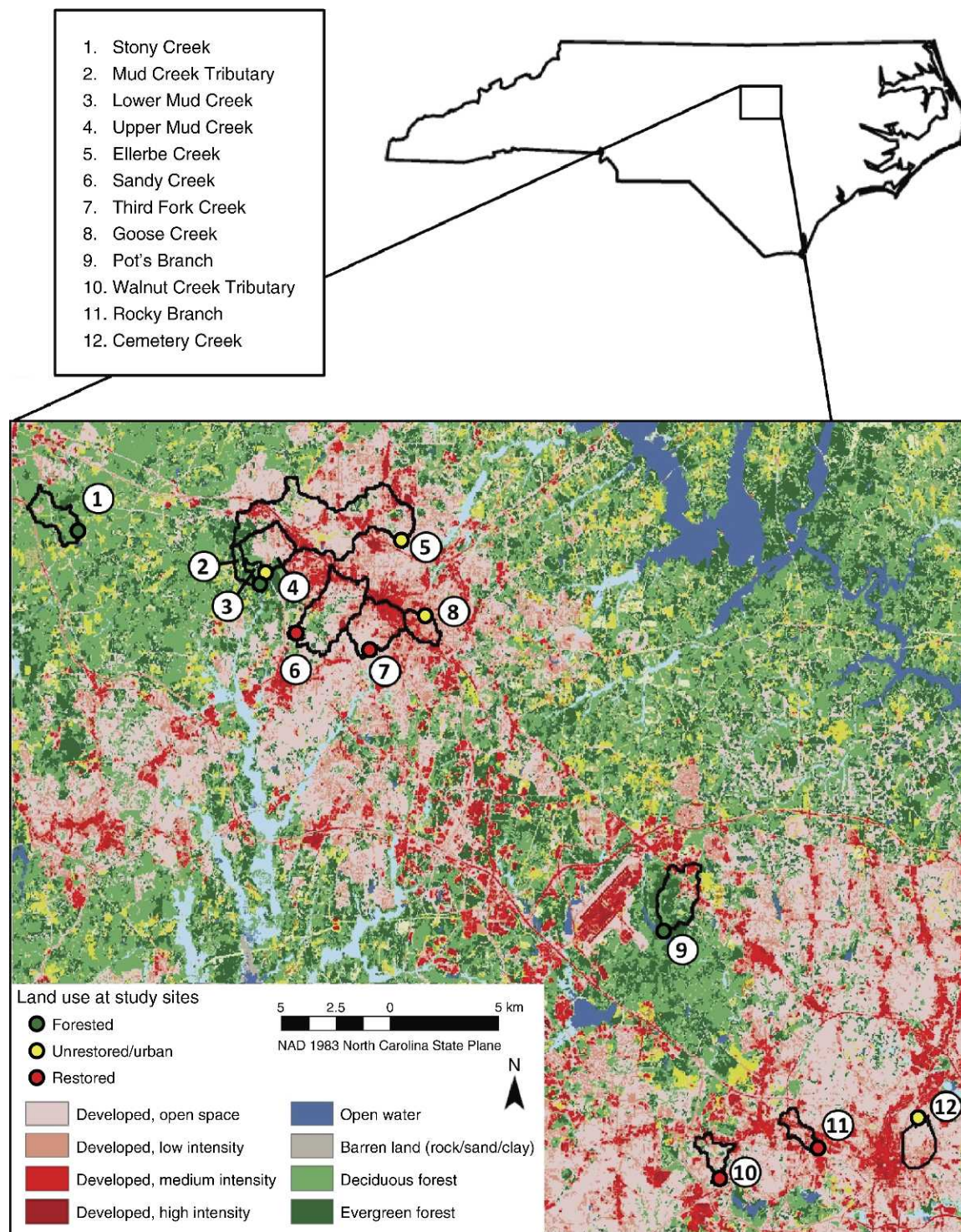


FIG. 1. Study site locations and watershed boundaries in the Raleigh–Durham area in the Piedmont region of North Carolina.

rather our “forested” streams are secondary growth and represent the post-agricultural, pre-urban landscape. Due to the lack of undisturbed Piedmont streams, we included one forested stream reach, Lower Mud Creek that had urban development more than 1.5 km upstream

of the study reach, and for which the entire watershed within that 1.5 km was ~100-year-old mixed deciduous forest. Because Lower Mud Creek is far from an ideal reference stream, we performed all statistical comparisons both with and without this stream.

TABLE 1. Study sites listed by block, stream type, EPA level-IV ecoregion, channel, and catchment characteristics.

Status	Site name	Ecoregion	Reach length (m)	Estimated discharge (L/s)	Watershed size (km ²)	Developed (%)	% ISC
Block 1							
Forested	Stony Creek	45c	100	0.66	6.9	24.4	3.4
Urban restored	Third Fork Creek	45g	80	4.41	4.4	99.5	32.4
Urban degraded	Ellerbe Creek	45c/g	50	10.41	7.6	88.7	20.8
Block 2							
Forested	Pot's Branch	45f	140	5.83	4.2	27.4	9.9
Urban restored	Walnut Creek Tributary	45f	200	5.47	1.7	84.5	17.8
Urban degraded	Cemetery Creek	45f	100	11.54	2.2	98	19.1
Block 3							
Forested	Mud Creek Tributary	45c/g	54	2.08	0.9	4.4	0.5
Urban restored	Rocky Branch	45f	50	1.54	1.5	99.2	34.8
Urban degraded	Goose Creek	45g	35	3.72	1.7	100	39.4
Block 4							
Forested	Lower Mud Creek	45c/g	102.5	11.58	4.1	58.6	9.5
Urban restored	Sandy Creek	45g	60	12.00	6.7	76.9	16.8
Urban degraded	Upper Mud Creek	45c/g	140	4.86	3.5	66.9	11

Notes: Ecoregions are described in Griffith et al. (2002); %ISC is the percentage of the watershed with impervious surface cover.

Land use characterization

We acquired the 1/3 arc-second (10 m) digital elevation model for Durham, Orange, and Wake counties in North Carolina from the USGS Seamless Server and performed analysis using the ArcHydro extension of ArcGIS (ESRI, Redlands, California, USA) to calculate flow direction and flow accumulation, and define streams based on a 1000-pixel threshold and delineate watersheds for all sites. Land use and impervious surface cover within study watersheds were analyzed based on 2001 National Land Cover Dataset (NLCD), and the associated Impervious Surface Cover data set from the USGS Seamless Server (Homer et al. 2004). We classified riparian land use in a 30-m buffer around each stream segment using the same technique. NLCD was reclassified into four categories: developed, agriculture, undeveloped, and water and for each watershed we calculated the percentage of each land use type and percentage of impervious surface cover. The percentage of catchment developed and catchment impervious cover (ISC) were used as predictor variables in subsequent analyses.

Habitat surveys

In each stream, we delineated experimental reaches encompassing at least one hour of travel time under June 2006 base-flow conditions. We selected the upstream end of each reach by locating an area of constricted flow with the greatest downstream extent of channel uninterrupted by tributary inputs or road crossings. Reach travel time was determined by calculating water travel times using a rhodamine dye release. We used rhodamine tracers because traditional salt tracers proved problematic in several of our urban streams due to high spatial and temporal variation in stream water chloride concentrations. Our study reaches

were standardized by water residence time and varied in length from 35 to 200 m. We delineated our study reaches in this manner in order to correctly measure ecosystem function variables (see Sudduth et al. 2011). Habitat surveys were performed in July and August of 2006. We created habitat maps (see Appendix B for examples) of all experimental reaches by determining the longitudinal boundaries and channel widths of riffle, run, pool, and debris dam habitats within each reach (Vermont Water Quality Division 2009). We used a stadium rod and level to survey longitudinal slope for the entire reach and to generate cross-sectional profiles for five randomly selected points within the reach. Reach canopy cover was measured at each cross section using a spherical densiometer. We conducted pebble count surveys of 100 randomly selected sediment particles spaced evenly throughout the study reach (Wolman 1954) to estimate variation in sediment grain size within each stream reach.

Hydrologic data

We created fine-scale flow-habitat maps by measuring velocity and depth values at five evenly spaced points across the active channel, with a sixth measurement in the thalweg, at 30 cross-section locations evenly spaced longitudinally in each reach. In October 2006, we deployed Solinst levelloggers (Solinst Canada, Georgetown, Ontario, Canada) in each stream reach to collect continuous measurements of water level. We used HEC-RAS (US Army Corps of Engineers) to estimate discharge water level and surveyed channel dimensions (software *available online*).⁷ We used these data to create a flashiness index (Baker et al. 2004) for use as a predictive variable in macroinvertebrate community analyses.

⁷ www.hec.usace.army.mil/software/hec-ras/

Functional measures

Nutrient and organic matter dynamics were measured concurrently in the same study reaches (for methods and results, see Sudduth et al. 2011). Functional measures were used as potential predictor variables in ordination analyses.

Macroinvertebrate sampling

Macroinvertebrate sampling was conducted at the 12 study sites between May and September 2006 ("summer" sample) and February and March 2007 ("winter" sample) from the same reach as physical and functional measurements were taken. Macroinvertebrate communities were sampled once each season using the North Carolina Department of Water Quality Qual 4 semi-quantitative protocol (North Carolina Division of Water Quality 2006 [hereafter NC DWQ]). This sampling protocol is designed to assess macroinvertebrate diversity in small streams (drainage area <7.7 km²) and is conducted so that sampling effort is consistent among study sites. Each sample consisted of one 2–3 minute, 1-m², 1-mm mesh, kick-net sample from a characteristic riffle; one 500- μ m mesh triangular sweep net of stream marginal habitats such as root mats and bank vegetation; an approximately 500-g leaf pack sample collected from rock or snag habitats; and visual assessments of habitats not easily sampled with the above methods (e.g., large rocks or logs). Samples were field sorted and specimens were preserved in 95% ethanol. Non-chironomid taxa were identified at 45 \times magnification to the lowest possible taxonomic level, typically species (Pennak 1953, Brigham et al. 1982, Merritt et al. 2008). Chironomidae were mounted on slides in CMC-10 medium (Master's Chemical Company, Wood Dale, Illinois, USA), and identified at 400 \times magnification to genus or species (Epler 2001). Following the NC DWQ protocol, we classified taxa as abundant (>10 individuals), common (three to nine individuals), or rare (one or two individuals). One of the winter urban degraded samples, Goose Creek, was lost, however field notes conclusively indicate the absence of Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa in this sample. Thus, this sample was included in EPT richness analyses.

Data analyses

Physical data analyses.—Habitat complexity was determined by counting the number of transitions between different aquatic habitats (riffle, run, pool, and debris dam classifications) for each experimental reach. The transition counts were normalized for all reaches by converting the counts to number of transitions per 100 m reach length. The average number of transitions and standard error was determined for each stream type (F, R, and U). Velocity and point depth measurement averages for each reach were calculated and used to obtain an average and coefficient of variation for each stream type.

The ratio of active channel width to the active channel depth at the thalweg was determined from the field survey cross-section data for each experimental reach. Also, the maximum (smallest width:depth ratio value for each stream) and minimum (largest width:depth ratio value for each stream) incision value from the field survey data were calculated. The average and coefficient of variation of percent canopy cover were determined from spherical densiometer measurements. Physical metrics were compared among stream types using one-way ANOVA with stream type as a single factor (GraphPad Prism v4; GraphPad Software Inc., La Jolla, California, USA). Where the overall effect was significant, we performed post hoc pairwise comparisons (Student-Newman-Keuls) to test for differences among stream types and calculated the magnitude of effect as ω^2 , the variance component of the factor in the ANOVA relative to the total variance (Graham and Edwards 2001).

Macroinvertebrate data analyses.—In addition to total species richness, we calculated richness of the orders Ephemeroptera, Plecoptera, and Trichoptera (EPT) as a measure of pollution-sensitive taxa richness. We also calculated the biotic index (BI) for each site as a measure of overall macroinvertebrate community pollution tolerance. BI was calculated as a weighted mean of taxa tolerance values relative to their abundance, and higher BI values indicate a more pollution-tolerant assemblage (NC DWQ 2006). Individual taxon tolerance values were taken from the NC DWQ benthos standard operating protocol (Lenat 1993, NC DWQ 2006). Taxa for which BI information was not available represented a small minority of taxa and were excluded from the BI calculations. We compared community metrics among stream types using one-way analysis of variance (ANOVA) with stream type as a factor. Where the overall affect of stream type was significant, post-hoc pairwise comparisons and effect size calculations were performed as for physical metrics.

We used least-squares linear regression to quantify correlative relationships between macroinvertebrate metrics and environmental physical and functional variables. As sites were grouped into sampling blocks a priori according to watershed and geological variables, all analyses should include sampling block as a variable. However, as block was not found to be an important predictor of any of our habitat or macroinvertebrate community metrics (data not shown), it was not included in our analyses in order to maximize our power to detect differences among site types.

We examined seasonal macroinvertebrate species compositional similarity among sites using nonmetric multidimensional scaling (NMS) ordination of sites in species space, using Bray-Curtis similarities of square root transformed abundance values (PC-ORD v. 5; McCune and Mefford 2006). Solutions were obtained from 500 runs (250 randomized, 250 with real data) using random starting coordinates. We created joint plots incorporating a second matrix of physical and

TABLE 2. Mean values (\pm SE) of habitat complexity, flow heterogeneity, floodplain connectivity, and canopy cover of forested, urban restored, and urban degraded stream types.

Parameter	Forested	Urban restored	Urban degraded	df	<i>F</i>	ω^2
Number of habitat transitions per 100-m reach	20.75 \pm 1.89	9.250 \pm 2.14	9.75 \pm 1.11	2, 9	13.55*	41.1
Depth from point measurements (m)	0.065 \pm 0.0164	0.175 \pm 0.0131	0.158 \pm 0.012	2, 9	17.97**	48.5
%CV for depth point measurements	109.3 \pm 12.21	73.73 \pm 7.59	83.03 \pm 9.30	2, 9	2.29	
Velocity from point measurements (m/s)	0.035 \pm 0.008	0.023 \pm 0.007	0.026 \pm 0.012	2, 9	0.47	
%CV for velocity point measurements	209.2 \pm 46.43	139.4 \pm 5.38	237.0 \pm 46.43	2, 9	2.29	
Degree of incision	6.15 \pm 0.37	7.14 \pm 1.39	4.96 \pm 0.77	2, 8	1.64	
Maximum degree of incision	4.74 \pm 0.31	5.02 \pm 1.06	4.40 \pm 0.57	2, 8	0.23	
Longitudinal slope (%)	0.93 \pm 0.49	0.51 \pm 0.49	0.29 \pm 0.11	2, 8	0.78	
Canopy cover (%)	87.54 \pm 2.50	53.71 \pm 8.28	81.35 \pm 4.36	2, 9	10.37*	34.2
Median substrate size (mm)	35.75 \pm 11.35	8.0 \pm 6.35	4.75 \pm 3.75	2, 9	4.75*	17.3

Notes: Degree of incision is the width-to-depth ratio; the maximum degree of incision is the smallest width-to depth ratio. Results and effect sizes are from one-way ANOVAs with stream type as a factor. Where the overall effect was significant, we performed post hoc pairwise comparisons (Student-Newman-Keuls) to test for differences among stream types and calculated the magnitude of effect as ω^2 , the variance component of the factor in the ANOVA relative to the total variance.

* $P < 0.05$; ** $P < 0.01$.

functional variables. We set a minimum r^2 of 0.30 to identify geomorphological and functional parameters correlated with macroinvertebrate community structure at different sites.

We assessed the importance of time since restoration to macroinvertebrate recovery by evaluating separately collected macroinvertebrate monitoring data from Rocky Branch both within the restoration and at an unmanipulated upstream reference, and from Sal's Branch, a forested reference site in Umstead Park, North Carolina. Monitoring data were collected using the same NC DWQ Qual-4 protocol as for this study. Pre-restoration samples were collected for Rocky Branch in December 2000 and post-restoration data were collected in December 2003, November 2004, December 2005, and December 2006. Reference data were collected from Sal's Branch in March 2002, March 2003, and May 2004. We evaluated the importance of time since restoration to total species richness, EPT richness, and community BI for Rocky Branch. We calculated the change in each community metric by subtracting the pre-restoration value from the post-restoration value for each monitoring year ($\Delta_{\text{Rest}} = \text{Metric}_{\text{post[yr]}} - \text{Metric}_{\text{pre}}$). We accounted for community structure changes due to factors other than restoration by performing the same calculation for the upstream reference ($\Delta_{\text{Up}} = \text{Metric}_{\text{up[yr]}} - \text{Metric}_{\text{up pre}}$) and then calculated the effect of restoration by taking the difference of the two (Restoration Response = $\Delta_{\text{Rest}} - \Delta_{\text{Up}}$). This is similar to the "raw effect score" for taxon abundance calculations from impact assessment studies (Weiss and Reice 2005), but applied to community-level metrics. We evaluated species compositional similarity among these samples using the same NMS ordination protocol as above.

RESULTS

To test our overarching hypothesis that positive restoration outcomes would lead urban restored streams to become more similar to minimally impacted sites, we compared physical and biological attributes among the

three stream types. Excluding Lower Mud Creek did not change the conclusions of any of our relationships of physical metrics among stream types and there was no consistent pattern in the effect of removing this site. However, in every case, removing Lower Mud Creek from biological analyses increased the strength of the observed relationship (Fig. 3), and for some analyses, resulted in a stronger overall effect of stream type (Tables 3–4). For all analyses, we show comparisons with and without Lower Mud Creek included as a forested site.

Habitat

Urban streams had significantly deeper channels, smaller substrate sizes, and less reach-scale habitat variation (transitions between riffles, runs, and pools) than their forested counterpart (Table 2, Fig. 2). For each of these metrics, urban restored streams were indistinguishable from their urban degraded counterparts and significantly different from the forested streams. We found a significant difference between urban degraded and urban restored reaches in only a single habitat metric; restored urban streams had significantly lower riparian canopy cover than their unrestored counterparts.

Our hydrologic metrics did not differ between stream types. Stream velocities and flow heterogeneity were highly variable within stream types. There was no difference in either average or maximum degree of incision among stream types (Table 2).

Biological structure

Macroinvertebrate community richness was similar across stream types in summer, while in our winter sampling our three forested sites (excluding LMC) had significantly higher taxa richness than their restored or urban counterparts (Table 3, Fig. 3). In both seasons, species of Chironomidae made up 56.6% ($\pm 4.5\%$ [mean \pm SE]) and 44.9% ($\pm 2.6\%$) of the taxa found in urban and restored streams respectively, and only 26.7% ($\pm 5.0\%$) of the taxa in the forested streams (Appendix C).

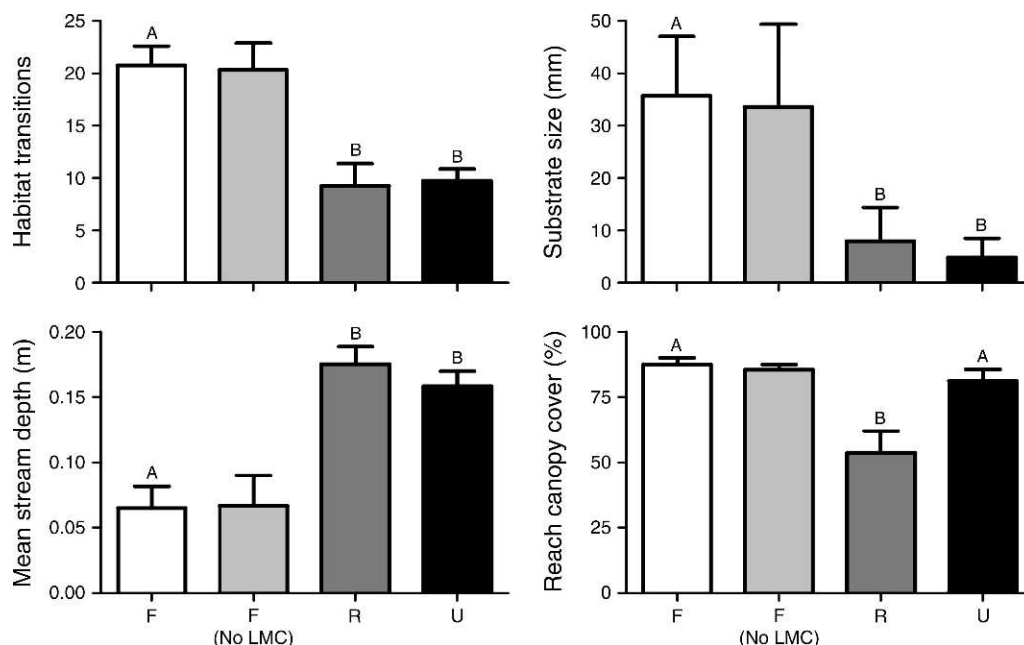


FIG. 2. Mean values (and SE) of habitat transitions (number of transitions per 100 m reach length), substrate size (mm), mean stream depth (m), and reach canopy cover (% covered) for forested sites (F), forested sites excluding Lower Mud Creek (LMC), urban restored (R), and urban degraded (U) sites (one-way ANOVA, $P < 0.05$). Differences among stream types are indicated by different letters (Student-Newman-Keuls post hoc multiple comparisons test).

The three forested sites had higher mean EPT richness than urban and restored sites in both summer and winter (Table 3, Fig. 3). Summer and winter biotic integrity scores were lower (higher number of sensitive taxa) in these three forested streams than in the urban restored and urban degraded streams (Table 3, Fig. 3).

Among quantified watershed variables, watershed imperviousness was found to be the best single predictor of EPT richness, although the trend was significant only in winter ($r^2 = 0.54$, $P < 0.01$). Biotic index was positively correlated with watershed imperviousness in both summer ($r^2 = 0.50$, $P < 0.01$) and winter ($r^2 = 0.40$, $P < 0.05$) (Table 4, Fig. 4).

Among the many in-channel structural and functional variables measured, number of habitat transitions per 100 m was the only reliable predictor of EPT richness, with habitat complexity positively correlated with the number of EPT in both summer ($r^2 = 0.54$, $P < 0.01$)

and winter ($r^2 = 0.46$, $P < 0.05$). Habitat complexity was strongly negatively correlated with BI scores in both summer ($r^2 = 0.70$, $P < 0.001$) and winter ($r^2 = 0.67$, $P < 0.01$). Removing Lower Mud Creek from the analyses increased the strength of the observed relationships, but had a stronger affect on EPT richness than BI (Table 4, Fig. 4).

NMS ordination results revealed large differences in community composition between stream types. Two-dimensional NMS solutions were best for both summer and winter. The summer NMS ordination had a final stress of 0.13 and explained 78.2% of compositional similarity, 40.2% along axis 1 and 37.8% along axis 2 (Fig. 5a). The winter NMS had a final stress of 0.078 and explained 88.2% of compositional similarity among sites, 58.6% along axis 1 and 29.7% along axis 2 (Fig. 5b). With the exception of Lower Mud Creek, forested sites clustered closer to one another in winter than in

TABLE 3. Mean values (\pm SE) of macroinvertebrate community metrics with and without Lower Mud Creek (LMC).

Parameter	Forested	Forested (no LMC)	Urban restored	Urban degraded	df	F	df (no LMC)	F (no LMC)	ω^2 (no LMC)
Summer species richness	20.0 \pm 2.3	22.3 \pm 0.3	15.3 \pm 3.9	15.0 \pm 2.6	2, 9	0.86	2, 8	1.70	
Summer EPT richness	5.8 \pm 2.0	7.3 \pm 1.8	1.8 \pm 0.6	1.5 \pm 0.9	2, 9	3.28	2, 8	8.79*	32.1
Summer BI	5.8 \pm 0.8	5.4 \pm 1.0	7.7 \pm 0.5	8.0 \pm 0.4	2, 9	4.35*	2, 8	5.19*	20.2
Winter species richness	29.3 \pm 3.4	32.3 \pm 1.9	15.5 \pm 4.7	14.7 \pm 3.8	2, 8	4.19	2, 7	5.88*	24.5
Winter EPT richness	10.3 \pm 2.4	12.3 \pm 1.7	1.5 \pm 0.5	1.3 \pm 0.9	2, 9	11.46**	2, 8	34.47***	67.0
Winter BI	5.6 \pm 0.3	5.4 \pm 0.3	7.0 \pm 0.4	7.3 \pm 0.5	2, 8	5.55*	2, 7	6.04*	25.1

Notes: Abbreviations are: EPT, Ephemeroptera, Plecoptera, and Trichoptera; BI, biotic index. Results and effect sizes are from one-way ANOVAs performed with and without LMC, with metric as a factor.

* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$.

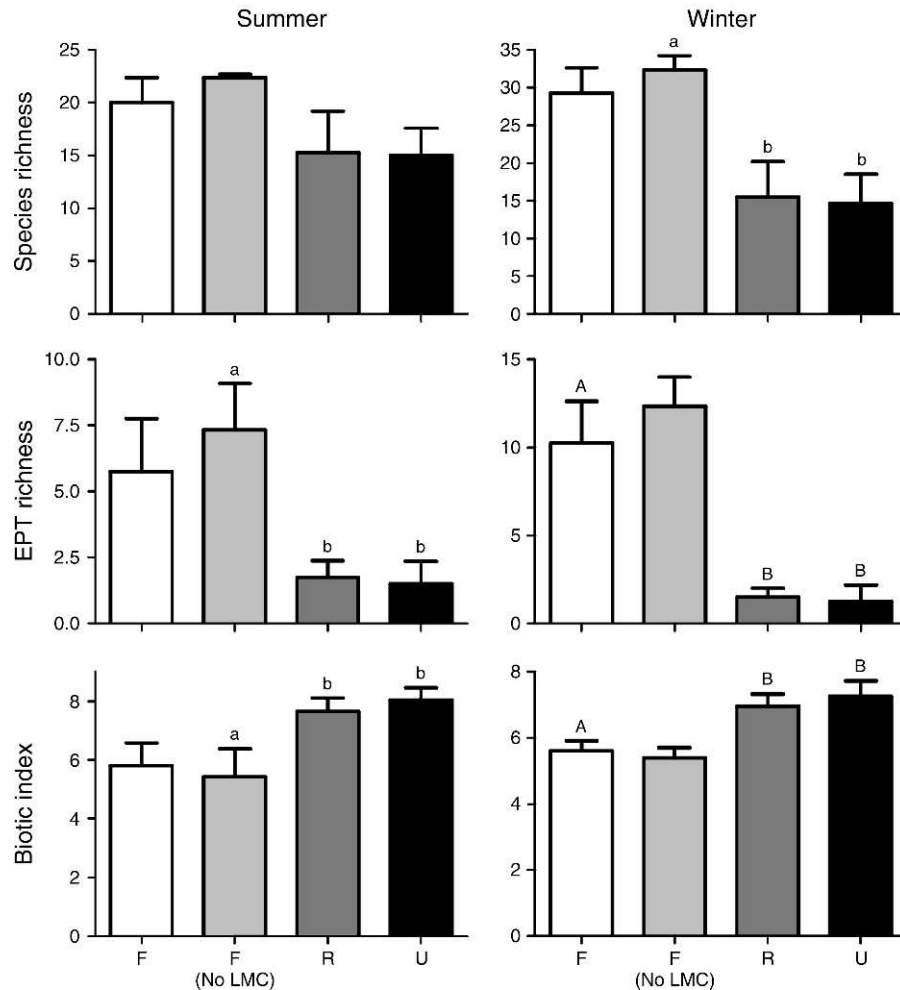


FIG. 3. Mean values (and SE) of summer and winter species richness (number of species); Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness (number of species); and biotic index for forested sites, forested sites excluding LMC, urban restored, and urban degraded sites (one-way ANOVA, $P < 0.05$). Differences among stream types is indicated by different letters (Student-Newman-Keuls post hoc multiple comparisons test). Different uppercase letters indicate differences among stream types for the entire data set; different lowercase letters indicate differences among stream types excluding LMC.

summer (Fig. 5). In spite of their close proximity (<50 m of forest between the reaches, Fig. 6), Lower Mud Creek and Mud Creek Tributary did not cluster together in either season.

In analyses of long-term data from Rocky Branch, we found no significant effect of time since restoration on total species richness, EPT richness, or BI. A three-dimensional NMS solution was best for explaining compositional similarity (final stress = 0.09, cumulative $r^2 = 0.866$) among Rocky Branch macroinvertebrate communities collected as part of this study, upstream and restored samples collected for restoration monitoring, and reference data from Sal's branch and the winter forested samples from this study. Regardless of year, restoration monitoring samples from Rocky Branch clustered more closely to restored samples collected as

part of this study and impacted upstream reference samples than to forested samples (Fig. 7).

DISCUSSION

We hypothesized that, if restoration is effective at improving degraded urban stream ecosystems, both the geomorphology and biota at restored sites would more closely resemble forested sites than would their urban counterparts. While it would be overly optimistic to expect restored stream reaches to become identical to reference sites, successful restoration ought to lead to stream habitat and biological communities that are distinguishable from unrestored urban streams. In this survey, urban restored streams differed significantly from their unrestored urban counterparts in only a single metric: having reduced canopy cover as a direct result of project implementation (Fig. 2). These results

TABLE 4. Results from least-squares linear regression analyses of macroinvertebrate community metrics with and without LMC.

Metric	With LMC			No LMC		
	r^2	df	F	r^2	df	F
Watershed imperviousness						
Summer EPT	0.26	1, 10	3.44	0.33	1, 9	4.43
Summer BI	0.50	1, 10	10.18**	0.51	1, 9	9.55*
Winter EPT	0.54	1, 10	11.85**	0.58	1, 9	12.23**
Winter BI	0.40	1, 9	6.03*	0.40	1, 8	5.24
Habitat transitions						
Summer EPT	0.54	1, 10	11.72**	0.86	1, 9	56.62***
Summer BI	0.70	1, 10	22.91***	0.81	1, 9	39.36***
Winter EPT	0.46	1, 10	8.61*	0.58	1, 9	12.60**
Winter BI	0.67	1, 9	18.01**	0.74	1, 8	22.41**

Note: Watershed imperviousness is the percentage of the watershed with impervious surface cover.

* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$.

suggest that despite expenditures of >US\$1 million per project, these restored streams did not have improved habitat complexity or detectable changes in their macroinvertebrate communities. The deep, sandy, simplified channels in urban catchments suggest that hydrological differences, particularly storm events, are the major habitat structuring force in our study channels. Stormwater is rarely, if ever, addressed by NCD, therefore this is likely a significant barrier to urban stream restoration success (Walsh et al. 2005a, Bernhardt and Palmer 2007).

The similarity in summer total species richness among stream types (Table 3, Fig. 3) is likely due to high richness of more-tolerant non-EPT taxa in urban and urban restored sites (Appendix C). Higher winter EPT richness probably accounts for the significant effect seen in winter (Table 3, Appendix C). Higher EPT richness at forested sites (Fig. 3) is consistent with the expectation that urbanization typically results in the loss of these sensitive taxonomic groups (Morse et al. 2003, Roy et al. 2003, Cuffney et al. 2010). Urban restored channels did not have higher EPT richness than urban degraded channels (Fig. 3), suggesting that natural channel design is not mitigating the factors responsible for sensitive taxa loss at these locations. The difference in biotic integrity between urban restored and forested channels, and their similarity to urban degraded channels (Fig. 3) indicates that in addition to having lower sensitive taxa richness (i.e., lower EPT richness), these channels contain more tolerant assemblages across all invertebrate groups.

Regression analyses revealed a strong relationship between EPT richness and watershed ISC (Table 4, Fig. 4). Watershed imperviousness is not something easily addressed by reach-scale restoration, thus prioritizing projects with lower ISC or evaluating the spatial arrangement of ISC (Moore and Palmer 2005) during the planning stages may increase the likelihood of successful restoration. However, although ISC cannot be easily altered, its effects may be mitigated by

catchment-based stormwater retention efforts (Walsh et al. 2005a).

Urban degradation leads to compositionally distinct macroinvertebrate communities, which is not successfully mitigated by reach-scale restoration. NMS plots revealed that species composition of restored streams were more similar to each other and to urban degraded streams than to forested streams (Fig. 5). The lack of grouping of forested sites in summer illustrates that although forested sites possess multiple sensitive EPT taxa that primarily delineate them from urban sites (Fig. 5a), there are inter-site compositional differences across all taxonomic groups (Appendix C). While this could be due to the fact that sites were sampled over several summer months, it is also likely local-scale habitat filters differed among forested sites and influenced community composition (Poff 1997). Additionally, although these sites are best-case scenarios of minimally impacted Piedmont streams, they are still subject to human impacts, the extent of which varies among catchments. The dissimilarity between Lower Mud Creek and Mud Creek Tributary in spite of their proximity further suggests that Mud Creek still experiences urban influences at the lower site. In fact, Lower Mud Creek is more similar to the urban degraded site Upper Mud Creek in both seasonal NMS plots (Fig. 5).

Joint plots suggest that summer species composition is explained first by underlying geology (axis 1), and second by catchment and reach-scale urban stressors (axis 2). Axis 1 represents a gradient of high percent dilution to high chloride ion concentration (Fig. 5a). Higher percent dilution is characteristic of streams with higher groundwater and hyporheic exchange (Griffith et al. 2002) and this variable correlated mainly with Northern Outer Piedmont sites. High chloride concentration is probably caused by low groundwater and hyporheic exchange and low summer baseflows characteristic of Triassic basin and Slate Belt streams (Griffith et al. 2002). Axis 2 represents an urban vector that encompasses differences in habitat and water quality,

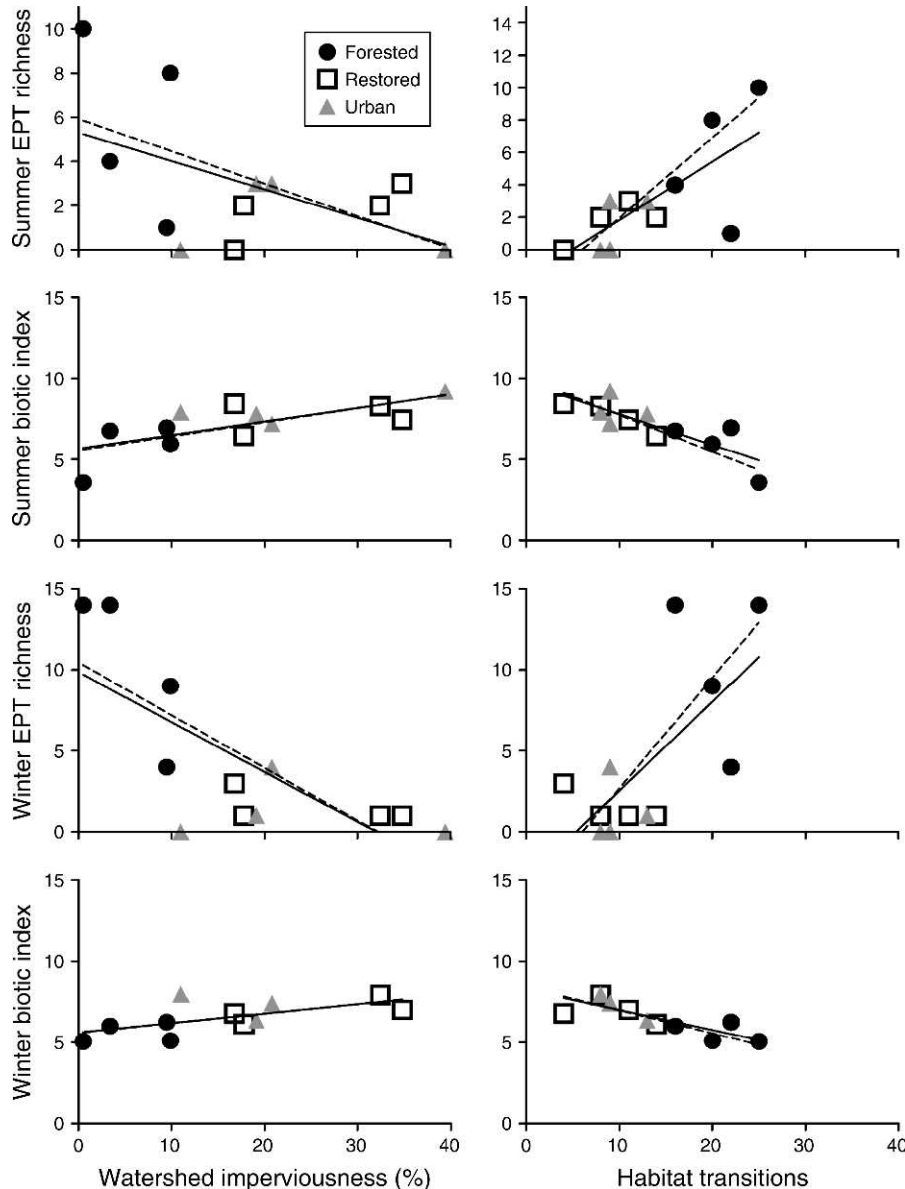


FIG. 4. Linear regression of seasonal macroinvertebrate community metrics vs. watershed imperviousness (percentage of impervious cover in watershed; left panels) and number of habitat transitions per 100-m reach (right panels) with (solid line) and without (dashed line) Lower Mud Creek. Table 4 gives r^2 values.

canopy cover, and hydrological differences and consequently separates forested sites from those in urban catchments (Fig. 5a). This axis best represents our original hypothesis that if urban restoration effectively addresses factors responsible for sensitive taxa loss, restored sites would be at least intermediate between forested and urban degraded endpoints. Our analysis finds no evidence of directional change in composition due to restoration.

The closer clustering of forested sites in winter (Fig. 5b) likely reflects the widespread winter prevalence of shredder taxa such as *Tipula*, *Gammarus*, and *Amphinemura*. An urban vector along axis 1 similar to

that found in summer was the primary axis separating forested sites from urban and restored sites, and once again we found no evidence that the restored stream benthos were distinct from their unrestored urban counterparts (Fig. 5b). Additionally, stream nitrogen concentrations (TN, $\text{NO}_3\text{-N}$) were correlated with urban and restored species composition. Urban catchments deliver more nutrients to streams than undeveloped ones, and nutrient pollution has long been known to influence macroinvertebrate community structure and impair aquatic communities (Bernhardt et al. 2008). There was no clear effect of underlying geology in the winter ordination, in this season chloride concentrations

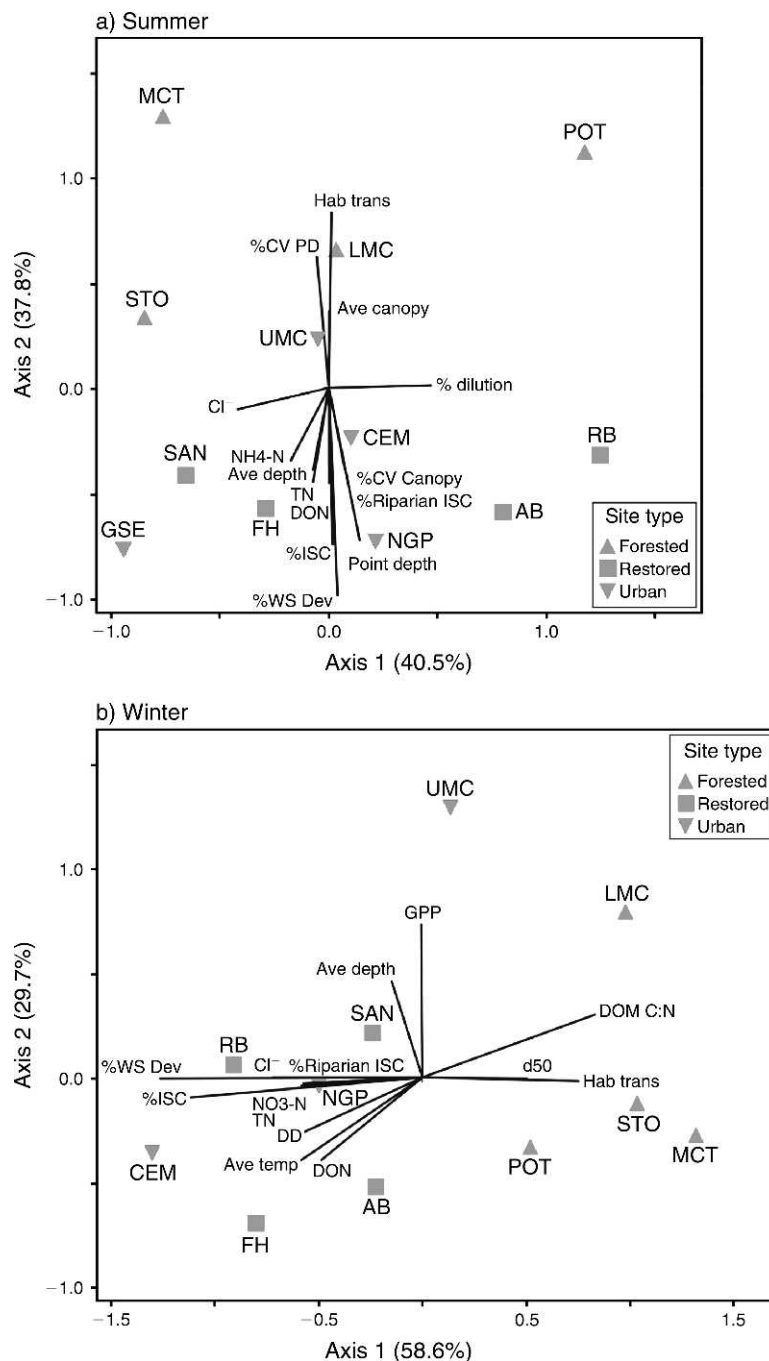


FIG. 5. Two-dimensional joint plots of nonmetric multidimensional scaling (NMS) ordination of (a) summer and (b) winter square-root transformed Bray-Curtis similarities with environmental variables. Minimum explanatory r^2 for environmental variables = 0.3. Final stress = 0.13 for summer and 0.078 for winter. Cumulative r^2 = 0.782 and 0.882 for summer and winter, respectively. Site abbreviations are: AB, Walnut Creek Tributary; CEM, Cemetery Creek; FH, Forest Hills; GSE, Goose Creek; LMC, Lower Mud Creek; MCT, Mud Creek Tributary; NGP, Ellerbe Creek; POT, Pot's Branch; SAN, Sandy Creek; STO, Stony Creek; UMC, Upper Mud Creek. Environmental variable abbreviations (followed by the units in which they were measured) are: Hab trans, habitat transitions; %CV PD, percent coefficient of variation of point depth; Ave canopy, average canopy cover; Cl⁻, chloride concentration (ppm); NH4-N, ammonium-N concentration (ppm); Ave depth, average water depth (m); DON, dissolved organic nitrogen concentration (ppm); %ISC, percentage of watershed with impervious surface cover; %WS Dev, percentage of watershed developed; Point depth, stream point depth (m); %Riparian ISC, percent riparian buffer impervious surface cover; %CV Canopy, percent coefficient of variation of canopy cover; GPP, gross primary production ($\text{g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$); NO3-N, nitrate-N concentration (ppm); TN, total nitrogen (ppm); DD, degree days; Ave temp, average stream temp ($^{\circ}\text{C}$); d50, median particle size (mm); DOM C:N, dissolved organic matter C:N.

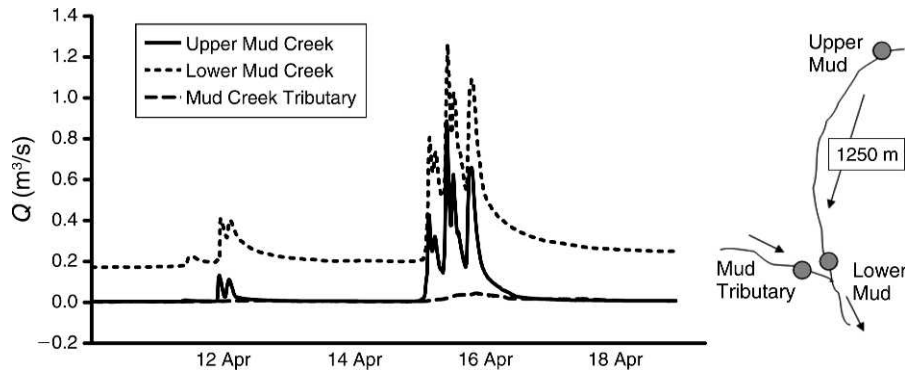


FIG. 6. Hydrographs (flow [Q] over time) from April 2007 and station map of Upper Mud Creek, Lower Mud Creek, and Mud Creek Tributary. Flow readings were made in 10-minute increments.

were highly correlated with impervious cover, suggesting that winter chloride concentrations are dominated by road salt use (e.g., Kaushal et al. 2005). Deep streams with high gross primary production (GPP) were delineated from other sites along axis 2 (Fig. 5b). Together the two axes appear to separate the open canopy urban streams with high nitrogen loading and higher temperatures from closed-canopy forested streams with the cooler temperatures and high stream-water C:N ratios characteristic of forested heterotrophic streams (Fisher and Likens 1973).

The restored streams we studied more closely resembled urban rather than forested endpoints both structurally and biologically, suggesting that restoration activities have not yet led to the recovery of sensitive macroinvertebrate taxa in these streams. The large number of metrics measured in the context of this study provides an unprecedented opportunity to explore what factors are most important for community recovery.

Lower Mud Creek: an unacceptable reference site proves an effective case study of the field of dreams hypothesis

Due to the extreme difficulty of locating watersheds without significant urban or agricultural activity in the North Carolina Piedmont, we made the decision to include Lower Mud Creek (LMC) as one of our forested streams. Based on its geomorphology and its location within ~100-year-old mixed deciduous forest within the protected Duke Forest, the segment we selected on Mud Creek (LMC) appeared to be an acceptable forested stream reach for inclusion in our study. Despite the high habitat heterogeneity, connected floodplain and high canopy cover of LMC, it supports a very depauperate faunal community. Although LMC proved to be a less than ideal forested stream replicate, the mismatch between physical habitat and biological community structure make it an ideal case study for investigating the limitations of reach-scale restoration. Lower Mud Creek was indistinguishable from other forested sites in most measured geomorphic variables, including habitat complexity. Among forested sites however, Lower Mud

Creek had the fewest EPT taxa, and the most tolerant macroinvertebrate assemblage (highest BI value).

The positive correlation between habitat complexity and species richness is well documented (MacArthur and MacArthur 1961, Minshall 1984, O'Connor 1991, Downes et al. 1998, Allan 2004) but may not be causal (Palmer et al. 2010). We speculate that our measure of habitat complexity serves as an indicator of hydrologic disturbance as well as a direct measure of habitat suitability. As such, we must caution that the observed strong positive correlation between habitat complexity and sensitive invertebrate taxa (Table 4, Fig. 4) does not necessarily support the assumption that an increase in habitat complexity will improve biological communities. Indeed, our findings suggest that habitat restoration will prove ineffective if urban stormwaters rapidly rehomogenize restored stream segments, as seen in previous urban restorations (Larson et al. 2001, Booth 2005).

Prior work has suggested that landscape or stream network fragmentation or habitat homogenization may represent an important barrier to macroinvertebrate dispersal in urban catchments and thus may inhibit community recovery in restored urban systems (Blakely et al. 2006, Urban et al. 2006, reviewed in Smith et al. 2009). The proximity of LMC to Mud Creek Tributary (<50 m, Fig. 6), the forested site within our data set with the least impervious cover and the highest diversity of sensitive macroinvertebrate taxa suggests that LMC is not dispersal limited. Indeed, several of the EPT taxa found in the tributary but not in LMC have been sampled in the riparian vegetation surrounding LMC suggesting that dispersal is likely (C. R. Violin, *unpublished data*). The apparent structural integrity and impaired macroinvertebrate community suggests that while habitat complexity is important to faunal diversity, it is, on its own insufficient to support the recovery of biotic communities. This is an important caveat to the utility of the "field of dreams" hypothesis. Mud Creek experiences urban influences along its length, and although the kilometer preceding our LMC study site is entirely forested, this site has a characteristic urban hydrograph due to upstream catchment urbanization

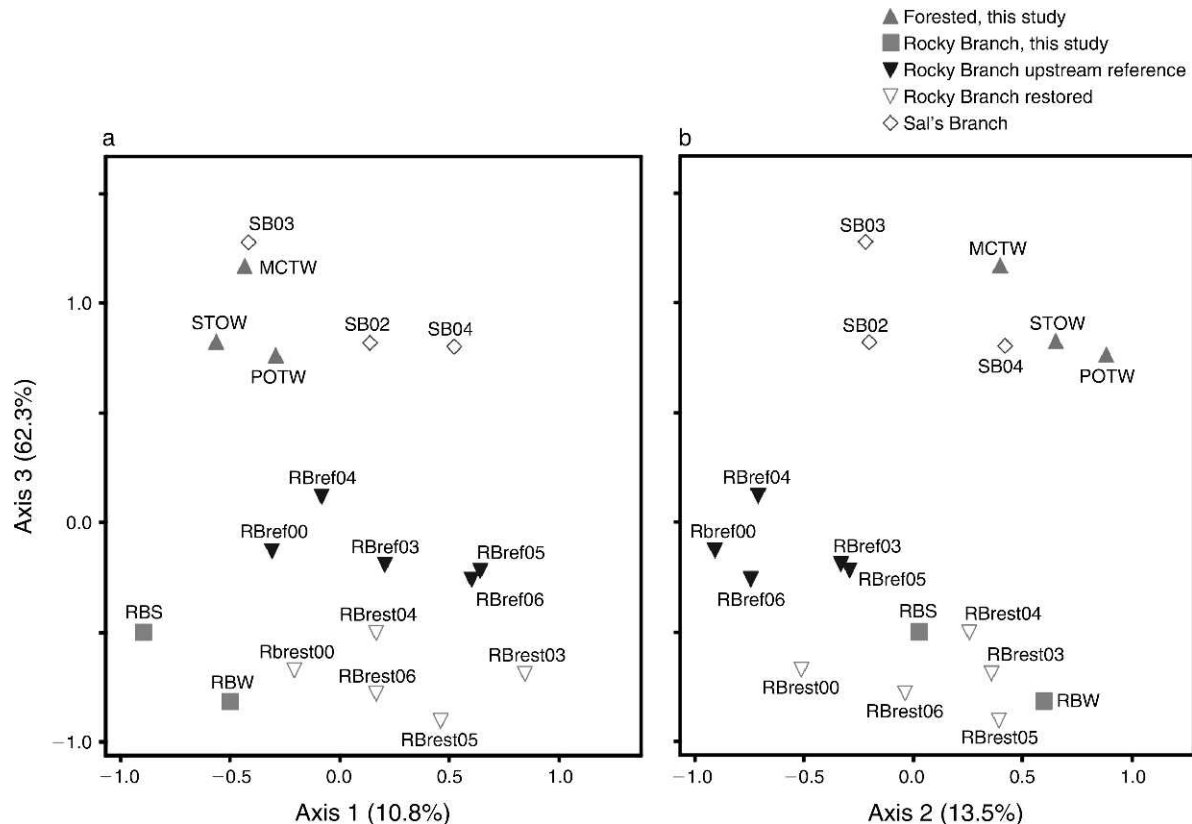


FIG. 7. Two-dimensional representations of (a) axis 3 vs. axis 1 and (b) axis 3 vs. axis 2 of three-dimensional NMS of all Rocky Branch and forested site data (final stress = 0.09, cumulative $r^2 = 0.866$). Axis 3 explained the majority of compositional differences ($r^2 = 0.623$); therefore two-dimensional plots are shown relative to this axis. Site abbreviations are: RBS, Rocky Branch summer sample (from this study); RBW, Rocky Branch winter sample (from this study); RBref00, Rocky Branch monitoring, upstream reference 2000 (pre-restoration); RBref03, Rocky Branch monitoring, upstream reference 2003; RBref04, Rocky Branch monitoring, upstream reference 2004; RBref05, Rocky Branch monitoring, upstream reference 2005; RBref06, Rocky Branch monitoring, upstream reference 2006; RBrest00, Rocky Branch monitoring, restored 2000 (pre-restoration); RBrest03, Rocky Branch monitoring, restored 2003; RBrest04, Rocky Branch monitoring, restored 2004; RBrest05, Rocky Branch monitoring, restored 2005; RBrest06, Rocky Branch monitoring, restored 2006; MCTW, Mud Creek Tributary winter; POTW, Pot's Branch winter; STOW, Stony Creek winter; SB02, Sal's Branch 2002; SB03, Sal's Branch 2003; SB04, Sal's Branch.

(Fig. 6). Hydrologic disturbance is a major driver of macroinvertebrate community structure, as species adapt to local hydrologic conditions (Resh et al. 1988, Townsend et al. 1997, Lake 2000). Previous work has shown that the effective discharge(s) responsible for in-stream habitat structure are not necessarily those responsible for ecological processes such as invertebrate dislodgement (Doyle et al. 2005). Thus, while the urban hydrology of LMC does not cause significant scouring or substrate/habitat homogenization, it may be sufficient to impair aquatic fauna. It is also possible that periods of storm flow introduce urban derived contaminants (e.g., Makepeace et al. 1995, Beasley and Kneale 2002, Kolpin et al. 2002) that may further stress sensitive aquatic taxa. The special case of Mud Creek suggests that even if a restoration project could build intact channels with high floodplain connectivity surrounded by 100-year-old trees and in close proximity to source populations; such an effort would be unsuccessful at promoting the recovery

of a diverse macroinvertebrate assemblage containing sensitive taxa unless the project also was able to mitigate storm flows or the associated pulses of sediments and contaminants through catchment-based efforts (Walsh et al. 2005a, 2007).

Time since restoration

Restoration itself is a catastrophic disturbance to already impaired stream ecosystems (Tullos et al. 2009), and as a result, we expect time lags between restoration implementation and community recovery. One possible explanation for the lack of significant recovery of habitat or biological communities within our restored streams is that insufficient time was allowed for recovery between the restoration implementation and our sampling effort. Our restored study sites were restored one to seven years prior to our sampling effort, however the small sample size and the lack of pre-restoration data precluded us from evaluating the potential role of time

lags in our initial data set. To address this question, we were able to examine long-term data from one of our study sites (Rocky Branch, Raleigh, North Carolina). For this data set we found no significant effect of time since restoration on total species richness, EPT species richness, or biotic index for the five-year post-restoration monitoring period. Three-dimensional NMS ordination of long term monitoring data for the restored reach of Rocky Branch revealed that macroinvertebrate communities from the restored reach of Rocky Branch remained similar in composition to the unrestored upstream urban reach during the five years of post restoration monitoring and remained consistently different from benthic communities in the closest reference stream Sal's Branch (a tributary to Pot's Branch) (Fig. 7). Thus, long-term monitoring of Rocky Branch further supports the conclusions of our synoptic sampling effort, with no evidence of directional change in restored stream reaches either away from the pre-restoration composition or toward reference stream conditions. All available evidence suggests that merely waiting longer prior to evaluating a restoration project is unlikely to lead to different conclusions.

CONCLUSIONS

Our results demonstrate the limited utility of reach-scale restoration to combat the overwhelming effects of watershed urbanization. Within this study, the only demonstrable effect of restoration activities was to remove riparian trees, a practice that may impede recovery. In our study, restoration failed to improve habitat over impaired urban channels, suggesting that watershed level hydrologic processes are degrading restoration efforts. Expanding urban restoration planning beyond the reach scale to include watershed-scale impacts will lead to better restoration design and more positive restoration outcomes.

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APPENDIX A

Study site UTM coordinates (*Ecological Archives* A021-087-A1).

APPENDIX B

Example habitat transition maps and photos for study site block 1 (*Ecological Archives* A021-087-A2).

APPENDIX C

Study site species lists (*Ecological Archives* A021-087-A3).

Twenty years of stream restoration in Finland: little response by benthic macroinvertebrate communities

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Abstract. The primary focus of many in-stream restoration projects is to enhance habitat diversity for salmonid fishes, yet the lack of properly designed monitoring studies, particularly ones with pre-restoration data, limits any attempts to assess whether restoration has succeeded in improving salmonid habitat. Even less is known about the impacts of fisheries-related restoration on other, non-target biota. We examined how restoration aiming at the enhancement of juvenile brown trout (*Salmo trutta* L.) affects benthic macroinvertebrates, using two separate data sets: (1) a before–after–control–impact (BACI) design with three years before and three after restoration in differently restored and control reaches of six streams; and (2) a space–time substitution design including channelized, restored, and near-natural streams with an almost 20-year perspective on the recovery of invertebrate communities. In the BACI design, total macroinvertebrate density differed significantly from before to after restoration. Following restoration, densities decreased in all treatments, but less so in the controls than in restored sections. Taxonomic richness also decreased from before to after restoration, but this happened similarly in all treatments. In the long-term comparative study, macroinvertebrate species richness showed no difference between the channel types. Community composition differed significantly between the restored and natural streams, but not between restored and channelized streams. Overall, the in-stream restoration measures used increased stream habitat diversity but did not enhance benthic biodiversity. While many macroinvertebrates may be dispersal limited, our study sites should not have been too distant to reach within almost two decades. A key explanation for the weak responses by macroinvertebrate communities may have been historical. When Fennoscandian streams were channelized for log floating, the loss of habitat heterogeneity was only partial. Therefore, habitat may not have been limiting the macroinvertebrate communities to begin with. Stream restoration to support trout fisheries has strong public acceptance in Finland and will likely continue to increase in the near future. Therefore, more effort should be placed on assessing restoration success from a biodiversity perspective using multiple organism groups in both stream and riparian ecosystems.

Key words: BACI design; benthic macroinvertebrates; fisheries restoration; long-term effects; restoration assessment; stream habitat enhancement.

INTRODUCTION

During the last two decades, restoration ecology has emerged as a novel scientific discipline with substantial applied importance for environmental managers and decision makers (Ormerod 2003). Few other branches of ecology attract as much interest among the public as does restoration ecology. Ecosystem restoration has raised great optimism among both environmental

managers and academics, not only as a way of mitigating the detrimental impacts of humans on natural ecosystems, but even as a means of preventing the loss of biodiversity (Wilson 1992). Running waters are among the most impacted ecosystems in the world, and restoration of degraded rivers and streams is a key component of many freshwater management programs. For example, the EU Water Framework Directive requires member countries to target restoration measures on streams that fail to reach good ecological status by year 2015. It is therefore alarming to notice that more than 90% of stream restorations in the United States, Australia, and Europe are still not monitored beyond

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visual estimation (Bernhardt et al. 2005, Brooks and Lake 2007), and even the best monitoring designs rarely incorporate data prior to restoration. In a recent meta-analysis, Miller et al. (2009) showed that increasing habitat heterogeneity may enhance benthic macroinvertebrate richness, while changes to density are negligible. However, because of the low number of properly designed and executed studies, and a relatively low effect size, the generality of this finding remains uncertain.

The primary focus of many stream restoration attempts is to enhance stream habitat diversity to create optimal nursery habitat for fish, particularly salmonids. Increased habitat diversity is in turn expected to increase biological diversity, and this assumption has gained empirical support in terrestrial literature (MacArthur and Wilson 1967, Tews et al. 2004). Although some stream studies also show this pattern (Li et al. 2001, Brown 2003), recent reviews have found only weak support for the success of in-stream restoration in enhancing benthic biodiversity, even though a majority of projects did enhance physical habitat heterogeneity (Miller et al. 2009, Palmer et al. 2010). Another complicating factor is that most attempts to monitor restoration impacts have been short, often no more than a few years. Although little is known about the time scale of community recovery (Tullos et al. 2009), it is possible that if new colonizers have to come from distant places, the time for full recovery may be long (Lake et al. 2007).

The most radical exploitation of Finnish rivers took place between 1850 and 1970 when a great majority of streams wider than about 5 m were channelized to facilitate water transport of timber. Timber floating ceased by the end of the 1980s, and legislation imposing restoration on channelized streams was soon established. Stream restoration in Fennoscandia has been motivated mostly by the enhancement of sport fisheries, and restoration is conducted using excavators to add boulders into the stream bed, creating deflectors, boulder dams, weirs, and other in-stream enhancement structures (Nilsson et al. 2005). The added physical heterogeneity enhances retention of organic material (Muotka and Laasonen 2002), but has only limited impact on macroinvertebrate communities (Muotka et al. 2002, Lepori et al. 2005). A conspicuous change to in-stream habitat is that the use of relatively heavy machinery during restoration works typically causes massive eradication of aquatic mosses from large areas of the stream bed, leaving the stream almost devoid of macrophytes (Muotka and Syrjänen 2007). As mosses have a key role in many stream ecosystem processes, Muotka et al. (2002) suggested that the recovery of invertebrate communities may only be possible once mosses have fully recovered. To test this hypothesis, however, a much longer period of monitoring than covered by our previous work (up to eight years) is needed.

In this study, we examined how restoration measures aiming at the enhancement of habitat for juvenile brown trout (*Salmo trutta*) affect other stream biota. For this purpose, we used (1) a well-replicated, balanced, BACI design with data from three years before and after restoration in differently restored and control reaches of six separate streams. This design lacked a natural reference state and, as noted by Downes et al. (2002), attempts to monitor restoration need to consider also the direction, not only the degree of change. Thus, a “double-control” is needed: a degraded control that does not receive restoration and a near-natural control that represents the target state of restoration. Furthermore, three years of post-restoration data may still be too short to detect biological responses, and we therefore complemented the BACI approach by (2) a space–time substitution design including channelized, restored, and near-natural reference streams with an almost 20-year perspective to the recovery of benthic invertebrate communities from restoration. Part of these streams were already visited by Laasonen et al. (1998), allowing a direct comparison of recently restored streams (one to three years prior to sampling by Laasonen et al. [1998]) to the same streams 15–17 years after restoration. Our specific questions were: (1) are total density and species richness of benthic macroinvertebrates higher in the restored than channelized streams (both 1 and 2 above), and do restored streams compare with near-natural streams in this respect (only 2); and (2) does macroinvertebrate community composition reflect a “restoration signal,” i.e., do communities in restored streams differ from those in channelized and/or near-natural streams, and if so, are there any species that might indicate these differences? We focused on density and taxonomic richness as the key response variables, because these are the most commonly recorded measures of the success of stream restoration, and restoration efforts are typically expected to enhance both richness and density (Miller et al. 2009). Also, taxonomic richness of macroinvertebrates, particularly of those involved in decomposition processes (Gessner et al. 2010), may have far-reaching effects on stream ecosystem processes. However, species richness alone may not provide an adequate presentation of community patterns but needs to be complemented by other measures that quantify different aspects of diversity (Magurran 2004). We therefore assessed the effects of restoration by also using Simpson’s index, which takes into account both species richness and species’ relative abundances, but gives weight to the most common taxa, being thus essentially a measure of dominance.

METHODS

Restoration measures

Restoration measures used were similar in both studies, and they closely followed the protocol outlined for stream restoration in Finland (Yrjänä 1998). These measures have been described in detail in several earlier

publications (Laasonen et al. 1998, Muotka et al. 2002, Muotka and Syrjänen 2007), and we therefore only provide a short summary. In-stream restoration in Finland aims at enhancing stream habitat diversity, and it consists of rearranging the stream bottom using boulders that were removed from the channel and placed along stream margins during channelization. This material is used to construct in-stream enhancement structures, e.g., boulder ridges and flow deflectors, placed across the channel. Excavations are also dug to serve as resting sites for fish. Wood is rarely used in stream restoration in Finland. These measures are known to be very effective in enhancing the structural diversity of in-stream habitat at multiple spatial scales (Muotka and Syrjänen 2007).

Study areas and sampling designs

A six-year BACI design.—A BACI study was carried out in six headwater streams in the Oulujoki watercourse, northeastern Finland (64° N, 28° E). These are second-order streams, with a mean annual flow of 0.29–0.70 m³/s and mean width of 2–5.5 m. All streams were channelized for timber floating in the 1950s and 1960s. Responses of juvenile brown trout to restoration measures in these streams are reported elsewhere (Vehanen et al. 2010). In this study, we used the same set up to monitor changes caused by restoration to species richness and composition of benthic macroinvertebrate communities across the six study years. In each stream, three reaches of ~80 m were selected for the study. The reaches were separated by at least 300 m of pool-like (deep, slow-flowing) sections to increase spatial independence among sites within a stream. In each reach, we sampled benthic macroinvertebrates twice a year (June and October) for six years: three years before (1999–2001) and three after (2001–2003) restoration. Each reach was divided into eight sub-reaches of equal length, and one randomly placed sample was collected in each sub-reach on each sample date using Surber sampler (0.04 m², net mesh size 0.3 mm, $n = 8$ samples per reach). Samples were preserved in 70% ethanol and animals were later sorted and identified in the laboratory. Owing to the high number of replicates ($n = 1728$ samples), only genus-level identification could be used. Chironomid midges were only identified to family level and were therefore excluded from most analyses (except total density). In August 2001, one randomly selected section in each stream was restored using boulders (>50 cm), another one using both boulders and large woody debris (diameter = 30–40 cm), while the third reach was left as a control remaining in a channelized state. In this design, experimental units within each stream shared largely similar background conditions of water quality and riparian vegetation. Restoration structures were added as cross-sectional weirs at approximately 5-m intervals, the distance being adjusted by stream slope. A more detailed description of

the study sites and experimental design is given in Vehanen et al. (2010).

Space–time substitution design.—Restored ($n = 15$) and channelized ($n = 8$) streams included in this survey are 3rd-order tributaries of Rivers Iijoki and Kiiminkijoki in northeastern Finland. These streams were restored in the early 1990s and, at the time of sampling, they spanned a post-restoration recovery period of 15 to 17 years. Rivers Iijoki and Kiiminkijoki were used extensively for log floating until the late 1980s, and most tributaries down to a few meters wide were channelized. Restoration works were initiated in the early 1990s, and since then, almost all previously channelized tributaries have been restored along their full length. As these two river systems do not contain any unmodified riffles (except in the upmost headwaters), we included eight near-pristine streams with intact channels in a neighboring watershed, River Koutajoki, as regional references for the rehabilitated and channelized streams. This is the nearest watershed to Rivers Iijoki and Kiiminkijoki that still remains in a near-natural state (for a detailed description of River Koutajoki, see Malmqvist et al. [2009]), so this would be the most likely source of new colonists if the species richness in the restored tributaries of Rivers Iijoki and Kiiminkijoki were to increase. Catchment areas of all the streams included in this survey, even the ones with unmodified channels in River Koutajoki, have been affected by forestry to some degree and no major attempt to restore catchment hydrology has been made in any of the basins. Thus, restoration measures were only taken to enhance in-stream habitat diversity.

Part of the streams (restored, $n = 8$; near-pristine, $n = 4$) included in this survey were sampled by Laasonen et al. (1998) in the early 1990s (1990–1992; hereafter referred to as 1991) when they had been restored one to three years prior to sampling. One of the main weaknesses of studies monitoring restoration success is that they tend to be too short to reliably document biological responses to restoration. We therefore used these repeated snapshots to examine whether streams >15 years after restoration resembled near-natural reference streams in species richness and community composition. The sites with repeated sampling are described in detail in Laasonen et al. (1998) and Muotka et al. (2002). Briefly, restoration consistently enhanced streambed diversity, generally increasing mean water depth, decreasing mean water velocity, and increasing variability in all stream habitat characteristics. In all cases, however, the use of bulldozers for restoration works caused a dramatic loss of aquatic mosses. Muotka et al. (2002) suggested the loss of mosses during, and their very slow recovery after, restoration works to be a key explanation for the lack (or nearly so) of responses by macroinvertebrate communities in restored streams. Therefore, to test whether the recovery of mosses had progressed between 1991 and 2007, we measured the percent cover of mosses

in 30 randomly placed quadrates of 0.25 m² in each stream.

At each site, we selected a relatively uniform riffle of ~50 m. We then collected a 2-minute kick-net sample (25 × 25 cm, net mesh size 0.25 mm) comprising four 30-s subsamples aiming to cover most benthic microhabitats present at a site. A sample of this size covers an area of 1.2 m², and it typically captures about 75% of macroinvertebrate species present at a site, mainly missing species that occur only sporadically in streams. As emphasized by Kappes et al. (2010), space-time substitution designs should be seasonally stratified. We therefore collected all our samples (both 1991 and 2007) during a two-week period in September–October 2007. Samples were preserved in 70% ethanol, and animals were identified in the laboratory, mainly to species level. Chironomid midges were only identified to family level and were therefore excluded from most analyses (except density estimates).

Statistical analyses

BACI design.—We calculated four macroinvertebrate metrics for each study reach on each sample date: number of species, Simpson's diversity index (a measure of dominance), total invertebrate density, and density of leaf-shredding invertebrates. We focused on shredders, because of all invertebrate feeding groups, they are likely to be most directly affected by the enhanced retention capacity of a stream after restoration (Muotka and Laasonen 2002). We also conducted similar analyses on other feeding groups (collector–gatherers, collector–filterers, scrapers, and predators) but as no significant responses to restoration (nonsignificant period × treatment interaction) were detected, we will not report these results in any detail. The impact of restoration on response variables was tested using a linear mixed-effect model (LME), which allows the incorporation of fixed factors and random effects that control for correlation in data arising from grouped observations (Pinheiro and Bates 2000). We incorporated three independent categorical variables as fixed factors: (1) period (B), before (temporal control) vs. after restoration; (2) treatment (TR), boulder enhancement structures vs. both boulders and large woody debris (LWD) vs. control (no enhancement structures; spatial control); and (3) seasons (S) (two surveys each year) nested within periods (S(B)) as an additive factor stabilizing variation among sampling times. Streams and sampling sites nested within streams were fitted as random effects to control for the nonindependence of observations within streams and sites.

Data interpretation followed the logic of beyond-BACI designs (Underwood 1994), with two simplifications: (1) our design was symmetric, and (2) we had independent replicates ($n = 6$) for each treatment and control. In this design, particular interest lies in the interaction ($B \times TR$), which, if significant, implies a long-lasting pulse effect of anthropogenic disturbance

(Underwood 1994). In our case, a significant period × treatment interaction indicates a difference between treatments in how they vary from before to after restoration. We further constructed separate contrasts for the interaction terms between each restoration type and the control. All data were log-transformed prior to analysis. LME models were fitted using the restricted maximum likelihood (REML) method. Statistical significance of period, treatment, season, and period × treatment interaction were tested using F statistics in lme function (nlme library) available in the free software R 2.11.1 (R Development Core Team 2010).

We used nonmetric multidimensional scaling (NMDS) ordination based on the Bray-Curtis dissimilarity coefficient to visualize patterns in macroinvertebrate community composition through time. To avoid the problem of local minima, we ran the analyses using the autopilot mode of PC-Ord (McCune and Mefford 2006), where the program chooses the best solution (i.e., solution with the lowest stress value) from 100 runs of the real data. We used a three-dimensional solution, because change in the stress value was minor with further dimensions. NMDS was performed on log ($x + 1$)-transformed macroinvertebrate data using PC-Ord version 5.31 (McCune and Mefford 2006). To further examine the influence of restoration on community structure, NMDS site scores on each of the three dimensions were analyzed using LME models similar to those for macroinvertebrate metrics. Mean between-group dissimilarity between before vs. after restoration samples was analyzed using MEANSIM6 (Van Sickle 1998).

Space-time substitution design.—We used a priori contrasts on log ($x + 1$)-transformed data to test the hypothesis that the streams restored 15–17 years prior to sampling in 2007 differed from the channelized streams and/or near-natural reference streams in macroinvertebrate density and species richness. To control for differences in abundance, samples were rarified to the lowest abundance observed (476 individuals) using EcoSim version 7.0 (Gotelli and Entsminger 2001). We assessed patterns in macroinvertebrate community structure using three-dimensional solution of NMDS based on Bray-Curtis dissimilarity coefficient. Differences in community composition among channel types were analyzed by multi-response permutation procedure (MRPP), which is a nonparametric method for testing differences in assemblage structure between a priori defined groups. The null hypothesis of no difference among groups was tested by a Monte Carlo procedure with 1000 permutations. If MRPP returned a significant difference, we used indicator value (IndVal) analysis to identify species that best discriminated among the groups (channel types). The indicator value of a species varies from 0 to 100, attaining its maximum value when all individuals of a species occur at all sites of a single group. The significance of the indicator value for each species was tested by a Monte Carlo randomization test

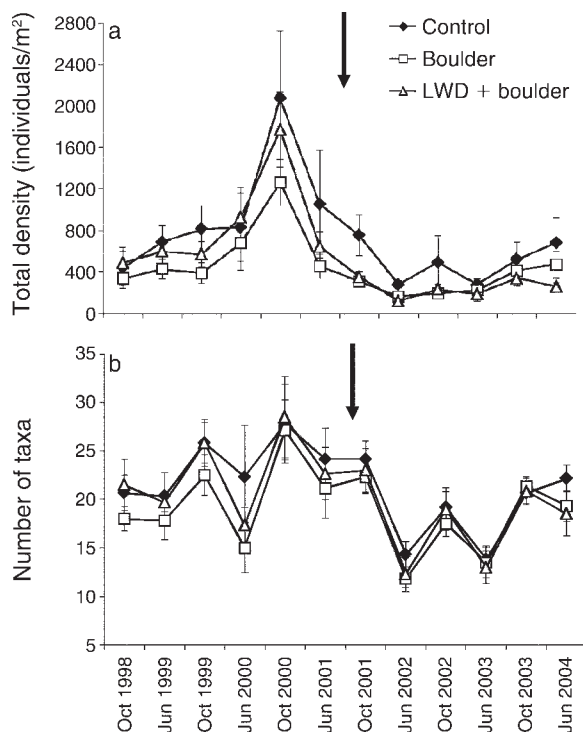


FIG. 1. Before–after–control–impact (BACI) design with differently restored reaches (boulder vs. boulder + large woody debris, [LWD]) and unmodified control reaches in six study streams, sampled from October 1998 to June 2004: (A) mean densities and (B) taxonomic richness of benthic invertebrates. The arrow indicates the timing of restoration works in late summer 2001.

with 1000 permutations. We then ran the same set of analyses using only the subset of the sites (eight restored, four near-natural) that were sampled by Laasonen et al. (1998) in the early 1990s, one to three years after restoration. Unfortunately, only one of the initially four channelized streams sampled by Laasonen et al. (1998) remained channelized in 2007 (others had been restored), so we only included channelized streams ($n = 1$) in NMDS analysis but not in any significance tests. All multivariate analyses were performed on PC-Ord version 5.31 (McCune and Mefford 2006). Mean-between group dissimilarities among channel types were analyzed using MEANSIM6 (Van Sickle 1998).

RESULTS

BACI design

A detailed habitat survey (water velocity, depth, substratum size, cover of aquatic vegetation) was carried out in all study reaches of three streams at summer low flow conditions both before and after restoration. Results from these surveys are reported in Vehanen et al. (2010), and they show that restoration clearly increased water depth and stream habitat heterogeneity, as indicated by, for example, substantially higher CV of water velocity and macrophyte cover after restoration.

For example, the percent increase in CV of water velocity from before to after restoration, corrected for any concurrent change in controls, was 36% for boulder additions and 51% for LWD + boulder additions. The most conspicuous immediate change caused by restoration was, however, the loss of aquatic vegetation (mainly mosses; average decrease of percent moss cover was 42% in boulder additions and 39% in LWD + boulder additions), which showed no sign of recovery during the three years of post-restoration monitoring.

Total macroinvertebrate density varied significantly through time, exhibiting both short-term variation among sampling times and difference from before to after restoration (Fig. 1A, Table 1). The interaction term ($B \times TR$) was also significant, reflecting the fact that although densities decreased in all treatments after restoration, they did less so in the controls than in the restored sections. Interaction contrasts further showed that this difference was mainly caused by a difference between controls and LWD + boulder additions (Fig. 1A, Table 1). The same pattern of significant differences among treatments and periods (and their interaction) remained even if the peak density of October 2000 (Fig. 1A) was removed. This overall pattern reflected mainly densities of leaf-shredding invertebrates, whose densities also decreased more in the LWD-restored streams than in the controls (Table 1). Taxonomic richness also differed significantly between periods (higher richness before than after restoration), as well as between seasons (higher richness in October than in June; Fig. 1B, Table 1). The interaction term was, however, nonsignificant, indicating that temporal changes in richness were unrelated to restoration. Similar to taxonomic richness, Simpson's index displayed temporal variation both between periods (higher values, implying less dominance, before than after restoration) and seasons (higher dominance in June), but there were no differences among treatments in how their dominance patterns varied through time (nonsignificant $B \times TR$ interaction; Table 1).

In NMDS ordination, differences in community composition between restored and controls sites were weak, but samples before and after restoration tended to be slightly separated (MRPP for difference between periods: $A = 0.034$, $P < 0.001$; Fig. 2). This trend was also indicated by the significant main effect of period (before vs. after) for NMDS1, NMDS2, and NMDS3 site scores in the LME model (results not shown). However, the $B \times TR$ interaction term was not significant for any of the ordination dimensions (all $P > 0.25$), showing that treatments did not differ in how their invertebrate communities varied through time.

Space–time substitution design

Total macroinvertebrate abundance was clearly highest in the channelized streams, whereas restored and natural streams did not differ (ANOVA, $F_{2,28} = 3.71$, $P = 0.037$; a priori contrasts: restored vs. channelized, $t_{28} =$

TABLE 1. (A) *F* values and level of significance of linear mixed-effects model testing for the effects of restoration on the density of all macroinvertebrates, density of leaf-shredding invertebrates, taxonomic richness, and Simpson's index, and (B) interaction contrasts.

Source of variation	df	Total density			Shredder density			No. taxa			Simpson's index		
		<i>t</i>	<i>F</i>	<i>P</i>	<i>t</i>	<i>F</i>	<i>P</i>	<i>t</i>	<i>F</i>	<i>P</i>	<i>t</i>	<i>F</i>	<i>P</i>
Main effects													
Before vs. after = B	1, 190		99.97	<0.001		127.71	<0.001		32.96	<0.001		22.75	<0.001
Among treatments = TR	2, 10		2.53	0.129		1.10	0.371		3.40	0.075		0.86	0.454
Seasons (B) = S(B)	5, 190		11.79	<0.001		40.675	<0.001		22.60	<0.001		7.56	<0.001
B × TR	2, 190		4.50	0.012		3.59	0.030		1.24	0.292		1.16	0.316
B) Interaction contrasts													
Boulder vs. Control	190	−0.09		0.925	−0.39		0.694	0.80		0.427	0.80		0.424
LWD vs. Control	190	−2.64		0.009	−2.49		0.014	−0.78		0.437	1.52		0.130

Notes: Treatments (TR): control (no added structures) vs. boulder enhancement structures vs. both boulders and large woody debris (LWD) added. Period (B): before vs. after restoration. Seasons (S): sampling occasions (June and October each year) nested within period S(B). Sites within streams were included as a random variable; other factors were treated as fixed.

2.44, $P = 0.021$; restored vs. natural, $t_{28} = 0.29$, $P = 0.776$; Fig. 3A). By contrast, species richness showed no difference between the channel types ($F_{2,28} = 1.286$, $P = 0.29$; Fig. 3B). Simpson's index was closely similar in all channel types (mean $D \pm 1$ SE, 0.822 ± 0.04 for channelized, 0.850 ± 0.01 for natural, and 0.869 ± 0.02 for restored streams), with no significant differences in any of the comparisons ($P > 0.15$). The three channel types did not form distinct clusters in the NMDS ordination space (Fig. 4); nevertheless, MRPP returned a significant, albeit weak, difference among the restored and natural streams ($A = 0.023$, $P = 0.015$), whereas restored and channelized streams did not differ significantly ($A = 0.010$, $P = 0.116$). IndVal identified two significant indicator species for the restored and six for the natural streams (Table 2). However, as multiple feeding modes were present in both channel types, this

analysis did not provide any clear ecological interpretation for the observed differences.

We then used a subset of the data consisting of streams visited by Laasonen et al. (1998) in the early 1990s, one to three years post-restoration, and re-sampled by us in 2007, allowing a 15–17-year recovery period from restoration. Moss cover in the recently restored streams was much lower than in the natural streams (a priori contrast, $t_{13} = 3.57$, $P = 0.003$; Fig. 5A). By 2007, mosses seemed to have recovered to some degree, but the difference between the restored and natural streams was still significant ($t_{13} = 2.40$, $P = 0.036$). Macroinvertebrate abundance was higher in the natural than restored streams in 1991 (mainly because of very high numbers of chironomid midges in some samples; a priori contrast, $t = 3.74$, $P = 0.002$) but not in 2007 ($t = 0.53$, $P = 0.606$; Fig. 5B). Species richness

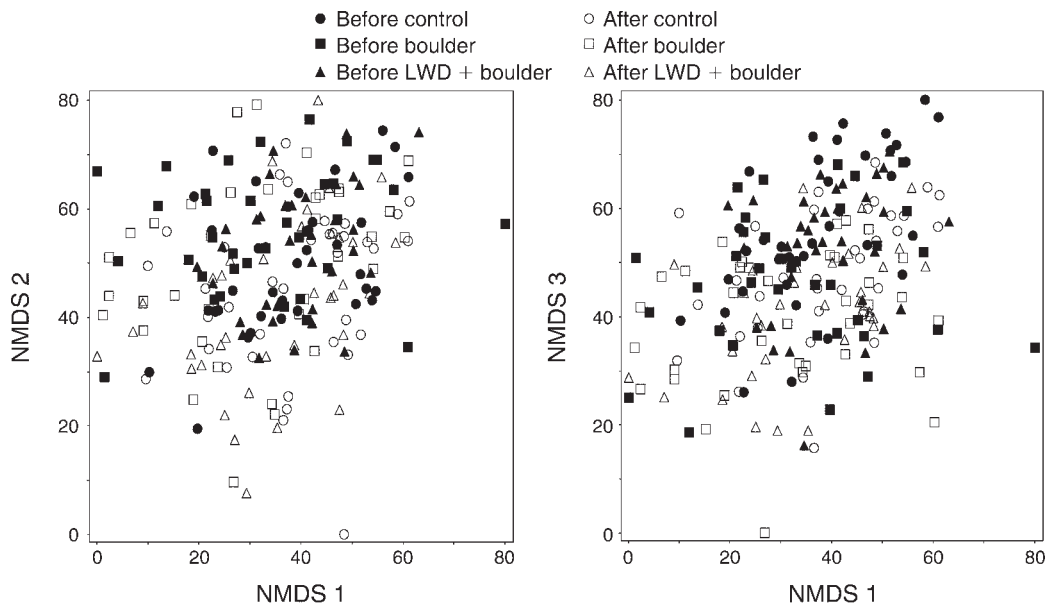


FIG. 2. Nonmetric multidimensional scaling (NMDS) ordination of the study reaches included in the BACI design, before (black symbols) and after (open symbols) restoration. Stress value for a three-dimensional solution was 15.2, and mean between-group dissimilarity (before vs. after) was 0.641.

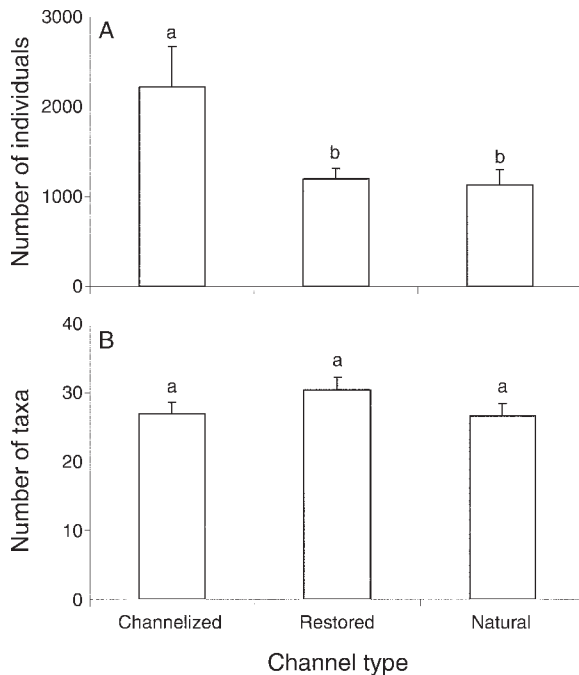


FIG. 3. (A) Abundance and (B) species richness of benthic macroinvertebrates in the space-time substitution design, including channelized streams ($n = 8$), streams restored 15–17 years prior to sampling ($n = 15$), and near-natural reference streams ($n = 8$). Values are means and SE. Stream types that do not share a letter differ significantly ($P < 0.05$).

was higher in the restored than natural streams in 1991 ($t = 2.73$, $P = 0.017$) but did not differ among the channel types on the latter sampling time ($t = 0.44$, $P = 0.667$; Fig. 5C). Comparisons between restored and channelized streams could not be done because only one of the initially four streams in this group remained channelized by 2007. For community composition, the difference between the restored and natural streams in 1991 bordered at significance ($A = 0.031$, $P = 0.06$), whereas in 2007 communities did not differ among the channel types ($A = 0.014$, $P = 0.164$). Interestingly, invertebrate communities in the natural streams did not differ between the two sampling years ($A = 0.006$, $P = 0.407$), whereas in the restored streams, they did ($A = 0.046$, $P = 0.002$; Fig. 6). In three of the restored streams, where moss cover differed strongly between years (at least 56% increase in moss cover), a conspicuous change in macroinvertebrate species composition could be seen, with the appearance of six species of filter-feeding caddis larvae (Hydropsychidae) in 2007 that were missing in the 1991 survey. Also the abundance of *Rhyacophila* caddis larvae in these three streams increased by 60–92%.

DISCUSSION

We detected little response by macroinvertebrate communities to in-stream restoration, both in the short-term BACI approach and in the long-term comparative study. Therefore, we can safely conclude that while the in-stream habitat enhancement measures used clearly increased stream habitat diversity, they had little impact on benthic biodiversity (see also Laasonen

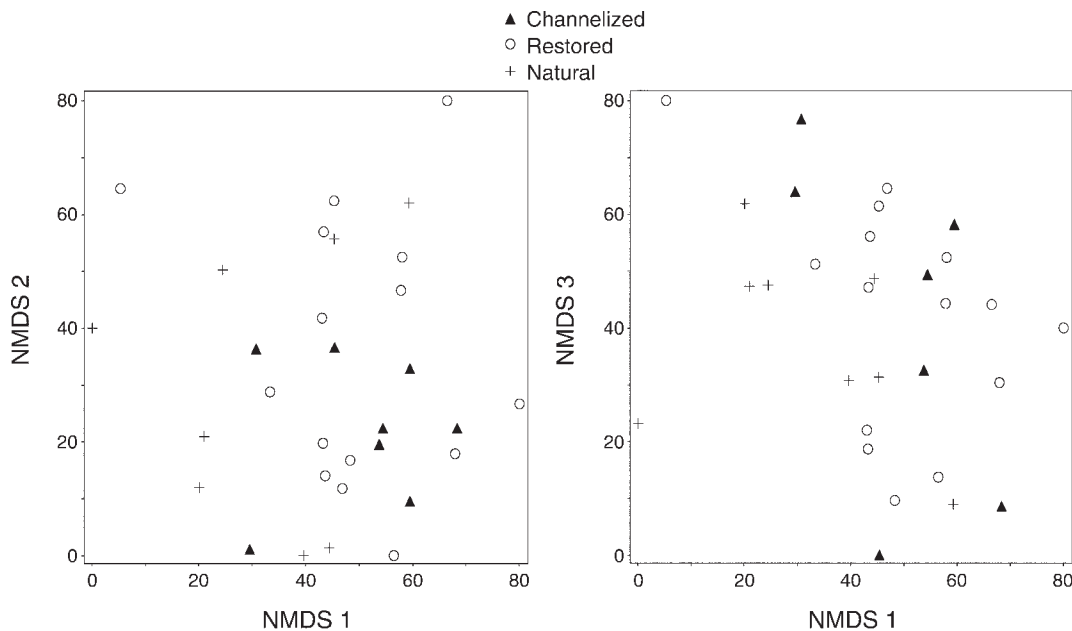


FIG. 4. NMDS ordination of the sites included in the space-time substitution design. Stress value for a three-dimensional solution was 15.6, and mean between-group dissimilarities were 0.704 for channelized vs. restored and 0.686 for restored vs. natural channels.

TABLE 2. Indicator values (IndVal; maximum value = 100%) for the most important indicator species discriminating between the restored (RE) and near-natural (NA) streams in the space–time substitution design.

Species	Observed IndVal (%)		IndVal from Monte Carlo tests			FFG
	RE	NA	Mean	SD	P	
<i>Ephemera mucronata</i>	82.5	4.0	59.0	12.50	0.017	C-G
<i>Hydropsyche pellucidula</i>	67.7	6.0	46.7	10.75	0.045	C-F
<i>Baetis muticus</i>	0	60.7	22.8	9.86	0.002	C-G/Scr
<i>Ameletus subalpinus</i>	0	50.0	18.1	7.70	0.008	C-G
<i>Protonemura intricata</i>	0	50.0	18.3	7.97	0.009	Shr
<i>Haprophlebia</i> sp.	0	37.5	13.6	7.02	0.025	C-G
<i>Hydropsyche saxonica</i>	0	37.5	15.1	6.85	0.023	C-F
<i>Capnia</i> sp.	0	37.5	15.9	6.77	0.034	Shr

Notes: Monte Carlo tests based on 1000 permutations were used to assess the significance of each species as an indicator for the respective stream group. Only species with an indicator value of at least 35% and significant at $\alpha = 0.05$ are given. The main functional feeding group (FFG) for each species is also indicated (C-G, collector–gatherer; C-F, collector–filterer; Scr, scraper; Shr, shredder).

et al. 1998, Muotka et al. 2002, Lepori et al. 2005). Although this observation is not in line with the general ecological theory, where a positive relationship between habitat diversity and species richness is often taken as granted, the same pattern was detected by Palmer et al. (2010) in a comprehensive review of stream restoration studies. These authors concluded that the direct effect of habitat diversity on stream macroinvertebrates is overwhelmed by several other factors, e.g., flow-related and/or anthropogenic disturbances. In boreal streams also, landscape-level disturbances (e.g., forestry) may well be more limiting to macroinvertebrate communities than lack of in-stream habitat diversity, and restoration efforts should therefore be prioritized according to a comprehensive watershed assessment (Roni et al. 2008). In-stream restoration may then not be a top priority but may still be important once other obstacles to successful restoration have been removed. Unfortunately, such watershed assessments are rare in Finland, where in-stream restoration is forced by legislation, with the implicit assumption that what is beneficial for trout, must be beneficial for other stream organisms as well. This, however, remains an untested hypothesis (see Muotka and Syrjänen 2007).

Why, then, should macroinvertebrates exhibit such weak responses to increased habitat heterogeneity? We can think of at least four potential, not mutually exclusive, explanations: (1) dispersal limitation by macroinvertebrates; (2) dispersal limitation by a habitat-forming organism group, aquatic bryophytes; (3) overriding influence of broad-scale climatic factors; and (4) modest changes to stream habitat heterogeneity caused historically by channelization. The “species credit” hypothesis (Hanski 2000) states that, because many organisms are inherently dispersal limited, it may take a long time for them to get established in newly restored sites, even though these sites may be perfectly suitable for them. The dispersal ability of stream macroinvertebrates varies considerably, some species being almost completely confined to freshwater environments, while others may disperse tens of kilometers as winged adults (Thompson and Townsend 2006). It is

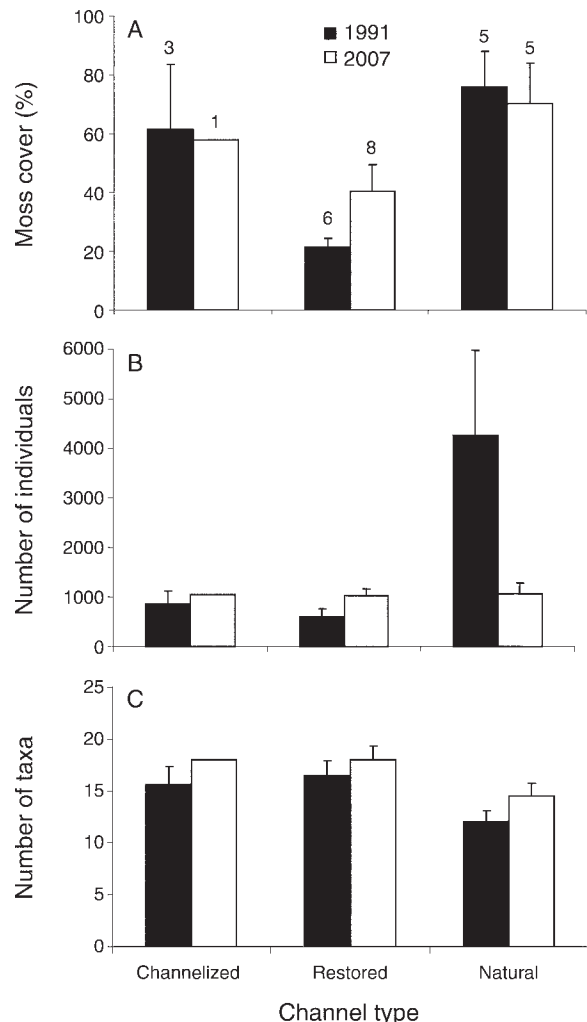


FIG. 5. (A) Moss cover, (B) total benthic macroinvertebrate abundance, and (C) species richness in channelized, restored, and natural streams sampled repeatedly in 1991 vs. 2007. Values are means and SE. Numbers above each bar indicate the number of sampling sites for each channel type.

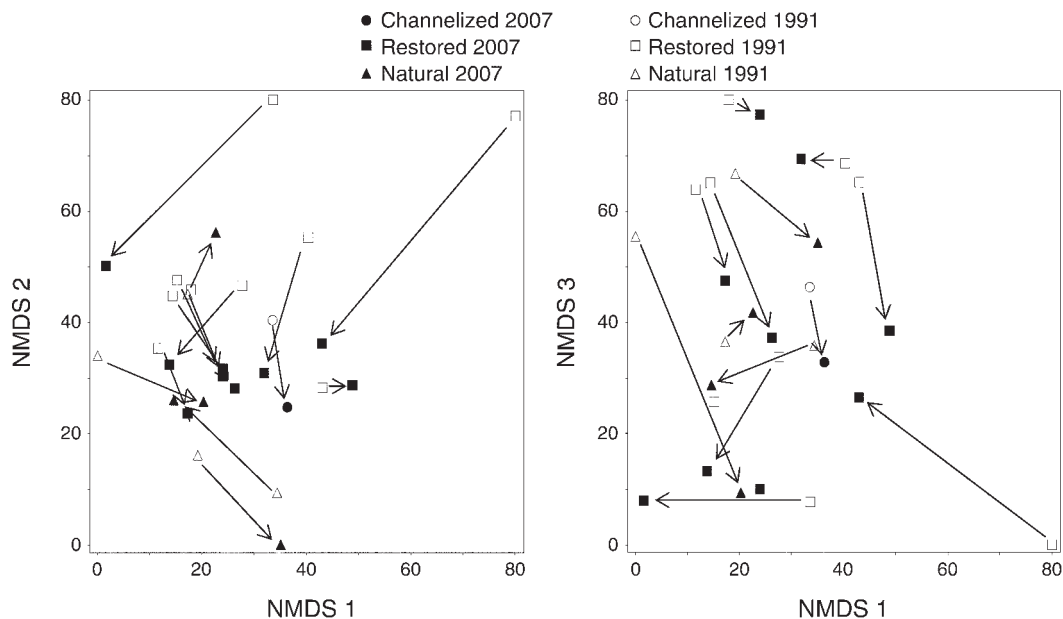


FIG. 6. NMDS ordination of benthic invertebrate communities in streams sampled in both 1991 (open symbols) and 2007 (black symbols). An arrow combines the two sampling occasions for each particular stream. Only one channelized stream could be sampled in both 1991 and 2007. Stress value for a three-dimensional solution was 13.7.

therefore conceivable that the time lag of our study (15–17 years) was still too short for all species in the regional species pool to have reached the restored sites. However, according to restoration practices in Finland, rivers are restored along their full length, so our study sites were not isolated stretches within otherwise degraded channels. Conversely, practically all riffles in rivers used for timber floating were channelized, and therefore the closest unmodified streams will rarely be in the same watershed. In our case, the closest intact streams are quite close (less than 100 km in River Koutajoki), however, and the colonization distances should not have been too long for most species to reach within almost 20 years. Lack of response by stream biota to restoration has been interpreted to reflect large-scale degradation of running water ecosystems (Jähnig et al. 2009). Then, the regional species pool no longer provides propagules for recolonization, and to reach the restored sites, colonizers should disperse very long distances from remote watersheds. In our study area, forestry is the only major form of land use, and although forestry may bear a historical legacy on stream macroinvertebrate communities for many decades (Harding et al. 1998), northern Finland is certainly one of the least impacted areas in Europe.

Another possibility is that the recovery of macroinvertebrate communities reflects the slow renewal of their key habitat resource, aquatic bryophytes. Many stream invertebrates are strongly dependent on bryophytes as their microhabitat, and owing to the use of heavy machinery, moss cover decreases radically during restoration works (Muotka and Laasonen 2002). Within-stream dispersal of stream bryophytes mainly

takes place via spores and fragments, and it is quite possible, as observed for macrophytes (Engström et al. 2009), that in-stream restoration enhances the retention of moss fragments travelling downstream, thus facilitating bryophyte dispersal. However, practically nothing is known about the among-stream dispersal rates and distances of stream bryophytes (Stream Bryophyte Group 1999), but if this is dependent on the presence of an atmospheric “spore cloud,” as suggested for terrestrial bryophytes (Hylander 2009), it may be a very slow and stochastic process. As restoration works do not destroy all the mosses, however, it is likely that most of the recolonization observed between 1991 and 2007 came from the same river system. Indeed, using genetic data, Hughes (2007) concluded that natural colonization of restored sites is only likely from within the same stream. It is therefore somewhat surprising that so little recovery occurred within the 15–17-year period between our two surveys. While this may certainly have repercussions on invertebrates, we suspect that it affects the relative abundances of moss-dependent organisms more than their presence/absence. Indeed, the slight adjustments in community composition of restored streams from 1991 to 2007 toward natural reference streams (Fig. 6) may reflect these changes. This is partly supported by the fact that in three of the restored streams where mosses became much more abundant between the years, five species of Hydropsychidae and two species of *Rhyacophila* spp. caddisfly larva were only detected, or became more abundant, in 2007. In several previous studies, these species have been reported to

show strong preference for moss-covered stones (Malmqvist and Sjöström 1984, Muotka 1993).

A growing body of literature shows that regional climatic variability may override localized management efforts (Bradley and Ormerod 2001, Suding et al. 2004), and because extreme hydrological events are expected to be more frequent in the future (Hintzman et al. 2005), many monitoring studies will be plagued by such unpredictable broad-scale variability. This was exactly what happened in our BACI design, because in 2002, a year after the restoration, discharge was at a record low in Finnish rivers (Finnish Environment Institute 2002). This partly confounded the detection of restoration-induced changes to macroinvertebrate density and diversity which both decreased after restoration. However, as a parallel decrease also occurred in unmodified controls, this response could not have been caused merely by restoration. Importantly, however, the response was stronger in restored than in control reaches, and densities decreased strongly already in October 2001, soon after restoration and well before the onset of drought. Therefore, restoration-induced disturbance must also have been involved. Tullios et al. (2009) suggested that channel reconfiguration acts as a “disturbance filter” that affects macroinvertebrates for many years after restoration. In our case, recovery from restoration may indeed have been slowed down by the subsequent drought. This may also explain the rather unexpected result of strongly decreased densities of shredding invertebrates in the LWD-restored streams; initially, we expected shredder densities to increase after wood addition because this should enhance the retention capacity of a stream to riparian litter inputs (Johnson et al. 2003). However, very low water levels in years following restoration shifted large areas of stream bed to slowly flowing pools, and this was particularly pronounced in reaches with added logs (Vehanen et al. 2010). These conditions may have been unsuitable for many shredders, most of which were relatively sensitive stonefly species.

When Fennoscandian streams were channelized for log floating, only major flow obstructions were removed from the channel, and the loss of habitat heterogeneity was only partial (Nilsson et al. 2005). Therefore, heterogeneity may not have been limiting macroinvertebrate communities in our streams to begin with. Furthermore, although we lack historical, pre-channelization data, as well as data from the active timber floating period (prior to the 1980s), it seems that if channelization and floating activities did cause any direct harm to benthic communities, the 20–30-year period after the cessation of timber floating until our first survey should have been enough to allow full community recovery. Therefore, one might ask whether these streams needed any restoration measures at all. In fact, although the channelized streams were hydro-morphologically degraded, their biological communities did not differ strongly from regional references and,

therefore, under current criteria emphasizing biodiversity, these streams would not be defined as in need of restoration. Twenty years ago, however, biodiversity did not have much weight in environmental decision making in Finland, and the primary motivation for restoring the streams used for timber transport was to enhance salmonid fisheries. In a companion paper, we used the same BACI set-up to show that stream habitat enhancement had little impact on brown trout populations (Vehanen et al. 2010). Similarly, Stewart et al. (2009) questioned the use of in-stream enhancement structures for fisheries purposes, maintaining that their widespread use is not based on scientific evidence. It thus seems that stream restoration using such structures may represent a case of “over-application of oversimplified concepts to complex systems” (Hilderbrand et al. 2005). Nevertheless, stream restoration in its present form has a strong public acceptance in Finland, mainly because it is considered beneficial for commercially important fish, ecotourism, and the well-being of local people (Oikio and Eloranta 2007). Thus, as numerous restoration projects are currently underway in Finland, and many more are likely to come, it is important that any damage to lotic biota, particularly aquatic mosses, be minimized by leaving areas of streambed untouched to aid the recolonization by mosses.

Some recent studies have shown that, through changed hydrology (e.g., increased frequency and duration of flooding) and creation of additional habitats, in-stream restoration may in fact have quite strong impacts on riparian plant and insect communities (Helfield et al. 2007, Jähnig et al. 2009). For example, Jähnig et al. (2009) showed that channel reconstruction in Central European rivers had no impact on benthic invertebrates, whereas species richness of floodplain vegetation and ground beetles increased strongly. These positive responses are perhaps not that surprising, given the recent evidence that stream and riparian food webs are tightly intertwined (Nakano and Murakami 2001, Marczak and Richardson 2007). As the trophic links between streams and their riparian forests are easily disturbed by human intervention (Baxter et al. 2004), future research should pay more attention to the effects of stream restoration on the reciprocal energy fluxes between these two strongly interconnected ecosystems. Assessment of stream restoration success may have been too strongly biased toward benthic macroinvertebrates and more emphasis in future projects should be given to multiple organism groups in both streams and stream-side forests and floodplains.

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River restoration success depends on the species pool of the immediate surroundings

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Abstract. Previous studies evaluating the success of river restorations have rarely found any consistent effects on benthic invertebrate assemblages. In this study, we analyzed data from 24 river restoration projects in Germany dating back 1 to 12 years and 1231 data sets from adjacent river reaches that lie within 0–5, 5–10, and 10–15 km rings centered on the restored sites. We calculated restoration success and recolonization potential of adjacent river reaches based on stream-type-specific subsets of taxa indicative for good or bad habitat quality. On average, the restorations did not improve the benthic invertebrate community quality. However, we show that restoration success depends on the presence of source populations of desired taxa in the surrounding of restored sites. Only where source populations of additional desired taxa existed within a 0–5 km ring around the restored sites were benthic invertebrate assemblages improved by the restoration. Beyond the 5-km rings, this recolonization effect was no longer detected. We present here the first field results to support the debated argument that a lack of source populations in the areas surrounding restored sites may play an important role in the failure to establish desired invertebrate communities by the means of river restorations. In contrast, long-range dispersal of invertebrates seems to play a subordinate role in the recolonization of restored sites. However, because the surroundings of the restored sites were far from good ecological quality, the potential for improvement of restored sites was limited.

Key words: aquatic communities; benthic invertebrates; dispersal; habitat heterogeneity; regional species pool; river rehabilitation; source population.

INTRODUCTION

Over the last decades, public awareness of not only the ecological but also the economic and social values of rivers has gradually risen, resulting in an increasing number of river restoration projects (Bernhardt et al. 2005, Palmer et al. 2005, Dudgeon et al. 2006, Jähnig et al. 2011). Commonly, restoration projects intend to reconstruct a channel form that is similar to a historic form or least-disturbed reference site (Palmer 2009). As degraded river sections often show comparatively monotonous conditions and a low availability of potential habitats, river restoration practices often apply the habitat heterogeneity hypothesis assuming that an increase in habitat heterogeneity will promote biodiversity (Ricklefs and Schluter 1993). However, the biotic effects of river restoration on aquatic assemblages are still poorly understood, and many projects do not yield the expected results. Palmer et al. (2010) reviewed 78

restoration projects, among which only two showed statistically significant increases in benthic invertebrate biodiversity, suggesting an uncoupling between structural and biological diversity.

The reasons for failure are diverse, but are most often argued along the following lines. First, even after an increase in habitat heterogeneity, some stressors persist and limit the development of the aquatic community (Lake et al. 2007). Such stressors include water pollution and inadequate sediment quality (Kowalik and Ormerod 2006). These often come along with persistent land use changes in the catchment area. Further stressors are temperature changes (Lehmkuhl 1972), and constraints in other habitats that species use during periods of their life cycle (Briers and Gee 2004). Second, long-lasting impacts lead to a loss of biodiversity, which results in the disappearance of many sensitive species from large parts of their former distribution range (Zwick 1992). Species colonizing a local site are drawn from the regional species pool (Cornell and Lawton 1992). In many cases, however, streams and rivers are impacted not only on the local scale but also on the catchment scale, so that the regional species pool will decrease and consist mainly of opportunistic, highly tolerant generalists (Lake et al. 2007).

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The recolonization of restored stream sections by benthic invertebrates therefore also depends on the availability of a regional species pool and the species dispersal propensities. Many studies examining the movement of adult insects assume dispersal to be limited, as the number of specimens rapidly decreases with increasing distance to the stream (Petersen et al. 2004, Blakely et al. 2006, Hughes 2007, Masters et al. 2007; but see Coutant 1982 and Ims 1995). Some recent genetic studies, however, indicate that dispersal over long distances within and across catchments may be common (Kovats et al. 1996, Kelly et al. 2001, Wilcock et al. 2003, 2007). Independent of dispersal, Lake et al. (2007) predicted that the existence of a diverse regional species pool is a prerequisite for a successful recolonization. The more additional taxa that are available in the regional species pool, the more likely it is that the number of taxa will rise after restoring a site.

By analyzing the outcome of 24 restoration projects (treatment–control approach) and the assemblages in the vicinity of the restored sites, our study investigates the relationship between the community improvement at restored sites and the regional species pool. Thereby we used a stream type-specific approach, which was not based on general species assemblage metrics, but on stream type specific sets of benthic invertebrate species indicating good or bad habitat quality to estimate the recolonization potential within three rings of different distances from the restored site. We then addressed the following questions: (1) Is restoration success affected by the quality of the communities in the surroundings? and (2) Is there a relationship between distance from a regional pool of desired taxa and the colonization of desired taxa at a restored site? We hypothesized that improvement of the benthic invertebrate assemblages through restoration will only be observed where source populations of habitat-specific desired taxa are present in close vicinity to the restored site. With this study, we aim at revising current and developing new restoration strategies so that they are more likely to produce the desired effects in benthic invertebrate communities.

MATERIAL AND METHODS

Restoration projects

We investigated 24 river restoration projects that have been carried out over the last 12 years with the goal of improving the habitat diversity (Table 1). At each restoration site, two river sections were sampled. One section was situated within the restored reach. A second section was selected further upstream outside of the restored reach and served as a control (space-for-time substitution; mean distance between sections 1 km). The still-degraded control sections were selected to best represent the conditions of the rehabilitated section prior to restoration.

The investigated restoration projects were quite diverse, including re-creation of natural features in straightened reaches, reconnection of side channels,

and improvement of lateral connectivity. The successful achievement of these objectives was documented by Jähnig et al. (2011), who investigated 26 restoration projects of which 19 were identical to those in our study. Unfortunately, the achievement of objectives of the remaining restoration projects were not followed up but there were no principal differences in the restoration practices compared to the other sites. A detailed overview of project goals and measures undertaken to reach them is given in Table 1. The mean length of rehabilitated river sections was 1.2 km, and the average cost was US\$670 500 (€528 000) per site. Information on the location of restoration sites, including their position within the river network, is provided along with the land use in the catchment (CORINE land cover data) in Appendices A and B. Settlement area, agricultural area (with sub-types arable land and permanent crops and pastures and heterogeneous agricultural areas), forested area, and surface water area were differentiated. The catchment of one site was dominated by settlement area, the catchment of eight sites by arable land and permanent crops, five by pastures and heterogeneous agricultural areas, and 10 by forested area. Averaged over all sites, agricultural areas (both sub-types combined, mean percentage 56%) dominated the landscape, which was also reflected in a high level of nutrients. In particular, total phosphorus was critical at 20 out of 24 sites (>0.3 mg/L; data available online).^{2,3} Stream order of the restoration sites were determined on 1:50 000 maps (TOP 50 Hessen and TOP 50 Nordrhein-Westfalen; Geogrid-Viewer 1.1, EADS Deutschland GmbH, Munich, Germany).

Sampling design

Benthic invertebrates were sampled in all sections ($N = 48$ sections), following the official EU Water Framework Directive (WFD) compliant sampling protocol applied in Germany (Haase et al. 2004a). Multi-habitat samples, which reflect the proportion of the microhabitat types that are present with $\geq 5\%$ cover, were taken from each river section. At each section, 20 sample units were taken, each notionally 25×25 cm in size, resulting in ~ 1.25 m² of river bottom being sampled. All samples were taken from March to July in 2007 and 2008, according to the protocol for collecting samples in river monitoring programs to assess the ecological status of rivers in Germany. At each site, samples of the restored and control sections were taken on the same day by the same person.

The sample material was preserved in 70% ethanol and was transferred to the laboratory where it was sorted following the method of Haase et al. (2004a, b). Taxa were identified to the level proposed by Haase et

² <http://www2.hmuenv.hessen.de/umwelt/wasser/wrrl/stadtlandfluss/hessen/ergebnisogdetail/>

³ <http://www.elwasims.nrw.de/ims/ELWAS-IMS/viewer.htm>

TABLE 1. Characterization of the restoration projects.

Site		Parameter					Goals		
		Size of catchment (km ²)	Stream order	Length of restored section (km)	Invested costs (thousand Euros)	Time since restoration (yr)	Increasing physical heterogeneity	Flood prevention	Longitudinal connectivity
1	5	25	4	1.0	312	5		x	
2	9	314	6	0.8	800	2			x
3	5	30	2	1.0	210	4	x		
4	5	14	3	0.3	55	2	x		x
5	9.2	2375	7	1.0	800	2	x	x	
6	9.2	1290	6	2.0	45	2	x		
7	19	154	6	1.2	650	1	x		x
8	9	173	4	0.9	104	4	x		
9	5.1	29	3	0.4	97	5	x		
10	9	885	6	0.2	12	7	x		
11	5	11	3	0.8	18	3	x	x	
12	9	650	6	0.3	111	7	x		
13	9	1200	6	0.5	253	6	x		
14	9	1168	6	1.5	320	1	x	x	x
15	9	226	5	2.5	1444	3	x	x	
16	5	153	5	0.3	84	5	x		
17	12	658	5	0.8	1000	7	x	x	
18	19	71	4	2.0	300	5	x	x	x
19	9	1000	7	0.8	798	3	x	x	
20	5	83	5	2.0	282	3	x		
21	19	116	4	2.6	1700	12	x		
22	12	250	4	4.3	2600	10	x	x	x
23	5	33	4	0.4	230	8		x	
24	9	384	5	0.4	457	1	x	x	
Mean		470	5	1.2	528				

Notes: Stream types are classified according to Pottgiesser and Sommerhäuser (2004): type 5.1, small fine substrate dominated siliceous highland rivers; type 5, small coarse substrate dominated siliceous highland rivers; type 9, mid-sized fine to coarse substrate dominated siliceous highland rivers; type 9.2, large highland rivers; type 12, mid-sized and large organic substrate-dominated rivers; type 19, small streams in riverine floodplains. Goals and measures for each site are marked with "x."

al. (2006a), which ensured that taxa lists were comparable with regard to their taxonomic resolution. For most taxa, the resolution was at the genus or species level, but selected families such as Chironomidae, Naididae, or Tubificidae were determined only to the sub-family or family level. This taxa list is compliant to the EU WFD.

Restoration success and recolonization potential of the surrounding

It is generally assumed that species are habitat limited and that species soon colonize or recolonize if suitable habitats are created in the process of restoration of structurally degraded streams. The set of species that is expected to recolonize a restored site is stream-type specific and depends on stream characteristics, e.g., stream size, catchment geology, water hardness, current velocity, and sediment size spectrum. For this reason, only a subset of the species occurring in the proximity of the restored site, although in habitats with different properties (e.g., in small tributaries while the restored section is in the main-stem river), will be able to recolonize restored river sections. Restoration measures should support the colonization of those species that are characteristic for the stream type under consideration, while establishment of foreign species to a habitat type

should be barred. The assessment of the recolonization potential from the surrounding area thus also needs to be based only on this specific subset of taxa that can be determined from species identity or traits, rather than on simplified assemblage metrics such as species richness or diversity. In this study, we used a novel approach to evaluate restoration success on a stream-type-specific basis. For each stream type (classified in Pottgiesser and Sommerhäuser 2004), each benthic invertebrate taxon has been assigned an indicator value between -2 and 2 (Lorenz et al. 2004). Positive values indicate a high correlation of the taxon's presence with a good ecological status of its habitat, while negative values indicate a correlation of its presence with a bad ecological status of the habitat (Haase et al. 2006b). Depending on stream type, 154–348 taxa were classified as diagnostic for ecological quality of their habitat. All other taxa were assigned a value of 0, as they can be linked to neither good or bad habitat quality. As indicator values of taxa are stream-type specific, these values may vary between different stream types. For example, *Leuctra nigra* (Plecoptera), a typical species in small streams, is assigned an indicator value of 2 for small streams (stream type 5.1, Table 1), however, as *L. nigra* does typically not occur in larger rivers, it is assigned a value of 0 for the corresponding main-stem

TABLE 1. Extended.

Goals	Measures							
Lowering of entrenchment depth	Removal of bank fixation	Wood placement	Installation of flow deflectors	Elongation of river length	Creating a new water course	Creation of multiple channels	Extensification of land use	Reconnection of back waters
				X	X			
		X			X		X	
X	X	X	X	X	X	X	X	
	X	X	X	X	X	X		
					X	X		X
								X
	X	X		X	X		X	
	X		X					
X		X					X	
								X
X				X	X			
X	X	X	X			X	X	
X	X		X	X			X	
X	X	X	X	X	X		X	
	X	X	X	X	X	X	X	
	X		X		X	X	X	
X	X	X	X	X	X	X		X
X	X	X	X			X	X	
	X		X				X	
	X			X	X		X	
	X			X	X		X	
	X				X			
X	X	X	X	X	X	X	X	

stream (stream type 9, Table 1). Based on this information, we derived which taxa to take into account when estimating the recolonization potential of surrounding sites. Thus, *L. nigra* was considered when the restoration was carried out in small streams (stream type 5), whereas it was ignored when the restored stream section belonged to larger streams (stream type 9). Hereafter, the taxa that were considered as a regional species pool for a certain restored site are called “diagnostic taxa.” Based on the abundances of all diagnostic taxa at a sampling site, we calculated an assemblage metric called the fauna index (FI; Lorenz et al. 2004) using the following formula:

$$FI = \frac{\sum_{i=1}^n sc_i a_i}{\sum_{i=1}^n a_i} \quad (1)$$

where sc_i is the indicator value of taxon i , and a_i the abundance class of taxon i . Abundance classes were defined as class 1, 1–3 individuals; class 2, 4–10 individuals; class 3, 11–30 individuals; class 4, 31–100 individuals; class 5, 101–300 individuals; class 6, 301–1000 individuals; class 7, >1000 individuals (Lorenz et al. 2004). As each taxon is assigned an indicator value, FI describes the quality of a community rather than being a diversity index. Its values can vary between 2, which indicates a community exclusively composed of typical taxa for the respective stream type and –2, which

would be the poorest community composition imaginable, containing only species typical for degraded habitats. In practice, however, in many stream types, typical reference communities at undisturbed conditions do not reach values of 2. Therefore, we also indicated empirical FI values from reference conditions.

To estimate the recolonization potential (RP) of the surroundings to a restored section we analyzed benthic invertebrate sampling data in the surrounding river reaches of the restored sites that were gathered by German governmental environmental agencies during the years 2004 to 2008. The sampling method followed the same routines as ours (Haase et al. 2004a, b, 2006a). Sampling sites within 0–5, 5–10, and 10–15 km rings around each restored section were identified. Only samples from sites within the same main catchment where the restoration project took place were considered. The resulting mean numbers of available samples were 11.5, 16.5, and 23.2 per ring, respectively (Table 2). Since an increasing number of samples may lead to more rare species being detected (Heatherly et al. 2007), we calculated FI for each sample separately and averaged individual FI values per ring afterward. SP was defined as the average difference in FI between all samples within $r = 0$ –5, 5–10, and 10–15 km rings around a restored site ($FI_{pot,r}$) and FI at the control site (FI_{cont}):

$$SP_r = FI_{pot,r} - FI_{cont} \quad (2)$$

TABLE 2. Number of available benthic invertebrate samples within 0–5, 5–10, and 10–15 km rings around each of the 24 restored sections.

Site no.	5 km	5–10 km	10–15 km	Total
1	19	13	22	54
2	5	14	10	29
3	17	18	10	45
4	13	20	31	64
5	5	16	11	32
6	5	14	15	34
7	9	11	13	33
8	10	16	58	84
9	9	5	21	35
10	39	27	20	86
11	26	8	8	42
12	10	10	27	47
13	14	25	36	75
14	17	30	37	84
15	7	22	57	86
16	7	11	16	34
17	1	4	1	6
18	10	36	47	93
19	0	6	15	21
20	16	18	20	54
21	12	26	35	73
22	1	9	9	19
23	22	29	37	88
24	2	8	3	13
Mean	11.5	16.5	23.3	51.3
SD	9.0	8.8	15.9	26.8

The restoration success (RS) was calculated as the difference in FI between the restored (FI_{rest}) and control site:

$$RS = FI_{\text{rest}} - FI_{\text{cont}} \quad (3)$$

For orientation, total abundances, number of taxa, and Shannon-Wiener diversity (Lloyd et al. 1968) of restored and control sections are also provided. Moreover, for each stream type, FI values under reference conditions are given. This allows estimating the dimension of improvement or deterioration of restored sections.

Statistics

Paired t tests were used to analyze overall restoration success by comparing total abundances, number of taxa, and Shannon-Wiener diversity and FI between the restored and control sections. The effects of the length of restored section, time since restoration, dominant land use type in the catchment, stream order, SP, and FI_{cont} on RS were tested with multiple regression analysis. Stepwise backward selection based on the Akaike information criterion (AIC) was used to find the best model. One-way ANOVA followed by Fisher LSD post hoc test were used to test differences in the average FI between the restoration project sites and the surroundings at different distances. Normal distributions of RS, FI_{cont} , and FI_{rest} were tested with the Shapiro-Wilk test (all $P > 0.34$), and homogeneity of the

variances of FI_{cont} , FI_{rest} , FI_{pot5} , FI_{pot10} , and FI_{pot15} were tested with Levene test ($F_{4,114} = 0.66$, $P = 0.62$).

RESULTS

On average, restored sections did not harbor more individuals or a higher number of taxa than control sections (paired t test; $t = 1.40$, $P = 0.18$ and $t = 1.72$, $P = 0.10$, respectively). Mean differences were 210 ± 2406 and 3 ± 8 , respectively, between the restored and control sites (Appendix C). The Shannon-Wiener diversity at restored sections did not differ significantly from that of control sections (paired t test; $t = 0.98$, $P = 0.34$). FI at the control sections ranged between -1.90 and 0.67 , and FI at the restored sections ranged between -1.86 and 0.81 (Fig. 1a). The effect of the river restoration on FI was inconsistent (paired t test; $t = 0.06$, $P = 0.95$). On average, restoration changed FI by 0.31 ± 0.22 units, and the maximum increase and decrease were $+0.68$ FI units (site 5) and -0.79 FI units (site 12), respectively. Except for site 7, all the restored river sections still had comparatively low FI values that differed considerably from reference conditions (Fig. 1a). A detailed analysis of FI in the surroundings of the 24 sites revealed that communities were deteriorated in almost all catchments as they were far from their own reference values (data not presented). They also had only low recolonization potentials to the restored sites (Fig. 2).

Multiple-linear-regression models showed that only the recolonization potential within a ring of 0–5 km around the restored sections could significantly explain RS (Table 3). None of the other variables, namely level of FI at the control site, stream order, land use in the catchment, time since restoration, and length of the restored river section, influenced success or failure of river restoration (Table 3, Fig. 3a).

With only a few exceptions, restoration improved FI when a positive recolonization potential existed within this distance while restoration typically decreased FI when the surroundings provided a negative recolonization potential to the restored site. This correlation faded with each increasing ring radius. For a ring radius of 5–10 km, the effect of SP on RS became marginally insignificant (Table 3, Fig. 3b), and when considering SP for ring radii of 10–15 km, no effect of SP on RS was found (Table 3, Fig. 3c).

DISCUSSION

We found a positive relationship between the regional species pool and the restoration outcome. Sites at which the community quality (i.e., FI) was increased after restoration were surrounded by river reaches that harbored a higher number and higher abundances of the stream type-specific taxa. Where no such pool of desired taxa existed, restoration efforts most often did not increase community quality. Therefore, our results support our hypothesis and confirm the prediction by Lake et al. (2007) that recolonization of restored sites depend on intact regional species pools. However, this

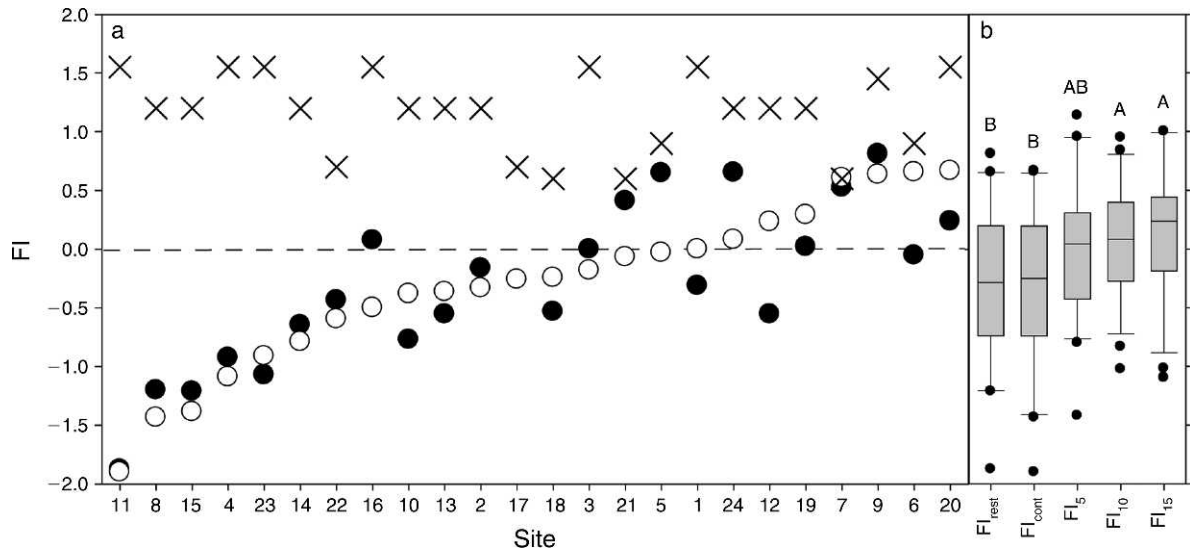


FIG. 1. (a) Fauna index (FI) of the individual restoration projects, at the restored (solid circles) and at the control (open circles) sites. FI values for stream-type-specific reference conditions are also given (X). The site names correspond to the numbers given in Table 1. (b) FI at the restored (FI_{rest}) and control (FI_{cont}) sites, as well as in the surroundings of the restoration projects at sites within 0–5, 5–10, and 10–15 km rings ($FI_{pot,5}$, $FI_{pot,10}$, and $FI_{pot,15}$, respectively). Significant differences according to ANOVA/Fisher LSD post hoc test are indicated with different letters above the boxplots. Boxes indicate the 25th and 75th percentiles, bars indicate the 10th and 90th percentiles, dots indicate the 5th and 95th percentiles, and mid lines show the mean.

study also raises the issue that the recolonization potential of a restored river section cannot be measured based on unspecific metrics like taxa richness or diversity indices—especially because only a subset of the species occurring in the proximity of the restored site will be able to recolonize formerly degraded sections. Moreover, many re-colonizing taxa do not necessarily mean that the quality of the community at restored sections is improved. Hence, a stream-type-specific approach is necessary, focusing on the set of diagnostic taxa for the individual stream type. In our study, the regional species pools of diagnostic taxa were however in most cases much less diverse than would have been desirable. This catchment-scale impoverishment of the species pool is likely the reason that, even in restoration projects that yielded an increase in FI, the increase was very limited.

We cannot preclude that with an average period of 4.5 years since the restoration projects were implemented, recolonization was still in progress at some sites, and the final result was not measurable. Langford et al. (2009) showed that recolonization of heavily deteriorated sites may still be in progress 30–50 years after the improvement of the habitat conditions. Furthermore, some species that recolonized the restored section at very low numbers may have remained undetected by our sampling method. In the long-term, especially aquatic insects with winged adult stages might have the potential to recolonize restored habitats from neighboring catchments (Briers et al. 2004).

On shorter time scales, however, our study indicates that dispersal of benthic invertebrate taxa over distances

larger than 5 km is uncommon. Field studies using mark–recapture experiments or light traps show that flight dispersal typically occurs along water bodies within a corridor of no more than a few meters to 100 m in width (Petersen et al. 1999, MacNeale et al. 2005). Direct observations of long distance overland dispersal in aquatic insects are rare, due to methodological problems with mark–recapture techniques in small specimens and the dilution effect in the landscape (but see Coutant 1982), among other reasons. Thus, direct detection of long-term dispersal will probably underestimate true dispersal. Studies analyzing the genetic relationship of aquatic populations evade these methodological weaknesses. Genetic studies, in particular those based on microsatellite markers, can provide insight into recent dispersal events, however, they may also suffer from other methodological and conceptual weaknesses (reviewed in De Meester et al. [2002] and Bohonak and Jenkins [2003]). Microsatellite studies analyzing the population structure of benthic invertebrates deliver contrasting results for different species. While *Plectrocnemia conspersa* (Trichoptera), for example, is suggested to be a strong disperser, facilitating gene flow within and between catchments, *Polycentropus flavomaculatus* showed strong genetic differentiation, indicating much more limited dispersal propensities (Wilcock et al. 2007). Even in the taxonomic group of damselflies, who are generally believed to be comparatively strong flyers, *Coenagrion mercuriale* (Odonata) turned out to be a poor disperser (Watts et al. 2005). Thus, our understanding of feasible dispersal distances and mechanisms driving dispersal patterns of benthic invertebrates in rivers is

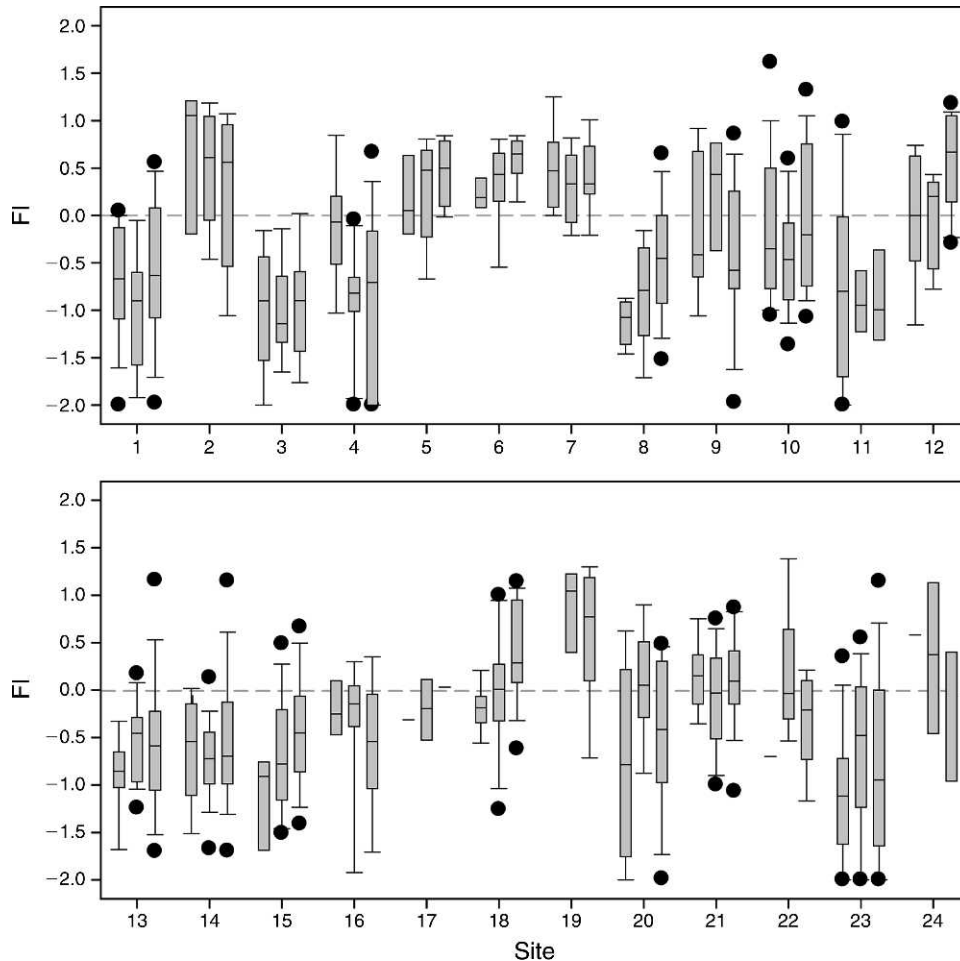


FIG. 2. FI in the surroundings of the 24 restoration sites within 0–5, 5–10, and 10–15 km rings centered on the restored sites (from left to right, respectively). At site 19, no data were available within the ring of 0–5 km. Site names correspond to the numbers in Table 1.

still restricted. Dispersal of aquatic stages of benthic invertebrate species is even more limited. For these, dispersal is generally believed to be more effective downstream because of passive drift (Williams and

Williams 1993, Turner and Williams 2000); yet, drifting distances are typically short (Elliott 2003).

In addition to dispersal capabilities of benthic invertebrates and the time that is needed for recoloniza-

TABLE 3. Results of the backward selection of variables explaining restoration success in a multiple linear modeling approach, minimizing the Akaike information criterion (AIC).

Model	AIC	Null dev	Res dev	Variables	df	<i>t</i>	<i>P</i>
Start	34.28			SP_r	1		
				FI_{cont}	1		
				stream order	1		
				land use	3		
				length	1		
				age	1		
0–5 km	21.69	3.3	2.7	SP_5	1	2.24	0.036
5–10 km	23.62	3.4	2.9	SP_{10}	1	1.84	0.079
10–15 km	25.04	3.4	3.1	FI_{cont}	1	–1.39	0.18

Note: The initial model is shown (start), as are the best models with minimal AIC after backward selection for radii of 0–5 km, 5–10 km, and 10–15 km. SP_r is the recolonization potential within the ring r , FI_{cont} is the fauna index at the control section, stream order is the stream order of the restored site, land use gives the dominant land use type in the catchment in which the restoration was performed, and age is the number of years since the restoration was carried out. For the outcome models, null deviance (null dev), residual deviance (res dev), and degrees of freedom (df) are given.

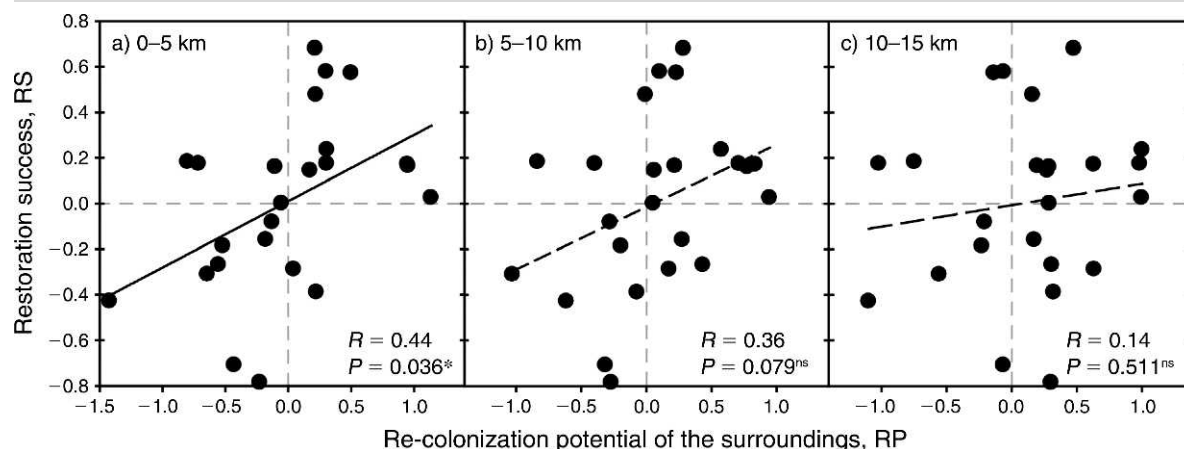


FIG. 3. Regression of restoration success (RS) vs. the recolonization potential (RP) of river reaches in the surrounding areas of each restored site within distances of (a) 0–5, (b) 5–10, and (c) 10–15 km. Significant regressions are shown as solid lines; nonsignificant regressions are shown as dashed lines.

tion, other aspects can also affect restoration success. These include the scale of the restoration and water quality. We need to ask whether in-stream restorations alone can provide suitable habitats for all life stages of benthic invertebrates; e.g., riparian vegetation is expected to have strong impact on survival and success on adult stages through alteration of microclimate, habitat structure, and potential food sources (Briers and Gee 2004). Hence, it has been argued that restorations should also incorporate the riparian stream interface as well as adjacent floodplains (Beechie et al. 2010, Sundermann et al. 2011). Another potential cause that may impede benthic invertebrates from establishing at restored sites is water pollution. In our study, many sites exceed thresholds for total phosphorus and elevated loads of other nutrients or chemicals occur. Organic or chemical loads were probably also part of the reason why benthic invertebrate communities, both at restored sites and in their surroundings, were far from reference conditions. More research is needed to establish the relationship between benthic invertebrate community richness and diversity, relevant scales of restorations, as well as important stressors that persist after physical river restoration, such as water pollution. This research should aspire to identify the suitable dimensions of restorations and the order in which multiple stressors should be tackled (water quality, physical habitat quality). Future restoration projects should therefore not only aim to improve the flow regime and the channel form, but require a more holistic approach, including consideration of the physical and biological potential of the site and the scale at which projects are implemented (Beechie et al. 2010). As one of the key components for success, the present study reveals the importance of evaluating the regional species pool to estimate the probability of restoration success before the restoration is carried out. Only where nearby source populations of the desired taxa persist does restoration have the

potential to increase the quality of benthic invertebrate assemblages in the short term. We, thus, recommend making use of available data (e.g., from environmental agencies) when considering the latter aspect. If there are only limited data available for the assessment of the catchment-wide species pool, the use of substitute parameters can be an option. For example, some studies found that the outcome of restoration measures is related to land use in the catchment (Tullos et al. 2009, Miller et al. 2010). Thus, land use may be used as a proxy for the degree to which the regional species pool is intact; the assumption is that catchments with a high percentage of urban areas should have a more depleted regional species pool than in agricultural areas, which again have more depleted species pools than in unattended areas. However, for this approach, land use should be rather diverse between, but monotonous within, catchments. In our study, where high percentages of settlement and agricultural areas dominated most catchments, catchment-wide land use was not correlated with the recolonization potential in the surroundings.

Selection of appropriate sites is crucial for the success of river restoration projects and our results show that projects in severely deteriorated rivers have a high probability of failure. The results of this study also indicate that, if done in a suitable environment (i.e., if there is a positive recolonization potential between the surroundings and the site that is to be restored), restoration of the physical river habitat and habitat diversification are successful tools for improving the benthic invertebrate community quality. To enhance biodiversity in the long term, it might be feasible to implement many small projects instead of one large project; this would create stepping-stones to facilitate dispersal from more remote species pools, and thereby the recolonization of depleted catchments (Rouquette and Thompson 2007, Jähnig et al. 2008). However,

dispersal via such stepping-stones may take many years (Langford et al. 2009), which is often too long with regard to the short-term assessment periods for restoration success under environmental directives such as the EU WFD (European Union 2000). Where populations are depleted at the catchment scale, assisted migrations from the nearest source population may offer a tool to support establishment of desired taxa (Layzer and Scott 2006, Smith et al. 2009, Vitt et al. 2009). Even though highly controversial (Ricciardi and Simberloff 2009, among others), this tool may be the only measure to attain the goal of increased biodiversity in large-scale degraded catchments over relevant time scales for water management, especially for species with very low dispersal capabilities. Therefore, we feel that a broader discussion is needed regarding the time frame for evaluating restoration success and the risks and benefits of assisted migrations as a potential tool in river restorations (McLachlan et al. 2007, Mueller and Hellmann 2008).

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APPENDIX A

Location of each restoration project in Germany, including its position within the river network and the land use in the catchment draining to each restoration site (*Ecological Archives* A021-088-A1).

APPENDIX B

Land use in the catchments of restored river sections (*Ecological Archives* A021-088-A2).

APPENDIX C

Abundances, number of taxa, and Shannon-Wiener diversity indices for restored and control sections (*Ecological Archives* A021-088-A3).

Testing the Field of Dreams Hypothesis: functional responses to urbanization and restoration in stream ecosystems

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Abstract. As catchments become increasingly urban, the streams that drain them become increasingly degraded. Urban streams are typically characterized by high-magnitude storm flows, homogeneous habitats, disconnected riparian zones, and elevated nitrogen concentrations. To reverse the degradation of urban water quality, watershed managers and regulators are increasingly turning to stream restoration approaches. By reshaping the channel and reconnecting the surface waters with their riparian zone, practitioners intend to enhance the natural nutrient retention capacity of the restored stream ecosystem. Despite the exponential growth in stream restoration projects and expenditures, there has been no evaluation to date of the efficacy of urban stream restoration projects in enhancing nitrogen retention or in altering the underlying ecosystem metabolism that controls instream nitrogen consumption. In this study, we compared ecosystem metabolism and nitrate uptake kinetics in four stream restoration projects within urban watersheds to ecosystem functions measured in four unrestored urban stream segments and four streams draining minimally impacted forested watersheds in central North Carolina, USA. All 12 sites were surveyed in June through August of 2006 and again in January through March of 2007. We anticipated that urban streams would have enhanced rates of ecosystem metabolism and nitrate uptake relative to forested streams due to the increases in nutrient loads and temperature associated with urbanization, and we predicted that restored streams would have further enhanced rates for these ecosystem functions by virtue of their increased habitat heterogeneity and water residence times. Contrary to our predictions we found that stream metabolism did not differ between stream types in either season and that nitrate uptake kinetics were not different between stream types in the winter. During the summer, restored stream reaches had substantially higher rates of nitrate uptake than unrestored or forested stream reaches; however, we found that variation in stream temperature and canopy cover explained 80% of the variation across streams in nitrate uptake. Because the riparian trees are removed during the first stage of natural channel design projects, the restored streams in this study had significantly less canopy cover and higher summer temperatures than the urban and forested streams with which they were compared.

Key words: *benthic organic matter; ecosystem functions; nitrate uptake; stream ecosystem metabolism; stream restoration; transient storage; urbanization.*

INTRODUCTION

In 2008, for the first time in human history, more people live in urban than in rural areas (United Nations 2008). As low-lying lines in the landscape, the streams draining urban watersheds are strongly affected by the altered hydrology of urban catchments, particularly the dramatic increases in stormwater runoff that result from increasing impervious surface cover (ISC; Booth and

Jackson 1997). Urban watershed hydrology leads to many physical and biological changes in the streams (Paul and Meyer 2001), which have together been termed the “urban stream syndrome” (Meyer et al. 2005, Walsh et al. 2005). The high peak flows characteristic of urban streams tend to homogenize streambed sediments, reduce fine- and coarse-scale flow habitat variation, and incise stream channels leading to disconnected riparian zones. At the same time that urban streams become more “pipe like,” urban infrastructure (drains, pipes, sewers, roads) routes watershed contaminants more directly from the landscape into receiving streams (reviewed in Bernhardt et al. 2008).

In the United States, a common strategy of stream management is some form of stream restoration (Bernhardt et al. 2005). Stream restoration in urban

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settings is frequently based on natural channel design (*sensu* Rosgen 1994, 2007) and aimed at reconfiguring the channel and restoring riparian vegetation to create a stream reach that structurally resembles a less impacted (or reference) reach, with the assumption that habitat enhancement or creation is sufficient to restore biotic integrity (Bernhardt and Palmer 2007). This approach has been termed the Field of Dreams hypothesis (Palmer et al. 1997), after the 1989 movie of the same title, whose catch phrase was “If you build it, they will come.” Although conceptual links between habitat complexity and biotic diversity have been suggested for some time (Gorman and Karr 1978), there is limited empirical evidence that natural channel design approaches can promote biological recovery (Miller et al. 2009). Even fewer studies have analyzed the efficacy of reach-scale habitat restoration on altering stream ecosystem functioning in order to reduce nutrient loading, even though water quality improvement is the most common goal of U.S. stream restoration projects (Bernhardt et al. 2005). A few studies have shown enhanced nutrient uptake following stream restoration (Bukaveckas 2007, Roberts et al. 2007, Kaushal et al. 2008).

Stream ecosystem functions

Ecosystem function metrics synthesize complex interactions with implications for nutrient cycling and water quality. Stream ecosystem gross primary production (GPP) is the total production of energy in a stream, usually expressed in units of grams of oxygen or carbon per square meter per day (Odum 1956, Bott 2006). It is primarily driven by nutrient and light availability to stream autotrophs (Hill et al. 2001, Mulholland et al. 2001, Flecker et al. 2002), and may be limited by the availability of stable habitat (Grimm and Fisher 1989, Uehlinger 2000). Ecosystem respiration (ER), expressed in the same units, is the total consumption of energy in a stream (Odum 1956, Bott 2006). One important component of ER is autotrophic respiration. ER is additionally driven by heterotrophic respiration, which is largely dependent upon supply and retention of benthic organic matter (BOM) and streamwater dissolved organic carbon (DOC; Webster and Benfield 1986, Wallace et al. 1997). Decomposition of organic matter in streams can be limited by nutrient availability (Tank and Webster 1998), and is also constrained by the availability of stable habitats and instream structures that are capable of retaining BOM (Webster et al. 1999, Small et al. 2008). Net daily metabolism (NDM), the net production or consumption of energy, is calculated as the difference between these two functions (Odum 1956, Bott 2006). Biomass accrual through autotrophic and heterotrophic production drives the uptake rate of nutrients, and previous studies have seen strong relationships in particular between GPP and nitrate uptake rates (Tank and Webster 1998, Hall and Tank 2003, Stelzer et al. 2003, Meyer et al. 2005).

Streams in urban catchments typically receive higher inputs of nutrients and light relative to nearby unimpacted stream ecosystems (Hatt et al. 2004, Walsh et al. 2005). Collectively these increased inputs could be expected to generate higher GPP and autotrophic respiration and a heightened capacity for ecosystem nutrient uptake (Bott et al. 1985, 2006, Bunn et al. 1999, Young and Huryn 1999; Fig. 1b). However, previous studies suggest a more complex relationship between urbanization and nutrient uptake functions, for instance urban stream nutrient uptake experiments have tended to measure reduced nitrogen uptake efficiencies in urban streams (Grimm et al. 2005, Meyer et al. 2005, Mulholland et al. 2008). The reduced nutrient removal efficiency of urban streams, despite increased nutrient loading and light availability, could be due to other impacts associated with urbanization. Increases in impervious surface cover in the watershed (ISC) leads to stormflows of increased magnitude and frequency (Dunne and Leopold 1978, Walsh et al. 2005), which may result in frequent bed-scouring disturbances that reduce algal biomass and productivity (Uehlinger et al. 2002, Atkinson et al. 2008) and decrease the retention of particulate organic matter and associated heterotrophic respiration (Meyer et al. 2005). The competing effects of increases in resource supply and increasingly frequent disturbance make it difficult to predict the direction and magnitude of changes in stream ecosystem function as a result of urbanization.

Urban stream restoration

Stream restoration “based on the principles of the Rosgen geomorphic channel design approach is most commonly accomplished by restoring the dimension, pattern, and profile of a disturbed river system by emulating the natural, stable river” (Rosgen 2007). Because these approaches seek to increase habitat heterogeneity and to reconnect the stream channel with its floodplain by reshaping the channel, successful restoration projects should reduce stormflow disturbances and promote organic matter retention. Because restoration is conducted at the reach scale, it does not alter watershed loading of nutrients, thus all effects of restoration on water quality must be due to alterations in reach scale ecosystem functions.

Although most urban stream restoration focuses on reach-scale habitat complexity, the causative relationship between habitat complexity and ecosystem function is poorly researched. There are many links in our conceptual model on the path between habitat restoration and nutrient uptake (Fig. 1b). Several previous studies have reported an increase in nutrient uptake rates as a result of channel restoration activities (Bukaveckas 2007, Roberts et al. 2007, Kaushal et al. 2008), with the authors concluding that enhanced nutrient uptake following stream restoration was due to increased transient storage, stimulation of metabolism, and hydrologic reconnection of riparian soils

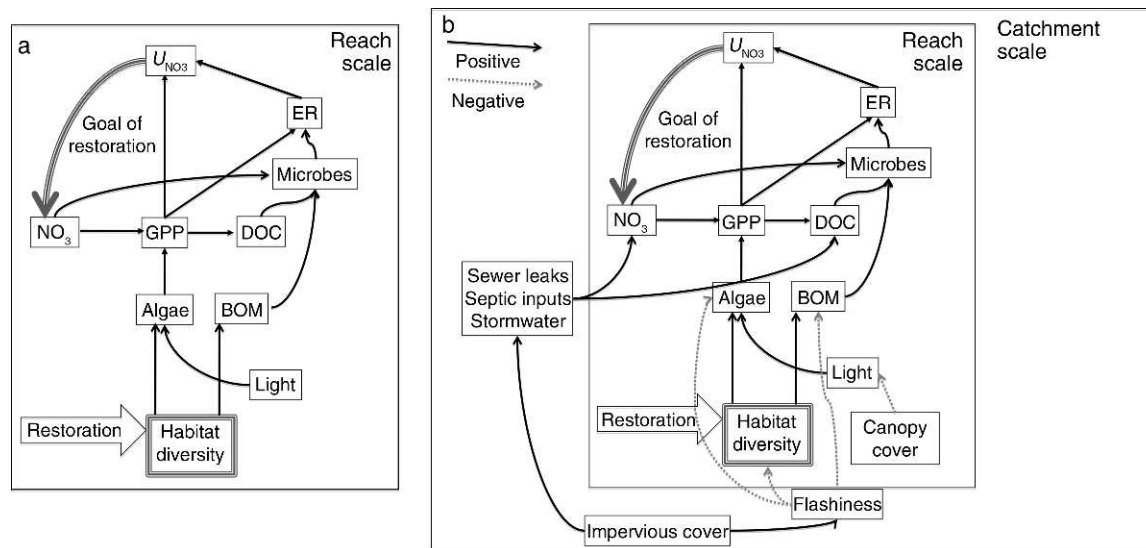


FIG. 1. Hypothetical causal model of stream ecosystem function (after Hall et al. 2009). Black arrows indicate that a variable has a positive effect, while gray dashed arrows indicate a negative effect. (a) Reach-scale controls on gross primary productivity (GPP), ecosystem respiration (ER), and nitrate uptake, indicating the management practice (habitat diversity) and the goal (water quality improvement by nitrate concentration reduction) of many stream restoration projects. (b) Catchment-scale effects of urbanization and their proposed reach-scale effects on ecosystem function. Variables are U_{NO_3} , nitrate uptake areal rate; DOC, dissolved oxygen content; BOM, benthic organic matter.

respectively. If the goal of restoration is water quality improvement, restoration would need to increase nutrient uptake capacity to levels above those seen in forested or unrestored urban streams, in order to maximize nutrient removal and improve water quality.

Study objectives

The overall objective of this study was to examine the effects of urbanization on stream hydrologic, metabolic, and nutrient uptake functions in order to (1) describe the alterations of these critical ecosystem functions by urbanization and (2) determine the extent to which urban stream restoration projects are able to mitigate these impacts.

We examined hydrology, water quality, heat, light, and the ecosystem functions GPP, ER, NDM, and nitrate uptake in four forested, four urban, and four urban restored streams of the North Carolina Piedmont. We expected that streams in urban catchments would have reduced canopy cover and higher nutrient loading, which would lead to higher GPP in urban streams. Predictions for ER are more difficult. We expected reduced supply and storage of allochthonous organic carbon and increased photorespiration and standing stocks of autochthonous carbon in urban streams. Thus, while we anticipated that the sources of carbon to stream microbes would differ between forested and urban streams, we could not formulate an a priori prediction for a change in the magnitude of ER. Nitrate uptake has been found to be strongly correlated with GPP (Hall and Tank 2003, Hall et al. 2009) and thus we

anticipated that urban streams would have higher nitrate uptake both as a function of heightened autotrophic metabolism and as a function of higher nitrate availability. Because restoration aims to enhance channel complexity and dampen storm flows, we expected reduced storm scour in restored urban streams, meaning they would store greater quantities of allochthonous carbon and have less variable rates of GPP than their unrestored counterparts. They would thus show enhanced metabolic activity and nitrate uptake relative to either forested streams or unrestored urban streams.

METHODS

Study sites

This study was designed with four blocks of sites (each block containing one forested, one urban unrestored, and one urban restored stream reach) with similar watershed size and ecoregion, and similar watershed percent impervious surface cover (ISC) between the urban unrestored (hereafter urban) and urban restored streams to control as much as possible for those variables (Table 1). We used this blocking strategy to stage our fieldwork, collecting the same data from all three streams in a block without any intervening storms. The study area spans three different ecoregions and many sites drain two different ecoregions (Table 1). This has several implications for this study. The soil types in the different regions are of different ages and origins, yielding differences in pH and available cations and anions (Griffith et al. 2002). Both the Carolina Slate Belt and Triassic Basin are characterized by low baseflow in

TABLE 1. Study sites, listed by stream type and ranked in order by percent impervious surface cover (ISC) in the watershed.

Block	Type	Site name	Site code	Watershed size (km ²)	Developed (%)	ISC (%)	Undeveloped riparian area in watershed (%)	Ecoregion
3	forested	tributary to Mud Creek	MC-t	0.9	4.4	0.5	86.6	Carolina Slate Belt/Triassic Basin
1	forested	Stony Creek	ST	6.9	24.4	3.4	70.9	Carolina Slate Belt
4	forested	Mud Creek (lower reach)	MC-L	4.1	58.6	9.5	41.2	Carolina Slate Belt/Triassic Basin
2	forested	Pott's Branch	PB	4.2	27.4	9.9	71.3	Northern Outer Piedmont
4	urban	Mud Creek (upper reach)	MC-U	3.5	66.9	11	30.8	Carolina Slate Belt/Triassic Basin
2	urban	Cemetery Creek	CM	2.2	98	19.1	5	Northern Outer Piedmont
1	urban	Ellerbe Creek	EC	7.6	88.7	20.8	10	Carolina Slate Belt/Triassic Basin
3	urban	Goose Creek	GC	1.7	100	39.4	0	Triassic Basin
4	restored	Sandy Creek	SA	6.7	76.5	16.8	21.8	Triassic Basin
2	restored	tributary to Walnut Creek	WC-t	1.7	84.5	17.8	14.7	Northern Outer Piedmont
1	restored	Third Fork Creek	3F	4.4	99.5	32.4	0.3	Triassic Basin
3	restored	Rocky Branch	RB	1.5	99.2	34.8	0.4	Northern Outer Piedmont

Note: Blocking is based on watershed size and ecoregion and is used in statistical analysis.

streams due to low yield of water from rocks in the Slate Belt and low permeability of clays in the Triassic Basin (Griffith et al. 2002). The Northern Outer Piedmont does not suffer from the same low baseflows, and so watersheds of comparable sizes but located in different ecoregions may produce streams of different sizes. Low baseflow and the tendency of streams to dry in summer have effects on biota; the Triassic Basin in particular is known for its comparatively low benthic macroinvertebrate richness (North Carolina Department of Environment and Natural Resources 2003).

We acquired a 1/3 arc-minute (10 m) digital elevation model for Durham, Orange, and Wake counties in North Carolina from the U.S. Geological Survey (USGS) Seamless Server and performed analysis using the ArcHydro extension of ArcGISv9.2 (ESRI, Redlands, California, USA) to calculate flow direction and flow accumulation, and define streams based on a 1000-pixel threshold and delineate watersheds for all sites. Land use and ISC within those watersheds as well as within the riparian area of the streams were analyzed based on 2001 National Land Cover Dataset (NLCD), and the accompanying ISC data set from the USGS Seamless Server. NLCD was reclassified into four categories—developed, agriculture, undeveloped, and water—and then the percentage of each land use type and ISC for each watershed was calculated. Riparian area land use was determined using the same techniques by buffering 30 m around each stream segment in the watershed.

The North Carolina Ecosystem Enhancement Project (EEP) provided the locations and basic information on all of their completed and planned stream restoration projects. We also accessed the National River Restoration Science Synthesis (NRRSS) database for stream restoration projects within our proposed study area and consulted with local stream restoration experts

(database *available online*).⁵ From these potential sites, we chose four stream restoration sites. Each had been restored using natural channel design approaches (Rosgen 1994, 1996), the dominant stream restoration approach in North Carolina (Sudduth et al. 2007). The Abbot restoration project was constructed in 1999 on a tributary to Walnut Creek that drains a 1.7-km² suburban (84.5% developed, 17.8% ISC) watershed in Raleigh, North Carolina. The goal of this project was “to restore the stream to the stable dimension pattern and profile for a C4 stream type as classified using Rosgen’s stream classification methodology (Rosgen 1996)...This type of restoration will reestablish the channel on a previous floodplain, or in this case, the basin of the old pond. Appropriate channel dimensions (width and depth), pattern (sinuosity, belt width, riffle-pool spacing), and profile (bed slope) of the new channel will be determined from reference reaches” (North Carolina Department of Transportation 1999). The Rocky Branch restoration project was constructed in 2001 on a stream draining a 1.5-km² watershed heavily impacted by the campus of North Carolina State University in Raleigh, North Carolina (99.2% developed, 34.8% ISC). The Rocky Branch restoration project goals included “Restore a stable self-maintaining morphologic pattern in the stream channel; stabilize streambanks using vegetation; create and improve habitat for fish and aquatic invertebrates; improve the quality of stormwater entering the creek through restoring and enhancing riparian buffers and establishing stormwater control within the creek’s watershed; provide safe and enjoyable access to the stream and passage through the campus by completing the greenway path adjacent to the creek” (Doll 2003).

⁵ (<http://nrrss.nbii.gov/>)

The Sandy Creek restoration project was constructed in 2004 on a stream draining a 6.7-km² watershed dominated by the campus of Duke University in Durham, North Carolina (76.9% developed, 16.8% ISC). The stated goals of the Sandy Creek project were to “Re-contour and restore more than 600 meters of degraded stream to hydrologically reconnect the stream with the adjacent floodplain to improve biogeochemical transformations and stream water quality” (Richardson and Pahl 2005).

The Third Fork Creek restoration project was constructed in 2005 on a stream that drains a highly urbanized 4.4-km² watershed (99.5% developed, 32.4% ISC). The planned goals of this restoration were to “restore stable channel morphology that is capable of moving the flows and sediment provided by its watershed; reduce sediment-related water quality impacts resulting from lateral bank erosion and bed degradation; improve aquatic habitat diversity through the reestablishment of riffle–pool bed variability and the use of in-stream structures; restore vegetative riparian buffers utilizing native plant species; and, improve natural aesthetics in an urban park setting” (KCI Associates 2003).

Although the ideal comparison would be with pre-restoration data from the same sites, no measures of ecosystem function were taken at these sites prior to restoration. Instead, after selection, each urban restored stream reach was paired with reaches draining two similarly sized watersheds, one urban reach without stream restoration and one non-urban forested reach. All four urban stream reaches were located in municipal parks or protected areas of the Duke Forest (a university-owned, ~7000-ha, research forest) to facilitate access and to accurately represent the “pre-restoration” condition, since most urban stream restoration projects in North Carolina take place on public land (BenDor et al. 2009). Indeed, two of our urban stream sites were chosen because the North Carolina EEP listed them as high-priority sites for future stream restoration efforts. Goose Creek drains a highly urban 1.7-km² watershed including parts of downtown Durham (100% developed, 39.4% ISC). Ellerbe Creek drains a high-density residential 7.6-km² watershed (88.7% developed, 20.8% ISC). Mud Creek (upper reach) drains a more recently developed (ca. 1980s) 3.5-km² watershed of housing subdivisions and apartment complexes (67% developed, 11% ISC). Cemetery Creek drains a 2.2-km² watershed with an older, high-density urban neighborhood in Raleigh, North Carolina (98% developed, 19.1% ISC).

Locating non-urban, forested watersheds of sufficient size to support perennial streams represents a significant challenge in the rapidly urbanizing landscape of the North Carolina Piedmont. We found one watershed in the area that was fully forested, a tributary to Mud Creek that drains a 0.9-km² watershed completely within the boundaries of the Duke Forest. With the exception

of Mud tributary we chose watersheds that had some level of watershed impacts in preference to watersheds that were significantly spatially removed from our urban and restored streams. Stony Creek drains a 6.9-km² watershed that is almost completely contained within a parcel of the Duke Forest near Hillsborough, North Carolina, but which has a residential area and a non-operational factory in a small portion of the upper watershed (3.4% ISC). Potts Branch is located in Umstead State Park near Raleigh, North Carolina, and drains a watershed that is primarily protected forest but is transversed by a major highway and associated development in the upper watershed (9.9% ISC). Our study site on Lower Mud Creek was located ~1.2 km downstream of the urban site on Mud Creek, thus capturing suburban impacts in its upper reaches (9.5% ISC). Because of the land use history of the North Carolina Piedmont, these are not pristine reference sites, but rather post-agriculture reforested streams with primarily forested watersheds. There may be legacy effects of prior agricultural land use on geomorphology, vegetation, and sediment (Maloney et al. 2008). There are no unmanaged forest remnants in the area of sufficient size to have a permanent stream; instead our “reference” streams are representative of the pre-urban landscape. Further information about the physical characteristics of all of these sites can be found in Violin et al. (2011).

Field methods: hydrology

Standard methods were used to analyze stream channel shape and habitat types (see Violin et al. 2011). Water level and temperature were continuously monitored beginning in January 2007 using either flow meters (Teledyne; ISCO, Lincoln, Nebraska, USA) or pressure transducers (Solinst Canada, Georgetown, Ontario, Canada) installed within each reach. Discharge was estimated from level based on surveyed channel dimensions using HEC-RAS (U.S. Army Corps of Engineers; software *available online*).⁶ Flashiness was estimated from changes in mean hourly discharge over a period encompassing all sampling days in a season using the following formula:

$$\text{Flashiness} = \frac{\sum_{i=1}^n |q_i - q_{i-1}|}{\sum_{i=1}^n q_i}$$

where q_i is mean hourly discharge and q_{i-1} is mean hourly discharge from the previous hour (modified from Baker et al. 2004). We then divided this flashiness index by watershed area to correct for the effects of watershed size on storm timing.

⁶ <http://www.hec.usace.army.mil/software/hecras/>

In the summer of 2006, we performed short-term whole stream conservative tracer additions using rhodamine dye (FWT Red 50 Liquid; Kingscote Chemicals, Miamisburg, Ohio, USA) to measure travel times and to estimate channel transient storage volume (Runkel 1998). A probe equipped with a rhodamine sensor (YSI Optical Monitoring System; YSI, Yellow Springs, Ohio, USA) was used to record rhodamine concentrations as the tracer was added, allowed to plateau, and then shut off and rhodamine concentration allowed to return to baseflow. Hydrodynamic properties (e.g., transient storage) of each experimental reach were estimated using a one-dimensional transport with inflow and storage model, OTIS-P (Runkel 1998). The storage zone area (A_s , in m^2), cross-sectional area (A in m^2), dispersion coefficient (D in m^2/sec), storage zone exchange coefficient (α in per second), and storage zone first-order decay coefficient (λ_s , in s^{-1}) were parameters modeled within OTIS-P for all 12 streams. Because rhodamine has the potential to adsorb to sediment surfaces (especially in the hyporheic zone) and photodegrade (R. Runkel, *personal communication*), the mass of dye recovered was determined for each reach. We used rhodamine rather than a salt tracer in these streams because we have found highly variable spatiotemporal patterns of Cl^- and Br^- in our urban streams that frustrated our initial attempts to model transient storage. We also measured average stream width on the sampling day and calculated specific discharge (Q/w in m^2/sec).

Water sample collection and analysis

Streamwater samples were collected from each stream reach on the day of ecosystem function sampling and at least once per month from June 2006 to June 2007. Samples were filtered in the field through Whatman GF/F syringe tip filters and returned to the lab on ice. All samples were stored at $4^\circ C$ and analyzed for NO_3^- , Br^- , PO_4^{3-} , SO_4^{2-} , and Cl^- with an ion chromatograph equipped with an AS4A anion column and KOH eluent generator (Dionex, Sunnyvale, California, USA). Total nitrogen and dissolved organic carbon were measured on a total organic carbon analyzer with a total nitrogen module (Shimadzu, Kyoto, Japan). Ammonium analysis was performed using the OPA fluorometric technique (Holmes et al. 1999) using a field fluorometer (10-AU, Turner Designs, Sunnyvale, California, USA). Dissolved organic nitrogen (DON) was calculated by subtracting NO_3^- -N and NH_4^+ -N from TN.

Energy inputs

Canopy cover was estimated in mid-summer using a spherical densitometer, measuring in four directions at five randomly chosen points in the thalweg of each stream; canopy cover was used as a proxy for a more direct measure of solar energy input such as photosynthetically active radiation. Degree-days were calculated from daily maximum and minimum temperature

data using the double triangle method (Roltsch et al. 1999), both for the individual sampling days for metabolism and nutrient uptake (to estimate short term ecosystem effects of temperature) and for continuous levellogger data records for windows of time around the sampling (to estimate longer-term effects of temperature). Organic matter was collected in summer for three blocks of study sites (nine sites total). We quantified habitat types as riffle, run, or pool and stratified sampling by type to obtain a representative sample of the reach (further details in Violin et al. 2011). Transects were taken at ten points along the stream reach, randomly chosen and stratified by habitat type. At each selected transect, surface organic matter sampling involved collecting by hand all visible coarse organic matter on the streambed across a 1-m wide transect. At sites with mostly fine sediments, benthic core samples for organic matter were taken using five 5-cm diameter cores evenly distributed across each transect. At sites where this was not possible because of large particle sizes, we took two 5-cm diameter cores and one 30-cm diameter core. In each case, we inserted the corers as far as possible into the streambed, and recorded the depth sampled. Samples were returned to the lab where surface samples were dried in an oven at $60^\circ C$ then weighed. A subsample of each core was ashed for 3 hours at $500^\circ C$ to determine percent ash free dry mass (AFDM). Benthic core samples were elutriated and preserved in 10% formalin for macroinvertebrate removal; following picking, samples were dried, weighed, and ashed for AFDM. Data were then normalized by total streambed surface area sampled for analysis.

Primary production and ecosystem metabolism

Algal biomass was sampled in summer at the same transects as organic matter sampling. At four points spread at even distances across each transect, algae were sampled by either scraping a 2.5-cm circle off a rock or removing the top 5-mm of sediment in a 2.5-cm circle. Samples from each transect were combined and analyzed for chlorophyll *a* with a fluorometer (10AU, Turner Designs, Sunnyvale, California, USA) using acetone extraction (Arar and Collins 1997). In one block the reference stream had gone dry due to summer drought so algae were not sampled (eight sites sampled total).

Gross primary production, ecosystem respiration, and net daily metabolism (NDM) were estimated once in summer and once in winter in each stream using the single station oxygen method (Odum 1956, Bott 2006). This method estimates metabolism for a presumed homogenous downstream reach whose length is determined by water velocity and reaeration rate. The small streams in this study were homogenous for an extensive distance downstream, well encompassing the reach for which metabolism was estimated. Estimates were made at baseflow on days without precipitation and at least 2 days following a storm event. Estimates within each

block were made on consecutive days without storms between them when possible. Although daily production and metabolism tend to be highly variable even at a single forested site (Roberts et al. 2007), logistics and equipment limitation meant that this study used only single-day estimates, in keeping with methods used in other multiple-site studies (e.g., Mulholland et al. 2001, 2008).

Dissolved oxygen and temperature were monitored every 10 minutes using a sonde (model 600XLM; YSI, Yellow Springs, Ohio, USA). These numbers were corrected for reaeration using propane releases (Marzolf et al. 1994, Young and Huryn 1998). At some sites the propane release failed or produced unrealistic numbers and at those sites reaeration estimation was repeated at similar baseflow conditions using sulfur hexafluoride (SF₆; Cole and Caraco 1998). Propane and SF₆ concentrations were measured on a gas chromatograph equipped with an ECD (Shimadzu, Kyoto, Japan). Ecosystem metabolism rates were determined from the changes in DO concentration every 10 minutes using the following equation:

$$\Delta DO = GPP - ER + E$$

where ΔDO is the change in DO concentration (g O₂/m³), GPP is gross primary production (g O₂/m³), ER is ecosystem respiration (g O₂/m³) and E is net exchange of oxygen with the atmosphere (g O₂/m³). E is the product of the reaeration coefficient and the average DO deficit over the 10-minute sampling interval. GPP is zero at night, when all metabolism flux is considered to be ER. During the day, ER was interpolated from metabolism flux for the hour before dawn and the hour after dusk. GPP for daytime intervals was the difference between net metabolism flux and interpolated ER for the interval. Daily rates of GPP and ER (g·m⁻²·d⁻¹) were calculated by summing all interval calculations of GPP or ER and dividing by the mean water depth observed on the sampling day.

Nutrient uptake

Nutrient uptake was measured through a series of short-term nutrient enrichment experiments using standard approaches (Newbold et al. 1981). Stream reach length for nutrient release was determined as 60-minute travel time based on a dye slug to estimate stream velocity in summer 2006. For consistency, the same reach length was used in winter 2007. In summer 2006, we used a modification of methodology used in enrichment experiments to estimate NO₃⁻ uptake by using the regression approach for estimating ambient uptake lengths (Bernhardt 2002, Thomas et al. 2003, Payn et al. 2005). In this approach, enrichment experiments use progressive levels of enrichment. For each enrichment level, uptake length is calculated as the inverse of the slope of the regression relating tracer corrected nutrient concentration to distance downstream of the addition point. Ambient (non-enrichment) uptake lengths are

estimated as the y-intercept of the curve relating estimated uptake lengths to the level of nutrient enrichment. To successfully employ the method it is necessary to calculate at least three uptake lengths. In our 12-stream survey, we encountered difficulties in employing the regression approach in five of 12 streams where demand was rapidly saturated and uptake lengths could only be calculated for one or two of the three levels of enrichment. Because of this difficulty we used data from the lowest level enrichment experiments only in all subsequent analyses (target enrichment ~25 µg/L NO₃⁻-N). In winter 2007 releases, we performed NO₃⁻ enrichment experiments only at single low-level enrichments (target enrichment ~25 µg/L NO₃⁻-N).

Briefly, to estimate the relative demand for inorganic N across our 12 streams, we measured uptake length of nutrients in each stream from which we calculated nutrient uptake velocities using established equations (Stream Solute Workshop 1990). All injections added NO₃⁻ as NaNO₃ and included a co-injected hydrologic tracer (chloride [NaCl] or bromide [NaBr]) in order to distinguish uptake from dilution and dispersion (Stream Solute Workshop 1990). Average stream wetted width was measured at ten sites within each reach. Initial estimates of discharge for estimating target addition rate and solution concentration were determined from flow meter estimates across the channel cross section. We used the tracer concentrations at plateau to accurately calculate Q for nutrient uptake calculations. Water sample collection and analysis was performed as previously described.

The nutrient uptake areal rate (U_{NO_3}) and nutrient uptake velocity (V_{NO_3}) were calculated for each nutrient in each stream using the established method of nutrient enrichment releases (Newbold et al. 1981, Stream Solute Workshop 1990). Nutrient uptake lengths were calculated as the inverse slope of the line describing the decline in the nutrient concentrations relative to the tracer over distance downstream (Newbold et al. 1981):

$$\ln N_x = \ln N_0 - k_x$$

where N_x is the concentration at the sampling site, N_0 is the concentration at the injection site, and k_x is the per meter uptake site. The uptake length (S_w), the average distance a molecule of NO₃⁻ or NH₄⁺ travels before it is removed from the water column, is calculated as k^{-1} . Uptake rate per square meter is calculated as

$$U = \frac{QN_0}{S_w w}$$

where Q is discharge and w is average stream width. Uptake velocity (V_f), the mass transfer coefficient (Stream Solute Workshop 1990), is calculated as

$$V_f = \frac{Q}{w S_w}$$

This calculation is especially useful in comparative studies like this one as it allows comparison across

TABLE 2. Physical properties of study sites.

Site	Type	Canopy cover† (%)	Reach length (m)	Median discharge‡ (L/s)	Summer total degree-days§	Summer flashiness§	Winter total degree-days¶	Winter flashiness¶	As/(As + A)	F_{med}^{200}
MC-t	forested	87.94	54	5	11 018	0.04	5127	0.04	0.484	48.37
ST	forested	87.10	100	6	10 691	0.01	4990	0.01	0.416	41.65
MC-L	forested	93.66	100	8	11 418	0.01	4740	0.02	0.234	21.52
PB	forested	81.46	140	12	11 020	0.04	4778	0.10	0.297	28.53
MC-U	urban	92.23	140	5	11 450	0.26	4583	0.07	0.429	42.91
CM	urban	71.79	80	13	11 470	0.14	5988	0.15	0.438	43.70
EC	urban	77.69	50	14	12 167	0.09	5131	0.08	0.168	14.37
GC	urban	83.67	50	9	12 899	0.17	5942	0.06	0.334	33.21
SA	restored	48.60	60	3	12 137	0.10	4633	0.10	0.229	22.91
WC-t	restored	64.74	200	4	12 164	0.06	5003	0.07	0.311	30.98
3F	restored	32.61	80	11	nd#	nd	5197	0.30	0.472	47.11
RB	restored	68.88	50	15	12 604	0.37	6074	0.22	0.222	19.52

† Summer 2006.

‡ January–July 2007.

§ Data for 5 May–1 June 2007 (data not available for 2006).

¶ Data for 27 February–19 March 2007.

Data loss due to instrument failure.

streams of different sizes and discharges (Hall et al. 2002). This enrichment technique assumes that the gross uptake rate and the nutrient demand increase at the same rate for the duration of the enrichment and no asymptote in uptake capacity is reached in the course of the release. Uptake velocities are a measure of nutrient uptake rate relative to nutrient concentration in the water column normalized for physical differences in depth and velocity among streams (Stream Solute Workshop 1990, Davis and Minshall 1999, Hall et al. 2002).

Statistical methods

Data throughout this study were analyzed using a blocked one-way ANOVA by site type (forested, urban, restored), with post-hoc Student's *t* tests to look for significant differences among types using JMP version 7.0.2 (SAS Institute, Cary, North Carolina, USA). Ecosystem functions were also analyzed for correlations with the structural variables hypothesized to be their predictors. Multiple regressions were used to examine the predictability of functional variables based on the hypothesized causal variables. Forward stepwise multiple regressions (alpha to enter = 0.25, alpha to exit = 0.1) were run based on our conceptual model of ecosystem functions (Fig. 1b). Predictive variables in models were evaluated for multicollinearity using a correlation matrix, and the best model was chosen based on Akaike information criteria (AIC). Adjusted R^2 and AIC provide a measure of model fit while penalizing the addition of more model parameters.

RESULTS

Hydrology

Although we saw many differences in channel shape between forested streams and urban and urban restored streams (Violin et al. 2011), and in watershed variables

commonly considered to affect flashiness, and despite a wide range of flashiness across our stream reaches (0.04–0.3), our flashiness index showed no significant difference among stream types (Table 2). Contrary to expectations, there were no significant differences among stream types in transient storage zone volume (Table 2). Flashiness was significantly correlated with ISC for both summer ($R^2 = 0.37$, $P = 0.0287$) and winter ($R^2 = 0.34$, $P = 0.0263$). We found no strong correlations between specific discharge (Q/w) or transient storage zone volume and any watershed land use characteristics.

Water quality

Monthly water quality sampling revealed generally higher nitrogen concentrations in urban streams than was seen at forested sites (Table 3). Mean monthly nitrate concentrations in the forested streams ranged from 0.13 to 0.38 mg/L NO_3^- -N while in the urban streams it ranged between 0.15 and 1.53 mg/L NO_3^- -N. DOC was not significantly different between stream types, ranging between 2.07 and 7.54 mg/L across all streams and dates.

Heat and light

At the watershed scale, the proportion of undeveloped land in the riparian buffer was significantly lower for the restored and urban streams than for their forested counterparts ($P = 0.001$, Table 1). However, direct canopy cover over the study reaches did not differ between urban and forested streams, but was significantly lower in restored reaches due to clearing for stream restoration construction ($P = 0.01$; Table 2). Across all stream types in summer 2007 (5 May–1 June, matching the timing of 2006 intensive sampling efforts), total degree-days were significantly higher in restored and urban streams than in forested streams ($P = 0.0175$; Table 2). Summer total degree-days were strongly correlated with ISC in the watershed ($R^2 = 0.83$, $P <$

TABLE 3. Average of monthly sampled ambient nutrient concentrations \pm SE for one year (June 2006–June 2007).

Stream	Class	ISC (%)	NO ₃ ⁻ -N (mg/L)	NH ₄ ⁺ -N (mg/L)	DON (mg/L)	DOC (mg/L)	Cl ⁻ (mg/L)
MC-t	forested	0.5	0.18 \pm 0.06	0.17 \pm 0.09	0.03 \pm 0.05	5.91 \pm 0.88	10.59 \pm 1.54
ST	forested	3.4	0.38 \pm 0.20	0.05 \pm 0.04	0.28 \pm 0.08	4.49 \pm 0.52	8.09 \pm 0.74
MC-L	forested	9.5	0.14 \pm 0.03	0.12 \pm 0.07	0.1 \pm 0.05	5.4 \pm 0.90	7.39 \pm 1.41
PB	forested	9.9	0.13 \pm 0.05	0.07 \pm 0.06	0.06 \pm 0.02	5.1 \pm 0.49	7.37 \pm 1.52
MC-U	urban	11	0.15 \pm 0.03	0.07 \pm 0.04	0.24 \pm 0.03	6.11 \pm 0.88	9.95 \pm 1.35
CM	urban	19.1	1.53 \pm 0.14	0.09 \pm 0.06	0.16 \pm 0.11	2.07 \pm 0.24	17.95 \pm 1.50
EC	urban	20.8	0.25 \pm 0.05	0.08 \pm 0.03	0.34 \pm 0.04	7.54 \pm 0.55	13.27 \pm 1.94
GC	urban	39.4	0.48 \pm 0.12	0.14 \pm 0.05	0.47 \pm 0.03	7.27 \pm 0.49	18.78 \pm 2.11
SA	restored	16.8	0.19 \pm 0.03	0.06 \pm 0.04	0.29 \pm 0.04	6.56 \pm 0.38	15.23 \pm 2.33
WC-t	restored	17.8	0.24 \pm 0.04	0.06 \pm 0.03	0.16 \pm 0.03	3.21 \pm 0.39	7.94 \pm 0.52
3F	restored	32.4	0.52 \pm 0.10	0.12 \pm 0.05	0.61 \pm 0.10	6.83 \pm 0.61	18.44 \pm 2.47
RB	restored	34.8	0.50 \pm 0.09	0.11 \pm 0.06	0.27 \pm 0.06	2.75 \pm 0.44	10.60 \pm 1.84

0.0001). For the time period surrounding sampling in winter 2007 (27 February–19 March), there were no significant differences in total degree-days among stream types, although again total winter degree-days were significantly positively correlated with ISC ($R^2 = 0.39$, $P = 0.02$). For temperature data taken only from the days of nutrient uptake and metabolism measurements, there were no significant differences in degree-days in summer; however, in winter, restored streams had significantly higher degree-days for the sampling day than either forested or urban streams ($P < 0.0001$, Appendix A). Stream temperatures (in degree-days) on the days of intensive sampling were significantly correlated with percent canopy cover for winter dates ($R^2 = 0.28$, $P = 0.04$) but not for summer dates.

Organic matter

Benthic organic matter standing stocks in these streams ranged between 10 and 3039 g AFDM/m² surface BOM, 88 and 1252 g AFDM/m² buried coarse benthic organic matter (CBOM), and 464 and 3832 g AFDM/m² buried fine benthic organic matter (FBOM). There were no significant differences in the total amount of surface BOM or in buried CBOM among stream types (Fig. 2). However, restored streams had significantly higher standing stocks of FBOM buried in the sediments ($P = 0.022$; Fig. 2). Normalizing the buried BOM numbers by sample volume rather than surface area sampled revealed no other significant differences in organic matter distribution (data not shown). Average water column depth was a significant positive predictor of CBOM ($R^2 = 0.54$, $P = 0.015$), as was, surprisingly, summer flashiness ($R^2 = 0.74$, $P = 0.0038$). We found no other strong correlates of benthic organic matter amongst either watershed or reach scale descriptive variables.

Primary production

Summer GPP ranged from 0.16 g O₂·m⁻²·d⁻¹ to 7.63 g O₂·m⁻²·d⁻¹. Winter GPP ranged from 0.02 g O₂·m⁻²·d⁻¹ to 3.3 g O₂·m⁻²·d⁻¹. There were no significant differences in GPP among stream types (Fig. 3). The best fit stepwise multiple regression model found that stream

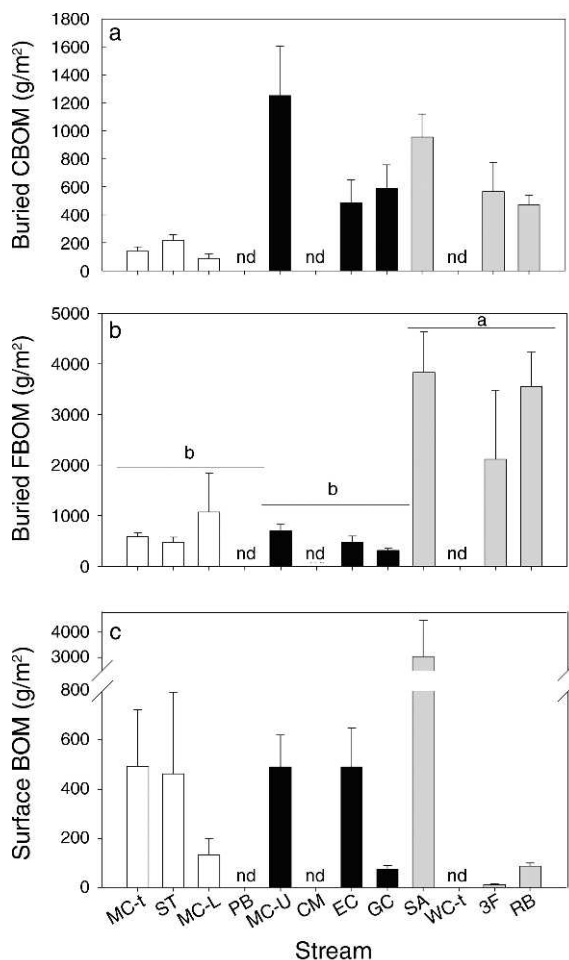


FIG. 2. Benthic organic matter ash free dry mass for 12 streams. (a) Buried coarse particulate organic matter mass (CBOM; >1 mm), (b) buried fine organic matter mass (FBOM; <1 mm), and (c) total mass of benthic organic matter on the surface of the streambed (BOM). White bars represent forested sites, black bars represent urban sites, and gray bars represent restored sites; nd indicates data not collected. Error bars represent standard error. Different lowercase letters indicate that restored streams as a group have significantly higher buried fine benthic organic matter than forested or urban streams ($P = 0.022$). Site codes are explained in Table 1.

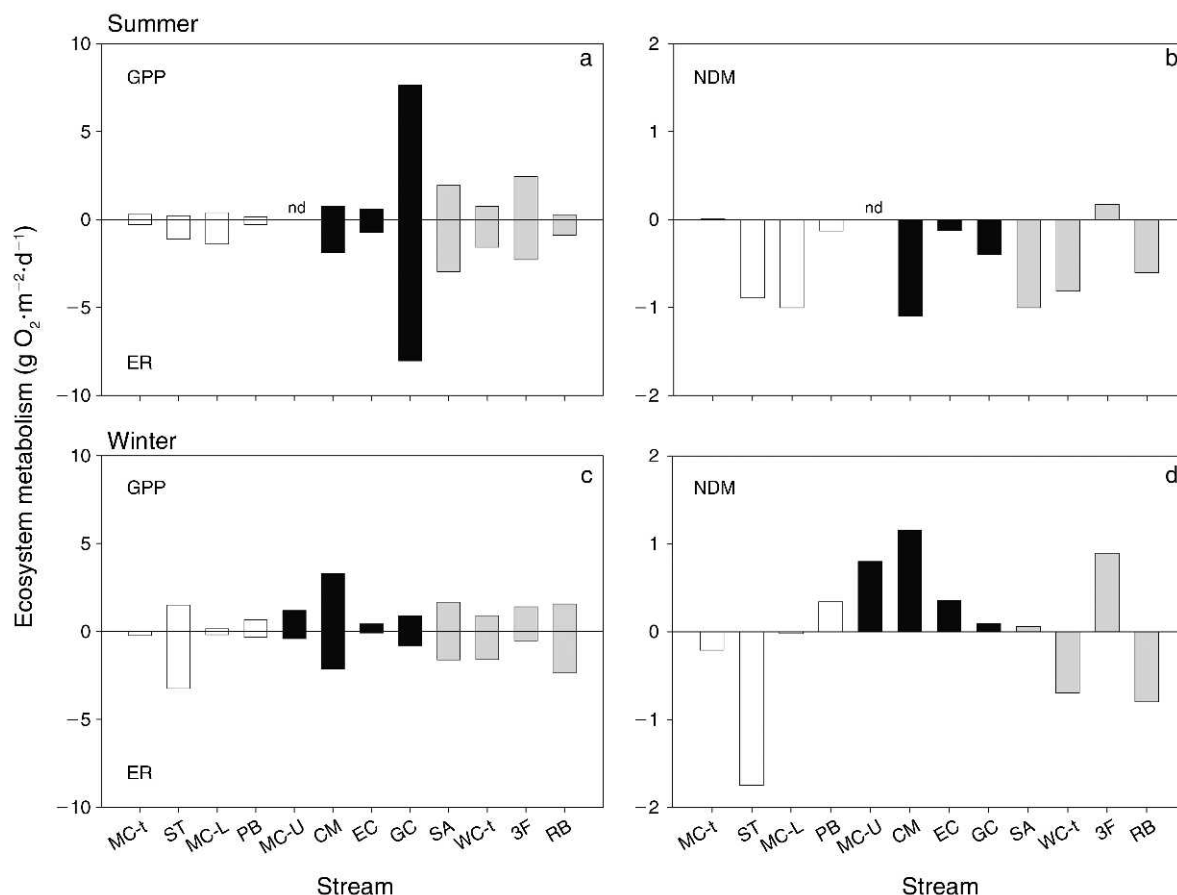


FIG. 3. Ecosystem metabolism for 12 streams: (a) GPP and ER in summer 2006, (b) net daily metabolism (NDM) in summer 2006, (c) GPP and ER in winter 2007, (d) NDM in winter 2007. There are no significant differences among stream types. White bars represent forested sites, black bars represent urban sites, and gray bars represent restored sites; nd indicates data lost due to instrument failure.

temperature (as degree-days on the day of sampling) and ISC explained 45% of the variation in summer GPP ($P = 0.037$, Table 4). Summer GPP was positively correlated with ISC ($R^2 = 0.36$; $P = 0.03$) but including the negative correlation with stream temperature improved model fit. Winter GPP was not significantly correlated with ISC. For the winter data set, the best stepwise multiple regression model found that temperature (as degree-days on the day of sampling) and stream NO_3^- together explained 73% of the variation in winter GPP ($P = 0.001$, Table 4). Both factors were positively correlated with winter GPP.

Ecosystem respiration and net daily metabolism

Summer ER ranged from -8.02 to -0.28 g $\text{O}_2\cdot\text{m}^{-2}\cdot\text{d}^{-1}$. Winter ER ranged from -3.26 to -0.08 g $\text{O}_2\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ (Fig. 3). There were no significant differences in ER between site types in either season. Summer ER was significantly correlated with ISC ($R^2 = 0.39$, $P = 0.041$), although that regression was strongly driven by our most highly urbanized stream, Goose Creek. There

was no correlation between ISC and winter ER. Stepwise multiple regression modeling found that GPP together with specific discharge (Q/w) explained 96% of the variation in summer ER ($P < 0.0001$, Table 4) but this relationship was strongly affected by Goose Creek, which overshadowed differences among sites (e.g., Goose ER was -8.02 g $\text{O}_2\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ while the average of other sites was -1.34). After removing Goose Creek from the data set the best explanatory model was still GPP and Q/w , which together explained 72% of the variation in summer ER ($P = 0.005$, Table 4). For the winter data set, DOC alone was positively correlated with winter ER ($R^2 = 0.35$; $P = 0.03$); but the best fit multiple regression model incorporated a negative correlation with GPP along with the positive correlation with DOC, collectively explaining 44% of the variation in winter ER ($P = 0.03$, Table 4).

Summer NDM ranged from -1.09 to 0.16 g $\text{O}_2\cdot\text{m}^{-2}\cdot\text{d}^{-1}$. Winter NDM ranged from -1.75 to 1.16 g $\text{O}_2\cdot\text{m}^{-2}\cdot\text{d}^{-1}$ (Fig. 3). There were no significant differences in NDM between stream types in either season.

TABLE 4. Comparison of multiple regression models for predicting ecosystem function variables.

Variable explained	Parameters tested†	Parameters kept	Prediction equation
Summer GPP	degree-days of sampling day, % canopy cover, [NO ₃ -N], Q/w , ISC	degree-days of sampling day, ISC	$3.44 - 0.01(\text{degree-days of sampling day}) + 0.16 \times \text{ISC}$ $0.68 + 0.11 \times \text{ISC}$
Winter GPP	[NO ₃ -N], degree-day of sampling day, flashiness, habitat transitions	[NO ₃ -N], degree-days of sampling day	$-0.18 + 1.33 \times [\text{NO}_3\text{-N}] + 0.004(\text{degree-days of sampling day})$ $0.67 + 1.17 \times [\text{NO}_3\text{-N}]$
Summer ER	degree-days of sampling day, [DOC], Q/w , ISC, GPP	GPP, Q/w	$-0.26 - 0.98 \times \text{GPP} - 0.12 \times Q/w$
Summer ER (outlier excluded)	degree-days of sampling day, [DOC], Q/w , ISC, GPP	GPP, Q/w	$0.58 - 0.97 \times \text{GPP}$ $-0.31 - 0.9 \times \text{GPP} - 0.13 \times Q/w$
Winter ER	[DOC], flashiness, habitat transitions, GPP	GPP, [DOC], GPP	$-0.62 - 0.91 \times \text{GPP}$ $-1.51 + 0.27 \times [\text{DOC}] - 0.52 \times \text{GPP}$
Summer V_{NO_3}	%canopy cover, degree-days of sampling day, [NO ₃ -N], GPP, Q/w	GPP, [DOC] % canopy cover, degree-days of sampling day, % canopy cover	$-0.23 + 0.78 \times \text{GPP}$ $-2.66 + 0.43 \times [\text{DOC}]$ $14.92 - 0.11 \times \% \text{canopy cover} - 0.01 \times \text{degree-days}$
Winter V_{NO_3}	% canopy cover, [NO ₃ -N], degree-day for sampling day, GPP, Q/w	[NO ₃ -N], Q/w	$6.41 - 0.06 \times \% \text{canopy cover}$ $-6.65 + 12.02 \times [\text{NO}_3\text{-N}] + 2.95 \times Q/w$

Note: Q/w is specific discharge.

† All explanatory variables from the same season as ecosystem function variables.

Overall NDM was more positive in winter, indicating greater autotrophy in winter at most sites, however this seasonal shift was only significant for the urban stream category. There was no correlation between ISC and summer or winter NDM. We found no single or multiple predictor models that could explain a significant amount of the variation for either summer or winter NDM.

Nitrogen cycling

Nitrate uptake lengths varied between 19 and 270 m in the summer and between 6 and 161 m in the winter and were not significantly different among stream types (data not shown). Estimated nitrate uptake rates (U_{NO_3}) across sites and seasons ranged from 0.01 to 75 mg $\text{NO}_3^- \cdot \text{N} \cdot \text{m}^{-2} \cdot \text{minute}^{-1}$, while NO_3^- uptake velocities (V_{NO_3}) ranged from 0.1 to 38 mm/minute (Fig. 4). Both the lowest and highest values for both metrics were measured during our winter sampling (Fig. 4). Summer U_{NO_3} was significantly higher for urban restored stream reaches than for forested streams but neither was significantly different from urban degraded streams ($P = 0.03$, Fig. 4). V_{NO_3} showed the same pattern of significant differences ($P = 0.003$, Fig. 4). In the winter there were no significant differences in either U_{NO_3} or V_{NO_3} among stream types; instead, we had two streams that had extremely high rates of NO_3^- uptake while there was little variation among the other 10 streams. We measured the highest winter NO_3^- uptake in one of our forested streams and one of our urban degraded streams.

Canopy cover was the best single predictor of summer V_{NO_3} ($R^2 = 0.32$; $P = 0.035$) but adding temperature (as the degree-days on the day of sampling) dramatically

improved explanatory power, with these two variables collectively explaining 71% of the variation in summer V_{NO_3} ($P = 0.003$, Table 4). FBOM was a significant predictor for U_{NO_3} in both summer ($R^2 = 0.59$, $P = 0.0091$) and winter ($R^2 = 0.75$, $P = 0.002$), but this relationship was driven by the outliers and limited by sample size ($n = 9$ for BOM). There was no single predictor that explained a significant proportion of winter V_{NO_3} , but together specific discharge (Q/w) and streamwater NO_3^- concentration explained 49% of the variation in V_{NO_3} ($P = 0.0189$, Table 4).

DISCUSSION

Contrary to expectations, we found no significant functional differences between forested and urban stream categories, although ISC was significantly correlated with both ER and GPP in the summer. We found that nitrate uptake velocities were higher in restored stream reaches during our summer sampling period. Otherwise, restored streams were not functionally different from forested or urban streams in any of the variables that we measured. This is in contrast with the results we found for physical habitat in the companion research, in which forested streams were significantly shallower, had more habitat transitions, and larger mean sediment size than urban streams; restoration of urban streams did not alter mean depth, number of habitat transitions, or mean sediment size, or in any way make physical structure significantly different from the unrestored urban streams (Appendix B; Violin et al. 2011). We found that the best predictors of summer nitrate uptake were canopy cover and stream temperature, both of which were significantly altered by

TABLE 4. Extended.

df	<i>F</i>	Adjusted <i>R</i> ²	<i>P</i>	AIC
2, 8	5.1261	0.4521	0.0369	12.7801
1, 9	6.6292	0.3437	0.03	14.0822
2, 9	15.5482	0.7257	0.0012	-16.2253
1, 10	15.9467	0.5761	0.0025	-11.7381
2, 8	124.4232	0.9611	<0.0001	-16.5337
1, 9	197.6383	0.9516	<0.0001	-14.6307
2, 7	12.8181	0.7242	0.0022	-15.1445
1, 8	18.1648	0.656	0.0028	-14.0513
2, 9	5.3241	0.4402	0.0298	-3.4174
1, 10	7.4463	0.3695	0.0212	-2.7268
1, 10	6.979	0.3521	0.0247	-2.4009
2, 8	13.1451	0.7084	0.003	2.4049
1, 10	6.1182	0.3175	0.0329	8.1776
2, 9	6.3648	0.4938	0.0189	56.2418

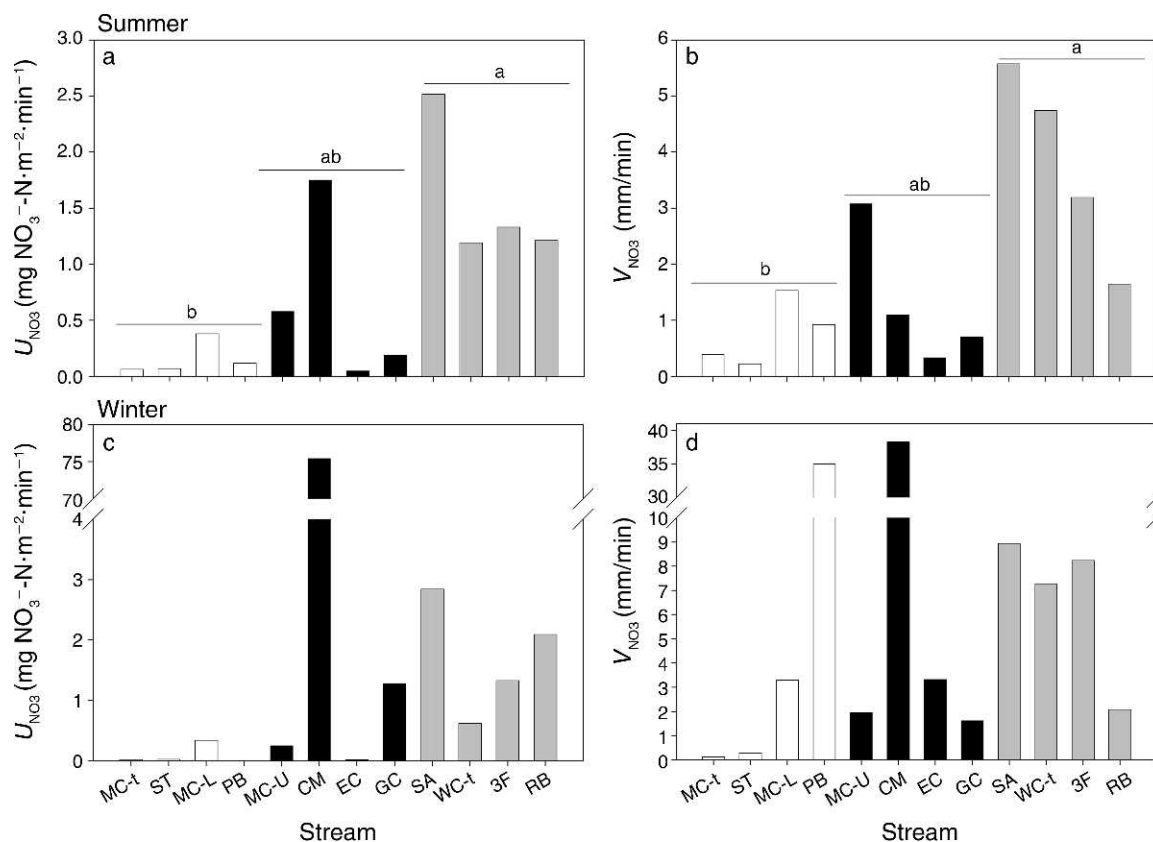


FIG. 4. Nitrate uptake measurement for 12 streams: (a) U_{NO_3} for summer 2006, (b) nutrient uptake velocity (V_{NO_3}) for summer 2006, (c) U_{NO_3} for winter 2007, and (d) V_{NO_3} for winter 2007. White bars represent forested sites, black bars represent urban sites, and gray bars represent restored sites. Letters indicate that summer restored streams as a group have significantly higher areal uptake rates ($P = 0.03$) and uptake velocities ($P = 0.003$) than forested streams, while urban streams are indistinguishable from both.

restoration activities. Restoration also led to an increase in streambed FBOM, which has previously been shown to be a strong correlate of nitrate uptake (Meyer et al. 2005). We hypothesize that the major ecosystem impact of natural channel design restoration projects in these systems has been an increase in light and heat, which, while it may stimulate nutrient uptake, is likely to also have severe consequences for sensitive biota (Violin et al. 2011). The significant effect of restoration on nutrient uptake was only seen in summer, not winter, suggesting that it is primarily an effect of light and heat, and that the alteration of the physical shape of the channel by restoration was not the driver of this difference in biogeochemical cycling.

One of the most interesting findings of this study was the significant positive relationship between ISC and streamwater temperatures. Stream temperature (measured as degree-days) was significantly correlated with all ecosystem functions in the summer (GPP, ER, and nitrate uptake). This suggests that the urban heat island (Kalnay and Cai 2003) has important implications for the functioning of aquatic ecosystems as well as their surrounding catchments, an aspect of the heat island problem that has received little attention to date. Natural channel design restoration projects typically remove the local riparian vegetation to ease construction, which will exacerbate the thermal pollution of urban watersheds; this could potentially further degrade stream habitat, or at least constrain biological recovery of sensitive taxa. Our results suggest that this should be an area of concern for restoration practitioners. In reconsidering our conceptual model (Fig. 1b), we need to add consideration of both the catchment-scale heat island effects of urbanization and the reach-scale effects of loss of canopy cover on stream temperature.

As previously seen by Meyer et al. (2005) and Bernhardt and Palmer (2007), we expected urbanization to decrease BOM due to reduced inputs and retention, limiting the fuel available for heterotrophic respiration. We discovered that in these catchments urbanization did not decrease BOM when compared with forested streams, and that restoration may indeed increase buried FBOM. The significant positive correlation of buried CBOM standing stocks with both water depth and hydrologic flashiness suggests that altered urban hydrology may be leading to increased burial and storage of CBOM in the sediments, leading to the very high levels of CBOM seen at some of the urban sites in this study. There was, however, no correlation between BOM and ER among our sites. Instead, increased DOC and ISC in urban watersheds seemed to be the strongest drivers of differences in ER. ER was also significantly correlated with GPP. It seems that in our more urban streams the energy budget is altered with autochthonous OM and allochthonous DOC (perhaps from leaking infrastructure) serving as the primary source of carbon, rather than allochthonous BOM. As in Hall and Tank (2003), we found increased nutrient uptake with increased GPP.

However, the relationship was only significant in the winter and was not related to urbanization factors. We detected a positive relationship between BOM and nutrient uptake as seen in other studies (Mulholland et al. 1985, Webster et al. 2000, Meyer et al. 2005), but this correlation was skewed by outliers and our ability to draw conclusions is also limited by lack of statistical power ($n = 9$ replicates for BOM in our study).

Based on enhanced carbon and nutrient supply together with higher heat and light, we expected to see higher rates of GPP and ER in urban streams and expected that these factors would collectively lead to higher nutrient uptake. However, in the extreme example of Goose Creek in the summer, a very warm urban stream with high nutrient and organic matter loading leading to high GPP and ER, we saw no proportional increase in nitrate uptake. This suggests that our conceptual model (Fig. 1) failed to consider some other impacts of urbanization that could be limiting nutrient uptake in this system. We hypothesize that urban contaminants or frequent stormwater disturbance may suppress microbial activity in streambed sediments. The conceptual understanding of the effects of urbanization on stream ecosystem functioning needs to be revised to better take into account the many possible impacts of peak flow conditions and urban contaminants on stream ecosystems.

Effects and impacts of restoration

Although there are no clear significant differences in ecosystem function between forested and urban streams, there were some significant differences between the restored sites and either forested or urban sites. Some of these appear to be due to the impacts of restoration, including loss of canopy cover and further simplification of substrate and channel structure (Violin et al. 2011). It is not uncommon to see lingering detrimental effects of a variety of stream restoration practices (Biggs et al. 1998, Bednarek 2001, Colangelo and Jones 2005, McKie et al. 2006). The restoration projects in this study are still quite new (<10 years old); as planted riparian trees grow, the canopy cover will presumably return at least to levels seen in the unrestored urban sites. There is no evidence, however, that the channel will gain in complexity with time, so we expect simplified channel structure to be a lasting impact of stream restoration at these sites.

There were surprisingly few significant effects of urbanization or restoration on hydrologic variables measured in this study. The lack of differences among stream types in transient storage seems particularly surprising, given the significant differences in channel shape between forested and urban streams (Appendix B; Violin et al. 2011). This is likely due to the inability of transient storage estimates with OTIS-P to distinguish between hyporheic storage and in-channel storage (Runkel 1998). It is likely that the forested streams have more hyporheic storage while the deeper, slower urban streams have more in-channel storage. Restoration likely

had no effect on this, since the restored streams were similarly deep and slow, and likely also had predominantly in-channel storage. A stated goal of these projects, as with most natural channel design projects, was reconnection of the stream with a floodplain. In addition to somewhat mitigating the physical effects of peak stormflows, more contact with the floodplain could lead to higher uptake rates of nutrients during stormflow. Because of logistics and safety, as well as the steady-state assumptions of the nutrient tracer method, we were unable to measure uptake during stormflow at these sites. However, since the urban restored streams retain their flashy urban hydrograph, as restoration did nothing to address watershed scale hydrologic alteration, any effect of floodplain reconnection would be relatively brief for any given storm.

The one clear significant effect of restoration on ecosystem function that was seen in this study was that restored sites had significantly higher U_{NO_3} and V_{NO_3} at baseflow than their forested counterparts in the summer. Unlike Bukaveckas (2007), we found no effect of restoration on transient storage (Table 2) or water depth or velocity (Appendix B; Violin et al. 2011) and thus we cannot conclude that higher nitrate uptake in restored streams was due to channel alterations of restoration. Instead, nitrate uptake rates seem to be correlated both with higher FBOM at these sites and with increased temperatures due to loss of canopy cover. Uptake rates are generally correlated with temperature (Butturini et al. 2000). Bukaveckas (2007) also saw significantly higher temperatures in a restored stream (presumably due to loss of canopy cover) when compared to both channelized and reference streams, which may call into question the study's conclusion that restoration of channel structure was the sole cause of increased nutrient uptake in the restored stream. Kaushal et al. (2008), saw significantly higher denitrification rates within streambanks of restored reaches with low banks vs. restored and unrestored reaches with high banks, a change they attribute to greater hydrologic connectivity with the floodplain. Although they saw no change in groundwater temperature, altered streambank height could also affect channel shading and thus affect streamwater and sediment temperatures, which were not measured in that study. Because high temperatures can exacerbate dissolved oxygen problems and exclude organisms with lower thermal tolerance (Jones et al. 2006), we caution that using stream temperature manipulation to facilitate nitrate retention is not an effective restoration strategy.

Previous work has shown FBOM to be a strong predictor of nutrient uptake in both forested (Mulholland et al. 1985, Webster et al. 2000) and urban streams (Meyer et al. 2005). It is not clear why the restored sites in this study had higher FBOM than the other sites. One possibility is that the loss of canopy cover is significantly correlated with and likely drives higher algal biomass observed in the restored streams,

which over time would lead to higher detrital levels of FBOM. Roberts et al. (2007) saw increased nutrient uptake following restoration by coarse woody debris addition, which also led to increased BOM storage in the restored channels. Future restoration projects with the goal of increasing nitrate uptake to improve water quality could use innovative methods, such as coarse woody debris addition, to increase FBOM to fuel nutrient removal. However, as was seen in the Roberts et al. (2007) study, without management of altered stormwater peak flows, newly retained BOM can quickly be buried or scoured and the benefit of this type of restoration lost.

Field of Dreams hypothesis

The original goal of this study was to test the Field of Dreams hypothesis, answering the question: Is restoration of stream physical habitat sufficient to restore ecosystem function? We were unable to answer this question for several reasons. First, as shown in Violin et al. (2011), the stream restoration projects examined in this study had no significant effect on stream depth, habitat, or particle size when compared to urban unrestored stream habitat. In addition, in this study, there were no clear significant differences in ecosystem function between urban unrestored and forested streams as groups and thus the question of "restoring" ecosystem function may not be a real one, or at least not as straightforward as we had expected. We saw a clear correlation between ISC and GPP across all stream types, suggesting that urbanization does have quantifiable effects of ecosystem functions. However, these effects are gradual and cumulative, and thus better suited to a regression approach than the ANOVA approach of stream types that was the primary focus of this study. Finally, we saw that restoration did have effects on nitrate uptake in the summer, but likely due to loss of stream canopy cover, an unintentional impact of stream restoration, rather than direct beneficial effects of habitat restoration. Although, ultimately, our study is not a true test of the Field of Dreams hypothesis, it raises questions about the effectiveness of natural channel design stream restoration as it was performed at these sites, as well as questions about the assumptions behind stream restoration in general in terms of the trade-offs between habitat restoration and enhancing ecosystem functions like nutrient uptake.

CONCLUSIONS

In a study of 12 streams in the North Carolina Piedmont, we found surprisingly few significant functional differences between forested, urban, and urban restored streams. We found that ecosystem functions (GPP, ER, and nitrate uptake) were primarily controlled by temperature, which was significantly correlated with ISC. There was also evidence to suggest that natural channel design stream restoration is causing a significant increase in stream temperature through the removal of

riparian vegetation. Although this increase in temperature was strongly correlated with nitrate uptake, which could be a benefit of restoration, the extreme alteration of the thermal regime is likely to have severe consequences on the biotic community, which could inhibit successful biological restoration. In contrast to previous studies, which documented lower standing stocks of benthic organic matter in urban streams, we saw no such reduction. However, autochthonous production and dissolved organic carbon, rather than allochthonously derived benthic organic matter, seem to be the primary drivers of ecosystem respiration, suggesting a fundamental shift in the base of the food web in urban streams. Our study suggests a clear need for a reconsideration of urban streams that better includes the effects of heat due to loss of riparian vegetation at the reach scale and the watershed scale effects of the urban heat island. It also clearly demonstrates fundamental flaws in natural channel design stream restoration projects as practiced at these sites, both because removal of riparian vegetation exacerbates the urban heat effect, and because implementation of habitat-focused stream restoration with a goal of water quality improvement neglects to use all available knowledge of the underlying processes of stream ecosystem functions.

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APPENDIX A

Chemical and physical characteristics for summer 2006 and winter 2007 sampling days (*Ecological Archives* A021-089-A1).

APPENDIX B

Physical data for study sites collected during summer 2006 (*Ecological Archives* A021-089-A2).

Assessing stream restoration effectiveness at reducing nitrogen export to downstream waters

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Abstract. The degradation of headwater streams is common in urbanized coastal areas, and the role these streams play in contributing to downstream pollution is a concern among natural resource managers and policy makers. Thus, many urban stream restoration efforts are increasingly focused on reducing the downstream flux of pollutants. In regions that suffer from coastal eutrophication, it is unclear whether stream restoration does in fact reduce nitrogen (N) flux to downstream waters and, if so, by how much and at what cost. In this paper, we evaluate whether stream restoration implemented to improve water quality of urban and suburban streams in the Chesapeake Bay region, USA, is effective at reducing the export of N in stream flow to downstream waters. We assessed the effectiveness of restored streams positioned in the upland vs. lowland regions of Coastal Plain watershed during both average and stormflow conditions. We found that, during periods of low discharge, lowland streams that receive minor N inputs from groundwater or bank seepage reduced in-stream N fluxes. Furthermore, lowland streams with the highest N concentrations and lowest discharge were the most effective. During periods of high flow, only those restoration projects that converted lowland streams to stream–wetland complexes seemed to be effective at reducing N fluxes, presumably because the design promoted the spillover of stream flow onto adjacent floodplains and wetlands. The observed N-removal rates were relatively high for stream ecosystems, and on the order of 5% of the inputs to the watershed. The dominant forms of N entering restored reaches varied during low and high flows, indicating that N uptake and retention were controlled by distinctive processes during different hydrological conditions. Therefore, in order for stream restoration to effectively reduce N fluxes exported to downstream waters, restoration design should include features that enhance the processing and retention of different forms of N, and for a wide range of flow conditions. The use of strategic designs that match the dominant attributes of a stream such as position in the watershed, influence of groundwater, dominant flow conditions, and N concentrations is crucial to assure the success of restoration.

Key words: *Chesapeake Bay; headwater streams; Maryland; nitrogen export; nitrogen management; restoration assessment; stream restoration; urban streams.*

INTRODUCTION

Human activities have impacted watersheds worldwide, such that countless rivers and streams are polluted, have highly altered flows, or are severely degraded geomorphically. Many countries are now making substantial investments to improve environmental conditions in these waterways, particularly with respect to the reduction of nutrients. A variety of strategies are being employed including the regulation of point-source pollutant discharge, implementation of land preservation programs, and improvements to stormwater management (Daniels and Lappins 2005, Taylor and

Fletcher 2007). Other strategies target point and non-point sources of pollution by reducing the application of excess fertilizer to agricultural land, restoring wetlands or by repairing leaky sewer systems (Verhoeven et al. 2006, Dietz and Clausen 2008). Projects are also being undertaken to improve water quality in streams by reconfiguring their channels, reconnecting floodplains or simply planting riparian vegetation (Spruill 2000, Roni et al. 2002, Rheinhardt et al. 2009).

These efforts all have in common the goal of improving water quality but they differ fundamentally in that some of them are designed to prevent pollutants from reaching tributaries (e.g., land-based best management practices [BMPs]) while others must reduce pollutants that are already in tributaries (e.g., channel restoration projects). Enhancing stormwater infrastructure or reforesting uplands is known to lead to positive

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hydrological changes that help cleanse polluted water before it reaches a stream and also reduce flood-induced channel disturbances that often degrade water quality (Walsh et al. 2005). Riparian reforestation has also been extensively studied, and while projects may not have immediate effects or work in all settings, over time they can reduce the movement of nutrients and sediments to streams and thus lead to improved water quality (Lowrance et al. 1985, Jordan et al. 1993, Meyer et al. 1998). There are also abundant data to show that, if properly implemented, agricultural BMPs can result in substantial reductions in the movement of nutrients and sediments to adjacent waterways (Staver and Brinsfield 1995, 2000). In contrast, data on the water quality benefits of restoration projects that involve in-stream manipulations such as altering channel form or adding in-stream structure are largely absent despite the frequency of the practice (Bernhardt et al. 2005). Stream restoration is a relatively young science and it has only been in the last several years that studies quantifying its ecological effectiveness are starting to be published, with most focusing on benthic invertebrates or geomorphic channel stability (e.g., Moerke et al. 2004, Smith and Prestegard 2005, Sudduth and Meyer 2006, Tullos et al. 2009).

Various channel design options to improve water quality have been suggested (Craig et al. 2008) and there have been a few short-term or small-scale measurements using experimental releases of nutrients or core-scale measurements of denitrification. The experimental releases are used to quantify nutrient "uptake rate," which is a measure of the rate at which dissolved nutrients decline as they move downstream from the release site. This decline could be due to utilization by microbes and algae, temporary storage in the sediments, permanent loss from the stream in the case of nitrogen (due to denitrification), or a combination of these three. Bukaveckas (2007) measured uptake rates in three streams in Kentucky (USA) when flows were low (April–June) and found small increases in uptake rates of nitrate and phosphate in restored vs. channelized stream reaches. Klocker et al. (2009) found no consistent pattern in various measures of nitrate uptake rates in two restored stream reaches in Maryland (USA) during the summer of 2006; however, Kaushal et al. (2008) found higher denitrification rates in the stream banks of one of these stream reaches compared to unrestored reaches. Roberts et al. (2007) added coarse woody debris to disturbed streams as a restoration "experiment" and showed that ammonium uptake rates were higher one month after the wood was added. To date, however, there have been no comprehensive studies that determine if in-stream restoration projects can reduce the net export of N to downstream waters, especially during periods of both high and low flows, when retention efficiency can vary dramatically (Shields et al. 2006, Lewis and Grimm 2007).

We evaluated the effectiveness of restored urban and suburban Coastal Plain streams at reducing the export of N to downstream waters over a three-year period and during high and low flows. For the different hydrological conditions, we examined fluxes of N in different forms, including nitrate (NO_3^-), ammonium (NH_4^+), and dissolved organic and particulate N (DON and PN, respectively) to explore possible mechanisms that control N export to downstream waters. We also quantified restoration effectiveness as a function of position of the restored reach in the watershed. We refer to these positions as upland vs. lowland because the former are positioned higher up in the watershed and the latter are positioned near the tidal boundary. While they differ only slightly in terms of channel grade and valley-floor width, these differences have important hydrologic and geomorphic implications. We tested the hypotheses that (1) restored streams are more effective at reducing N exports than non-restored degraded streams and (2) the N-removal capacity of a restored stream varies between upland and lowland reaches.

The study was conducted in streams in the Chesapeake Bay region, where excess nutrients exported from the watershed associated with an ever growing population and intensive agriculture have caused the bay to become increasingly eutrophic over the last several decades (Kemp et al. 2005, Fisher et al. 2006, Williams et al. 2010). Consequently, this region has one of the highest number of stream restorations (Hassett et al. 2005) and diversity of restoration approaches implemented, including restoration in upland and lowland reaches of the Coastal Plain region.

MATERIALS AND METHODS

Site descriptions

The study was conducted in eight streams located within a radius of 40 km in the Coastal Plain of western Maryland in Anne Arundel County (38°51' N, 76°32' W), which is one of the most urbanized regions in the United States, and where annual precipitation and temperature average 1130 mm and 13.2°C, respectively. The study streams drained dominantly urban and suburban catchments that ranged in size from about 0.4 to 1.4 km² (Table 1), and were serviced either by private septic systems, public sewage treatment facilities, or both. All stream reaches included in the study were in headwaters (first and second order), however, four reaches were positioned higher up in the watershed (upland reaches) and four positioned near the tidal zone (lowland reaches). Headwater streams make up the majority of the channels in the Coastal Plain drainage network and, despite relatively low gradients compared to streams in mountainous terrain, they can destabilize with increased runoff because they lack bedrock controls and are free to adjust with changes in flow. Degradation of stream reaches positioned higher up in the watershed (upland reaches) result in gully-like conditions, where channels lack adjacent floodplains and are characterized

TABLE 1. Names and geographic position (coordinates) of study streams, lengths of study reaches, and respective surface drainage area.

Stream name	Reach length (m)	Catchment area (ha)	GPS coordinates	
			Above study reach	Below study reach
Upland stream reaches				
Weems Creek, Bristol (BRI)	196	44	38.98576° N, 76.51951° W	38.98723° N, 76.51892° W
Weems Creek, Moreland (MOR)	173	32	38.98444° N, 76.52677° W	38.98570° N, 76.52726° W
Weems Creek, Mall (MAL)	489	36	38.99177° N, 76.54061° W	38.99203° N, 76.53613° W
Control-1	209	130	38.97401° N, 76.52145° W	38.97309° N, 76.51717° W
Lowland stream reaches				
Howard's Branch (HBR)	295	90	39.02053° N, 76.55075° W	39.02138° N, 76.54795° W
Wilelinor Stream Valley (WIL)	416	80	38.96797° N, 76.54473° W	38.96596° N, 76.54100° W
Spa Creek (SPA)	239	85	38.97401° N, 76.52145° W	38.97330° N, 76.51955° W
Control-2	214	137	38.97310° N, 76.51761° W	38.97311° N, 76.51650° W

Note: Codes in parentheses refer to the abbreviated names used in the text.

by tall banks subject to erosion. Stream reaches positioned near the tidal zone (lowland reaches) are commonly degraded by high sediment deposition from upstream associated with land use changes and channel erosion. These lowland reaches also have relatively wide valleys and gentle slopes.

Three of the upland stream reaches in our study (Bristol [BRI], Moreland [MOR], and Mall [MAL]; Table 1) were “geomorphically restored” using channel design methods (e.g., Gillilan 1996, Rosgen 1996, Federal Interagency Stream Restoration Working Group 1998) commonly employed to reduce streambed and bank erosion including some level of channel reconfiguration, bank armoring, boulder placement, and grade controls to increase hydraulic resistance (Fig. 1). In contrast, each restored lowland study reach was restored with a unique approach. One lowland reach (Spa Creek [SPA]) was restored by regrading its banks, planting riparian grasses, and placing small cobbles and stones along the stream bed, while the other two (Howard's Branch [HBR] and Wilelinor [WIL]) were restored with less-conventional methods that involved sculpting a combination of back-watered “step-pools” of varied sizes connected by small “riffles,” adding rock weirs in various places, and establishing vegetated floodplains. These two lowland streams have been transformed into what we call a stream-wetland complex, and also include ponds and shallow wetlands adjacent to the restored reaches (Fig. 1). One degraded upland and one degraded lowland stream (“controls”) were also monitored. The study reaches in the control (unrestored) streams were similar to the restored reaches in terms of order and length.

Study design

From January 2007 to January 2010, data were collected above, below, and in some cases, within the study reaches. For each sampling site, data were collected during periods of base flow (hereafter referred

to as periods of average flow), which we define as the periods when the effect of precipitation on stream volume was minimal and streamflow was relatively stable, and during periods of stormflow. During average flow conditions, water samples were collected and discharge measured above and below each study reach, in surface-flowing lateral inlets, and in groundwater and bank seepages along the reaches. The combination of water (and N) entering a reach upstream, and from lateral inlets or groundwater and bank seepage comprised the “inputs”; the water and N leaving a reach represented the “output.” The net export of N along each reach was subsequently estimated using an input-output budget approach, where the sum of inputs entering the reach was subtracted from the amount exported downstream.

During periods of storm flow, discharge and N concentration data were collected on an event basis, but only in restored streams that seemed effective at reducing N fluxes during average flow conditions (based on data from the preliminary phase of the study), and that were also effective at attenuating storm hydrograph peak flows (assessed by comparing peak flows above and below the restored reach). Nitrogen retention effectiveness of the monitored streams was estimated as the difference in fluxes between the two ends of the restored reach during a series of storm events.

Sample collection and analyses

During average flows, water samples were collected biweekly above and below each study reach, and also in any surface-flowing lateral inlets. Water samples were collected manually in 1-L pre-leached polyethylene bottles, and a portion of each sample was immediately filtered through pre-rinsed glass-fiber filters (GF/F, Whatman 0.45 μ m nominal pore size) to separate dissolved from particulate N. Filtered water samples were stored in prewashed, high-density, polyethylene bottles and kept on ice in the dark until returned to the

laboratory (within 6 hours) where they were frozen until analyzed for dissolved N. Particulate material in the remaining unfiltered samples was collected on pre-combusted (500°C for 1.5 hours) glass-fiber filters after pouring a measured volume of sample water (≥ 200 mL) into a filtration apparatus in increments of 50 mL and filtering with a vacuum pressure ≤ 25.4 cm Hg. When the filter was saturated, it was removed from the base of the apparatus using forceps, placed in labeled glassine envelopes to be frozen and stored until analyses could be performed. Groundwater and bank-seepage samples were collected during average flow conditions during nine different synoptic sampling events in streams where measurable increases in discharge along the study reach were observed. Groundwater samples were collected using mini-piezometers in the streambed over multiple days in the fall of 2009 and spring of 2010. Bank seepage samples were collected when visible in macro-pores, using a syringe to minimize disturbance. Both, groundwater and seepage samples were processed following the protocol used for grab samples, except that no PN samples were collected.

Stormflow samples were collected simultaneously above and below the restored reaches of two streams (HBR and WIL), during entire storm hydrographs. Samples were collected on an event basis during 2008, 2009, and spring of 2010 using automated pump samplers (ISCO 6712; Teledyne Isco, Lincoln, Nebraska, USA) configured to collect 24 or more samples during each event. Sample collection started as soon as stream stage increased 1.5 cm above average flow level and continued at 15–30-minute intervals for the duration that stream stage remained elevated. When a storm event lasted for more than 6 to 12 hours, ISCO bottles were replaced so sampling could continue throughout the falling hydrograph. During warm months, ISCOs were filled with ice until samples could be retrieved a few hours after the end of each storm event. After collection, all samples were stored in a dark and cool container during transport to the laboratory, where they were kept at 4°C. All stormflow samples were filtered within 24 to 48 hours for N analyses.

Nitrate plus nitrite (NO_2^-) concentrations were determined using the cadmium reduction method on a flow injection analyzer (Lachat QuikChem 8000; Hach Company, Loveland, Colorado, USA). Ammonium concentrations were determined using the Berthelot Reaction method (Kerouel and Aminot 1987). Total dissolved N (TDN) concentrations were determined using the persulfate digestion method (D'Elia et al. 1977), and the dissolved organic N fraction was calculated by subtracting N in dissolved inorganic forms ($\text{NO}_3^- + \text{NH}_4^+$) from TDN concentrations. Particulate N was measured with a Perkin Elmer 2400 CHN elemental analyzer (Perkin Elmer, Waltham, Massachusetts, USA). Total N (TN) was calculated as the sum of TDN and PN. All analyses were performed at the Analytical Laboratory of the Chesapeake Biological Laboratory.

Hydrological measurements

The cross-sectional area method (Gordon et al. 2004) was used to measure instantaneous discharge above and below each study reach, and in lateral inlets immediately after water samples were collected during average flow conditions. Discharge from groundwater and bank seepage was estimated as the difference of average discharge measured above and below each reach during average flow conditions. In streams where discharge was lower downstream than upstream, the groundwater and bank seepage input was assumed to be zero. The cross-sectional area method was also used to measure discharge during storm flow, and during successive stream stages in order to construct hydrologic rating curves for each sample site (Rantz 1982). Continuous stream stage height data were recorded at 5–15-minute intervals by automated stage loggers (Onset HOBO Water Level Logger; Onset Computer Corporation, Pocasset, Massachusetts, USA) installed in each stream. Errors associated with the instability of the relationship between stage height and discharge (e.g., due to aquatic vegetation, ice, debris) were evaluated by examining the maximum, minimum, average and standard deviation of each data set. Values outside the expected range were either discarded or substituted by an average of neighboring values.

DATA ANALYSES

Calculation of discharge

Hydrologic rating curves were used to generate continuous records of discharge for each sample site. The total discharge that occurred during a stormflow event was estimated as the sum of the discharge recorded at consistent intervals over a stormflow hydrograph. Total annual discharge was estimated by adding all the interval discharges recorded during a period of one year. We estimated total annual discharges for 2007 to 2010 and used the grand mean to calculate the annual N flux in each stream. The relative contribution of average flow to annual discharge was estimated using the sum of discharges recorded in one year that were less than or equal to the maximum average flow value measured during the year, whereas the relative contribution of storm flow to annual discharge was calculated as the sum of discharges greater than the maximum average flow value measured in a year.

Calculation of discharge-weighted mean concentrations and fluxes

Discharge-weighted mean concentrations of N in stream water were computed as

$$\text{DWMC} = (\Sigma C_i Q_i) / \Sigma Q_i \quad (1)$$

where C_i is the observed concentration of instantaneous stream flow i , Q_i is the discharge (L/s) for the interval i when water sample was collected, and the denominator is the sum (Σ) of discharge. To estimate the annual flux



FIG. 1. Stream restoration designs used in Anne Arundel County, Maryland, USA. The upper panels show restoration features such as (a) rip-raps and (b) cross vanes used in traditional restoration designs commonly implemented in upland reaches (e.g., BRI, MOR, and MAL). Traditional designs are based on a geomorphic template that refers to a least disturbed reference site (i.e., a semi-pristine stream) to reduce peak flow and control erosion. The lower panels show restoration features such as (c) shallow step pools and (d) riffle weirs, which are part of restoration designs implemented in lowland reaches such as HBR and WIL. Reaches are described in Table 1.

of N exported from each stream during average flow conditions, we first assumed that the discharge-weighted mean concentrations (DWMC) of the every-other-week samples represented concentrations during all periods of discharge equivalent to or below the maximum value observed during biweekly sampling. Similarly, we assumed that the DWMC of samples collected during storm flow represented concentrations during all high-flow periods. Total N flux (F_j) for each site was then calculated separately for high and average flows as

$$F_j = (\text{DWMC})(Q_j) \quad (2)$$

where DWMC is the discharge-weighted concentration for each study site, and Q_j is the total discharge during either high or average flows during the year. The sum of fluxes from high and average flows yielded the annual fluxes.

Assessment of restoration efficiency

The efficiency of stream restoration at reducing the net export of N downstream was determined by comparing N fluxes in and out of the study reaches during average flow conditions and during storm events. Net N export during average flow was calculated by subtracting the sum of N inputs to the reach (i.e., water from upstream, lateral inlets, bank seepage, and ground-

water) from the flux of N at the outflow. Fluxes from upstream, at the outflow, and from lateral inlets were calculated as the product of volume-weighted mean TN concentrations and average biweekly discharge. Groundwater and bank seepage fluxes were calculated as a product of average TDN concentrations of samples collected during synoptic sampling and the average discharge increase along each reach. During high-flow conditions, we evaluated the performance of restoration by calculating N retention/loss along the restored reaches. In addition, because stream degradation in urban catchments is associated with severe-flow-pattern alterations, our assessment of restoration efficiency during high-flow conditions also included the evaluation of storm hydrographs (peak flow characteristics and magnitude) during different storm events.

Uncertainty assessment

Uncertainties associated with the mass balances were estimated by determining the propagation of individual errors generated from the calculation of fluxes included in the input–output model. The error for each component of the mass balance was computed using the Tukey's jackknife procedure for mean concentrations and discharge. To obtain an approximate estimate of the overall error associated with fluxes, a propagation of the component errors was performed as described in

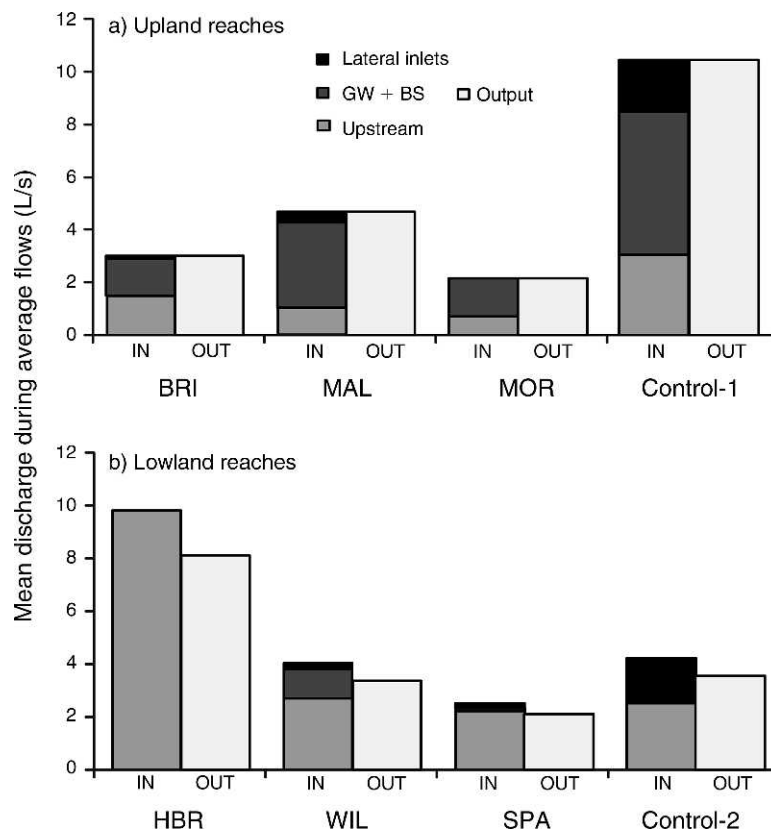


FIG. 2. Mean annual discharge of water entering (IN) and exiting (OUT) (a) upland reaches and (b) lowland reaches during average flow conditions. The different sources of water entering each reach (IN) are represented by shades of gray and include stream water upstream, groundwater + bank seepage (GW + BS), and lateral inlets. Output includes only stream water exiting the reach downstream.

Williams and Melack (1997), where the reduced term of the propagation equation is

$$S_T = (V^2 S_c^2 + C^2 S_v^2)^{1/2} \quad (3)$$

where V is the volume of water, C is the concentration of water, and S is the standard error with the subscripts of volume (v) concentration (c) and total flux (T). After calculating errors in N fluxes from upstream (UF), groundwater + bank seepage flow (GW + BS), lateral inlets flow (LF), and downstream (DF), the sum of errors (S_{total}) from these four components of the mass balance was estimated for each stream as follows:

$$S_{\text{total}} = (S_{\text{UF}}^2 + S_{\text{GW+BS}}^2 + S_{\text{LF}}^2 + S_{\text{DF}}^2)^{1/2}. \quad (4)$$

RESULTS

Discharge

During average flow conditions, discharge was relatively low in all streams, averaging 4.5 and 7.5 L/s above and below the study reaches, respectively, regardless of the stream position (Fig. 2). However, in upland channels, discharge was about two to four times higher below than above the study reach, while in

lowland channels, discharge was higher below the study reach only in the control stream (Control-2) and Wilelinor (WIL; Fig. 2). Despite the substantial discharge gains along upland channels, the contribution from lateral inlets was usually less than 5% of the outflow, indicating that discharge increased along these reaches due to gains from groundwater and bank seepage (Fig. 2).

In the control lowland stream, the observed increase in discharge was due to large inputs from a lateral inlet in the study reach, while at WIL, groundwater and bank seepage seemed to play an important role. However, it is important to mention that what we describe as groundwater and bank seepage at WIL is, in large part, water transferred through a berm from two large stormwater ponds located adjacent to the restored channel. The ponds are slightly elevated in relation to the channel and also constructed of sand and wood chips to allow a constant transfer of water to the stream.

During stormflow conditions, maximum peak discharge in the different streams ranged from 470 to more than 1500 L/s (Table 2). Lowland reaches that were "restored" to become stream-wetland complexes (HBR and WIL) had less flashy hydrographs during storms

TABLE 2. Estimated average annual discharge, maximum peak flow, and relative contribution of average and high flows to total annual discharge in each study stream.

Stream reach	Mean annual discharge (m ³ /yr)	Maximum peak flow (L/s)	Contribution to annual discharge (%)	
			Average flows	High flows
Upland stream reaches				
BRI	407 123	1800	20	80
MAL	420 834	1000	25	75
MOR	na	na	na	na
Control-1	347 712†	800	40	60
Lowland stream reached				
HBR	608 523	850	40	60
WIL	530 802	470	10	90
SPA	na	na	na	na
Control-2	765 350	1760	15	85

Note: The abbreviation “na” stands for “not available.”

† Data provided by Anne Arundel County, Maryland, USA.

than did restored upland reaches (Fig. 3), despite the fact that WIL drained a highly impervious catchment. Among the upland reaches, paradoxically, only the control (Control-1) had an attenuated hydrograph.

N concentrations and fluxes

Average flow conditions.—Discharge-weighted mean concentrations of TN entering the study reaches upstream varied between approximately 0.5 and 2.2 mg/L, regardless of the stream position in the watershed (Fig. 4). In both upland and lowland reaches, the highest concentrations were observed in the streams that received water directly from street drainage networks (BRI and SPA). As water moved downstream, discharge-weighted TN concentrations either decreased or increased in the different restored streams, but the largest decreases were observed in streams with relatively high concentrations upstream (Fig. 4).

Nitrate was the dominant form of N in stream water entering all lowland reaches, but it was the dominant form in only one upland reach (BRI; Fig. 4). In the remaining upland reaches, DON and NH₄⁺ were the main forms entering the channel upstream; these streams received water directly from retention ponds or storm-water wetlands. Nitrate was also the main form of N exported from most study reaches (outputs; Fig. 4), however, concentrations usually changed downstream. In upland channels, NO₃-N concentrations increased downstream, except for BRI, while in lowland channels concentrations either decreased or remained the same (Fig. 4).

Because TN fluxes are a product of TN concentration and discharge, patterns for fluxes were somewhat different than those for concentrations. In upland channels, TN fluxes below the restoration (outputs; Fig. 5) increased considerably in relation to fluxes above the restoration (bars labeled “upstream” as part of inputs; Fig. 5). In most cases, increases in TN fluxes in upland reaches corresponded to gains from lateral inlets and groundwater plus bank seepage (GW + BS; Fig. 5).

The gains from groundwater and bank seepage were relatively large in upland reaches even though their concentrations were often low in relation to those observed in water entering the channel above the study reach (Table 3). In two lowland channels (HBR and SPA, Fig. 5), TN fluxes below the restored reach decreased in relation to fluxes above. Both of these reaches had relatively small inputs from groundwater and bank seepage (Fig. 5).

High-flow conditions.—Our stormflow data summarize hundreds of discrete streamflow samples collected during 10 storm events at HBR and 12 events at WIL ranging in depth from 1 to 62 mm. At HBR, the discharge-weighted mean TN concentrations upstream of the restored reach were similar to those below the reach, except during larger storms (i.e., >18 mm), when concentrations were markedly lower downstream vs. upstream of the restoration project (Fig. 6). During these large storms, PN concentrations in streamflow above the restoration were markedly high in comparison to concentrations observed in smaller storms, which contributed to high TN concentrations. Consequently, there was a positive correlation between storm size and discharge-weighted mean TN concentrations above the HBR restored reach.

At WIL, PN concentrations in stormflow were not particularly high during storm events sampled (Fig. 7), and there was no significant correlation between storm depth and TN concentrations. Above the restored reach at WIL, discharge-weighted TN concentrations varied between about 0.78 and 1.5 mg/L and, apart from a couple of storm events, they were similar to concentrations downstream. However, the relative contribution of the different species of N changed, especially for NO₃⁻ and NH₄⁺. While NO₃⁻ concentrations were relatively high upstream of the restoration, they were low downstream. In contrast, NH₄⁺ concentrations were substantially lower upstream than downstream. When PN was the dominant form of N entering the restored reach at HBR, TN concentrations decreased substan-

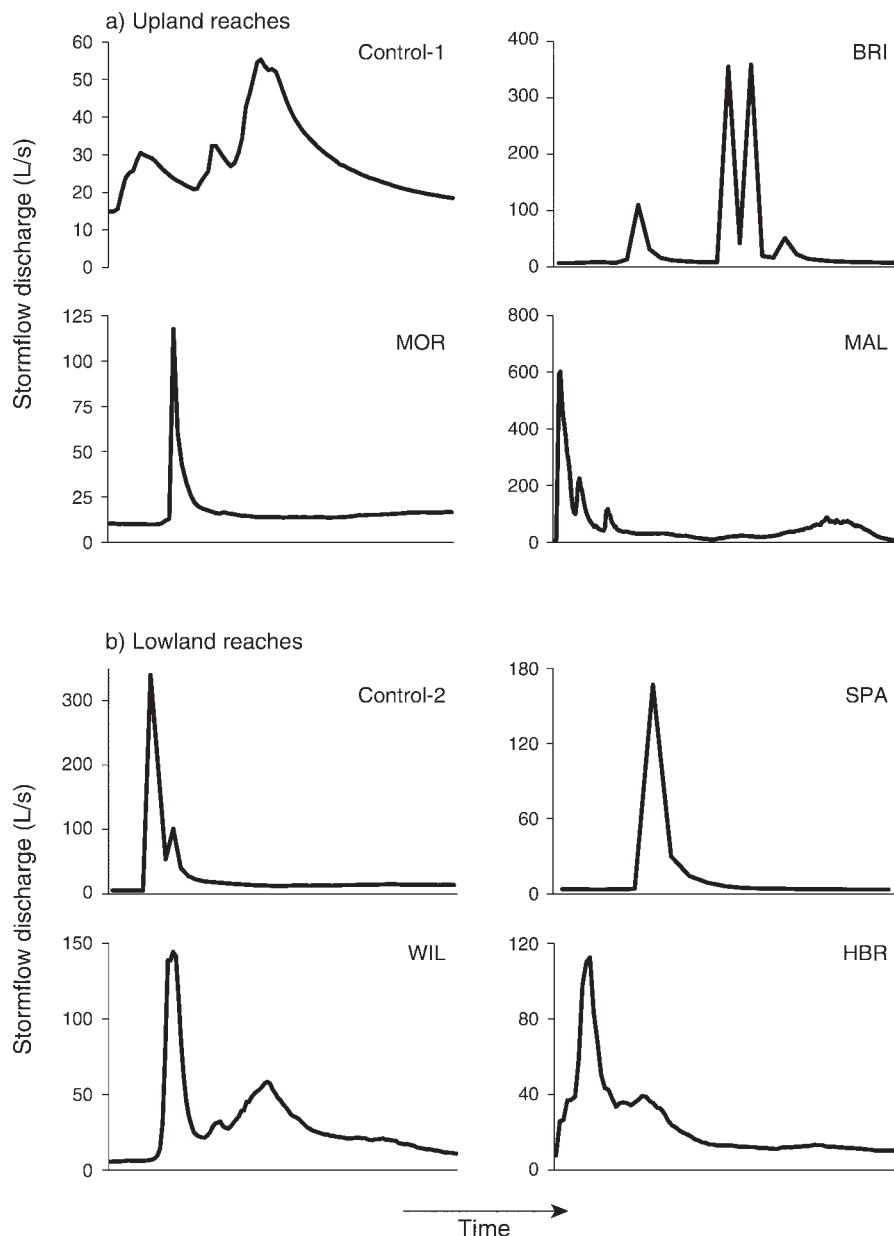


FIG. 3. Storm hydrographs illustrating peak-flow characteristics, and stormflow volume downstream of the restored reach. (a) The four upper plots represent upland reaches while (b) the four bottom plots represent lowland reaches.

tially downstream, suggesting that the efficiency with which restored reaches retained TN during storms increased with PN concentrations upstream (Fig. 8). In contrast, the efficiency of TN retention at WIL tended to decrease with increasing PN concentrations upstream (Fig. 8).

Evaluation of stream restoration effectiveness

Average flow conditions.—When the mass of N entering the study reaches (estimated as the sum of inputs from water entering the reach from upstream, lateral inlets, groundwater, and bank seepage) is subtracted

from the mass exported downstream, we observe that there were net losses of N in all restored lowland reaches and in one upland reach during average flow conditions (Fig. 9). Therefore, for these channels, our data indicate that more N entered than exited each restored reach. However, because the accumulation of errors (S_T) associated with the calculation of fluxes in the mass balances was relatively high relative to fluxes for most streams (Table 4), we cannot conclude that net retention was significant, except for HBR and SPA, and possibly WIL (Fig. 9). The upland degraded (control) reach was a distinct source of N (i.e., net N export, Fig. 9), while

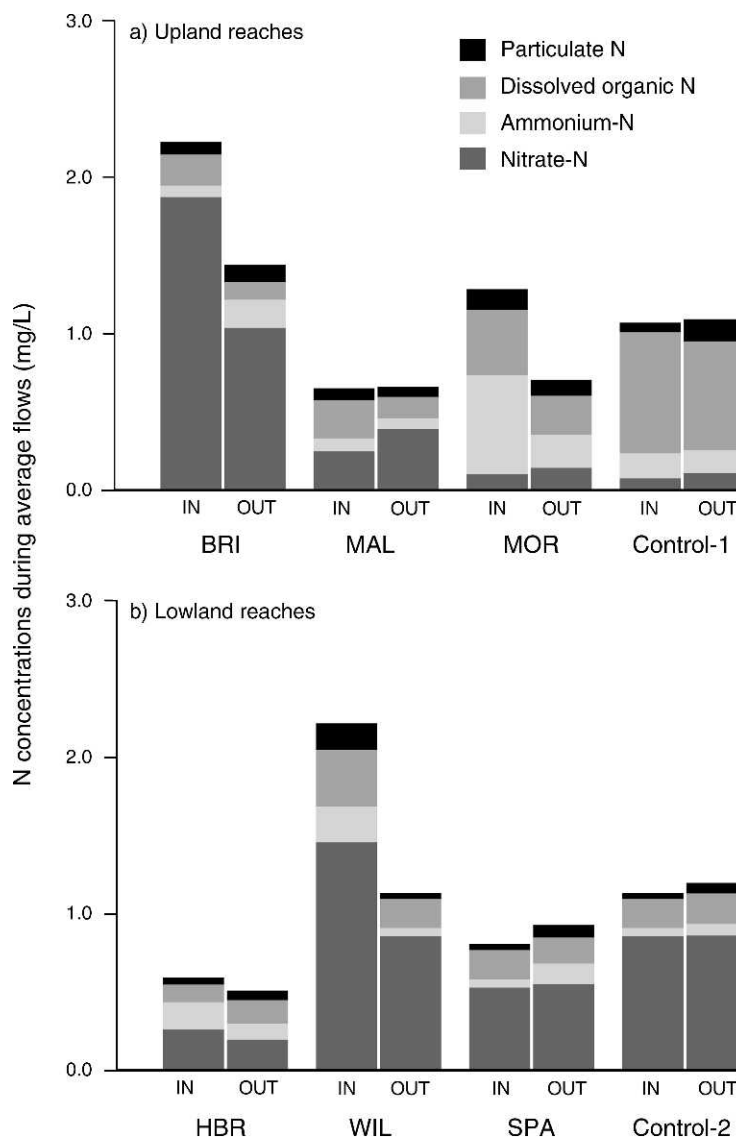


FIG. 4. Discharge-weighted mean TN concentrations (DWMC) in stream flow entering (IN) and exiting (OUT) the study reaches: (a) concentrations in the upland reaches; (b) concentrations in lowland reaches. The different shades of gray refer to different forms of N (particulate N [PN], dissolved organic N [DON], ammonium $[\text{NH}_4]$, and nitrate $[\text{NO}_3]$). For each stream, DWMC was calculated from 39 samples except for SPA, which was calculated from 25.

the lowland degraded (control) reach was much less so (Fig. 9).

In general, upland streams had a higher accumulation of errors (S_T) relative to fluxes than the lowland reaches because of additional errors associated with quantifying fluxes from lateral inlets, and groundwater and bank seepages (Table 4). Therefore, in streams where these sources were small relative to the sum of fluxes (e.g., SPA and HBR), the propagation of errors was lower. The relatively stable nature of water quality in the lowland channels above and below the study reaches during average flow conditions also lowered the magnitude of errors. On average, the lowland restored reaches reduced loads by $0.38 \text{ g N} \cdot \text{m}^{-1} \cdot \text{d}^{-1}$ (Fig. 9),

which represents about 23% of the N entering the stream reaches upstream during average flow conditions. This is a conservative estimate since it includes the modest load reduction observed at WIL.

High-flow conditions.—Among the 10 events sampled along the restored reach at HBR during stormflow conditions, we observed a decrease in TN fluxes in nine (Table 5), indicating that the restored reach functioned as a net sink for TN during stormflow conditions. On average, 1.16 g N was retained/removed per linear meter of restored stream per storm event, totaling about 0.34 kg N per storm event. Out of the 12 storm events sampled at WIL, net retention was observed in only four. On average, the WIL reach was a net source rather

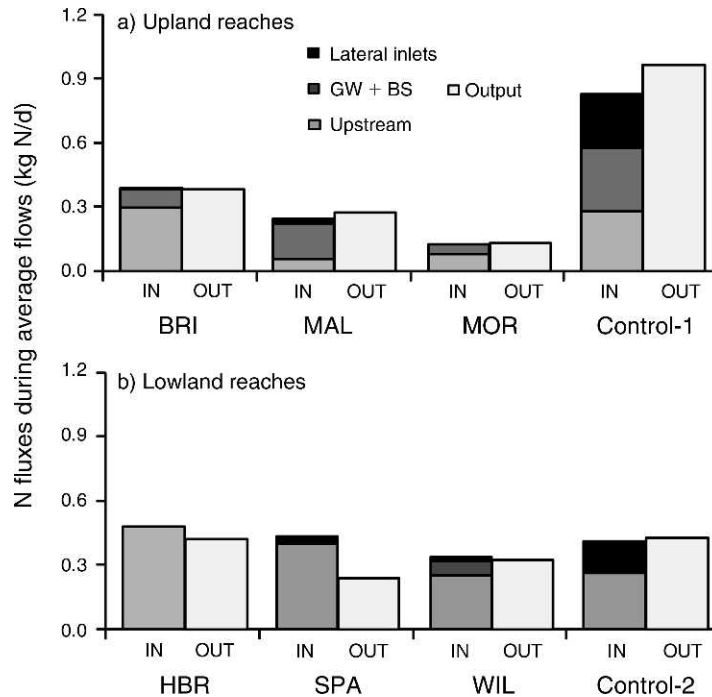


FIG. 5. Nitrogen fluxes in and out of the study reaches during average flow conditions. Inputs include water from upstream, groundwater + bank seepage (GW + BS), and lateral inlets. Output includes only stream water exiting the reach downstream. (a) Fluxes in upland reaches; (b) fluxes in lowland reaches. Fluxes were calculated as a product of concentration and discharge.

than a net sink of N, exporting about 0.77 g N per linear meter of restored stream per event. However, since not all inputs to the WIL restored reach were accounted for during stormflow conditions, it is possible that the inputs into the reach were underestimated and, therefore, this reach functioned more as a N sink than our data demonstrate.

DISCUSSION

Our study evaluated the effectiveness of restoration in upland and lowland channels in the Coastal Plain region of Maryland, and the results suggest that two out of six restored reaches (HBR and SPA) were clearly effective at reducing the export of TN to downstream waters. During average flow conditions, these two lowland

reaches exported significantly less N downstream of the restoration than the sum of inputs from external sources into the restored reach, but only HBR was considered to be effective during stormflow conditions. Net retention was at best suggested at WIL (i.e., the variability was too high to be conclusive), and we did not find clear evidence that the upland restored reaches reduced N export during any hydrological conditions.

The low gradient and wide stream valleys associated with the position of lowland channels in the watershed were likely crucial factors controlling the export of N in the restored reaches, especially during large storm events when loads of N in particulate form may be high, as we observed at HBR. However, as we will discuss, many other factors are sure to have influenced N-removal

TABLE 3. Average concentrations of total nitrogen (TN) entering the study reaches upstream, in lateral inlets, and exiting downstream, and average concentrations of total dissolved N (TDN) entering the study reaches as groundwater and bank seepage (GW + BS) during average flow conditions.

Stream reach	Upstream	Downstream	Lateral inlets	GW + BS
MAL	0.64	0.66	1.52	0.62
BRI	2.19	1.46	2.20	0.71
MOR	1.28	0.66	na	0.75
Control-1	1.15	0.95	1.32	1.10
HBR	0.60	0.52	na	na
WIL	0.93	0.81	1.13	0.53
SPA	2.15	0.94	2.25	na
Control-2	1.13	0.8	1.48	na

Note: All concentrations are expressed in mg N/L.

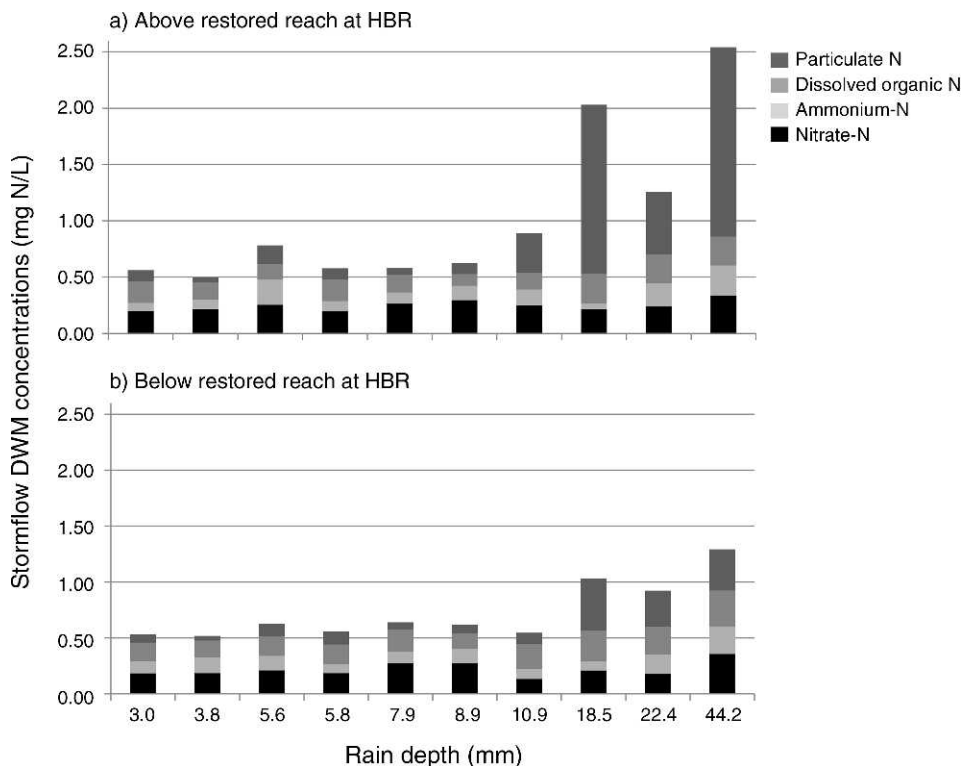


FIG. 6. Partitioning of N species in stormflow samples collected (a) above and (b) below the restored reach at HBR during 10 different storms varying in size. Each column represents the discharge-weighted mean (DWM) concentrations of stream water samples collected along an entire storm hydrograph; data are sorted on the x-axis according to rainfall depth of the storm event.

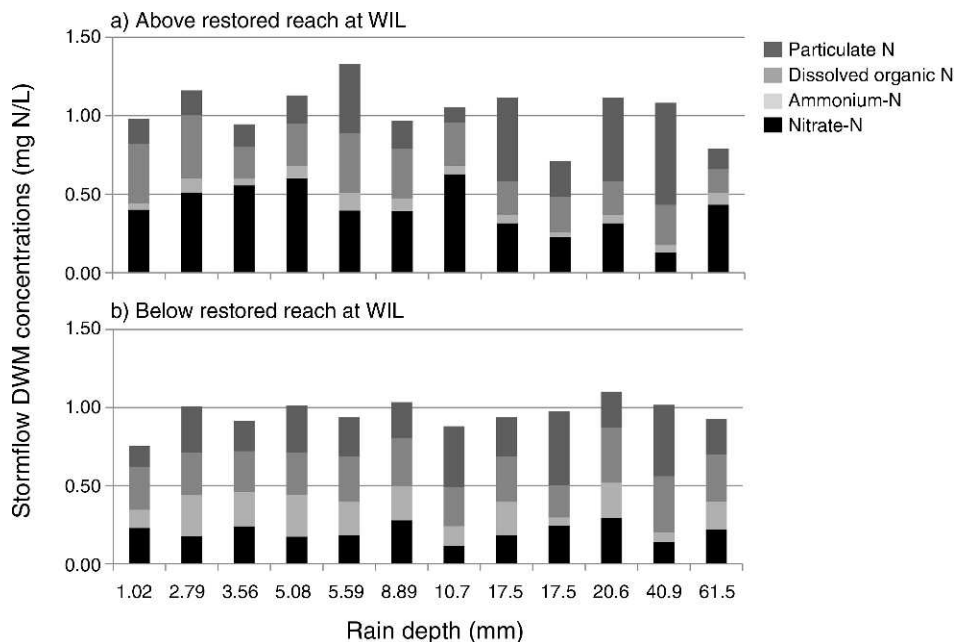


FIG. 7. Concentrations and composition of N in stormflow samples collected at WIL during different storm events (a) above and (b) below the restored reach. Each column represents the discharge-weighted mean (DWM) concentrations of samples collected during a stormflow hydrograph; data are sorted on the x-axis according to rainfall depth of the storm event.

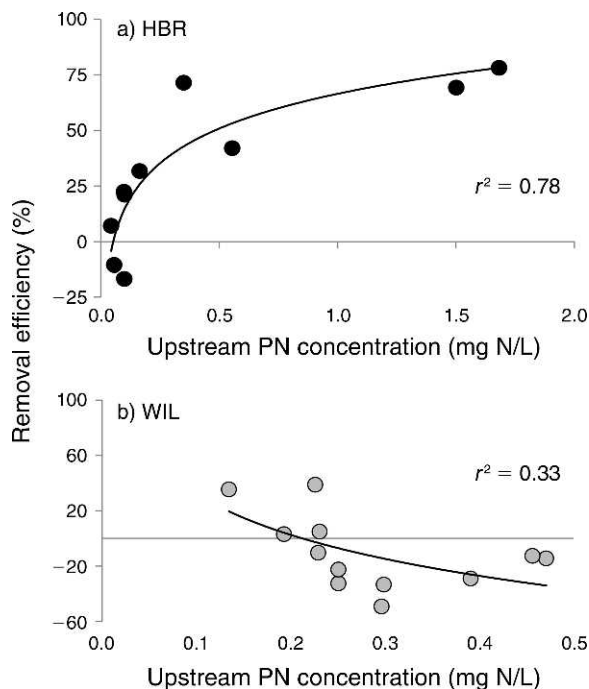


FIG. 8. Efficiency of N removal during stormflows for restored stream reaches as a function of discharge-weighted mean concentrations upstream of the restored reach: (a) HBR; (b) WIL. Both curves are significant ($P < 0.05$).

capacities of the study reaches. Clearly, comparisons among sites and between restored and control streams in all studies of real-world restoration projects are complicated by inevitable intrinsic preexisting differences among control and restored streams, or by differences in restoration design or implementation at each site. Because we lack pre-restoration data and stormflow data from some sites, the interpretation of our findings is further complicated. Despite these caveats, our findings suggesting that most of the restoration projects did not seem to result in significant decline in the downstream export of N along the restored reach have important implications for N management and for future studies on the water quality benefits of stream restoration.

A possible explanation for the higher retention rates observed at HBR and SPA vs. the other reaches during average flows is that most of the N transported in their channels originated upstream of the restoration as opposed to within the restored reach (i.e., from groundwater and lateral inputs) and, therefore, had more opportunity for in-channel processing. In contrast, as upland restored channels received relatively large amounts of new N from groundwater and bank seepage inputs along the entire length of the reach, it is likely that in-stream biotic uptake of N was insufficient to balance groundwater inputs and internal N cycling (Brookshire et al. 2009), resulting in a rapid increase in loads transported downstream. Therefore, unless resto-

ration efforts prevent some of the N in groundwater and bank seepage from entering the restored streams, the effectiveness of restoration in upland channels may be compromised by a limited capacity for in-stream N processing to compensate for groundwater inputs.

Inputs of N from groundwater and bank seepage at HBR and SPA were relatively small either because discharge from these sources was small, which was the case at Spa Creek (SPA), or because most of the groundwater and bank seepage that would have entered the lowland channels directly was retained in seepage wetlands and shallow ponds built adjacent to the restored reach. These wetlands and shallow ponds increased the opportunities for N processing and losses, and also for water losses through evaporation, evapotranspiration, and seepage. In the case of WIL, groundwater and bank seepage also accumulated in two large stormwater ponds and subsequently percolated to the stream through the berms separating the ponds from the restored channel. The berms were made of sand and wood chips with Atlantic white cedar (*Chamaecyparis thyoides*) planted throughout, and likely promoted N loss and retention via denitrification and plant uptake.

Higher water residence time associated with the lower gradient of lowland channels is another important factor controlling the effectiveness of stream restoration at reducing fluxes exported downstream because it provides opportunities for substantial in-channel N removal

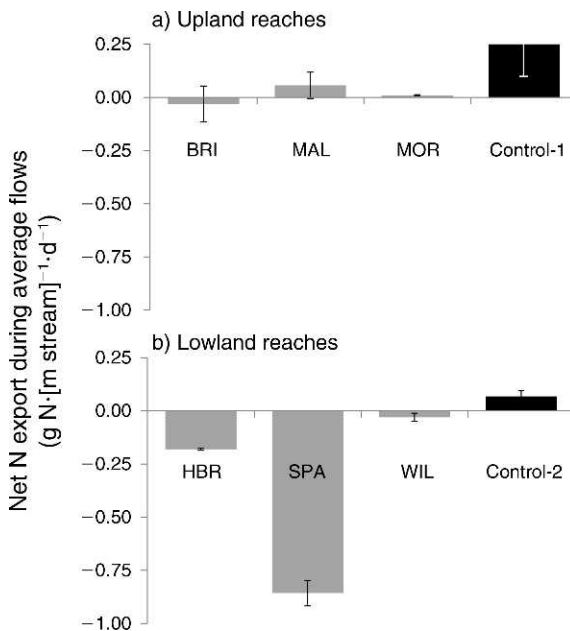


FIG. 9. Net export of total N during average flow conditions in study reaches. The mass balance includes inputs from surface water upstream of the study reach, groundwater + bank seepage, and lateral inlets. Output is the mass of N exported downstream of the study reach. Net export is shown for (a) upland reaches and (b) lowland reaches. Error bars represent the error percentage of flux, based on error analyses.

TABLE 4. Results of error propagation estimates and cumulative error (S_{total}) for each component of the mass balance for each study stream.

Stream reach	S_{US}	S_{DS}	$S_{\text{GW+BS}}$	S_{LI}	S_{total}	Error percentage of flux
MAL	0.017	0.036	0.010	0.605	0.184	277
BRI	0.010	0.041	0.047	0.234	0.029	108
MOR	0.044	0.025	0.083		0.005	16
CONTR-1	0.396	1.212	0.705	0.210	1.083	39
HB	0.112	0.073			0.009	3
WIL	0.037	0.042	0.014	0.1311	0.010	63
SPA	0.121	0.038			0.008	4
CONTR-2	0.026	0.050		0.561	0.317	34

Note: S represents the standard error associated with each flux included in the mass balance (US, upstream of study reach; DS, downstream; GW+BS, groundwater + bank seepage; LI, lateral inlets). Empty cells occur where the specific flux was not measureable and is not included in the balance for a particular stream.

and retention, as illustrated by the results from the addition experiments performed in the lowland study channels during average flow conditions (Appendix). However, we have found that the low gradient and higher residence time that characterize lowland channels seem to be also crucial at controlling the export of N in restored reaches during high-water conditions, especially when loads of N in particulate form are high, as we observed during stormflow conditions associated with large rain events at HBR.

We acknowledge that our evaluation of restoration effectiveness at reducing TN export during high flows has limitations because of the small number of restored streams monitored during stormflow conditions, and

also because of the limited number of storm events sampled in the streams monitored (Table 5). However, because we often observed both N concentrations and stream discharge notably lower below than above the restored reach at HBR during most storm events, our data provide strong evidence that stream restoration potentially plays an important role at reducing the export of N to downstream waters during stormflow conditions. Whether or not most degraded streams, at least in the lowlands of the Maryland Coastal Plain, transport high loads of N in particulate form, which was the main form of N reduced along the restored reach at HBR during stormflow conditions, is a matter for further investigation.

TABLE 5. TN fluxes up- and downstream of the restored reach at HBR and WIL, estimated for the duration of stormflow hydrographs for different storm events.

Storm event	Upstream flux ($\text{g N}\cdot\text{m}^{-1}\cdot\text{event}^{-1}$)	Downstream flux ($\text{g N}\cdot\text{m}^{-1}\cdot\text{event}^{-1}$)	Net export ($\text{g N}\cdot\text{m}^{-1}\cdot\text{event}^{-1}$)	Retention (%)
HBR				
1	2.16	2.34	0.19	9
2	3.52	1.28	-2.23	-63
3	10.37	4.48	-5.89	-57
4	1.68	1.26	-0.42	-25
5	1.39	1.12	-0.26	-19
6	1.31	1.00	-0.30	-23
7	6.80	4.56	-2.24	-33
8	1.45	1.34	-0.11	-8
9	2.66	2.45	-0.20	-8
10	0.87	0.75	-0.12	-13
Average	3.22	2.06	-1.16	-24 (+7)
WIL				
1	2.40	3.17	0.78	32
2	2.18	2.29	0.11	5
3	3.41	3.18	-0.22	-7
4	2.03	2.04	0.01	1
5	0.95	1.52	0.57	60
6	1.42	1.37	-0.05	-4
7	1.40	1.12	-0.28	-20
8	13.44	16.22	2.78	21
9	1.17	1.06	-0.11	-9
10	7.73	12.81	5.08	66
11	3.33	3.79	0.47	14
12	0.84	0.88	0.04	5
Average	3.36	4.12	0.77	14 (+8)

*Different forms of N must be considered
in restoration efforts*

Nitrogen retention is one of the essential ecosystem services (Grimm et al. 2005) that restoration seeks to reestablish in streams. Moreover, promoting dissimilative uptake of N or denitrification is the most common goal in restorations that target N reduction because it leads to the permanent removal of N from the system (Kaushal et al. 2008, Pickett et al. 2008, Klockner et al. 2009). However, while a fundamental condition for denitrification to occur is availability of NO_3^- (Meyer et al. 2005), organic N is the dominant N pool within rivers across most of the United States and is important even in basins with high anthropogenic sources of N (Scott et al. 2007).

In streams associated with urbanization, nitrate is commonly the main N form (Grimm et al. 2005). However, in our study reaches, much of the N was present in other forms. Out of the eight reaches monitored in this study, NO_3^- was the dominant form of N in half of them. Where the contribution of NO_3^- was relatively low, NH_4^+ and DON were the main forms entering and exiting the restored reach. Thus, given the diverse forms of nitrogen that may exist in streams, restoration efforts that focus mainly on promoting in-stream denitrification, such as establishing organic debris dams (Groffman et al. 2005) and inducing hyporheic exchange (Kasahara and Hill 2006), may fail to reduce N loads if NO_3^- is a minor component of TN in stream water.

We speculate that, for our study sites, dissolved oxygen and DOC may have controlled NO_3^- production in stream reaches where streamflow originated from stormwater ponds and stormwater wetlands (Thouin et al. 2009). Stormwater systems have proven to be effective at removing N in dissolved and particulate forms (Lee et al. 2009), but they can create an anoxic environment that promotes mineralization and limits nitrification. Water from stormwater ponds and wetlands also has large quantities of particulate carbon relative to N in stream water, which can slow nitrification (Starry et al. 2005). In streams not linked to stormwater ponds and wetlands, DON and NH_4^+ could also have originated from septic system discharge and leaking sewage pipes (Böhlke et al. 2006, Bernhardt and Palmer 2007).

*Restoration is only one of many factors
influencing N-removal capacity in streams*

Most N that enters watersheds is removed in terrestrial and riparian areas before entering streams and, while processing N in streams can be substantial, net N removal along a stream reach depends on hydrologic, geomorphic, and biological conditions in the watershed (Wollheim et al. 2008). Retention capacity of watersheds is assumed to be relatively high, but it is also proportional to inputs (Boyer et al. 2006); as inputs to the watershed increase, export to aquatic ecosystems

are likely to increase as well. Once excess N enters stream channels, in-stream N removal, especially of NO_3^- , increases with N concentrations in stream water. However, the relationship between concentrations and removal is nonlinear, hence, above a certain concentration level, N-removal capacity in streams becomes saturated (Claessens et al. 2009).

In our study streams, we observed that the reaches with the highest concentrations of dissolved inorganic N were those where most streamflow originated directly from street drainage networks, without any stormflow management control upstream (e.g., BRI and SPA). In these reaches, N concentrations decreased substantially along the restored reach during average flows (Fig. 4), yet, whether or not restoration was responsible for promoting N removal in both streams is unclear. We observed that at BRI, large portions of the restored reach were covered by iron oxidizing bacteria mats, which may be an indication that the groundwater supplied high levels of reduced iron to the stream channel. Iron oxidizing bacteria can mediate NO_3^- reduction coupled to iron oxidation in aquatic ecosystems (Burgin and Hamilton 2007), hence, anoxic groundwater with high levels of iron in reduced (ferrous) form rather than stream restoration may have caused the enhanced removal of NO_3^- at BRI.

At SPA, the high NO_3^- concentrations combined with a low channel gradient were probably the most important factors yielding the high removal rates observed (Fig. 9). It is important to note, however, that our results from the NO_3^- -N addition experiment at SPA (Appendix) suggest that removal rates would not continue to increase with increasing N concentrations since we found a decrease in N uptake when we elevated NO_3^- -N substantially above background levels (i.e., suggesting uptake saturation is likely to occur in these restored channels if N inputs increase). In addition, at the peak of the growing season, thick mats of grassy vegetation grow in this shallow and narrow channel, which likely remove a substantial amount of N from stream water. By contrast, in the winter, after this vegetation has died, N-removal rates decrease (S. Filoso, unpublished data).

Besides the magnitude of N inputs from the watershed and ambient N concentrations, discharge is also known to influence the N-removal capacity of streams. Generally, N-removal rates are lower during high flows because there is little time for stream water and sediments to be in contact (low hydraulic retention). Therefore, as N export in streams occurs increasingly during periods of high flows, the ability of streams to remove N decreases.

As our data show, stormflow contributed the majority of the discharge in the urban and suburban streams we studied (Table 2), and stormflow N concentrations in the streams that we monitored during stormflow conditions were comparable to or greater than those of average flow conditions (Figs. 7 and 8). Consequently, we

deduce that most of the N in the restored streams was exported during high flows, which means that, restoration efforts that reduce the magnitude of stormflow discharges are more likely to reduce the total export of N on an annual basis.

We observed that, along a relatively short restored reach at HBR (295 m, Table 1), peak flows seemed to be attenuated enough for TN loads to decrease by an average of 24% or 1.16 g N/m/storm event, especially because of retention of relatively high amounts of PN exported in stream water during high flows in this stream. On an areal basis, this decrease was on the order of 193 mg N·m⁻²·event⁻¹, assuming an average width of 6 m for the entire restoration reach. For the sake of comparison, we calculated that, during average flow conditions, the same reach reduced N fluxes by an average of 11%, or 1.39 mg N·m⁻²/h⁻¹, while according to Mulholland et al. (2009), areal denitrification rates in urban/suburban streams average about 2 mg N·m⁻²/h⁻¹.

The restored design used at WIL was similar to that of HBR, however, we did not observe such consistent net retention in the system during stormflow conditions. One of the reasons may be that, as we did not account for N inputs from lateral inlets during large storms, we presumably underestimated inputs into the system during stormflow conditions and, consequently, the N removal or deposition along the restored reach. Another reason may be that the ponds adjacent to the restored stream channel at WIL are much larger than the shallow ponds at HBR, so they accumulate large volumes of water that overflow into the channel during certain storms. When pond water overflows into the stream channel during storms, additional N is loaded into the restored reach and, consequently, N export is likely to increase downstream.

Whereas restored lowland streams converted to stream-wetland complexes such as HBR and WIL seemed to effectively reduce peak discharges, it is certain that steeper gradients and narrow adjacent floodplains associated with upland reaches may restrict the potential for certain streams to effectively reduce N export to downstream waters during high-flow conditions. Therefore, restoration of these types of streams may need to be accompanied by other measures at the watershed scale that help reduce excess overflow and manage N loads to streams.

Putting urban stream restoration for N management in perspective

While effective management to reduce excess N loads from anthropogenic sources in urban catchments should include a variety of measures at the landscape scale designed to capture, retain and ultimately denitrify this N (Bernhardt and Palmer 2007), failures at several levels have lead managers to increasingly turn to streams restoration as the main attempt to minimize N loads to downstream or coastal waters (Palmer 2009). If stream restoration were to be implemented along entire river

TABLE 6. Characteristics of a typical urbanizing headwater catchment in the study region, which we used to estimate the amount of N input to a catchment and compare with the amount of N flux reduced along a restored study stream.

Characteristic	Value
Surface area	90 ha
Restored stream	400 m
Average flow discharge	4.5 L/s
Impervious cover	33 ha
Residential land	16 ha
Lawn cover	8 ha
Parking lots and roads	6 ha
Residential lot size	0.375 acres (0.152 ha)
Average household	2.5 people
Average annual rainfall	1016 mm
Fertilizer application rate	50 kg N·ha ⁻¹ ·yr ⁻¹
N retention rate in catchment	65–80%

networks and at the watershed scale, this management approach would be likely be quite effective at reducing N loads to coastal waters. However, stream reaches targeted for restoration are usually short, have small drainage areas (commonly <1 km²) and, consequently, contribute relatively little to N export at the watershed scale. Therefore, stream restoration has a limited capacity to substantially reduce the delivery of excess N to coastal waters.

In order to provide some perspective about the magnitude of N flux reduction via stream restoration in a typical urbanizing catchment in the study region, we constructed a back-of-the-envelope N budget based on the catchment characteristics of WIL (Table 6) and the best-case scenario for in-stream N retention observed in this study for a restored reach (i.e., net retention for HBR). The main sources of N in the budget and typical for urban and suburban catchments include atmospheric deposition, human waste (septic or sewage), lawn fertilizer, and pet waste (Bernhardt et al. 2008).

Assuming that in the Chesapeake Bay watershed atmospheric deposition contributes about 10–14 kg N·ha⁻¹·yr⁻¹, sewage 0.81 kg N·ha⁻¹·yr⁻¹, septic systems 1.55 kg N·person⁻¹·yr⁻¹, and fertilizer to lawns 100 kg N·ha⁻¹·yr⁻¹ (Boyer et al. 2002, Wollheim et al. 2005, Chesapeake Bay Program 2008), we estimated that the total annual input of N to the catchment was approximately 1700 kg. We also assumed that about 72% of the N inputs to the landscape were retained in the catchment (including terrestrial system, riparian zone); therefore, only about 480 kg of N was loaded into the restored stream. Of the total N loaded into the stream reach, we estimated that 43%, or about 206 kg N/yr, was exported during average flow conditions, and 57%, or 273 kg N/yr, during high water conditions. Therefore, if we consider that, during average flow conditions the restored stream reduced the N export by 13% or about 27 kg N/yr, and during stormflows the removal rate was about 20%, or 55 kg N/yr, the restored reach theoretically removed/retained about 82 kg N/year. The retention/removal is equivalent to 17% of

the annual N export and to less than 5% of the total N input to the catchment, a catchment-scale figure consistent with recent estimates by Claessens et al. (2009) for streams in other catchments. Consequently, we reiterate the conclusion Claessens et al. (2009) and others (Bernhardt et al. 2008, Craig et al. 2008) reached: efforts to substantially reduce watershed N export should focus on riparian and upland interventions on the catchment scale as opposed to focusing only on in-stream channel processes, especially on a reach scale.

CONCLUSIONS

Despite evidence from our study that only some restoration projects are effective at reducing the export of N in urban/suburban streams, we have shown that load-reduction efficiency can be obtained, especially if restoration reduces the export of N during high water conditions. However, in order to compensate for the increasing pace of anthropogenic N inputs and the concomitant loss in the capacity of N processing in the drainage area (Bernhardt et al. 2008), streams may need to be increasingly manipulated or highly engineered to manage high N loads, at the expense of losing some of the fundamental functions associated with stream ecosystems. Thus, innovative projects such as those implemented at WIL and HBR could be implemented to reduce N flux in lowland areas if it is acceptable to convert them to dramatically different ecological systems (i.e., more like created wetlands than restored streams). While it remains to be tested, such designs may have limited effectiveness in the uplands unless increases in hydraulic retention are sufficient to remove not only N inputs from upstream but those from groundwater and lateral inlets. It is more likely that success will come from preventing excess N from being loaded into the stream channel in the first place by restoring the riparian vegetation and implementing watershed scale best management practices such as improved stormwater infrastructure. These actions are more likely to promote N removal/retention under stormflows than channel projects. While streams are part of a continuum on a drainage network, restoration is largely implemented in stream reaches as isolated units, where the proximity and magnitude of sources to the restored reach are ignored. Using a combination of approaches that target the source of the problem (i.e., lowering N inputs to the watershed and subsequent delivery to streams) will in the long run be much more likely to provide nutrient reduction and other restoration benefits than focusing on restoring streams one reach at a time (Bernhardt and Palmer 2011).

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APPENDIX

A table containing NO₃-N uptake rates in restored lowland streams calculated from N addition experiments conducted during average flow conditions (*Ecological Archives* A021-090-A1).

River restoration success: a question of perception

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Abstract. What defines success and failure of river restoration measures is a strongly debated topic in restoration science, but standardized approaches to evaluate either are still not available. The debate is usually centered on measurable parameters, which adhere to scientific objectivity. More subjective aspects, such as landscape aesthetics or recreational value, are usually left out, although they play an important role in the perception and communication of restoration success. In this paper, we show that different perceptions of restoration success exist by analyzing data from 26 river restoration measures in Germany. We addressed both objective parameters, such as hydromorphological changes and changes in fish and benthic invertebrate assemblages, from field investigations, and subjective parameters, such as opinions and perceptions, from water managers via an online survey. With regard to the objective hydromorphological and biotic parameters, our results agree with many studies that have reported improvements in the hydromorphology following restoration; however, there is no similar agreement between results concerning changes in the benthic invertebrate and fish assemblages. The objective results do not correspond to the subjective parameters because self-evaluation of the restoration projects by water managers was overly positive. Indeed, 40% of the respondents admitted that their evaluation was based on gut feeling, and only 45% of the restoration measures were monitored or occasionally checked. This lack of objectively recorded data meant that the water managers were not able to reasonably evaluate restoration success. In contrast, some self-evaluation responses reflected a different perception of the restoration success that was based on landscape aesthetic values or on benefit for the public; others adopted a general “condemned to success” attitude. Based on our data, we argue (1) that goals should be thoughtfully formulated prior to restoration implementation and (2) that it is necessary to monitor river restoration success from different perspectives.

Key words: assessment; EU Water Framework Directive; failure; Germany; goals; monitoring; online survey; ordination; river restoration; success.

INTRODUCTION

River restoration has recently become a billion-dollar business, and it is a driving force behind applied ecological science, which is evident from rapidly increasing numbers of publications. There are different drivers for river restoration: increasing awareness of aquatic biodiversity and its value, the wide range of ecosystem services provided by near-natural rivers and floodplains, and legal requirements or international obligations. In the 1970s and 1980s, river restoration was usually limited to improving water quality by reducing organic pollution, eutrophication or inflow of

toxic substances. More recently, the focus in North America and Europe has shifted to improving the hydrological and morphological character of channels and floodplains, which is mainly accomplished by restoring local-scale aquatic and riparian habitats either by adding boulders or large woody debris or by re-creating the morphological features of channels. The claim of either success or failure of these restoration measures has been debated for decades (e.g., Palmer et al. 2010), but the picture is incomplete because it remains uncommon to evaluate success with a sound monitoring program or a standardized before-and-after sampling program (Bernhardt et al. 2005, Alexander and Allan 2007, Feld et al. 2007, Roni et al. 2008). This study will neither add to the growing body of restoration effectiveness studies nor will we again debate and hypothesize possible reasons for success or failure;

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instead, we will shed light on another dimension: the perception and communication of success.

“Success” of river restoration measures can be defined with widely different parameters as the basis: abiotic aspects (e.g., related to hydromorphology, hydrology, or substrate), biotic aspects (e.g., biodiversity, functional characteristics, or ecological traits) and socioeconomic aspects (e.g., recreational value, ecosystem services, or policy). More generally, river restoration success can be judged based on objective (i.e., measurable) parameters and on subjective parameters, which involve preferences, feelings, or aesthetic values. Both the objective and subjective parameters may be subject to change, e.g., if definitions of reference conditions are adapted or if societal attitudes change.

In a recent study, Woolsey et al. (2007) developed an extensive indicator set of 49 parameters that covers the three aspects mentioned above; though, only measurable, quantifiable parameters were included, which is typically the case. However, there is no doubt that subjective parameters play an important role in the perception and communication of restoration success. Water managers or municipal politicians often communicate a “success story” of a river restoration project, while results based on objective parameters might be more divergent concerning the effects on habitats and on assemblages (Muotka and Syrjänen 2007, Jähnig et al. 2009b, Weber et al. 2009). It is therefore obvious, but rarely acknowledged, that restoration success is a matter of perspective; however, the societal debate regarding what the ultimate success-indicators in river restoration should be has been missing thus far.

To advance this debate we analyzed the performance of different success indicators for 26 relatively large river restoration measures in Germany; we included both analysis of hydromorphological and biotic parameters based on field investigations and the perception of water managers recorded via an online survey. Our goal was to (1) compare and relate the objective and subjective parameters to each other and (2) disentangle the different perceptions of success as a basis to discuss the consequences of river restoration practice and monitoring.

MATERIAL AND METHODS

General approach

We sampled and analyzed data from 26 restoration projects, which consisted of both non-restored and restored river sections (“paired sites”) in German lower mountain areas and lowlands. The data collected included the following: (1) the perception of success of the restoration measures by water managers via an online questionnaire, (2) hydromorphological changes between the restored and non-restored sections, and (3) biological changes in fish and benthic invertebrate communities between the sections. The selected restoration projects were relatively large, consisted of hydro-

morphological improvements, and had to be well documented (Appendix A).

Subjective parameters: online questionnaire

Water managers play a key role in restoration projects in Germany with large influence and decision power, thus representing very important players in restoration projects. Getting insights into their perception is therefore crucial for the development of future restoration strategies. An online questionnaire was directed at the executive water managers of the investigated restoration projects, which aimed to (1) evaluate the definition of goals (three questions), (2) evaluate the individual’s perception of fulfillment of these goals (five questions), (3) obtain information on barriers to achieving project success and poll major improvement suggestions (three questions), and (4) assemble information on the projects, the institutions, and the role of the respondent within the projects (four questions). From these questions, we chose a subset of the most relevant questions for comparing the perception of success (Table 1).

A total of 139 water managers of the 26 projects were approached, of which 69 water managers from 22 projects responded. If several answers were available per project, the median of the answer was calculated, which resulted in one value for each site.

Objective parameters: hydromorphological and biological data

Data on hydromorphology and biota prior to restoration were not available; therefore, we followed a space-for-time-substitution approach to quantify hydromorphological and biological changes. Each restored section was compared to an upstream non-restored section, which was similar in terms of geology, adjacent land use, and catchment size. Non-restored sections were selected on basis of three parameters: (1) the non-restored section should be similar to the restored section prior to restoration, (2) the non-restored section must be located up-stream of the restored section to avoid an (positive) influence of by drifting organisms from the restored section, and (3) in between the two sections several riffle–pool segments must be present to reduce the risk of organism interchange. Since this parameter depends on stream size the inter-site distances varied between 300 m and 2100 m (one outlier by 7000 m), mean was 750 m without and 1050 m including the outlier. Although this design cannot completely exclude any organism interchange it reflects a compromise between the desired colonization by dispersing organisms into restored sites and a suitable study design. Additionally some bias is accounted for in the German assessment system, which is focusing on reproducing units (e.g., juvenile fish).

All sections were sampled using the same field protocol. Habitat composition was surveyed at two spatial scales, with mesohabitat characteristics of river and floodplains being recorded on 10 equidistant trans-

TABLE 1. Subset of the most relevant questions and the possible answers used in the online questionnaire in this analysis.

Number	Question	Possible answers
1	Were there clear (measurable) goals defined prior to conducting the restoration measures?	none, unclear, medium, precise, very precise, don't know
2	Please state and prioritize these goals from 1 (high) to 5 (low priority).	open answers
3	How do you evaluate overall project success?	none, little, medium, high, very high, don't know
4	What is the basis for your evaluation?	monitoring, spot checks, gut feeling, others, please explain your method
5	How much did changes to the following aspects contributed to your evaluation? (a) Continuity, (b) structure, (c) floodplains, (d) recreation.	none, little, medium, high, very high, don't know
6	How much contribution did your project add to complement the following directives and legal requirements? (a) FFH and birds directive, (b) EU-Waterframework directive, (c) biodiversity strategy, (d) natural conservation (general), (e) landscape (general).	none, little, medium, high, very high, don't know
7	Which improvements could be implemented in future projects?	open answers

Note: FFH stands for Fauna, Flora, and Habitats Directive.

ects within the bankfull discharge area (see Jähnig et al. [2008] for details). Aquatic microhabitats were described by substrate type (Hering et al. 2003), depth and velocity pattern, and these data were recorded at 5 or 10 points along these transects, depending on river width. From these data, eight aggregated metrics were derived: (1) Shannon-Wiener index of channel feature composition, (2) spatial diversity index (Fortin et al. 1999) of substrate composition, (3, 4) coefficient of variation of depth and velocity data, (5) sum of all channel features, (6) mean width, (7) non-fixed shoreline, and (8) profile type (data provided in Appendix B). Generally, restoration measures should increase all metric values. In the subsequent data analysis, we selected metrics that fulfilled two criteria (Table 2): (1) are appropriate on different scales, and (2) cover various habitat aspects, such as substrate distribution, velocity, floodplain habitats and others (Jähnig et al. 2008).

To determine the hydromorphological differences between the non-restored and restored sections we followed the procedure suggested by Jongman et al. (1995): detrended correspondence analysis (DCA) was first applied to analyze the length of the gradient in the data set. Based on the length of this gradient (gradient length <2) principal component analysis (PCA) was applied (CANOCO 4.51; ter Braak and Smilauer 2003). Prior to this analysis, the data were transformed (see Table 2).

Benthic invertebrates and fish were chosen as biological indicators because they provide complementary information because of their different life cycle lengths, different relevant habitat scales, and differing mobility. Both groups have been extensively studied in the river restoration context and are mandatory to use for EU Water Framework Directive (WFD) ecological assessments. The groups were sampled following the national standardized procedures for assessment (Dußling et al. 2004, Haase et al. 2004a, b). Sampling was performed in

the late spring and early summer of 2007 and 2008, and paired sites were sampled on the same day. To evaluate assemblage changes between the restored and non-restored sections, organism data were ranked using nonmetric multidimensional scaling (NMS) with Bray-Curtis dissimilarity as a distance measure (PC-ORD v5.10; McCune and Mefford 2006) and using the flexible beta linkage method. Dissimilarity was calculated with log-transformed abundance data.

As a major driver for restoration in Europe, we were also interested in achievements under the EU Water Framework Directive (WFD), which aims at a “good ecological status” for all rivers by 2015. This status is defined through the biota (aquatic vegetation, benthic invertebrates, and fish fauna) and for all components field, laboratory, and calculation rules are available, including free software to calculate assessment results. Using this freeware Asterics v3.1.1 and Fibs v8.0.6 we calculated the official assessment results (software available online).^{7,8}

RESULTS

Responses overview

Participants in the online survey agreed that prior to conducting the restoration measure, clear and measurable goals had been either precisely (55.1%) or very precisely (8.7%) formulated (Fig. 1). Success was rated as high or very high (43.5% and 36.2%, respectively). The success rating classes “no” and “little success” were only chosen four times. The evaluations were mainly based on the participants’ gut feelings (close to 40%), followed by monitoring and spot checks (25.0% and 22.2%, respectively) and by some other unspecified measures. According to their methodological explana-

⁷ www.fliessgewaesserbewertung.de

⁸ www.landwirtschaft-bw.info

TABLE 2. Spearman's ρ values of aggregated hydromorphological metrics and the scale.

Hydro-morphological metrics	Transformation	Scale	Hydromorphological metrics							
			CV _{velocity}	CV _{depth}	SDI	SWI	Profile type	Non-fix	Mean width	Sum
CV_{velocity}	none	micro	1.00							
CV_{depth}	none	micro	0.57	1.00						
SDI	none	micro	0.53	0.31	1.00					
SWI	none	meso	0.49	0.60	0.46	1.00				
Profile type	square root	meso	-0.48	-0.63	-0.55	-0.50	1.00			
Non-fix	arcsin(sqrt)	meso	-0.06	-0.15	-0.28	-0.28	0.50	1.00		
Mean width	log	meso	0.23	0.19	0.04	0.25	-0.08	0.19	1.00	
Sum	square root	meso	0.49	0.57	0.47	0.83	-0.43	-0.25	0.25	1.00

Notes: Transformations were applied prior to PCA. Two metrics were removed because they showed autocorrelation during PCA; those selected for PCA are shown in boldface type. Metric definitions are the following: micro, aquatic microhabitats; meso, aquatic and riparian mesohabitats; CV, coefficient of variation; SDI, spatial diversity index (describes composition and spatial pattern of substrate composition); SWI, Shannon-Wiener index of channel feature composition; non-fix, non-fixed shoreline (%), recorded on right and left shore. Profile type ranges from 1 to 7, with more natural profiles having a higher value; Mean width is the mean overall width of sections recorded in meters; and Sum is the sum of all different channel features recorded within a section.

ations, these never followed a standardized procedure. The respondents considered structural changes to have contributed most to this success, with the proportion of high and very high structural change contribution responses summing to 84.1%. Water managers were aware that their projects served a range of directives, and 40% and 65% of respondents reported a high and

very high contribution, respectively. However, they viewed the contribution of their projects to add to complement the WFD as significantly larger compared to contributions to other directives and legal requirements (t test of ratings, $P < 0.05$).

Hydromorphological improvements were stated most often as the goal, which was followed by measures to

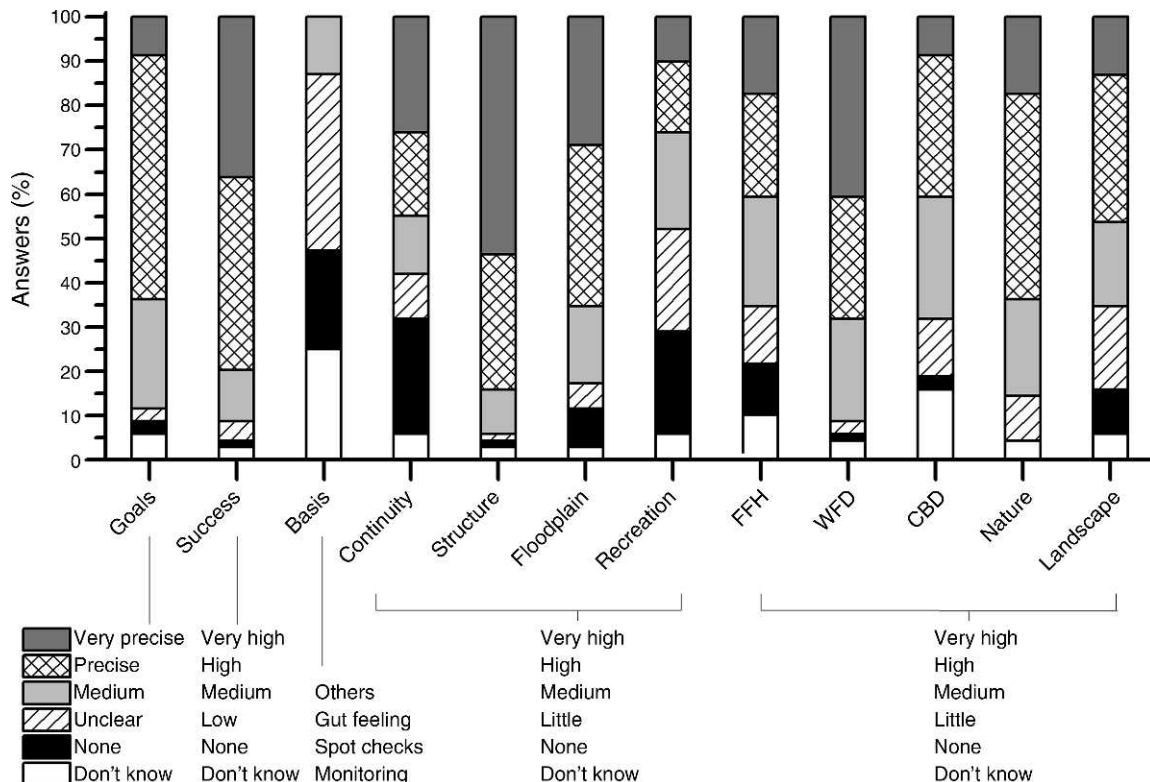


FIG. 1. Distribution of answers to the questions (shortened, details available in Table 2; words shown in *italics* here denote x-axis labels): Were there clear measurable *goals* defined? How do you evaluate overall project *success*? What is the *basis* for your evaluation? How much did changes in [*continuity*, *structure*, *floodplain*, *recreation*] contribute to your evaluation? Which contribution did your project add to complement [*FFH*, *WFD*, *CBD*, *nature* conservation, *landscape* conservation]? Abbreviations are: FFH, Fauna, Flora and Habitats Directive; WFD, Water Framework Directive; CBD, Convention on Biological Diversity.

TABLE 3. Number and priority of the restoration goals derived from question 2 (Table 2): “Please state and prioritize goals from 1 (high) to 5 (low priority)”.

Categorized answer	Priority				
	1	2	3	4	5
Hydromorphology: structure, dynamics	24	20	4	2	
Continuity	12	3	2	1	2
Floodplain development	7	13	14	6	1
Habitat diversity	7	6	6	5	2
Project development	7	5	1	5	2
Biodiversity	4	5	8	2	
Flood protection	3	5	6	7	5
Water quality		2	5	1	1
Recreational value		1	3	3	4
Public participation			4	2	1

Notes: The answers given were categorized by the authors and ordered from most to least frequent entries within each priority class. Values are the number of responses at that priority level.

improve continuity (Table 3). Goals related to ecological improvements, such as biodiversity or habitat improvement, with the implicit hope of increasing biodiversity, were mentioned far less often.

Perception of hydromorphological changes

Generally, the non-restored and restored sections were hydromorphologically separated; there is an overall clear left-right division in Fig. 2. Four restored and four non-restored sections deviated from this pattern. The first and second PCA axes of six hydromorphological metrics at the 50 stream sections (Fig. 2A) explain 63.9% of the variance in the data set. Eigenvalues were 0.466 for the first and 0.173 for the second PCA axes (eigenvalues for the third and fourth PCA axes were 0.134 and 0.086, respectively). The first axis correlated with both meso- and micro-scale parameters, such as channel element diversity (Shannon-Wiener index channel elements), profile type, substrate distribution (spatial diversity index), and velocity variability (CV velocity). The second axis only vaguely correlated with the proportion of non-fixed shoreline and overall mean width. Restored and non-restored sections are separated. There was no pattern recognizable in the way water managers had rated the success, and even the restored sections that appeared within the non-restored division had been rated as highly successful. In other words, the restoration measures per se were given a successful rating.

Perception of biological changes

The results of the NMS analyses for both benthic invertebrate and fish assemblages show no tendency of restored sections to diverge from their non-restored counterparts in any systematic way (Fig. 2B and C). The numbers of taxa, abundances, and Shannon-Wiener indices only show minute and nonsignificant changes (Sundermann et al. 2011). Again, when comparing the biological changes with the water managers' self-evaluations, no pattern is recognizable to describe why one project was rated as more successful than others.

Achievements of the Water Framework Directive, in terms of their assessment results (Fig. 3), show that 48% of the restored sections achieved a better assessment result for fish, while 20% of sites have a better assessment result in the restored section for invertebrates. In 40% (fish) and 60% (benthic invertebrates) of the sites the assessment result did not change; in 12% (fish) and 20% (benthic invertebrates) of the sites the assessment result showed deterioration. The WFD-required “good ecological status” has not yet been reached in 84% (fish) and 77% (invertebrates) of the restored sections.

DISCUSSION

With regard to the measured hydromorphological and biotic parameters, our results agree with many studies that have reported improvements in hydromorphology; however, the results are mixed with regard to the changes in benthic invertebrate and fish assemblages following restoration (Entekin et al. 2009, Miller et al. 2010). This does not correspond with the subjective parameters, i.e., the self-evaluation of restoration projects by the water managers, which were mostly positive. In fact, managers rated measures per se as being successful or highly successful. This supports our original assumption that subjective parameters play an important role in the perception and communication of restoration success.

Although some hydromorphological parameters are relatively easy to evaluate, water managers predominantly expressed subjective evaluations. One general rationale might simply be the lack of objectively recorded data: almost no measures were monitored by the executive boards (Fig. 1) and the results presented in this study were not available during the online survey. Additionally, water managers might have had a different idea about restoration success that was based on landscape aesthetic values or benefit for the public, which are commonly reflected by hydromorphological features.

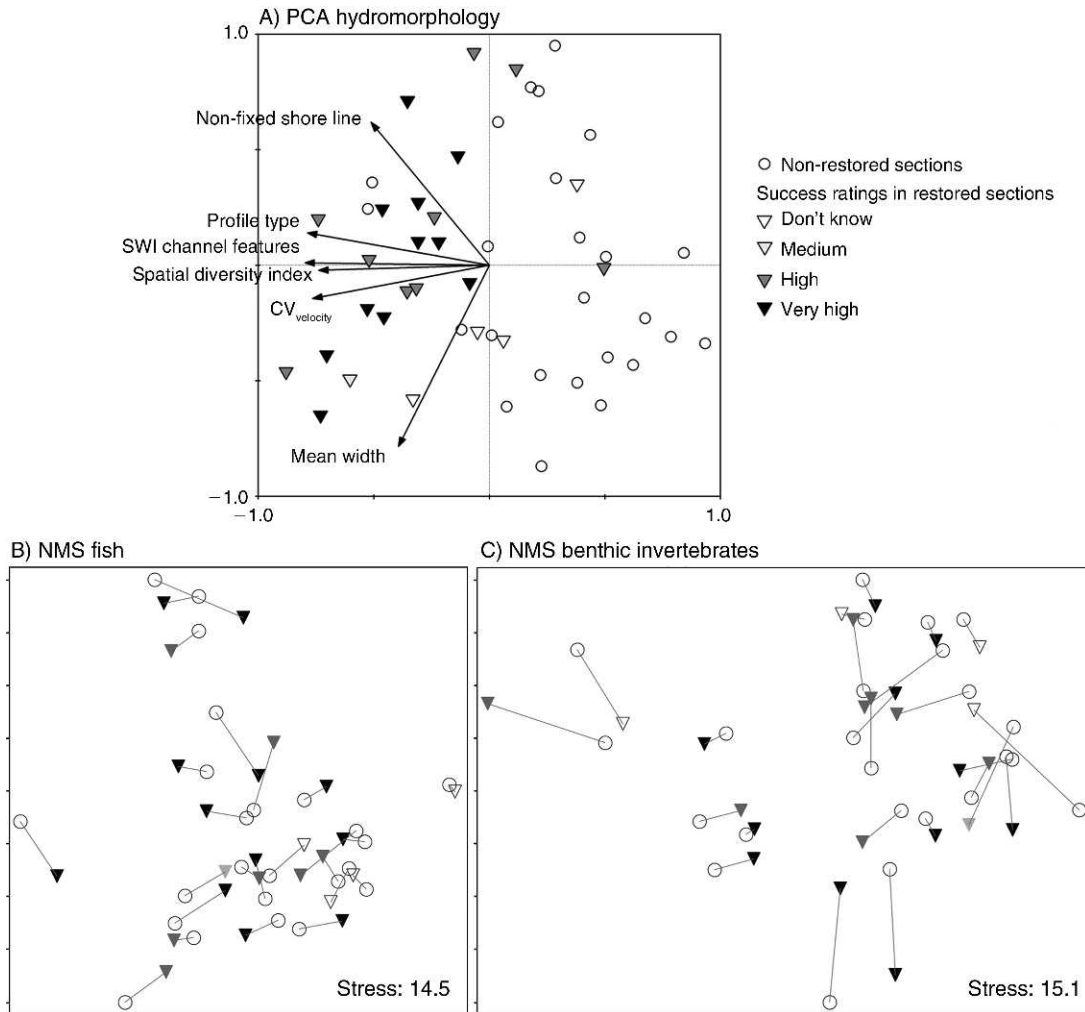


FIG. 2. Ordination diagrams for (A) hydromorphological parameters, (B) fish, and (C) benthic invertebrates. Open circles represent non-restored sections; triangles represent restored sections. The answers “no success” and “little success” dropped out during median calculation. The site of the Rhine River was excluded due to the very different size of the river. Abbreviations are: SWI, Shannon-Wiener index; CV, coefficient of variation. In panels B and C, paired sites are connected by a gray line.

The mismatch between subjective self-evaluation and objective measurements of biological parameters is more difficult to explain. Several projects did not aim to improve biodiversity or to fulfill the biodiversity-related legal requirements; indeed, of the 229 replies regarding their project's goals, only 22 answers from nine of the projects were related to ecology, which included increased biodiversity, improved spawning habitat, or the aims of the Water Framework Directive (Table 3). However, about 60% of the measures were conducted by regional environmental agencies, which are obliged to fulfill the legal requirements of the WFD, i.e., reaching a “good ecological status” in all rivers by 2015. This goal is likely to be missed by 70% of the rivers in Europe (EU Commission 2007), including most of the investigated sections (Fig. 3). It thus seems more likely that the respondents were not able to estimate how biotas respond to restoration. Effects on river hydromorphol-

ogy, on the other hand, can be estimated more easily and were correspondingly more often defined as project goals (yet their success was not measured, see above). This can be further inferred from the answers given to question 2 (“Please prioritize these goals from one [high] to five [low priority]”), where the ecological aims were overambitious, e.g., establishing a near-natural assemblage. There was little perception of the requirements to improve assemblages beyond merely generating in-stream habitats at a local scale. This includes goals such as attaining the required length of near-natural sections to have self-sustaining populations, establishing source populations to recolonize restored sections, and meeting the time spans required for recolonization (e.g., Bond and Lake 2003, Pretty et al. 2003, Harrison et al. 2004, Hughes 2007, Jähnig and Lorenz 2008), or goals generated by considering the “ghost of land use past” and the remaining stressors (Harding et al. 1998, Giller 2005,

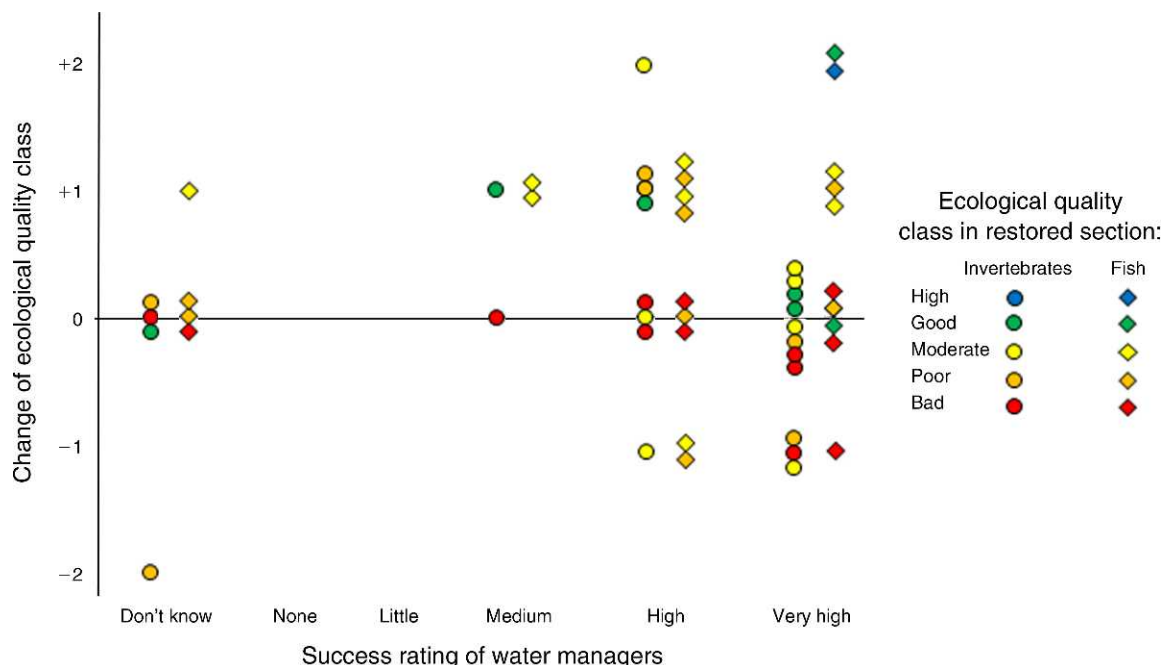


FIG. 3. Comprehensive diagram showing the official ecological quality class (EQC) results for invertebrates and fish in the restored sections, the answer of water managers to question 3 from Table 2, "How do you evaluate overall project success?" (x-axis), and the differences in the EQC results between non-restored and restored values (y-axis; values above zero indicate improvement; values below zero indicate deterioration).

Hilderbrand et al. 2005, Sondergaard and Jeppesen 2007, Langford et al. 2009). All these parameters have to be considered when planning restoration measures with the goal to improve biodiversity. Scientists have the urgent task of identifying and quantifying the parameters required for an ecologically successful restoration (Palmer 2009) because, at the moment, hypotheses addressing the poor biological response to restoration are rarely based on empirical data.

In addition to the unintended overestimation of restoration success, there is a third highly subjective argument: from a water manager's perspective, restoration measures are "condemned to success." If measures are not immediately successful in reaching legal requirements, such as those of the Water Framework Directive, then river restoration might be seen as a waste of money and might come to a halt. This argument is supported by the water managers' surprising awareness of problems and their numerous suggestions for improvements of restoration efforts, despite their generally positive evaluation of the project's success (Table 1: question 7). Their suggestions include the following: (1) to improve political and legal factors, especially related to integrated river management or land use changes; (2) to improve the means of project handling, e.g., to achieve more binding results from preliminary discussions; (3) to achieve a better estimation of the biological effectiveness of measures; (4) to increase area availability and pursue large-scale measures, including reallocation of land use; (5) to speed up the approval of projects and shift the

approval to lower administrative levels; (6) to strengthen or impose an obligation for post project monitoring and communication of results; (7) to supply more financial means; and (8) to improve participation of stakeholders during all project phases.

From an objective point of view, the poor definition and monitoring of river restoration success (Zedler 2007) has prevented a shift to more successful restoration approaches, especially in terms of biological changes. However, as laid out in the introduction, a societal debate as to what *all* stakeholders might perceive as successful is missing. Definitions of success could include aspects of legal requirements, socioeconomic reasoning, health issues, landscape aesthetics, and nature conservation. We thus suggest extended assessment systems to evaluate restoration success. The highly sophisticated long developed assessment systems within the WFD do not account for time lags, ecological constraints (e.g., related to dispersal capabilities or hysteresis effects), or subjective parameters. One of the essential requirements is to formulate clear goals for each individual assessment measure and to then employ an adapted set of success indicators (e.g., Woolsey et al. 2007, England et al. 2008). To enhance the probability of achieving desired biological outcomes, a long-term vision and a linkage of measures to the context of the catchment are required. The agreed-upon indicator set must also consider ecosystem compartments that are not covered in the standard freshwater assessment or monitoring approaches. Riparian vegetation and arthro-

pod, land–water interactions, self purification, denitrification, functional aspects, and ecosystem services have all been shown to react more quickly and more positively to restoration measures than aquatic assemblages do (Lepori et al. 2005, Rohde et al. 2005, Kaushal et al. 2008, Aldridge et al. 2009, Jähnig et al. 2009a, Klockner et al. 2009, Palmer and Filoso 2009, Tullos et al. 2009). In addition to these objective parameters, subjective parameters should also be included, such as the sense of landscape beauty perceived by the public and the priceless opportunity to teach children about nature. On the other hand, there could also be a feeling of nuisance from biting midges or a perception of a dangerous riverscape (Le Lay et al. 2008). To include such perspectives would require (expensive) representative surveys of all stakeholders and is an interesting and promising future research topic.

By addressing the contradictory perspectives of success and by disentangling the current different approaches to assess it, we have shown that perspective is extremely important when evaluating river restoration measures. With this study, we hope to contribute to a societal debate regarding what should be considered to be river restoration success indicators and to highlight the necessity of implementing monitoring standards.

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APPENDIX A

Characterization of the restoration projects (*Ecological Archives* A021-091-A1).

APPENDIX B

Diversity of mesohabitats and microhabitats at paired sites (*Ecological Archives* A021-091-A2).