EFFECTS OF BIOENGINEERED BANK STABILIZATION ON URBAN STREAMS

by

ELIZABETH BALLARD SUDDUTH

(Under the Direction of Judy L. Meyer)

ABSTRACT

Bioengineered streambank stabilization is among the most commonly performed of stream restoration practices. The goal of this study was to assess the effects of bioengineering on bank and benthic macroinvertebrate communities and their habitat. I studied four bioengineering sites on Peachtree and Nancy Creeks in Atlanta, GA, and compared them to an unrestored site and an urban reference site in the same watershed. The bank macroinvertebrate community was found to have higher abundance, biomass, and richness at the reference site and several of the bioengineering sites; in addition, these values were higher on organic habitats than inorganic habitats. The benthic macroinvertebrate community was more dense, but less diverse than the bank community. At the reach scale, percent organic bank habitat proved to be a strong predictor for many aspects of the bank and benthic macroinvertebrate communities. These results suggest that joint planting is not an ecologically beneficial form of bioengineering. Overally, this study shows that bioengineering can have positive effects on urban streams; however, bioengineering alone cannot mitigate the effects of urbanization.

INDEX WORDS: Stream ecology, Aquatic macroinvertebrates, Bank habitat, Bank stabilization, Bioengineering, Stream restoration, Bed mobility, Urban streams, Peachtree Creek

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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

Around the world, streams face impacts from a variety of human activities including agriculture, silviculture, and urbanization (Allan and Flecker 1993). Urbanization in particular has many direct impacts—including point source pollution and piping of streams—as well as indirect impacts—such as increased stormflows due to impervious surface cover and sedimentation from upland erosion on construction sites (Paul and Meyer 2001). Many different governmental and non-governmental agencies are trying to address the impacts of urbanization on our water resources (Baer and Pringle 2000).

In addition to policy changes and better growth management, many agencies are turning their attention towards stream restoration techniques to mitigate anthropogenic effects, including the effects of urbanization (Gore et al. 1995). Stream restoration ecology is a very young science and currently faces many impediments to its growth as a science, especially the lack of hypothesis testing, monitoring, and reporting of results (Lake et al. 2002). It is estimated that currently only about half of restoration projects collect baseline data prior to restoration or monitor any parameters in the stream following restoration (Bash and Ryan 2002). Monitoring is much more common among projects done for ecological goals (e.g., fish habitat) rather than "engineering" goals (e.g., bank stabilization). If stream restoration projects are not treated as experiments, monitored, and the results are not reported, stream restoration ecology cannot advance as a science and the practice of stream restoration will be doomed to repeat mistakes (Kondolf and Micheli 1995).

Studies of stream restoration projects have shown great successes (e.g., Charbonneau and Resh 1992) and great failures (e.g., Kondolf et al. 2001). Most projects, however, have mixed results: they frequently meet some goals and not others. For instance, a study of large woody debris addition projects in urban streams in the Pacific Northwest found that the goal of habitat creation had been achieved, but the expected accompanying biological improvements had not occurred (Larson et al. 2001). Another study of large woody debris addition in Mississippi found that the project achieved its physical and biological goals for the first year, but the woody debris was washed out in the second year (Shields et al. 2003). The successes and failures of these projects provide useful lessons for both scientists and practitioners.

Bioengineered streambank stabilization is among the most commonly performed of all stream restoration practices (Brown 2000). It involves the use of non-structural means, such as geotextile fabrics, wood, plantings, and live cuttings, to increase stability and complexity of eroding streambanks (Li and Eddleman 2002). It is done in agricultural settings as well as in urban streams by government agencies and community groups (Firehock and Doherty 1995). It is also frequently used in combination with other restoration techniques, such as Natural Channel Design (Doll et al. 2003). While some bank erosion is a natural part of alluvial streams (Schumm 1985), accelerated streambank erosion is a common feature of urban streams (Whipple et al. 1981).

Many studies have evaluated the success of bioengineering at its primary goal of stabilizing streambanks (e.g., Henderson 1986, Abernethy and Rutherford 1998, Schaefer 2000, Brown 2000). Because bioengineering often does not have expressed ecological goals, its ecological effects are not nearly as well understood. For that reason, I undertook a study of the effects of bioengineered bank stabilization on urban streams. The most likely ecological benefits of bioengineering are improved bank habitat and higher abundance, biomass, diversity, and richness of the bank macroinvertebrate community (Chapter 2). It is also possible for this practice to have more indirect effects on streams, including benefits for the benthic macroinvertebrate community as it mitigates other impacts of urbanization, such as low retention of benthic organic matter, sedimentation, and high peak stormflows and accompanying bed mobility (Chapter 3).

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CHAPTER 2

EFFECTS OF BIOENGINEERED STREAMBANK STABILIZATION ON BANK HABITAT AND MACROINVERTEBRATES IN URBAN STREAMS[1]

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Abstract

Non-structural streambank stabilization, or bioengineering, is a stream restoration practice commonly done to remedy eroding streambanks. Little research has been done assessing its ecological effects. We surveyed bank habitat and sampled bank macroinvertebrates at four bioengineering sites, an unrestored site, and a reference site in the urban Peachtree-Nancy Creek catchment in Atlanta, GA, USA. The amount of organic bank habitat, wood and roots, was much higher at the reference site and three of the bioengineering sites than at the unrestored site or the other bioengineering site, where joint planting was the primary bioengineering technique used. At all sites we saw high abundance of tolerant taxa, especially chironomids and oligochaetes, and low richness and diversity of the bank macroinvertebrate community. Total biomass, insect biomass, and non-chironomid insect biomass were highest at the reference site and two of the bioengineering sites ($p<0.05$). Higher total biomass, insect biomass, and odonate biomass were found on organic habitats (wood and roots) versus inorganic habitats (mud, sand, and rock) across all sites. Percent organic bank habitat at each site proved to be strongly positively correlated with many factors, including taxon richness, total biomass, insect biomass, and shredder abundance, and negatively correlated with percent chironomid biomass. These results suggest that bioengineered bank stabilization can have positive effects on bank habitat and macroinvertebrate communities in urban streams, but it cannot completely mitigate the impacts of urbanization.

Keywords: Stream restoration, Bioengineering, Bank stabilization, Bank habitat, Macroinvertebrates, Urban streams, Peachtree Creek

Introduction

Streambank erosion is a natural part of the dynamic system that is an alluvial stream channel (Schumm 1985). Because the banks of an alluvial channel are composed primarily of material previously transported by the stream, the material is easily eroded and redeposited as the channel migrates across its floodplain (Schumm 1985). Both undercut streambanks and woody debris falling into the stream due to this natural erosion provide important, stable habitat for macroinvertebrates and fish.

While some streambank erosion is natural in all alluvial streams, accelerated erosion of streambanks is common in streams impacted by urbanization (Whipple et al. 1981). This is due to a combination of anthropogenic effects, especially change in stormflow timing and quantity and removal of riparian vegetation (Bledsoe and Watson 2001). From a human perspective, the results—ugly, bare dirt and loss of real estate—are undesirable. Accelerated streambank erosion also greatly increases downstream sedimentation (Trimble 1997). Increased fine sediments from accelerated streambank erosion can directly affect the stream ecosystem by decreasing primary productivity and faunal diversity and abundance (Wood and Armitage 1997).

Many different techniques have been developed to slow streambank erosion. Streambank stabilization using non-structural methods or bioengineering, such as geotextile fabrics and live cuttings, is one of the activities most commonly performed in the name of stream restoration (Brown 2000). Its intention is to mimic the natural structure and function of a vegetated bank in a non-impacted system, including underground soil reinforcement, surface protection, reduction of shear stress, and anchoring to more stable strata (Li and Eddleman 2002). This practice can be done as a simple project by groups ranging from city governments to scout troops (Firehock and Doherty 1995). In addition, it is frequently used with other stream restoration techniques, such as Natural Channel Design (Doll et al. 2003). One reason for its use is widespread concern about the effects of more traditional, structural, or "hard" streambank stabilization methods such as riprap and concrete. These range from ecological effects, such as loss of habitat, to aesthetics (Li and Eddleman 2002). In addition, hard streambank stabilization has been shown to only exacerbate accelerated erosion downstream (Henderson 1986).

Because it does not directly address the causes of accelerated erosion, bioengineering can only be a "band-aid" on the problem. Other small scale restoration practices do not have large ecosystem effects, as reach-scale approaches can't compensate for catchment effects (Larson et al. 2001, Moerke and Lamberti 2003, Shields et al. 2003). However, by recreating quality bank habitat, bioengineering has the potential to contribute to a more diverse and functional stream ecosystem. Although many studies have addressed the effectiveness of bioengineering practices at reducing accelerated erosion (e.g. Henderson 1986, Abernethy and Rutherford 1998, Schaefer 2000, Brown 2000), little is known of the ecological effects, despite its widespread use.

The objective of this study was to examine the effects of bioengineered streambank stabilization on the available bank habitat and macroinvertebrate communities in an urban catchment in Atlanta, GA. We characterized bank habitat and sampled benthic invertebrates in unrestored and reference stream reaches, as well as reaches restored 1, 3, 7, and 9 years ago. An increase in the amount of stable, natural bank habitat in the form of roots and wood would be expected in bioengineered streams. We hypothesized that abundance, biomass, richness, and diversity of bank macroinvertebrates would be greatest at the reference site and least at the unrestored site, with values increasing with time since restoration at the other sites. As the bioengineered habitat became vegetated, stabilized, and was colonized, one would expect that the macroinvertebrate community at restored sites would come to resemble that of the reference site. Organic bank habitat, such as roots and wood, would be expected to have higher abundance and biomass of macroinvertebrates, whereas these measures should be lower on inorganic habitats, such as sand, rock, and mud. Finally, we expected to see a positive relationship between abundance, biomass, diversity, and richness of the macroinvertebrate community and the total amount of this organic habitat at each site. Sampling of microhabitats, such as a meter reach of woody debris, tells us only what is living in that patch at that point in time; macroinvertebrates are mobile creatures, so it is also appropriate to look at their population in comparison to reach-scale measures such as total amount of organic habitat.

Methods

Study Sites

Peachtree-Nancy Creek flowing into the Chattahoochee River in the Atlanta metropolitan area was chosen for this study because it is an urban stream network where stream restoration is becoming common. In the past few years, a number of new community watershed groups have formed, and local city and county governments have made water quality improvement a priority. A reference site was chosen on South Peachtree Creek at South Peachtree Creek Nature Preserve (Figure 1) to represent a likely goal of restoration projects: a fairly natural area in an urban setting. An unrestored site was chosen on Nancy Creek in Murphey Candler Park. This site resembles the pre-restoration conditions of the restored sites—inadequate riparian zone and eroding banks—and was used for comparison because no pre-restoration data were available for the sites themselves.

Four sites (Figure 1 and Table 1) were identified in the Peachtree-Nancy Creek catchment where bioengineering techniques had been used for stream restoration. The length of the bank stabilization completed at each site varied. In each case, the bottom 150 meters of the restored reach was used as the study reach. These four sites represent a range of ages, from 1 to 9 years since completion of bank stabilization and are numbered according to their age (e.g., bank stabilization at Site 9 was completed nine years ago). Because bioengineering uses a variety of techniques, there is also some variation in practices performed at each site. Site 9 features a combination of geotextile fabric and live cuttings on the banks. These were also used at Site 7, as well as "logs" of geotextile fabric. Site 3 had been riprapped previously and there the primary restoration technique was "joint planting"—plantings and live cuttings inserted into the riprap. Site 1 restorers used live cuttings, geotextile fabrics, and tree revetments, in which a large tree is buried in an unstable bank. For site photographs, see Appendix A.

The entire Peachtree-Nancy Creek catchment is about 225 km^2 (Rose and Peters 2001). It is 53% residential, 22% commercial, 11% open space, and 14% other, with a total of about 26% impervious surface (MAUWI Guidance Document 2002). Both streams carry substantial amounts of stormwater: 88% of Nancy's annual flow and 76% of Peachtree's annual flow is stormwater (MAUWI Guidance Document 2002). During high stormflows, Peachtree Creek has 30 to 100% greater peak flows than other Georgia Piedmont streams of similar size in less impacted catchments (Rose and Peters 2001).

Bank Habitat Assessment

Bank habitat was assessed by a visual survey. We walked both sides of 150-m reaches of stream, measuring with a tape measure and categorizing habitat. We considered available habitat to be whatever was present at the water's edge at baseflow. Categorization was based on primary bank materials: bedrock, natural rocks, riprap, gravel, sand, clay, silt, native vegetation, exotic vegetation (kudzu and ivy), fine roots, large roots, undercut roots, small woody debris, large woody debris, and exposed geotextile fabric (remaining from bioengineering). In some cases, multiple categories were chosen for a given meter of stream—for instance, an area might be predominantly sand with a quantity of small woody debris.

For data analysis, these fine categories were combined into broader categories: rock, sand, clay, roots and vegetation, and wood. For some analyses, these categories were further combined into inorganic or organic. In the cases in which the initial field categories overlapped (e.g. sand and wood), a set of rules was developed to ensure consistent grouping into the broader categories (see Appendix F).

Macroinvertebrate Collection

Macroinvertebrates were collected at all sites four times: July and November 2002, and March and June 2003. Four 1-m reaches of streambank were randomly selected at each site on each sampling date and the type of habitat present noted using the same terminology and procedures as in the overall bank habitat assessment. Although we attempted to sample a 10 cm by 1 m section of streambank, because of the differences between the different habitat types, it is impossible to estimate exact surface area sampled (i.e., roots and rocks have much more surface area than sand over the same spatial area). In addition, our primary concern was investigating improvements made by bioengineering to a specific length of streambank. For these reasons, both habitat and macroinvertebrate measures are expressed by linear meters of streambank throughout this study.

The 1-m reach was surrounded by a 1-mm mesh kick-net in order to block off the area of stream and capture macroinvertebrates dislodged from the bank. We then scraped the habitat to approximately 10 cm below the water's surface along the margin of the stream with a 1-mm mesh D-net while brushing the habitat with a plastic brush. The contents of the brush and both the kick-net and the D-net were rinsed, collected, preserved in 95% ethanol, and returned to the lab. In the lab, samples were stained with Phloxine B, then hand picked under a dissecting microscope, and preserved in 70% ethanol.

All macroinvertebrates were measured and identified to genus wherever possible using Merritt and Cummins (1996), Peckarsky et al. (1990), and Thorp and Covich (1991). Chironomidae were identified as Tanypodinae or non-Tanypodinae. Biomass was calculated using length-mass regressions (Benke et al. 1999). Functional feeding groups were assigned based on Merritt and Cummins (1996) for insects and Thorp and Covich (1991) for non-insects. Although they were collected and measured, crayfish were excluded from biomass calculations because of their large size and relative rarity.

Data Analysis

Data were analyzed using JMP version 5.0.1 (SAS Institute Inc, Cary NC). Similarity was calculated using Percentage Similarity Index (PSC) for each site in each season and then averaged over the year of samples (Washington 1984). Diversity was calculated using Shannon's H' for each site in each season (Washington 1984). Total taxon richness and insect richness were calculated for each site in each season and then averaged over the year of samples. One-way and two-way ANOVA's were used with an alpha value of 0.05. Abundance and biomass data were log transformed, and percent data were arcsine square root transformed for analysis.

Results

Bank Habitat

The unrestored site had a variety of habitat types, but was dominated by sand and rock, from all of the riprap present there (Figure 2). The reference site had primarily roots and wood habitat types, but also some sand and some mud—a good mix of habitats. Sites 9, 7, and 1 also had a mix of habitat types, with some rock and slightly more sand than the reference. Site 3 was the exception among the bioengineered sites because it was dominated by rock habitat from the riprap in which the joint plantings were done. This suggests that all of the practices commonly called bioengineering do not necessarily increase the presence of roots and wood bank habitat to the same extent. Examining habitat using the broader categories of organic and inorganic, the reference site had the highest percent of organic habitat (Table 2). Sites 9, 7, and 1 had about 50% organic habitat; this is more than site 3 and the unrestored site, but not as much as the reference. Sites 9, 7, and 1 were more than 70% similar to the reference by habitat composition, whereas the unrestored and 3 year old sites were far less similar (Table 2).

Differences in Macroinvertebrates among Sites

Overall, the most common taxa were oligochaetes, Tanypodinae and non-Tanypodinae chironomids, copepods, and the limpet *Ferrissia*. Most sites had several genera of Odonata. Hydropsychid caddisflies were found at all sites but Site 9. Hydroptilid caddisflies, a more sensitive family of caddisfly, were found at the unrestored site and Site 3 only on the last sampling date. Baetid mayflies were found at all sites but Site 7. Stoneflies, the third order of the common EPT index, were not found at all in this study. At all sites, the percent community composition was quite similar to the composition of the reference site by abundance, but not very similar to the reference site by biomass (Table 3). Sites 9, 7, and 1 always had greater than 50% similarity to the reference site by abundance, while Site 3 and the unrestored site both dropped below 50% similarity to the reference in November. The unrestored site was the only site to never get above 50% similarity to the reference site by biomass. There were no significant differences among the annual means of the similarities.

We sampled in different seasons throughout a year to gather information on the natural variation of these systems. Although there were some significant differences among the seasons—abundance was significantly higher in the two summer samples than in November and March (see Appendix C)—two-way ANOVAs revealed no significant interaction between season and habitat or site. Because there were no differences in response to seasonal variation by either habitat or site, we do not consider seasonal variation further and focus instead on the effects of bioengineering on these streams.

Total abundance, diversity, and taxon richness of bank macroinvertebrates were not significantly different among sites (Table 4). Macroinvertebrate biomass (excluding crayfish), however, was highest at Site 9, Site 7, and the reference site and lowest at Site 3 ($p<0.05$). The sites with the highest total biomass also had the highest insect biomass; though chironomids made up much of the biomass at these sites, the higher biomass was also due to the less common but larger taxa, such as Ephemeroptera, Odonata, Trichoptera, and Tipulidae. All sites had high abundance and biomass of tolerant oligochaetes and chironomids, and there was no significant difference in abundance or biomass of these tolerant taxa among sites. The unrestored site had the highest percent chironomid abundance, but it was only significantly higher than Site 1. Site 9 and the reference site had the highest non-chironomid insect biomass, significantly higher than Site 3.

Functional feeding group composition is another useful metric for examining community differences among sites. All sites were dominated by collector-gatherers and there were no significant differences among sites in the percent abundance or biomass of this group (Table 5). Scrapers and shredders were quite uncommon at all sites and there were no significant differences in their biomass or abundance among sites. Collector-filterers were also quite rare, but there were some differences among sites. By abundance, filterers made up a significantly higher proportion of the community at the reference site and Site 3 than at all other sites. By biomass, however, filterers made up a significantly higher percent at Site 3 than at the reference site, Site 7, or Site 1. Predators were more common than scrapers, shredders, or filterers at all sites. There were significantly more predators at Site 7 than at Site 3, by both percent abundance and biomass.

Habitat Effects on Macroinvertebrates

Total macroinvertebrate abundance was significantly higher on wood and roots than on sand (Table 6). These abundance data show a pattern that is evident throughout the habitat data, namely that the organic habitats—wood and roots—group together and have values of many metrics that are significantly higher than those found in the inorganic habitats—mud, rock, and sand. Similarly, total biomass was significantly higher on roots than rock and sand, though total biomass on wood was only significantly higher than sand. Both insect abundance and biomass showed a similar pattern. Higher abundance and biomass on organic habitats is likely due to both the higher surface area and more diverse habitat provided. Just as in the examination of the differences among sites, the differences among habitats were largely driven by the large and relatively more sensitive Ephemeroptera, Trichoptera, Odonata, and Tipulidae; non-chironomid insect abundance and biomass were significantly higher on organic habitats than inorganic habitats. However, chironomids and oligochaetes were also more abundant on organic habitats than on inorganic habitats.

There were few differences among habitat types in functional feeding group composition (Table 7). Percent predator biomass was significantly higher on roots than on rock, and percent scraper biomass showed the opposite pattern. Percent shredder abundance was significantly higher on roots than on rock or sand, and the preference for organic habitat over inorganic habitat was true for percent shredder biomass as well. Roots and wood are more likely than the other habitat types to retain the leaves that shredders eat.

Reach Scale Effects of Organic Habitat on Macroinvertebrates

The previous examination of habitat effects was based on the habitat from which each sample was taken and revealed important differences between organic and inorganic habitats. To consider the effect of the overall habitat at the site on macroinvertebrates throughout the year, we regressed annual mean macroinvertebrate metrics against percent organic habitat measured at the site (Table 8, for percent organic habitat values see Table 2). We found significant ($p<0.05$) positive relationships between percent organic habitat and taxon richness, total biomass without crayfish, insect biomass, non-chironomid insect abundance and biomass, Diptera biomass, chironomid biomass, shredder abundance, collector-gatherer biomass, and shredder biomass. We found significant negative relationships between percent organic habitat and percent Diptera biomass, percent chironomid biomass, and percent scraper biomass.

Discussion

The overall goal of this study was to assess the effectiveness of bioengineered streambank stabilization in restoring macroinvertebrate communities, yet few differences in macroinvertebrate communities were observed among the stream reaches studied, including differences between the unrestored and reference sites. Although the reference site has the widest wooded riparian area and highest percent organic habitat on the banks of the sites in this study, it is not in an undisturbed reach of stream. A forested stream in the Piedmont region of North Carolina, which should be comparable to the streams in this study, had a taxon richness of 75, whereas the reference site in this study had a taxon richness of 23 (Lenat and Crawford

1994). The North Carolina stream population was also more than 60% Ephemeroptera, Trichoptera, and Plecoptera, whereas at this reference site Ephemeroptera and Trichoptera barely made up 5% of the abundance, and no Plecoptera were ever found.

The small differences between sites in this study suggests that in this highly impacted catchment, absence of bank habitat is not the primary limitation on the macroinvertebrate community; other effects of urbanization, such as overall water quality, stormflow timing and volume, and bedload transport, have severe impacts on aquatic macroinvertebrates in urban streams like these (Jones and Clark 1987). New habitat created by structures installed to restore habitat in urban streams in Washington had no effect on the macroinvertebrate community (Larson et al. 2001). The authors concluded that the extent of overall catchment development strongly limits the success of small-scale projects. The preliminary results of a study of reach-scale channel reconstructive restoration suggest that in urban streams reach-scale restoration may not adequately address the catchment effects, such as stormwater runoff, which lead to stream degradation (NCDENR 2003).

One important difference among sites in this study was that overall biomass, insect biomass, and non-chironomid insect biomass were all higher at the reference site and the two oldest restoration sites (Sites 9 and 7) than at the unrestored site. Biomass in streams seems to be largely determined by availability of habitat and food (Benke 1984). Streams with poor water quality due to organic enrichment can have extremely high biomass, though very low diversity (e.g., Kimerle and Anderson 1971), and so biomass alone is not an ideal indicator of biological health. However, the macroinvertebrate communities at these sites suggest no obvious differences in organic enrichment. The combination of higher total number of taxa along with highest overall biomass, insect biomass, and non-chironomid insect biomass suggests that at the reference site and Sites 9 and 7 the macroinvertebrate community is doing substantially better than at the other sites. There are many factors which could contribute to this difference, including techniques of restoration and position in the catchment. However, the data in this study suggest that higher macroinvertebrate biomass, insect biomass, non-chironomid insect biomass, taxa richness, and insect richness are found on organic habitats and that higher values of these metrics occur at sites where there is a higher proportion of organic habitat overall. From these results we conclude that bioengineering can be of substantial benefit to bank macroinvertebrates by creating preferred habitat. It seems likely that in a less urbanized catchment this response would be even stronger.

Increasing the amount of wood and root habitat on banks is an ecologically beneficial aspect of bioengineering. However, this benefit was not realized at Site 3. The style of bioengineering done at Site 3, called joint planting, involves inserting live cuttings and plantings into riprap. Unfortunately, this means that most of the bank habitat available to macroinvertebrates is still rock, rather than wood or roots. The amount of organic habitat at Site 3 (4.9%) was by far the lowest of this study, lower than even the unrestored site. Site 3 was consistently one of the lowest sites on many of the macroinvertebrate metrics. Riprap with moss growing on it can be a high quality macroinvertebrate habitat (Linhart et al. 2002), but moss was not observed to be growing on the riprap at Site 3. Yearling fish are much more likely to prefer unaltered shoreline habitats to riprap (Garland et al. 2002), and our study suggests that macroinvertebrates prefer organic habitat to rock. If the goals of a bioengineering project include ecological restoration, joint planting will probably not meet those goals. However, only three years have passed since the joint planting at Site 3, and so it is possible that over a longer time period the trees could become a more important component of the bank habitat, and joint planting could create more organic habitat.

Other studies have shown that in naturally sandy streams, stable woody debris is very important as a habitat and a refuge in storms (Benke et al. 1984, Benke et al. 1985, Borchardt 1993, Benke 2001). In a large river, the lower hydraulic stress and more stable habitat of the area along the shore were shown to be a refuge in stormflows (Rempel et al. 1999). Bank habitat was found to be a refuge from another type of disturbance in an urbanizing catchment in the Georgia piedmont; increasing fine sediments in riffles due to urbanization led to an increase in macroinvertebrate richness on the banks as former riffle inhabitants fled to the only available stable habitat (Roy et al. 2003). Because urbanization also leads to increased stormflows, the importance of banks as refugia from both high flows and sedimentation likely increases with increasing urbanization. Although bioengineering does not restore the lost stable benthic habitat or reduce stormflows, it does provide an alternative habitat which may allow macroinvertebrates to persist in a stream in which they otherwise could not.

Conclusions

Overall, these results suggest that bioengineered streambank stabilization can have positive effects on bank habitat and macroinvertebrate communities in urban streams, but it cannot solve all the problems facing these streams. We predicted that bioengineering would increase the amount of stable, natural bank habitat in the form of roots and wood. This proved to be largely true, with the notable exception of the “joint planting” at Site 3, which appeared to have negligible effects on the amount of available organic bank habitat. We hypothesized that abundance, biomass, richness and diversity of bank macroinvertebrates would be greatest at the reference site and least at the unrestored site, with values increasing with time since restoration at the other sites. While we did find some significant differences between the reference and unrestored sites, there were surprisingly few, suggesting that our “reference” site is probably in need of some “restoration” itself. The bioengineered sites did not follow a pattern based on time since restoration; rather, amount of organic habitat created appeared to be driving differences among sites. We predicted and observed higher abundance and biomass on organic bank habitat than on inorganic habitats. Finally, as predicted, we saw a positive relationship between abundance, biomass, diversity, and richness of the macroinvertebrate community and the total amount of this organic habitat at each site. From this we can conclude that the total amount of organic habitat on the reach scale is an important determinant of macroinvertebrate communities.

This study has several implications for implementers of stream restoration projects. First, it provides little justification for the use of the technique known as “joint planting.” It does not appear that planting a few trees and live cuttings can mitigate the severe effects of riprap on bank macroinvertebrate communities. Greater use of wood, plants, and roots in bank stabilization projects has the potential to enhance the bank macroinvertebrate communities of urban streams. However, bank erosion is a naturally occurring part of alluvial streams, and erosion is important in creation of undercut root and woody debris habitats; therefore complete elimination of streambank erosion should not be a goal of stream restoration. Finally, although bioengineered streambank stabilization can have positive impacts on the bank macroinvertebrate community, the streams in this study suffered from many more impacts than poor bank habitat, and bank habitat improvement is not sufficient to counteract the negative effects of urbanization seen in these streams. To truly benefit these urban streams, more attention must be paid to mitigating the impacts of urbanization catchment-wide.

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Table 2.1. Characteristics of the six study sites in the Peachtree-Nancy Creek catchment, Atlanta, GA, USA

Site Name	Years Since Restoration	Length of Bioengineering (m)	Site Location	Stream Name	Catchment Size (km^2)
Site 9	9	330	Medlock Park	South Peachtree Creek	26
Site 7	7	200	Dresden Park	North Peachtree Creek	8
Site 3	3	300	Starlight Neighborhood	Nancy Creek	57
Site 1	1	250	Blue Heron Nature Preserve	Nancy Creek	65
Unrestored	--	--	Murphey Candler Park	Nancy Creek	36
Reference	--	--	S. Peachtree Cr Nature Preserve	South Peachtree Creek	24

Table 2.2. Percent organic habitat (roots and wood) as surveyed on the banks at each site. Percent similarity (PSC) of surveyed habitat at each site to habitat at the reference site.

	Reference	Unrestored	9	7	3	1
% Organic	71.4	25.4	50.6	42.1	4.9	53.9
% Similarity to Reference		45.3	77.9	70.8	21.7	73.4

Table 2.3. Percent similarity (PSC) of macroinvertebrate community at each site in each season to reference site macroinvertebrate community by abundance and biomass.

	Un	9	7	3	1
	% Similarity to Reference by Abundance				
July 2002	81.5	87.5	74.9	70.8	60.4
Nov 2002	38.0	68.8	82.3	46.0	83.2
Mar 2003	80.6	57.8	63.3	72.1	60.4
June 2003	64.3	78.3	84.0	51.0	87.7
Mean	66.1	73.1	76.1	60.0	72.9
	% Similarity to Reference by Biomass				
July 2002	49.1	70.8	23.5	60.2	58.6
Nov 2002	8.0	39.1	5.8	7.2	23.4
Mar 2003	38.5	67.1	34.5	28.0	38.8
June 2003	13.5	22.6	55.0	14.0	14.2
Mean	27.3	49.9	29.7	27.3	33.7

Table 2.4. Annual mean±standard error of diversity (Shannon's H'), richness, abundance (Ab, #/meter of bank), and biomass (Bm, mg DM/meter of bank) of macroinvertebrates at each site. Values with different letters are significantly different based on one-way ANOVA ($p<0.05$). Abundance and biomass were log transformed, and percents were arcsine square root transformed for analysis. Crayfish were excluded from biomass analyses.

	Diversity (H')	Insect Diversity	Abundance	Insect Ab	Non-Chironomid Insect Ab	Oligochaete Ab	Chironomid Ab	% Chironomid Ab
Ref	1.4± 0.1 a	1.0± 0.2 a	225.6± 50.2 a	113.7± 33.7 a	16.6± 5.5 a	96.9± 28.0 ab	97.1± 30.1 a	40.7± 7.0 ab
Un	1.2± 0.2 a	0.7± 0.1 a	122.7± 44.1 a	89.3± 38.1 a	6.2± 2.4 a	20.2± 6.0 b	83.1± 36.9 a	59.1± 7.9 a
9	1.4± 0.3 a	1.2± 0.1 a	150.1± 42.1 a	74.7± 23.0 a	16.1± 3.9 a	64.6± 34.9 ab	58.6± 20.6 a	35.9± 7.1 ab
7	1.5± 0.1 a	1.2± 0.2 a	110.6± 29.1 a	52.6± 16.6 a	8.0± 1.7 a	41.8± 15.8 ab	44.6± 15.4 a	38.1± 6.1 ab
3	1.5± 0.2 a	0.7± 0.1 a	88.6± 41.3 a	56.3± 29.8 a	5.0± 2.1 a	17.3± 6.5 b	51.3± 27.9 a	46.6± 6.9 ab
1	1.3± 0.2 a	1.0± 0.2 a	276.9± 74.5 a	90.9± 33.3 a	13.6± 4.1 a	155.4± 43.2 a	77.3± 29.9 a	28.1± 6.8 b

	Richness	Insect Richness	Biomass	Insect Bm	Non-Chironomid Insect Bm	Oligochaete Bm	Chironomid Bm	% Chironomid Bm
Ref	23.0± 3.8 a	16.0± 3.5 a	44.1± 17.6 a	39.8± 17.4 a	33.1± 16.9 a	3.3± 1.7 a	6.8± 2.0 a	32.5± 7.4 a
Un	14.8± 4.0 a	9.0± 2.3 a	11.4± 3.6 ab	10.3± 3.5 ab	4.9± 2.8 ab	0.8± 0.4 ab	5.4± 2.1 a	53.6± 10.1 a
9	21.0± 2.8 a	14.3± 2.9 a	38.8± 13.5 a	35.2± 13.2 a	27.7± 13.3 a	2.7± 1.5 a	7.6± 3.1 a	35.9± 8.2 a
7	14.8± 3.4 a	12.5± 2.0 a	36.2± 14.5 a	32.4± 14.6 a	26.8± 14.8 ab	2.6± 1.0 a	5.6± 2.2 a	31.3± 8.0 a
3	19.5± 2.3 a	8.5± 3.0 a	4.5± 1.9 b	3.4± 1.4 b	0.8± 0.4 b	0.2± 0.1 b	2.6± 1.0 a	57.0± 7.0 a
1	21.8± 3.2 a	13.0± 3.1 a	27.6± 11.1 ab	20.3± 10.4 ab	14.3± 10.5 ab	6.0± 2.4 a	6.0± 2.0 a	37.7± 8.3 a

Table 2.5. Annual mean±standard error of percent functional feeding group composition of macroinvertebrates by abundance (Ab) and biomass (Bm) at each site. Values with different letters are significantly different based on one-way ANOVA ($p<0.05$). Percents were arcsine square root transformed for analysis. Crayfish were excluded from biomass analyses. (CF=Collector-Filterer, CG=Collector-Gatherer, P=Predator, SC=Scaper, SH=Shredder)

	% CF Ab	% CG Ab	% P Ab	% SC Ab	% SH Ab
Ref	1.3 ± 0.6 a	87.3 ± 2.3 a	7.5 ± 1.6 ab	1.4 ± 0.7 a	0.5 ± 0.2 a
Un	1.6 ± 1.6 b	75.5 ± 7.0 a	20.9 ± 6.5 ab	1.5 ± 0.8 a	0.3 ± 0.2 a
9	0.4 ± 0.2 b	87.2 ± 2.1 a	10.3 ± 2.0 ab	0.6 ± 0.3 a	0.9 ± 0.3 a
7	1.0 ± 0.5 b	70.6 ± 6.0 a	22.3 ± 5.1 a	4.9 ± 3.1 a	0.6 ± 0.3 a
3	7.0 ± 2.2 a	82.1 ± 3.3 a	7.1 ± 1.7 b	3.2 ± 1.2 a	0.0 ± 0.0 a
1	1.0 ± 0.5 b	89.4 ± 1.9 a	8.0 ± 1.5 ab	0.4 ± 0.2 a	0.6 ± 0.3 a

	% CF Bm	% CG Bm	% P Bm	% SC Bm	% SH Bm
Ref	2.5 ± 1.7 b	63.9 ± 8.2 a	25.1 ± 8.2 ab	1.1 ± 0.6 a	7.4 ± 3.7 a
Un	2.9 ± 2.8 ab	64.1 ± 10.1 a	30.8 ± 9.5 ab	1.9 ± 0.9 a	0.3 ± 0.2 a
9	0.1 ± 0.1 b	53.3 ± 8.7 a	31.5 ± 8.8 ab	2.0 ± 1.5 a	13.1 ± 6.0 a
7	1.4 ± 0.7 ab	47.7 ± 8.7 a	44.1 ± 8.6 a	1.2 ± 0.6 a	5.6 ± 3.7 a
3	9.0 ± 2.6 a	73.8 ± 4.4 a	11.9 ± 3.6 b	5.3 ± 2.1 a	0.0 ± 0.0 a
1	3.3 ± 1.9 b	73.6 ± 6.5 a	14.8 ± 3.9 ab	1.0 ± 0.4 a	7.3 ± 4.9 a

Table 2.6. Annual mean±standard error of diversity (Shannon's H'), richness, abundance (Ab, #/meter of bank), and biomass (Bm, mg DM/meter of bank) of macroinvertebrates by habitat type. Values with different letters are significantly different based on One-way ANOVA ($p<0.05$). Abundance and biomass were log transformed, and percents were arcsine square root transformed for analysis. Crayfish were excluded from biomass analyses.

	Abundance	Insect Ab	Non-Chironomid Insect Ab	Odonate Ab	Chironomid Ab	Oligochaete Ab	% Chironomid Ab
Roots	180.5 ± 27.3 a	78.4 ± 15.3 a	14.1 ± 2.7 a	3.6 ± 1.3 a	64.3 ± 13.3 a	83.0 ± 19.8 a	34.6 ± 3.9 ab
Wood	287.2 ± 68.7 a	159.8 ± 47.3 a	14.4 ± 4.2 a	1.7 ± 0.6 a	145.4 ± 46.0 a	111.2 ± 46.5 ab	54.8 ± 10.0 ab
Mud	240.1 ± 141.9 ab	118.0 ± 71.5 ab	10.7 ± 5.9 a	1.3 ± 0.8 a	107.3 ± 65.9 ab	80.1 ± 42.6 ab	40.1 ± 13.9 ab
Rock	106.1 ± 38.8 ab	66.4 ± 27.0 ab	7.2 ± 2.7 a	0.3 ± 0.2 a	58.9 ± 24.9 b	24.3 ± 9.3 ab	57.6 ± 4.8 a
Sand	50.9 ± 17.7 b	19.3 ± 5.6 b	4.5 ± 1.2 a	0.2 ± 0.1 a	15.1 ± 5.0 b	28.8 ± 13.8 b	31.2 ± 6.9 b
Organic	206.2 ± 26.9 a	98.0 ± 16.7 a	14.2 ± 2.3 a	3.1 ± 1.0 a	83.8 ± 15.5 a	89.8 ± 18.6 a	39.4 ± 3.9 a
Inorganic	106.1 ± 30.2 b	55.9 ± 17.0 b	6.7 ± 1.6 b	0.4 ± 0.2 b	49.2 ± 15.7 b	35.4 ± 10.0 b	44.0 ± 4.5 a

	Biomass	Insect Bm	Non-Chironomid Insect Bm	Odonate Bm	Chironomid Bm	Oligochaete Bm	% Chironomid Bm
Roots	44.5 ± 10.4 a	39.8 ± 10.3 a	32.7 ± 10.3 a	24.9 ± 9.8 a	7.1 ± 1.5 a	3.6 ± 1.2 a	34.4 ± 4.8 b
Wood	30.3 ± 8.2 ab	25.7 ± 7.8 a	16.3 ± 8.4 a	12.9 ± 8.4 a	9.4 ± 2.2 a	3.4 ± 1.9 ab	30.7 ± 9.6 ab
Mud	13.9 ± 6.2 abc	10.6 ± 5.3 ab	2.1 ± 1.3 a	1.2 ± 0.9 a	8.5 ± 5.1 ab	2.4 ± 0.9 ab	43.0 ± 15.7 ab
Rock	5.8 ± 1.7 bc	4.2 ± 1.2 b	1.1 ± 0.4 a	0.2 ± 0.2 a	3.2 ± 0.9 ab	0.6 ± 0.4 b	61.8 ± 6.0 a
Sand	10.6 ± 4.7 c	8.7 ± 4.4 b	7.9 ± 4.4 a	6.4 ± 4.0 a	0.8 ± 0.2 b	1.8 ± 0.9 ab	45.9 ± 9.3 b
Organic	41.1 ± 8.1 a	36.4 ± 8.0 a	28.7 ± 8.1 a	22.0 ± 7.7 a	7.7 ± 1.3 a	3.6 ± 1.0 a	37.2 ± 4.3 a
Inorganic	9.1 ± 2.3 b	7.1 ± 2.1 b	4.0 ± 1.8 b	2.9 ± 1.7 a	3.1 ± 1.0 b	1.4 ± 0.4 b	46.4 ± 5.6 a

Table 2.7. Annual mean±standard error of percent functional feeding group composition of macroinvertebrates by abundance (Ab) and biomass (Bm) by habitat type. Values with different letters are significantly different based on one-way ANOVA ($p<0.05$). Percents were arcsine square root transformed for analysis. Crayfish were excluded from biomass analyses. (CF=Collector-Filterer, CG=Collector-Gatherer, P=Predator, SC=Scaper, SH=Shredder)

	% CF Ab		% CG Ab		% P Ab		% SC Ab		% SH Ab	
Roots	0.9 ±	0.3 a	78.3 ±	2.7 a	16.3 ±	2.4 a	2.3 ±	1.2 a	1.0 ±	0.2 a
Wood	1.0 ±	0.4 a	87.5 ±	2.9 a	9.7 ±	2.6 a	1.3 ±	0.8 a	0.2 ±	0.1 ab
Mud	1.0 ±	1.0 a	90.4 ±	3.2 a	7.2 ±	2.7 a	0.8 ±	0.5 a	0.6 ±	0.5 ab
Rock	4.5 ±	1.8 a	83.3 ±	2.7 a	8.5 ±	1.5 a	3.1 ±	1.0 a	0.1 ±	0.1 b
Sand	3.2 ±	1.9 a	82.2 ±	7.2 a	13.2 ±	6.5 a	1.0 ±	0.8 a	0.1 ±	0.1 b
Organic	0.9 ±	0.3 a	80.5 ±	2.3 a	14.7 ±	2.0 a	2.1 ±	1.0 a	0.8 ±	0.2 a
Inorganic	3.4 ±	1.1 a	84.1 ±	3.1 a	10.1 ±	2.7 a	1.9 ±	0.6 a	0.2 ±	0.1 b

	% CF Bm		% CG Bm		% P Bm		% SC Bm		% SH Bm	
Roots	0.9 ±	0.3 a	52.5 ±	5.0 a	33.2 ±	4.9 a	1.3 ±	0.6 b	12.0 ±	3.4 a
Wood	5.6 ±	2.9 a	64.4 ±	9.2 a	27.6 ±	9.8 ab	1.7 ±	0.7 ab	0.7 ±	0.4 a
Mud	1.1 ±	0.9 a	85.0 ±	6.2 a	12.0 ±	6.1 ab	1.5 ±	0.7 ab	0.5 ±	0.3 a
Rock	5.8 ±	2.3 a	75.6 ±	5.4 a	13.2 ±	4.8 b	5.1 ±	1.7 a	0.3 ±	0.3 a
Sand	4.7 ±	3.0 a	62.6 ±	10.3 a	29.8 ±	9.5 ab	1.3 ±	1.1 ab	1.7 ±	1.6 a
Organic	2.1 ±	0.8 a	55.4 ±	4.4 a	31.9 ±	4.4 a	1.4 ±	0.5 b	9.3 ±	2.7 a
Inorganic	4.6 ±	1.5 a	72.1 ±	4.9 a	19.5 ±	4.5 b	3.0 ±	0.9 a	0.9 ±	0.7 b

Table 2.8—Significant ($p<0.05$) regressions of annual mean of abundance (#/m of bank) and biomass (mg DM/m of bank) by site versus percent organic habitat. Crayfish were excluded from all biomass values for analysis.

	+/-	adj R^2	p-value
Biomass	+	0.81	0.009
Taxon Richness	+	0.86	0.004
Insect Biomass	+	0.73	0.019
Non-Chironomid Insect Abundance	+	0.76	0.014
Non-Chironomid Insect Biomass	+	0.68	0.026
Diptera Biomass	+	0.78	0.012
Chironomid Biomass	+	0.68	0.026
% Diptera Biomass	-	0.74	0.017
% Chironomid Biomass	-	0.71	0.022
Shredder Abundance	+	0.85	0.005
Collector-gatherer Biomass	+	0.85	0.005
Shredder Biomass	+	0.70	0.024
% Scraper Biomass	-	0.61	0.040

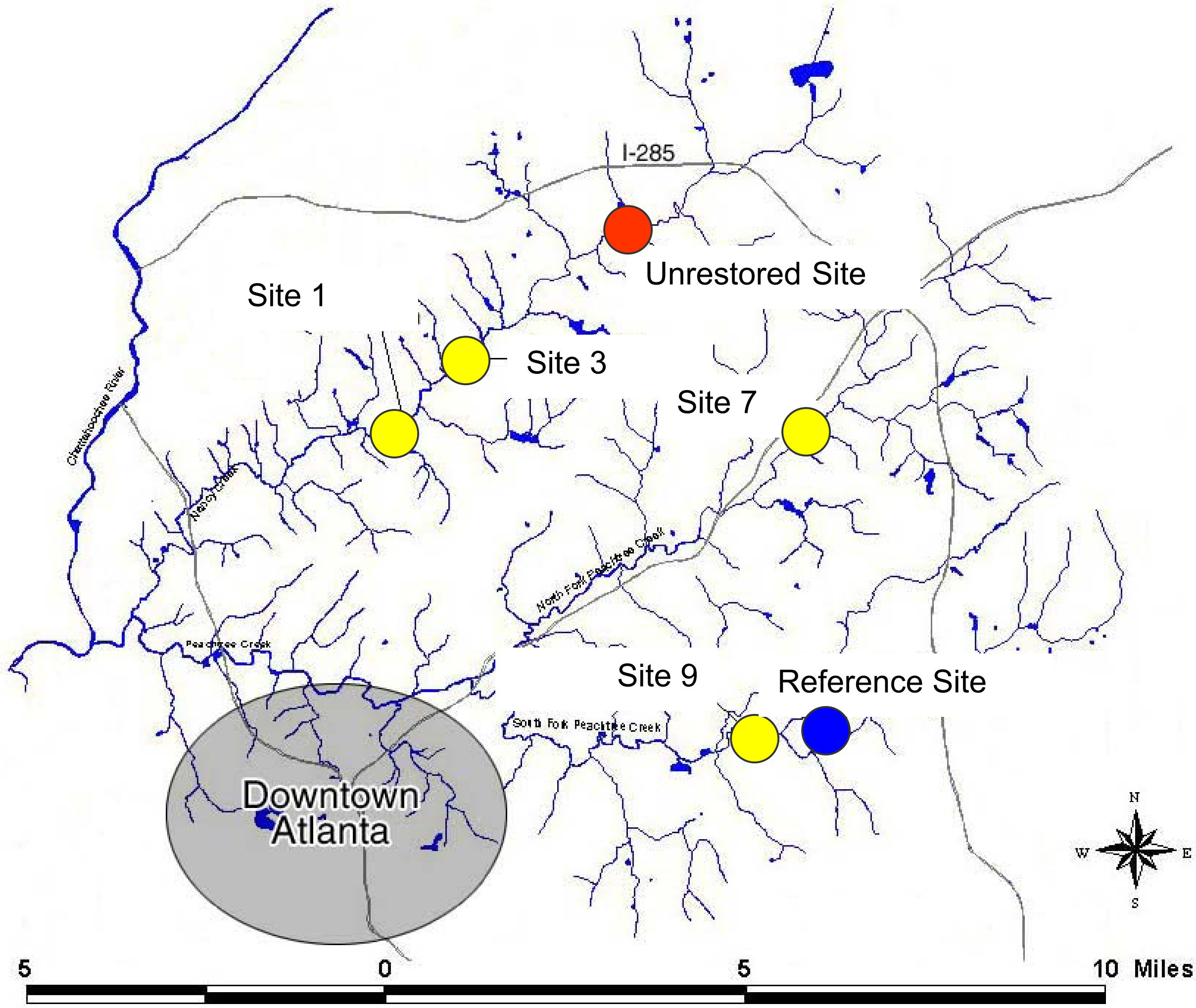

Figure 2.1. Map of Peachtree-Nancy Creek Catchment in DeKalb and Fulton Counties, GA, USA, showing location of six study sites

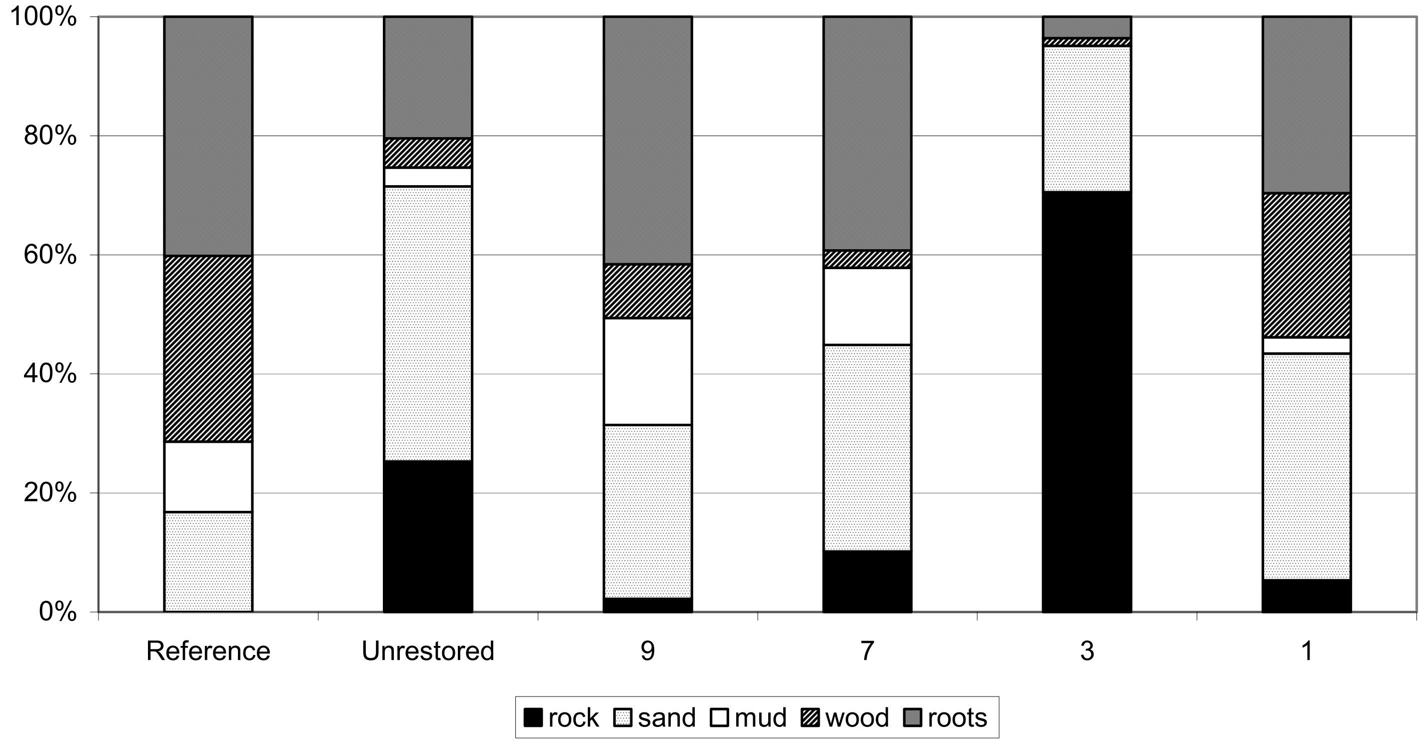

Figure 2.2—Surveyed percent composition of banks at the six study sites.

CHAPTER 3

INDIRECT EFFECTS OF BIOENGINEERED BANK STABILIZATION ON URBAN STREAMS[1]

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Abstract

Bioengineered bank stabilization can have positive effects on bank habitat and bank macroinvertebrate communities in urban streams. However, urbanization has many impacts on streams besides bank erosion, including high stormflows, high bed mobility, and low retention of allochthonous inputs. The objective of this study was to examine the effects of bioengineering on benthic macroinvertebrates and consider the possibility that bioengineering streambanks could indirectly mitigate some effects of urbanization. If bioengineering were having a strong effect on benthic macroinvertebrates, we would expect abundance, biomass, richness, and diversity to be highest at a reference site, lowest at an unrestored site, and to increase with time since restoration at bioengineered sites. These hypotheses were tested at six sites in Peachtree-Nancy Creek, an urban stream network in Atlanta, GA, USA. The benthic community did not follow the predicted pattern; abundance was highest at the reference site, but richness was actually lowest at the reference site and highest at two of the restored sites. This suggests that other factors such as particle size and bed mobility are also influencing benthic macroinvertebrate taxa richness. Stormflows were high at all sites, and several sites had very low mean particles sizes. HEC-RAS modeling indicated that streambeds would be mobilized in the 0.5-year stormflow at all sites. We expected that increased organic bank habitat, canopy cover, and allochthonous input from bioengineering would increase benthic organic matter levels (BOM) and that this would be beneficial to benthic macroinvertebrates. Although there were few differences in BOM among sites, there was a strong correlation between percent organic bank habitat and benthic macroinvertebrate abundance and biomass. Organic bank habitat appears to serve as a refuge from high peak stormflows, so that sites with high amounts of this habitat have a larger population to recolonize the benthos following storm events.

Introduction

There are many pathways by which urbanization affects stream ecosystems and few easy answers for how to mitigate these effects (Baer and Pringle 2000, Paul and Meyer 2001). Increased impervious surface cover increases runoff, leading to higher peak stormflows (Dunne and Leopold 1978), which can increase channel erosion and downstream redeposition of eroded sediment (Trimble 1997). Clearing of both upland and riparian vegetation increases surface erosion, leading to increased fine sediments in stream channels (Wolman 1967). The combined effects of increased stormflows and increased benthic fine sediment lead to much greater bed mobility than is seen in less impacted streams (Bledsoe and Watson 2001).

Urbanization can also have more direct effects upon the biological communities in streams. Fish and macroinvertebrate communities can be altered by poor water quality, loss of habitat through the previously mentioned channel changes, and changes in food resource availability (Paul and Meyer 2001). Stream food webs are based on both instream primary production and input of organic matter from outside of the stream (Allan 1995). Primary production may be reduced by increased bed mobility and turbidity, or it may be increased by elimination of riparian shading and increased nutrient loading (Dodds et al. 2000). Allochthonous input may decrease due to loss of riparian vegetation, and retention of any organic material that does enter the stream may decrease due to loss of instream structure (e.g., woody debris) and increased bed mobility (Paul 1999). Human wastewater inputs can greatly increase the levels of fine organic matter and dissolved organic carbon in urban streams, particularly following storms in cities with combined sewer overflows (McConnell 1980).

There is much debate about the best methods for mitigating these effects of urbanization on streams, including whether it is possible at all. Since many of the problems can be traced back to increased stormflows, some argue that the emphasis should be placed on stormwater mitigation (Booth and Jackson 1997). Others argue that the most efficient management decision is a restructuring of the channel to make it more stable and better able to carry the current flow regime without eroding (e.g., Doll et al. 2003). Both stormwater management and channel restructuring are large-scale and expensive, whereas other approaches to stream enhancement are done at a smaller scale. Many of these practices, such as grade control structures, flow deflection structures, and bank stabilization practices, have been shown to be effective at meeting goals such as preventing channel degradation, creating pools, or stabilizing banks (Brown 2000), but their ecological effects have received less study. Many municipalities and community groups are focused on smaller scale projects, including nonstructural streambank stabilization, or bioengineering, to repair eroded banks. These practices are chosen as an alternative to more traditional structural practices, such as concrete and riprap, for aesthetic reasons, as well as in the hope of providing some ecological benefit to the stream (Li and Eddleman 2002).

The most obvious possible ecological benefit of bioengineering would be new habitat created through establishment of vegetation on the banks. In Chapter 2, we examined the direct effects that bioengineering can have on bank habitat and the bank macroinvertebrate community. Bioengineering could have other indirect benefits for streams, although restabilizing and revegetating a small reach of stream may not mitigate for whole catchment effects. Reduced bank erosion could decrease fines entering streams, although the direct input from one bank area is probably small compared to the sediment export of the entire catchment. Increased riparian vegetation could increase allochthonous inputs; however, it is unlikely to increase retention of organic matter if the streambed is still highly mobile. Woody debris inputs from mature riparian vegetation could help stabilize the streambed and provide more organic retention, though the wood itself might not be retained under high peak stormflow conditions. These possible indirect benefits have been discussed but they have not yet been quantified.

By studying stream reaches with bioengineered bank stabilization projects of different ages, and comparing them to unrestored and reference reaches in the same urban catchment, we examined whether this practice could mitigate effects of urbanization, particularly poor retention of allochthonous inputs and high bed mobility. We predicted that benthic organic matter would be highest at the reference site, lowest at the unrestored site, and increase with time since restoration at the bioengineered sites. Benthic organic matter standing crop, percent organic habitat on the banks, and percent canopy cover could all have been increased by successfully executed bioengineered bank stabilization and we expected to see a clear correlation between them. Bioengineered bank stabilization could also have some positive benefits for the benthic macroinvertebrate community; benthic macroinvertebrate abundance, biomass, diversity, and taxon richness would be expected to be greatest at the reference site, least at the unrestored site, and increasing with time since restoration at the bioengineering sites. Because bioengineering has a clear direct mechanism for benefiting bank macroinvertebrates, we hypothesized that abundance, biomass, richness, and diversity would all be significantly higher on the banks than in the benthos. We predicted that differences among macroinvertebrate communities among the sites would be driven primarily by particle size, stormflow discharge, and bed mobility, but that benthic organic matter standing crops, percent canopy cover, and percent organic bank habitat would also be important. Many studies have suggested that stormflows and sedimentation are the most severe effects of urbanization on streams; our goal was to determine the extent to which these impacts can be reduced by bioengineered bank stabilization projects.

Methods

Study Sites

The Peachtree-Nancy Creek catchment in Atlanta, GA, USA was chosen for this study, as it is a large urban catchment in which bioengineering and other restoration practices are becoming common. We chose four sites where bioengineering had been completed (Table 1); for ease of reference, sites will be referred to by their time since restoration (e.g., Site 9 was restored 9 years ago). These were compared to an unrestored reach that was similar to the prerestoration condition of these sites and a reference reach that had a large wooded riparian area and relatively stable banks and was thus an example of a likely goal of bioengineering. Although the bioengineering sites varied in length, at each site the bottom 150-m reach of stream was used for sampling. Sampling occurred at baseflow in July and November 2002, and June 2003. For a more detailed description of these sites, see Chapter 2. For photographs of sites, see Appendix A.

Geomorphic, Hydrologic, and Chemical Methods

Discharge was measured at each site on each sampling date using a Flo-Mate Model 2000 Portable Flowmeter (Marsh-McBirney, Inc.). All sites were surveyed at baseflow using a Sikka self-leveling level and stadia rod. A typical cross-section was surveyed at each site. Longitudinal slope was measured along tops of riffles, except at Site 3, which lacked riffles, where top of water slope was measured at 35-m intervals through the reach. Samples of riffle and non-riffle bed material were collected throughout each reach. The material was dried and shaken through sieves, and the sorted sediment fractions were weighed in phi categories: <250 μm, <500 μm, <1 mm, <2 mm, <4 mm, <8 mm, <16 mm, <32 mm, <64 mm. These categories were plotted by percent composition to obtain the 16th, 50th, and 84th percentile particle size (d16, d50, and d84). These three values were averaged to calculate the mean particle size.

Catchment area was obtained from USGS gaging stations located at Sites 1 (02336360) and 9 (02336180). For other sites, catchment area was calculated from topographic maps and confirmed by comparison to sites 1 and 9. Peak discharge for 0.5. 2, 5, 10, 25, 50, 100, 200, and 500-year recurrence interval storms was calculated from catchment area using published flood frequency relations for urban streams in Georgia (Region 1, 26% impervious surface cover) (Inman 1995).

The survey data, peak and baseflow discharges, and the sediment profile were entered into the hydrologic modeling program HEC-RAS version 3.1.1 (US Army Corps of Engineers) in order to model baseflow and peak discharges. HEC-RAS calculated stage, mean velocity, and bedload transport for each discharge level. Mean velocities from each discharge were used to calculate velocity near the benthic or bank surface for each site at each level of discharge (benthic velocity =0.7*mean velocity). Mean particle size was used to calculate the critical velocity for each site (critical velocity =0.155*square root of mean particle size). The ratio of benthic velocity to critical velocity is a measure of bed mobility (Gordon et al. 1992 eq 7.13-7.15).

On each site visit, turbidity was measured using a Hach 2100P turbidimeter and conductivity was measured using an Orion Model 130 conductivity meter. Water samples were collected at baseflow in February and June 2003, filtered in the field, and brought on ice back to the laboratory where they were frozen for later analyses. Analysis of ammonium, nitrate, and total phosphorus were done using an Alpkem autoanalyzer by the University of Georgia Institute of Ecology Chemical Analysis Laboratory following Standard Methods protocol. Spherical densiometer readings were taken in February and July 2003 at all sites to estimate winter and summer canopy cover.

Macroinvertebrate Sampling

Benthic macroinvertebrate sampling was done in July and November 2002 and June 2003 with a 22-cm diameter core sampler. Four random points were chosen in each stream reach on each sampling date, and each sample point was classified as riffle, run, or pool and as silt, sand, gravel, cobble, or rock. A 10-cm deep core was removed, placed on ice, and taken to the lab for sorting. Core samples were washed through 1 mm and 250 µm sieves and elutriated to remove organics and macroinvertebrates. The fractions >1 mm and 250 µm-1 mm were preserved in ethanol for later removal of macroinvertebrates. Subsamples of the <250 µm and the 250 µm-1 mm size fractions were filtered, dried, ashed, and weighed to measure very fine benthic organic matter (VFBOM) and fine benthic organic matter (FBOM). Both >1mm and 250 µm-1 mm samples were dyed with Phloxine B and picked under a dissecting microscope at 15x. After macroinvertebrates were removed from the >1 mm fraction, the remaining material was dried, ashed, and weighed for measurement of coarse benthic organic matter (CBOM).

All macroinvertebrates were measured and identified to genus wherever possible using Merritt and Cummins (1996), Peckarsky et al. (1990), and Thorp and Covich (1991). Chironomidae were identified as Tanypodinae or non-Tanypodinae. Biomass was calculated using length-mass regressions (Benke et al. 1999). Functional feeding groups were assigned based on Merritt and Cummins (1996) for insects and Thorp and Covich (1991) for non-insects. Because they are large and not present at all sites, *Corbicula* were excluded from biomass and abundance calculations.

Data Analysis

Data were analyzed using JMP version 5.0.1 (SAS Institute Inc, Cary NC). Diversity was calculated using Shannon's H' for each site in each season and then averaged over the year (Washington 1984). Total richness and insect richness were calculated for each site in each season and then averaged over the year. One-way and two-way ANOVA's were used with an alpha value of 0.05. Abundance and biomass data were log transformed for analysis. Percent data were arcsine square root transformed for analysis. Correlations were calculated using annual means of macroinvertebrate data versus environmental variables.

Results

Physical and Chemical Characteristics

All sites had fairly low slope—0.001 or 0.002—except Site 7, which also has the smallest catchment and lowest baseflow discharge (Table 1). The reference site and Site 9 had a surprisingly finer sediment profile than several of the restoration sites (Table 2, Appendix D). Several of the sites had more cobble and gravel substrate, which should provide better habitat and is less likely to be transported in storm events.

Hydrologic modeling reveals that all of these sites experience high stormflows at even the 0.5-year recurrence interval (Table 3, Appendix E). Peachtree Creek is known to have stormflows 30-100% greater than other Georgia Piedmont streams of similar size (Rose and Peters 2001). Velocities near the streambed get very high during stormflows, but velocities near the surface of the banks can be much lower (Table 4). Bed mobility > 1 indicates that the velocity on the bed is greater than the critical velocity for movement of the mean particle size. At all sites bed mobility was fairly high at baseflow; at Site 9, the bed velocity is high enough to move the mean particle size at baseflow (Table 5). In the 0.5-year or higher stormflow, it is apparent that the bed mobilization occurs at all sites.

There were no significant differences in baseflow turbidity (Table 6). Site 7 had significantly higher conductivity than any of the other sites. Site 7 also had much higher ammonium and nitrate levels than any of the other sites. Total phosphorus was not very different among sites. It is possible that differences in water chemistry might be more apparent during stormflows, but the quantity of water in these streams at stormflow made it very hard to sample.

Benthic Organic Matter

Standing crop of benthic organic matter (BOM) differed little among sites, with the range of mean values from 169.2 to 461.8 g AFDM/m^2 (Table 7). Site 7 was significantly higher than the reference and unrestored sites in FBOM. Sites 9 and 1 were significantly higher than all sites except the unrestored site in CBOM levels. Total BOM was significantly higher in July 2002 than in June 2003, but there was no obvious seasonal trend (such as a peak in BOM level at leaffall). Very few correlations were seen between BOM and environmental variables. Baseflow bed mobility and FBOM+VFBOM were inversely related ($r = -0.815$, $p = 0.045$). Contrary to expectations, no significant relationship existed between percent organic bank habitat and BOM. Percent canopy cover in winter and summer was highest at the reference site and Site 9 (Table 1). The only significant correlation between percent canopy cover and BOM was a negative correlation with FBOM ($r = -0.8593$, $p = 0.028$) and FBOM+VFBOM ($r = -0.9321$, $p = 0.007$). Large woody debris in the streambed was present only at the reference site and was not measured; this would have increased the total amount of BOM present at the reference site.

Benthic Macroinvertebrates

The most common benthic taxa by abundance were chironomids, oligochaetes, *Corbicula*, ceratopogonids, and Tanypodinae chironomids. The most common benthic taxa by biomass were *Corbicula*, chironomids, oligochaetes, gastropods, and ceratopogonids. Hydropsychid caddisflies and baetid mayflies were the only somewhat sensitive taxa commonly found in the benthos; both were found at least once at each site. One tipulid and one gomphid were found in July 2002, and one elmid larva was found in June 2003 at the reference site.

Diversity of benthic macroinvertebrates was highest at Sites 1 and 3 and lowest at the reference site ($p<0.05$) (Table 8). Overall taxon richness, insect diversity, and insect taxon richness followed this pattern as well, but the differences were not significant. Total macroinvertebrate abundance was highest at the reference site and significantly lower at Site 3. There were no significant differences among sites in biomass once *Corbicula* were excluded. *Corbicula* had highest abundance at Site 1 and Site 3 but highest biomass only at Site 1 ($p<0.05$) (Table 9). Insect abundance was highest at the reference site and Site 9, where abundance was significantly higher than at Sites 7 and 3. Insects made up greater than 85% of the abundance at the reference site, the unrestored site, and Site 9; this was significantly higher than all of the other sites. This difference in insect abundance was driven by chironomids, although ceratopogonid midges were also very common at some sites. Oligochaetes made up more than 40% of the benthic community at Site 7, both by abundance and biomass. The benthic community at all of these sites was dominated by collector-gatherers (Table 10). Predators, primarily ceratopogonids, made up a large part of the population by both abundance and biomass at Sites 1 and 3. Some collector-filterers and scrapers were also found at all sites, but never in large numbers. Shredders were absent from the benthos at all of these sites.

Although total abundance and biomass of benthic macroinvertebrates at all sites was significantly higher in July 2002 (at the end of a very long drought) than in November 2002 or June 2003 (during a very wet year), two-way ANOVA's revealed no significant interaction between season and site. We also considered differences in macroinvertebrate communities on different benthic habitats and in different flow types to see if they could be driving differences among the sites. Very few significant differences among habitat types or flow types were present (Appendix C). The benthic macroinvertebrate community was correlated with a few environmental variables. Baseflow width was a good predictor for both total richness ($r = 0.94$, $p<0.01$) and insect richness ($r = 0.91$, $p<0.01$). Total abundance was inversely correlated with mean particle size ($r = -0.88$, $p<0.05$). There was also a positive relationship between total BOM and benthic richness ($r = 0.85$, $p< 0.05$).

Discussion

Differences in Bank and Benthic Macroinvertebrate Communities

The bank macroinvertebrate community at these sites was described in Chapter 2. Overall, benthic abundance per square meter of stream was around 20 times greater than bank abundance per meter of streambank. Benthic biomass was 10-20 times greater than bank biomass. Considering that these streams were several meters wide, it's clear that far more macroinvertebrates live within the streambed sediments than on the banks. However, the community composition in the two habitats is quite different. Richness and diversity are much higher on the banks, both overall and when only insects are considered. The biggest difference between the bank and benthic macroinvertebrate communities is the absence of some taxa in the benthos. The most obvious absences are odonates and all shredders. Although shredders were never very abundant on the banks, they composed up to 13% of the biomass, with most of that biomass made of large tipulid larvae; only one tipulid was found in the benthos on one sample day, and it was a predator, not a shredder. Similarly, although odonates generally made up only 1 or 2% of the abundance on the banks, they composed up to a quarter of the biomass. The bank samples contained twelve different genera of odonate in five different families, and all sites had at least one odonate. In contrast, only one individual odonate, *Progomphus*, was found in the benthos; it only occurred at the reference site and it only occurred in July 2002.

In a reference stream at the Coweeta Hydrologic Laboratory in North Carolina, benthic macroinvertebrate abundances and biomasses were 10 to 100 times greater than those in Peachtree-Nancy Creek (Wallace et al. 1999). In impacted streams north of Atlanta in the Etowah basin, both bank and benthic abundances were similar to those in Peachtree-Nancy Creek but both benthic and bank insect richness were on average twice as high in the Etowah (Roy et al. 2003). In the subset of the Etowah sites identified as sediment-impacted, the benthic and bank macroinvertebrate abundance and richness were similar to those in this study.

The pattern seen here of significantly higher abundance in the benthos and significantly higher richness on the banks, mirrors what is commonly seen in sandy-bottomed blackwater Coastal Plain streams, where woody debris provides the stable habitat. In the Satilla River in south Georgia, diversity is much higher on snag habitats and includes caddisflies, black flies, mayflies, dragonflies, and beetles, all of which were present in bank community in this study, as well as hellgrammites and stoneflies, which were absent from Peachtree-Nancy Creek (Benke et al. 1984). As in this study, the benthic community in the Satilla was mostly chironomids, oligochaetes, and ceratopogonids, with an occasional dragonfly. However, their density was much higher in the Satilla, with total density sometimes greater than 20,000/m^2. In the Satilla, the snag community was dominated by collector-filterers, while in Peachtree-Nancy Creek both the bank and benthic communities were overwhelmingly dominated by collector-gatherers, with very few filterers present, except *Corbicula*. Although there are interesting parallels between a naturally sandy stream and these streams impacted by urbanization, differences in hydrology, water quality, and food abundance have led to several differences in community structure.

The correlation of percent organic bank habitat with benthic abundance and biomass suggests that bioengineering can have an impact on the benthos as well as directly on bank macroinvertebrates (Table 11). This is a rather surprising result, as it might not be expected that such a small-scale practice on the banks would have a clear effect in the streambed of these streams that have so many impacts from urbanization. The effects of bank bioengineering on the benthic macroinvertebrate community are likely due to banks serving as a refuge from high velocities and bed mobilities during storm events rather than some direct effect of the banks on the benthos. More stable banks probably provide more colonists for the benthos following a storm event.

Particle Size and Bed Mobility

The strong negative correlation that we found between mean particle size and bank insect richness confirms what Roy et al. (2003) concluded from their study: poor benthic habitat can actually result in higher richness of bank taxa as benthic taxa look for more stable habitats. The strong negative correlation between mean benthic particle size and benthic abundance is surprising at first, but easily explained when the percent composition of the benthic community is examined. At the high benthic abundance sites, up to 97% of the benthos was small chironomids, which flourish in fine sediments. The fact that benthic biomass was not negatively related to mean particle size suggests that finer sediments provide conditions under which many small chironomids can live, but a more diverse community of larger individuals cannot.

The critical velocity at which macroinvertebrates are transported may be even lower than the critical velocity we used for sediment, suggesting that some macroinvertebrates on the benthic surface may be transported even before the bed begins to move. Laboratory measures of critical velocities for macroinvertebrates range from as high as 2.4 m/s for limpets or 1.79 m/s for Ephemeroptera to as low as 0.3 m/s for amphipods (Statzner et al. 1988). The mean velocity of the overbank flow calculated for all stormflows is below 1 m/s (Table 4), suggesting that the bank could indeed be a refuge for macroinvertebrates in stormflow, assuming the bank vegetation were sufficient to prevent mobilization of the bank sediments. It is likely that benthic macroinvertebrates are washed downstream as the bed is mobilized in each storm event, whereas many bank macroinvertebrates are able to remain.

Benthic Organic Matter

It should be noted that BOM levels were significantly higher on the first sample day than on the subsequent days (see Appendix C). A multi-year drought ended after the first samples were taken, and an extremely wet year followed. The November 2002 and June 2003 samples had lower BOM because high bed mobility during multiple storms and lack of instream structure such as wood or even rocks leads to decreased BOM retention.

BOM values in Peachtree and Nancy Creeks are extremely low when compared to streams in less impacted catchments. Reported BOM storage in streams throughout North America range from 20 g BOM/m^2 in an Alaskan stream to 35,000 g BOM/m^2 in a stream in Oregon (Jones 1997). BOM values for Southeastern streams range from 600 to 10,500 g BOM/m^2 (Jones 1997). These numbers include the large amounts of wood generally found in Southeastern streams; woody debris was only found at the reference site and was not measured in this study. In addition, because the organic matter in this study was stored in 70% ethanol for

6 months to a year, some loss of mass would be expected. Macroinvertebrates stored in 70% ethanol for this period of time generally have a 15-30% dry mass loss (Leuven et al. 1985) and a similar pattern could be expected in preserved organic matter. In spite of this, it still seems clear, however, that the amounts of BOM at these sites are quite low and that this is due to a combination of low inputs and low retention.

After four years of excluding all inputs of leaf litter, BOM in a forest stream was still nearly twice that found in the Peachtree-Nancy catchment (Wallace et al. 1999). In that stream, macroinvertebrate abundance and biomass levels are 5 to 20 times greater than the streams in this study. Of particular interest is the persistence of shredders in the litter exclusion stream while no benthic shredders were found in this study. There were shredders on the banks of the streams (Chapter 2), but they were all tipulids. The streams at Coweeta feature many shredders of the orders Plecoptera and Trichoptera, which are considered intolerant to organic pollution and are rarely found in urban streams. The absence of benthic shredders in these streams is likely due to a combination of food limitation, poor habitat, and poor water quality.

Implications for Restoration Practices

Laboratory studies have shown that the availability of refugia is the most important factor in the persistence of macroinvertebrates in sandy streams (Borchardt 1993). Although stream margins may be adequate refugia for many macroinvertebrates, some benthic macroinvertebrates are not mobile enough to utilize them (Borchardt 1993). Therefore attempts to restore sandy bottom streams should address instream refugia for benthic macroinvertebrates (Borchardt 1993). An experimental stream restoration added large woody debris (LWD) to a sand-bed stream in Mississippi, increasing the total amount of wood by an order of magnitude, in an effort to improve fish habitat in an incised reach of stream which had shallow water, shifting sand, few pools, and no shelter for fish (Shields et al. 2003). This wood addition increased flow resistance, moderated velocities, and increased retention time of a dye tracer. In Peachtree and Nancy Creeks, LWD was only present in the channel at the reference site, probably due to a combination of lack of input from the riparian zone and lack of retention due to extremely high stormflows. If high channel velocities and low BOM retention are the main problems in this catchment, addition of LWD in the channel might complement bioengineered bank stabilization, if it were well anchored against the stormflows. However, in a similar study in a more urban system in the Pacific Northwest, the only habitat improvement seen with LWD addition was an increase in the number of pools, with no subsequent biological improvements (Larson et al. 2001). These authors suggest LWD addition is only appropriate when lack of LWD is the most obvious cause of the problem (i.e., when management practices included removing wood from the stream). In general, their recommendation is to look for the cause of impairment to a stream, and correct it. In the Peachtree-Nancy Creek catchment, stormwater management would reduce the severe peak flows, which may be the root source of the high bed mobility and lack of BOM in these streams.

Conclusions

This study evaluated the extent to which bioengineered streambank stabilization mitigated the effects of urbanization, in this case hydrologic and geomorphic changes and loss of BOM retention. Although we predicted that benthic organic matter would be highest at the reference site, lowest at the unrestored site, and increase with time since restoration at the bioengineered sites, there was no significant difference among sites in total BOM retention, and the slight differences in retention of FBOM and CBOM showed no clear pattern. Because bioengineering should increase riparian and bank vegetation, leading to increased allochthonous input, a clear correlation among benthic organic matter standing stock, percent organic habitat on the banks, and percent canopy cover would be expected. Yet there was very little correlation, suggesting that the most obvious connection between bioengineering and BOM is not significant, probably because of a lack of retention.

We hypothesized that bioengineered bank stabilization could have some positive benefits for the benthic macroinvertebrate community and that benthic macroinvertebrate abundance, biomass, diversity, and taxon richness would be greatest at the reference site, least at the unrestored site, and increase with time since restoration at the bioengineered sites. The reference site indeed had the highest abundance, but it was dominated by chironomids and in fact had the lowest richness; highest richness was at the two most recently restored sites. Taxa richness and insect richness were much higher on the banks than in the benthos, but benthic abundance and biomass were much higher than on the banks, owing to the very large numbers of chironomids and oligochaetes. We expected that lower particle size, higher stormflow discharge, and higher bed mobility, would lead to lower benthic abundance, biomass, and richness. The opposite proved to be true: there was a significant inverse relationship between mean particle size and benthic abundance. Benthic richness was best predicted by wetted channel width at baseflow. However, the strongest predictor of both bank and benthic macroinvertebrate abundance and biomass was percent organic bank habitat. This is a surprising result, particularly since there was no direct correlation between bioengineering and BOM levels. It is likely that organic bank habitat provides the best refuge from peak stormflows, allowing more rapid recolonization of the benthos following storms. Our results indicate that bioengineering does have positive effects on both benthic and bank macroinvertebrate communities; however, bioengineering cannot mitigate urbanization's myriad of effects.

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Table 3.1. Site descriptions of six study sites in the Peachtree-Nancy Creek catchment, Atlanta, GA, USA including physical characteristics.

	Reference	Unrestored	Site 9	Site 7	Site 3	Site 1
Years Since Restoration	--	--	9	7	3	1
Length of Bioengineering (m)	--	--	330	200	300	250
Site Location	S. Peachtree Creek Nature Preserve	Murphey Candler Park	Medlock Park	Dresden Park	Starlight Neighborhood	Blue Heron Nature Preserve
Stream	S. Peachtree Creek	Nancy Creek	S. Peachtree Creek	N. Peachtree Creek	Nancy Creek	Nancy Creek
Catchment area (km2)	24	36	27	8	57	65
Baseflow (m3/s)	0.23	0.49	0.25	0.03	0.79	0.95
Wetted width (m)	6.01	6.74	7.28	4.86	9.05	11.43
Slope	0.001	0.002	0.001	0.007	0.001	0.001
Winter % Canopy Cover	70	43	73	27	30	35
Summer % Canopy Cover	88	47	91	77	49	43
% Organic Bank Habitat	71.4	25.4	50.6	42.1	4.9	53.9

Table 3.2. Particle sizes (mm) at the 16th, 50th, and 84th percentiles overall and in riffles only for each site. Riffles were absent from Site 3. Mean particle size is the average of d16, d50, and d84.

	Ref	Un	9	7	3	1
d16	0.25	0.20	0.23	0.23	0.25	0.25
d50	0.75	2.00	0.50	0.90	2.00	1.75
d84	3.70	16.00	1.00	12.00	20.00	16.00
mean particle	1.57	6.07	0.58	4.38	7.42	6.00
d16-riffle	0.28	1.00	0.20	0.25	--	0.75
d50-riffle	1.50	8.00	0.40	2.00	--	12.00
d84-riffle	6.00	32.00	1.50	16.00	--	24.00
riffle mean particle	2.59	13.67	0.70	6.08	--	12.25

Table 3.3. Baseflow and calculated peak discharges (m^3/s) for the study sites. Peak discharge values were calculated using catchment area and 26% impervious surface cover following published equations for urban streams in Region 1 of Georgia (Inman 1994)

Recurrence Interval	Ref	Un	9	7	3	1
Baseflow	0.23	0.49	0.25	0.03	0.79	0.95
0.5-year	18.0	23.5	19.0	7.0	32.5	37.0
2-year	69.7	93.8	74.8	29.0	128.1	144.0
5-year	113.9	151.3	121.8	49.1	204.1	228.4
10-year	132.5	176.7	141.8	56.4	239.3	268.2
25-year	143.5	190.6	153.4	61.8	257.1	287.6
50-year	160.3	212.1	171.2	69.9	284.8	318.2
100-year	190.0	251.3	202.9	82.8	337.5	377.0
200-year	203.6	268.3	217.3	89.8	358.8	400.1
500-year	227.4	299.5	242.6	100.3	400.6	446.7

Table 3.4. Velocity (m/s) near the benthic and bank surface at baseflow and during floods with various recurrence intervals. Values were calculated by HEC-RAS using surveyed cross-sections and slope and calculated recurrence interval floods from Table 3. (LB=Velocity at surface of left bank; MC=Velocity at benthic surface in main channel; RB=Velocity at surface of right bank)

Recurrence	Reference			Unrestored			Site 9			Site 7			Site 3			Site 1		
Interval	LB	MC	RB	LB	MC	RB	LB	MC	RB	LB	MC	RB	LB	MC	RB	LB	MC	RB
baseflow		0.18	0.02		0.29			0.15	0.01		0.21			0.20	0.04	0.04	0.22	
0.5-year	0.15	0.82	0.11	0.25	0.95	0.19	0.18	0.97	0.14	0.34	0.85	0.04	0.27	0.75	0.32	0.27	0.83	0.27
2-year	0.12	1.39	0.18	0.38	1.46	0.36	0.35	1.37	0.29	0.39	1.40	0.06	0.50	1.24	0.54	0.27	1.39	0.20
5-year	0.27	1.65	0.22	0.49	1.86	0.50	0.53	1.73	0.44	0.63	1.67	0.07	0.33	1.55	0.43	0.50	1.60	0.47
10-year	0.24	1.59	0.18	0.46	1.76	0.47	0.48	1.63	0.41	0.57	1.61	0.07	0.61	1.48	0.34	0.45	1.54	0.41
25-year	0.28	1.69	0.24	0.51	1.93	0.53	0.55	1.78	0.46	0.67	1.72	0.08	0.36	1.58	0.46	0.53	1.62	0.50
50-year	0.31	1.74	0.26	0.53	2.00	0.55	0.59	1.86	0.50	0.71	1.79	0.08	0.41	1.61	0.50	0.57	1.66	0.54
100-year	0.34	1.82	0.29	0.30	2.13	0.60	0.64	1.99	0.55	0.78	1.88	0.08	0.48	1.67	0.57	0.63	1.73	0.60
200-year	0.36	1.86	0.31	0.35	2.18	0.61	0.67	2.04	0.57	0.82	1.92	0.08	0.50	1.70	0.59	0.65	1.75	0.62
500-year	0.38	1.91	0.34	0.42	2.25	0.25	0.71	2.14	0.61	0.87	1.99	0.08	0.55	1.74	0.63	0.69	1.80	0.67

Table 3.5. Bed mobility at baseflow and for floods of various recurrence intervals at the study sites. Bed mobility is the ratio of benthic velocity (see Table 4) and critical velocity (calculated from mean particle size). Bed mobility > 1 means the mean particle size sediment is likely to be mobilized.

	Ref	Un	9	7	3	1
baseflow	0.9	0.8	1.3	0.6	0.5	0.6
0.5	4.2	2.5	8.3	2.6	1.8	2.2
2	7.1	3.8	11.6	4.3	2.9	3.7
5	8.5	4.9	14.7	5.2	3.7	4.2
10	8.2	4.6	13.9	5.0	3.5	4.1
25	8.7	5.0	15.1	5.3	3.7	4.3
50	8.9	5.2	15.8	5.5	3.8	4.4
100	9.4	5.6	16.9	5.8	4.0	4.6
200	9.6	5.7	17.4	5.9	4.0	4.6
500	9.9	5.9	18.2	6.1	4.1	4.7

Table 3.6. Average water chemistry values for the year. n=5 for turbidity and conductivity. n=2 for nutrients.

	Ref	Un	9	7	3	1
Turbidity (NTU)	11.4	16.7	13.9	7.5	11.6	14.1
Conductivity (µS/cm)	103.3	92.4	106.3	242.3	90.7	95.6
NH_4-N (µg/L)	613	437	835	1232	211	550
NO_3-N (µg/L)	43	31	38	849	29	42
Total P (µg/L)	15	9	19	16	13	16

Table 3.7. Annual mean±standard error for benthic organic matter (g AFDM/m^2) at the study sites (n=12). VFBOM is < 250 µm. FBOM is 250 µm-1 mm. CBOM is > 1 mm. Values with different letters are significantly different based on one-way ANOVA with alpha of 0.05.

	VFBOM	FBOM	CBOM	Total BOM
Ref	53.7 ± 12.9 a	16.4 ± 3.9 b	115.4 ± 86.7 bc	185.5 ± 90.2 a
Un	94.5 ± 35.2 a	22.9 ± 4.4 b	51.9 ± 8.5 ab	169.2 ± 42.3 a
9	35.8 ± 6.0 a	22.4 ± 2.9 ab	282.9 ± 72.1 a	341.1 ± 73.3 a
7	50.2 ± 7.0 a	218.2 ± 98.8 a	15.7 ± 4.3 c	284.1 ± 101.7 a
3	133.6 ± 73.4 a	98.6 ± 43.4 ab	80.0 ± 29.1 bc	312.2 ± 107.2 a
1	27.0 ± 3.5 a	91.6 ± 69.1 ab	343.2 ± 128.7 a	461.8 ± 135.6 a

Table 3.8. Annual mean±standard error of diversity (Shannon's H'), richness, abundance (#/m^2), and biomass (mg DM/m^2) of macroinvertebrates at each site. Values with different letters are significantly different based on one-way ANOVA ($p<0.05$). Abundance and biomass were log transformed for analysis. *Corbicula* were excluded from total abundance and biomass analyses.

	Abundance		Diversity		Insect Ab		Insect Diversity	
Ref	7195.1 ±	1777.4 a	0.31 ±	0.05 c	6875.0 ±	1680.3 a	0.12 ±	0.03 a
Un	2929.9 ±	895.5 ab	0.71 ±	0.11 b	2729.2 ±	880.2 ab	0.39 ±	0.10 a
9	5630.7 ±	1346.3 ab	0.39 ±	0.05 bc	5416.7 ±	1302.5 a	0.20 ±	0.07 a
7	3121.2 ±	1027.3 ab	0.79 ±	0.06 b	1858.0 ±	697.5 b	0.42 ±	0.24 a
3	1918.6 ±	595.8 b	1.34 ±	0.04 a	1585.2 ±	554.9 b	0.62 ±	0.17 a
1	4072.0 ±	1426.2 ab	1.37 ±	0.14 a	3522.7 ±	1277.7 ab	0.63 ±	0.13 a

	Biomass		Richness		Insect Bm		Insect Richness	
Ref	386.2 ±	145.3 a	8.33 ±	2.40 a	341.6 ±	134.2 a	5.00 ±	1.15 a
Un	120.5 ±	27.3 a	8.67 ±	3.48 a	95.7 ±	25.1 a	4.33 ±	1.86 a
9	316.9 ±	110.1 a	9.67 ±	3.18 a	296.6 ±	107.3 a	5.00 ±	2.08 a
7	258.2 ±	64.8 a	9.00 ±	2.08 a	153.7 ±	49.1 a	4.00 ±	1.15 a
3	161.5 ±	64.3 a	11.33 ±	3.28 a	122.6 ±	50.0 a	5.33 ±	1.76 a
1	421.2 ±	131.0 a	12.67 ±	3.93 a	238.7 ±	88.6 a	6.00 ±	2.08 a

Table 3.9. Annual mean±standard error of abundance (#/m^2), and biomass (mg DM/m^2) of macroinvertebrates at each of the six study sites. Values with different letters are significantly different based on one-way ANOVA ($p<0.05$). Abundance and biomass were log transformed, and percents were arcsine square root transformed for analysis. *Corbicula* were excluded from % abundance and biomass analyses.

	Non-Chironomid Insect Ab		Chironomid Ab		% Chironomid Ab		Oligochaete Ab		Corbicula Ab	
Ref	92.80 ±	37.2 ab	6782.2 ±	1672.0 a	94.4 ±	1.7 a	267.0 ±	114.3 b	1.9 ±	1.9 b
Un	128.79 ±	44.9 ab	2600.4 ±	881.9 abc	76.8 ±	6.4 ab	157.2 ±	19.9 b	0.0 ±	0.0 b
9	178.03 ±	55.0 ab	5238.6 ±	1269.6 ab	91.7 ±	1.7 a	181.8 ±	46.6 b	7.6 ±	4.3 b
7	51.14 ±	33.5 b	1806.8 ±	672.9 bc	52.9 ±	8.5 bc	1206.4 ±	473.8 a	7.6 ±	4.3 b
3	219.70 ±	58.3 ab	1365.5 ±	568.5 c	41.3 ±	7.2 c	200.8 ±	87.8 b	844.7 ±	484.2 a
1	320.08 ±	100.0 a	3202.7 ±	1274.4 bc	48.2 ±	6.8 c	401.5 ±	129.9 ab	994.3 ±	469.1 a

	Non-Chironomid Insect Bm		Chironomid Bm		% Chironomid Bm		Oligochaete Bm		Corbicula Bm	
Ref	176.6 ±	113.0 a	165.0 ±	43.3 a	72.1 ±	8.0 ab	40.2 ±	29.4 ab	3.1 ±	3.1 b
Un	10.0 ±	3.1 a	85.7 ±	25.9 a	64.8 ±	8.3 ab	13.3 ±	4.4 ab	0.0 ±	0.0 b
9	19.5 ±	7.2 a	277.1 ±	103.1 a	82.8 ±	5.7 a	15.7 ±	7.2 b	0.9 ±	0.6 b
7	3.7 ±	3.2 a	150.1 ±	47.4 a	46.3 ±	9.0 bc	96.0 ±	24.7 a	1.0 ±	0.8 b
3	21.9 ±	5.7 a	100.7 ±	51.4 a	23.2 ±	5.1 cd	36.9 ±	22.6 ab	376.2 ±	179.4 b
1	48.4 ±	25.3 a	190.3 ±	86.6 a	14.5 ±	6.4 d	78.9 ±	43.7 ab	1104.8 ±	267.0 a

Table 3.10. Annual mean±standard error of percent functional feeding group composition of macroinvertebrates by abundance and biomass at each site. Values with different letters are significantly different based on a one-way ANOVA (p<0.05). Percents were arcsine square root transformed for analysis. *Corbicula* were excluded from abundance and biomass analyses. (CF=Collector-filterer, CG=Collector-Gatherer, Pred=Predator, Scr=Scaper)

	% CF Ab	% CG Ab	% Pred Ab	% Scr Ab
Ref	0.4 ± 0.3 a	97.6 ± 1.0 a	1.9 ± 0.8 c	0.14 ± 0.1 a
Un	0.4 ± 0.4 a	87.2 ± 5.7 abc	12.0 ± 5.7 abc	0.36 ± 0.2 a
9	0.0 ± 0.0 a	94.9 ± 1.6 a	5.0 ± 1.6 bc	0.18 ± 0.1 a
7	0.2 ± 0.2 a	93.5 ± 2.2 ab	5.6 ± 2.3 bc	0.71 ± 0.3 a
3	0.1 ± 0.1 a	71.1 ± 8.0 c	28.0 ± 8.1 a	0.69 ± 0.4 a
1	1.4 ± 1.0 a	73.7 ± 8.4 bc	23.7 ± 8.6 ab	1.19 ± 0.6 a

	% CF Bm	% CG Bm	% Pred Bm	% Scr Bm
Ref	0.3 ± 0.2 a	81.5 ± 8.2 ab	17.5 ± 8.3 ab	0.7 ± 0.7 a
Un	0.9 ± 0.9 a	76.6 ± 7.7 ab	16.1 ± 6.0 ab	6.4 ± 5.8 a
9	0.0 ± 0.0 a	92.1 ± 2.1 b	7.4 ± 2.1 ab	0.5 ± 0.4 a
7	0.4 ± 0.4 a	91.4 ± 3.5 a	6.0 ± 3.4 b	2.2 ± 1.1 a
3	0.0 ± 0.0 a	61.6 ± 9.4 b	35.2 ± 9.7 a	3.2 ± 2.0 a
1	6.0 ± 4.3 a	62.0 ± 9.8 ab	20.9 ± 9.1 ab	11.1 ± 7.0 a

Table 3.11. Correlation coefficients (r) between environmental and macroinvertebrate values. Bank macroinvertebrate values and % bank habitat taken from Chapter 2. (* indicates p<0.05. ** indicates p<0.01.)

	Benthic				Bank			
	Abundance	Biomass	Richness	Insect Richness	Abundance	Biomass	Richness	Insect Richness
Baseflow Width			+0.937**	+0.909**				
% Organic Bank Habitat	+0.943**	+0.849*			+0.825*	+0.931**	+0.944**	+0.958**
Mean Particle Size	-0.876*							-0.844*

CHAPTER 4

CONCLUSION

If stream restoration projects are not monitored and evaluated, and the results are not disseminated, the practice of stream restoration will persist in making the same mistakes repeatedly, and the science of stream restoration ecology will never progress beyond its infancy (Kondolf and Micheli 1995, Lake et al. 2002). In this study, I evaluated both the direct and indirect ecological effects of bioengineered bank stabilization on four reaches of stream in an urban watershed. These sites were compared to an unrestored site and to a reference site in the same impacted urban watershed.

In Chapter 2, the direct effects of bioengineering on bank habitat and bank macroinvertebrates were examined. At some sites, bioengineering increased the percent organic bank habitat, so that it approached that of the reference site. At the site where the technique joint planting was used, the amount of organic bank habitat was lower than at the unrestored site. The macroinvertebrate community was found to have higher abundance, biomass, and richness both at the reference site and at several of the bioengineering sites. Abundance, biomass, and richness were also higher in samples taken from organic habitats than inorganic habitats. At the reach-scale, percent organic bank habitat proved to be a strong predictor for many aspects of the macroinvertebrate community.

In Chapter 3, other effects of bioengineering on urban streams were considered. In particular, I explored possible benefits of bioengineering to the benthic macroinvertebrate community through mitigation of the effects of high peak stormflows and sedimentation and the increase of allochthonous inputs. No differences were seen in benthic organic matter levels among sites. The benthic macroinvertebrate communities were more dense and less diverse than those on the banks. A strong correlation between percent organic bank habitat and benthic macroinvertebrate abundance and biomass suggests that bioengineered streambanks have the potential to serve as refugia during storms and therefore to provide the benthos with a larger recolonizing population following the flood.

Bioengineered streambank stabilization has been shown to have direct benefits for bank habitat and macroinvertebrates, and indirect benefits for benthic macroinvertebrates. Complete elimination of bank erosion is not a sound goal of stream restoration; some bank erosion is a natural part of an alluvial stream system (Schumm 1985), and bank erosion is important for creating undercut root and woody debris habitats. This study shows that in urban streams impacted by accelerated bank erosion, bioengineered streambank stabilization can mitigate some of the impacts of urbanization.

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APPENDICES

APPENDIX A

SITE PHOTOGRAPHS

a)

b)

c)

Figure A.1. Photographs of Reference Site, South Peachtree Creek at South Peachtree Creek Nature Preserve, Decatur, DeKalb County, GA. a)study reach, b)upstream view, c)downstream view.

a) 

b) 

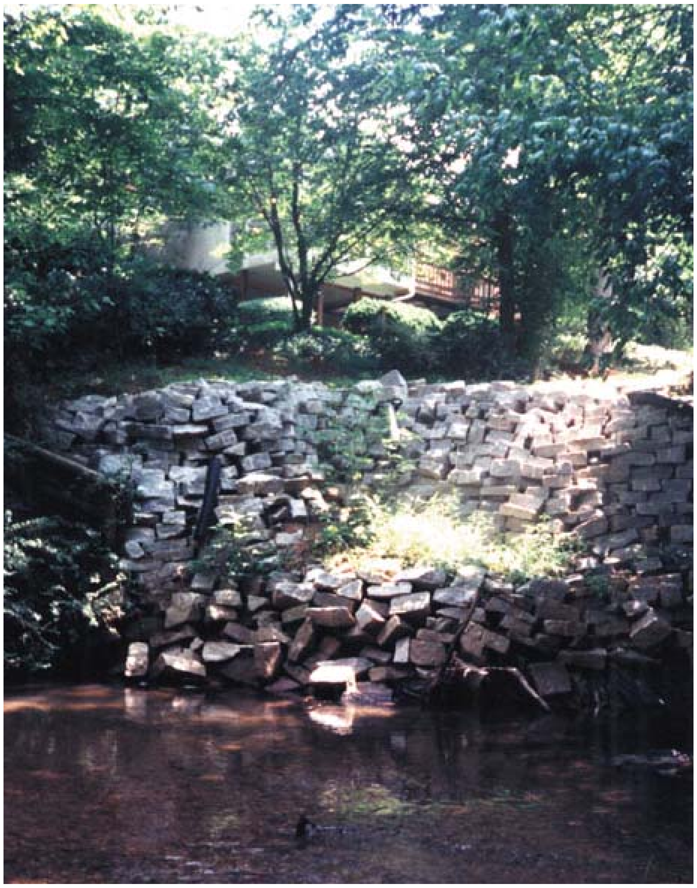
c) 

Figure A.2. Photographs of Unrestored Site, Nancy Creek at Murphey Candler Park, DeKalb County, GA. a)severe bank erosion, b)rip-rap and baseball fields on right bank, c)little riparian vegetation and failed bank stabilization on left bank.

a)

b)

c)

Figure A.3. Photographs of Site 9, South Peachtree Creek at Medlock Park, Decatur, DeKalb County, GA. a)willows and bioengineering, b)failed bioengineering with bank erosion, c)riparian vegetation.

a)

b)

c)

Figure A.4. Photographs of Site 7, North Peachtree Creek at Dresden Park, DeKalb County, GA. a)bioengineering, and rocky streambed, b)downstream structural stabilization, c)riparian vegetation.

a)

b)

c)

Figure A.5. Photographs of Site 3, Nancy Creek at Starlight Drive, Fulton County, GA. a)riparian vegetation, b)joint planting and riprap, c)joint planting and riprap.

a) 

b) 

c)

Figure A.6. Site 1, Nancy Creek at Blue Heron Nature Preserve, Atlanta, Fulton County, GA. a)willow stakes and brush revetments immediately following installation (March 2002), b)tree revetments, willow posts, and willow stakes, six months after installation (August 2002), c)riparian vegetation, six months after installation (August 2002).

APPENDIX B

MACROINVERTEBRATE ASSEMBLAGE BY SITE BY SEASON

Table B.1. Reference Site bank macroinvertebrate assemblage. Numbers are mean bank abundance (#/m of bank) per season (n=4). Annual mean is abundance (#/m of bank) for all four samples (n=16). Richness is number of taxa per sample site per season.

Taxa		Summer 2002	Fall 2002	Spring 2003	Summer 2003	Annual Mean
OTHER						
Corbiculidae	*Corbicula* spp.	1.5	0.0	0.3	0.0	0.4
Collembola		0.8	1.0	0.0	4.3	1.5
Cladocera		0.0	0.0	0.3	0.0	0.1
Copepoda		3.3	8.0	1.3	9.5	5.5
Decapoda		0.0	0.8	1.0	0.0	0.4
Gastropoda		0.0	0.8	0.0	0.0	0.2
Ancylidae	*Ferrissia* spp.	11.3	1.8	0.0	0.5	3.4
Hirudinea		0.8	2.5	0.3	1.8	1.3
Hydracarina		2.5	5.3	0.0	5.8	3.4
Isopoda		0.5	0.0	0.0	0.3	0.2
Oligochaeta		48.0	78.0	12.3	249.3	96.9
Turbellaria		0.0	0.5	0.0	0.3	0.2
COLEOPTERA						
Dysticidae	*Hydrovatus* spp.	0.0	0.0	0.0	0.5	0.1
Noteridae	*Notomicrus* spp.	0.0	0.0	0.0	0.5	0.1
DIPTERA						
Ceratopogonidae	*Atrichopogon* spp.	0.3	0.0	0.0	0.0	0.1
	Dasyhelea spp.	1.3	0.0	0.0	0.0	0.3
	Probezzia spp.	0.0	0.0	0.3	0.0	0.1
	Serromyia spp.	0.3	0.8	0.3	0.0	0.3
Chironomidae	Non-Tanypodinae	179.0	28.3	8.5	146.0	90.4
	Tanypodinae	17.3	4.3	0.0	5.3	6.7
Culicidae	*Aedes* spp.	0.0	0.3	0.0	0.0	0.1
	Culex spp.	0.0	0.0	0.0	0.3	0.1
Empididae	*Hemerodromia* spp.	0.8	0.5	0.0	0.5	0.4
Simulidae	*Simulium* spp.	0.0	0.0	0.0	13.3	3.3
Stratiomyidae	*Allognosta* spp.	0.8	0.0	0.0	0.5	0.3
	Nemotelus spp.	0.0	0.0	0.0	0.3	0.1
	Stratiomys spp.	0.0	0.3	0.5	0.0	0.2
Tipulidae	*Pilaria* spp.	0.5	0.0	0.0	0.0	0.1
	Rhabdomastix spp.	0.3	0.0	0.0	1.3	0.4
	Tipula spp.	2.8	1.3	0.0	0.3	1.1
EPHEMEROPTERA						
Baetidae	*Baetis* spp.	2.0	4.8	0.0	1.8	2.1
	Procleon spp.	0.0	0.0	0.3	3.8	1.0
HEMIPTERA						

Veliidae	*Microvelia* spp.	1.8	0.5	0.0	4.0	1.6
ODONATA						
Aeshnidae	*Boyeria* spp.	0.0	0.0	0.0	0.8	0.2
Caleopterygidae	*Caleopteryx* spp.	0.5	0.3	0.0	0.0	0.2
Coenagrionidae	*Argia* spp.	2.3	2.8	0.0	0.0	1.3
	Enallagma spp.	0.0	0.3	0.0	0.0	0.1
Cordulidae	*Cordulegaster* spp.	0.3	0.3	0.0	0.0	0.1
Gomphidae	*Progomphus* spp.	0.3	0.3	1.0	0.0	0.4
TRICHOPTERA						
Hydropsychidae	*Cheumatopsyche* spp.	0.8	0.0	0.0	0.0	0.2
	Hydropsyche spp.	1.3	0.5	0.0	1.0	0.7
Hydroptilidae	*Hydroptila* spp.	0.0	0.0	0.0	0.8	0.2
Total Abundance (#/m of bank)		281	144	26	452	
Total Richness (# of taxa per sampling day)		26	25	12	25	

Table B.2. Unrestored Site bank macroinvertebrate assemblage. Numbers are mean bank abundance (#/m of bank) per season (n=4). Annual mean is abundance (#/m of bank) for all four samples (n=16). Richness is number of taxa per sample site per season.

Taxa		Summer 2002	Fall 2002	Spring 2003	Summer 2003	Annual Mean
OTHER						
Cladocera		3.0	0.0	0.0	2.0	1.3
Collembola		0.3	0.0	0.3	1.3	0.4
Copepoda		1.8	0.0	0.0	1.8	0.9
Decapoda		1.3	0.0	0.0	0.0	0.3
Gastropoda		1.0	0.0	0.0	1.0	0.5
Ancylidae	*Ferrissia* spp.	0.5	0.3	0.0	1.8	0.6
Hirudinea		1.5	0.3	0.5	0.0	0.6
Hydracarina		0.5	0.3	0.0	2.0	0.7
Hydridae	*Hydra* spp.	0.0	0.0	0.0	32.0	8.0
Oligochaeta		27.5	1.3	5.3	46.8	20.2
Ostracoda		1.5	0.0	0.0	0.0	0.4
Turbellaria		0.3	0.0	0.0	0.0	0.1
COLEOPTERA						
Psephenidae	*Ectopria* spp.	0.0	0.0	0.0	0.3	0.1
Ceratopogonidae	*Atrichopogon* spp.	2.3	0.0	0.0	3.8	1.5
	Culicoides spp.	0.0	0.0	0.0	6.0	1.5
Chironomidae	Non-Tanypodinae	232.8	9.0	3.3	67.3	78.1
Chironomidae	Tanypodinae	17.0	0.3	0.3	2.5	5.0
Empididae	*Hemerodromia* spp.	0.0	0.3	0.0	0.0	0.1
Stratiomyidae	*Allognosta* spp.	0.3	0.0	0.0	0.0	0.1
	Nemotelus spp.	0.0	0.0	0.0	0.5	0.1
Tipulidae	*Antocha* spp.	0.0	0.0	0.3	0.0	0.1
	Dicronota spp.	0.0	0.0	0.0	0.3	0.1
	Tipula spp.	0.5	0.0	0.0	0.3	0.2
EPHEMEROPTERA						
Baetidae	*Baetis* spp.	1.5	0.0	0.0	0.0	0.4
HEMIPTERA						
Gerridae	*Gerris* spp.	0.3	0.0	0.0	0.0	0.1
Veliidae	*Microvelia* spp.	0.0	0.0	0.0	0.3	0.1
ODONATA						
Aeshnidae	*Boyeria* spp.	0.5	0.0	0.0	0.0	0.1
Caleopterygidae	*Caleopteryx* spp.	0.8	0.0	0.0	0.0	0.2
Coenagrionidae	*Argia* spp.	1.3	0.0	0.0	0.0	0.3
	Enallagma spp.	0.5	0.3	0.0	0.0	0.2
Gomphidae	*Progomphus* spp.	1.5	0.0	0.0	0.3	0.4
TRICHOPTERA						
Hydropsychidae	*Hydropsyche* spp.	0.8	0.3	0.0	0.0	0.3

Hydroptilidae	*Hydroptila* spp.	0.0	0.0	0.0	0.3	0.1
Total Abundance (#/m of bank)	Total		12.0	9.8	170.0	
Total Richness (# of taxa	Total Richness (# of	24	9	6	19	

Table B.3. Site 9 bank macroinvertebrate assemblage. Numbers are mean bank abundance (#/m of bank) per season (n=4). Annual mean is abundance (#/m of bank) for all four samples (n=16). Richness is number of taxa per sample site per season.

Taxa		Summer 2002	Fall 2002	Spring 2003	Summer 2003	Annual Mean
OTHER						
Amphipoda		0.0	0.0	0.0	0.3	0.1
Corbiculidae	*Corbicula*	1.3	0.5	0.3	0.0	0.5
Collembola		3.0	17.3	0.5	2.8	5.9
Copepoda		1.5	5.8	5.8	0.8	3.4
Decapoda		0.3	0.5	0.8	1.0	0.5
Gastropoda		0.3	0.3	0.3	0.0	0.2
Ancylidae	*Ferrissia*	3.8	0.0	0.0	1.3	1.3
Hirudinea		4.5	1.3	2.0	0.8	2.1
Hydracarina		3.8	1.8	2.5	2.3	2.6
Hydridae	*Hydra*	0.0	0.0	0.0	0.8	0.2
Oligochaeta		30.0	31.5	161.3	35.5	64.6
COLEOPTERA						
Dysticidae	*Hydroporus*	0.0	0.0	0.0	0.5	0.1
Elmidae	*Stenelmis*	0.0	0.0	0.0	0.3	0.1
DIPTERA						
Ceratopogonidae	*Serromyia*	0.0	0.5	1.0	0.3	0.4
Chironomidae	Non-Tanypodinae	154.5	7.5	9.8	37.8	52.4
Chironomidae	Tanypodinae	21.0	1.0	0.0	3.0	6.3
Dixidae	*Dixella*	0.3	0.0	0.0	0.0	0.1
Empididae	*Hemerodromia*	1.8	0.0	0.0	0.0	0.4
Simulidae	*Simulium*	0.0	0.0	0.0	0.8	0.2
Stratiomyidae	*Nemotelus*	0.0	0.0	0.0	0.5	0.1
Tipulidae	*Pilaria*	0.5	0.0	0.0	0.0	0.1
	Rhabdomastix	0.0	0.5	0.0	0.3	0.2
	Tipula	0.0	0.0	0.0	0.8	1.0
EPHEMEROPTERA						
Baetidae	*Baetis*	0.3	1.3	0.0	0.5	0.5
	Procleon	0.0	0.0	0.0	1.8	0.4
HEMIPTERA						
Veliidae	*Microvelia*	0.3	0.0	0.3	1.0	0.4
ODONATA						
Aeshnidae	*Boyeria*	0.5	0.3	0.0	0.0	0.2
Caleopterygidae	*Caleopteryx*	0.0	0.0	0.0	0.3	0.6
Coenagrionidae	*Argia*	11.5	0.3	0.0	0.0	2.9
	Enallagma	1.0	0.0	0.8	0.3	0.5

	Ishnura	0.3	0.0	0.0	0.0	0.1
Cordulidae	*Cordulegaster*	1.0	0.0	0.0	0.0	0.3
Gomphidae	*Dromogomphus*	1.8	0.0	0.0	0.0	0.4
	Erpetogomphus	0.0	0.0	0.0	0.3	0.1
	Progomphus	0.3	1.3	1.0	0.3	0.7
TRICHOPTERA						
Hydroptilidae	*Hydroptila*	0.0	0.0	0.0	0.5	0.1
Total Abundance (#/m of bank)		243.3	71.5	186.0	94.5	
Total Richness (# of taxa per sampling day)		24	17	13	27	

Table B.4. Site 7 bank macroinvertebrate assemblage. Numbers are mean bank abundance (#/m of bank) per season (n=4). Annual mean is abundance (#/m of bank) for all four samples (n=16). Richness is number of taxa per sample site per season.

Taxa		Summer 2002	Fall 2002	Spring 2003	Summer 2003	Annual
OTHER						
Corbiculidae	*Corbicula*	2.3	0.0	1.3	0.3	0.9
Collembola		2.0	0.5	0.0	0.5	0.8
Copepoda		17.0	1.8	0.3	3.0	5.5
Decapoda		0.0	0.3	0.0	0.0	0.1
Gastropoda		1.0	0.0	0.3	0.3	0.4
Ancylidae	*Ferrissia*	18.3	0.3	0.0	1.8	5.1
Hirudinea		11.0	0.5	0.3	0.0	2.9
Hydracarina		1.3	0.0	0.8	0.8	0.7
Hydridae	*Hydra*	0.3	0.0	0.0	0.0	0.1
Isopoda		1.8	0.0	0.3	0.0	0.5
Oligochaeta		20.3	19.0	34.0	93.8	41.8
Ostracoda		0.0	0.0	0.0	0.3	0.1
COLEOPTERA						
Dysticidae	*Hydroporus*	0.3	0.0	0.0	0.3	0.1
	Hydrovatus	0.0	0.0	0.0	0.3	0.1
DIPTERA						
Ceratopogonidae	*Atrichopogon*	0.3	0.0	0.0	0.0	0.1
	Culicoides	0.0	0.0	5.3	4.0	2.3
	Dasyhelea	0.0	0.3	0.0	0.0	0.1
	Serromyia	0.0	0.0	0.3	0.0	0.1
Chironomidae	Non-Tanypodinae	99.5	10.0	7.5	35.5	38.1
	Tanypodinae	16.8	1.5	2.8	5.0	6.5
Empididae	*Hemerodromia*	0.5	0.3	0.0	0.0	0.2
Stratiomyidae	*Stratiomys*	0.3	0.3	0.0	0.0	0.1
Tabanidae		0.8	0.0	0.0	0.0	0.2
Tipulidae	*Antocha*	0.0	0.0	0.0	0.3	0.1
	Dicronota	0.0	0.3	0.0	0.0	0.1
	Hexatoma	0.0	0.3	0.0	0.0	0.1
	Ormosia	0.0	0.0	0.8	0.5	0.3
	Pilaria	0.0	0.0	0.5	0.0	0.1
	Rhabdomastix	0.0	0.3	0.0	0.0	0.1
	Tipula	0.8	0.0	0.3	1.0	0.5
HEMIPTERA						
Gerridae	*Limnoporus*	0.0	0.3	0.0	0.0	0.1
Veliidae	*Microvelia*	3.3	0.3	0.0	0.5	1.0
ODONATA						
Aeshnidae	*Aeshna*	0.3	0.0	0.0	0.0	0.1
	Boyeria	0.0	0.0	0.0	0.5	0.1

Caleopterygidae	*Caleopteryx*	0.5	0.8	0.0	0.0	0.3
Coenagrionidae	*Argia*	2.5	0.0	0.0	0.0	0.6
	Enallagma	0.3	0.0	0.0	0.0	0.1
	Ishnura	0.0	0.3	0.0	0.0	0.1
Gomphidae	*Progomphus*	0.5	0.0	0.0	0.0	0.1
TRICHOPTERA						
Hydropsychidae	*Hydropsyche*	0.0	0.3	0.0	0.0	0.1
Hydroptilidae	*Hydroptila*	0.0	0.0	0.0	0.3	0.1
Total Abundance (#/m of bank)		149.0	34.5	51.5	142.0	
Total Richness (# of taxa per sampling day)		18	15	9	13	

Table B.5. Site 3 bank macroinvertebrate assemblage. Numbers are mean bank abundance (#/m of bank) per season (n=4). Annual mean is abundance (#/m of bank) for all four samples (n=16). Richness is number of taxa per sample site per season.

Taxa		Summer 2002	Fall 2002	Spring 2003	Summer 2003	Annual
OTHER						
Corbiculidae	*Corbicula*	13.0	0.8	0.0	0.3	3.5
Cladocera		0.0	0.0	1.0	0.0	0.3
Hydridae	*Hydra*	0.0	0.0	1.3	0.3	0.4
Collembola		1.5	0.8	1.0	0.3	0.9
Copepoda		4.3	0.3	1.5	0.0	1.5
Decapoda		0.0	0.3	0.0	0.3	0.1
Gastropoda		1.0	0.0	0.0	0.3	0.3
Ancylidae	*Ferrissia*	2.8	0.5	0.0	21.0	6.1
Hirudinea		0.0	0.3	0.0	0.0	0.1
Hydracarina		1.3	0.0	1.3	7.5	2.5
Oligochaeta		5.5	1.5	28.5	33.8	17.3
Ostracoda		1.0	0.0	0.0	0.0	0.3
COLEOPTERA						
Elmidae	*Ancyronyx*	0.0	0.0	0.0	0.3	0.1
DIPTERA						
Ceratopogonidae	*Atrichopogon*	0.8	0.0	0.0	1.3	0.5
	Culicoides	0.0	0.0	0.3	0.0	0.1
	Monohelea	0.3	0.0	0.0	0.0	0.1
	Serromyia	1.3	0.0	0.0	0.0	0.3
Chironomidae	Non-Tanypodinae	42.3	3.3	9.0	144.5	49.8
	Tanypodinae	5.0	0.0	0.0	1.3	1.6
Empididae	*Hemerodromia*	0.0	0.0	0.0	0.5	0.1
Simulidae	*Simulium*	0.0	0.0	0.0	0.3	0.1
Stratiomyidae	*Nemotelus*	0.0	0.3	0.0	0.3	0.1
Tipulidae	*Antocha*	0.0	0.0	0.0	1.0	0.3
EPHEMEROPTERA						
Baetidae	*Baetis*	0.8	0.0	0.0	2.5	0.8
HEMIPTERA						
Gelastocoridae	*Gelastocorus*	0.3	0.0	0.0	0.0	0.1
Veliidae	*Microvelia*	0.8	0.0	0.0	0.8	0.4
ODONATA						
Coenagrionidae	*Argia*	0.0	0.0	0.0	0.3	0.1
	Enallagma	0.8	0.0	0.0	0.0	0.2
TRICHOPTERA						
Hydropsychidae	*Hydropsyche*	1.0	0.3	0.0	2.3	0.9
Hydroptilidae	*Hydroptila*	0.0	0.0	0.0	0.5	0.1

Total Abundance (#/m of bank)	70.5	7.25	42.75	218.75
Total Richness (# of taxa per sampling day)	18	9	7	20

Table B.6. Site 1 bank macroinvertebrate assemblage. Numbers are mean bank abundance (#/m of bank) per season (n=4). Annual mean is abundance (#/m of bank) for all four samples (n=16). Richness is number of taxa per sample site per season.

Taxa		Summer 2002	Fall 2002	Spring 2003	Summer 2003	Annual
OTHER						
Amphipoda		0.0	0.5	0.0	0.0	0.1
Corbiculidae	*Corbicula*	24.8	0.3	0.0	0.0	6.3
Cladocera		19.8	0.5	1.0	0.0	5.3
Collembola		12.8	1.8	1.0	0.3	3.9
Copepoda		19.0	2.8	5.0	0.0	6.7
Decapoda		0.0	0.5	0.0	0.3	0.2
Gastropoda		1.0	0.0	0.0	0.0	0.3
Ancylidae	*Ferrissia*	10.5	0.5	0.0	0.8	2.9
Hirudinea		3.0	1.5	1.5	1.8	1.9
Hydracarina		1.8	2.3	1.8	1.3	1.8
Hydridae	*Hydra*	0.0	0.0	3.0	0.0	0.8
Oligochaeta		215.3	30.8	230.8	144.8	155.4
Ostracoda		13.3	0.0	0.8	0.0	3.5
Turbellaria		0.3	0.0	3.8	0.0	1.0
COLEOPTERA						
Elmidae	*Ancyronyx*	0.0	0.0	0.0	0.3	0.1
	Stenelmis	0.0	0.0	0.0	0.3	0.1
DIPTERA						
Ceratopogonidae	*Atrichopogon*	8.0	0.0	0.0	0.3	2.1
	Dasyhelea	0.5	0.0	0.0	0.5	0.3
	Serromyia	2.5	0.3	1.0	0.3	1.0
Chironomidae	Non-Tanypodinae	190.8	6.3	28.5	66.5	73.0
	Tanypodinae	14.3	0.3	0.5	2.3	4.3
Empididae	*Hemerodromia*	0.0	0.8	0.0	0.3	0.3
Stratiomyidae	*Nemotelus*	0.0	0.0	0.0	1.5	0.4
	Stratiomys	0.8	0.3	0.3	0.0	0.3
Tipulidae	*Antocha*	0.0	0.0	1.0	0.3	0.3
	Ormosia	0.3	0.0	0.0	0.0	0.1
	Pilaria	0.5	0.0	0.0	0.0	0.1
	Rhabdomastix	0.0	0.0	0.0	0.3	0.1
	Tipula	0.5	1.5	0.0	0.5	0.6
EPHEMEROPTERA						
Baetidae	*Baetis*	4.3	0.0	0.0	1.3	1.4
HEMIPTERA						
Gelastocoridae	*Gelastocorus*	0.0	0.0	0.0	0.5	0.1
Veliidae	*Microvelia*	3.8	0.0	0.0	0.3	1.0
ODONATA						
Caleopterygidae	*Caleopteryx*	0.0	0.5	0.0	0.0	0.1

Coenagrionidae	*Argia*	0.0	0.8	0.0	0.3	0.3
	Enallagma	0.5	0.0	0.0	0.3	0.2
Cordulidae	*Cordulegaster*	0.0	0.3	0.0	0.0	0.1
Libellulidae	*Plathemis*	0.3	0.0	0.0	0.0	0.1
Micromiidae	*Macromia*	0.5	0.0	0.0	0.0	0.1
TRICHOPTERA						
Hydropsychidae	*Hydropsyche*	0.0	0.0	0.0	2.8	0.7
Hydroptilidae	*Hydroptila*	0.0	0.0	0.0	0.3	0.1
Total Abundance (#/m of bank)		548.5	52.0	279.8	227.5	
Total Richness (# of taxa per sampling day)		25	19	14	25	

Table B.7. Reference Site benthic macroinvertebrate assemblage. Numbers are mean benthic abundance (#/m^2) per season (n=4). Annual mean is abundance (#/m^2) for all four samples (n=12). Richness is number of taxa per sample site per season.

Taxa		Summer 2002	Fall 2002	Summer 2003	Annual Mean
OTHER					
Corbiculidae	*Corbicula*	0	6	0	2
Collembola		0	6	0	2
Copepoda		11	0	0	4
Gastropoda		17	0	0	6
Ancylidae	*Ferrissia*	17	0	0	6
Hirudinea		34	0	0	11
Hydracarina		68	0	11	27
Oligochaeta		545	85	170	267
COLEOPTERA					
Elmidae	*Stenelmis*	0	0	6	2
DIPTERA					
Ceratopogonidae	*Serromyia*	74	51	6	44
Chironomidae	Non-Tanypodinae	13699	3545	2858	6701
	Tanypodinae	244	0	0	81
Simulidae	*Simulium*	0	0	6	2
EPHEMEROPTERA					
Baetidae	*Baetis*	51	0	0	17
HEMIPTERA					
Veliidae	*Microvelia*	6	0	0	2
ODONATA					
Gomphidae	*Progomphus*	23	0	0	8
TRICHOPTERA					
Hydropsychidae	*Hydropsyche*	28	0	23	17
Total Abundance (#/m^2)		14818	3693	3080	
Total Richness (# of taxa per sampling day)		13	5	7	

Table B.8. Unrestored Site benthic macroinvertebrate assemblage. Numbers are mean benthic abundance (#/m^2) per season (n=4). Annual mean is abundance (#/m^2) for all four samples (n=12). Richness is number of taxa per sample site per season

Taxa		Summer 2002	Fall 2002	Summer 2003	Annual Mean
OTHER					
Cladocera		0	0	6	2
Collembola		0	0	11	4
Copepoda		23	0	11	11
Gastropoda		0	0	17	6
Ancylidae	*Ferrissia*	6	0	0	2
Hirudinea		28	0	0	9
Hydracarina		17	0	6	8
Hydridae	*Hydra*	11	0	0	4
Oligochaeta		216	102	153	157
Ostracoda		6	0	0	2
DIPTERA					
Ceratopogonidae	*Serromyia*	34	142	97	91
Chironomidae	Non-Tanypodinae	5625	403	1511	2513
	Tanypodinae	261	0	0	87
Empididae	*Hemerodromia*	11	0	0	4
Tipulidae	*Dicronota*	6	0	0	2
EPHEMEROPTERA					
Baetidae	*Baetis*	6	0	0	2
ODONATA					
Gomphidae	*Progomphus*	6	0	0	2
TRICHOPTERA					
Hydropsychidae	*Hydropsyche*	74	0	0	25
Total Abundance (#/m^2)		6330	648	1813	
Total Richness (# of taxa per sampling day)		15	3	8	

Table B.9. Site 9 benthic macroinvertebrate assemblage. Numbers are mean benthic abundance ($\#/m^2$) per season (n=4). Annual mean is abundance ($\#/m^2$) for all four samples (n=12). Richness is number of taxa per sample site per season.

Taxa		Summer 2002	Fall 2002	Summer 2003	Annual Mean
OTHER					
Corbiculidae	*Corbicula*	11	6	6	8
Copepoda		28	0	0	9
Gastropoda		6	6	0	4
Ancylidae	*Ferrissia*	11	6	0	6
Hirudinea		23	0	0	8
Hydracarina		11	0	6	6
Oligochaeta		261	57	227	182
DIPTERA					
Ceratopogonidae	*Atrichopogon*	6	0	0	2
	Serromyia	250	131	23	134
Chironomidae	Non-Tanypodinae	9625	1881	3994	5167
	Tanypodinae	205	0	11	72
Empididae	*Hemerodromia*	11	0	0	4
Stratiomyidae	*Stratiomys*	6	0	0	2
EPHEMEROPTERA					
Baetidae	*Baetis*	85	0	0	28
HEMIPTERA					
Veliidae	*Microvelia*	0	0	11	4
ODONATA					
Coenagrionidae	*Argia*	6	0	0	2
TRICHOPTERA					
Hydropsychidae	*Hydropsyche*	6	0	0	2
Total Abundance ($\#/m^2$)		10551	2085	4278	
Total Richness (# of taxa per sampling day)		16	6	7	

Table B.10. Site 7 benthic macroinvertebrate assemblage. Numbers are mean benthic abundance (#/m^2) per season (n=4). Annual mean is abundance (#/m^2) for all four samples (n=12). Richness is number of taxa per sample site per season.

Taxa		Summer 2002	Fall 2002	Summer 2003	Annual Mean
OTHER					
Corbiculidae	*Corbicula*	0	11	11	8
Collembola		0	0	6	2
Gastropoda		6	0	6	4
Ancylidae	*Ferrissia*	97	17	0	38
Hirudinea		6	0	6	4
Hydracarina		6	0	17	8
Isopoda		0	0	6	2
Oligochaeta		1494	511	1614	1206
Turbellaria		6	0	0	2
DIPTERA					
Ceratopogonidae	*Atrichopogon*	0	0	11	4
	Serromyia	0	0	6	2
Chironomidae	Non-Tanypodinae	3926	1051	222	1733
	Tanypodinae	188	11	23	74
Empididae	*Hemerodromia*	17	0	23	13
TRICHOPTERA					
Hydropsychidae	*Hydropsyche*	91	0	0	30
Total Abundance (#/m^2)		5835	1602	1949	
Total Richness (# of taxa per sampling day)		10	5	12	

Table B.11. Site 3 benthic macroinvertebrate assemblage. Numbers are mean benthic abundance (#/m^2) per season (n=4). Annual mean is abundance (#/m^2) for all four samples (n=12). Richness is number of taxa per sample site per season.

Taxa		Summer 2002	Fall 2002	Summer 2003	Annual Mean
OTHER					
Corbiculidae	*Corbicula*	2290	193	51	845
Cladocera		63	0	0	21
Collembola		6	0	0	2
Copepoda		11	0	6	6
Gastropoda		0	6	6	4
Gastropoda (Ancylidae)	*Ferrissia*	0	0	6	2
Hirudinea		23	0	0	8
Hydracarina		17	0	28	15
Hydridae	*Hydra*	11	0	0	4
Oligochaeta		420	80	102	201
Ostracoda		210	0	11	74
COLEOPTERA					
Elmidae	*Ancyronyx*	6	0	0	2
DIPTERA					
Ceratopogonidae	*Dasyhelea*	11	0	0	4
	Serromyia	85	199	324	203
Chironomidae	Non-Tanypodinae	3068	170	727	1322
	Tanypodinae	74	0	57	44
Empididae	*Hemerodromia*	0	0	6	2
EPHEMEROPTERA					
Baetidae	*Baetis*	6	0	6	4
ODONATA					
Gomphidae	*Progomphus*	6	0	0	2
TRICHOPTERA					
Hydropsychidae	*Hydropsyche*	0	0	6	2
Total Abundance (#/m^2)		6307	648	1335	
Total Richness (# of taxa per sampling day)		16	5	13	

Table B.12. Site 1 benthic macroinvertebrate assemblage. Numbers are mean benthic abundance (#/m^2) per season (n=4). Annual mean is abundance (#/m^2) for all four samples (n=12). Richness is number of taxa per sample site per season.

Taxa		Summer 2002	Fall 2002	Summer 2003	Annual Mean
OTHER					
Corbiculidae	*Corbicula*	2597	176	210	994
Cladocera		17	0	0	6
Collembola		0	0	11	4
Copepoda		34	0	40	25
Gastropoda		11	6	17	11
Gastropoda (Ancylidae)	*Ferrissia*	34	0	6	13
Hydracarina		114	0	23	45
Hydridae	*Hydra*	11	0	0	4
Isopoda		0	0	17	6
Oligochaeta		960	45	199	402
Ostracoda		97	0	17	38
DIPTERA					
Ceratopogonidae	*Dasyhelea*	11	0	0	4
	Serromyia	131	205	261	199
Chironomidae	Non-Tanypodinae	8403	165	818	3129
	Tanypodinae	210	0	11	74
Empididae	*Hemerodromia*	34	0	6	13
Simulidae	*Simulium*	0	0	11	4
EPHEMEROPTERA					
Baetidae	*Baetis*	68	0	0	23
TRICHOPTERA					
Hydropsychidae	*Hydropsyche*	199	0	23	74
Total Abundance (#/m^2)		12932	597	1670	
Total Richness (# of taxa per sampling day)		16	5	15	

APPENDIX C

MACROINVERTEBRATES AND ORGANIC MATTER BY SEASON, SITE, AND HABITAT

Table C.1. Seasonal mean±standard error of abundance (Ab, #/meter of bank) and biomass (Bm, mg DM/meter of bank) of macroinvertebrates by site. Values with different letters are significantly different based on One-way ANOVA ($p<0.05$). Abundance and biomass were log transformed, and percents were arcsine square root transformed for analysis. Crayfish were excluded from biomass analyses. a)Summer 2002, b)Fall 2002, c)Spring 2003, 4)Summer 2003

a)

Site	Total Ab	Insect Ab	Non-Chironomid Insect Ab	% Chironomid Ab	Total Bm	Insect Bm	Non-Chironomid Insect Bm	% Chironomid Bm
Ref	280.5 ± 79.4 ab	212.8 ± 75.2 a	16.5 ± 7.3 a	60.5 ± 13.6 ab	29.1 ± 12.3 ab	25.8 ± 12.2 ab	11.2 ± 8.8 a	53.6 ± 22.4 a
Un	298.8 ± 131.5 ab	260.0 ± 117.7 a	10.3 ± 2.8 a	82.8 ± 3.7 a	32.2 ± 6.1 ab	31.4 ± 6.2 a	16.2 ± 10.0 a	53.1 ± 42.0 a
9	246.5 ± 64.3 ab	200.3 ± 53.9 a	24.8 ± 11.4 a	70.6 ± 6.9 ab	43.5 ± 14.5 a	39.4 ± 13.3 a	13.2 ± 9.7 a	66.6 ± 18.0 a
7	201.8 ± 63.8 ab	128.8 ± 46.6 a	12.5 ± 4.1 a	49.7 ± 9.4 ab	81.2 ± 50.8 a	76.1 ± 51.8 a	60.8 ± 56.2 a	54.8 ± 39.5 a
3	83.5 ± 24.1 b	54.8 ± 17.0 a	7.5 ± 1.0 a	54.7 ± 5.8 ab	6.1 ± 1.2 b	4.3 ± 0.8 b	0.7 ± 0.1 a	59.7 ± 2.1 a
1	548.5 ± 172.7 a	240.0 ± 101.4 a	35.0 ± 8.1 a	31.7 ± 9.5 b	24.2 ± 5.5 ab	17.2 ± 4.7 ab	4.6 ± 1.9 a	43.9 ± 29.3 a

b)

Site	Total Ab	Insect Ab	Non-Chironomid Insect Ab	% Chironomid Ab	Total Bm	Insect Bm	Non-Chironomid Insect Bm	% Chironomid Bm
Ref	144.0 ± 70.6 a	46.5 ± 20.5 a	14.0 ± 6.3 ab	30.3 ± 9.6 ab	53.0 ± 24.8 a	44.4 ± 23.2 ab	42.1 ± 23.4 ab	8.0 ± 6.2 ab
Un	12.0 ± 6.4 a	10.0 ± 6.0 a	0.8 ± 0.5 c	59.5 ± 13.4 a	2.0 ± 1.1 a	1.9 ± 1.1 b	1.0 ± 1.0 b	52.2 ± 51.3 a
9	74.3 ± 13.6 a	33.0 ± 5.3 a	24.5 ± 6.9 a	10.5 ± 3.8 b	80.1 ± 46.8 a	78.4 ± 46.9 a	77.6 ± 46.7 a	1.3 ± 1.5 b
7	37.8 ± 24.8 a	16.0 ± 9.0 a	4.5 ± 0.9 abc	29.7 ± 7.1 ab	31.4 ± 22.8 a	30.8 ± 22.6 ab	29.8 ± 22.0 ab	8.3 ± 7.0 ab
3	8.0 ± 3.4 a	4.5 ± 2.1 a	3.3 ± 1.3 bc	38.3 ± 20.4 ab	0.6 ± 0.3 a	0.6 ± 0.3 b	0.3 ± 0.1 b	30.6 ± 33.3 ab
1	52.0 ± 37.6 a	12.5 ± 10.0 a	6.0 ± 5.0 bc	7.0 ± 4.1 b	47.7 ± 44.6 a	44.7 ± 42.4 ab	44.4 ± 42.2 ab	0.9 ± 1.5 b

c)

Site	Total Ab	Insect Ab	Non-Chironomid Insect Ab	% Chironomid Ab	Total Bm	Insect Bm	Non-Chironomid Insect Bm	% Chironomid Bm
Ref	26.0 ± 15.6 a	10.8 ± 8.1 a	2.3 ± 2.3 a	42.7 ± 19.7 a	7.7 ± 6.8 a	6.1 ± 5.6 a	4.7 ± 4.7 a	44.6 ± 43.6 a
Un	10.0 ± 6.6 a	4.3 ± 2.2 a	0.8 ± 0.5 a	51.5 ± 25.1 a	1.8 ± 1.0 a	0.9 ± 0.7 a	0.2 ± 0.2 a	46.4 ± 48.2 a
9	185.8 ± 152.9 a	13.5 ± 8.9 a	3.8 ± 3.1 a	13.8 ± 7.3 a	18.5 ± 17.4 a	11.6 ± 10.9 a	10.9 ± 10.9 a	42.1 ± 34.2 a
7	54.3 ± 21.5 a	17.3 ± 2.9 a	7.0 ± 4.0 a	29.2 ± 18.7 a	12.0 ± 3.8 a	6.8 ± 4.2 a	3.4 ± 2.5 a	23.2 ± 26.0 a
3	43.8 ± 16.3 a	10.3 ± 6.7 a	1.3 ± 0.6 a	25.5 ± 10.4 a	1.4 ± 0.9 a	1.1 ± 0.9 a	0.0 ± 0.0 a	63.1 ± 38.7 a
1	279.8 ± 106.1 a	32.3 ± 20.8 a	3.3 ± 1.3 a	14.2 ± 5.4 a	17.1 ± 9.1 a	5.3 ± 3.6 a	0.5 ± 0.5 a	56.6 ± 39.1 a

d)

Site	Total Ab	Insect Ab	Non-Chironomid Insect Ab	% Chironomid Ab	Total Bm	Insect Bm	Non-Chironomid Insect Bm	% Chironomid Bm
Ref	452.0 ± 73.1 a	184.8 ± 81.5 a	33.5 ± 18.3 a	29.5 ± 9.7 a	86.8 ± 65.3 a	83.1 ± 65.0 a	74.2 ± 63.9 a	23.9 ± 15.9 a
Un	170.0 ± 45.6 a	82.8 ± 30.8 a	13.0 ± 8.2 a	42.7 ± 11.5 a	9.6 ± 3.0 a	7.1 ± 1.7 a	2.4 ± 2.0 a	62.7 ± 34.5 a
9	93.8 ± 36.1 a	52.0 ± 17.8 a	11.3 ± 3.4 a	48.7 ± 7.9 a	12.8 ± 4.0 a	11.6 ± 3.5 a	8.9 ± 3.0 a	33.6 ± 31.0 a
7	148.5 ± 75.5 a	48.5 ± 20.6 a	8.0 ± 4.0 a	43.8 ± 12.4 a	20.1 ± 11.1 a	16.0 ± 10.0 a	13.1 ± 10.1 a	38.9 ± 35.8 a
3	219.0 ± 157.3 a	155.8 ± 112.0 a	10.0 ± 7.7 a	68.0 ± 7.1 a	9.7 ± 7.1 a	7.5 ± 5.4 a	2.0 ± 1.7 a	68.0 ± 14.9 a
1	227.5 ± 162.0 a	78.8 ± 27.2 a	10.0 ± 5.3 a	59.4 ± 16.0 a	21.4 ± 13.2 a	14.1 ± 7.0 a	7.6 ± 4.9 a	49.2 ± 24.4 a

Table C.2. Seasonal mean±standard error of percent functional feeding group composition of macroinvertebrates by abundance (Ab) and biomass (Bm) by site. Values with different letters are significantly different based on one-way ANOVA ($p<0.05$). Percents were arcsine square root transformed for analysis. Crayfish were excluded from biomass analyses. (CF=Collector-Filterer, CG=Collector-Gatherer, P=Predator, SC=Scaper, SH=Shredder). a)Summer 2002, b)Fall 2002, c)Spring 2003, d)Summer 2003.

a)

Site	n	% CF Ab	% CG Ab	% P Ab	% SC Ab	% SH Ab
Ref	4	2.1 ± 1.1 b	83.6 ± 4.4 a	8.9 ± 2.3 a	4.1 ± 2.3 ab	1.2 ± 0.6 a
Un	4	0.3 ± 0.1 b	82.0 ± 7.1 a	16.4 ± 7.5 a	0.4 ± 0.3 b	0.2 ± 0.1 a
9	4	0.3 ± 0.2 b	78.4 ± 3.9 a	18.8 ± 3.5 a	1.2 ± 0.7 ab	0.8 ± 0.6 a
7	4	1.0 ± 0.6 b	61.0 ± 12.2 a	19.0 ± 2.2 a	18.1 ± 10.6 a	0.9 ± 0.6 a
3	4	15.0 ± 2.1 a	67.7 ± 1.4 a	13.7 ± 2.4 a	3.5 ± 1.5 ab	0.0 ± 0.0 a
1	4	3.1 ± 1.6 b	88.1 ± 4.0 a	7.3 ± 2.0 a	1.3 ± 0.9 ab	0.1 ± 0.1 a
		% CF Bm	% CG Bm	% P Bm	% SC Bm	% SH Bm
Ref	4	8.5 ± 6.2 ab	63.8 ± 11.7 a	8.4 ± 1.3 a	3.5 ± 1.7 a	15.7 ± 12.2 a
Un	4	0.5 ± 0.3 b	52.2 ± 21.8 a	46.1 ± 22.6 a	0.9 ± 0.6 a	0.4 ± 0.4 a
9	4	0.1 ± 0.1 b	66.7 ± 10.9 a	21.0 ± 7.9 a	1.4 ± 1.0 a	10.8 ± 6.1 a
7	4	1.0 ± 0.6 ab	52.2 ± 17.9 a	42.8 ± 19.6 a	3.4 ± 1.7 a	0.6 ± 0.4 a
3	4	15.1 ± 1.5 a	61.3 ± 4.1 a	17.9 ± 4.6 a	5.8 ± 3.3 a	0.0 ± 0.0 a
1	4	2.9 ± 1.3 ab	74.6 ± 5.1 a	18.7 ± 6.3 a	3.4 ± 0.6 a	0.3 ± 0.3 a

b)

Site	n	% CF Ab	% CG Ab	% P Ab	% SC Ab	% SH Ab
Ref	4	0.1 ± 0.1 a	84.0 ± 4.9 a	13.2 ± 4.7 a	1.5 ± 0.6 a	0.7 ± 0.4 a
Un	4	6.3 ± 6.3 a	73.4 ± 17.0 a	17.2 ± 11.2 a	3.1 ± 3.1 a	0.0 ± 0.0 a
9	4	0.7 ± 0.7 a	87.7 ± 3.4 a	8.9 ± 3.6 a	0.4 ± 0.4 a	2.2 ± 1.0 a
7	4	0.2 ± 0.2 a	75.2 ± 5.5 a	22.1 ± 4.7 a	0.2 ± 0.2 a	0.0 ± 0.0 a
3	4	13.3 ± 6.7 a	76.7 ± 5.1 a	2.8 ± 2.8 a	4.4 ± 4.4 a	0.0 ± 0.0 a
1	4	0.2 ± 0.2 a	87.7 ± 4.9 a	9.4 ± 3.4 a	0.3 ± 0.3 a	1.5 ± 0.9 a
		% CF Bm	% CG Bm	% P Bm	% SC Bm	% SH Bm
Ref	4	0.3 ± 0.3 a	52.8 ± 20.2 a	36.9 ± 20.0 a	0.8 ± 0.5 a	9.2 ± 8.2 a
Un	4	11.3 ± 11.3 a	54.1 ± 25.8 a	33.4 ± 19.8 a	1.2 ± 1.2 a	0.0 ± 0.0 a
9	4	0.0 ± 0.0 a	16.0 ± 10.1 a	48.7 ± 27.0 a	0.2 ± 0.2 a	35.1 ± 20.4 a
7	4	1.1 ± 1.1 a	22.7 ± 14.7 a	76.1 ± 14.3 a	0.0 ± 0.0 a	0.0 ± 0.0 a
3	4	16.3 ± 9.6 a	77.8 ± 10.9 a	0.1 ± 0.1 a	5.7 ± 5.7 a	0.0 ± 0.0 a
1	4	0.0 ± 0.0 a	57.2 ± 24.2 a	15.1 ± 13.1 a	0.0 ± 0.0 a	27.7 ± 17.2 a

c)

Site	n	% CF Ab		% CG Ab		% P Ab		% SC Ab		% SH Ab	
Ref	4	0.3 ±	0.3 a	88.4 ±	6.1 a	4.1 ±	2.4 a	0.0 ±	0.0 a	0.0 ±	0.0 a
Un	4	0.0 ±	0.0 a	70.5 ±	23.6 a	28.6 ±	23.9 a	0.0 ±	0.0 a	0.9 ±	0.9 a
9	4	0.0 ±	0.0 a	95.0 ±	2.0 a	4.9 ±	2.0 a	0.0 ±	0.0 a	0.0 ±	0.0 a
7	4	1.9 ±	1.9 a	61.8 ±	17.4 a	34.7 ±	18.2 a	0.4 ±	0.4 a	1.3 ±	1.3 a
3	4	0.0 ±	0.0 a	93.5 ±	3.9 a	6.5 ±	3.9 a	0.0 ±	0.0 a	0.0 ±	0.0 a
1	4	0.0 ±	0.0 a	90.4 ±	5.1 a	9.6 ±	5.1 a	0.0 ±	0.0 a	0.0 ±	0.0 a
		% CF Bm		% CG Bm		% P Bm		% SC Bm		% SH Bm	
Ref	4	0.0 ±	0.0 a	83.7 ±	16.2 a	16.2 ±	16.2 a	0.0 ±	0.0 a	0.0 ±	0.0 a
Un	4	0.0 ±	0.0 a	72.9 ±	24.3 a	26.6 ±	24.5 a	0.0 ±	0.0 a	0.5 ±	0.5 a
9	4	0.0 ±	0.0 a	76.6 ±	14.9 a	23.1 ±	14.6 a	0.3 ±	0.3 a	0.0 ±	0.0 a
7	4	2.7 ±	2.7 a	60.5 ±	16.1 a	22.9 ±	6.2 a	0.1 ±	0.1 a	13.8 ±	13.8 a
3	4	0.0 ±	0.0 a	80.6 ±	11.2 a	19.4 ±	11.2 a	0.0 ±	0.0 a	0.0 ±	0.0 a
1	4	0.0 ±	0.0 a	88.5 ±	5.8 a	11.5 ±	5.8 a	0.0 ±	0.0 a	0.0 ±	0.0 a

d)

Site	n	% CF Ab		% CG Ab		% P Ab		% SC Ab		% SH Ab	
Ref	4	2.4 ±	2.2 a	93.1 ±	2.6 a	4.0 ±	0.7 a	0.1 ±	0.1 a	0.2 ±	0.1 a
Un	4	0.0 ±	0.0 a	76.2 ±	7.6 a	21.1 ±	8.4 a	2.4 ±	0.7 a	0.2 ±	0.2 a
9	4	0.6 ±	0.6 a	87.8 ±	3.2 a	8.6 ±	3.4 a	0.9 ±	0.7 a	0.6 ±	0.6 a
7	4	0.9 ±	0.9 a	84.5 ±	10.1 a	13.4 ±	9.5 a	0.7 ±	0.5 a	0.4 ±	0.3 a
3	4	1.1 ±	0.7 a	89.2 ±	4.7 a	4.3 ±	2.0 a	5.2 ±	3.0 a	0.0 ±	0.0 a
1	4	0.8 ±	0.5 a	91.4 ±	2.2 a	5.8 ±	1.5 a	0.1 ±	0.1 a	0.9 ±	0.9 a
		% CF Bm		% CG Bm		% P Bm		% SC Bm		% SH Bm	
Ref	4	1.2 ±	1.0 a	55.1 ±	18.8 a	38.8 ±	21.4 a	0.0 ±	0.0 a	4.9 ±	4.2 a
Un	4	0.0 ±	0.0 a	77.3 ±	12.6 a	17.2 ±	12.8 a	5.3 ±	3.1 a	0.1 ±	0.1 a
9	4	0.3 ±	0.3 a	53.9 ±	19.4 a	33.2 ±	19.2 a	6.4 ±	6.0 a	6.3 ±	6.3 a
7	4	0.5 ±	0.5 a	55.3 ±	20.5 a	34.7 ±	18.3 a	1.5 ±	1.5 a	8.0 ±	6.2 a
3	4	6.2 ±	3.6 a	76.6 ±	8.3 a	7.4 ±	3.3 a	9.8 ±	5.7 a	0.0 ±	0.0 a
1	4	10.2 ±	6.9 a	74.1 ±	6.5 a	13.8 ±	7.3 a	0.6 ±	0.6 a	1.3 ±	1.1 a

Table C.3. Seasonal mean±standard error of abundance (Ab, #/meter of bank) and biomass (Bm, mg DM/meter of bank) of macroinvertebrates by habitat. Values with different letters are significantly different based on One-way ANOVA ($p<0.05$). Abundance and biomass were log transformed, and percents were arcsine square root transformed for analysis. Crayfish were excluded from biomass analyses. a)Summer 2002, b)Fall 2002, c)Spring 2003, 4)Summer 2003, e)organic vs inorganic habitat.

a)

Category	n	Total Ab	Insect Ab	Non-Chironomid Insect Ab	% Chironomid Ab	Total Bm	Insect Bm	Non-Chironomid Insect Bm	% Chironomid Bm
mud	3	539.3 ± 250.1 a	270.3 ± 124.2 a	23.3 ± 10.3 a	46.3 ± 19.5 a	30.1 ± 6.0 ab	24.2 ± 6.2 ab	4.7 ± 2.6 a	56.7 ± 23.4 a
rock	4	83.5 ± 24.1 b	54.8 ± 17.0 a	7.5 ± 1.0 a	54.7 ± 5.8 a	6.1 ± 1.2 b	4.3 ± 0.8 b	0.7 ± 0.1 a	59.7 ± 1.1 a
roots	10	228.1 ± 40.1 ab	163.5 ± 34.5 a	18.4 ± 5.2 a	57.0 ± 6.7 a	57.2 ± 20.8 a	52.7 ± 21.0 a	33.4 ± 22.3 a	56.2 ± 8.7 a
wood	7	343.6 ± 81.9 ab	245.9 ± 71.9 a	20.3 ± 6.5 a	67.4 ± 9.6 a	25.6 ± 4.3 a	22.9 ± 4.8 a	10.8 ± 6.0 a	50.9 ± 11.8 a

b)

Category	n	Total Ab	Insect Ab	Non-Chironomid Insect Ab	% Chironomid Ab	Total Bm	Insect Bm	Non-Chironomid Insect Bm	% Chironomid Bm
mud	1	18.0 ±	5.0 ±	3.0 ±	11.1 ±	1.8 ±	1.0 ±	0.7 ±	13.6 ±
rock	4	14.0 ± 5.8 a	11.0 ± 6.0 ab	1.8 ± 1.4 b	74.9 ± 6.4 a	2.5 ± 0.9 ab	2.3 ± 0.8 ab	1.3 ± 0.9 b	49.3 ± 20.2 a
roots	10	103.9 ± 31.3 a	37.3 ± 8.9 a	16.9 ± 4.2 a	21.8 ± 4.8 b	66.4 ± 24.2 a	61.4 ± 23.8 a	59.9 ± 23.8 a	4.9 ± 1.8 a
sand	7	16.3 ± 10.0 a	7.7 ± 3.8 b	4.1 ± 2.1 ab	22.7 ± 9.1 b	10.2 ± 9.5 b	9.9 ± 9.3 b	9.7 ± 9.2 b	18.9 ± 15.4 a
wood	2	42.5 ± 31.5 a	7.0 ± 7.0 ab	2.0 ± 2.0 ab	6.8 ± 6.8 b	56.0 ± 55.8 ab	54.5 ± 54.5 ab	54.0 ± 54.0 ab	0.4 ± 0.4 a

c)

Category	n	Total Ab	Insect Ab	Non-Chironomid Insect Ab	% Chironomid Ab	Total Bm	Insect Bm	Non-Chironomid Insect Bm	% Chironomid Bm
mud	3	15.0 ± 8.1 a	3.3 ± 1.9 a	0.7 ± 0.7 a	43.5 ± 28.3 a	1.7 ± 1.2 a	0.2 ± 0.2 a	0.0 ± 0.0 a	39.0 ± 30.5 a
rock	4	30.5 ± 16.7 a	11.0 ± 6.5 a	1.3 ± 0.6 a	45.7 ± 16.7 a	1.8 ± 1.0 a	1.7 ± 1.0 a	0.2 ± 0.2 a	76.2 ± 13.4 a
roots	12	138.6 ± 59.7 a	15.8 ± 3.7 a	4.3 ± 1.7 a	21.8 ± 6.6 a	14.8 ± 6.2 a	8.3 ± 4.0 a	6.3 ± 3.8 a	40.8 ± 9.4 a
sand	3	47.3 ± 32.5 a	4.3 ± 3.0 a	2.7 ± 2.7 a	3.8 ± 3.0 a	5.7 ± 5.1 a	0.8 ± 0.7 a	0.6 ± 0.6 a	10.9 ± 10.1 a
wood	2	213.0 ± 210.0 a	48.0 ± 45.0 a	3.0 ± 3.0 a	60.3 ± 39.7 a	13.6 ± 12.9 a	8.3 ± 7.6 a	0.1 ± 0.1 a	79.7 ± 20.3 a

d)

Category	n	Total Ab	Insect Ab	Non-Chironomid Insect Ab	% Chironomid Ab	Total Bm	Insect Bm	Non-Chironomid Insect Bm	% Chironomid Bm
rock	6	232.8 ± 99.3 a	147.2 ± 71.9 a	16.0 ± 6.5 a	56.0 ± 6.0 a	10.4 ± 4.6 a	7.1 ± 3.5 a	1.8 ± 1.1 a	62.0 ± 9.4 a
roots	9	268.8 ± 68.9 a	112.8 ± 41.1 a	19.2 ± 8.9 a	40.7 ± 7.3 a	45.9 ± 29.7 a	43.3 ± 29.4 a	36.7 ± 28.7 a	34.4 ± 7.7 a
sand	7	87.1 ± 36.9 a	38.0 ± 9.1 a	5.6 ± 1.7 a	51.3 ± 9.8 a	13.0 ± 7.0 a	10.9 ± 6.0 a	9.1 ± 5.9 a	49.2 ± 14.4 a
wood	2	408.5 ± 302.5 a	123.0 ± 17.0 a	17.5 ± 7.5 a	53.4 ± 37.2 a	37.7 ± 22.7 a	24.2 ± 9.2 a	13.9 ± 8.1 a	39.9 ± 20.9 a

e)

Season	Category	n	Total Ab	Insect Ab	Non-Chironomid Insect Ab	% Chironomid Ab	Total Bm	Insect Bm	Non-Chironomid Insect Bm	% Chironomid Bm
Summer 2002	Inorganic	7	278.9 ± 132.6 a	147.1 ± 64.7 a	14.3 ± 5.1 a	51.1 ± 8.2 a	16.4 ± 5.4 b	12.8 ± 4.7 b	2.4 ± 1.3 a	58.4 ± 8.9 a
	Organic	17	275.6 ± 42.0 a	197.4 ± 36.0 a	19.2 ± 4.0 a	61.3 ± 5.5 a	44.2 ± 12.7 a	40.4 ± 12.7 a	24.1 ± 13.3 a	54.0 ± 6.8 a
Fall 2002	Inorganic	12	15.7 ± 5.9 b	8.6 ± 2.9 b	3.3 ± 1.3 b	39.1 ± 9.4 a	6.9 ± 5.5 b	6.6 ± 5.4 b	6.2 ± 5.3 b	29.5 ± 11.5 a
	Organic	12	93.7 ± 27.0 a	32.3 ± 8.1 a	14.4 ± 3.9 a	19.3 ± 4.4 a	64.7 ± 21.1 a	60.3 ± 20.8 a	58.9 ± 20.7 a	4.2 ± 1.6 b
Spring 2003	Inorganic	10	30.9 ± 11.4 a	6.7 ± 2.8 b	1.5 ± 0.8 a	32.5 ± 11.4 a	2.9 ± 1.5 b	1.0 ± 0.5 b	0.3 ± 0.2 a	45.5 ± 13.3 a
	Organic	14	149.2 ± 55.9 a	20.4 ± 6.5 a	4.1 ± 1.5 a	27.3 ± 7.9 a	14.6 ± 5.5 a	8.3 ± 3.5 a	5.4 ± 3.3 a	46.4 ± 9.1 a
Summer 2003	Inorganic	13	154.4 ± 52.0 a	88.4 ± 35.5 a	10.4 ± 3.4 a	53.5 ± 5.7 a	11.8 ± 4.1 b	9.1 ± 3.5 b	5.8 ± 3.3 a	55.1 ± 8.7 a
	Organic	11	294.2 ± 71.1 a	114.6 ± 33.3 a	18.9 ± 7.3 a	43.0 ± 7.9 a	44.4 ± 24.2 a	39.8 ± 23.9 a	32.6 ± 23.4 a	35.4 ± 6.9 a

Table C.4. . Seasonal mean±standard error of percent functional feeding group composition of macroinvertebrates by abundance (Ab) and biomass (Bm) by habitat type. Values with different letters are significantly different based on one-way ANOVA ($p<0.05$). Percents were arcsine square root transformed for analysis. Crayfish were excluded from biomass analyses. (CF=Collector-Filterer, CG=Collector-Gatherer, P=Predator, SC=Scaper, SH=Shredder). a)Summer 2002, b)Fall 2002, c)Spring 2003, d)Summer 2003., e)by organic/inorganic habitat

a)

Category	n	% CF Ab	% CG Ab	% P Ab	% SC Ab	% SH Ab
mud	3	2.4 ± 2.2 b	89.5 ± 5.0 a	6.1 ± 1.8 a	1.8 ± 1.0 a	0.2 ± 0.2 ab
rock	4	15.0 ± 2.1 a	67.7 ± 1.4 a	13.7 ± 2.4 a	3.5 ± 1.5 a	0.0 ± 0.0 b
roots	10	1.0 ± 0.4 b	72.7 ± 5.7 a	16.9 ± 2.1 a	8.1 ± 4.8 a	1.1 ± 0.4 a
wood	7	1.5 ± 0.7 b	82.5 ± 4.4 a	13.4 ± 4.3 a	2.0 ± 1.5 a	0.2 ± 0.1 ab

		% CF Bm	% CG Bm	% P Bm	% SC Bm	% SH Bm
mud	3	2.5 ± 1.8 ab	77.4 ± 8.5 a	16.1 ± 9.6 a	3.4 ± 0.6 a	0.5 ± 0.5 a
rock	4	15.1 ± 1.5 a	61.3 ± 4.1 a	17.9 ± 4.6 a	5.8 ± 3.3 a	0.0 ± 0.0 a
roots	10	1.2 ± 0.7 b	59.1 ± 8.6 a	27.2 ± 8.9 a	2.0 ± 0.8 a	10.4 ± 5.4 a
wood	7	4.7 ± 3.7 ab	59.2 ± 12.4 a	32.5 ± 13.7 a	2.8 ± 1.1 a	0.8 ± 0.6 a

b)

Category	n	% CF Ab	% CG Ab	% P Ab	% SC Ab	% SH Ab
mud	1	0.0 ±	77.8 ±	22.2 ±	0.0 ±	0.0 ±
rock	4	5.0 ± 5.0 a	83.0 ± 3.9 a	6.8 ± 2.6 a	3.1 ± 3.1 a	0.0 ± 0.0 a
roots	10	0.2 ± 0.1 a	80.4 ± 3.2 a	15.6 ± 3.1 a	0.7 ± 0.3 a	1.6 ± 0.5 a
sand	7	7.9 ± 4.7 a	79.0 ± 11.9 a	10.9 ± 8.0 a	2.2 ± 2.2 a	0.0 ± 0.0 a
wood	2	0.0 ± 0.0 a	87.3 ± 3.6 a	10.6 ± 1.5 a	1.4 ± 1.4 a	0.7 ± 0.7 a

		% CF Bm	% CG Bm	% P Bm	% SC Bm	% SH Bm
mud	1	0.0 ±	65.0 ±	35.0 ±	0.0 ±	0.0 ±
rock	4	8.3 ± 8.3 a	69.1 ± 20.3 a	21.4 ± 20.4 a	1.2 ± 1.2 a	0.0 ± 0.0 a
roots	10	0.6 ± 0.5 a	22.7 ± 7.1 a	47.8 ± 12.9 a	0.4 ± 0.2 a	28.5 ± 10.3 a
sand	7	10.2 ± 7.5 a	62.7 ± 18.9 a	24.3 ± 16.4 a	2.8 ± 2.8 a	0.0 ± 0.0 a
wood	2	0.0 ± 0.0 a	50.1 ± 47.1 a	48.4 ± 45.5 a	0.0 ± 0.0 a	1.5 ± 1.5 a

c)

Category	n	% CF Ab	% CG Ab	% P Ab	% SC Ab	% SH Ab
mud	3	0.0 ± 0.0 a	95.5 ± 2.3 a	3.4 ± 1.9 a	0.0 ± 0.0 a	1.1 ± 1.1 a
rock	4	0.0 ± 0.0 a	95.1 ± 2.4 a	4.9 ± 2.4 a	0.0 ± 0.0 a	0.0 ± 0.0 a
roots	12	0.8 ± 0.6 a	78.3 ± 6.7 a	18.0 ± 6.8 a	0.1 ± 0.1 a	0.4 ± 0.4 a
sand	3	0.0 ± 0.0 a	65.2 ± 32.6 a	34.8 ± 32.6 a	0.0 ± 0.0 a	0.0 ± 0.0 a
wood	2	0.0 ± 0.0 a	97.9 ± 2.1 a	2.1 ± 2.1 a	0.0 ± 0.0 a	0.0 ± 0.0 a

		% CF Bm	% CG Bm	% P Bm	% SC Bm	% SH Bm
mud	3	0.0 ± 0.0 a	99.2 ± 0.7 a	0.1 ± 0.1 b	0.0 ± 0.0 a	0.7 ± 0.7 a
rock	4	0.0 ± 0.0 a	89.2 ± 8.8 a	10.8 ± 8.8 ab	0.0 ± 0.0 a	0.0 ± 0.0 a
roots	12	0.9 ± 0.9 a	69.4 ± 8.2 a	25.0 ± 6.6 a	0.1 ± 0.1 a	4.6 ± 4.6 a
sand	3	0.0 ± 0.0 a	64.3 ± 32.2 a	35.7 ± 32.2 ab	0.0 ± 0.0 a	0.0 ± 0.0 a
wood	2	0.0 ± 0.0 a	85.6 ± 14.4 a	14.4 ± 14.4 ab	0.0 ± 0.0 a	0.0 ± 0.0 a

d)

Category	n	% CF Ab	% CG Ab	% P Ab	% SC Ab	% SH Ab
rock	6	0.2 ± 0.2 a	86.0 ± 3.5 a	8.5 ± 3.0 a	4.9 ± 1.8 a	0.2 ± 0.2 a
roots	9	1.7 ± 1.0 a	82.1 ± 5.4 a	14.3 ± 5.5 a	0.5 ± 0.3 b	0.7 ± 0.4 a
sand	7	0.4 ± 0.4 a	92.2 ± 2.2 a	6.0 ± 2.2 a	0.5 ± 0.4 b	0.2 ± 0.1 a
wood	2	1.6 ± 0.3 a	94.6 ± 1.2 a	3.4 ± 1.3 a	0.3 ± 0.3 ab	0.1 ± 0.1 a
		% CF Bm	% CG Bm	% P Bm	% SC Bm	% SH Bm
rock	6	1.8 ± 1.8 ab	80.5 ± 5.9 a	6.2 ± 2.2 a	10.5 ± 3.5 a	1.0 ± 1.0 a
roots	9	0.9 ± 0.5 b	55.9 ± 10.3 a	34.8 ± 10.9 a	2.9 ± 2.7 b	5.5 ± 3.1 a
sand	7	2.0 ± 2.0 ab	61.7 ± 14.1 a	32.1 ± 12.9 a	0.4 ± 0.4 ab	3.8 ± 3.7 a
wood	2	20.3 ± 8.9 a	75.6 ± 12.0 a	2.6 ± 1.7 a	1.2 ± 1.2 ab	0.3 ± 0.3 a

e)

Season	Category	n	% CF Ab	% CG Ab	% P Ab	% SC Ab	% SH Ab
Summer 2002	Inorganic	7	9.6 ± 2.9 a	77.1 ± 4.8 a	10.5 ± 2.1 a	2.8 ± 0.9 a	0.1 ± 0.1 b
	Organic	17	1.2 ± 0.4 b	76.7 ± 3.9 a	15.5 ± 2.1 a	5.6 ± 2.9 a	0.7 ± 0.2 a
	Inorganic	12	6.2 ± 3.0 a	80.3 ± 6.4 a	10.4 ± 4.5 a	2.3 ± 1.6 a	0.0 ± 0.0 b
Fall 2002	Organic	12	0.2 ± 0.1 a	81.5 ± 2.8 a	14.8 ± 2.6 a	0.8 ± 0.3 a	1.5 ± 0.5 a
Spring 2003	Inorganic	10	0.0 ± 0.0 a	86.2 ± 9.7 a	13.4 ± 9.7 a	0.0 ± 0.0 a	0.3 ± 0.3 a
	Organic	14	0.7 ± 0.5 a	81.1 ± 6.0 a	15.7 ± 6.0 a	0.1 ± 0.1 a	0.4 ± 0.4 a
Summer 2003	Inorganic	13	0.3 ± 0.2 b	89.4 ± 2.1 a	7.2 ± 1.8 a	2.5 ± 1.0 a	0.2 ± 0.1 a
	Organic	11	1.7 ± 0.8 a	84.3 ± 4.6 a	12.3 ± 4.7 a	0.5 ± 0.3 a	0.6 ± 0.4 a
			% CF Bm	% CG Bm	% P Bm	% SC Bm	% SH Bm
Summer 2002	Inorganic	7	9.7 ± 2.8 a	68.2 ± 5.1 a	17.1 ± 4.4 a	4.8 ± 1.8 a	0.2 ± 0.2 a
	Organic	17	2.6 ± 1.6 b	59.2 ± 7.0 a	29.4 ± 7.5 a	2.3 ± 0.6 a	6.5 ± 3.3 a
	Inorganic	12	8.6 ± 4.8 a	65.2 ± 12.0 a	24.2 ± 10.9 a	2.0 ± 1.6 a	0.0 ± 0.0 b
Fall 2002	Organic	12	0.5 ± 0.4 a	27.3 ± 8.8 b	47.9 ± 12.0 a	0.3 ± 0.2 a	24.0 ± 9.0 a
Spring 2003	Inorganic	10	0.0 ± 0.0 a	84.7 ± 10.1 a	15.1 ± 10.1 a	0.0 ± 0.0 a	0.2 ± 0.2 a
	Organic	14	0.8 ± 0.8 a	71.7 ± 7.3 a	23.5 ± 5.9 a	0.1 ± 0.1 a	3.9 ± 3.9 a
Summer 2003	Inorganic	13	1.9 ± 1.3 a	70.4 ± 8.2 a	20.2 ± 7.7 a	5.1 ± 2.1 a	2.5 ± 2.0 a
	Organic	11	4.5 ± 2.7 a	59.5 ± 8.8 a	28.9 ± 9.7 a	2.6 ± 2.2 a	4.5 ± 2.6 a

Table C.5. Seasonal differences in bank macroinvertebrates. a)Seasonal mean±standard error of abundance (Ab, #/meter of bank) and biomass (Bm, mg DM/meter of bank) of macroinvertebrates. Values with different letters are significantly different based on One-way ANOVA ($p<0.05$). Abundance and biomass were log transformed, and percents were arcsine square root transformed for analysis. Crayfish were excluded from biomass analyses. b) Seasonal mean±standard error of percent functional feeding group composition of macroinvertebrates by abundance (Ab) and biomass (Bm). Values with different letters are significantly different based on one-way ANOVA ($p<0.05$). Percents were arcsine square root transformed for analysis. Crayfish were excluded from biomass analyses. (CF=Collector-Filterer, CG=Collector-Gatherer, P=Predator, SC=Scaper, SH=Shredder)

a)

Season	Total Ab	Insect Ab	Non-Chironomid Insect Ab	% Chironomid Ab	Total Bm	Insect Bm	Non-Chironomid Insect Bm	% Chironomid Bm
Summer 2002	276.6 ± 47.0 a	182.8 ± 31.3 a	17.8 ± 3.1 a	58.3 ± 4.6 a	36.1 ± 9.4 a	32.4 ± 9.4 a	17.8 ± 9.6 a	55.3 ± 5.4 a
Fall 2002	54.7 ± 15.8 b	20.4 ± 4.9 b	8.8 ± 2.3 b	29.2 ± 5.5 c	35.8 ± 12.3 bc	33.5 ± 11.9 a	32.5 ± 11.8 a	16.3 ± 6.1 b
Spring 2003	99.9 ± 34.6 b	14.7 ± 4.1 b	3.0 ± 1.0 c	29.5 ± 6.5 bc	9.7 ± 3.4 c	5.3 ± 2.1 b	3.3 ± 2.0 b	46.0 ± 7.5 a
Summer 2003	218.5 ± 44.5 a	100.4 ± 24.2 a	14.3 ± 3.8 ab	48.7 ± 4.8 ab	26.7 ± 11.5 ab	23.2 ± 11.3 a	18.0 ± 10.9 a	46.1 ± 5.9 a

b)

	% CF Ab	% CG Ab	% P Ab	% SC Ab	% SH Ab
Summer 2002	3.6 ± 1.2 a	76.8 ± 3.1 a	14.0 ± 1.7 a	4.8 ± 2.1 a	0.5 ± 0.2 a
Fall 2002	3.0 ± 1.6 ab	80.9 ± 3.3 a	12.7 ± 2.5 a	1.5 ± 0.8 b	0.8 ± 0.3 a
Spring 2003	0.4 ± 0.3 b	83.3 ± 5.2 a	14.7 ± 5.2 a	0.1 ± 0.1 b	0.4 ± 0.2 a
Summer 2003	1.0 ± 0.4 ab	87.1 ± 2.4 a	9.5 ± 2.3 a	1.6 ± 0.6 ab	0.4 ± 0.2 a

	% CF Bm	% CG Bm	% P Bm	% SC Bm	% SH Bm
Summer 2002	4.7 ± 1.5 a	61.8 ± 5.1 ab	25.8 ± 5.5 a	3.0 ± 0.7 a	4.6 ± 2.4 a
Fall 2002	4.3 ± 2.4 ab	45.4 ± 8.2 b	36.6 ± 8.4 a	1.1 ± 0.8 bc	12.5 ± 5.3 a
Spring 2003	0.5 ± 0.5 b	77.1 ± 6.0 a	20.0 ± 5.4 a	0.1 ± 0.0 c	2.4 ± 2.3 a
Summer 2003	3.1 ± 1.4 ab	65.4 ± 6.0 ab	24.2 ± 6.0 a	3.9 ± 1.5 ab	3.4 ± 1.6 a

Table C.6. Benthic macroinvertebrate abundance and biomass. Seasonal mean±standard error of abundance (Ab, #/m^2) and biomass (Bm, mg DM/m^2) of macroinvertebrates by site. Values with different letters are significantly different based on One-way ANOVA ($p<0.05$). Abundance and biomass were log transformed, and percents were arcsine square root transformed for analysis. Corbicula were excluded from all analyses. a)Summer 2002, b)Fall 2002, c) Summer 2003.

a)

Site	n	Total Ab	Insect Ab	Non-Chironomid Insects	% Chironomid Ab	Total Bm	Insect Bm	Non-Chironomid Insect Bm	% Chironomid Bm
Ref	4	652.0 ± 100.5 a	621.5 ± 91.2 a	8.0 ± 0.6 a	93.9 ± 2.6 a	42.3 ± 11.3 a	37.1 ± 11.7 a	22.7 ± 12.7 a	46.0 ± 15.5 bc
Un	4	278.5 ± 69.0 ab	265.0 ± 70.4 ab	6.0 ± 0.7 a	91.6 ± 2.4 ab	7.9 ± 2.4 b	7.4 ± 2.3 a	0.4 ± 0.2 a	85.7 ± 4.8 ab
9	4	463.8 ± 109.9 ab	448.8 ± 105.3 ab	16.3 ± 0.4 a	92.8 ± 1.2 ab	31.9 ± 9.7 ab	30.7 ± 9.3 a	1.9 ± 0.7 a	90.2 ± 3.0 a
7	4	256.8 ± 105.5 ab	185.8 ± 64.6 ab	4.8 ± 0.7 a	74.2 ± 7.2 abc	20.1 ± 5.0 ab	14.2 ± 3.4 a	0.4 ± 0.4 a	68.4 ± 7.7 abc
3	4	176.8 ± 54.8 b	143.5 ± 59.7 b	5.3 ± 0.5 a	44.3 ± 12.5 c	17.4 ± 5.9 ab	12.7 ± 5.1 a	0.7 ± 0.2 a	33.1 ± 7.1 c
1	4	454.8 ± 69.4 ab	398.5 ± 69.3 ab	19.5 ± 0.9 a	66.7 ± 6.2 bc	37.1 ± 4.7 a	27.3 ± 4.9 a	4.3 ± 3.2 a	39.5 ± 11.3 c

b)

Site	n	Total Ab	Insect Ab	Non-Chironomid Insects	% Chironomid Ab	Total Bm	Insect Bm	Non-Chironomid Insect Bm	% Chironomid Bm
Ref	4	162.3 ± 26.3 a	158.5 ± 26.6 a	2.5 ± 0.3 a	95.8 ± 1.9 a	5.2 ± 0.6 ab	4.6 ± 0.7 a	0.4 ± 0.1 a	76.3 ± 7.0 a
Un	4	28.5 ± 11.7 bc	24.0 ± 10.1 bc	6.3 ± 0.7 a	59.5 ± 15.8 ab	4.0 ± 2.1 ab	2.9 ± 1.6 a	0.5 ± 0.4 a	53.1 ± 15.5 ab
9	4	91.5 ± 25.0 ab	88.5 ± 23.2 ab	5.8 ± 0.4 a	89.1 ± 3.9 a	5.5 ± 1.5 ab	5.0 ± 1.4 a	0.5 ± 0.3 a	77.5 ± 8.8 a
7	4	70.0 ± 32.8 abc	46.8 ± 23.6 abc	0.0 ± 0.0 a	66.2 ± 10.1 ab	10.5 ± 3.7 a	5.7 ± 2.6 a	0.0 ± 0.0 a	56.2 ± 16.4 ab
3	4	20.0 ± 5.8 c	16.3 ± 6.5 c	8.8 ± 0.8 a	23.3 ± 7.4 b	1.6 ± 0.5 b	1.4 ± 0.6 a	1.1 ± 0.6 a	11.2 ± 7.7 bc
1	4	18.5 ± 1.7 c	16.3 ± 2.5 c	9.0 ± 0.8 a	27.9 ± 13.6 b	2.8 ± 0.8 ab	1.6 ± 0.4 a	1.0 ± 0.6 a	1.8 ± 1.2 c

c)

Site	n	Total Ab	Insect Ab	Non-Chironomid Insects	% Chironomid Ab	Total Bm	Insect Bm	Non-Chironomid Insect Bm	% Chironomid Bm
Ref	4	135.5 ± 10.3 a	127.5 ± 7.7 ab	1.8 ± 0.5 b	93.5 ± 4.4 a	3.5 ± 1.4 a	3.4 ± 1.4 a	0.2 ± 0.2 a	94.2 ± 4.0 a
Un	4	79.8 ± 23.5 a	71.3 ± 23.7 ab	4.8 ± 0.4 ab	79.5 ± 4.8 ab	4.0 ± 1.4 a	2.3 ± 0.6 a	0.4 ± 0.1 a	55.7 ± 16.7 ab
9	4	188.0 ± 31.7 a	177.8 ± 32.2 a	1.5 ± 0.3 b	93.2 ± 3.2 a	4.5 ± 1.0 a	3.4 ± 0.7 a	0.2 ± 0.1 a	80.7 ± 15.4 a
7	4	85.3 ± 55.7 a	12.8 ± 5.3 c	2.0 ± 0.3 ab	18.2 ± 4.9 c	3.5 ± 2.2 a	0.3 ± 0.1 b	0.1 ± 0.0 a	14.3 ± 4.8 bc
3	4	56.5 ± 6.7 a	49.5 ± 7.0 b	15.0 ± 0.5 a	56.4 ± 13.1 b	2.3 ± 0.6 a	2.0 ± 0.6 a	1.2 ± 0.5 a	25.3 ± 9.7 bc
1	4	64.3 ± 13.1 a	50.3 ± 10.5 b	13.8 ± 0.4 a	49.8 ± 5.6 bc	15.7 ± 12.0 a	2.6 ± 1.1 a	1.1 ± 0.9 a	2.1 ± 0.9 c

Table C.7. Benthic functional feeding group composition. Seasonal mean±standard error of percent functional feeding group composition of macroinvertebrates by abundance (Ab) and biomass (Bm) by site. Values with different letters are significantly different based on one-way ANOVA ($p<0.05$). Percents were arcsine square root transformed for analysis. *Corbicula* were excluded from all analyses. (CF=Collector-Filterer, CG=Collector-Gatherer, P=Predator, SC=Scaper, no shredders were found in the benthos). a)Summer 2002, b)Fall 2002, c) Summer 2003.

a)

Site	n	% CF Ab	% CG Ab	% P Ab	% SC Ab	% CF Bm	% CG Bm	% P Bm	% SC Bm
Ref	4	0.3 ± 0.3 a	95.7 ± 2.6 a	3.7 ± 2.1 a	0.3 ± 0.2 a	0.1 ± 0.1 a	54.7 ± 19.3 a	45.0 ± 19.2 a	0.2 ± 0.1 a
Un	4	1.2 ± 1.2 a	91.6 ± 4.1 a	7.1 ± 3.1 a	0.1 ± 0.1 a	2.6 ± 2.6 a	85.4 ± 4.9 a	11.8 ± 4.2 ab	0.2 ± 0.2 a
9	4	0.0 ± 0.0 a	94.4 ± 1.8 a	5.3 ± 1.8 a	0.2 ± 0.1 a	0.1 ± 0.1 a	93.2 ± 2.3 a	6.3 ± 2.0 ab	0.4 ± 0.3 a
7	4	0.7 ± 0.7 a	93.8 ± 1.1 a	4.4 ± 1.0 a	1.0 ± 0.6 a	1.2 ± 1.2 a	94.5 ± 2.3 a	2.4 ± 0.3 b	1.8 ± 1.1 a
3	4	0.0 ± 0.0 a	89.7 ± 3.9 a	10.3 ± 3.9 a	0.0 ± 0.0 a	0.0 ± 0.0 a	89.5 ± 6.5 a	10.5 ± 6.5 ab	0.0 ± 0.0 a
1	4	2.6 ± 2.6 a	91.3 ± 4.2 a	5.7 ± 1.8 a	0.5 ± 0.1 a	11.7 ± 11.7 a	82.9 ± 12.9 a	4.6 ± 1.8 b	0.8 ± 0.4 a

b)

Site	n	% CF Ab	% CG Ab	% P Ab	% SC Ab	% CF Bm	% CG Bm	% P Bm	% SC Bm
Ref	4	0.0 ± 0.0 a	98.5 ± 0.5 a	1.5 ± 0.5 a	0.0 ± 0.0 a	0.0 ± 0.0 a	93.2 ± 1.6 a	6.8 ± 1.6 a	0.0 ± 0.0 a
Un	4	0.0 ± 0.0 a	77.5 ± 16.8 a	22.5 ± 16.8 a	0.0 ± 0.0 a	0.0 ± 0.0 a	81.6 ± 15.7 a	18.4 ± 15.7 a	0.0 ± 0.0 a
9	4	0.0 ± 0.0 a	91.3 ± 3.8 a	8.3 ± 3.9 a	0.3 ± 0.3 a	0.0 ± 0.0 a	86.6 ± 5.1 a	12.3 ± 5.4 a	1.1 ± 1.1 a
7	4	0.0 ± 0.0 a	98.7 ± 0.8 a	0.9 ± 0.9 a	0.5 ± 0.5 a	0.0 ± 0.0 a	97.7 ± 2.2 a	0.1 ± 0.1 a	2.3 ± 2.3 a
3	4	0.0 ± 0.0 a	58.4 ± 19.1 a	40.5 ± 19.8 a	1.1 ± 1.1 a	0.0 ± 0.0 a	46.1 ± 18.7 a	48.9 ± 21.6 a	5.0 ± 5.0 a
1	4	0.0 ± 0.0 a	53.8 ± 21.2 a	44.4 ± 22.1 a	1.8 ± 1.8 a	0.0 ± 0.0 a	41.7 ± 20.1 a	42.5 ± 24.4 a	15.9 ± 15.9 a

c)

Site	n	% CF Ab	% CG Ab	% P Ab	% SC Ab	% CF Bm	% CG Bm	% P Bm	% SC Bm
Ref	4	0.8 ± 0.8 a	98.6 ± 1.4 a	0.5 ± 0.5 c	0.2 ± 0.2 a	0.7 ± 0.7 a	96.6 ± 3.4 a	0.8 ± 0.8 b	2.0 ± 2.0 a
Un	4	0.0 ± 0.0 a	92.5 ± 2.9 ab	6.6 ± 3.3 abc	0.9 ± 0.4 a	0.0 ± 0.0 a	62.9 ± 17.0 ab	18.0 ± 11.3 ab	19.1 ± 16.8 a
9	4	0.0 ± 0.0 a	98.8 ± 0.6 a	1.2 ± 0.6 bc	0.0 ± 0.0 a	0.0 ± 0.0 a	96.4 ± 1.8 ab	3.6 ± 1.8 b	0.0 ± 0.0 a
7	4	0.0 ± 0.0 a	87.9 ± 5.7 ab	11.5 ± 6.0 abc	0.6 ± 0.6 a	0.0 ± 0.0 a	82.0 ± 8.7 ab	15.4 ± 9.0 ab	2.6 ± 2.6 a
3	4	0.4 ± 0.4 a	65.4 ± 11.8 b	33.3 ± 12.6 a	0.9 ± 0.6 a	0.1 ± 0.1 a	49.2 ± 14.1 b	46.1 ± 15.0 a	4.6 ± 3.9 a
1	4	1.7 ± 1.7 a	76.1 ± 8.7 b	20.8 ± 7.6 ab	1.3 ± 0.5 a	6.3 ± 6.3 a	61.6 ± 14.6 ab	15.5 ± 8.2 ab	16.6 ± 15.2 a

Table C.8. Benthic macroinvertebrates by season a)Mean±standard error of abundance (Ab, #/m^2) and biomass (Bm, mg DM/m^2) of macroinvertebrates by season. Values with different letters are significantly different based on One-way ANOVA ($p<0.05$). Abundance and biomass were log transformed, and percents were arcsine square root transformed for analysis. Corbicula were excluded from all analyses. b)Mean±standard error of percent functional feeding group composition of macroinvertebrates by abundance (Ab) and biomass (Bm) by season. Values with different letters are significantly different based on one-way ANOVA ($p<0.05$). Percents were arcsine square root transformed for analysis. *Corbicula* were excluded from all analyses. (CF=Collector-Filterer, CG=Collector-Gatherer, P=Predator, SC=Scaper; no shredders were found in the benthos).

a)

Season	n	Total Ab	Insect Ab	Non-Chironomid Insect Ab	% Chironomid Ab	Total Bm	Insect Bm	Non-Chironomid Insect Bm	% Chironomid Bm
Summer 2002	24	380.4 ± 45.9 a	343.8 ± 44.5 a	10.0 ± 2.4 a	77.3 ± 4.4 a	26.1 ± 3.6 a	21.6 ± 3.4 a	5.1 ± 2.6 a	60.5 ± 5.7 a
Fall 2002	24	65.1 ± 13.0 c	58.4 ± 12.5 b	5.4 ± 1.4 a	60.3 ± 6.8 a	4.9 ± 0.9 b	3.5 ± 0.6 b	0.6 ± 0.2 a	46.0 ± 7.3 a
Summer 2003	24	101.5 ± 14.2 b	81.5 ± 13.1 b	6.5 ± 1.8 a	65.1 ± 6.1 a	5.6 ± 2.1 b	2.4 ± 0.4 b	0.5 ± 0.2 a	45.4 ± 8.0 a

b)

Season	n	% CF Ab	% CG Ab	% P Ab	% SC Ab	% CF Bm	% CG Bm	% P Bm	% SC BM
Summer 2002	24	0.8 ± 0.5 a	92.8 ± 1.2 a	6.1 ± 1.0 a	0.4 ± 0.1 a	2.6 ± 2.0 a	83.3 ± 4.6 a	13.5 ± 4.3 a	0.6 ± 0.2 a
Fall 2002	24	0.0 ± 0.0 a	79.7 ± 6.2 a	19.7 ± 6.3 a	0.6 ± 0.3 a	0.0 ± 0.0 a	74.5 ± 6.6 a	21.5 ± 6.6 a	4.0 ± 2.7 a
Summer 2003	24	0.5 ± 0.3 a	86.5 ± 3.5 a	12.3 ± 3.4 a	0.7 ± 0.2 a	1.2 ± 1.1 a	74.8 ± 5.6 a	16.5 ± 4.5 a	7.5 ± 3.8 a

Table C.9. Benthic macroinvertebrates by bed sediment particle size. a) Mean±standard error of abundance (Ab, #/m^2) and biomass (Bm, mg DM/m^2) of macroinvertebrates by sediment type. Values with different letters are significantly different based on One-way ANOVA ($p<0.05$). Abundance and biomass were log transformed, and percents were arcsine square root transformed for analysis. Corbicula were excluded from all analyses. b) Mean±standard error of percent functional feeding group composition of macroinvertebrates by abundance (Ab) and biomass (Bm) by sediment type. Values with different letters are significantly different based on one-way ANOVA ($p<0.05$). Percents were arcsine square root transformed for analysis. *Corbicula* were excluded from all analyses. (CF=Collector-Filterer, CG=Collector-Gatherer, P=Predator, SC=Scaper; no shredders were found in the benthos).

a)

Bed Sediment	n	Total Ab	Insect Ab	Non-Chironomid Insect Ab	% Chironomid Ab	Total Bm	Insect Bm	Non-Chironomid Insect Bm	% Chironomid Bm
cobble	6	245.2 ± 93.8 a	193.2 ± 71.8 a	20.8 ± 7.6 a	63.5 ± 6.8 a	21.1 ± 9.3 a	17.9 ± 8.7 a	10.5 ± 6.8 a	25.9 ± 8.3 a
gravel	20	178.0 ± 38.5 a	157.3 ± 38.3 a	5.6 ± 1.7 a	79.5 ± 5.9 a	10.3 ± 3.0 a	8.8 ± 2.9 a	2.9 ± 2.3 a	65.1 ± 7.3 a
rock	2	28.0 ± 0.0 a	21.0 ± 5.0 a	0.0 ± 0.0 a	73.1 ± 19.8 a	4.4 ± 0.5 a	3.1 ± 1.7 a	0.0 ± 0.0 a	65.6 ± 31.6 a
sand	42	189.4 ± 32.8 a	171.6 ± 31.3 a	6.6 ± 1.2 a	64.0 ± 4.7 a	12.7 ± 2.4 a	8.7 ± 1.9 a	0.6 ± 0.1 a	47.9 ± 5.5 a
silt	2	44.0 ± 22.0 a	28.0 ± 21.0 a	4.5 ± 3.5 a	28.3 ± 2.3 a	2.9 ± 1.7 a	1.9 ± 1.4 a	0.7 ± 0.6 a	22.7 ± 10.5 a

b)

Bed Sediment	n	% CF Ab	% CG Ab	% P Ab	% SC Ab	% CF Bm	% CG Bm	% P Bm	% SC Bm
cobble	6	3.5 ± 1.7 a	82.0 ± 6.1 a	13.6 ± 5.3 a	0.9 ± 0.4 a	12.9 ± 7.9 a	57.4 ± 12.0 a	28.1 ± 12.2 a	1.5 ± 0.7 a
gravel	20	0.2 ± 0.2 b	91.4 ± 4.0 a	8.2 ± 4.0 a	0.3 ± 0.1 a	0.1 ± 0.1 b	84.7 ± 5.5 a	14.2 ± 5.6 a	0.9 ± 0.5 a
rock	2	0.0 ± 0.0 b	100.0 ± 0.0 a	0.0 ± 0.0 a	0.0 ± 0.0 a	0.0 ± 0.0 b	100.0 ± 0.0 a	0.0 ± 0.0 a	0.0 ± 0.0 a
sand	42	0.1 ± 0.1 b	83.9 ± 3.6 a	15.4 ± 3.6 a	0.6 ± 0.2 a	0.3 ± 0.3 b	76.2 ± 4.4 a	17.7 ± 4.0 a	5.8 ± 2.6 a
silt	2	0.0 ± 0.0 b	86.4 ± 4.5 a	11.4 ± 6.8 a	2.3 ± 2.3 a	0.0 ± 0.0 b	71.7 ± 1.7 a	18.2 ± 11.8 a	10.1 ± 10.1 a

Table C.10. Benthic macroinvertebrates by flow type. a) Mean±standard error of abundance (Ab, #/m^2) and biomass (Bm, mg DM/m^2) of macroinvertebrates by flow type. Values with different letters are significantly different based on One-way ANOVA ($p<0.05$). Abundance and biomass were log transformed, and percents were arcsine square root transformed for analysis. Corbicula were excluded from all analyses. b) Mean±standard error of percent functional feeding group composition of macroinvertebrates by abundance (Ab) and biomass (Bm) by flow type. Values with different letters are significantly different based on one-way ANOVA ($p<0.05$). Percents were arcsine square root transformed for analysis. *Corbicula* were excluded from all analyses. (CF=Collector-Filterer, CG=Collector-Gatherer, P=Predator, SC=Scaper; no shredders were found in the benthos).

a)

Flow Type	n	Total Ab	Insect Ab	Non-Chironomid Insect Ab	% Chironomid Ab	Total Bm	Insect Bm	Non-Chironomid Insect Bm	% Chironomid Bm
pool	15	156.9 ± 55.2 b	142.0 ± 52.2 a	6.1 ± 1.9 a	64.6 ± 7.4 a	15.1 ± 5.2 ab	10.6 ± 4.3 a	0.6 ± 0.2 b	55.3 ± 9.2 a
riffle	10	334.2 ± 71.5 a	275.7 ± 71.7 a	15.6 ± 5.2 a	72.4 ± 9.4 a	23.8 ± 6.6 a	20.7 ± 6.7 a	11.3 ± 5.7 a	49.5 ± 11.9 a
run	47	158.2 ± 25.7 b	143.0 ± 24.5 a	5.9 ± 1.1 a	67.5 ± 4.3 a	8.8 ± 1.5 b	6.2 ± 1.1 a	0.6 ± 0.1 b	49.4 ± 5.1 a

b)

Flow Type	n	% CF Ab	% CG Ab	% P Ab	% SC Ab	% CF Bm	% CG Bm	% P Bm	% SC Bm
pool	15	0.0 ± 0.0 b	83.9 ± 5.9 a	15.6 ± 6.0 a	0.5 ± 0.2 a	0.0 ± 0.0 b	77.1 ± 7.6 a	18.1 ± 7.2 a	4.7 ± 4.1 a
riffle	10	2.4 ± 1.1 a	90.0 ± 4.5 a	7.0 ± 3.7 a	0.6 ± 0.3 a	8.1 ± 5.0 a	69.4 ± 10.6 a	20.8 ± 10.0 a	1.7 ± 0.8 a
run	47	0.1 ± 0.1 b	86.3 ± 3.1 a	13.0 ± 3.2 a	0.5 ± 0.2 a	0.2 ± 0.2 b	79.4 ± 3.8 a	16.1 ± 3.5 a	4.3 ± 2.0 a

Table C.11. Seasonal mean±standard error for benthic organic matter (g AFDM/m^2) by site. VFBOM is < 250 µm. FBOM is 250 µm-1 mm. CBOM is > 1 mm. Values with different letters are significantly different based on one-way ANOVA ($p< 0.05$). a)Summer 2002, b)Fall 2002, c) Summer 2003.

a)

Site	n	VFBOM	FBOM	CBOM	Total BOM
Ref	4	45.8 ± 5.9 a	20.6 ± 11.5 a	288.8 ± 259.6 ab	355.2 ± 271.5 a
Un	4	36.9 ± 5.5 a	18.6 ± 4.1 a	35.8 ± 8.6 ab	91.3 ± 16.4 a
9	4	29.6 ± 10.6 a	23.8 ± 6.9 a	504.4 ± 106.5 a	557.8 ± 115.5 a
7	4	34.3 ± 12.2 a	299.4 ± 185.1 a	28.0 ± 9.2 b	361.7 ± 189.9 a
3	3	145.7 ± 118.2 a	166.3 ± 80.7 a	156.3 ± 93.1 ab	468.3 ± 220.9 a
1	4	28.5 ± 4.8 a	31.3 ± 6.7 a	485.2 ± 152.6 ab	544.9 ± 145.9 a

b)

Site	n	VFBOM	FBOM	CBOM	Total BOM
Ref	4	69.4 ± 31.2 a	9.6 ± 1.4 a	29.3 ± 7.2 ab	108.4 ± 28.7 a
Un	4	110.9 ± 48.7 a	23.7 ± 13.0 a	71.7 ± 20.0 ab	206.3 ± 65.0 a
9	4	32.9 ± 5.1 a	18.9 ± 4.7 a	106.8 ± 51.3 a	158.6 ± 57.8 a
7	4	57.0 ± 10.5 a	273.8 ± 249.4 a	15.3 ± 4.5 b	346.1 ± 255.6 a
3	4	234.0 ± 186.9 a	124.2 ± 103.2 a	84.2 ± 25.2 ab	442.4 ± 220.7 a
1	4	18.4 ± 6.1 a	221.4 ± 209.9 a	90.9 ± 48.7 ab	330.6 ± 204.4 a

c)

Site	n	VFBOM	FBOM	CBOM	Total BOM
Ref	4	45.9 ± 26.3 a	19.1 ± 3.5 a	27.9 ± 13.6 abc	92.9 ± 21.0 a
Un	4	135.7 ± 97.3 a	26.3 ± 4.0 a	48.1 ± 10.0 ab	210.1 ± 110.1 a
9	4	45.0 ± 14.6 a	24.4 ± 4.3 a	237.3 ± 125.7 ab	306.8 ± 123.3 a
7	4	59.3 ± 12.7 a	81.4 ± 38.9 a	3.9 ± 0.9 c	144.5 ± 51.4 a
3	4	24.1 ± 5.1 a	22.2 ± 5.1 a	18.7 ± 8.8 bc	65.0 ± 7.8 a
1	4	34.1 ± 5.8 a	22.2 ± 6.3 a	453.5 ± 353.0 a	509.8 ± 360.9 a

Table C.12. Mean±standard error for benthic organic matter (g AFDM/m^2). VFBOM is < 250 µm. FBOM is 250 µm-1 mm. CBOM is > 1 mm. Values with different letters are significantly different based on one-way ANOVA with alpha of 0.05. a)by season, b)by bed sediment particle size, and c)by flow type.

a)

Season	n	VFBOM	FBOM	CBOM	Total BOM
Summer 2002	23	49.5 ± 15.4 a	90.14 ± 37.64 a	253.8 ± 65.5 a	393.4 ± 71.96 a
Fall 2002	24	87.1 ± 32.6 a	112 ± 54.94 a	66.37 ± 13.4 b	265.4 ± 64.51 a
Summer 2003	24	57.4 ± 17 a	32.6 ± 7.498 a	131.6 ± 65 ab	221.5 ± 67.01 a

b)

Substrate	n	VFBOM	FBOM	CBOM	Total BOM
cobble	6	31.7 ± 5.9 a	294.4 ± 171.5 a	48.1 ± 9.4 a	374.2 ± 166.6 a
gravel	20	48.6 ± 10.4 a	32.5 ± 14.4 a	61.8 ± 22.8 a	142.9 ± 27.9 a
rock	2	39.3 ± 4.3 a	30.2 ± 1.6 a	14.8 ± 10.2 a	84.3 ± 16.2 a
sand	41	78.5 ± 22.3 a	63.8 ± 25.5 a	218.2 ± 51.6 a	360.4 ± 59.0 a
silt	2	73.7 ± 37.7 a	225.5 ± 207.3 a	43.7 ± 39.1 a	342.8 ± 284.1 a

c)

Flow Type	n	VFBOM	FBOM	CBOM	Total BOM
pool	15	53.9 ± 12.9 a	60.2 ± 20.8 a	241.0 ± 109.3 a	355.1 ± 108.0 a
riffle	10	31.3 ± 4.7 a	102.9 ± 80.0 a	81.8 ± 42.4 a	216.0 ± 85.0 a
run	46	75.7 ± 20.0 a	78.5 ± 29.6 a	133.8 ± 32.4 a	288.0 ± 46.9 a

APPENDIX D

BENTHIC SEDIMENT PARTICLE SIZE COMPOSITION

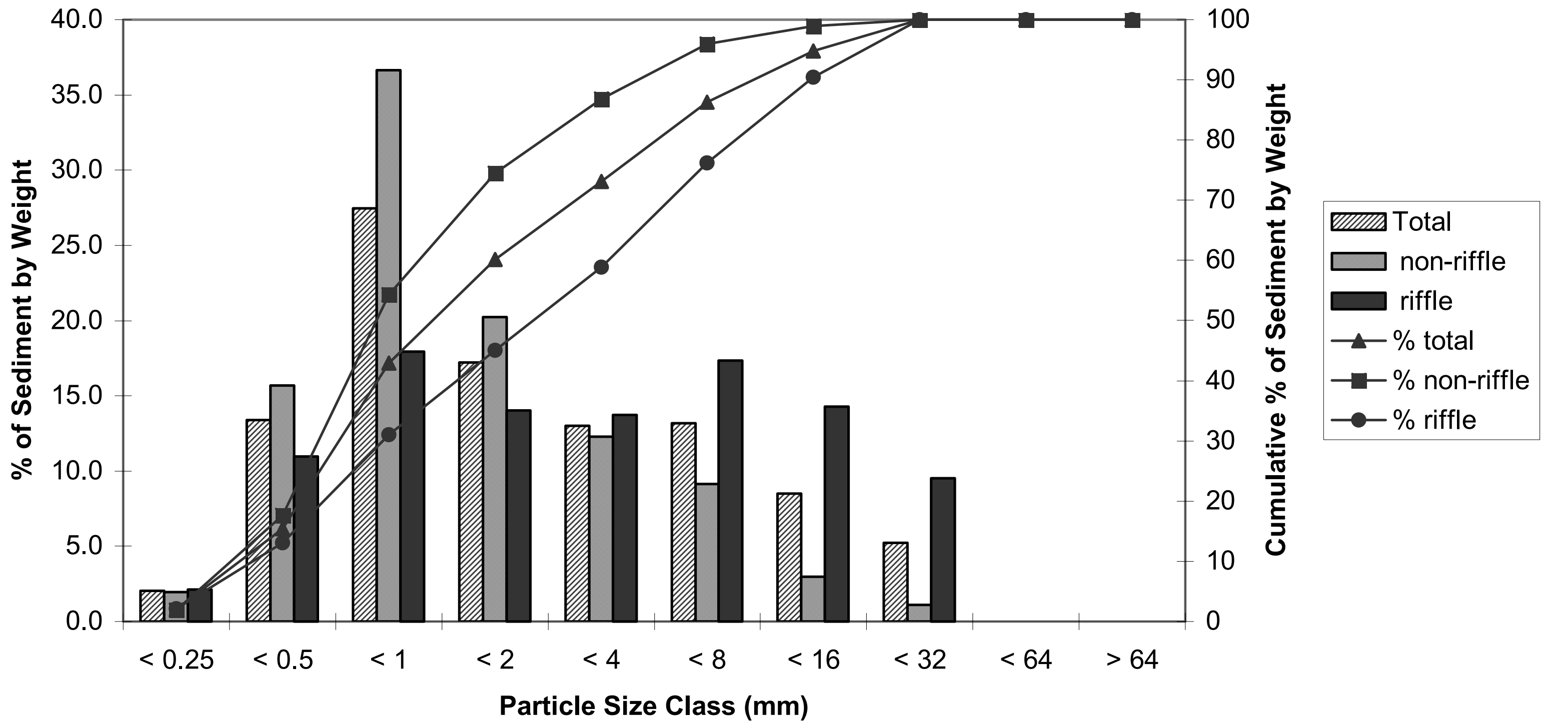


Figure D.1. Reference Site sediment composition. Bars indicate percent composition by weight in total, in non-riffle areas, and in riffle areas. Lines indicate cumulative percent composition by weight in total, in non-riffle areas, and in riffle areas.

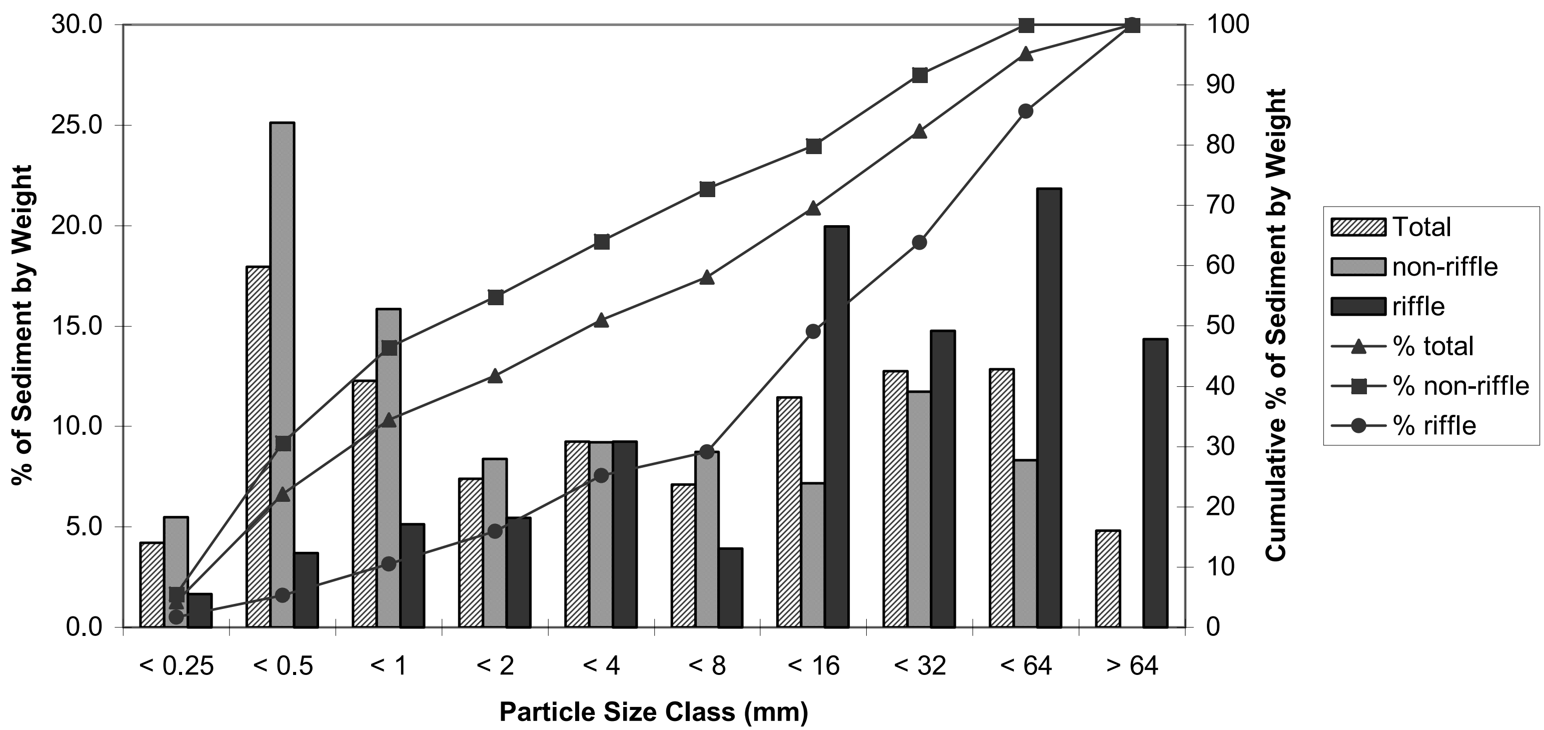


Figure D.2. Unrestored Site sediment composition. Bars indicate percent composition by weight in total, in non-riffle areas, and in riffle areas. Lines indicate cumulative percent composition by weight in total, in non-riffle areas, and in riffle areas.

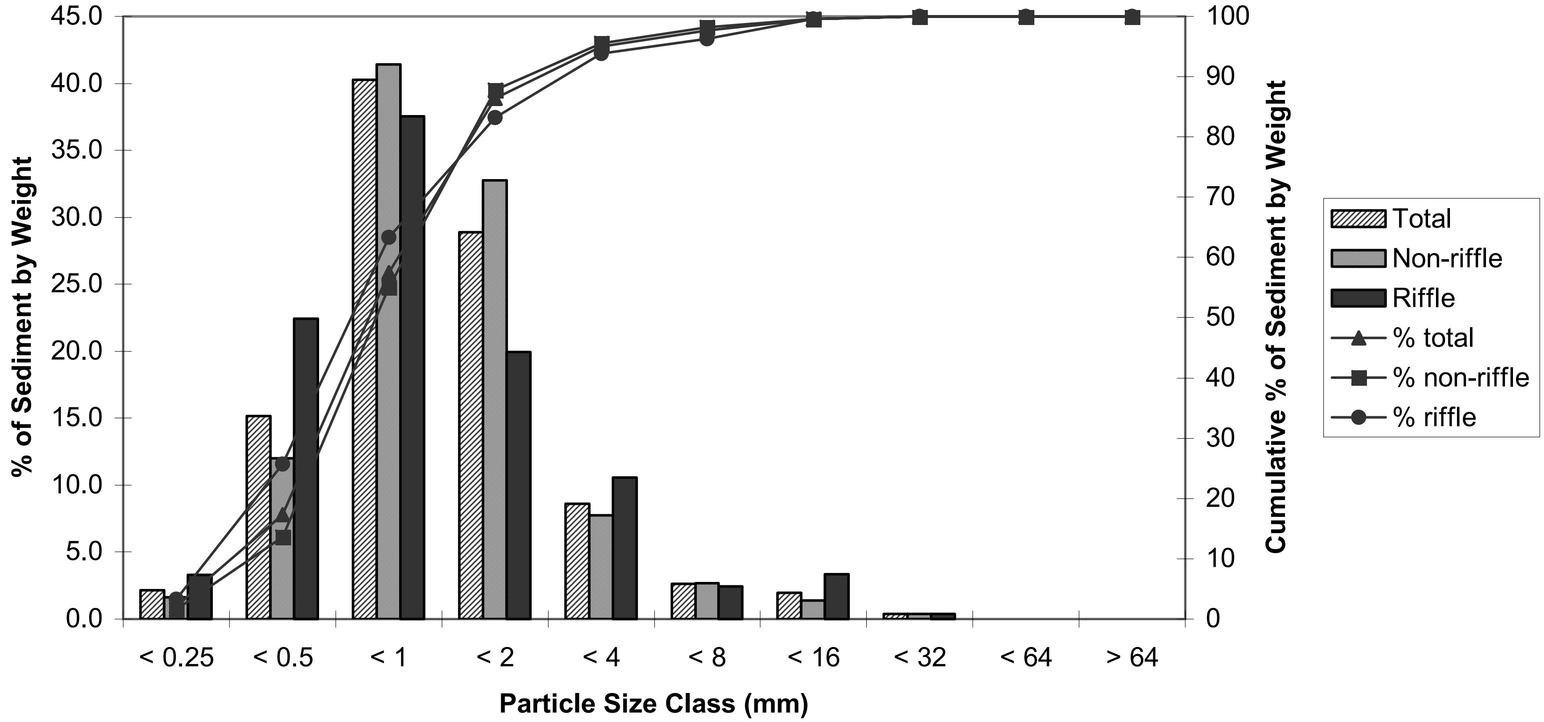


Figure D.3. Site 9 sediment composition. Bars indicate percent composition by weight in total, in non-riffle areas, and in riffle areas. Lines indicate cumulative percent composition by weight in total, in non-riffle areas, and in riffle areas.

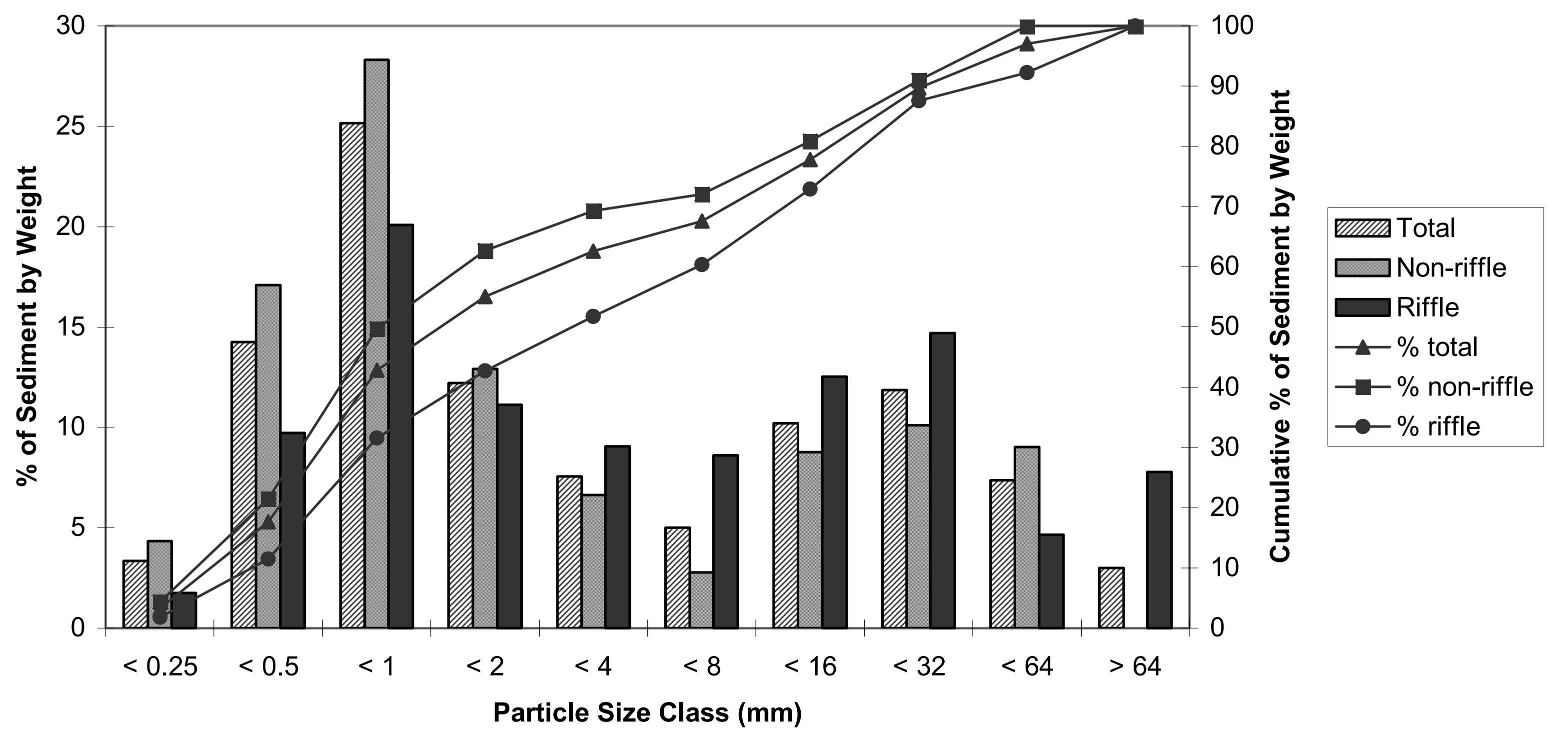


Figure D.4. Site 7 sediment composition. Bars indicate percent composition by weight in total, in non-riffle areas, and in riffle areas. Lines indicate cumulative percent composition by weight in total, in non-riffle areas, and in riffle areas.

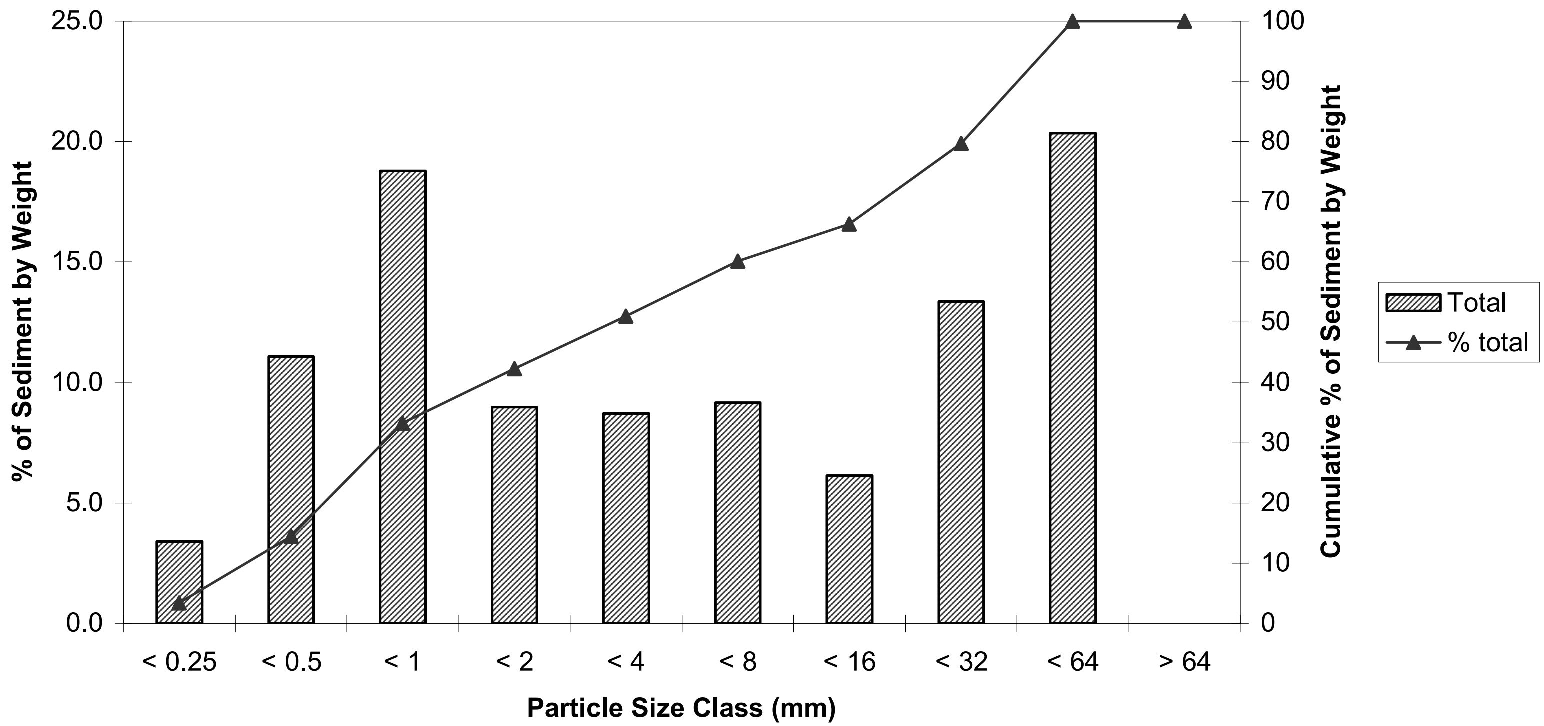


Figure D.5. Site 3 sediment composition. Bars indicate percent composition by weight. Lines indicate cumulative percent composition by weight. No riffles were found in the study reach at Site 3 during sediment sampling.

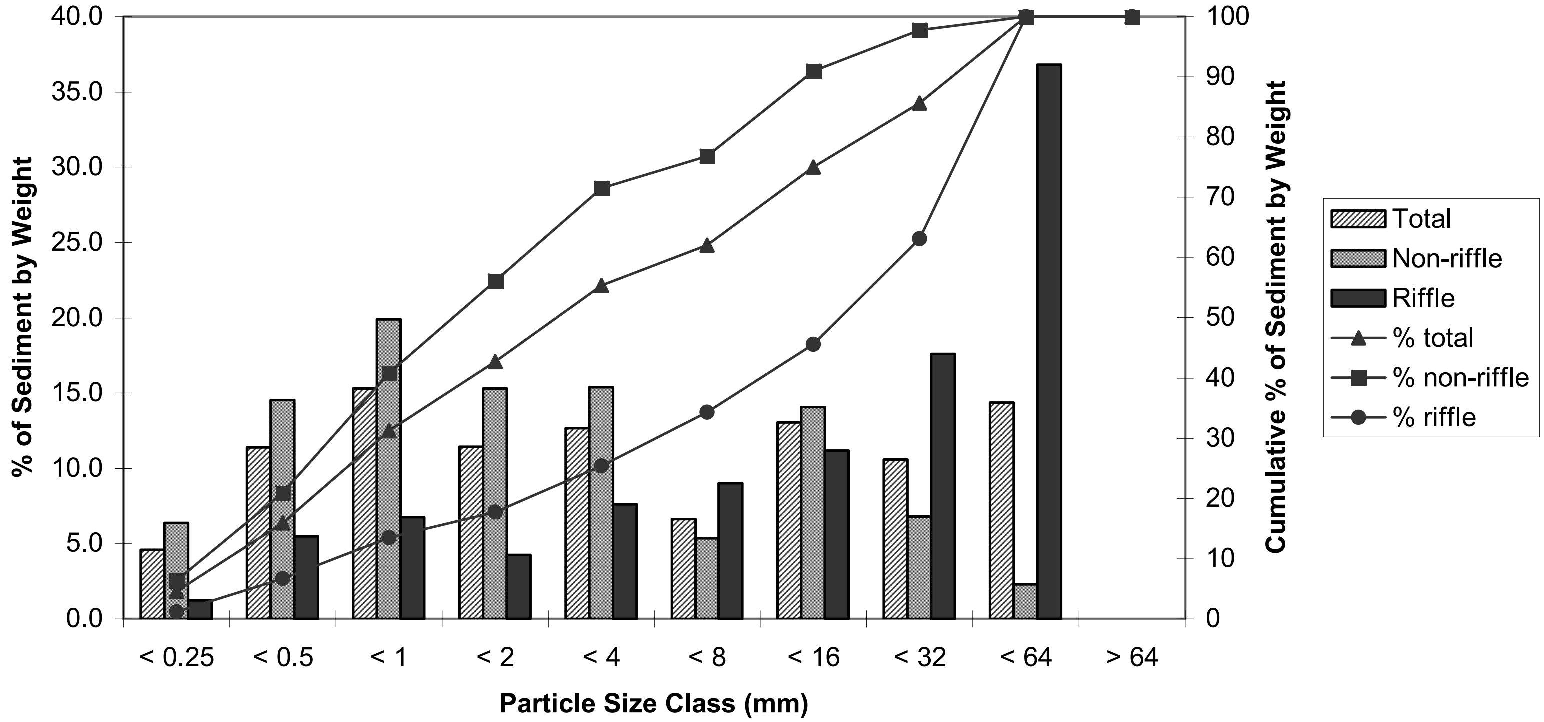


Figure D.6. Site 1 sediment composition. Bars indicate percent composition by weight in total, in non-riffle areas, and in riffle areas. Lines indicate cumulative percent composition by weight in total, in non-riffle areas, and in riffle areas.

APPENDIX E

SITE CROSS-SECTIONS AND STORMFLOWS

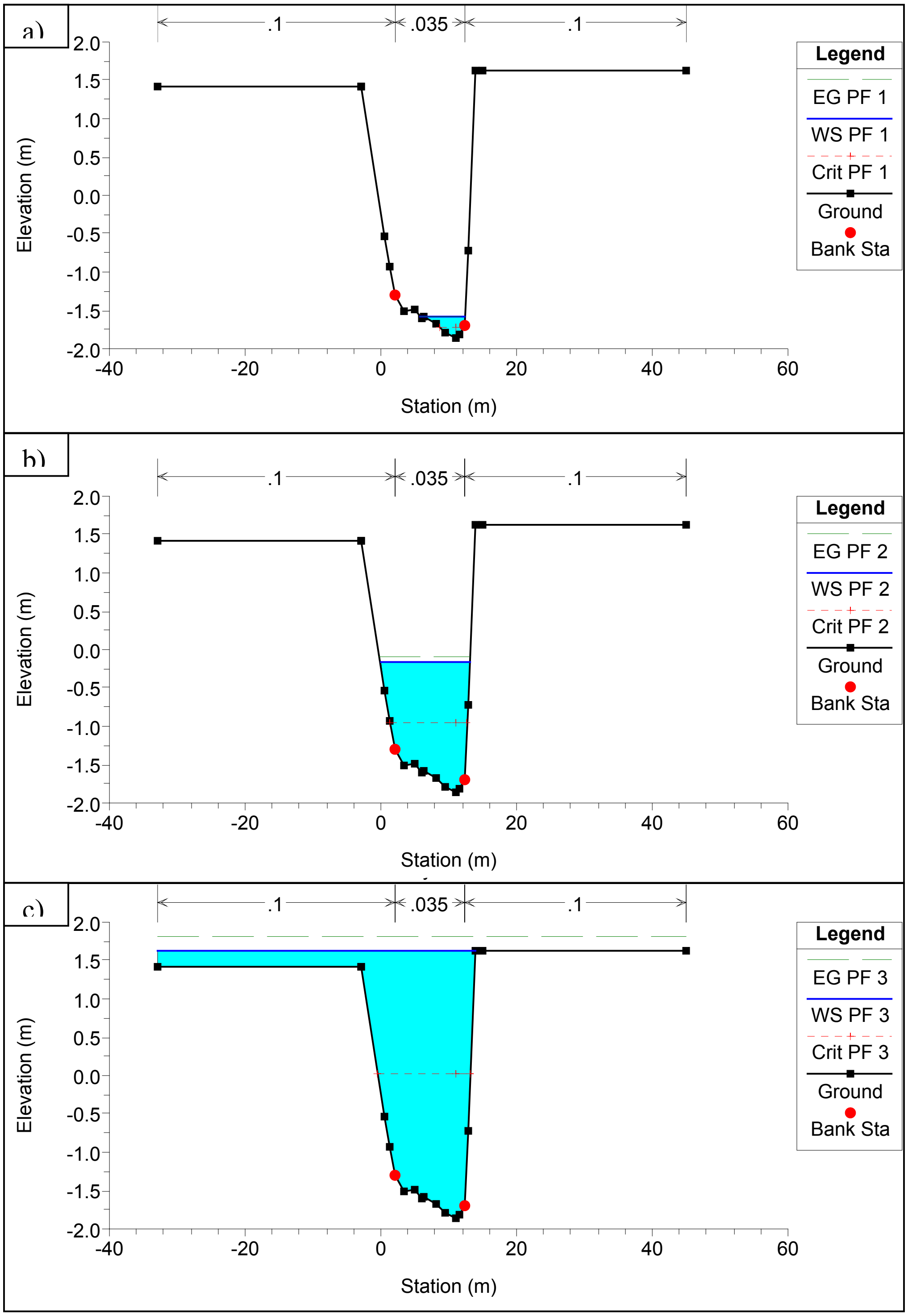


Figure E.1. Reference Site cross-sections. a)baseflow: 0.23 m^3/s, b)1/2-year recurrence interval flood: 18.0 m^3/s, c)2-year recurrence interval flood: 69.7 m^3/s.

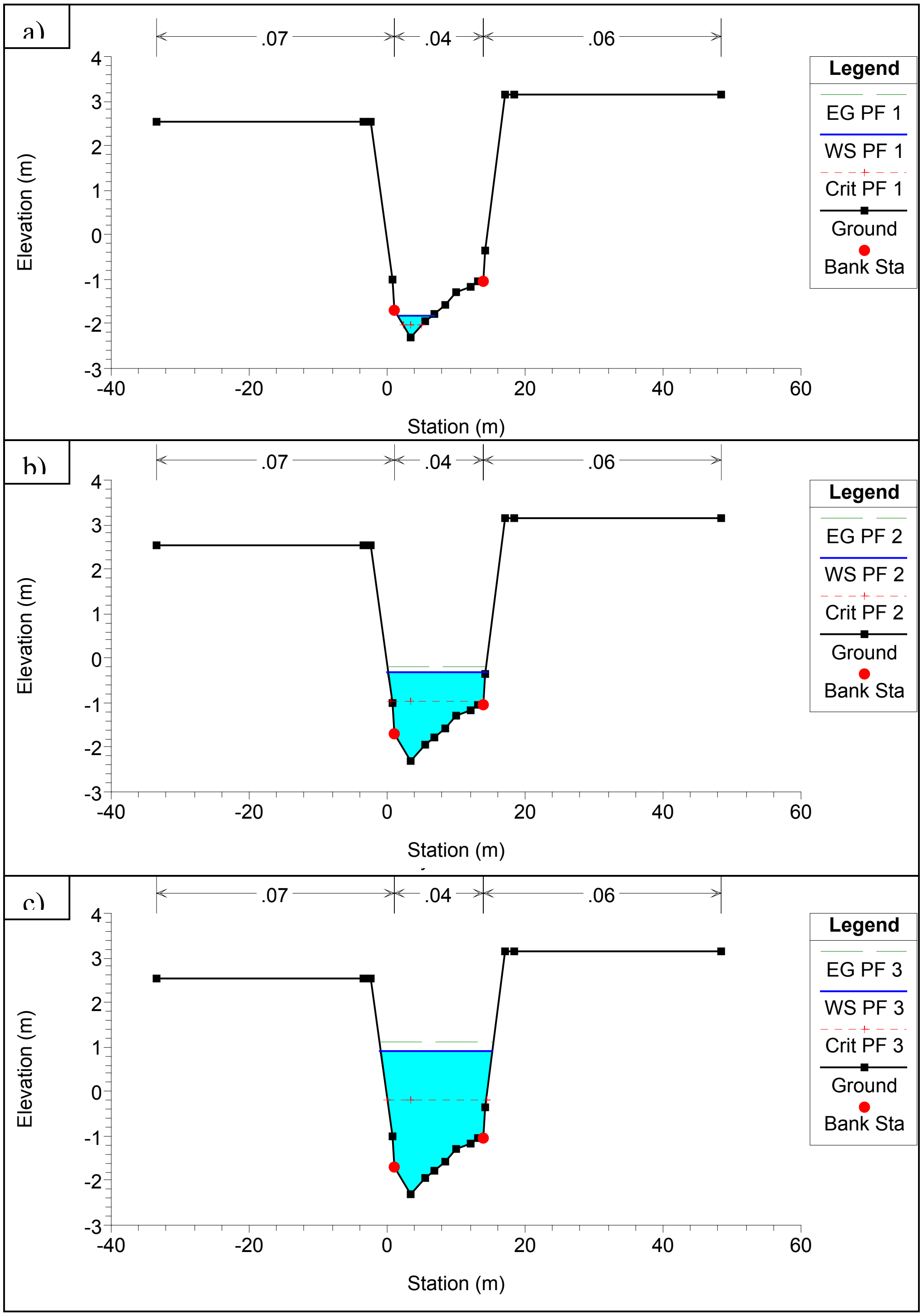


Figure E.2. Unrestored Site cross-sections. a)baseflow: 0.49 m^3/s, b)1/2-year recurrence interval flood: 23.5 m^3/s, c)2-year recurrence interval flood: 93.8 m^3/s.

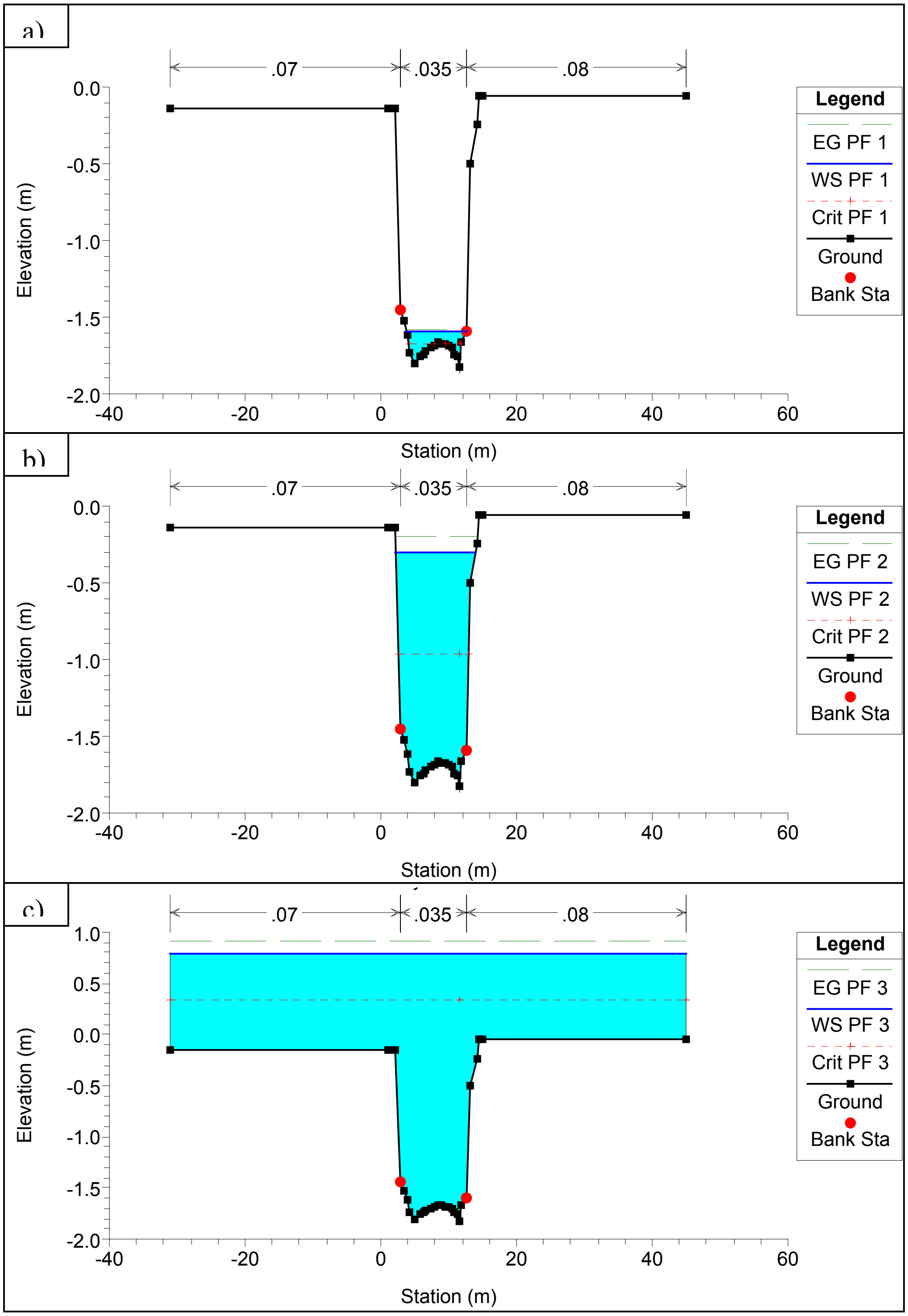


Figure E.3. Site 9 cross-sections. a)baseflow: 0.25 m^3/s, b)1/2-year recurrence interval flood: 19.0 m^3/s, c)2-year recurrence interval flood: 74.8 m^3/s.

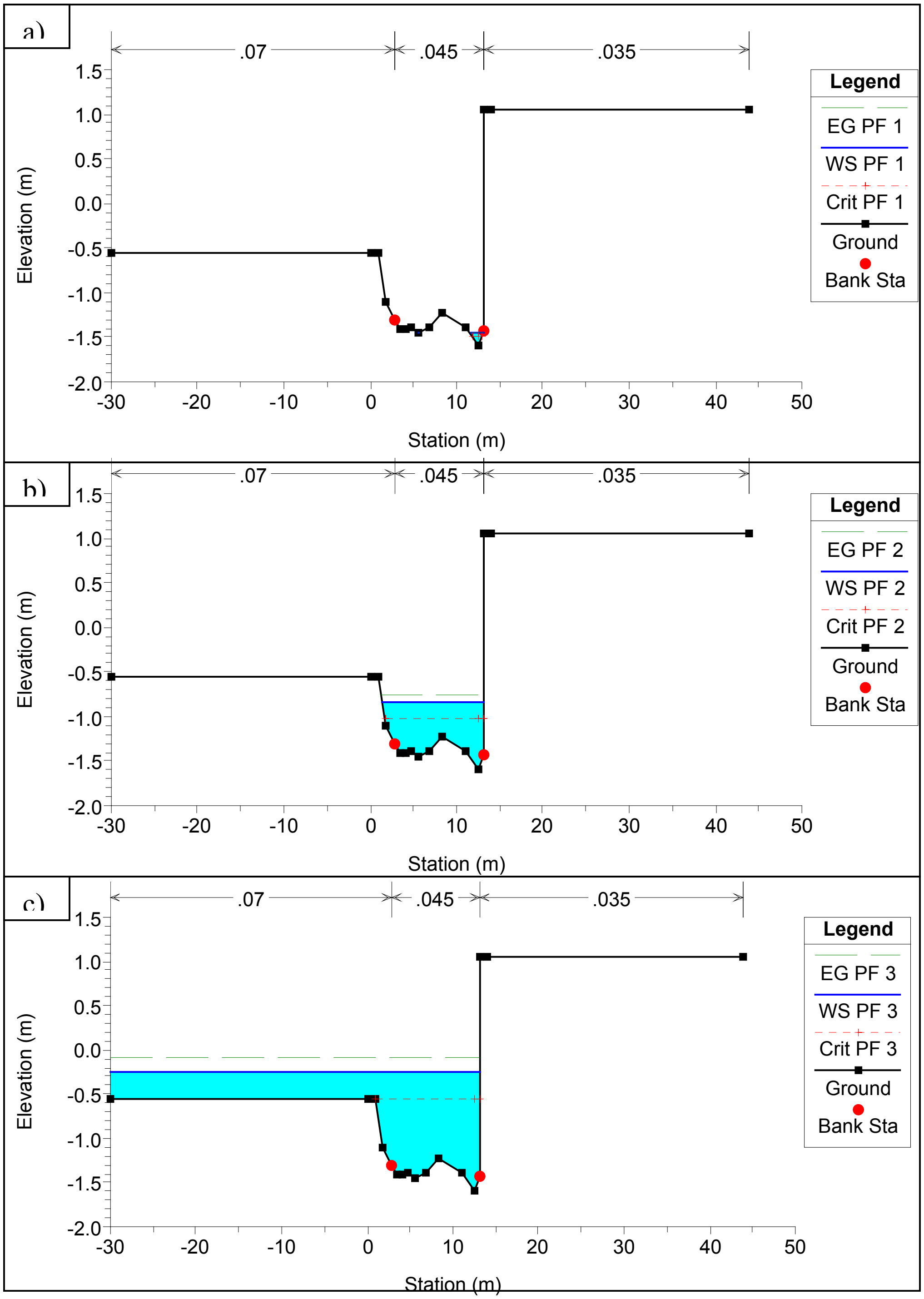


Figure E.4. Site 7 cross-sections. a)baseflow: 0.03 m^3/s, b)1/2-year recurrence interval flood 7.0 m^3/s, c)2-year recurrence interval flood: 29.0 m^3/s.

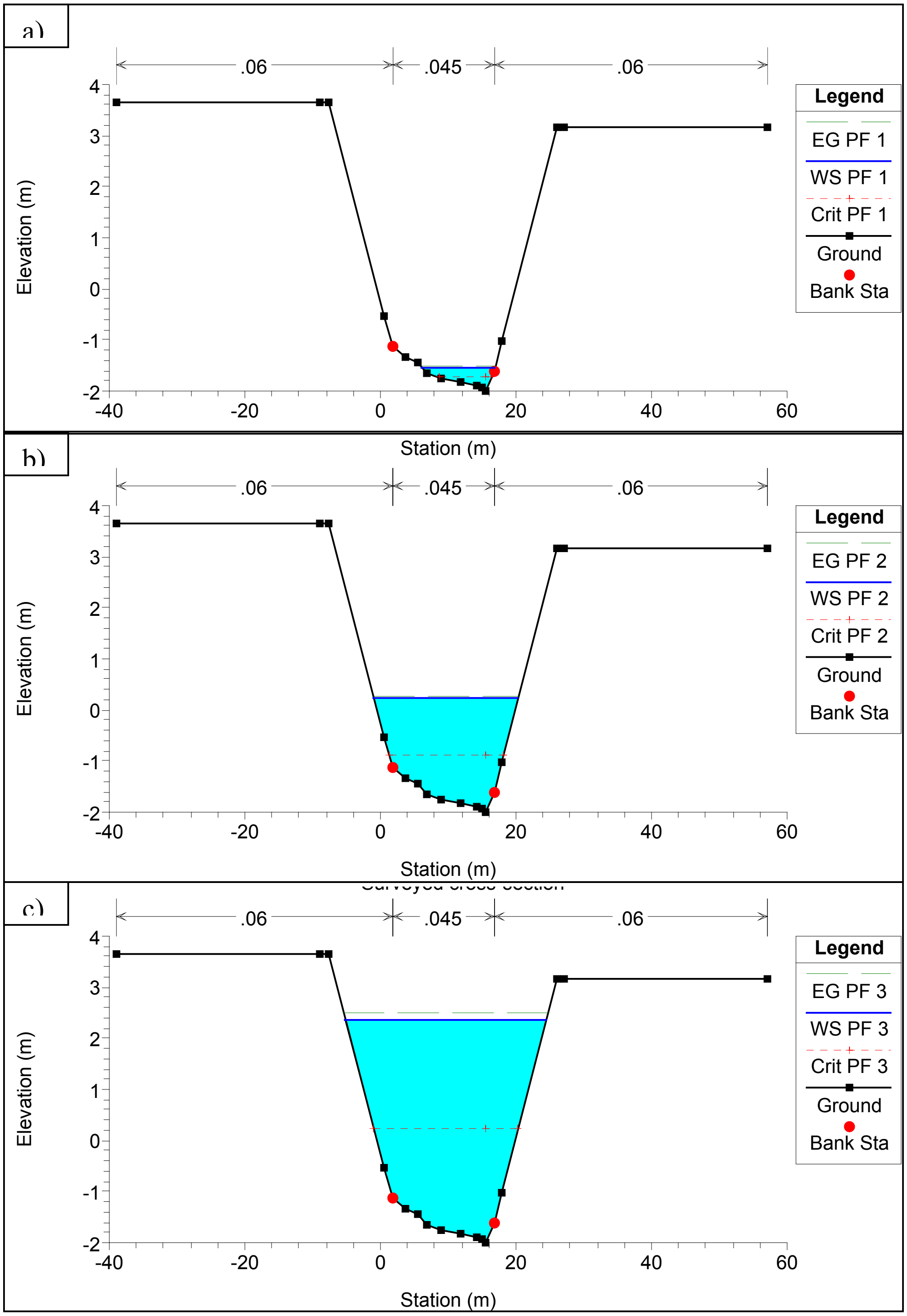


Figure E.5. Site 3 cross-sections. a)baseflow: 0.79 m^3/s, b)1/2-year recurrence interval flood: 32.5 m^3/s, c)2-year recurrence interval flood: 128.1 m^3/s.

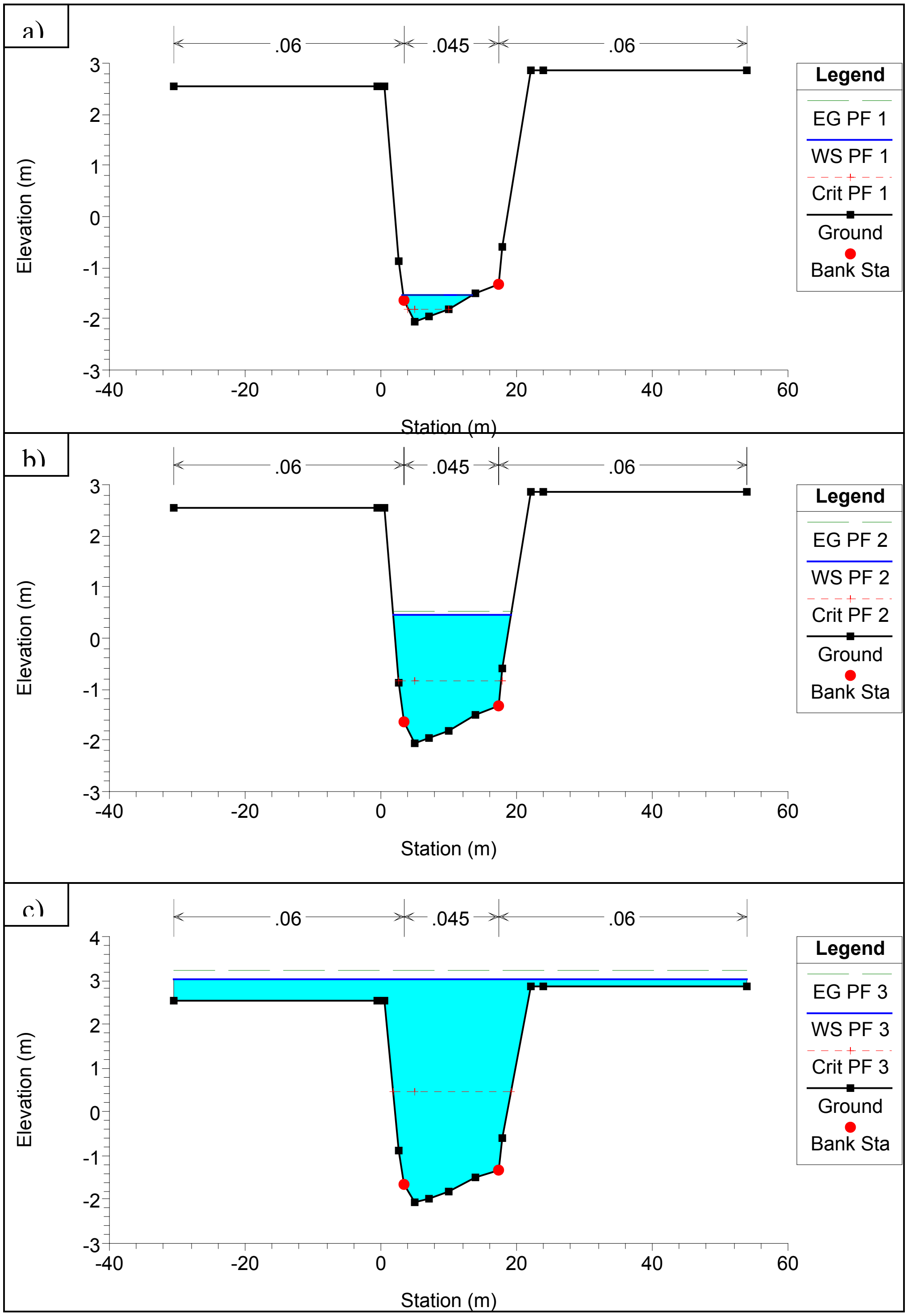


Figure E.6. Site 1 cross-sections a)baseflow: 0.95 m^3/s, b)1/2-year recurrence interval flood: 37.0 m^3/s, c)2-year recurrence interval flood: 144.0 m^3/s.

APPENDIX F

RULES FOR BANK HABITAT CLASSIFICATION

Bank habitat was surveyed at all sites by visual, qualitative descriptions. An overall survey was done at each site, and the same descriptive system was used for each bank sample taken. Bank habitat descriptions consisted of at most three habitat types—the primary habitat present, and up to two others also present (e.g., undercut clay with roots, sand with wood and rocks). For analyses, these were classified into five groups: roots, wood, mud, sand, and rock.

Roots—Any habitat with roots as the first or second habitat type. This includes fine roots, large tree roots, and undercut clay with roots. Any habitat with vegetation as the first or second habitat type. This includes aquatic vegetation, willows, and kudzu. If roots is first habitat type and wood is second, the site is classified as roots.

Wood—Any habitat with wood as the first or second habitat type. This includes large woody debris, small woody debris, and sand with wood. If wood is first habitat type and roots is second, the site is classified as wood.

Sand—Any habitat with sand as the first habitat type that doesn't have roots or wood as the second habitat type.

Rock—Any habitat with rock as the first habitat type that doesn't have roots or wood as the second habitat type. This includes all sizes from gravel to bedrock.

Mud—Any inorganic habitat that is not sand or rock as the first habitat type and that doesn't have roots or wood as the second habitat type. This includes eroding clay or soil banks as well as silty (not sandy).