IMPACTS OF SUSPENDED AND DEPOSITED SEDIMENT ON BENTHIC INVERTEBRATES AND FISHES IN A MISSOURI OZARK STREAM

A Thesis Presented to the Faculty of the Graduate School

University of Missouri-Columbia

In Partial Fulfillment

of the Requirements for the Degree

Master of Science

By

ZACHARY L. FORD

Dr. Charles F. Rabeni, Thesis Supervisor

MAY 2006

The undersigned, appointed by the Dean of the Graduate School, have examined the thesis entitled

IMPACTS OF SUSPENDED AND DEPOSITED SEDIMENT ON BENTHIC INVERTEBRATES AND FISHES IN A MISSOURI OZARK STREAM

Presented by Zachary L. Ford

A candidate for the degree of Master of Science

And hereby certify that in their final opinion it is worthy of acceptance.

__________Charles F. Rabeni_________
Professor Charles F. Rabeni

___________David L. Galat___________
Professor David L. Galat

_________Allen L. Thompson _________
Professor Allen L. Thompson

ACKNOWLEDGEMENTS

I would like to thank a mosaic of people who were instrumental in my professional development and success during graduate school. First, and foremost, I thank my advisor, Dr. Charles Rabeni, for his guidance, determination, and friendship and for providing many opportunities that helped inspire my professional development. This research would not have been possible without funding from the U. S. Fish and Wildlife Service which was part of a larger Missouri Department of Conservation project (Restoration of the Osage River Basin in Missouri). Many thanks are extended to Leanna Zweig for her valued assistance and advice.

I am very grateful for the service of my committee members, Dr. David Galat and Dr. Allen Thompson. I had several technicians (Andy Turner, Rex Jackson, Mike Kromrey, Gregory Rublaitus, and Matt Keener) that stayed with me through many circumstances and deserve much thanks. I especially thank Greg Wallace for his expert and expeditious identification of macroinvertebrate samples. Without the hard work and assistance of these people in the field and laboratory, this project would not have been a success. Further, the advice, support, and assistance in the field (with short term notice) provided by my fellow colleagues (Kevin O'Donnell, Jason Vokoun, Kathy Doisy, Shannon Brewer, and Emily Tracy) were invaluable during this process. I also thank Dr. Mark Ellersieck and Mike McManus for their valuable consultations.

Finally, I thank my wonderful wife and family who provided unconditional support and encouragement every step of the way. Without this stable foundation, I would not be where I am today.

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ABSTRACT

Sediment is suspected in the decline of sensitive-aquatic organisms in the Osage River basin of Missouri. Although transporting sediment is a primary function of a stream, it becomes a problem when this sediment load increases to the point where it is greater than the stream's historic-sediment regime. In this study, I monitored suspended and deposited sediment dynamics and evaluated corresponding linkages with the benthic invertebrate and fish assemblages as it related to highway construction activity adjacent to a small Ozark stream in southwest Missouri. Additional anthropogenic effects were examined due to agricultural land use in the watershed.

Water samples were collected using single-stage and automated samplers during 20 rain events throughout the study at sites upstream and downstream of the highway construction. Deposited sediment was quantified using pit-type traps in which the sediment was collected during alternate months throughout the study. The sediment sampling methods used in this study were determined to be effective at quantifying the sediment dynamics in a stream for a long duration.

The most notable effect of road construction on the sediment dynamics in the Brush Creek watershed was the overall change in suspended sediment concentration which was 53% greater downstream of the highway versus upstream after the start of road construction. Additionally, significant increases in suspended sediment existed at

downstream compared to upstream sites for 44% of rain events during construction activity. No significant trends in deposited sediment measurements were evident after the start of road construction, however, significant differences were found between sample periods and habitats. A strong correlation with discharge suggested the deposited sediment sampler used in this study collected primarily bedload sediment rather than sediment that settled out of suspension.

The lack of a marked shift in biomonitoring metrics and composition of biotic assemblages during road construction reflected similar, non-significant trends in deposited sediment. Significant correlations between sediment measurements and community metrics were found for macroinvertebrate and fish assemblages but these differed among habitats. Ordination analysis showed suspended sediment and surface cover of deposited sediment influenced the composition of the macroinvertebrate assemblage immediately before and after the start of road construction.

The results of my study provide needed information regarding the concentration, variation, and distribution of lotic sediment which will help future investigators identify normal and excessive sediment conditions in other Ozark highland streams with similar landuse types. Furthermore, the additional resolution of sediment dynamics and linkages with aquatic biota gained by this study will aid in the development of water-quality standards for sediment in Missouri streams.

CHAPTER ONE

INTRODUCTION

Sediment has been documented as a primary cause of impairment of streams (USEPA 2000) and is suspected in the decline of sensitive-aquatic organisms in the Osage River basin of Missouri. Increased sediment loads in streams resulting from road construction are well documented (Waters 1995), but little is known about the amount of sediment that enters streams in the Midwest U.S.A. or its effects on aquatic life (Berkman and Rabeni 1987).

Federal, state, and county highway construction makes significant amounts of soil vulnerable to erosion and transport into streams. Forman (2000) estimates that 22 % of the contiguous United States is “ecologically altered” in some way by the highway network. Road construction crosses small drainages on the landscape that focus and transmit sediment to larger tributaries. Bridge and culvert construction associated with these crossings may appreciably alter stream habitat and biotic assemblages both upstream and downstream of the perturbation (Forman and Alexander 1998; Wellman et al. 2000).

Anecdotal evidence suggests large quantities of sediment eroded from the landscape have entered Missouri waters during road and bridge construction (Missouri Department of Conservation, personal communication). Impacts resulting from road construction on Missouri streams remain undocumented. Missouri State Highway 13 expansion in southwest Missouri is continuing over the next 3 years. The highway is being expanded from a two to four-lane road between Clinton, MO, and Springfield, MO, with the final 9.2 miles of construction near Humansville, MO, which began in October

2004. The remaining-construction segment (and the focus of this thesis) crosses directly over Brush Creek, Panther Creek (St. Clair and Polk counties) and many other small tributaries upstream of designated-critical habitat for the Niangua darter *Etheostoma nianguae*.

In order to understand how road construction and sediment inputs affect aquatic biota, we need good estimates of the volume of sediment produced by road construction projects. Brush Creek contains eight miles of critical habitat for the state endangered and federally threatened Niangua darter. The Niangua darter is listed as threatened throughout its entire range within the Osage River basin and the last recorded observation of this species in Brush Creek was in 1997 (MDC, personal communication). Monitoring sediment inputs from the construction project and effects on water quality and physical habitat will provide information valuable to protecting Niangua darter and other aquatic life.

OBJECTIVES

1. To monitor the quantity, timing, duration, and movement of sediment in the Brush Creek watershed as it relates to road construction development before and during the activity, and, additionally, as it relates to agricultural landuse.
2. To evaluate corresponding changes in water quality and substrate composition.
3. To relate changes in water quality and physical habitat attributable to sediment inputs to changes in benthic fish and invertebrate assemblages.
4. To examine any longitudinal shift in species composition along the stream course and specifically monitor Niangua darter abundance.

LITERATURE REVIEW

Effects of Excess Sediment on Aquatic Biota

Suspended Sediment.—In Missouri, 65 % of impaired-stream miles are attributed to suspended solids (USEPA 2005). Although transporting sediment is a primary function of a stream, it becomes a problem when this sediment load increases to the point where it is greater than the stream's historic-sediment regime. Doisy and Rabeni (2004) define excessive sediment as,

> the concentration of particles < 2 mm in size entrained in the water column of a stream for a period that deviates from the normal concentrations and durations for that stream type to the extent that it has a detrimental effect on native-aquatic life.

Excessive sediments introduced into streams in the suspended state often result in physiological impacts on the biota (Redding et al. 1987) and the degradation or elimination of important habitats as it deposits (Zweig and Rabeni 2001). The effects of such sediment on aquatic biota are well documented in the literature; however most research has examined coldwater fisheries with a focus on Salmonids (Waters 1995). In its suspended state, fine sediment impacts fish and invertebrate assemblages by altering water quality, impacting species distributions, reducing feeding and growth, impairing respiration, decreasing reproductive success, reducing tolerance to disease and toxicants, and causing physiological stresses (Bowlby et al. 1987; Newcombe and MacDonald 1991; Newcombe and Jensen 1996; Wood and Armitage 1997;). An adverse-synecological outcome of altered-primary productivity due to turbidity and subsequent deterioration of water quality has also been suggested as a plausible effect (Cordone and Kelley 1961; Bruton 1985; Wood and Armitage 1997).

Newcombe and MacDonald (1991) categorized the effects of suspended sediments on fishes as behavioral, sublethal, and lethal effects. Increases in suspended sediment may result in behavioral effects such as an alarm reaction, and an avoidance response that causes dispersal and desertion of cover (Newcombe and Jensen 1996). Sublethal effects include a reduction in feeding rate or success, and physiological stresses such as increased respiration, coughing, and changes in blood chemistry. The synthesis by Cordone and Kelley (1961) suggest the sublethal effects of impaired circulation, respiration, excretion, and osmotic regulation can be induced by the clogging of gill filaments by a layer of silt. In addition, the presence of excessive suspended sediment may cause further gill damage through abrasion and matting, therefore resulting in increased susceptibility to disease as described by Ellis (1936). Finally, lethal effects ultimately result in the loss of particular species from a system due to either direct mortality (as a result of extremely high turbidity) or long-term stresses like those associated with sublethal effects (Doisy and Rabeni 2004).

*Deposited Sediment.—*Once introduced into a stream, deposited sediment can alter or eliminate aquatic habitat for several organisms (Cline et al. 1982; Culp et al. 1986). Zweig and Rabeni (2001) demonstrated the importance of physical habitat (i.e., substrate composition) to the biological integrity of four Missouri Ozark streams by documenting a significant decline in benthic invertebrate densities (and other biomonitoring metrics) with increasing deposited sediments in each study stream.

The role of hyporheic exchange, i.e., the flux of water into and out of the alluvium surrounding the stream channel, in sustaining benthic fauna is becoming better understood in lotic systems (Rehg et al. 2004). Clogging of interstitial spaces in the streambed due to deposition of suspended sediment can greatly reduce hyporheic exchange and, subsequently, have deleterious effects on the benthic community (Wood and Armitage 1997; Rehg et al. 2004). The smothering of suitable spawning habitat and the mortality of developing eggs are ultimate consequences of deposited sediment that clogs interstitial spaces and reduces hyporheic exchange (Alabaster and Lloyd 1982).

Deterioration of instream-habitat quality, limited distribution, small population size, and dam construction are the primary reasons for the federally threatened status of the Niangua darter *Etheostoma nianguae* endemic to the Osage River basin in south-central, Missouri (Pflieger 1997). Mattingly and Galat (2002) found the Niangua darter less frequently occupied substrate with > 2 mm of silt coverage. One conclusion was that Niangua darters used microhabitats (i.e., areas of hyporheic exchange) that increase the probability of encountering preferred prey types such as mayfly and stonefly larvae.

I intend to contribute additional information from this thesis regarding sediment-biota linkages that will help elucidate (1) the natural-background sediment conditions that exist in an Ozark highland stream and (2) the biotic response to conditions that may deviate from the natural-sediment regime. Such information is sorely lacking in Missouri and is required to develop effective-sediment standards.

Benthic Fauna as Descriptors of Biological Integrity

The term biological integrity refers to an ecosystem that has remained unchanged both structurally and functionally by man and remains in its pristine condition (Hocutt 1981). The federal Clean Water Act Amendments of 1972 shifted the focus to the welfare of aquatic biota in assessing the condition of a water body rather than being solely concerned about the pollution inputs to a system (Rabeni et al. 1997). The increase in anthropogenic perturbations of lotic systems over the last few decades has necessitated biological monitoring to detect potential degradation.

The use of benthic macroinvertebrates as water quality monitors to detect anthropogenic-watershed disturbances in aquatic ecosystems appears more frequently in the literature (Barton 1977; Lenat et al. 1981; Cline et al. 1982; Klemm 2002; Carline 2003), than the use of benthic fishes. This is due to several traits associated with macroinvertebrates that directly contribute to their sensitivity to ecosystem change. Invertebrates are vital to the metabolic activity of lotic systems and the trophic transfer of energy (Vannote et al. 1980; Cummins and Merritt 1996). A relatively sedentary nature, long life history, ease of qualitative sampling, and overall-ecosystem importance are the primary characteristics that make sampling macroinvertebrates very appealing in water-quality studies (Hellawell 1977; Rabeni et al. 1997).

The trophic link between benthic fishes and macroinvertebrates along with the abundant life-history information available for many fishes makes the use of benthic-fish assemblages advantageous in monitoring the integrity of streams (Berkman et al. 1986). Additional support for the use of fishes as biological monitors includes the ability to identify and enumerate specimens in the field, the ability to examine both acute toxicity and stress effects, the presence of fishes in most waters, and the general public conception that fishes are important (Karr 1981). Moreover, Karr (1981) argues that by monitoring fishes on a regular basis, an assessment of water-resource quality can be obtained if limited resources (i.e., time and funding) do not allow for more comprehensive evaluations of other taxonomic groups.

Berkman et al. (1986) evaluated the sensitivities of fishes to perturbations by comparing the use of fishes and invertebrates as biomonitors of lotic systems in agricultural regions of northeast Missouri. Three rural streams were studied that differed in both habitat type and quality but had similar morphologies. They examined the relation between the values obtained using a modified-index of biotic integrity (IBI) and the habitat quality, which was measured by a habitat-quality index (HQI) that quantitatively described environmental quality at each study site. The invertebrate assemblages sampled formed three distinct groupings, according to a detrended correspondence analysis (DCA), showing some relation to HQI scores. No such distinctive groupings were exhibited by the fish assemblages and no relation to habitat quality was observed. Furthermore, there was a lack of significant correlation between the Shannon diversity index and the HQI for fishes while the opposite was true for invertebrates. Other results from this study showed a reduced abundance of benthic-riffle fishes as siltation increased due to erosion from adjacent-agricultural land. Furthermore, the feeding guilds most inversely correlated with siltation were those associated with the benthos. They concluded fishes were sensitive to anthropogenic disturbance when ecological rather than structural measures, such as IBI, were included in the study.

Rabeni and Smale (1995) provided further evidence of the sensitivities of benthic feeding and reproductive-fish guilds to excessive sedimentation in northeastern Missouri streams. They found herbivores, benthic insectivores, and simple-lithophilous spawners to be the most sensitive guilds to siltation, which was primarily attributed to the life histories of species included within these guilds. Such life-history traits rely upon the composition and quality of the substratum.

The monitoring of benthic macroinvertebrate and fish assemblages in this thesis provides a holistic approach in determining the effects of anthropogenic disturbance on the biological integrity of Brush Creek. The literature provides broad support for focusing specifically on the response of benthic biota to habitat alteration especially in relation to sedimentation.

Anthropogenic-Sedimentation Effects in Streams

*Highway-Construction Effects.—*A meta-analysis conducted by Doisy and Rabeni (2004) compiled 124 datasets pertaining to the effects of suspended sediments on native Missouri fishes. They concluded that in 5th order streams, median concentrations of suspended sediment were highest in the Central Plains, followed by the Ozark Border, Osage Plains, and Ozark Highlands. These median sediment conditions were not determined to be harmful to most adult-fish species throughout the state; however, juvenile largemouth bass, smallmouth bass, and bluegill were susceptible to behavioral and sublethal effects at these normal concentrations found statewide. Data presented from two studies that examined the effects of road construction showed behavioral, sublethal and moderate mortality were predicted to occur in warmwater-native Missouri fishes if the disturbance lasted one year or more. Lethal responses of juvenile-warmwater fishes were exhibited for concentrations of 100 mg/L in only 24 hours.

Two-highway construction studies conducted in vastly different geographic regions found similar linkages between lotic sediment and biota. Lenat et al. (1981) examined the effects on biota of a highway-expansion project that intersected two streams in North Carolina and found convincing evidence of an overall reduction in benthic invertebrate density, taxa richness and diversity, and a concomitant increase in suspended solids at study sites below the road construction area. Barton (1977), with a very similar study design to the one conducted here, reported a substantial increase in suspended and deposited sediment concentrations during road construction activities (in Ontario, Canada) as well as associated reductions in fish abundance and shifts in species composition of riffle macroinvertebrates.

A recent study by Carline et al. (2003) monitored anthropogenic effects during and immediately after a road and bridge construction project that intersected Spring Creek, a coldwater stream in Pennsylvania. The sediment load, stream-substrate composition, benthic-macroinvertebrate community, and distribution and density of trout spawning redds upstream and downstream of the construction activity were monitored. They reported an average of 182 metric tons/year of introduced sediment associated with the road construction. Rock check-dams were installed to dampen sediment movement through concrete culverts (i.e., artificial drainage channels) that led directly into Spring Creek. They found significant increases in total suspended solids (TSS) in 55% of storms after the drainage channels were installed as compared to significant increases in TSS of only 23% of storms before the installation of the channels. None of the other habitat or biotic variables monitored in the study were markedly affected by the dramatic increase in sediment loads during the road construction. Perhaps significant effects may have been detected in the biotic community if sampling had been initiated further in advance of the road construction activities to allow reference conditions to be measured.

*Agricultural Effects.—*In addition to road construction, the presence of agricultural grazing is another-significant anthropogenic activity that serves as a potential origin of excessive sediment and is a widespread activity throughout the Brush Creek watershed in southwest Missouri. With a regional economy based primarily on agriculture (e.g., 59% of landowners in the basin are agricultural landowners), the land use within the Brush Creek watershed is 50% agricultural with 46% as pasture and 4% as cropland (Groshens 1997). Much of the riparian corridor throughout the basin is open to grazing by cattle, and timber harvesting (i.e., selective cutting) is very active in riparian areas throughout the basin (Groshens 1997; ZLF personal observation).

Bergthold (2001) examined the macroinvertebrate assemblage in three headwater streams in the prairie ecoregion of Missouri which has a significant-agricultural influence. The objective of this study was to determine if there was an influence of adjacent landuse, which consisted of forest, pasture, and row crop, on the diversity and composition of the invertebrate assemblage. The study found that the pasture sites (where cattle had unrestricted access to the stream) consistently had homogenous substrates comprised of sand with some gravel, the least canopy cover (i.e., open-local riparian zone), and increased amounts of algae compared to sites with other land uses. Although differences were found in taxonomic composition associated with land use, no significant effects of land use were detected on the community metrics examined in the study. It was concluded that larger-scale (i.e., watershed level) factors may have greater influence in structuring the invertebrate assemblage rather than the local level.

Several studies have examined the correlation between land use, habitat, and biological integrity of streams in agricultural drainages at various spatial scales. In a study designed to address the influence of riparian condition at both the local and upstream (i.e., upstream of the sampling reach) riparian zones on the fish assemblage, Lee et al. (2001) compared 18 streams in the agricultural Minnesota River basin. They found species richness and IBI scores were significantly greater at wooded-riparian sites at both the local and upstream scales compared to sites that were open at both scales. Not only was wooded-riparian cover a factor in structuring the fish assemblage, but streams with local-wooded riparian zones also had greater amounts of instream habitat and diversity of mesohabitats than streams with open-riparian zones.

Stewart et al. (2001) examined correlations between watershed, riparian-corridor land cover, and reach-scale habitat versus fish and macroinvertebrate assemblages in 38 warmwater streams in Wisconsin whose watersheds were greatly influenced by agricultural land use. Bottom-land and near stream-cattle grazing was a common practice. Their results showed near stream grasslands and agriculture had a negative influence on fish diversity and density. Similarly, the macroinvertebrate assemblage exhibited declines in EPT taxa and increased Hilsenhoff Biotic index scores which both indicate poorer water quality associated with percent grassland.

Clearly, agricultural activities such as cattle grazing can alter the aquatic biota of the adjacent stream. The presence of this activity, in addition to road construction, in the Brush Creek watershed provides a unique opportunity to examine the sediment dynamics, habitat quality, and biological integrity of a stream that has anthropogenic similarities to numerous Ozark Highland streams.

INFORMATION NEED

A lack of research exists in the literature pertaining to sediment inputs in Midwestern aquatic ecosystems due to landscape-altering activities such as road and bridge construction. Failure of proper development and effective implementation of BMP's associated with road construction activities (e.g., installation of silt fences) is a tautological concern commonly implied and suggested in the literature regardless of geographic location (see Dallaire 1976; Burton et at. 1977; Barton 1977; Lenat et al. 1981). Moreover, Rabeni and Smale (1995) suggest sound riparian management, in conjunction with other watershed efforts to reduce erosion, is vital to enhance the ecological health of stream ecosystems. Such pandemic effects on the biota of lotic systems widely suggested in the literature further substantiate the holistic-ecological approach needed to fully understand the affinity between sediment and biota (Bruton 1985; Wood and Armitage 1997).

BENEFITS OF RESEARCH

The advanced notice of the construction and the potential impacts and the presence of a threatened species in the watershed make this a unique opportunity to study sediment/biota linkages in Missouri. A finer resolution of the impact of sediment on aquatic life, especially in Missouri Ozark streams, will be enhanced by a better understanding of instream sediment dynamics associated with watershed-level perturbations. Results will quantify the magnitude, frequency, and duration of sediments both before and after road construction activities.

Meeting project objectives will notably aid preservation efforts for the threatened Niangua darter and other silt-sensitive species in the ecoregion. This information will ultimately be used to begin development of water-quality standards for sediment in Missouri in an effort to prevent or reduce future impacts of construction produced sediment on lotic systems. The knowledge of natural-background sediment conditions, critically lacking in Missouri, will hopefully aid Missouri in establishing turbidity standards like those of other states and help protect the biological integrity of Ozark streams during anthropogenic-press disturbances (Bruton 1985).

CHAPTER TWO

APPROACH AND METHODOLOGY

Spatial Arrangement of Study Sites

The Brush Creek watershed is situated in the southwest Missouri counties of Cedar, Hickory, Polk, and St. Clair, where Brush Creek flows approximately 35-km (22 miles) northwest to its confluence with the Sac River (Figure 1). The watershed drains 223 km^2 (86 mi^2) of land and is located in the Ozark Highlands Section, Osage River Hills Subsection, and Lower Sac River Oak Woodland Hills landtype association (LTA) according to the classification by Nigh and Schroeder (2002). The geology of the landtype association consists of Pennsylvanian sandstones in the uplands with Ordovician chert and shale dolomites in the valleys (Nigh and Schroeder 2002). Loamy to clayey soils exist adjacent to streams in this LTA. Mean annual precipitation in the region is 42-43 inches.

Study sites were chosen in a manner that focused on the mainstem of Brush Creek and its principal, fourth order tributary, Panther Creek (Figure 1). Two locations (sites 8 and 9) were established as control sites located directly upstream of Highway 13 with one site stationed in Brush Creek (site 8) and the other in Panther Creek (site 9). Downstream of Highway 13, sites 5, 6, and 7 were positioned in the mainstem of Brush Creek but upstream of the confluence with Panther Creek. A site located in the mouth of a small tributary at site 6 was added to monitor suspended sediment. Additionally, site 4 was located directly at the confluence of Panther Creek and Brush Creek which was a site of intense cattle disturbance.

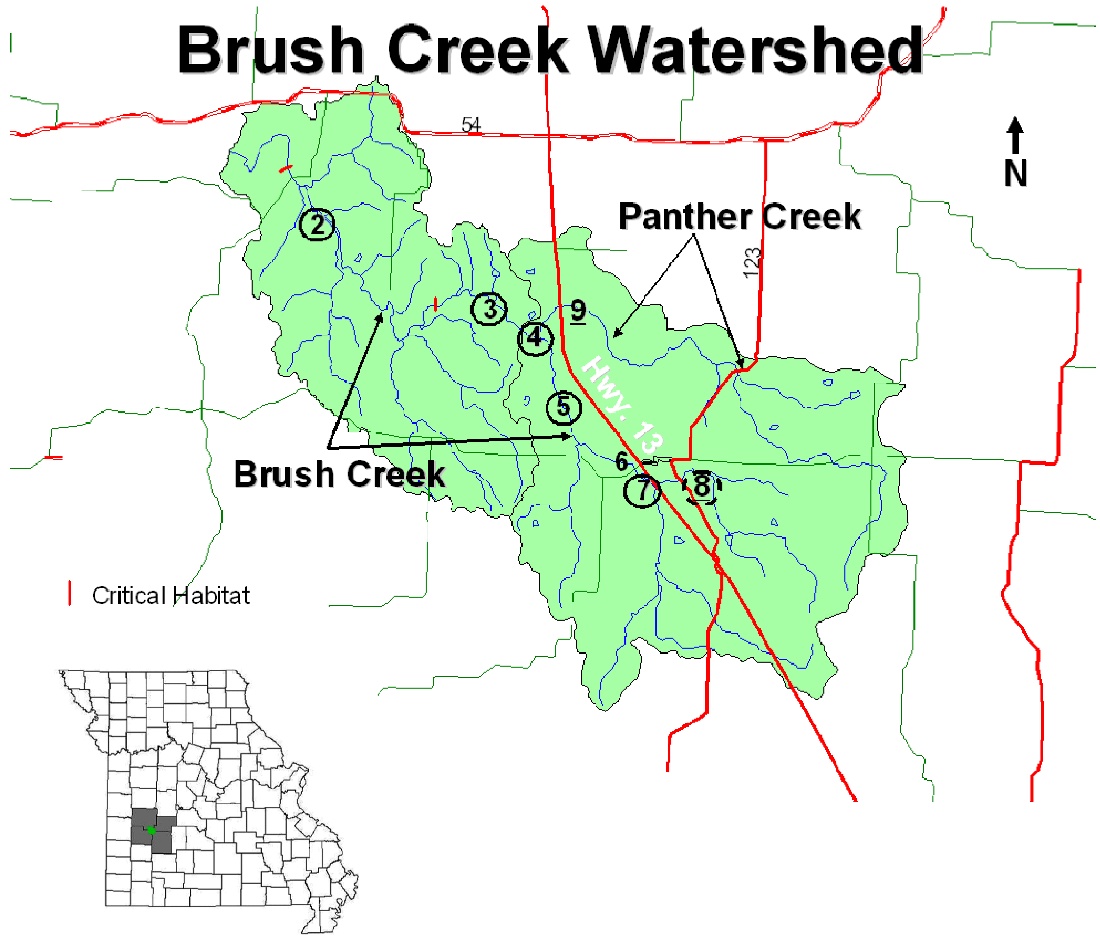


Figure 1.—Brush Creek watershed with location of study sites. Circled sites represent those where biotic sampling was conducted in addition to sediment sampling. Brush Creek flows in a NW direction. Sites 8 and 9 are upstream of the road construction. The boundaries of the critical habitat for the Niangua darter are designated with two-red dashes.

Sites 2 and 3 were downstream of this confluence with site 2 located in the eight-mile federally designated critical habitat for the Niangua darter. Site 2 is also situated within the Birdsong Conservation Area managed by the MDC.

A total of eight study sites (excluding the tributary at site 6) were chosen to adequately characterize the sediment dynamics within Brush and Panther Creek with the two most upstream locations representing the un-impacted sites, respective to the Highway 13 road-construction activity (see Appendix 6). Site 4 was directly associated with a sizeable area of excessive riparian and instream cattle impact. The location of all study sites remained consistent throughout the timeline of this study. Additionally, most sites (excluding site 3 and the tributary at site 6) are the same as those being sampled in conjunction with an ongoing, ecological study in the watershed that was initiated in 1996. This long-term study is being conducted by the Missouri Cooperative Fish and Wildlife Research Unit and provides a valuable data set regarding the macroinvertebrate assemblage in Brush Creek. This database supplements the pre-construction or baseline data for this study. Table 1 provides further details for each study site.

Table 1.—Location of study sites in the Brush Creek watershed.

Stream	Site	County	Location	GPS Coordinates
Brush Creek	2	St. Clair	T36N R25W Sec.20	N37-52-04.6 W93-42-30.9
Brush Creek	3	St. Clair	T36N R25W N Sec.35	N37-50-63.2 W93-39-53.4
Brush Creek	4	St. Clair	T36N R25W E Sec.36	N37-49-36.0 W93-37-16.5
Brush Creek	5	Polk	T35N R24W Sec.6	N37-49-12.9 W93-37-17.2
Brush Creek	6	Polk	T35N R24W W Sec.16	N37-47-35.7 W93-35-27.4
Brush Creek	7	Polk	T35N R24W S Sec.16	N37-47-16.6 W93-35-02.4
Brush Creek	8	Polk	T35N R24W W Sec.15	N37-47-22.5 W93-34-22.3
Panther Creek	9	St. Clair	T36N R24W N Sec.31	N37-50-29.1 W93-37-1.77

Suspended Sediment Collection Methods

<u>Sampler Design And Function.—</u>Single-stage, US U-59B, suspended sediment bottle samplers were constructed in-house and installed at all study sites to collect water samples during the ascending limb of the storm hydrograph. The single-stage sampler was originally designed and thoroughly tested by the Federal Inter-Agency Sedimentation Project (Subcommittee on Sedimentation 1961). The sampler functions by siphoning water into a plastic-one pint sample bottle once the water-surface elevation reaches the crown of the intake (i.e., shorter) tube (Figure 2). The bottle continues to fill with water until the sample reaches the outlet of the exhaust tube below the rubber stopper inserted in the bottle. When the stream rises to the level of the exhaust nozzle, air is trapped in the exhaust (i.e., longer) tube. This trapped air effectively prevents water circulation through the bottle even after the water stage rises above the exhaust tube inlet. As long as sufficient air remains in the tubes, no additional water can enter the bottle to alter the sample unless excessive velocities persist (Subcommittee on Sedimentation 1961). A detailed list of materials required to construct the sampler can be found in Appendix 1.

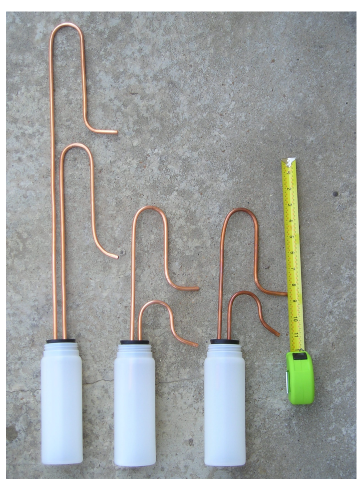


Figure 2.—Photograph of single-stage bottle samplers. Two different designs are shown. The design with the short-copper tubes (on the right) was used in this study based on potential water velocities. The measuring tape provides a reference of 12 inches.

*Sampling Protocol.—*Bottles were attached to a 2.4 m (8 ft.) piece of 1.27 cm (0.5 in) rebar that was anchored in the stream adjacent to a vertical bank where a tree (or tree roots) served as an anchor point. Sampler bottles were mounted (i.e., stacked) vertically in a manner that allowed for collection of discrete water samples every 20 cm (8 in.) during the rising hydrograph. This provided a range of samples that included the entire scope of baseflow to bankfull stages. One complete set of single-stage samplers consisted of 8-10 bottles per site, and the position of the bottle set within a site remained unchanged throughout the study (Figure 3). Additionally, at two-less-accessible study sites, I used SIGMA 900MAX automated, portable water samplers that acquired 24-water samples throughout the entire storm hydrograph at selected-time intervals (e.g., one sample every 2-4 hours), which helped evaluate the effectiveness of the single-stage bottle samplers. Bottles from both-sampling methods were collected for analysis and replaced after each-significant rain event that occurred in the watershed.

*Sample Processing.—*The Standard Methods (APHA 1995) protocol for filtration and total suspended solids (TSS) analysis was used to process aliquots of suspended-sediment samples and resulted in the determination of mg/L of suspended sediment per sample. The organic and inorganic fractions of these sediments were determined by measuring the oven-dry weight and ash-free dry weight. This helped elucidate the presence of anthropogenic sediments (Barton 1977) and determine any seasonal fluctuations in organic matter. Aliquots were also analyzed for turbidity, measured in nephelometric turbidity units (NTU), using a Hach turbidity meter. Total processing time was ~11 minutes per bottle (sample) with eight minutes required for the filtration procedure and three minutes to analyze turbidity.

Figure 3.—Photograph that shows the single-stage bottle sampler arrangement at one study site during the rising limb of the hydrograph. Some bottles were painted black.

Stage-Discharge Methods

Water-level Loggers.— Given that no gauging stations are installed in this basin, I collected stage information throughout the study by using pressure-transducer type water-level loggers that were installed at sites 2, 3, 6, 7, and 8 (Figure 4). These loggers record depth and time at specified-time intervals. Additional information about the logger is provided in Appendix 1. These were installed throughout the basin to encompass headwater and downstream-depth variability. The logger was positioned so that the water-level sensor was in place at the thalweg of the channel (if one existed).

Stage-Discharge Relation.—Stage height and velocity measurements were taken along a transect at the five sites where level loggers were installed. This was done for a few-initial rain events of varying magnitude to estimate discharge according to the equation,

$$Q = \sum(vA),$$

where Q = volumetric flow rate, v = fluid velocity, and A = individual, partial cross-section area of flow (Buchanan and Somers 1969). Velocity was measured using a Marsh-McBirney (electromagnetic) flow meter at six-tenths depth (0.6*D*) where the mean velocity within each vertical is approximately found (Gordon et al. 2004). The midsection method of computing cross-section area described by Buchanan and Somers (1969) was used in the estimation of discharge at each location. The number of subsections across the section perpendicular to the flow depended on the variability of depth and velocities within the channel. The resulting stage-discharge relations were useful in estimating discharge based on stream depth at sites located throughout the basin.

Figure 4.—Photographs of the procedure for measuring stream discharge along a transect at site 6 (top) and the sampling and measuring equipment at site 7 (bottom).

Deposited Sediment Collection and Estimation Methods

<u>*Sampler Design and Collection.—*</u>The pit-type sediment sampler that was used is a modified version of the one used in a similar-sediment-monitoring study (Hedrick et al. 2005) and parallels the method of using a Whitlock-Vibert box sampler design (Wesche 1989). The pit-type sampler that I constructed was 5-cm (2 in.) deep and 10.2-cm (4 in.) in diameter and was constructed of schedule-40 PVC pipe (Figure 5). Contained within each sampler was a collection of homogenous-artificial gravels >12.7-mm (0.5 in.) that represented the unconsolidated-stream substrate and allowed for collection of particles smaller than medium gravel (Gordon et al. 2004). The use of artificial gravel served to reduce variability in the gravel mixtures contained within each sampler and allow for homogeneity among sites. Lisle (1989) showed no difference in accumulation of deposited sediment existed between artificial and natural gravels. To prevent these gravels from being removed from the sampler by scour action, a layer of galvanized-hardware cloth, with mesh size 12.7-mm (0.5 in.), was stapled to the top of each sampler. The hardware cloth was a modification to the original sampler design used by Hedrick et al. (2005). A removable cap inserted on the bottom side held the gravels in the sampler. Each sampler was placed and anchored flush with the streambed in an excavated depression. Additionally, a two foot piece of steel rebar was driven into the stream adjacent to each sampler and plastic-cable ties were used to attach each sampler to the rebar stake (Figure 5). Flagging tape tied to the rebar stake marked the location of each sampler for easier retrieval. A detailed list of materials required to construct the sampler can be found in Appendix 1.

Upon retrieval of deposited sediment samples, the sampler was slowly removed from the streambed by cutting the cable ties, and the sampler was inverted directly into a plastic-tupperware container (Figure 5). The sediment was then washed from the artificial gravels with a squirt bottle into the transportable container which was then appropriately labeled. Additionally, I measured the displacement volume (i.e., interstitial space) of each individual sampler with ambient-stream water after the sediment was removed from the gravels (Appendix 4). This simply involved filling the sampler completely with water and transferring the water to a graduated cylinder for a volume measurement. Finally, the insert cap was replaced, and the sampler was inserted back into the same position in the streambed and re-attached to the rebar stake.

Figure 5.—Photographs that show the installation of a deposited-sediment sampler in the streambed (top and middle) and the collection of a sampler with the bottom-insert cap removed (bottom).

*Sampling Protocol.—*Deposited-sediment samples were collected and substrate-composition estimates were made at six-study sites during the months of March, May, July, September, and December 2004, and during the months of March and May 2005 for a total of seven-sampling periods (excluding September 2003 when pilot samples were collected). The two and one-half hour travel distance to the stream did not allow the ability to collect this data after each storm event that occurred in the watershed. This periodic sampling in conjunction with the biota monitoring provided information on the spatial and temporal dimensions of the relation of deposited to suspended sediment and the movement of sediment through the system. This intermittent-sampling regime also allowed adequate time for deposition of sediments to occur during and after any storm events within this defined period.

Study sites 2, 3, 4, 5, 7, and 8 contained enough non-bedrock riffle, run, and pool habitat to collect deposited sediment samples using the pit-type sediment traps. Two to four of these constructed samplers were placed in each mesohabitat and spaced at least one meter apart from each other for a total of six to twelve samplers per site. I performed a power analysis with data from early-sample collections to determine the number of samplers per habitat that provided high statistical power at an alpha level of 0.05 (Appendix 2).

Additionally, all samplers were placed in the same approximate position within each mesohabitat; for example, just above the downstream end of a pool or the riffle/run thalweg (if one existed). Thomas (1985) suggested these mesohabitat positions should be selected to monitor sediment because they tend to exhibit homogenous sediment and flow profiles that allow for more-accurate data to be collected. Samplers were also positioned in this manner to reduce hydrologic variability associated with dewatering of riffle/run sequences during drought conditions that were common throughout the study.

*Sample Processing.—*The deposited-sediment samples were dried in the laboratory for 24 hours at 105°C and burned for one hour at 500°C to determine an oven-dry weight and ash-free-dry weight for each sample, respectively. Therefore, a weight-per-sampler (g/sampler) of all sediments and the inorganic fraction was obtained. Additionally, particle size distribution was evaluated for each deposited-sediment sample by using a series of nested sieves. The size classes examined (from coarse to fine) were fine gravel (≥ 4.0 mm), very fine gravel (2.0-4.0 mm), coarse sand (0.5-1 mm), fine sand (0.125-0.25 mm), very fine sand (0.0625-0.125 mm), and coarse silt (<0.0625-mm) (Gordon et al. 2004). Processing time was approximately 15 minutes per sample in addition to the hours spent in the drying oven and muffle furnace.

*Visual Sediment Estimations.—*In addition to the collection of deposited sediment samples, deposited sediment estimations were made using two methods that have shown their value in Missouri streams: percent-surface cover and embeddedness (Zweig and Rabeni 2001). Percent-surface cover of deposited sediment (in 5% increments) and substrate embeddedness were visually estimated by a single observer (ZLF) within a 0.1-m^2 -circular quadrat in riffle, run, and pool habitats at each site. Embeddedness was defined as the percent in which gravel, cobble, and boulder particles were surrounded by fine sediment or sand, and the following four percentage classes were used: 0-25% (rating 1), 25-50% (rating 2), 50-75% (rating 3), and >75% (rating 4) (from Platts et al. 1983). A minimum of eight estimates for each parameter (per habitat type) was randomly made in a systematic fashion at each site. These estimations were made at the time of the deposited-sediment collections and provided further resolution of the sediment dynamics within the basin.

Data Analysis

Suspended Sediment.—Concentration-duration curves that accompany the hydrograph were developed from the suspended-sediment data to evaluate sediment concentrations (TSS) and durations associated with discharge These hydrograph-sedigraph curves were only constructed for rain events that resulted in a significant rise in water level (i.e., > 40 cm) at sites where level loggers and automated samplers were installed. Statistical analysis involved the use of the Kruskal-Wallis rank test to compare mean-suspended sediment concentrations at un-impacted versus impacted study sites for each rain event throughout the study period, and to examine seasonal differences in mean TSS across study sites. This nonparametric test uses a chi-square distribution approximation (Neter et al. 1996). An alpha level of 0.05 was used in all significance tests. All analysis in this thesis was done using SAS v.8 software (SAS 1999).

In addition, simple-linear regression was used to examine the relation between TSS and NTU. All data was Log_{10} transformed to meet assumptions. Residual plots were used to detect outlying *X* observations, and a bonferroni test procedure was used to detect outlying *Y* observations (Neter et al. 1996). Secondly, the *DFFITS* and *DFBETAS* measures were used to ascertain if outliers actually influenced the regression coefficients. Outliers were deemed influential and removed from the data set if the absolute value of these two measures was greater than one (Neter et al. 1996).

_Deposited Sediment and Substrate Composition.—_Two-different ANOVA methods were used to analyze deposited sediment data. First, a two-way mixed-model ANOVA (Model III) was used to analyze spatial and temporal differences of deposited sediment dry weight in each habitat across time. The fixed effects in the model were habitat, sampling period (i.e., time), and the interaction term. The random effects were site, site*habitat, and site nested within habitat and time. Sample date (i.e., time) was a repeated effect also incorporated in the model. Analysis of deposited sediment particle sizes and percent surface cover of fine sediments were examined using the same-two-way Model III ANOVA methods. Secondly, a one-way ANOVA (Model I) was used to test for habitat differences of deposited sediment at each site for every sampling period. For all ANOVA methods in this thesis, the Kolmogorov-Smirnov test was used to determine normality, and Bartlett's test was used to examine the homogeneity of variances. All raw data and proportion data was square-root transformed, with the exception of surface cover data which was rank transformed, to meet the model assumptions.

Since the embeddedness data were categorical in nature, the GENMOD procedure was used, which is an ordinal-multinomial model in SAS (SAS 1999). Habitat and sample date were the fixed effects in this model.

Correlations among all environmental variables were examined using Spearman's rank correlation coefficients. Significant correlations were determined at an alpha of 0.05, while moderately significant correlations were noted with a less conservative alpha of 0.25. A detailed description of each-environmental variable used in the correlation analysis throughout this study is provided in Appendix 5.

Benthic Macroinvertebrate Sampling Methods

Sampling Protocol.—I used the Rapid Bioassessment (RBA) sampling protocol that was used to develop biological criteria for streams of Missouri (Rabeni et al. 1997). This sampling protocol has been used in Brush Creek for eight years by Gregory Wallace (MO Fish and Wildlife Coop Unit) in conjunction with a long-term ecological study that is ongoing in the watershed; therefore, I was able to combine my pre-construction data to this earlier data set in order to better determine baseline conditions.

This sampling protocol was carried out at the six-study sites where deposited-sediment samples were collected (i.e., sites 2, 3, 4, 5, 7 and 8). The protocol involves the use of a bottom-aquatic kick-net with an 18 x 8" frame and 800 x 900 mμ mesh net. Two characteristic habitat types were sampled at each site. These consisted of coarse substrates in flowing water (i.e., riffles and runs) and habitats in non-flowing water defined as depositional areas; hereafter referred to as coarse-flow and non-flow habitats, respectively. An area of 1-m^2 directly upstream of the net was kicked to displace the substrate to a depth of approximately 15-25 cm. The organisms were then dislodged and carried into the kick-net by the water current or sweeping motion of the net. Six-individual net samples (sample replicates) were collected and composited within each habitat type at a variety of depths, current velocities (excluding non-flowing water), and substrate mixtures to account for the variety of conditions at each site. For each habitat type, the contents were combined and transferred to a glass jar for field preservation in 10% formalin. Approximately 10 minutes per replicate were required to collect benthic invertebrates in either habitat.

Sample Processing.—A detailed explanation of the laboratory processing of invertebrate samples can be found in the report by Rabeni et al. (1997). In summary, a 150 organism sub-sample was obtained with a standardized-sub-sampling procedure and identifications were made to the lowest possible taxonomic level. After the sub-sampling procedure was conducted for samples collected during this study period, the remaining portion of every sample was examined under a dissecting scope and all organisms considered to be "large and/or rare" in the sample were identified but not enumerated. Experienced personnel were recruited to help expedite the identification process. All specimens, including chironomids, were identified in each subsample.

Benthic Fish Sampling Methods

Quadrat Sampler Design.—I used a 1-m^2 benthic-quadrat sampler to obtain density estimates of benthic fishes in both riffle and run mesohabitats at the same-six study sites where deposited sediment and macroinvertebrate collections were collected. The quadrat sampler has proven to be a very accurate way to estimate densities of benthic fishes, especially in Ozark streams, and the sampling efficiency and bias of this gear has been extensively examined (Peterson and Rabeni 2001; Rettig 2003). The sampler consists of a 1-m^2 frame that is 0.5m tall and covered with 3-mm (1/8th in) diameter netting on all sides but the top and bottom, which are open. Each corner of the sampler has 0.25-m leg extensions at the bottom, which serve to anchor the sampler in position. The downstream side of the sampler has a 3-mm mesh bag net attached to it, which is the area of fish collection (see Figure 6).

Figure 6.—Photograph of the 1-m^2 quadrat sampler used to sample benthic fishes in riffle and run mesohabitats. Notice the collection bag on the downstream side of the sampler.

*Sampling Protocol.—*To avert some of the known, inherent biases associated with this particular gear, I followed the same methodology used by Rettig (2003), which exhibited approximately 80% sampling efficiency for most *Cottus* and *Etheostoma* species in riffle, run, and pool mesohabitats. Preliminary sampling at a few of my study sites in Brush Creek indicated that approximately 8-m^2 of riffle and 8-m^2 of run habitats (16 total replicates) needed to be sampled in order to obtain density estimates within 30% of the true mean (Figure 7). A total of 16-sample replicates per site were considered the maximum acquirable due to reduced-habitat availability at several-study sites (e.g., intermittent sites) during dry periods.

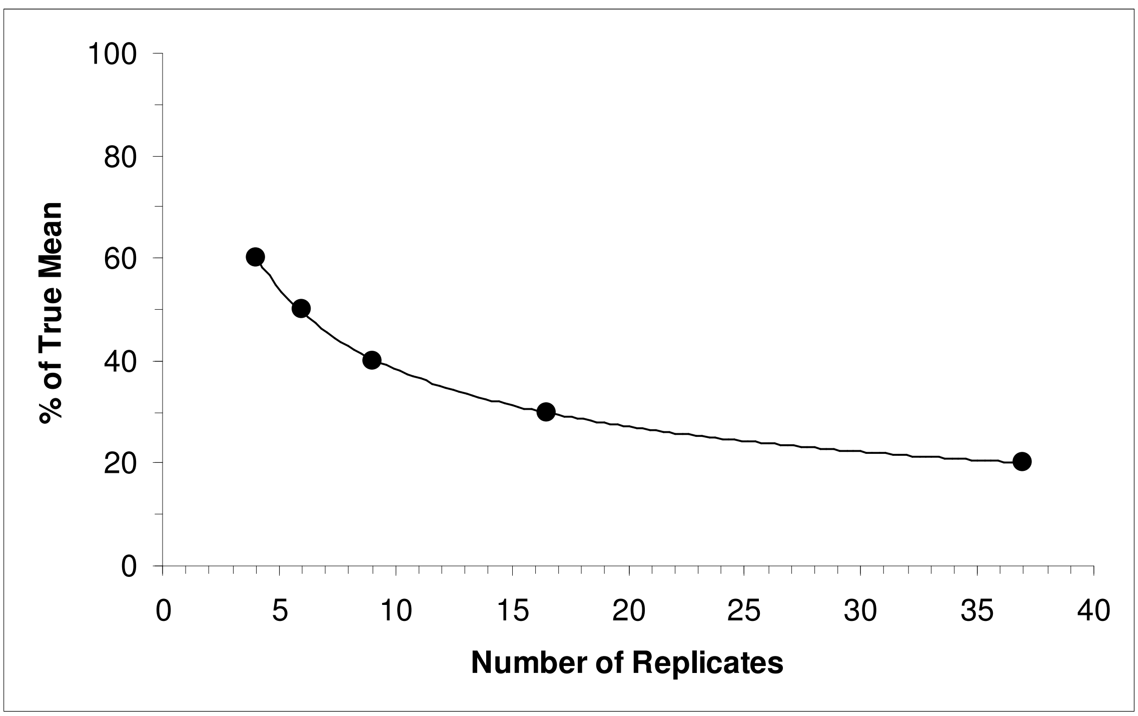


Figure 7.—A display of the estimated sampling precision obtained in relation to effort needed to sample benthic fishes at each site according to preliminary data.

The standardized-sampling procedure I used is as follows. Placement of the sampler began at the farthest downstream riffle or run within each site and a gradual progression upstream was made in a manner that sufficiently represented each habitat type both longitudinally and laterally. Each individual subsample consisted of deliberate, swift placement of the quadrat sampler and subsequent positioning of adjacent rocks along the inside-lower perimeter of the sampler to weight down the bottom edge of the mesh netting. The placement of rocks in this manner effectively reduced fish escapement. Once the quadrat sampler was firmly in place and the lower edge was sealed off along the stream bottom, I or a technician used a backpack electrofisher to capture fish inside the 1-m^2 quadrat. The area was electrofished for a period of one minute after which any large rocks, debris, etc. were removed and shocking resumed for one-additional minute. Immediately following the last minute of shocking, the substratum was intensively disturbed within the sampled area to dislodge any fish that remained. All fish were flushed into the collection bag. Finally, the quadrat sampler was carried away from the stream and rushed to the nearest bank. The fish were removed from the collection bag as quickly as possible and placed in a 5-gallon bucket that contained ambient-stream water. All captured fish were identified and enumerated in the field. The total length of each fish was measured and recorded. Approximately 15 minutes per replicate were needed to collect benthic fishes in this manner.

<u>*Niangua Darter Monitoring.—*</u>Specific monitoring of Niangua darter (*Etheostoma nianguae*) presence-absence was conducted on 17 June 2004. This involved the use of a mask and snorkel to observe benthic fishes at two sites in Brush Creek and one site in Panther Creek. Snorkeling is proven to be a very effective and least invasive technique

for observing the presence of the Niangua darter (Mattingly and Galat 2002) and is currently the method used in a long-term monitoring effort being carried out by MDC research personnel. The information collected at each site included: a drawing of the snorkeled reach, GPS coordinates of the reach boundaries, description of habitat where Niangua darter was observed (although none were found), list of other fish species observed, and additional habitat measurements (e.g., riparian width, distance of eroding bank, canopy cover, etc). This monitoring procedure was only carried out once due to low-water levels in Brush Creek during the regular MDC snorkeling seasons. All information gathered was given to MDC personnel directly involved with Niangua darter monitoring efforts.

Analytical Procedures.—Macroinvertebrate data analysis was done separately for the long-term data collected in the spring of each year (1998-2005) and for the sample dates associated with this study period (May 2004-May 2005). Site 3 was the only site not sampled until May 2004. Total taxa richness, the EPT index (i.e., the sum of all Ephemeroptera, Plecoptera, and Trichoptera taxa), the modified Biotic Index (Hilsenhoff 1987), and Shannon diversity index (Shannon and Weaver 1949) were calculated for both data sets, individually. Taxa richness and the EPT index both increase with improving water and habitat quality. Biotic Index values range from 0-10 and increase as the perturbation increases. The Biotic Index takes into account a tolerance value assigned to each taxa (Sarver and McCord 2001). Shannon diversity increases as the number of taxa increases and as individuals become more evenly distributed among those taxa.

The two-way ANOVA (model III) used to analyze the abiotic data was also used to test for differences in invertebrate-biomonitoring metrics. EPT richness data for this study period was $\log_{10}$ transformed to meet model assumptions; whereas, all other metric data for both study periods remained untransformed.

An ordination of the invertebrate assemblage was made using Detrended Correspondence Analysis (DCA), which is an indirect ordination technique based on reciprocal averaging, and rare species were downweighted (Gauch 1982). PCORD v.4.14 software was used to perform the DCA (McCune and Medford 1999). Axis 1 and 2 site scores were related to the biomonitoring metrics and environmental variables using Spearman rank correlation coefficients. The ordination was also used to examine the relative abundances of a few-dominant taxa that are known to be sediment intolerant (Zweig 2000).

Species richness and Shannon diversity were calculated for the benthic-fish assemblage throughout the study. These metrics were related to the environmental data using Spearman rank correlation coefficients. Additionally, the two-way model III ANOVA was used to test for changes in biomonitoring metrics and densities of adult fishes that composed greater than 10% of the total abundance (i.e., *E. spectabile, E. caeruleum,* and *E. flabellare*. The raw data for *E. spectabile* and *E. flabellare* was log10(x+1) transformed to meet the model assumptions and the autoregressive covariance structure was used. The raw data for *E. caeruleum* remained untransformed and the unstructured covariance structure was used.

Aerial Photograph and Video Monitoring.—To document stream channel and riparian changes as well as erosion control practices, aerial photographs and video were taken (via the MDC helicopter) on 5 October 2004 and 29 April 2005, immediately before and during road construction activities, respectively.

CHAPTER THREE

RESULTS

Suspended Sediment

Upstream versus Downstream Comparisons.—Twenty rain events were successfully sampled at both upstream and downstream study sites that varied in magnitude (Table 2). Eleven of these rain events occurred prior to road construction, while nine rain events occurred after the start of road construction. Hereafter, upstream and downstream sites will directly refer to those sites (relative to the highway construction) pooled together. The mean TSS for individual rain events upstream of the highway construction ranged from 19-460 mg/L before construction and from 58-727 mg/L after construction started (Table 2). In contrast, the mean TSS downstream ranged from 42-919 mg/L before construction and from 123-734 mg/L during construction (Table 2). Only seven-rain events had significantly different-mean TSS upstream vs. downstream of the road construction. One of those rain events had a significantly higher-sediment concentration upstream of the highway compared to downstream. However, most statistically significant rain events had an increased-mean TSS downstream of the highway especially after the start of road construction. There was a general trend for greater mean and max TSS downstream of Highway 13 versus upstream for most rain events (Table 2).

Table 2.—Descriptive statistics for all significant rain events (TSS) at upstream sites (pooled together) and downstream sites (pooled together). Asterisks identify significant differences between upstream and downstream for individual rain events, as indicated by the Kruskal-Wallis test. An alpha of 0.05 was used to test for significance.

Rain Date	Number of samples collected	Upstream TSS (mg/L) Mean	Upstream TSS (mg/L) Max	Number of samples collected	Downstream TSS (mg/L) Mean	Downstream TSS (mg/L) Max	p-value	
8/30/2003	*0*	-	-	6	203	470	-	
11/17/2003	3	280	415	34	638	1626	0.148	
3/3/2004	5	359	660	36	919	11185	0.719	
3/26/2004	5	291	469	37	371	2353	0.923	
4/24/2004	8	81	134	39	176	725	0.022	*
5/1/2004	2	47	72	10	50	78	0.667	
5/13/2004	3	21	29	11	44	72	0.024	*
5/18/2004	*1*	*19*	-	6	61	166	-	
7/1/2004	3	77	129	12	42	115	0.083	
7/9/2004	2	99	151	8	285	768	0.794	
7/23/2004	8	460	580	19	229	613	0.009	**
8/24/2004	7	295	530	26	336	907	0.792	
Start of Road Construction								
10/27/2004	11	727	5340	33	734	8590	0.278	
11/12/2004	7	58	106	24	123	267	0.044	*
11/23/2004	11	178	330	42	237	843	0.259	
1/4/2005	12	253	470	32	495	1002	0.001	*
1/12/2005	9	239	534	37	364	742	0.048	*
2/12/2005	5	92	206	24	270	496	0.009	*
4/6/2005	3	225	242	27	426	1880	0.284	
6/6/2005	8	272	760	28	732	4799	0.138	
6/9/2005	12	280	616	42	512	3607	0.061	

* Downstream TSS is significantly higher than upstream TSS

** Upstream TSS is significantly higher than downstream TSS

A Kruskal-Wallis test (using the untransformed data) showed a significant increase in the overall-mean TSS downstream of the highway after the start of road construction activity (Prob > ChiSq < 0.0001); whereas, no difference existed upstream before versus during construction (Prob > ChiSq = 0.5473) (Figure 8). There was no difference in the overall-mean TSS upstream versus downstream before the start of construction (Prob > ChiSq = 0.1698); however, the overall-mean TSS was 53% greater downstream of the highway during construction compared to upstream (Prob > ChiSq < 0.0001) (Figure 8).

When downstream sites were examined individually for each rain event, no-single site consistently displayed elevated-mean TSS concentrations compared to other sites (Figure 9). It should be noted that samples at every site were not successfully collected for all-rain events. Sites 2 and 6 tended to have the greatest-mean TSS (around 1000 mg/L) for rain events that occurred after the start of road construction (Figure 8). In particular, site 2 (the most downstream site) had a mean TSS concentration approaching 3,000 mg/L for one pre-construction rain event, which notably exceeded all other sites. Additionally, site 6 (immediately downstream of Highway 13) typically had mean-sediment concentrations that exceeded the majority of downstream sites for most rain events before and during highway construction activities (Figure 9).

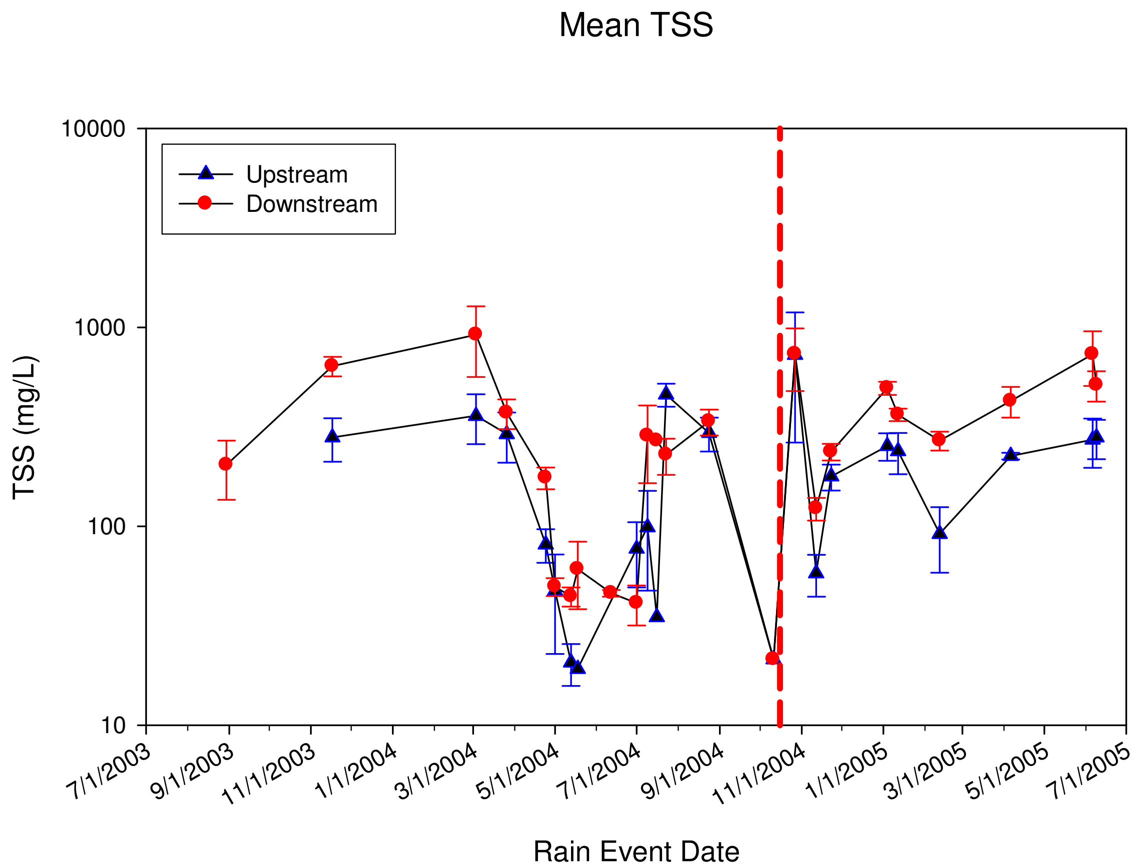


Figure 8.—Mean TSS (± S.E.) for each rain event throughout the study period for upstream and downstream study sites pooled separately. The vertical-dashed line represents the start of road construction activities.

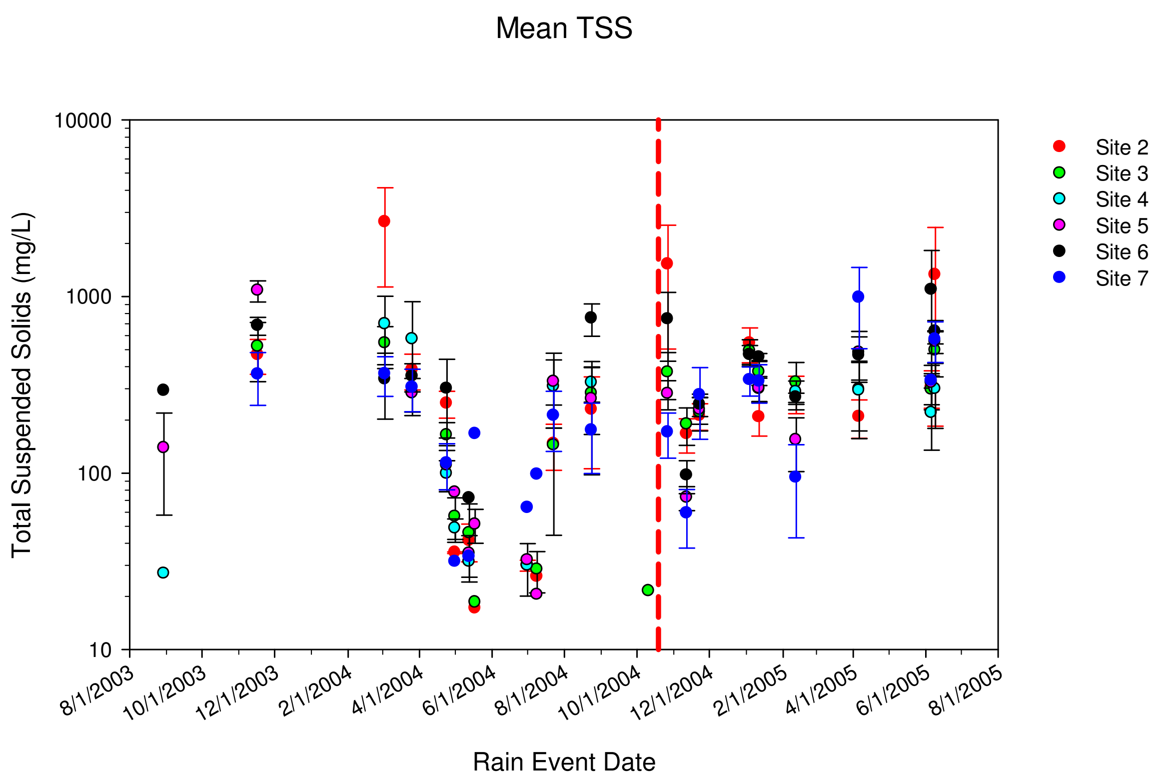


Figure 9.—Mean TSS (± S.E.) for each rain event throughout the study period at each-downstream-study site. The vertical, dashed line denotes the start of road construction.

*Sediment-Discharge Relation.—*A significant, positive relation existed between TSS and stream stage for suspended-sediment (untransformed) data gathered throughout the study with all sites combined (Linear Regression; $p < 0.0001$) (Figure 10). Furthermore, a general trend was evident in the relation between TSS and discharge for most-rain events (e.g., Figure 11). Numerous storm hydrographs exhibited a flashy nature made evident by the rapid rise and fall of Brush Creek which commonly occurred within a 24-hour time period (Figure 11). TSS concentrations tended to mirror the rising limb of the hydrograph for rain events of varying magnitude. The maximum sediment concentration occurred at, or immediately before, the peak in the hydrograph for successfully sampled rain events with data collected from the single stage-sediment samplers (Figure 11). Most significant rain events had a maximum TSS concentration that equaled or exceeded 100 mg/L with several events approaching 1000 mg/L.

The automated sampler at sites 3 and 7 provided further resolution of the suspended sediment dynamics during the storm hydrographs for rain events that occurred after May 2004. The TSS concentrations measured with the automated samplers accurately reflected that of the single-stage samplers during the rising limb of the hydrograph (Figure 11). The maximum-sediment concentration primarily occurred at the peak of the hydrograph for data collected with the automated sampler (Figure 11). There was little disparity in TSS concentrations between the single-stage and automated samplers at similar discharges. The automated sampler results show that TSS concentrations returned below 100 mg/L within 6-24 hours depending on the intensity of the storm event (Figure 11).

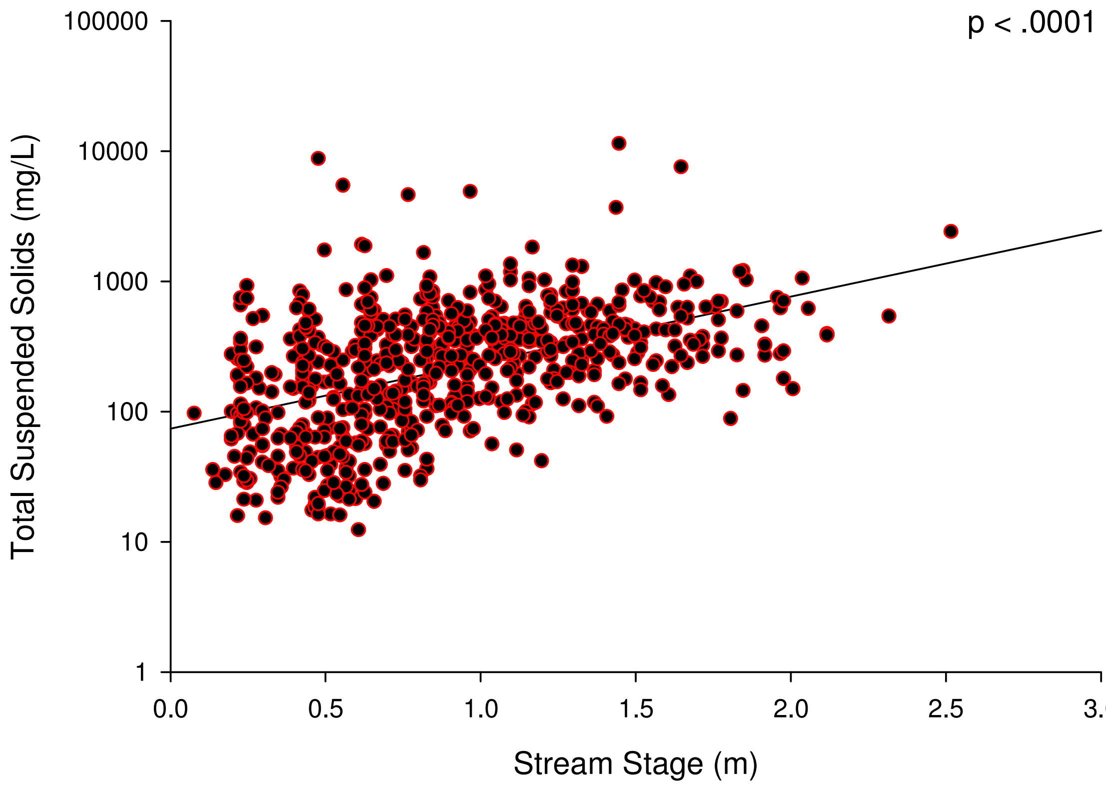
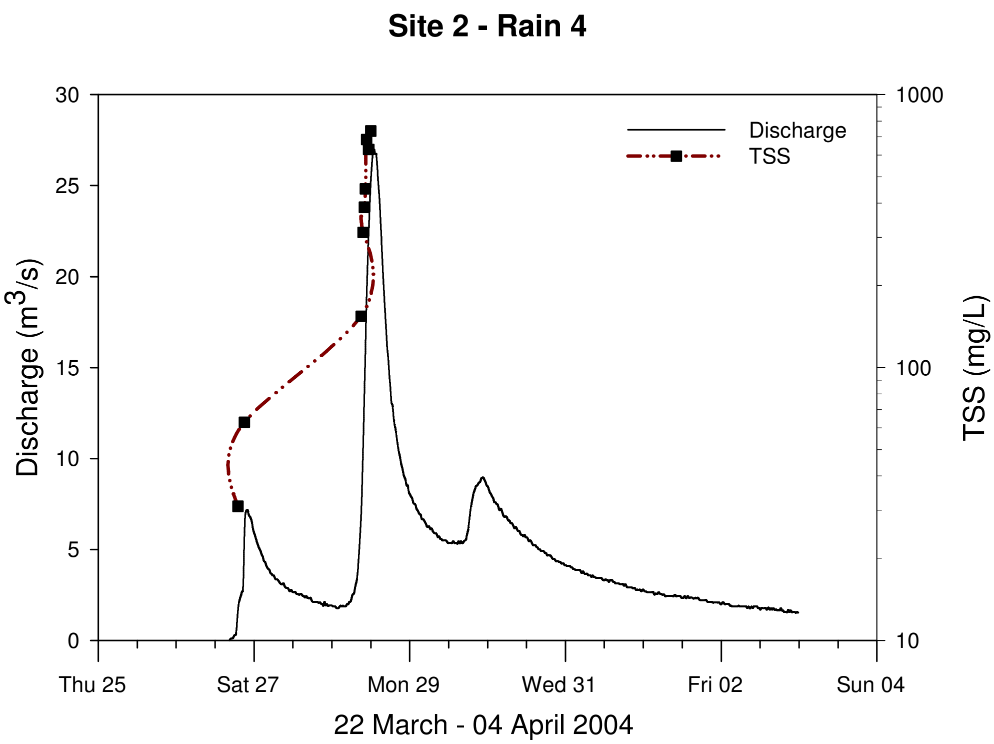


Figure 10.—Linear regression of TSS and stream stage for untransformed data with all sites pooled.



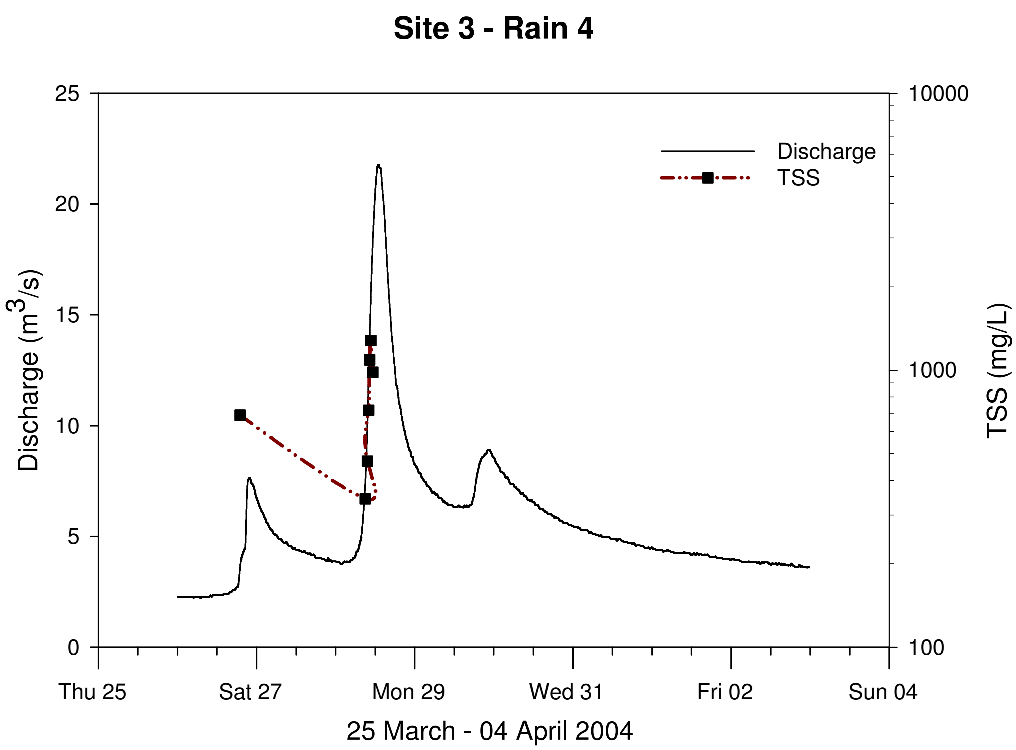
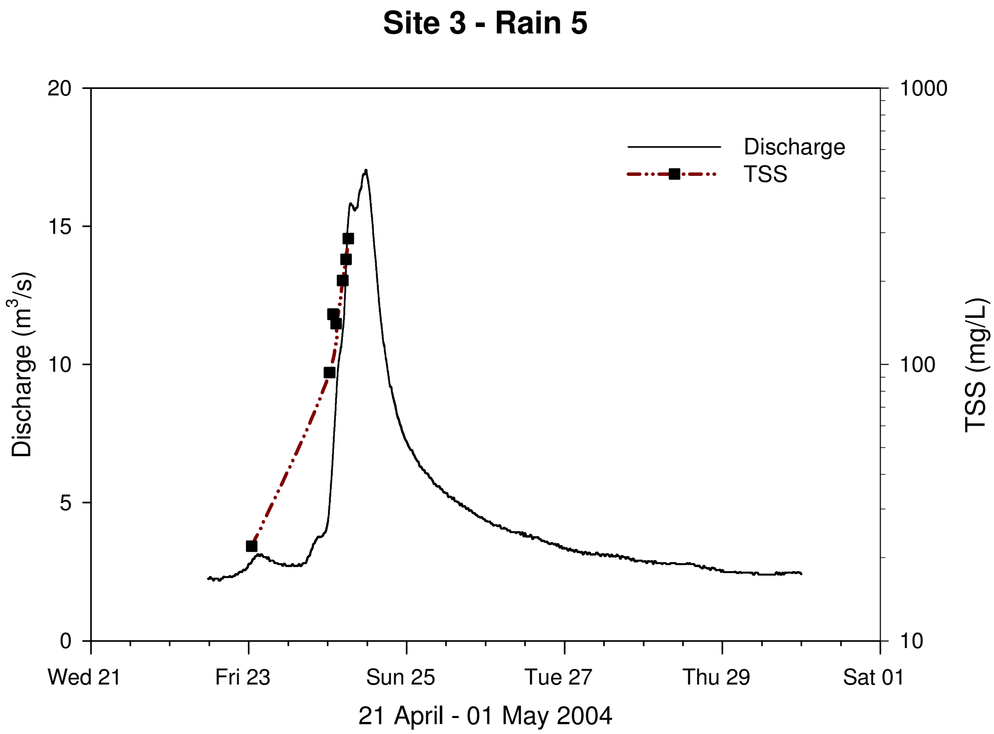


Figure 11.—The hydrograph and sedigraph for individual rain events at sites where uncompromised samples were collected. TSS was measured using only single-stage samplers for rain events 4 and 5. Both single-stage and automated samplers were used to measure TSS for all other rain events at sites 3 and 7.



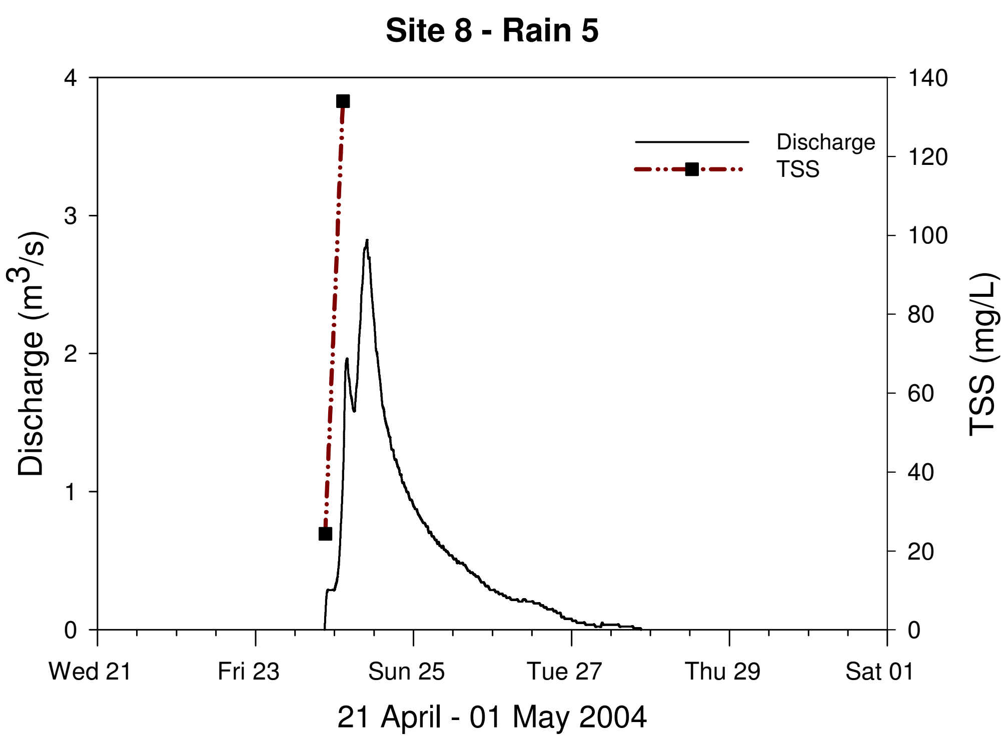
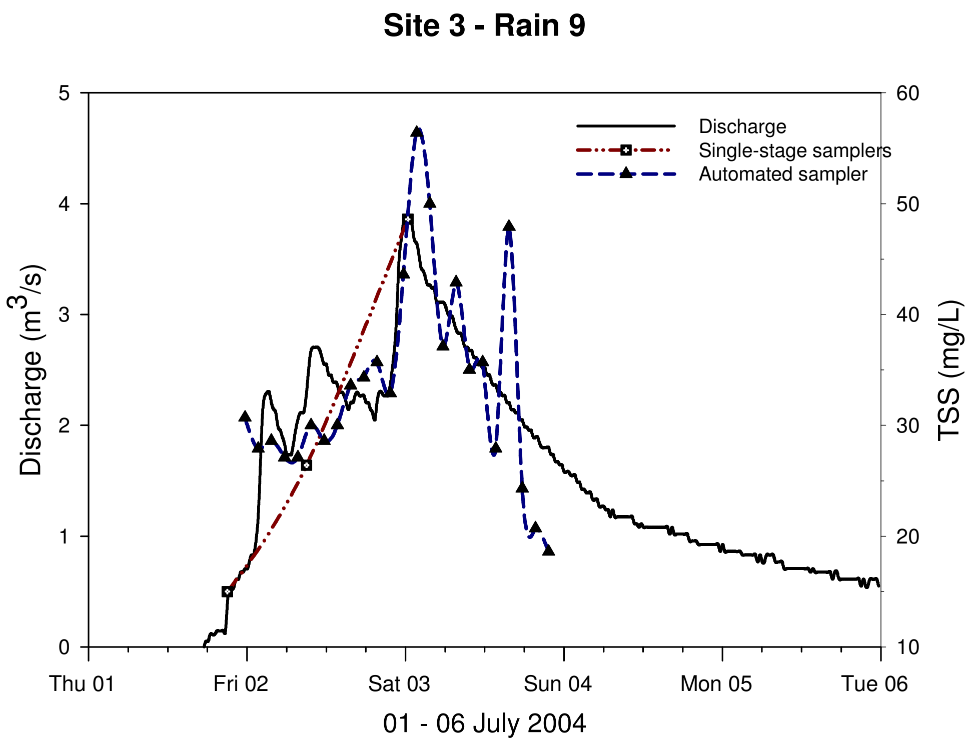


Figure 11.—continued.



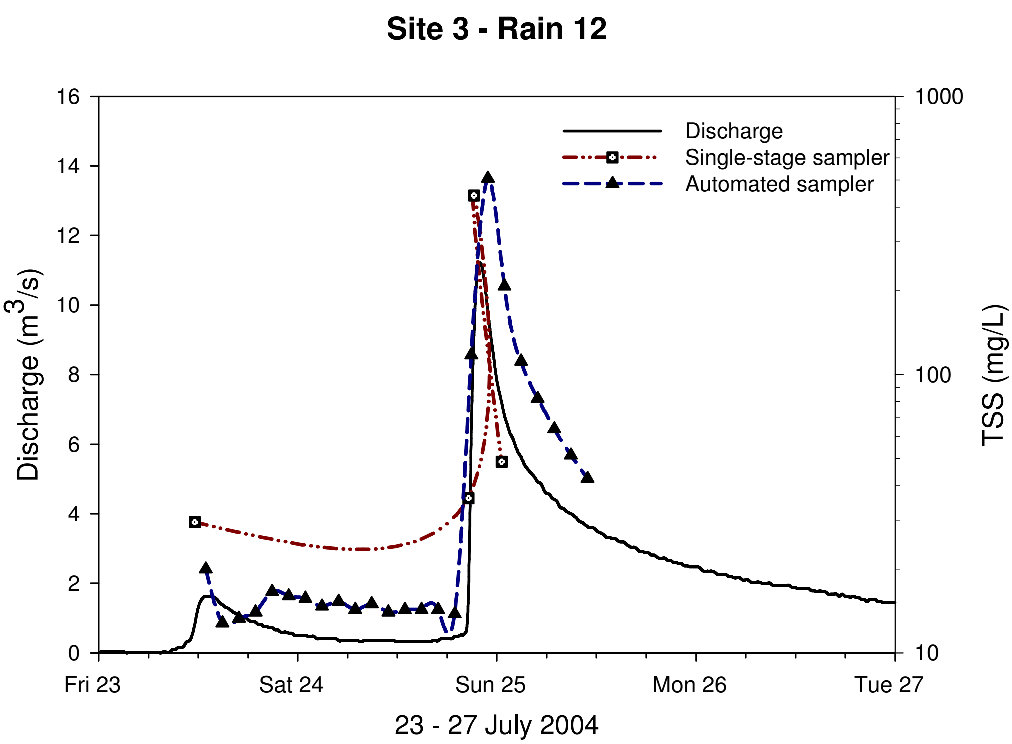
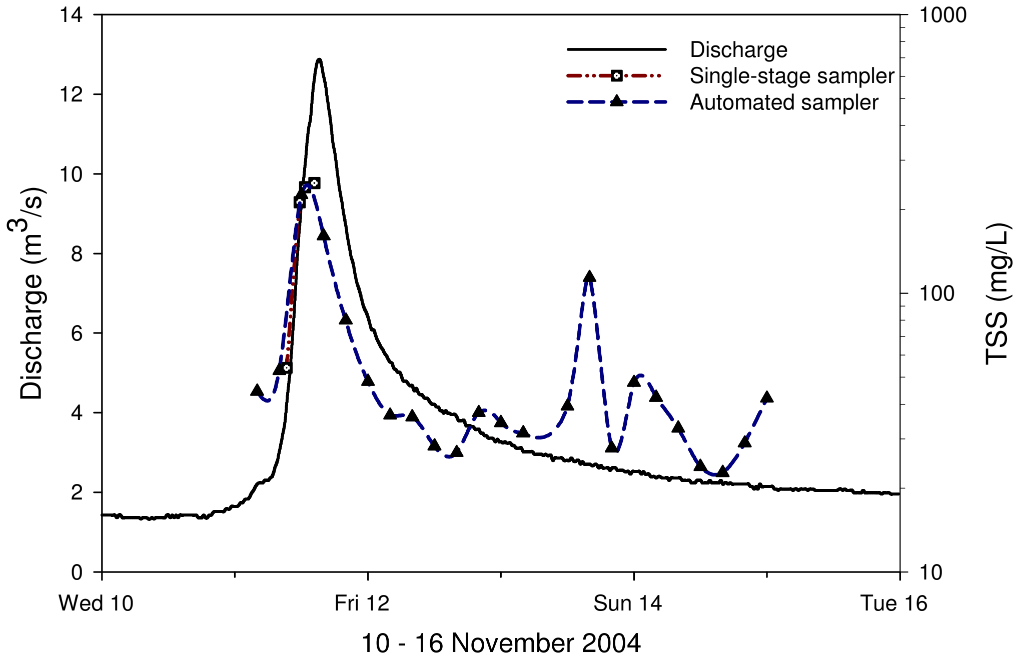
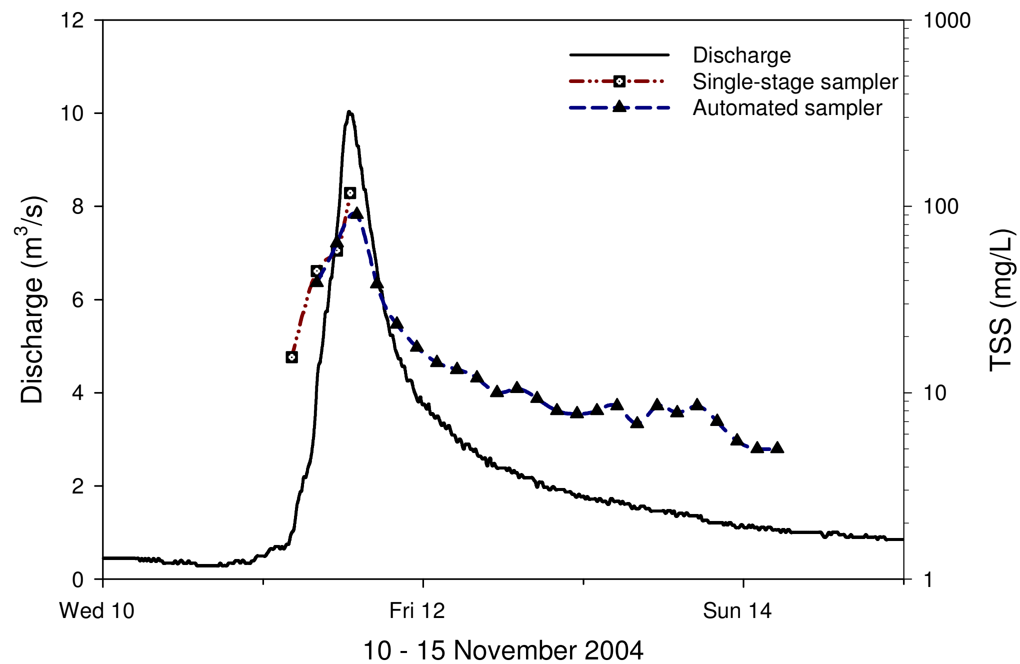


Figure 11.—continued.

Site 3 - Rain 16



Site 7 - Rain 16



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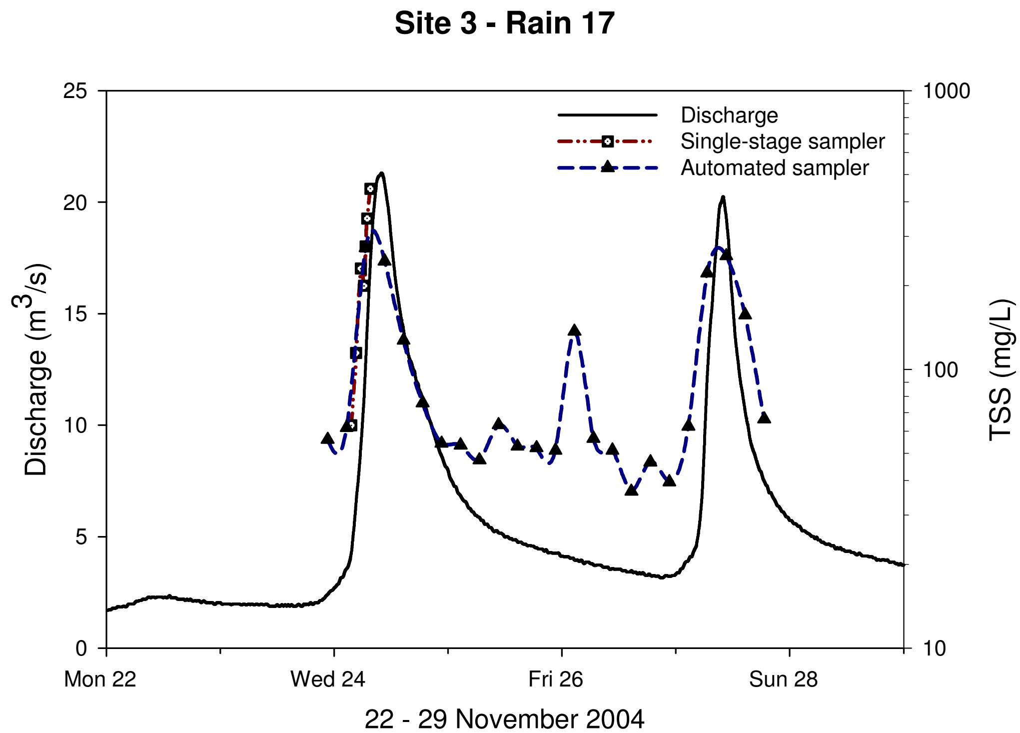

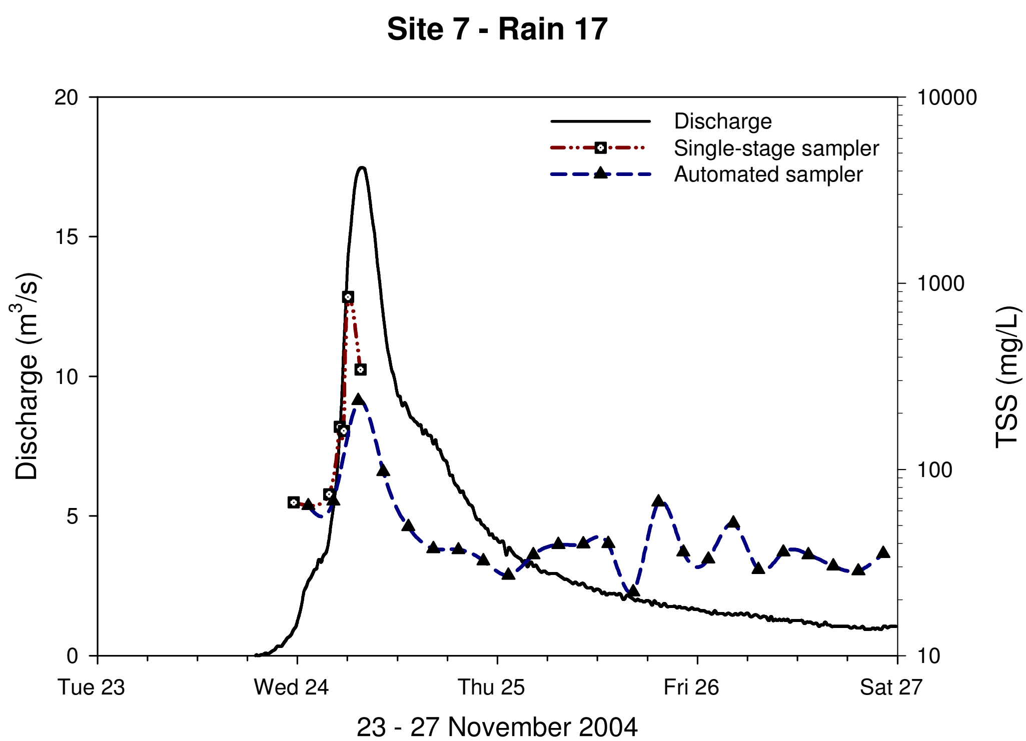

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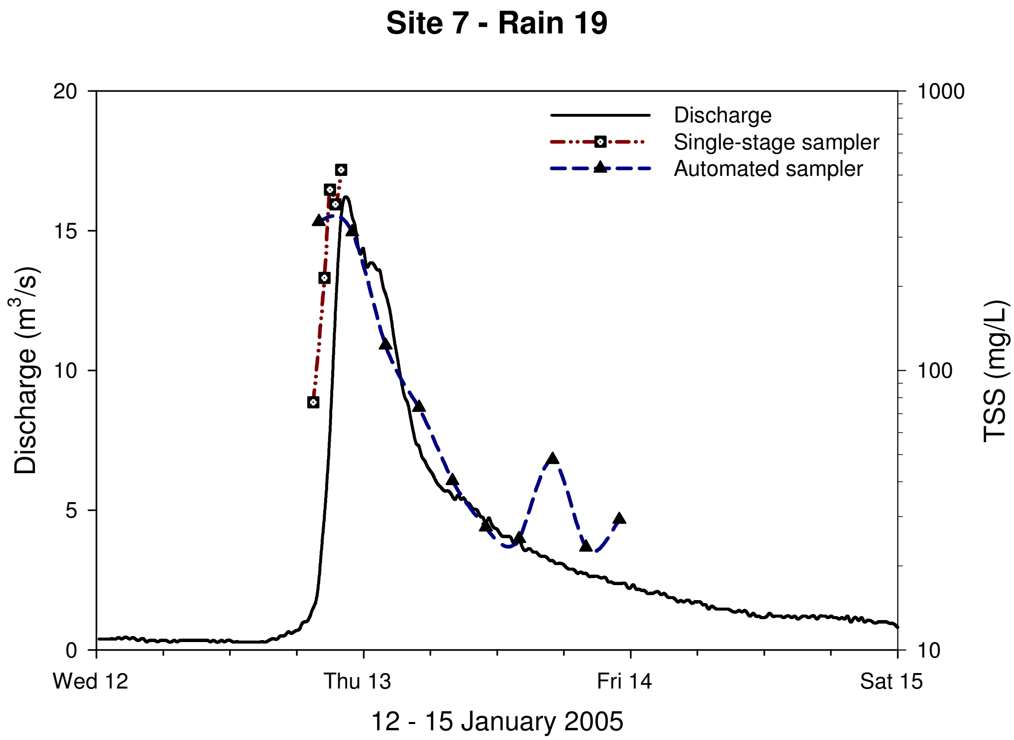

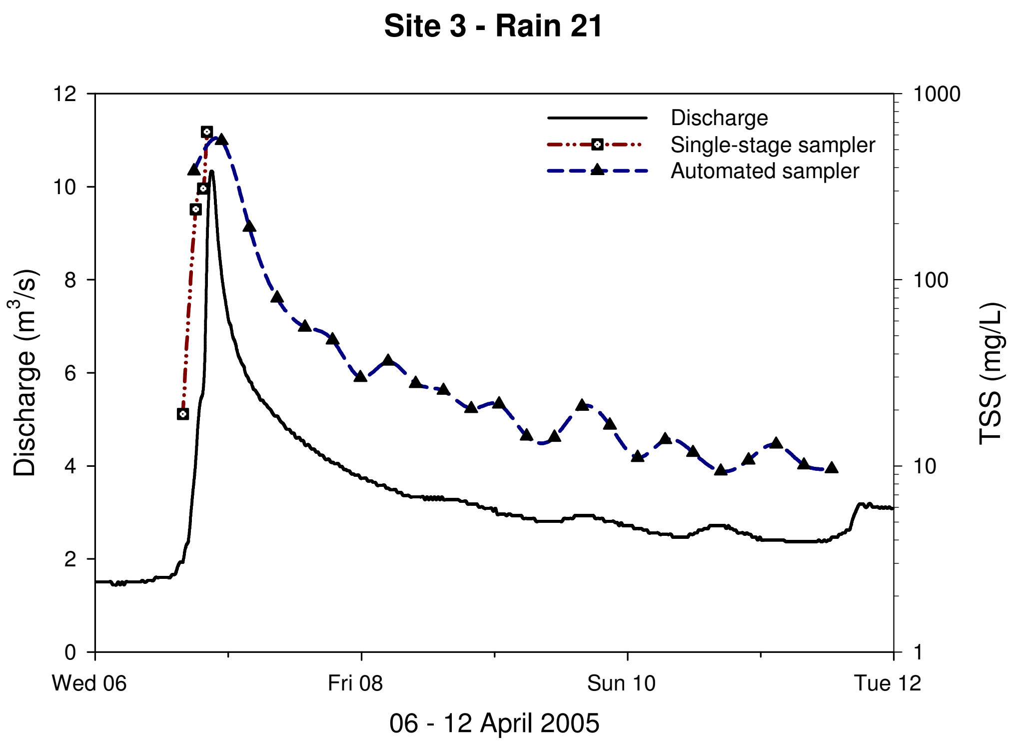

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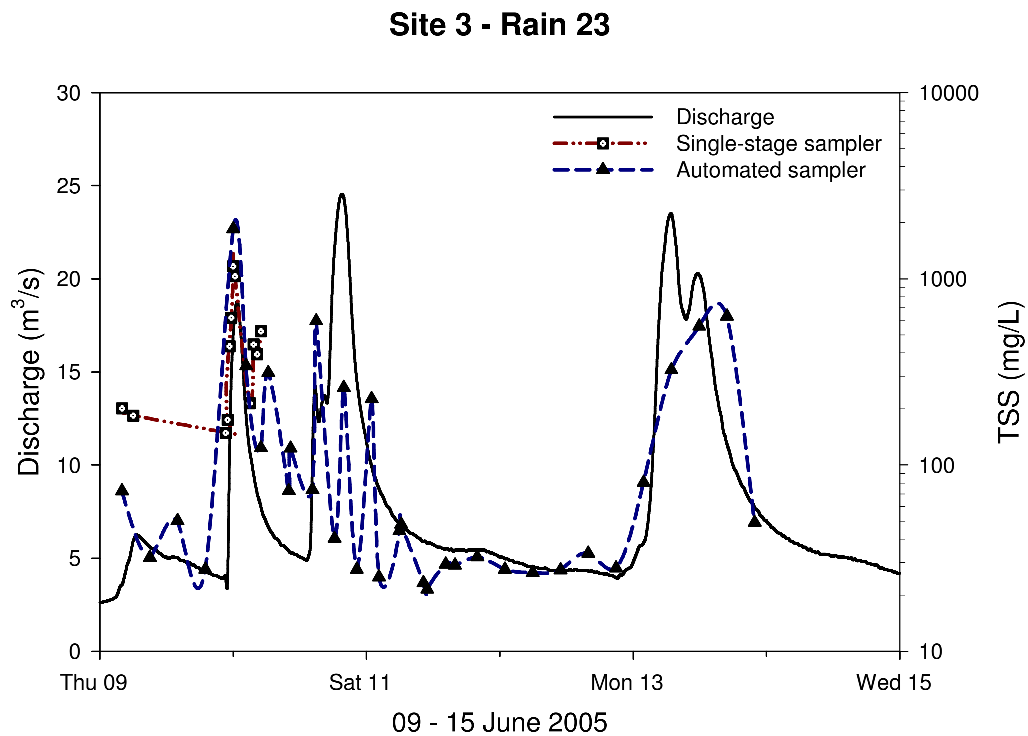


Figure 11.—continued.

Turbidity and TSS Correlations.—Turbidity and TSS were highly related for all data ($\log_{10}$ transformed) collected throughout the study [$R^2 = 0.93$, $p < 0.0001$, $\log_{10}(TSS) = 0.2596 + 1.0216 * \log_{10}(NTU)$] (Figure 12). The relation between NTU and TSS was nearly 1:1 for all data collected. Turbidity was a successful predictor of TSS concentrations.

Seasonal Characteristics of Suspended Sediment for Baseline Data.—The baseline TSS data (i.e., data from downstream sites before construction and all data from upstream sites) was broken into four seasons as follows: spring (March, April, and May), summer (June, July, and August), fall (September, October, and November), and winter (December, January, and February). A Kruskal-Wallis test showed there was a statistical difference in TSS concentrations between the four seasons (Prob > ChiSq = 0.0206).

A series of comparisons using the Kruskal-Wallis test showed the mean TSS concentration in the fall was significantly different from all other seasons (Table 3.1). More specifically, the mean TSS concentration in the fall was 34 %, 88 %, and 123 % greater than the spring, summer, and winter TSS concentrations, respectively. Winter exhibited the lowest variation in TSS concentrations in relation to all other seasons and also had the smallest-sample size (Table 3.1).

The inorganic solids (expressed as percentage of TSS) for baseline data composed the largest fraction of TSS during all seasons (Table 3.2). There were no seasonal differences in the proportion of inorganic solids for baseline conditions. The organic solids composed a small fraction of TSS and did not exhibit any seasonal differences (Table 3.3). In general, the mean proportion of TSS was approximately 84% inorganic solids and 16% organic solids across seasons for a ratio of 5.25:1, respectively.

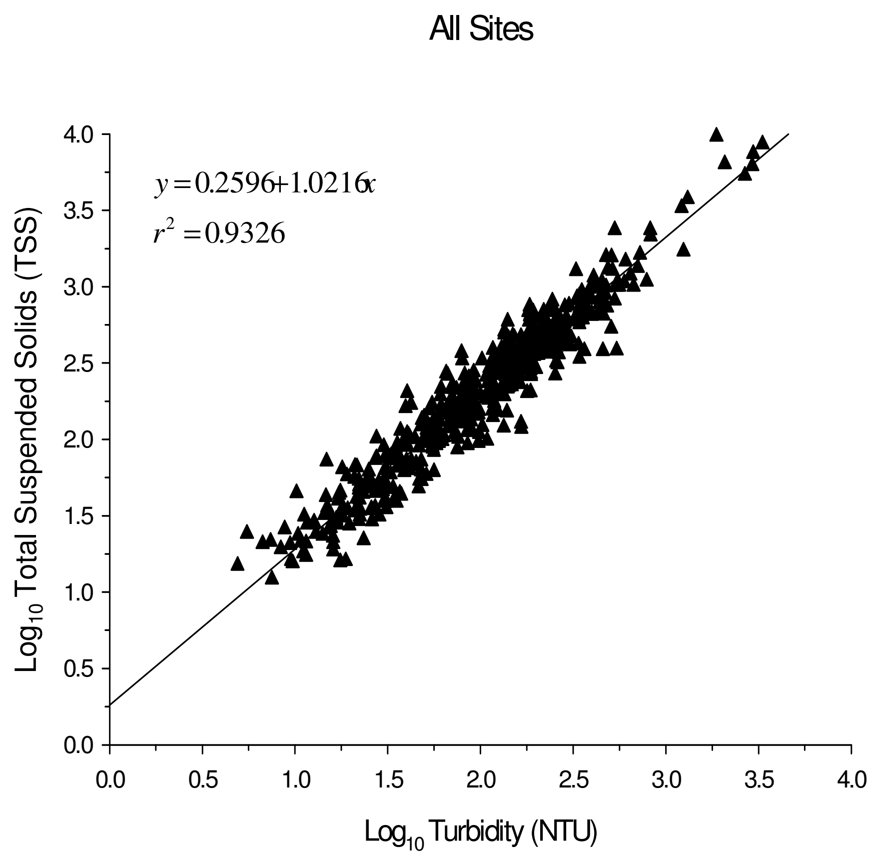


Figure 12.—Linear regression of Log_{10} NTU vs. Log_{10} TSS with data from all study sites and rain events throughout the study.

Table 3.1—The mean and standard deviation (S.D.) of TSS concentrations (mg/L) for all seasons during the baseline period of the study. Means with the same letter are not statistically different at an alpha of 0.05.

Season	n	Mean	Min	Max	S.D.
spring	166	361 (a)	12	11185	1051
summer	113	258 (a)	15	907	229
fall	68	484 (b)	16	5340	710
winter	26	217 (a)	27	534	149

Table 3.2—The mean and standard deviation (S.D.) for the proportion of inorganic solids expressed as a percentage of the TSS concentration for all baseline data across seasons.

Season	n	Mean %	Min	Max	S.D.
spring	163	84	53	100	8
summer	93	83	30	91	9
fall	68	84	61	97	6
winter	26	85	79	90	3

Table 3.3—The mean and standard deviation (S.D.) for the proportion of organic solids expressed as a percentage of the TSS concentration for all baseline data across seasons.

Season	n	Mean %	Min	Max	S.D.
spring	163	16	2	47	8
summer	93	17	8	70	9
fall	68	16	3	38	6
winter	26	15	10	21	3

RESULTS

Deposited Sediment

Characteristics of Deposited Sediment Dry Weight.—The Model III ANOVA showed that time (i.e., sampling period) was the only factor that resulted in significant differences (ANOVA; $p < 0.0001$) of deposited sediment dry weight when site, site*habitat, and site nested within habitat and time were the random effects of the model (Table 4). These results indicate that no significant differences in deposited sediment dry weight existed between riffle, run and pool habitats. The same results were found using data from the downstream sites only. Additionally, a separate contrast was performed to test for an overall difference in deposited sediment dry weight before (September 2003-September 2004) versus after (December 2004-May 2005) the start of road construction. This contrast showed that a significant difference existed (ANOVA; $p <0.0001$).

Figure 13 displays the significant differences across sampling periods for each habitat with all downstream sites pooled together. After the start of road construction, the most notable increases in deposited sediment dry weight were in pool and riffle habitat on December 2004 and March 2005, respectively (Figure 13). However, this did not appear to be a trend as deposited sediment in all habitats returned to pre-construction levels in May 2005.

Table 4.—Mixed-model ANOVA table for deposited sediment dry weights with habitat, time (i.e., sampling period), and habitat*time as the fixed effects for data from all sites and habitats throughout the study period. The random effects are site, site*habitat, and site nested within habitat/time. All data was square root transformed to meet model assumptions. The compound symmetry covariance structure was used due to the smallest Bayesian Information Criterion (BIC) value.

Fixed Effects	Num df	Den df	F value	Pr > F
habitat	2	8	1.11	0.3741
time	7	36	11.68	<.0001
habitat*time	14	36	0.59	0.8572

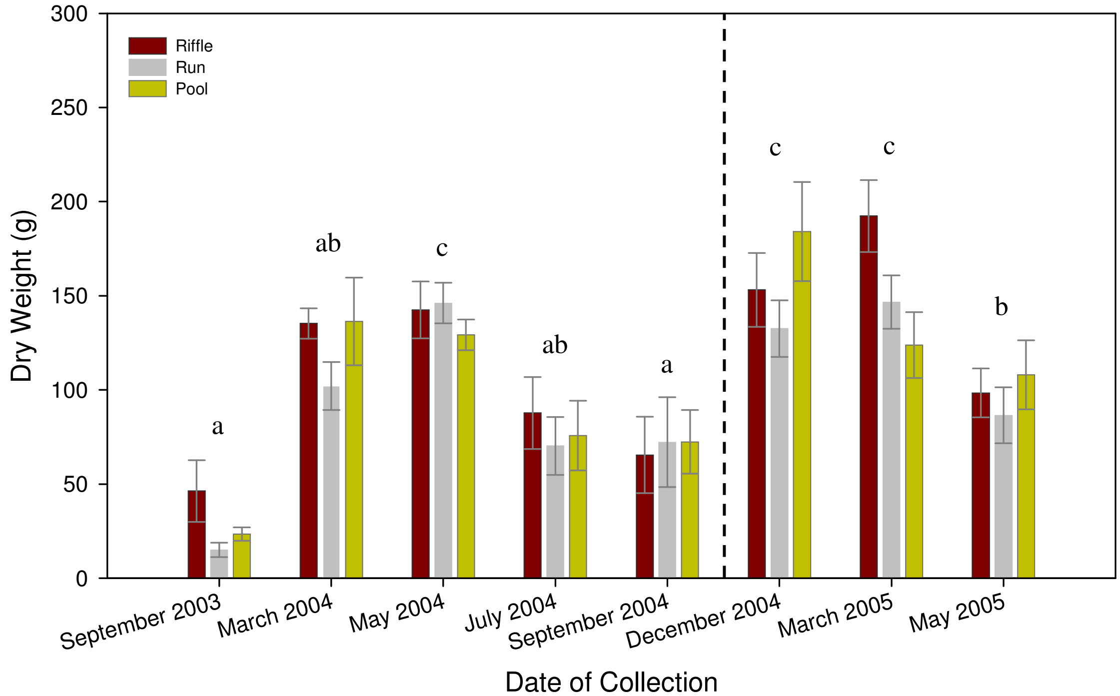


Figure 13.—Mean (± S.E.) of deposited sediment dry weight for each habitat and sample date with data from all downstream sites pooled. Sampling dates with the same letter above are not statistically different (α = 0.05). The dashed-vertical line denotes the start of road construction.

*Site Characteristics of Deposited Sediment Dry Weight.—*The variation in deposited sediment was low for most sites and habitats throughout the study period (Table 5). For all raw data where one or more samples were collected, the coefficient of variation ranged from 1% to 115% and (when expressed as a percentage) was rarely greater than the associated mean. No single habitat displayed notably lower variation across all sites and sampling periods (Table 5). However, slightly greater variation in deposited sediment measurements tended to exist in run habitat except for site 4 where pools displayed notably greater variation (Table 5). The only-significant difference between habitats was at site 4 for samples collected on July 2004. A small-sample size made seasonal comparisons of the variation in deposited sediment difficult to determine.

With data from all habitats pooled, no single site consistently displayed increased amounts of deposited sediment across sampling periods, where an adequate sample size was obtained (Figure 14). Sites 7 and 2 were the only sites that showed a significant increase in deposited sediment (relative to all other sites) on March and May 2005, respectively; however, this was not a consistent trend for all-sampling periods after construction began (Figure 14).

Table 5.—The mean and coefficient of variation (displayed as a fraction in parentheses) of deposited sediment dry weight (g/sampler) for riffle, run, and pool habitats at all sites across time. Sample size (n) ranged from two to four for the mean values. All missing values indicate a sample size ≤ 1. The asterisk notes a significant difference between habitats (α = 0.05).

		Collection Date							
Site	Habitat	Sep-03	Mar-04	May-04	Jul-04	Sep-04	Dec-04	Mar-05	May-05
2	riffle		131.2 (0.12)		73.6 (0.17)	144.3 (0.16)	189.5 (0.17)	175.2 (0.07)	144.7 (0.15)
2	run		102.1 (0.18)		126.1 (0.62)	155.0 (0.67)			131.3 (0.37)
2	pool		115.1 (0.20)		134.6 (0.19)	122.0 (0.32)		122.0 (0.33)	173.3 (0.11)
3	riffle	78.0 (0.20)		137.2 (0.13)	63.7 (1.13)	36.0 (0.07)		152.2 (0.08)	134.5 (0.30)
3	run	22.3 (0.58)		156.7 (0.12)	74.9 (0.53)	87.2 (0.98)	137.9 (0.40)		93.6 (0.55)
3	pool				99.9 (0.20)	76.6 (0.01)			144.8 (0.40)
4	riffle			130.0 (0.16)	184.0 (0.02)				75.2 (0.27)
4	run			157.2 (0.25)	40.0 (0.75)*	23.1 (0.49)	149.4 (0.20)	168.5 (0.11)	80.9 (0.88)
4	pool				8.4 (0.65)	39.9 (1.03)	210.5 (0.42)		50.8 (0.68)
5	riffle				49.3 (0.84)	12.6 (0.02)	107.2 (0.10)		72.4 (0.60)
5	run	10.6 (0.18)			54.9 (0.52)	28.0 (0.16)	88.5 (0.51)	101.5 (0.45)	
5	pool	24.4 (0.34)		121.2 (0.01)	60.0 (0.32)		122.3 (0.10)	125.5 (0.36)	62.9 (0.49)
7	riffle	32.8 (1.15)		166.4 (0.49)	80.1 (0.81)			249.6 (0.07)	59.9 (0.08)
7	run	8.6 (0.36)		124.5 (0.11)	65.0 (1.11)				60.8 (0.70)
8	riffle			127.1 (0.44)				126.9 (0.21)	37.5 (0.38)
8	run			101.9 (0.58)	12.0 (0.30)				5.9 (0.30)

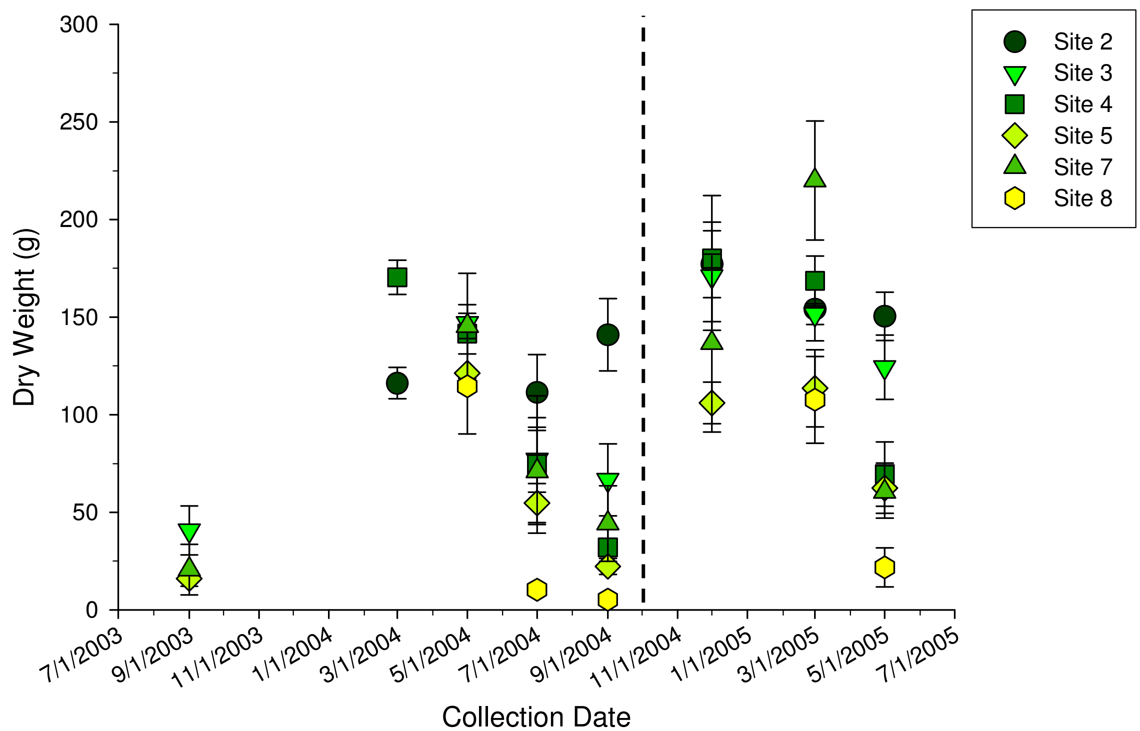


Figure 14.—Mean ± standard error of deposited sediment dry weight for each site and sampling period with all habitats pooled together. The vertical-dashed line denotes the start of road construction.

<u>*Particle-Size Characteristics of Deposited Sediment Dry Weight.—*</u>Fine sands (0.25-0.125 mm) composed the largest-mean proportion of deposited sediments in run habitat at sites 2, 4, 5, and 7 throughout the study period where an adequate sample size (n > 1) was obtained (Figure 15). This was also the dominant particle size in pool habitat at sites 2 and 5 throughout the study period. With data from all-downstream sites pooled together, fine sand was notably the dominant particle size in run and pool habitats across nearly all sampling periods, while fine sand was also the dominant-particle size in riffle habitat for five sampling periods (Figure 15).

Among habitats, there was no significant trend in the mean proportion of coarse silt (<0.063mm) particles (fines) for all downstream sites pooled together after the start of road construction (Figure 15). Additionally, the greatest mean proportion of coarse silt for pooled downstream sites was present in the collections made on July and September 2004 for all habitats (Figure 15). These-mean proportions were not exceeded after the start of road construction.

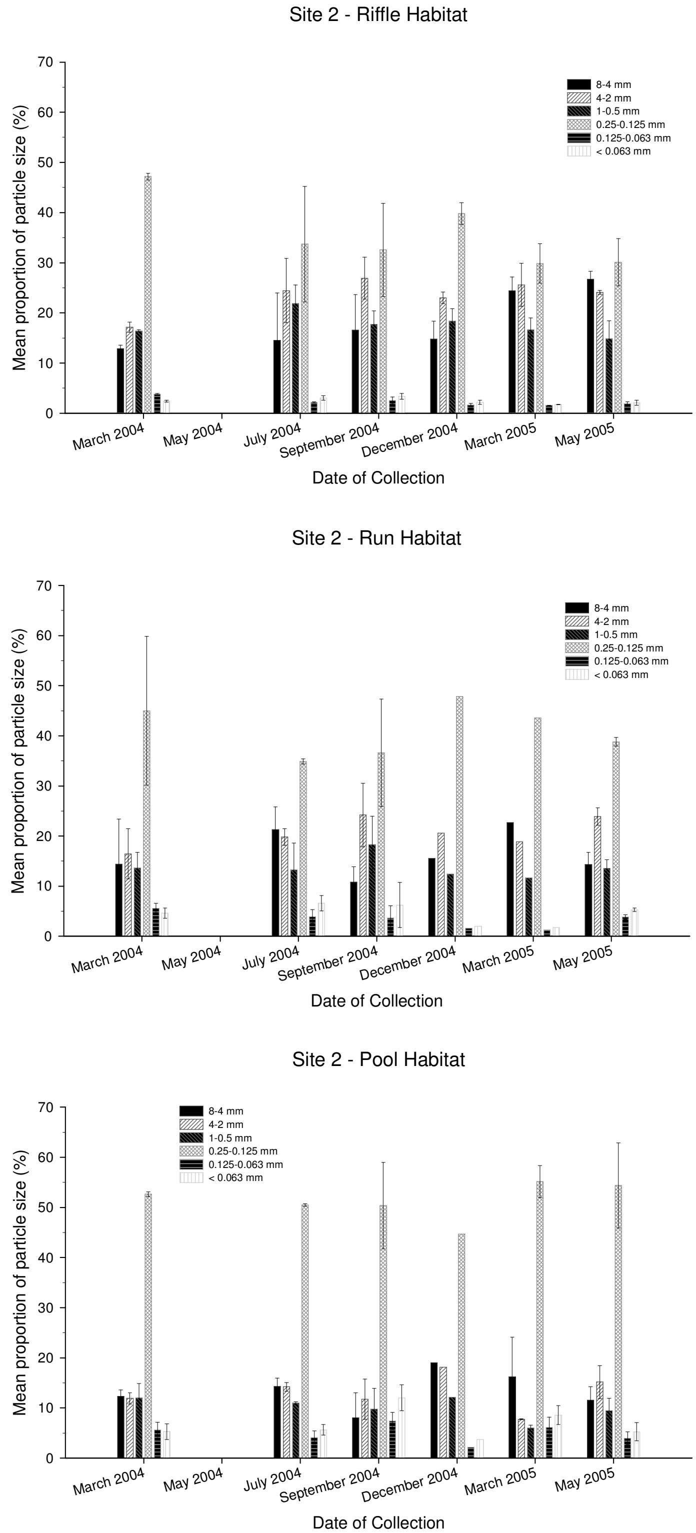
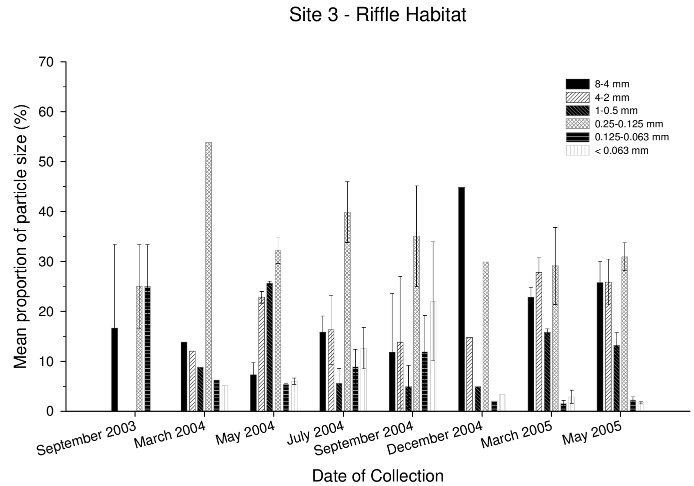
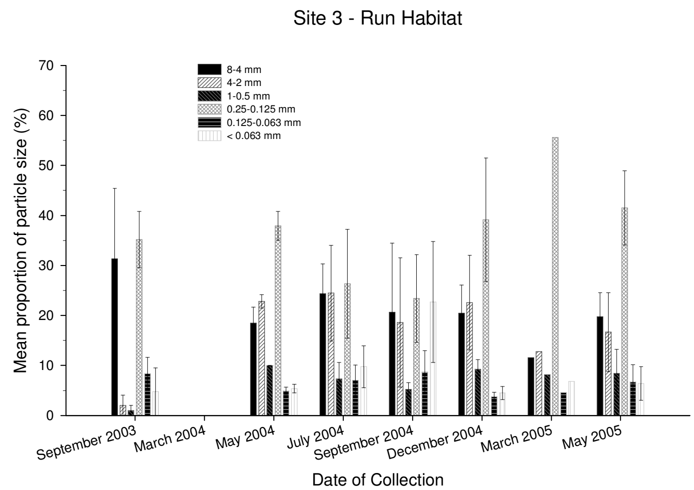


Figure 15.—Mean proportion (%) ± standard error of deposited sediment dry weight for each-particle-size class across collection dates and habitats. Particle size classes (from largest to smallest) are fine gravel, very fine gravel, coarse sand, fine sand, very fine sand, and coarse silt (fines). N = 1 to 4 for all sites and habitats.





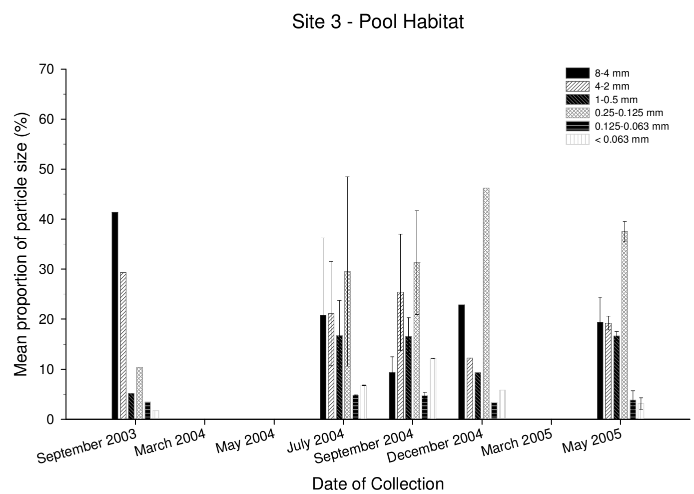


Figure 15.—continued for site 3, riffle, run, and pool habitats.

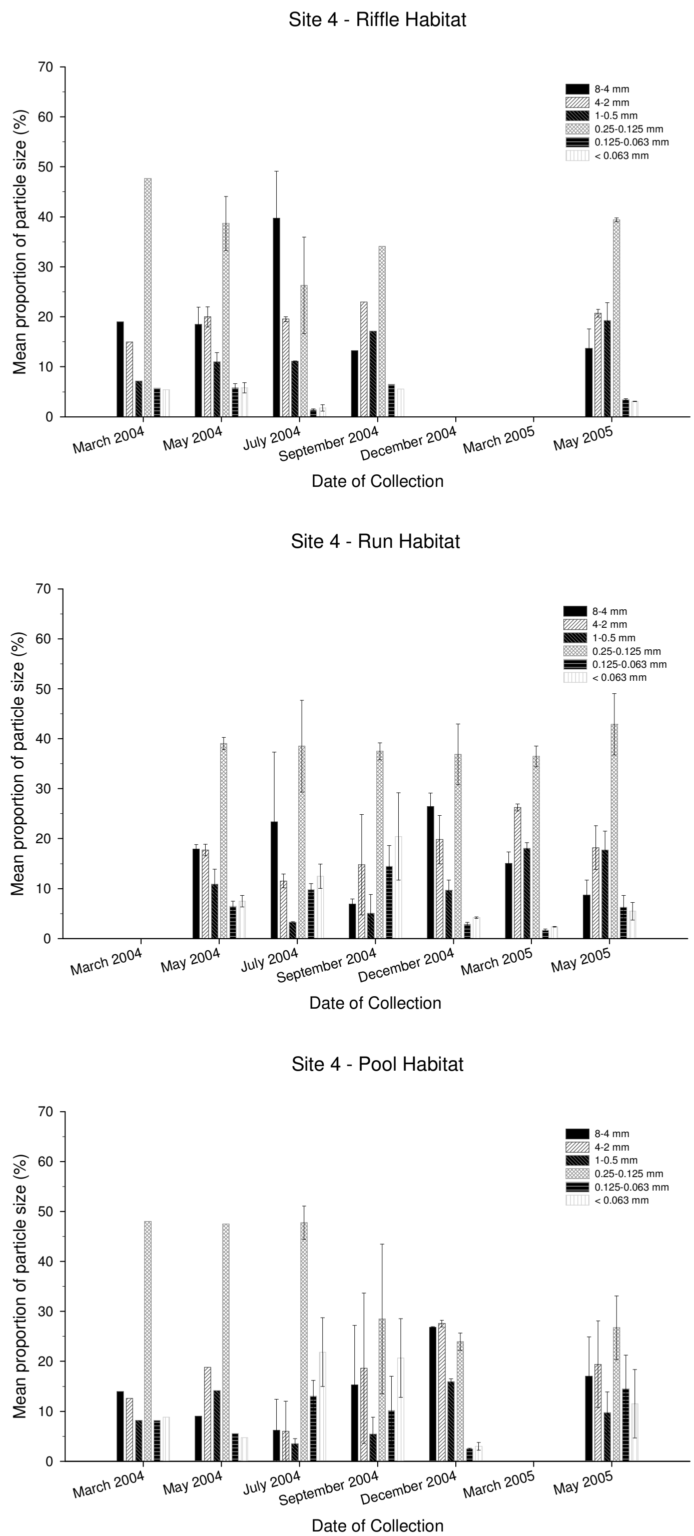


Figure 15.—continued for site 4, riffle, run, and pool habitats.

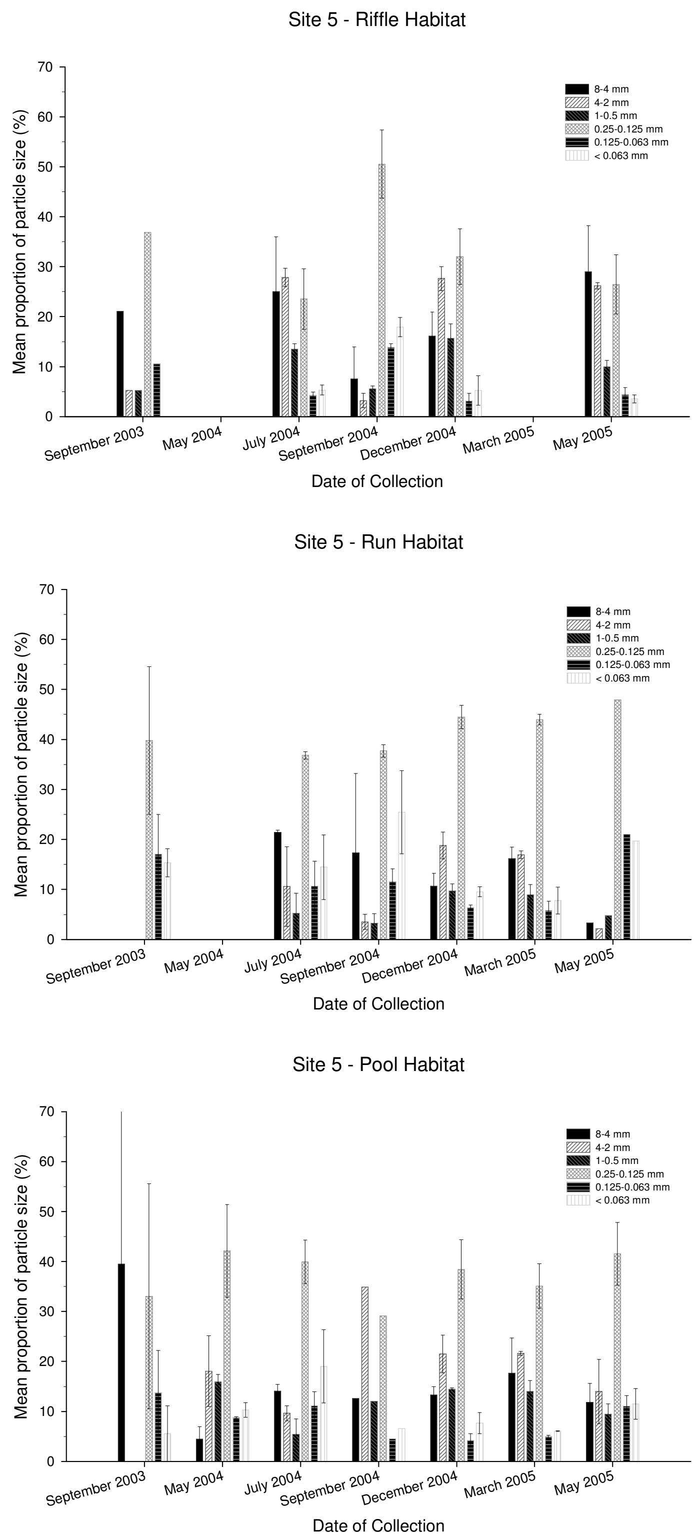
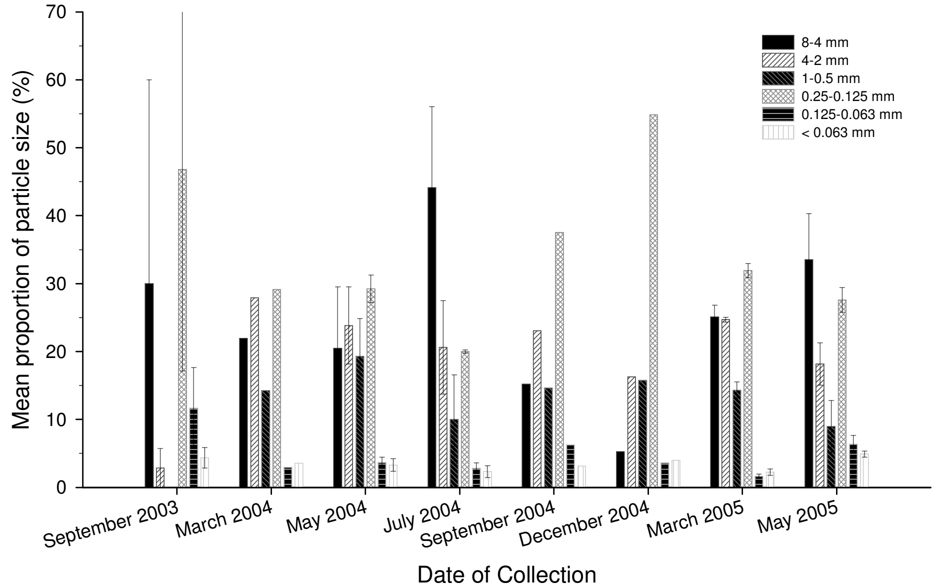


Figure 15.—continued for site 5, riffle, run, and pool habitats.

Site 7 - Riffle Habitat



Site 7 - Run Habitat

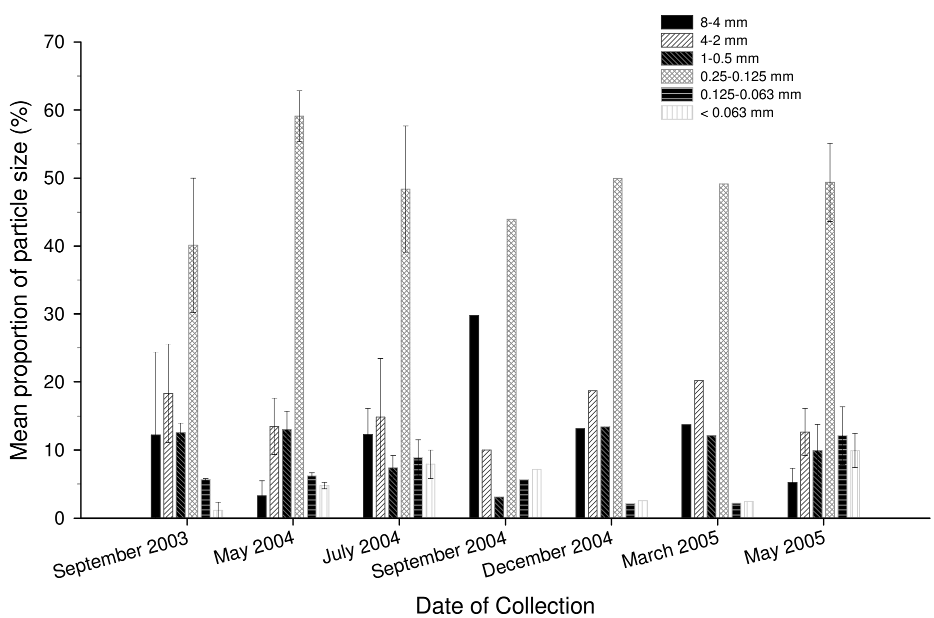
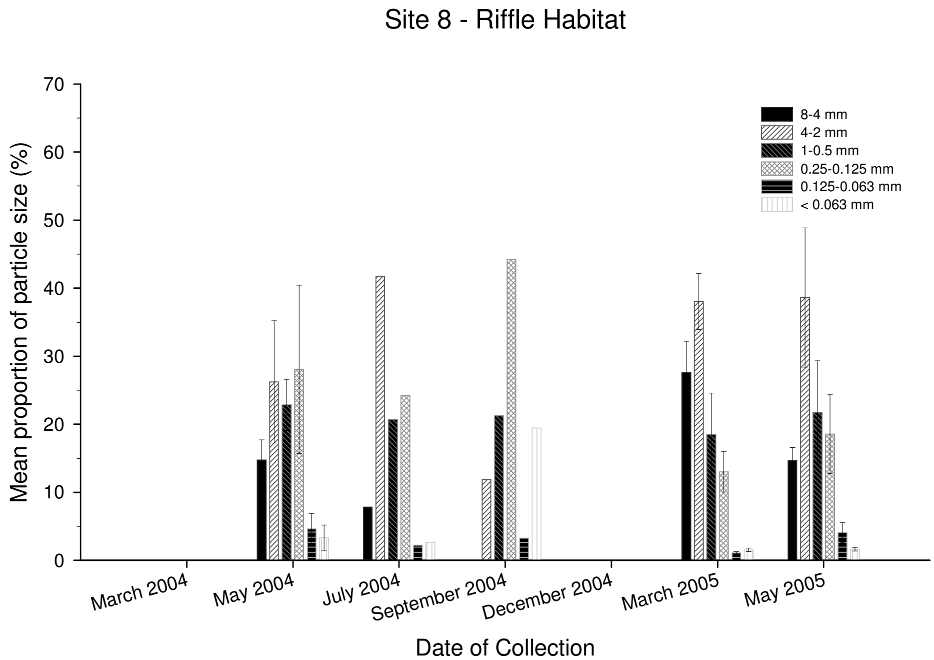


Figure 15.—continued for site 7, riffle and run habitats.



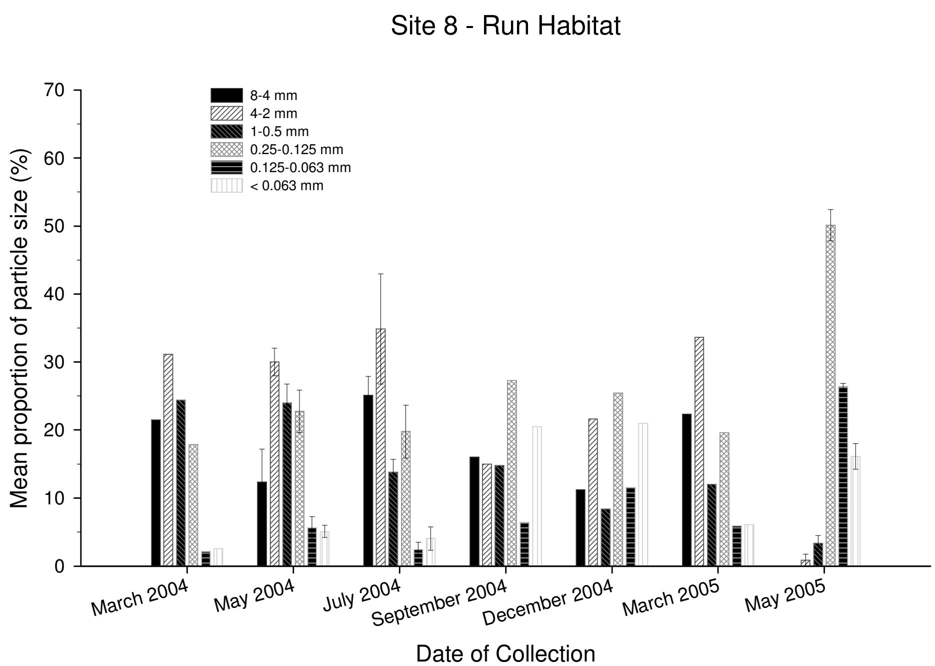


Figure 15.—continued for site 8, riffle and run habitats.

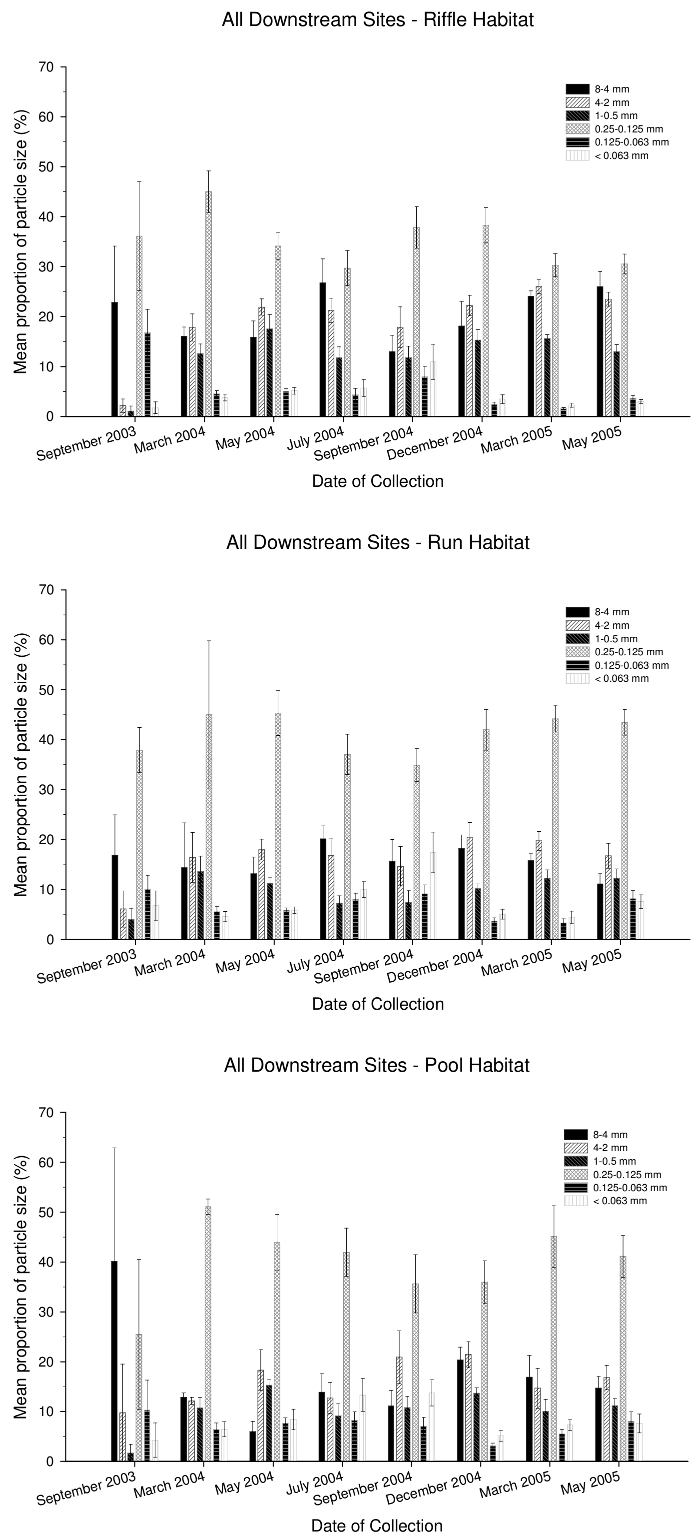


Figure 15.—continued for all downstream sites pooled; riffle, run, and pool habitats.

Characteristics of Deposited Sediment Particles < 2 mm in Size.—Due to a small-sample size for individual mesohabitats at each site, the two-way mixed-model ANOVA procedure used to analyze deposited sediment dry weight was also used to analyze the proportion of the sediment-dry weight composed of particles < 2mm in size. The results of this analysis showed no significant differences between habitats, sampling periods, or the interaction term at an alpha level of 0.05 (Table 6). The same (non-significant) results were found with data from the downstream sites only.

Additionally, a separate contrast was performed to test for an overall difference in the proportion of < 2mm size particles before (September 2003-September 2004) versus after (December 2004-May 2005) the start of road construction. This contrast showed that no significant difference existed (ANOVA; $p = 0.1912$).

Organic Proportion of Deposited Sediment.—The organic fraction of deposited sediment dry weight was only determined in the laboratory through December 2004. Throughout that time period, the organic proportion of deposited sediment dry weight was low for all habitats (Table 7). There were no significant differences in the organic fraction between habitats for individual-sampling periods as determined by a one-way ANOVA ($p > 0.05$). The organic fraction of the total dry weight ranged from 1% to 14% with an overall mean of 4%; thus, the mean inorganic proportion was 96%. Overall, the resulting inorganic:organic ratio was 24:1.

Retrieval of Deposited Sediment Samplers.—I was able to successfully retrieve an overall mean of 72% of the deposited sediment samplers across all sampling periods. The lowest retrieval was 34% in March 2004. Overall, run habitat had the highest mean retrieval rate (76%) while 70% of samplers were retrieved in both riffle and pool habitats.

Table 6.—Mixed-model ANOVA table with habitat, time (i.e., sampling period), and habitat*time as the fixed effects for % < 2mm particle size data from all sites and habitats. The random effects in the model are site, site*habitat, and site nested within habitat/time. All data was square-root transformed to meet the assumption of normality. The compound symmetry covariance structure was used due to the smallest Bayesian Information Criterion (BIC) value.

Fixed Effects	Num df	Den df	F value	Pr > F
habitat	2	8	2.05	0.1911
time	7	36	1.85	0.1079
habitat*time	14	36	1.13	0.3682

Table 7.—The mean-organic proportion (%) and coefficient of variation (in parentheses) of deposited sediment dry weight for all habitats across sampling periods with sites pooled. The organic proportion was not determined for collections made after December 2004.

	Collection Date					
	Sep-03	Mar-04	May-04	Jul-04	Sep-04	Dec-04
riffle	3.4 (70.8)	1.3 (43.3)	2.1 (15.8)	3.3 (45.3)	6.0 (63.0)	2.1 (32.2)
run	6.3 (32.8)	2.0 (0)	2.1 (30.2)	4.5 (45.5)	5.7 (49.5)	3.1 (85.7)
pool	6.7 (48.2)	1.7 (34.6)	2.3 (24.7)	5.9 (77.9)	5.0 (73.8)	2.0 (31.6)

Surface Cover and Embeddedness Estimates

<u>*Percent-Surface Cover Characteristics.—*</u>Results of the mixed-model ANOVA showed there were significant differences between habitats (ANOVA; $p = 0.0109$) and sampling periods (ANOVA; $p < 0.0001$) for the percent-surface-cover data (Table 8). With data from downstream sites pooled together, the percent-surface cover of fine sediments was significantly greater (ANOVA; $p < 0.05$) in run habitat for all sampling dates, except July 2004 (Figure 16). Mean-surface cover for run habitat was highest in the months of March (17%) and May 2004 (19%), while the greatest mean-surface cover for riffle habitat was in May (8%) and July 2004 (7%). No trend in the mean-surface cover was apparent for either habitat after the start of road construction (Figure 16).

Large variation in percent-surface cover among sites made significant differences difficult to detect on individual sampling dates for both habitats; however, some-site characteristics were evident (Figure 17). For riffle habitat, site 4 displayed a significantly greater-mean-surface cover on July 2004, relative to all other sites on that date; however, no single site consistently exhibited a greater-surface cover throughout the study. For run habitat, no site exhibited a significantly greater-surface cover for any sampling date (Figure 17). However, sites 2, 3, and 4 tended to have the highest-mean-surface cover (compared to all other sites) in run habitat across sampling dates, while site 8 tended to have the lowest-mean-surface cover across sampling dates (Figure 17). In general, all sites exhibited similar-mean-surface cover for most sampling dates in riffle habitat; whereas, there was greater separation among sites in mean-surface cover across sampling dates in run habitat.

Table 8.—Mixed-model ANOVA table with habitat, time (i.e., sampling period), and habitat*time as the fixed effects for % surface cover data from all sites and habitats. The random effects in the model are site, site*habitat, and site nested within habitat/time. All data was rank transformed to meet the assumption of normality. The compound symmetry covariance structure was used due to the smallest Bayesian Information Criterion (BIC) value.

Fixed Effects	Num df	Den df	F value	Pr > F
habitat	1	5	15.57	0.0109
time	6	58	6.80	< 0.001
habitat*time	6	58	1.34	0.2532

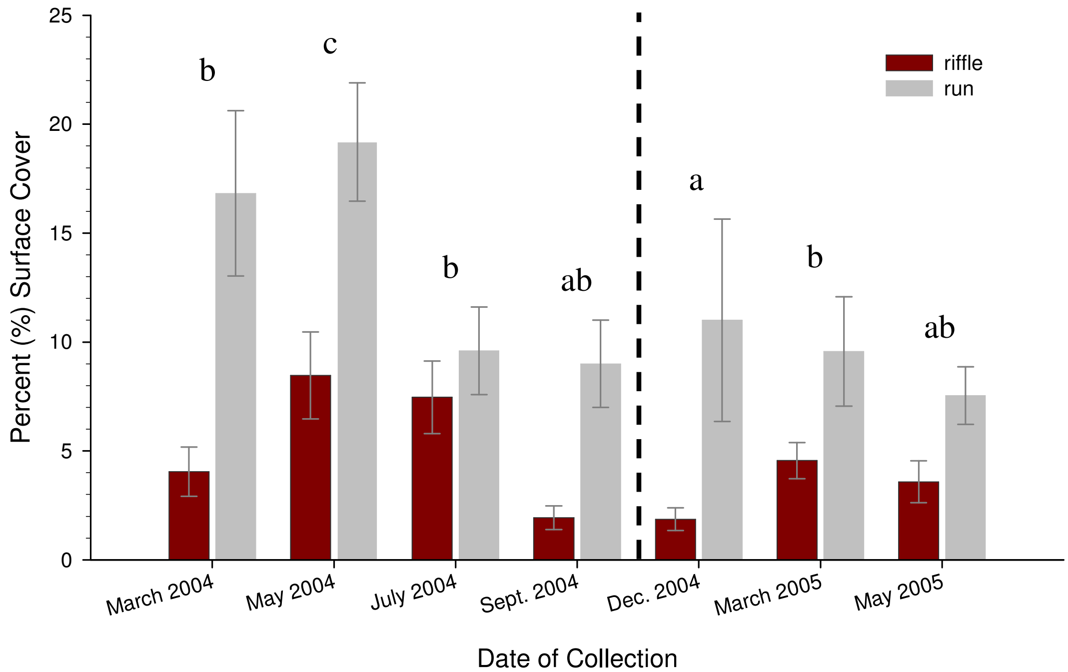


Figure 16.—Mean-percent-surface cover of fine sediments (± S.E.) for riffle and run habitats with data from all downstream sites pooled together across sampling periods. Least-squares-means comparisons confirm that all significant differences in habitat are represented by non-overlapping standard error bars (α = 0.05). Sampling periods with the same letter above are not statistically different (α = 0.05). The vertical, dashed line denotes the start of road construction.

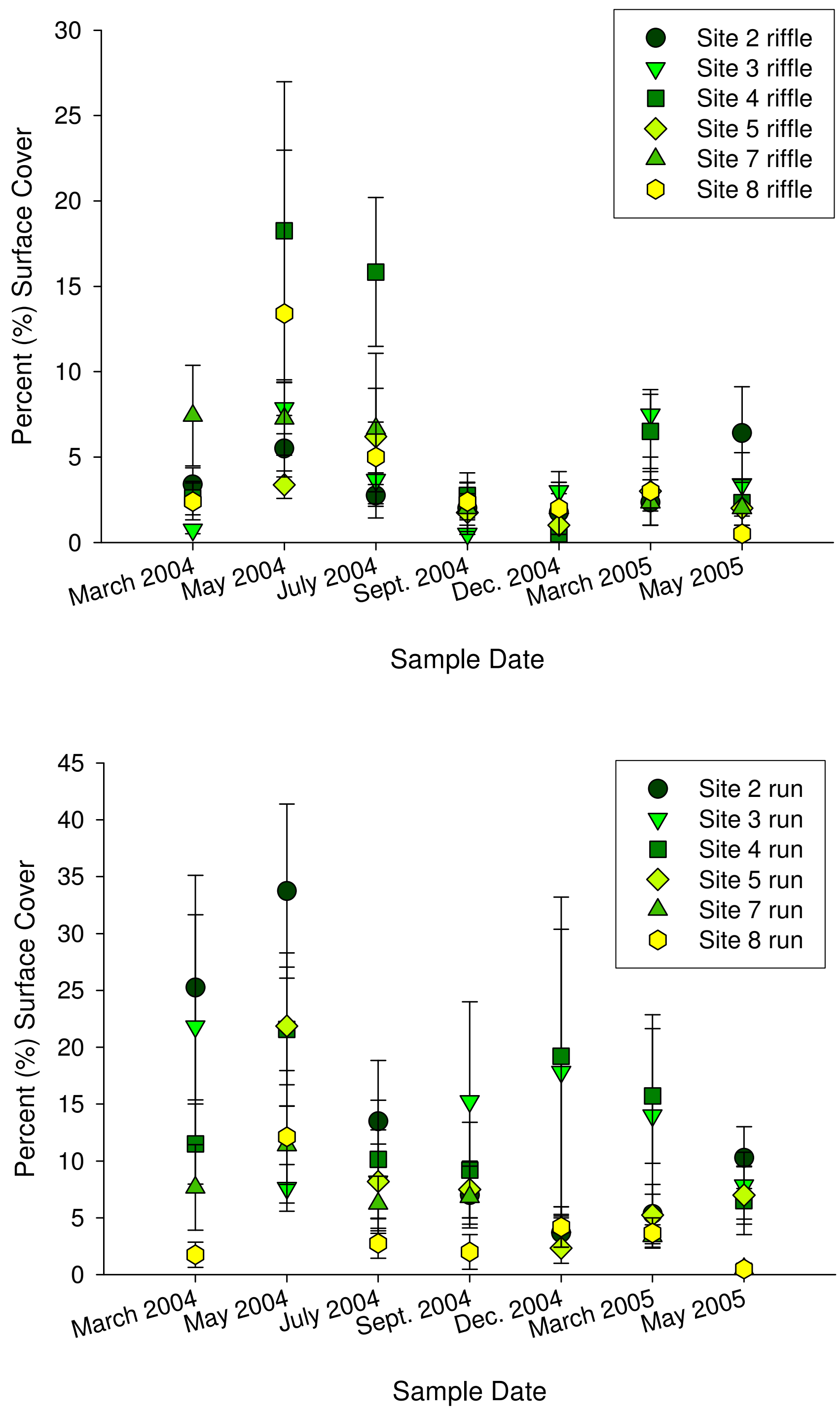


Figure 17.—Mean-percent-surface cover (± S.E.) of fine sediments for riffle (top) and run (bottom) habitats at each-study site across sampling periods (n = 2 to 8).

*Embeddedness Characteristics.—*Embeddedness was very low among sites and habitats with a rating of 1 (0-25%) comprising the greatest-total frequency for all data (Table 9). A rating of 4 (> 75%) only occurred at site 4 (four times) and site 7 (one time) on March 2004. No site consistently displayed an elevated-embeddedness rating throughout the study period.

Significant differences in embeddedness between habitats (Prob > ChiSq < 0.05) occurred on four occasions throughout the sampling period (Table 9). Three of those instances exhibited a higher-embeddedness rating for run habitat. Overall, both riffle and run habitats tended to have the same-average (mode) embeddedness ratings especially after May 2004.

A contrast was performed to test for an overall difference in embeddedness before (March 2004-September 2004) versus after (December 2004-May 2005) the start of road construction. Significant differences (Prob > ChiSq < 0.05) in this contrast were present at all sites except site 7. This is most likely attributed to the higher-average (mode) embeddedness ratings on March 2004 for most habitats at these sites since few differences existed thereafter (Table 9).

Table 9.—Summary statistics for embeddedness data across sites and sampling dates. The asterisk denotes a significant difference (Prob > ChiSq < 0.05) between habitats for a specific site and sampling period. A rating of 1 = 0-25%, 2 = 25-50%, 3 = 50-75%, and 4 = >75% embeddedness.

		Sampling Date													
		Mar-04		May-04		Jul-04		Sep-04		Dec-04		Mar-05		May-05	
Site	Habitat	Mode	Max	Mode	Max	Mode	Max	Mode	Max	Mode	Max	Mode	Max	Mode	Max
2	riffle	3	3	2	3	1	2	1	1	1	2	2	2	1	2
2	run	2	3	2	3	2	2	1	2	1	2	1	2	1	2
3	riffle	1	3	1	2	1	3	1	1	1	3	1	3	1	2
3	run	2	3	2 *	3	2	2	1	2	1	3	1	2	2	3
4	riffle	2	4	1	2	2 *	3	1	3	1	1	1	2	1	2
4	run	3	4	2	2	1	3	1	2	2 *	2	2	3	1	2
5	riffle	-		2	2	1	2	1	2	1	2	1	2	1	2
5	run	-		2	3	2	3	1	2	1	2	1	3	1	2
7	riffle	3	4	2	3	1	2	1	1	1	2	1	2	1	2
7	run	2	3	2	2	1	3	1	3	1	3	1	3	2	2
8	riffle	3	3	1	2	1	2	1	2	1	2	1	2	1	2
8	run	2	3	2	2	1	2	1	2	1	3	1	2	2 *	2

Correlations Between Physical Variables

<u>*Spearman Rank Correlations Among Physical Variables.*—</u>Deposited sediment dry weight was negatively associated with percent-coarse silt and positively associated with discharge, while suspended sediment (TSS) was not significantly correlated with any-physical variables (Table 10). A moderate-positive association was found, however, between TSS and deposited sediment dry weight at an alpha of 0.25. Surface cover and embeddedness had a strong-positive correlation, while surface cover and deposited sediment dry weight were moderately correlated ($p < 0.25$). Embeddedness was moderately correlated ($p < 0.25$) with deposited sediment dry weight and the percent of deposited sediment less than 2-mm in size. A high correlation with discharge was found with deposited sediment dry weight (positive correlation), percent-coarse silt (negative correlation), and embeddedness (positive correlation). The only significant correlation with precipitation (i.e., RAINMEAN) was found with discharge.

Table 10.—Spearman rank correlation coefficients, p-values, and sample size for environmental variables, all sites throughout study period (n = 34 to 37). P-values significant at α = 0.05 are bolded, while p-values significant at α = 0.25 are italicized.

	TSS	SURCOV	%<TWOMM	SILTPERC	EMBEDD	DISCHMEAN	DISCHMAX	RAINMEAN
DRYWTSED	0.22969 *0.1844* 35	0.33039 *0.0563* 34	-0.07944 0.6402 37	-0.45670 **0.0045** 37	0.27194 *0.1197* 34	0.63233 **<.0001** 34	0.60202 **0.0002** 34	0.14276 0.3993 37
TSS		-0.06903 0.6981 34	0.06776 0.6989 35	-0.15810 0.3644 35	0.15657 0.3765 34	0.11196 0.5284 34	0.17011 0.3361 34	0.16636 0.3395 35
SURCOV			0.10174 0.5670 34	0.04802 0.7874 34	0.46020 **0.0062** 34	0.23458 *0.1818* 34	0.19692 0.2643 34	-0.00288 0.9871 34
%<TWOMM				0.51978 **0.0010** 37	0.25238 *0.1499* 34	-0.01273 0.9430 34	-0.12718 0.4735 34	0.14799 0.3820 37
SILTPERC					-0.02802 0.8750 34	-0.47383 **0.0046** 34	-0.46884 **0.0052** 34	-0.08437 0.6196 37
EMBEDD						0.39930 **0.0193** 34	0.21164 *0.2295* 34	-0.12406 0.4845 34
DISCHMEAN							0.72296 **<.0001** 34	0.13727 0.4389 34
DISCHMAX								0.57396 **0.0004** 34

RESULTS

Macroinvertebrate Biomonitoring Metrics

<u>Biomonitoring Metrics for Long-term Data.—</u>Invertebrate metrics were calculated using the long-term data set (1998-2005) for Brush Creek from collections made in the spring (typically during the month of May) of each year and from the samples collected during this study (May 2004 to May 2005). The results will be discussed separately for each-time period.

A trend was not apparent after the start of road construction in biomonitoring metrics for the long-term period; however, significant differences between sample dates and habitats were found. The results of the two-way model III ANOVA (downstream sites pooled) showed significant differences between sample dates for total taxa richness (ANOVA; $p<0.0001$), EPT taxa richness (ANOVA; $p = 0.0006$), and Shannon diversity (ANOVA; $p = 0.0002$) (Figure 18). EPT richness was the only metric with significantly higher values in coarse-flow habitat (ANOVA; $p = 0.0011$) (Figure 18).

Total taxa richness and Shannon diversity were significantly higher in 2000 and lower in 2002 compared to all other sample years (Figure 18). EPT taxa richness was significantly higher in 2000 as well. EPT taxa richness and Shannon diversity in 2005 (i.e., during construction) were not significantly different from most-previous years. However, total taxa richness was significantly higher in 2005 compared to most-prior years. The results of a separate contrast to test for an overall effect before (1998-2004) versus during (2005) road construction confirmed no significant difference ($p > 0.05$) existed for any metric.

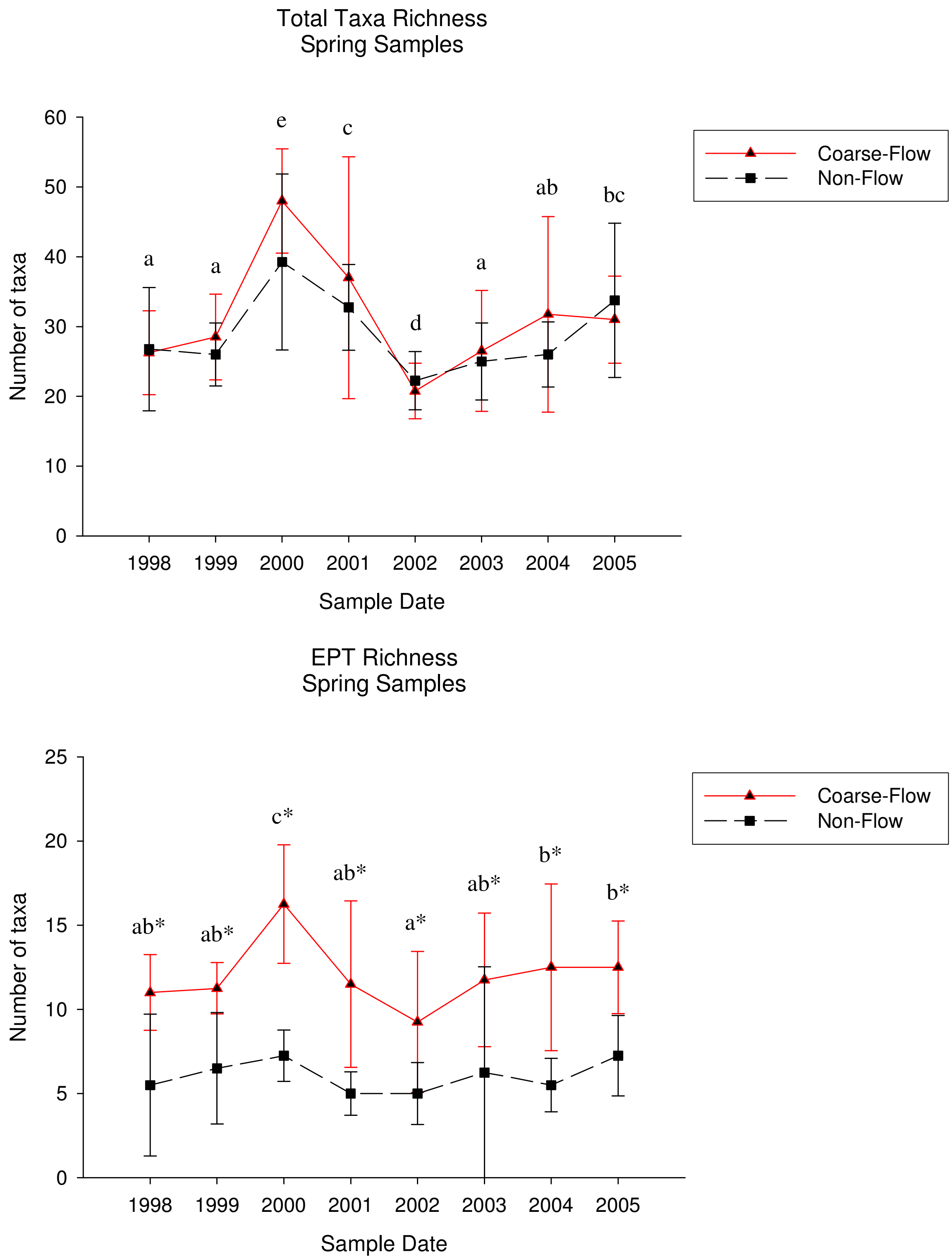


Figure 18.—The mean (± 95% C.I.) total taxa (top) and EPT taxa (bottom) richness with downstream sites pooled for the long-term study period. Sample dates with the same letter above are not statistically different. The asterisk notes a significant difference between habitats for an individual-sample date. An alpha level of 0.05 was used.

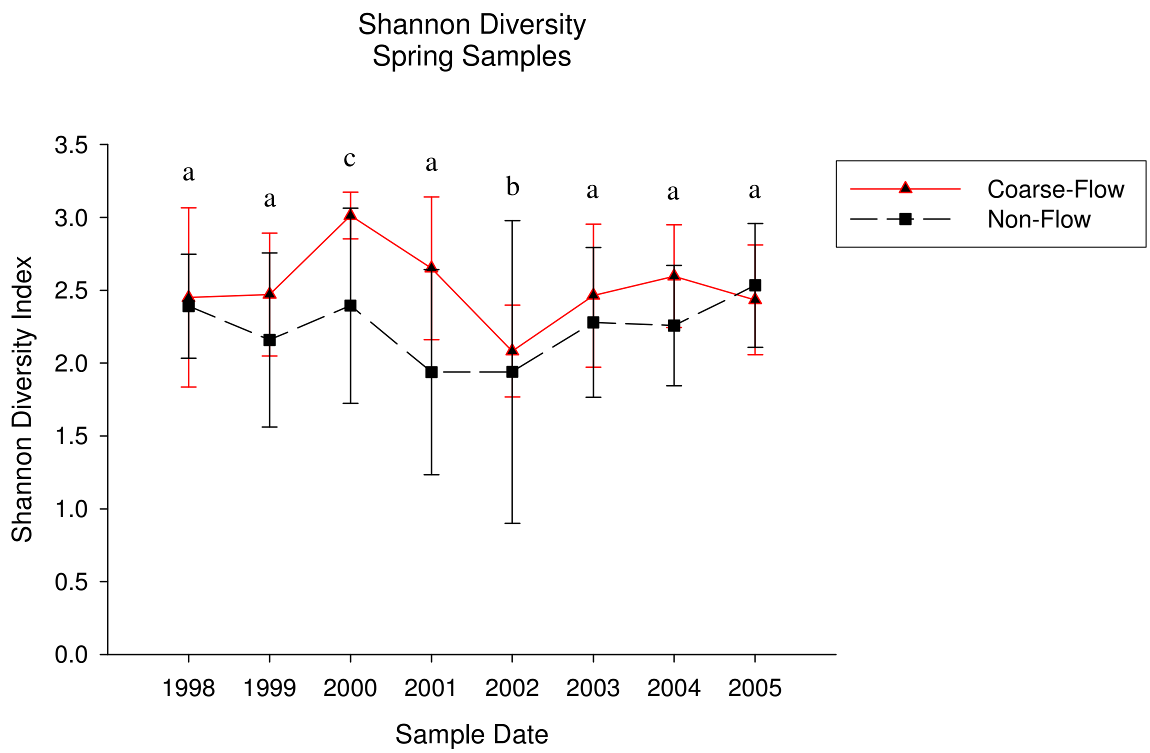
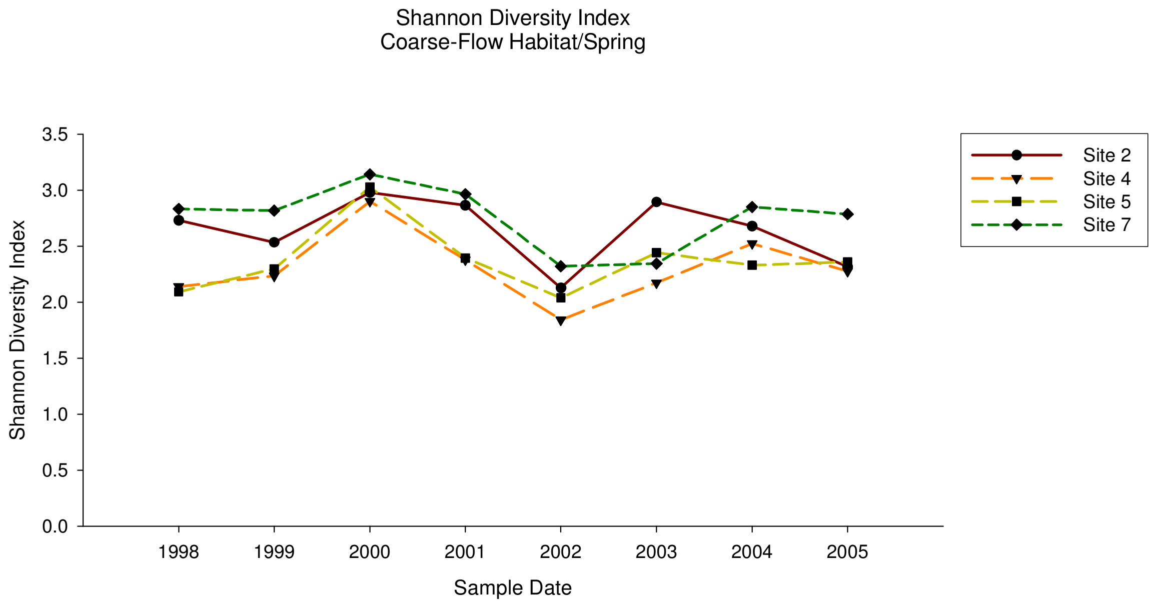


Figure 18 continued.—Mean (± 95% C.I.) for Shannon diversity with downstream sites pooled for the long-term study period. Sample dates with the same letter above are not statistically different.

Among Site Comparisons of Biomonitoring Metrics for Long-Term Data.— Several sites exhibited distinctive trends in macroinvertebrate metrics over time. Figure 19 displays the results of the Shannon diversity index for coarse-flow and non-flow habitats at each site with samples collected in the spring of each year from 1998-2005. In coarse-flow habitat, sites 2 and 7 consistently displayed the highest diversity across time with index values typically between 2.5 and 3.0 while site 4 consistently had the lowest diversity (Figure 19). In non-flow habitat, sites 2 and 4 frequently had the highest diversity and site 7 the lowest (Figure 19).

Study sites exhibited very similar trends in total taxa richness (Figure 20). In coarse-flow habitat, taxa richness was commonly the highest at sites 2 and 7 and lowest at site 4 (Figure 20). In non-flow habitat, the highest taxa richness was commonly at sites 2 and 4, while site 8 (upstream site) typically displayed the lowest taxa richness.

A greater disparity among sites was evident for EPT richness in both habitats (Figure 21). Site 2 frequently had the highest EPT index across years for both habitats with site 8 commonly displaying the lowest EPT index in non-flow habitat (Figure 21). Sites 2 and 5 exhibited the most stable (i.e., not highly fluctuating) EPT index in coarse-flow habitat whereas sites 4 and 7 exhibited a more erratic EPT index (Figure 21). In non-flow habitat, the EPT index was more stable among sites over time.



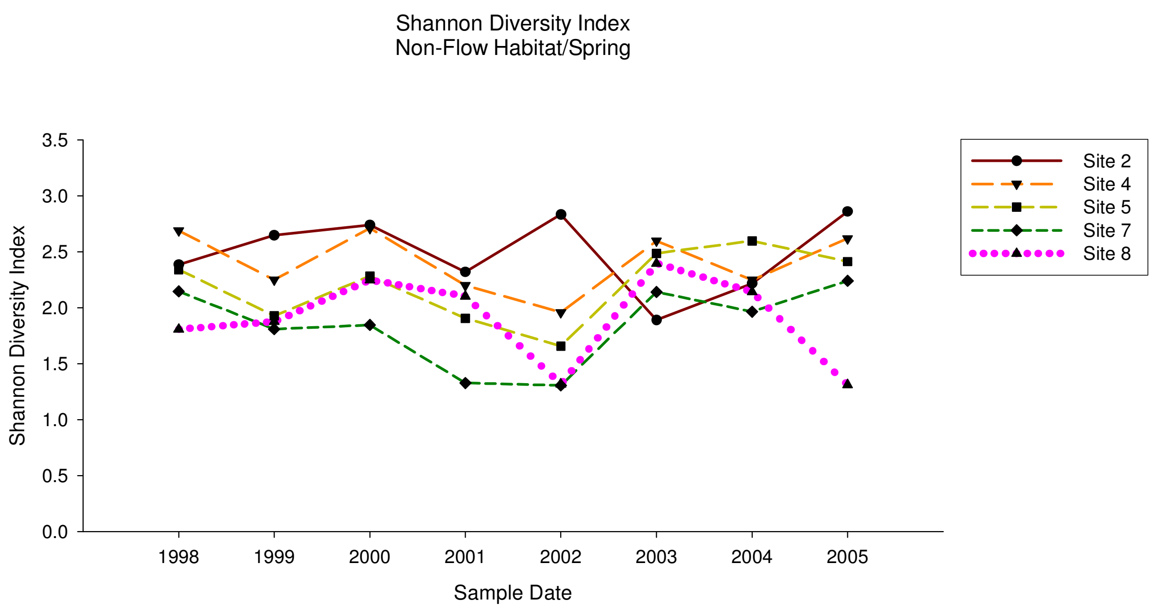


Figure 19.—Shannon's diversity index at each site for samples collected in the spring during the long-term study period (1998-2005) in coarse-flow (top) and non-flow (bottom) habitats.

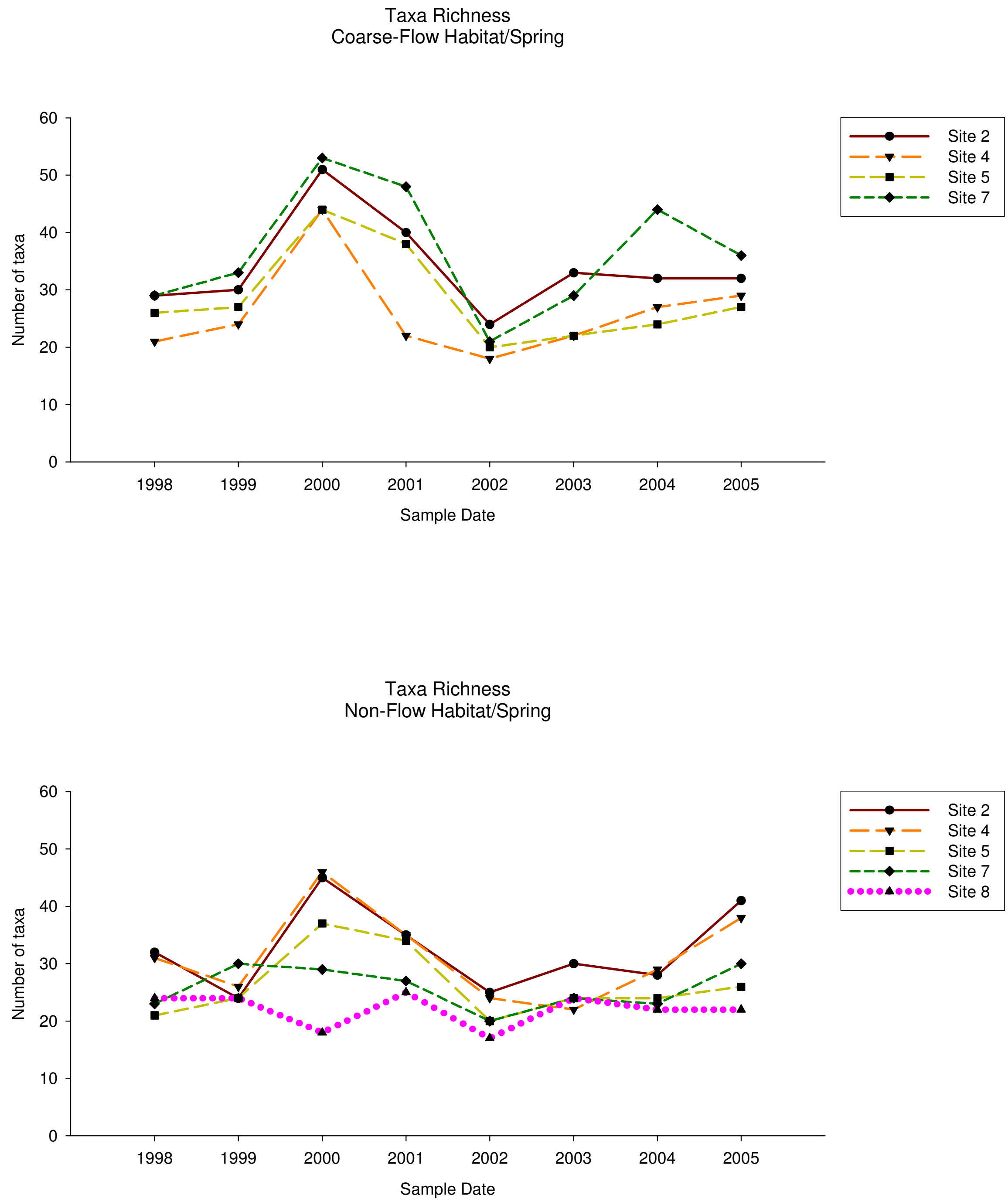
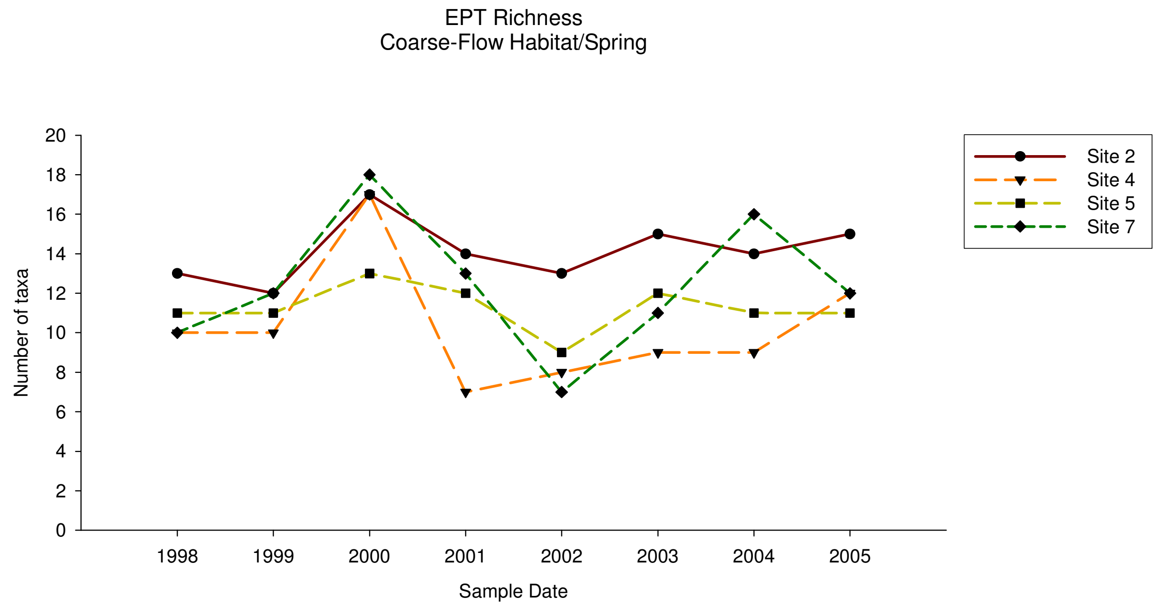


Figure 20.—Total taxa richness at each site for samples collected in the spring during the long-term study period (1998-2005) in coarse-flow (top) and non-flow (bottom) habitats.



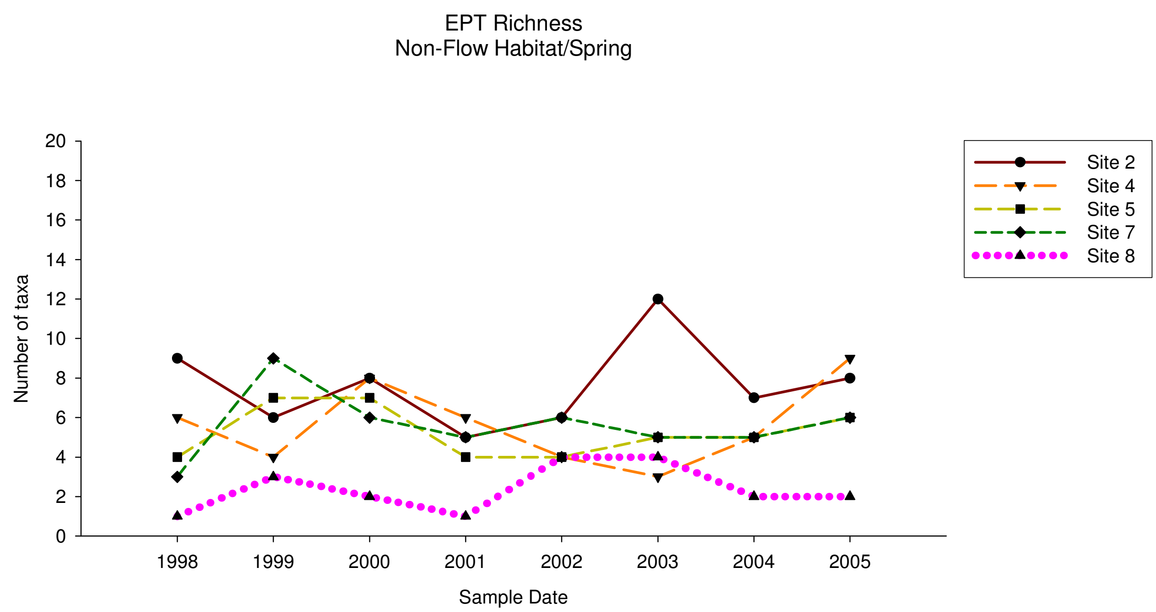
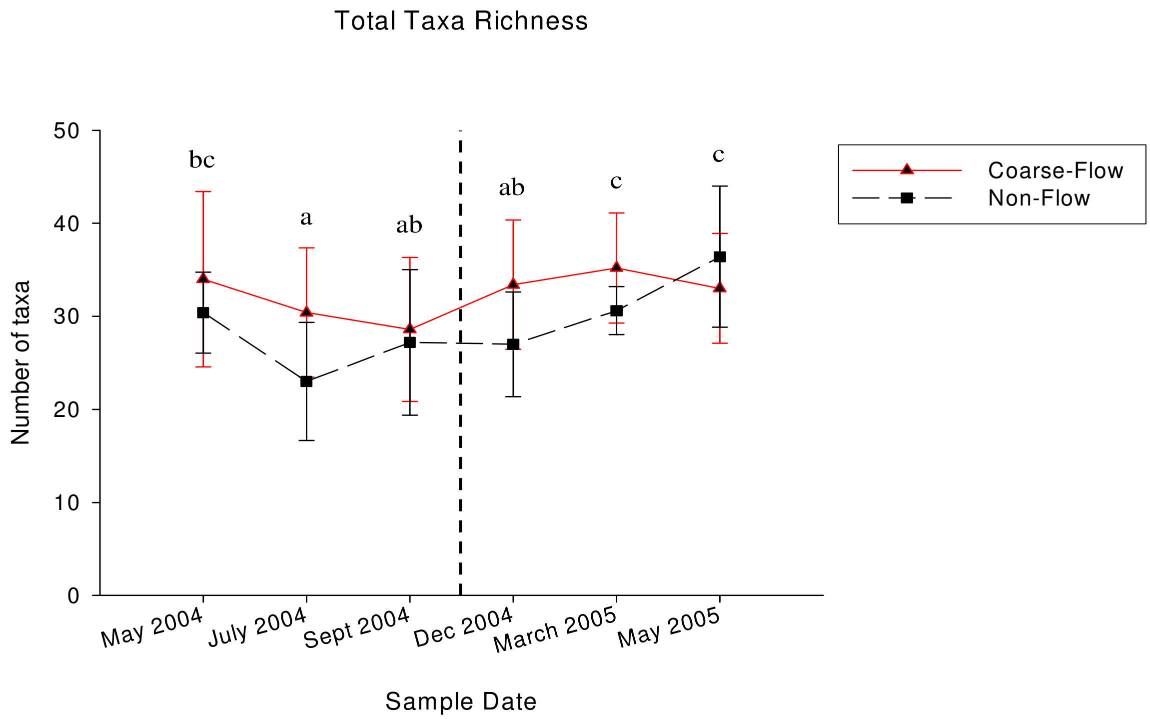


Figure 21.—EPT taxa richness at each site for samples collected in the spring during the long-term study period (1998-2005) in coarse-flow (top) and non-flow (bottom) habitats.

Biomonitoring Metrics for This Study Period.—Few trends were evident among biomonitoring metrics for this study period; however, significant differences between sample dates and habitats were found. Significant differences were found among sample dates (ANOVA; $p = 0.0033$) for total taxa richness (Figure 22). The EPT index had significantly higher values in coarse-flow habitat (ANOVA; $p = 0.0003$) for all but one sample date (December 2004), and also had significant differences in the interaction effect (i.e., time*habitat) (ANOVA; $p = 0.0185$) (Figure 22). No statistical differences existed with Shannon diversity for this study period.

After the start of road construction, total taxa and EPT taxa richness were not significantly different from most sampling dates prior to construction (Figure 22). The results of a separate contrast to test for an overall difference before versus during construction, however, showed a significant increase existed during construction for total taxa richness (ANOVA; $p = 0.0043$) and EPT taxa richness (ANOVA; $p = 0.0304$). The most apparent trend after the start of construction was the overall increase in total taxa richness for non-flow habitat, but this did not significantly exceed that of May 2004 (Figure 22). The overall increase in EPT richness during construction was much less pronounced.



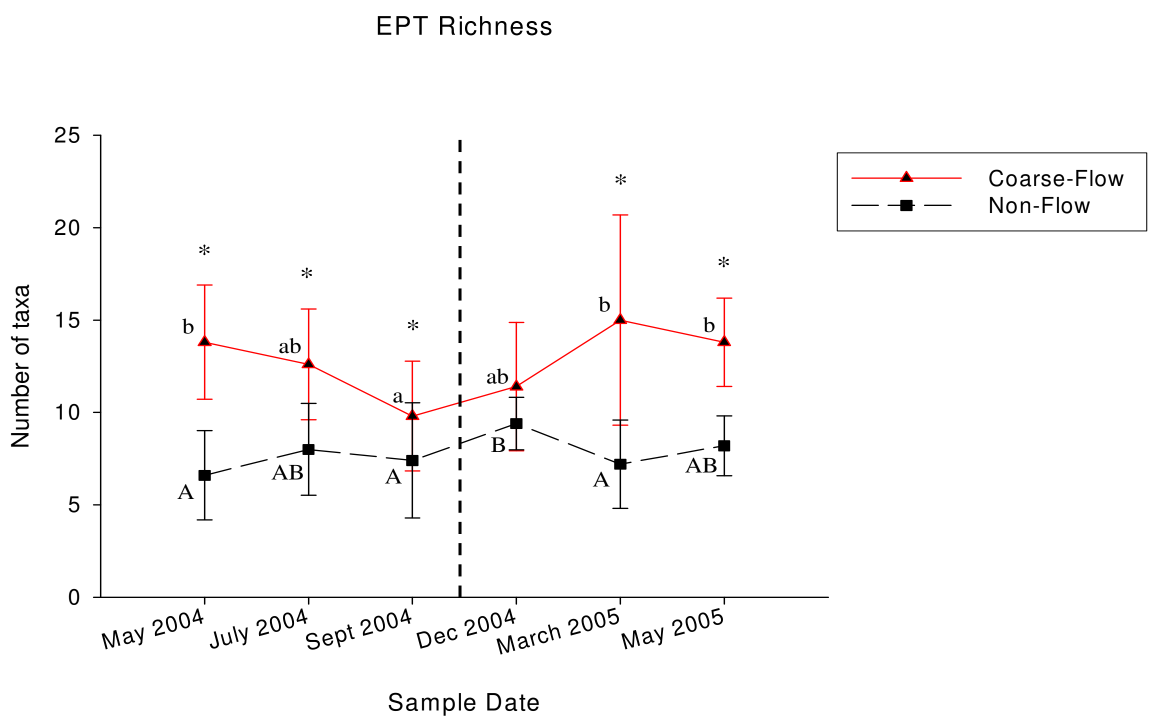


Figure 22.—The mean (± 95% C.I.) total taxa (top) and EPT taxa (bottom) richness with downstream sites pooled for this study period (May 2004 – May 2005). Sample dates and means associated with the same letter are not statistically different at $\alpha = 0.05$. The asterisk notes a significant difference between habitats for an individual sample date. The vertical-dashed line denotes the start of road construction.

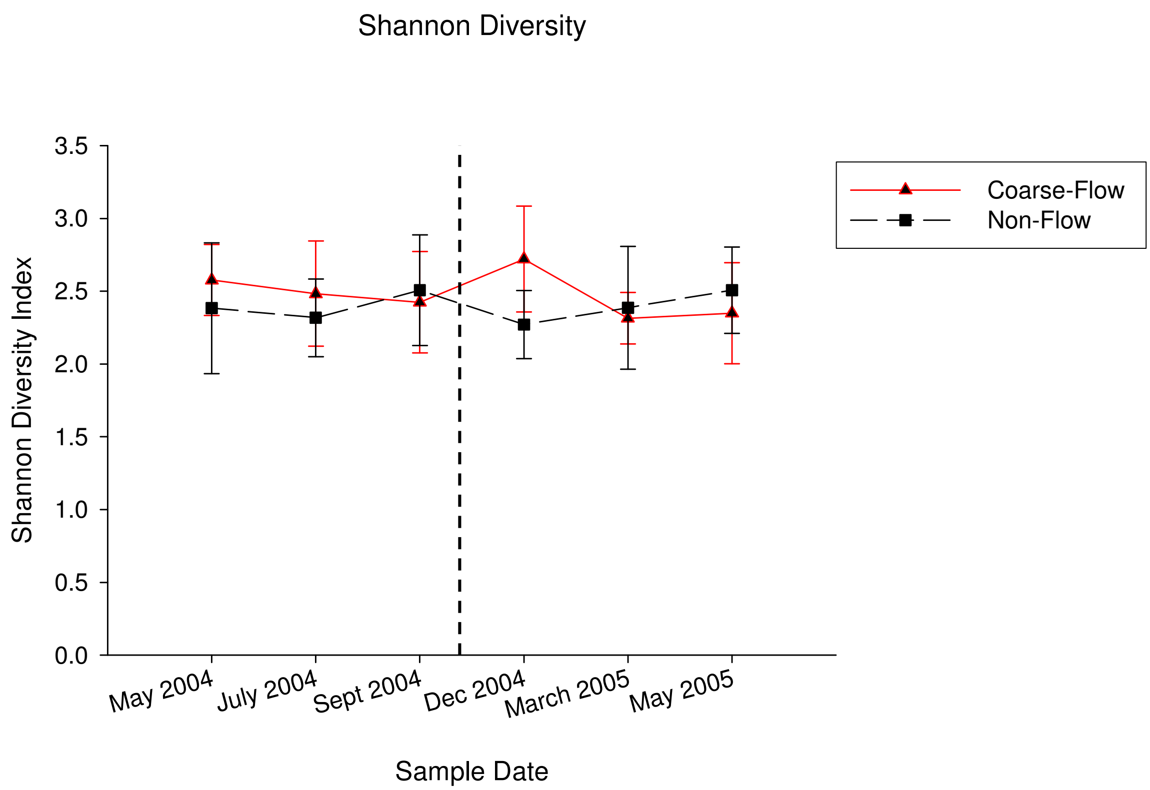
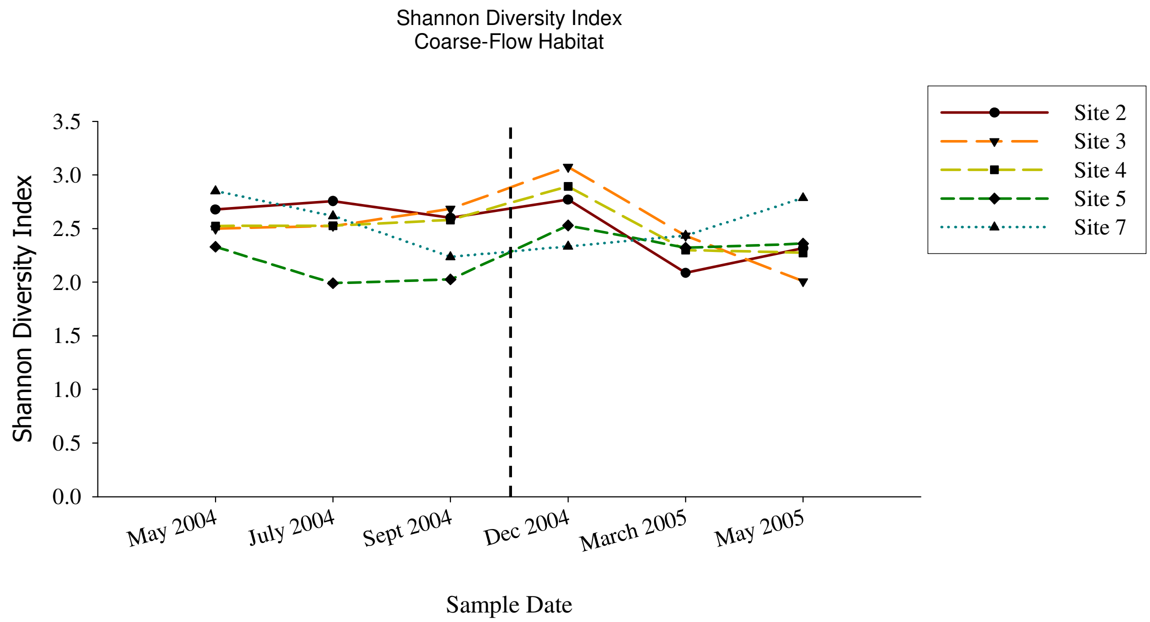


Figure 22 continued.—Mean (± 95% C.I.) for Shannon diversity with downstream sites pooled for this study period (May 2004 – May 2005). The vertical-dashed line denotes the start of road construction.

Among Site Comparison of Biomonitoring Metrics for This Study Period.—Few trends were evident in the response of invertebrate metrics at each site throughout the study period. There was little disparity among sites for Shannon's diversity index; however, prior to the start of construction, site 5 had the lowest diversity compared to all other sites in coarse-flow habitat (Figure 23). Site 8 displayed the lowest diversity in non-flow habitat throughout the majority of the study period (Figure 23).

Total taxa richness exhibited similar characteristics to the long-term data in non-flow habitat with the highest richness commonly exhibited at site 2, while site 8 had the lowest taxa richness throughout the study (Figure 24). Site 5 frequently had the lowest taxa richness in coarse-flow habitat. Site 2 also displayed the highest EPT index for most sample dates (especially after the start of road construction) in coarse-flow habitat, while site 8 had the lowest EPT index throughout the study in non-flow habitats (Figure 25).

The biotic index was calculated at downstream sites for samples collected between July 2004 and March 2005 (Figure 26). Minimal variability existed in biotic index values among sites with non-flow habitat consistently displaying higher values than coarse-flow habitat (Figure 26). For both habitats, the biotic index remained stable (i.e., no apparent trend) during this time period.



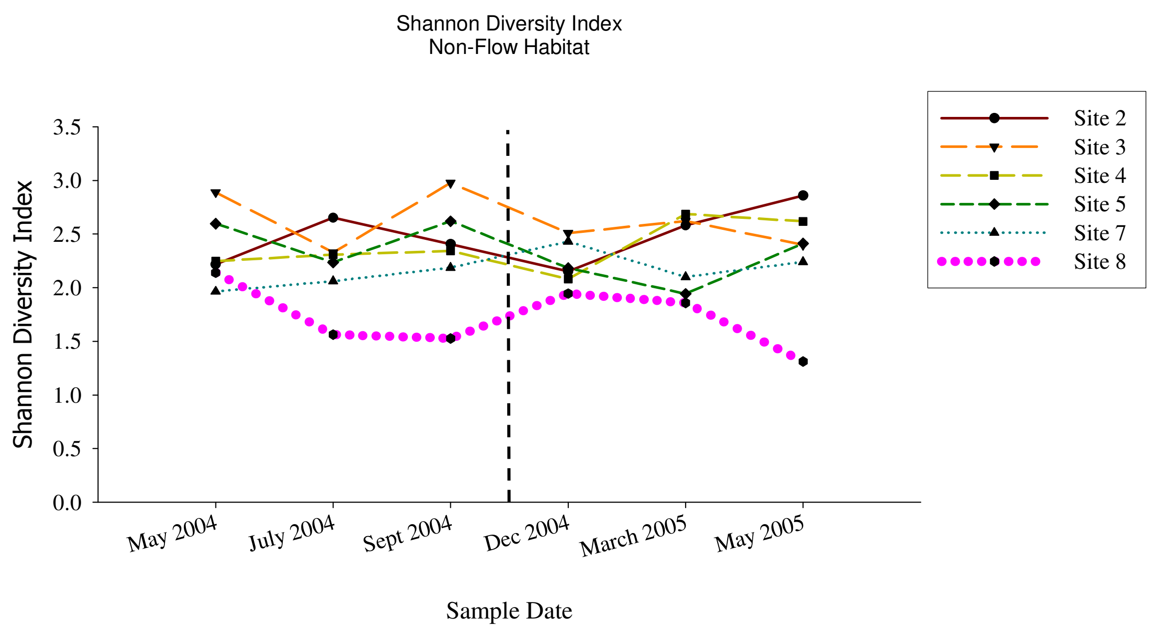
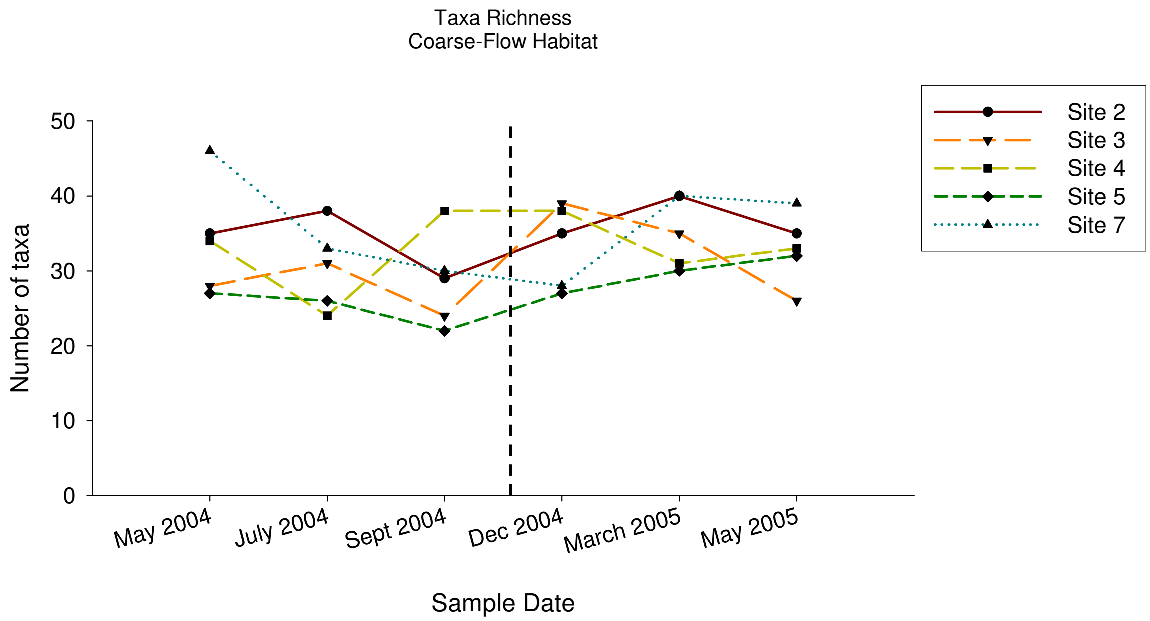


Figure 23.—Shannon's diversity index at each site throughout the study period (May 2004 – May 2005) for coarse-flow (top) and non-flow (bottom) habitats. The vertical-dashed line denotes the start of road construction.



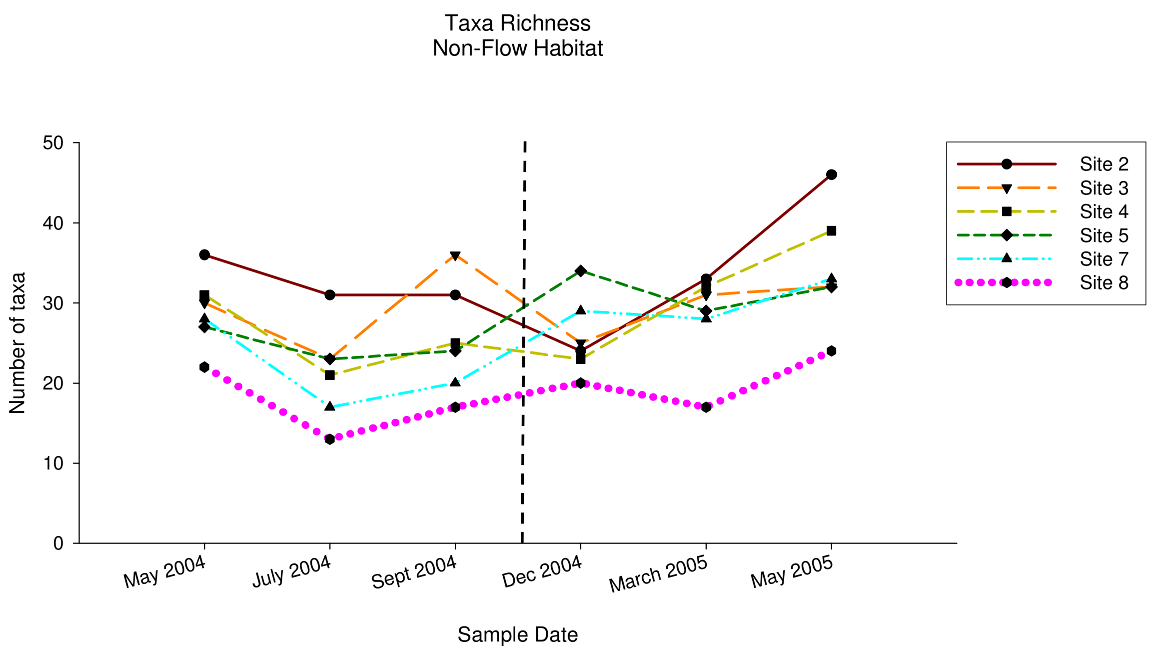
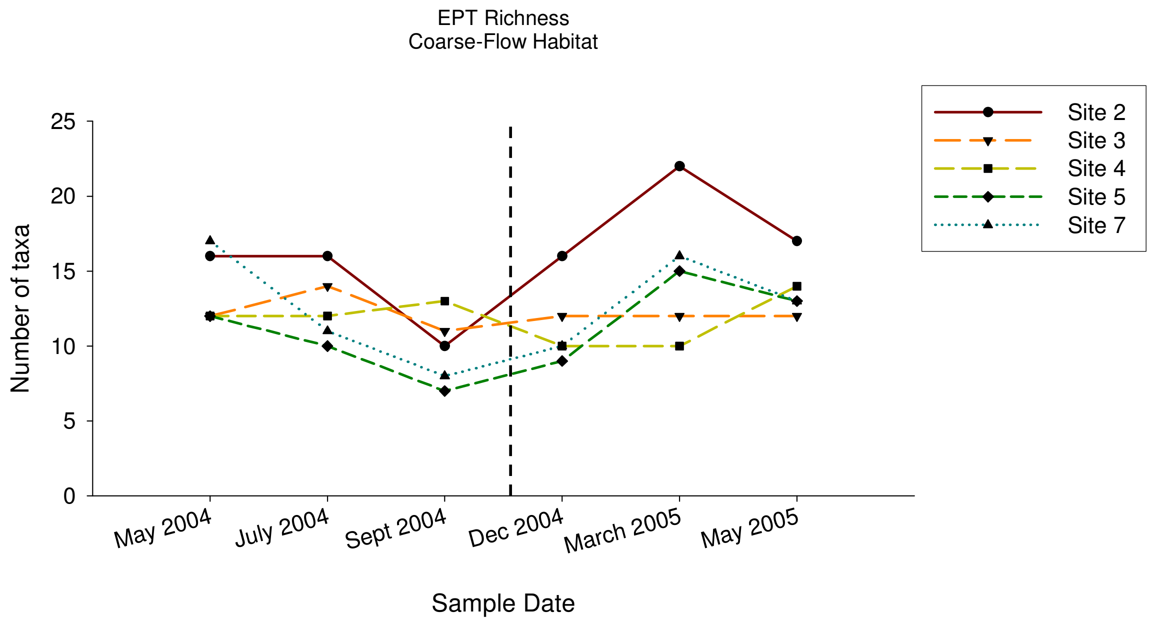


Figure 24.—Total taxa richness at each site throughout the study period (May 2004 – May 2005) for coarse-flow (top) and non-flow (bottom) habitats. The vertical-dashed line denotes the start of road construction.



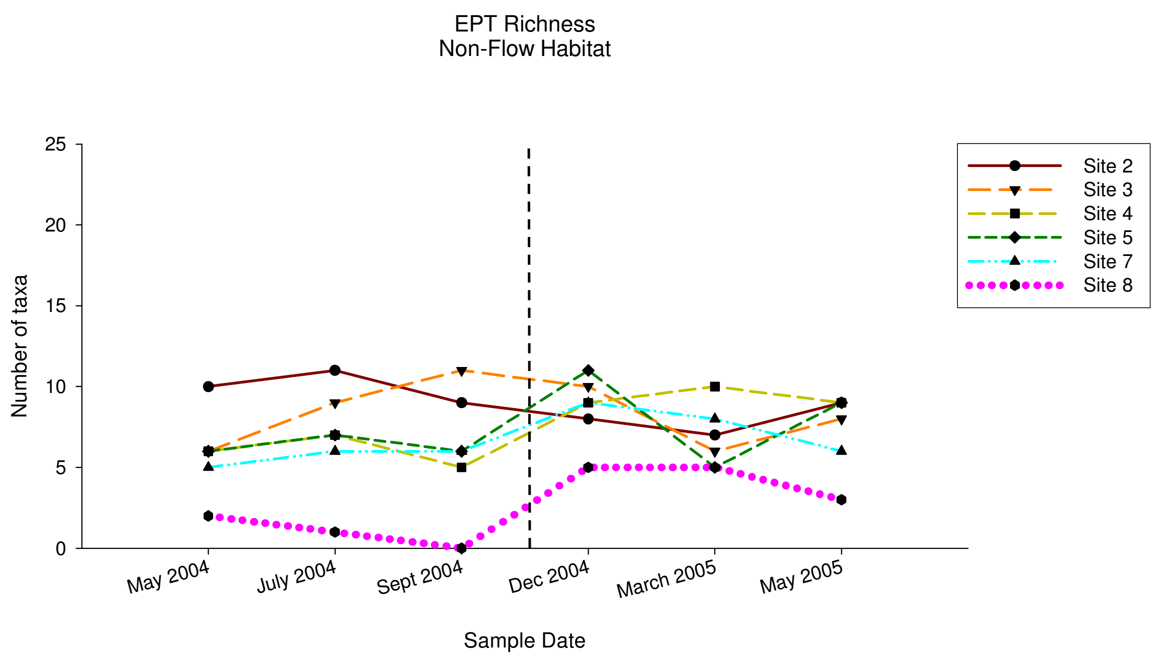
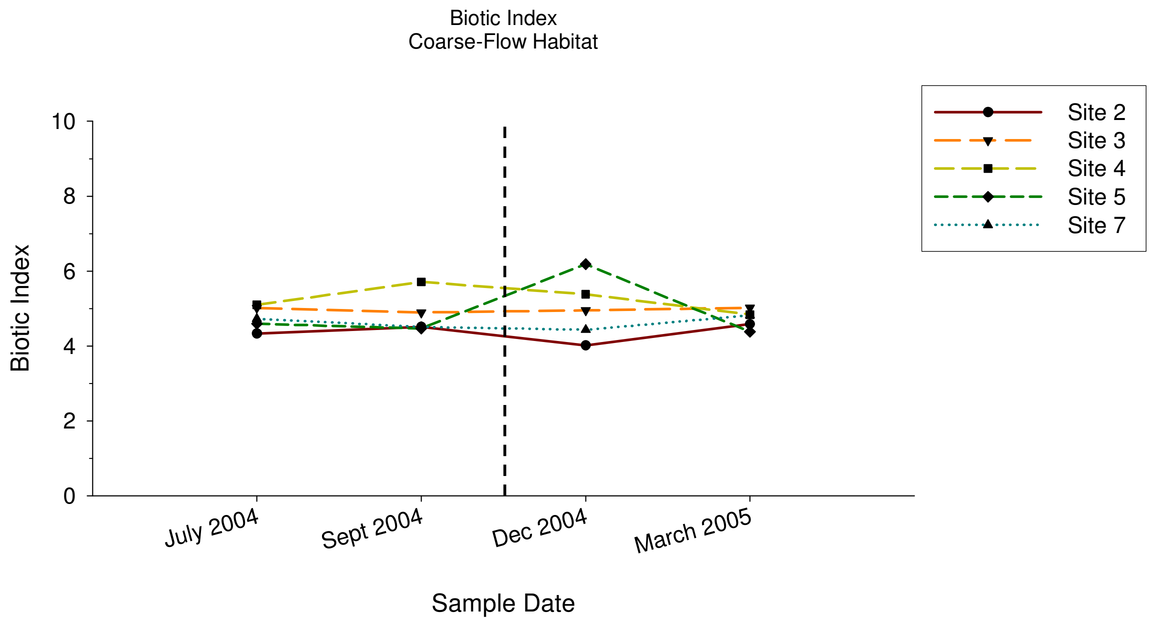


Figure 25.—EPT taxa richness at each site throughout the study period (May 2004 – May 2005) for coarse-flow (top) and non-flow (bottom) habitats. The vertical-dashed line denotes the start of road construction.



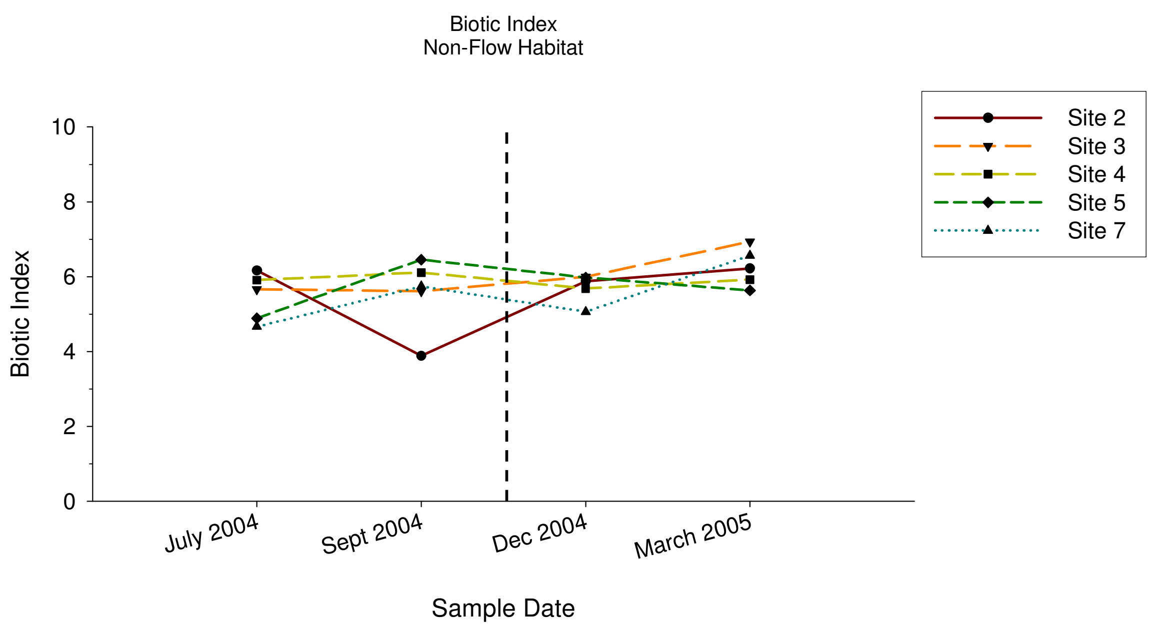


Figure 26.—The biotic index at each site throughout the study period (May 2004 – May 2005) for coarse-flow (top) and non-flow (bottom) habitats. The vertical-dashed line denotes the start of road construction.

Correlations with Macroinvertebrate Biomonitoring Metrics

Environmental and Biological Relations.—Biomonitoring metrics for coarse-flow habitat showed significant-positive relations existed between deposited sediment dry weight and both total and EPT taxa richness (Table 11). Total taxa and EPT taxa richness had significant-negative correlations with percent-coarse silt and mean precipitation, respectively. Total taxa richness was only moderately correlated ($p < 0.25$) with suspended sediment (TSS) and discharge, while the EPT index and embeddedness were also moderately correlated. Shannon diversity was not significantly (or moderately) correlated with any environmental variables; whereas, the biotic index (abbreviated BI) was only moderately correlated with percent-coarse silt and surface cover (Table 11).

The total taxa richness and EPT index in non-flow habitat had no significant correlations with the environmental variables (Table 12). Only moderately significant correlations ($p < 0.25$) were found with total and EPT taxa richness. Shannon diversity had a significant, positive correlation with surface cover, while the biotic index had a significant-positive correlation with suspended sediment and discharge (Table 12).

Table 11.—Spearman rank correlation coefficients, p-values, and sample size (n) of invertebrate metrics and environmental variables for samples collected throughout the study period in coarse-flow habitat (n = 20 or 29). Results significant at $\alpha = 0.05$ are in bold, while p-values significant at $\alpha = 0.25$ are italicized.

	TOTALTAXA	EPTTAXA	DIVERSITY	BI
DRYWTSED	**0.43688**	**0.38102**	0.14778	-0.15489
	0.0178	**0.0414**	0.4442	0.5144
	29	29	29	20
TSS	0.25966	0.17434	-0.08128	0.07068
	0.1737	0.3657	0.6751	0.7672
	29	29	29	20
SURCOVER	-0.04040	0.08982	0.08585	0.40557
	0.8352	0.6431	0.6579	*0.0760*
	29	29	29	20
%<TWOMM	-0.17364	-0.03714	0.15680	-0.02341
	0.3677	0.8483	0.4166	0.9220
	29	29	29	20
SILTPERC	**-0.43488**	-0.19085	0.05273	0.36220
	0.0184	0.3213	0.7859	*0.1166*
	29	29	29	20
EMBEDD	0.04158	0.23744	0.03689	0.17342
	0.8304	*0.2149*	0.8493	0.4647
	29	29	29	20
DISCHMEAN	0.25853	0.17813	-0.07121	-0.13833
	0.1757	0.3552	0.7136	0.5608
	29	29	29	20
DISCHMAX	0.14485	-0.04996	-0.06022	-0.13833
	0.4534	0.7969	0.7563	0.5608
	29	29	29	20
RAINMEAN	-0.12983	**-0.42286**	0.11319	-0.03264
	0.5021	**0.0223**	0.5588	0.8913
	29	29	29	20

Table 12.—Spearman rank correlation coefficients, p-values, and sample size (n) of invertebrate metrics and environmental variables for samples collected throughout the study period in non-flow habitat (n = 21 or 32). Results significant at α = 0.05 are in bold, while p-values significant at α = 0.25 are italicized.

	TOTALTAXA	EPTTAXA	DIVERSITY	BI
DRYWTSED	0.30896	0.30588	0.25733	0.32987
	0.0853	*0.0887*	0.1551	0.1442
	32	32	32	21
TSS	0.33284	0.04587	0.06672	**0.51558**
	0.0627	0.8031	0.7168	**0.0167**
	32	32	32	21
SURCOVER	0.04839	0.09766	**0.46826**	0.01859
	0.7925	0.5949	**0.0069**	0.9363
	32	32	32	21
%<TWOMM	0.28987	0.01157	0.25525	-0.11209
	0.1076	0.9499	*0.1586*	0.6286
	32	32	32	21
SILTPERC	0.02107	0.01420	0.16764	-0.21163
	0.9089	0.9385	0.3591	0.3571
	32	32	32	21
EMBEDD	0.09431	-0.31274	0.01882	0.16074
	0.6077	0.0814	0.9186	0.4864
	32	32	32	21
DISCHMEAN	0.23779	-0.22195	0.00316	**0.52781**
	0.1900	*0.2221*	0.9863	**0.0139**
	32	32	32	21
DISCHMAX	-0.07007	-0.17824	-0.03403	**0.52781**
	0.7031	0.3291	0.8533	**0.0139**
	32	32	32	21
RAINMEAN	-0.10194	0.09081	0.10432	0.12087
	0.5788	0.6211	0.5699	0.6017
	32	32	32	21

Macroinvertebrate Assemblage Composition

_Ordination Analysis and Associated Relations.—_The ordination of the invertebrate assemblage for May 2004 and 2005 in coarse-flow habitat (which represent pre- and during construction samples in the same season) showed a distinct separation of sample dates along each axis (Figure 27). Axis 1 showed there was between one-quarter and one-half species turnover (i.e., Beta diversity) from May 2004 to May 2005 in coarse-flow habitat (Figure 27). Axis 1 site scores were not significantly correlated ($p > 0.05$) with any of the biomonitoring metrics, whereas, axis 2 site scores were significantly correlated with total and EPT taxa richness (Table 13). Additionally, axis 1 site scores were significantly correlated with suspended sediment, surface cover, and discharge while a moderate association existed with embeddedness (Table 14).

The same ordination for samples collected in non-flow habitat showed a less exclusive separation of sample dates (pre- versus during construction) for all sites (Figure 28). Axis 1 site scores were significantly related to total taxa richness, EPT richness, and Shannon diversity (Table 15). Deposited sediment dry weight, surface cover, and TSS were also significantly correlated with Axis 1 site scores, while axis 2 was significantly related to percent-coarse silt, and precipitation (Table 16).

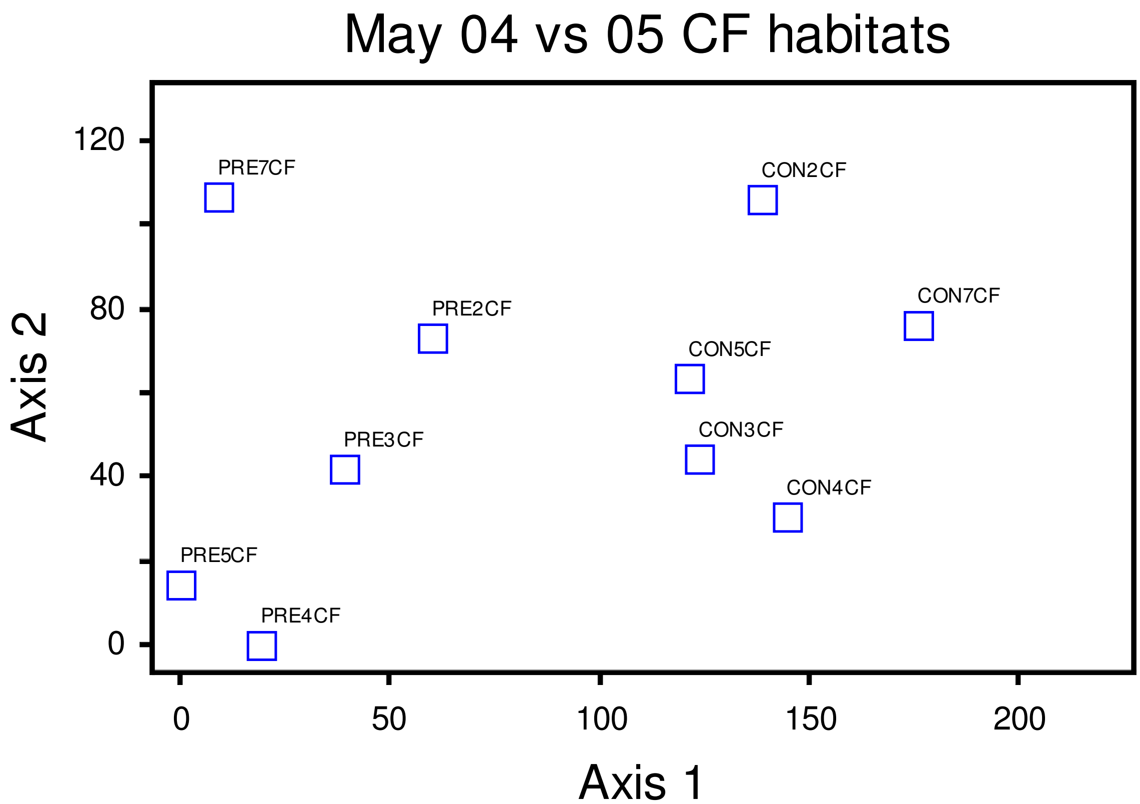


Figure 27.—Ordination of the May 2004 (labeled PRE) and May 2005 (labeled CON) invertebrate assemblages for coarse-flow habitat at all sites where coarse-flow habitat was sampled.

Table 13.—Spearman rank correlation coefficients and p-values for DCA axis scores and invertebrate metrics for samples collected in May 2004 and 2005 in coarse-flow habitat (n = 10). Results significant at α = 0.05 are in bold.

	AXIS1	AXIS2
TOTALTAXA	0.15806	**0.67478**
	0.6628	**0.0323**
EPTTAXA	0.26434	**0.78043**
	0.4605	**0.0077**
DIVERSITY	-0.29697	0.39394
	0.4047	0.2600

Table 14.—Spearman rank correlation coefficients and p-values for DCA axis scores and environmental variables for samples collected in May 2004 and 2005 in coarse-flow habitat (n = 9). Results significant at α = 0.05 are in bold, while p-values significant at α = 0.25 are italicized.

	AXIS1	AXIS2
DRYWTSED	-0.35000	0.15000
	0.3558	0.7001
TSS	**0.68333**	0.18333
	0.0424	0.6368
SURCOVER	**-0.89541**	-0.35984
	0.0011	0.3415
%<TWOMM	-0.35598	-0.05085
	0.3471	0.8966
SILTPERC	-0.00840	-0.41178
	0.9829	0.2708
EMBEDD	-0.45644	-0.18257
	0.2168	0.6382
DISCHMEAN	**-0.86603**	-0.34641
	0.0025	0.3611
DISCHMAX	**-0.86603**	-0.34641
	0.0025	0.3611
RAINMEAN	-0.86603	-0.34641
	0.0025	0.3611

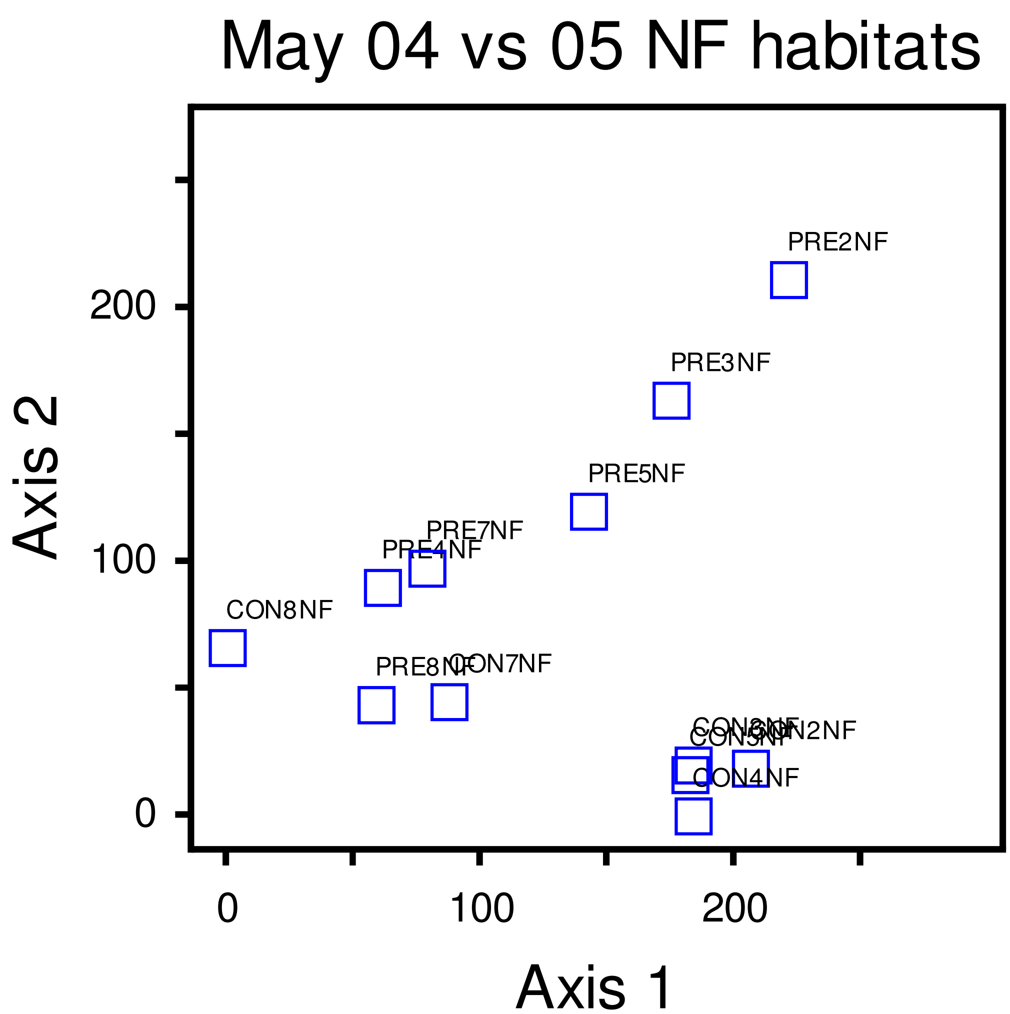


Figure 28.—Ordination of the May 2004 (labeled PRE) and May 2005 (labeled CON) invertebrate assemblages for non-flow habitat at all sites.

Table 15.—Spearman rank correlation coefficients and p-values for DCA axis scores and invertebrate metrics for samples collected in May 2004 and 2005 in non-flow habitat (n = 12). Results significant at α = 0.05 are in bold.

	AXIS1	AXIS2
TOTALTAXA	**0.83012**	-0.37128
	0.0008	0.2347
EPTTAXA	**0.93578**	-0.23305
	<.0001	0.4660
DIVERSITY	**0.59441**	-0.22378
	0.0415	0.4845

Table 16.—Spearman rank correlation coefficients and p-values for DCA axis scores and environmental variables for samples collected in May 2004 and 2005 in non-flow habitat (n = 11). Results significant at α = 0.05 are in bold, while p-values significant at α = 0.25 are italicized.

	AXIS1	AXIS2
DRYWTSED	**0.83012**	-0.37128
	0.0008	*0.2347*
TSS	**0.93578**	-0.23305
	<.0001	0.4660
SURCOVER	**0.59441**	-0.22378
	0.0415	0.4845
%<TWOMM	0.35455	0.30909
	0.2847	0.3550
SILTPERC	0.19091	**-0.68182**
	0.5739	**0.0208**
EMBEDD	-0.18679	0.48292
	0.5824	*0.1324*
DISCHMEAN	-0.34864	0.44039
	0.2934	*0.1752*
DISCHMAX	-0.05070	0.06452
	0.8823	0.8505
RAINMEAN	-0.35857	**0.65738**
	0.2789	**0.0279**

With both habitats included in the ordination, a distinct separation of habitats and sample dates was evident along each axis (Figure 29). A species turnover of approximately 75% was present with respect to Axis 1 (Figure 29). The EPT index was the only metric correlated with Axis 1, whereas no biomonitoring metric was correlated with Axis 2 (Table 17). No environmental variables were significantly related with Axis 1; however, Axis 2 was significantly related to TSS, surface cover, embeddedness, discharge, and precipitation (Table 18).

*Sediment Intolerant Taxa Response.—*The change in relative abundance of a few-dominant taxa considered to be sediment intolerant (Zweig 2000) was examined using the ordination for each habitat, separately. In coarse-flow habitat, *Stenonema femoratum* and *Isonychia* (both Ephemeroptera taxa), noticeably declined in abundance at downstream sites in May 2005 compared to May 2004. Similarly, there was a notable decline in the abundance of *Stenonema femoratum* in non-flow habitat at downstream sites in May 2005. *Isonychia* was not present in non-flow habitat for those sample dates.

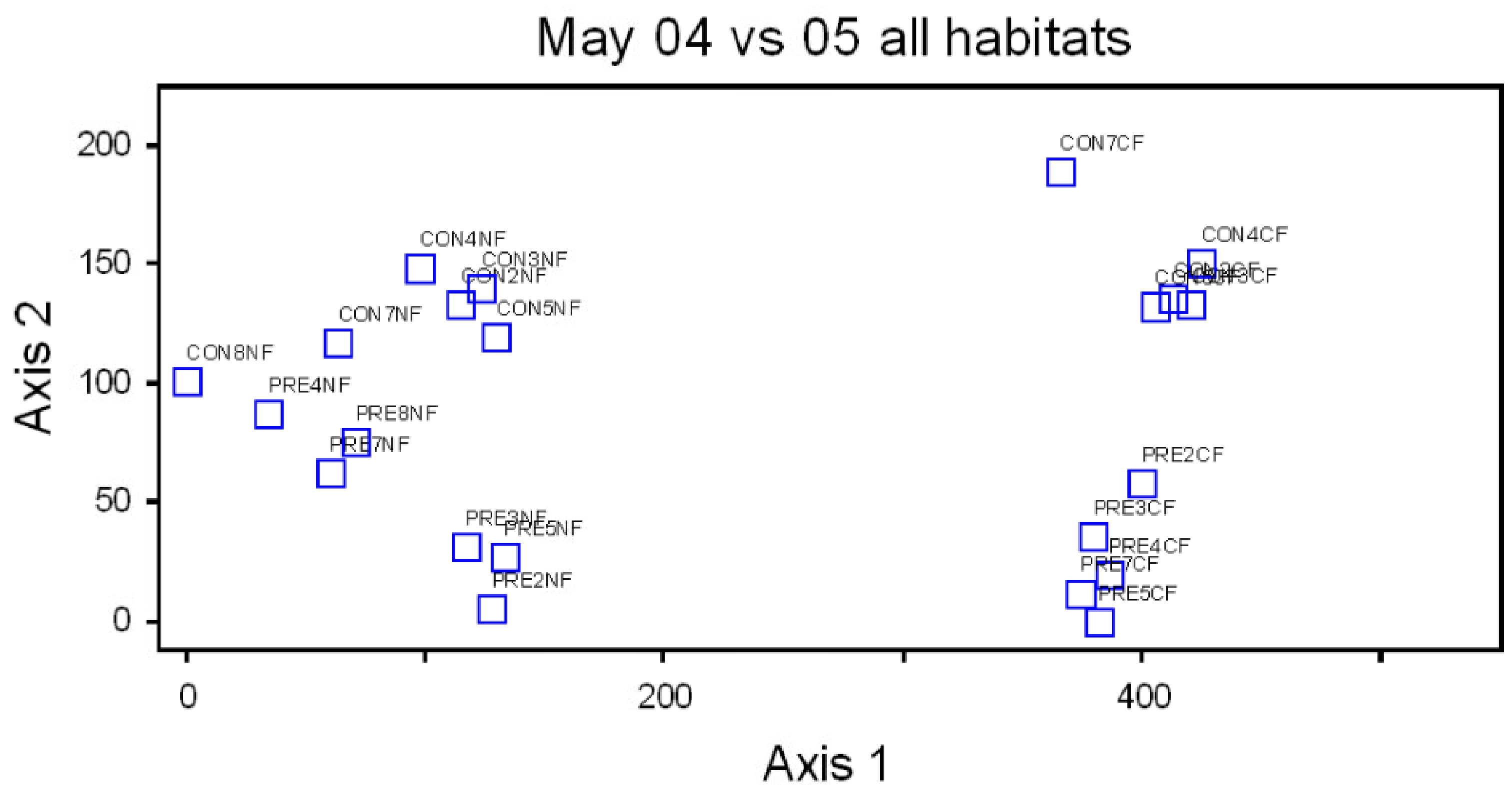


Figure 29.—Ordination of the May 2004 (labeled PRE) and May 2005 (labeled CON) invertebrate assemblages for coarse-flow and non-flow habitats at all sites.

Table 17.—Spearman rank correlation coefficients and p-values for DCA axis scores and invertebrate metrics for samples collected in May 2004 and 2005 in coarse-flow and non-flow habitats (n = 22). Results significant at α = 0.05 are in bold.

	AXIS1	AXIS2
TOTALTAXA	0.16931	0.21857
	0.4513	0.3284
EPTTAXA	**0.85440**	0.11206
	<.0001	0.6195
DIVERSITY	0.20045	-0.04574
	0.3711	0.8398

Table 18.—Spearman rank correlation coefficients and p-values for DCA axis scores and environmental variables for samples collected in May 2004 and 2005 in coarse-flow and non-flow habitats (n = 20). Results significant at α = 0.05 are in bold, while p-values significant at α = 0.25 are italicized.

	AXIS1	AXIS2
DRYWTSED	0.14938	-0.31083
	0.5296	*0.1822*
TSS	0.06186	**0.70163**
	0.7956	**0.0006**
SURCOVER	0.03178	**-0.74311**
	0.8942	**0.0002**
%<TWOMM	-0.18330	-0.38493
	0.4392	*0.0938*
SILTPERC	0.03738	-0.06103
	0.8757	0.7983
EMBEDD	-0.09999	**-0.51812**
	0.6749	**0.0193**
DISCHMEAN	-0.13072	**-0.86276**
	0.5828	**<.0001**
DISCHMAX	-0.13072	**-0.86276**
	0.5828	**<.0001**
RAINMEAN	-0.13072	**-0.86276**
	0.5828	**<.0001**

RESULTS

Benthic Fish Assemblage

Description of Benthic-fish assemblage.—Eight species representing a total of 3589 adult individuals were collected throughout the study period. Only young-of-year of *Phoxinus erythrogaster* and *Hypentelium nigricans* were captured and were not used in the final analysis. Of the eight species collected, *E. spectabile*, *E. caeruleum* and *E. flabellare* were dominant and comprised 12%, 22%, and 57% of the total individuals, respectively (Table 19). *E. punctulatum* and *Noturus exilis* comprised 4% and 3% of the total individuals, respectively, while all other species represented less than 1% of adult fishes collected. The majority of *Noturus exilis* collected were YOY and comprised 11% of the total individuals when YOY fishes were included in the data set. The overall mean species richness across sites, habitats and sample dates was 3.7 (S.D. 1.3). The overall mean density of adult fishes (regardless of species and habitat) collected throughout the study was 6.6/m^2 (Table 19).

Biomonitoring Metrics.—Results of a two-way mixed-model ANOVA showed significant differences in species richness occurred at the pooled downstream sites with regard to sample dates (ANOVA; $p = 0.0002$), habitats (ANOVA; $p = 0.0183$), and the interaction effect (ANOVA; $p = 0.0018$). Species richness was lowest for both habitats on March 2004 and was significantly higher in run habitat on three occasions (Figure 30). No trend was evident after the start of road construction as confirmed by an individual contrast that found no overall difference existed before versus during construction (ANOVA; $p = 0.6488$).

Table 19.—Number of adult fishes captured in each habitat at every site throughout the study period.

		Study Site and Habitat											
		2		**3**		**4**		**5**		**7**		**8**	
Sample Date	Species	riffle	run	riffle	run	riffle	run	riffle	run	riffle	run	riffle	run
Mar-04	*E. spectabile*			22	27								
	E. caeruleum	3	5	9	3	37	17			42	6	1	3
	E. flabellare	6	7	6	2	6	8			5	15		
	Noturus exilis	2								1			
	Total number of fishes	11	12	37	32	43	25			48	21	1	3
	Number of samples	6	6	5	5	6	6			6	6	1	1
	Density (no./m^2)	1.83	2	7.4	6.4	7.17	4.17			8	3.5	1	3
May-04	*E. spectabile*		1	10	22	3	20		15	1	2	1	15
	E. caeruleum	5	6	5	23	14	22	1	12	7	13	6	2
	E. flabellare	16	7	12	9	39	2	33	7	27	22	37	11
	E. blennoides				1		1						
	E. punctulatum								2	2	7	2	4
	E. zonale									1			
	Noturus exilis	5	4	6	5				1	2	9	3	2
	Total number of fishes	26	18	33	60	56	45	34	37	40	53	49	34
	Number of samples	6	6	8	8	8	8	8	8	8	8	6	5
	Density (no./m^2)	4.33	3	4.13	7.5	7	5.63	4.25	4.63	5	6.63	8.17	6.8
Jul-04	*E. spectabile*		3		1		16		3		8	2	4
	E. caeruleum	6	6	7	12	12	15		7	6	11	3	3
	E. flabellare	28	13	15	8	46	8	28	11	26	25	27	10
	E. blennoides		1		2				1		2		10
	E. punctulatum						1		3	2	15		
	Noturus exilis	1	2	1		1	4	1	1	2	2	1	2
	Cottus carolinae								1				
	Total number of fishes	35	25	23	23	59	44	29	27	36	63	33	29
	Number of samples	6	6	8	8	8	8	8	8	8	8	6	6
	Density (no./m^2)	5.83	4.17	2.88	2.88	7.38	5.5	3.63	3.38	4.5	7.88	5.5	4.83
Sep-04	*E. spectabile*	1	3		6	1	25		13		1		5
	E. caeruleum	7	22		18	31	21		4	5	11	3	6
	E. flabellare	10	10	9	27	176	33		8	48	31	16	7
	E. blennoides		1		1								
	E. punctulatum						1			5	12	2	14
	E. zonale		1						1				
	Noturus exilis	1			4	1	4		2		8	2	4
	Cottus carolinae		1										
	Total number of fishes	19	38	9	56	209	84		28	58	63	23	36
	Number of samples	3	7	1	8	8	7		8	8	8	2	4
	Density (no./m^2)	6.33	5.43	9	7	26.1	12		3.5	7.25	7.88	11.5	9

Table 19 continued.—

		Study Site and Habitat											
		2		**3**		**4**		**5**		**7**		**8**	
Sample Date	Species	riffle	run	riffle	run	riffle	run	riffle	run	riffle	run	riffle	run
Dec-04	*E. spectabile*			1	1	5	1	1	6	3	7	1	12
	E. caeruleum	2	11	2	11	1	5	1		5	6		7
	E. flabellare	9	13	6	11	23	8	8	14	22	10	27	10
	E. blennoides	1	1	1	1		1	2					
	E. punctulatum			1						1			
	Total number of fishes	12	25	11	24	29	15	12	20	31	23	28	29
	Number of samples	6	6	6	7	7	7	8	8	7	7	5	4
	Density (no./m^2)	2	4.17	1.83	3.43	4.14	2.14	1.5	2.5	4.43	3.29	5.6	7.25
Mar-05	*E. spectabile*	1	4	10	9	1	10	6	6	16	45	3	18
	E. caeruleum	11	4	12	13	16	10	4		9	7	12	2
	E. flabellare	38	13	27	8	43	10	24	3	49	50	60	29
	E. blennoides	1											
	E. punctulatum										1		
	E. zonale		1										
	Noturus exilis		1										
	Total number of fishes	51	23	49	30	60	30	34	9	74	103	75	49
	Number of samples	8	8	8	7	8	8	8	8	8	8	5	5
	Density (no./m^2)	6.38	2.88	6.13	4.29	7.5	3.75	4.25	1.13	9.25	12.9	15	9.8
May-05	*E. spectabile*	1	1		6		22			5	13		
	E. caeruleum	5	9	5	53	33	36			25	21	3	4
	E. flabellare	34	24	40	31	121	71	48	19	90	62	69	39
	E. blennoides	2	2		1		3	1	1		1		
	E. punctulatum					1	1		5	10	22	5	12
	E. zonale	2	3										
	Noturus exilis	3	1	1	1			1	1	2	4		
	Total number of fishes	47	40	46	92	155	133	50	26	132	123	77	55
	Number of samples	5	8	7	8	8	8	6	6	8	8	6	5
	Density (no./m^2)	9.4	5	6.57	11.5	19.4	16.6	8.33	4.33	16.5	15.4	12.8	11

The Shannon diversity index also exhibited significant differences with respect to sample date (ANOVA; p = 0.0007), habitat (ANOVA; p = 0.0065), and the interaction effect (ANOVA; p = 0.0004). Diversity was significantly higher in run habitat on four occasions (Figure 30). A contrast confirmed no significant difference in overall diversity occurred before versus during construction (ANOVA; p = 0.4744).

*Site Characteristics of Biomonitoring Metrics.—*Species richness ranged from zero to six throughout the study period in riffle habitat and exhibited few distinct characteristics among sites (Figure 31). Sites 2 and 7 were the only sites that had at least three species collected in riffle habitat throughout the study. Site 5 displayed the lowest richness of all sites before road construction started (Figure 31).

Species richness in run habitat was less variable among sites prior to construction; however, greater disparity between sites existed after the start of road construction (Figure 31). After construction started, sites 5 and 8 consistently displayed the lowest richness, while sites 2 and 7 had a noticeably greater richness in March and May 2005 relative to all other sites.

Shannon diversity was highly variable among sites in riffle habitat throughout the study period (Figure 32). Site 5 consistently displayed the lowest diversity in riffle habitat throughout the study, while sites 2 and 3 displayed the highest diversity for several sample dates in riffle habitat (Figure 32). Site 8 (upstream) exhibited a diversity in the middle range of most sites throughout the study period in riffle habitat.

Diversity was less disparate among sites in run habitat throughout the study (Figure 32). Site 5 often had the lowest diversity especially after the start of construction. No site consistently exhibited the highest diversity throughout the study (Figure 32).

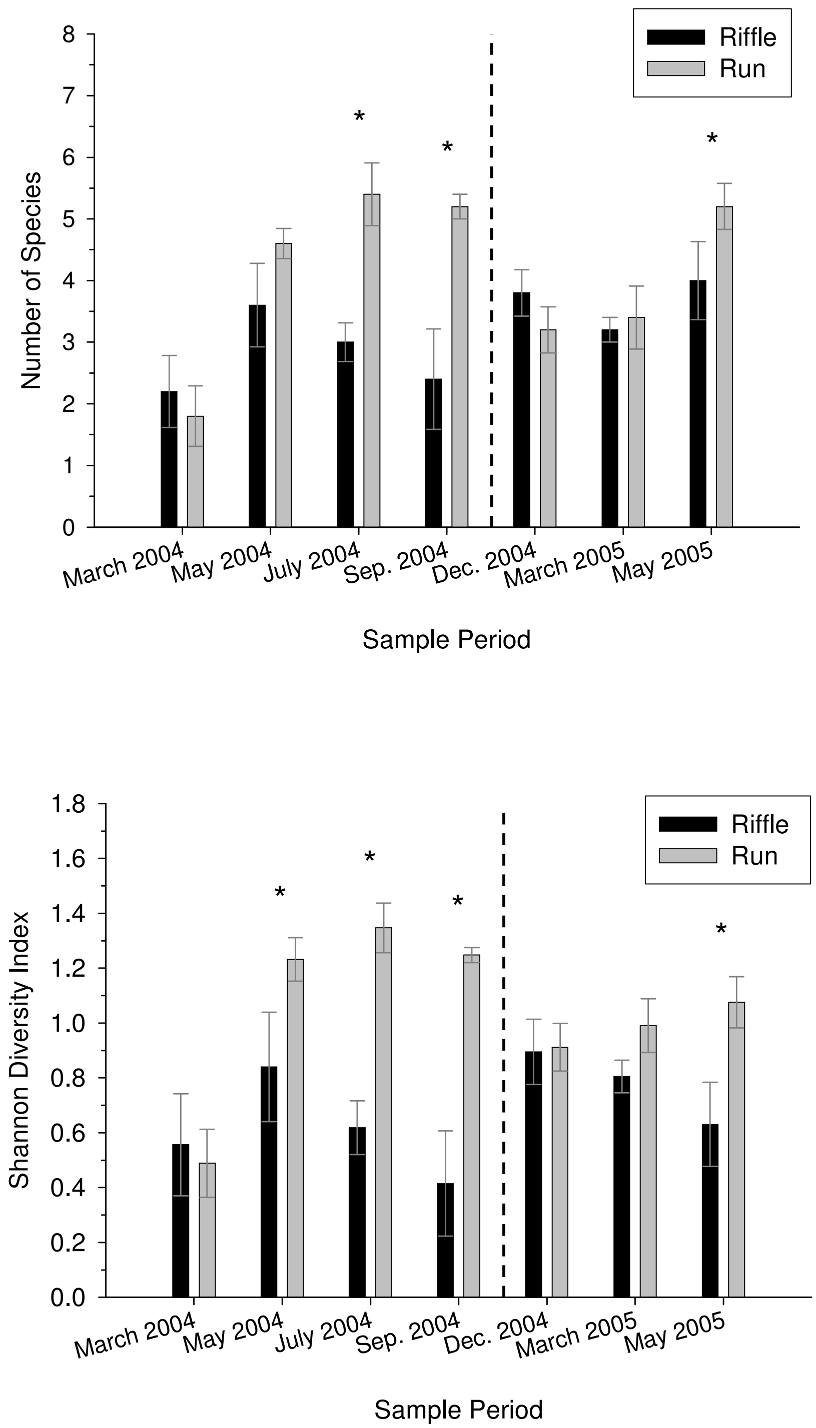
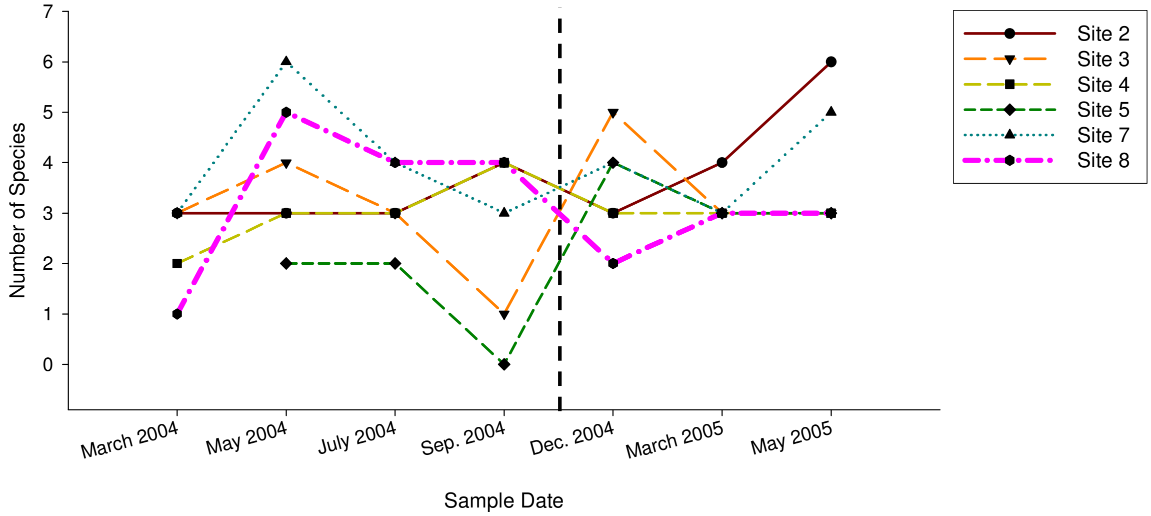


Figure 30.—Mean (± S.E.) species richness (top) and Shannon diversity (bottom) for adult fishes throughout the study period with downstream sites pooled. The asterisk notes a significant difference between habitats for an individual sample date. Comparison tests of least-squares means confirm all significant differences in time*habitat are represented by non-overlapping standard error bars. An alpha of 0.05 was used and the vertical-dashed line denotes the start of road construction.



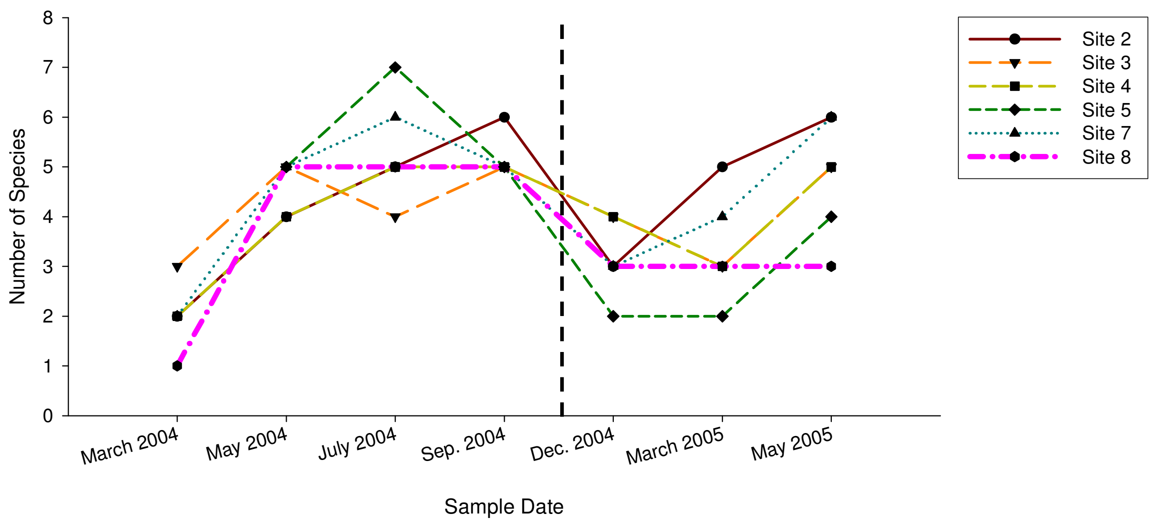
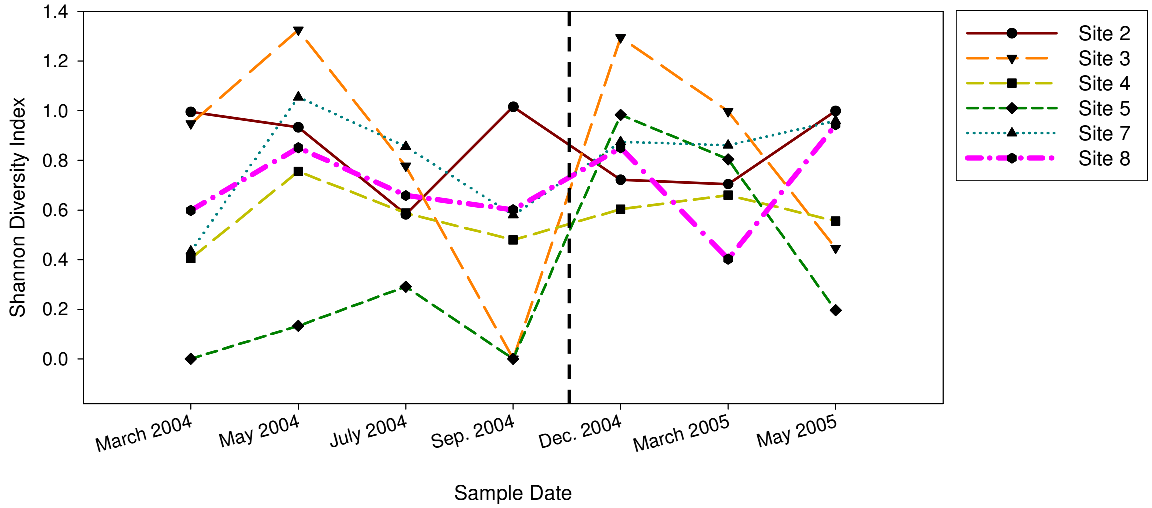


Figure 31.—Species richness for adult fishes at each site throughout the study period for riffle (top) and run (bottom) habitats. The vertical-dashed line denotes the start of road construction.



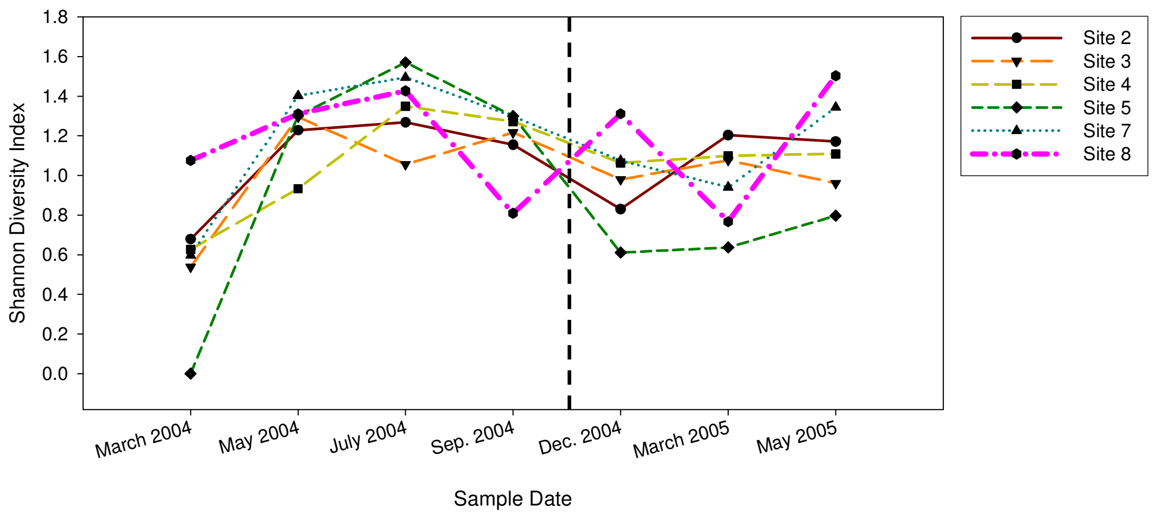


Figure 32.—Shannon diversity for adult fishes at each site throughout the study period for riffle (top) and run (bottom) habitats. The vertical-dashed line denotes the start of road construction.

Relations Among Metrics and Environmental Variables.—Since significant differences existed between habitats for both biomonitoring metrics, the relations with environmental variables were examined for each habitat separately. A significant, positive correlation was found between diversity and deposited sediment dry weight, while a significant, negative relationship existed between diversity and percent-coarse silt in riffle habitat (Table 20). Species richness had a moderately significant ($p < 0.25$), negative relation with percent-coarse silt; however, no significant correlations between species richness and environmental variables were found in riffle habitat (Table 20).

In run habitat, however, significant, negative correlations with diversity were found with deposited sediment dry weight, TSS, and discharge (Table 21). The only significant, negative relation with species richness in run habitat was found with discharge; although, moderately significant, negative correlations existed with deposited sediment dry weight, TSS, and precipitation (Table 21). Both richness and diversity had a moderately significant, positive relation to percent-coarse silt in run habitat.

Table 20.—Spearman rank correlation coefficients, p-values, and sample size for benthic fish metrics and environmental variables for samples collected throughout the study period in riffle habitat (n = 34). Results significant at α = 0.05 are in bold, while p-values significant at α = 0.25 are italicized.

	RICHNESS	DIVERSITY
DRYWTSED	0.17409	**0.42677**
	0.3248	**0.0118**
TSS	-0.20484	-0.18229
	0.2452	0.3021
SURCOVER	-0.07621	0.04005
	0.6684	0.8221
%<TWOMM	-0.20373	-0.10625
	0.2478	0.5498
SILTPERC	-0.33622	**-0.36565**
	0.0519	**0.0335**
EMBEDD	0.07568	0.06181
	0.6705	0.7284
DISCHMEAN	0.17422	0.31797
	0.3244	*0.0669*
DISCHMAX	-0.07410	0.16177
	0.6770	0.3607
PRECIPMEAN	-0.15726	-0.02753
	0.3744	0.8772

Table 21.—Spearman rank correlation coefficients, p-values, and sample size for benthic fish metrics and environmental variables for samples collected throughout the study period in run habitat (n = 34). Results significant at α = 0.05 are in bold, while p-values significant at α = 0.25 are italicized.

	RICHNESS	DIVERSITY
DRYWTSED	-0.30871	**-0.48083**
	0.0757	**0.0040**
TSS	-0.29772	**-0.42735**
	0.0873	**0.0117**
SURCOVER	0.03544	0.01199
	0.8423	0.9463
%<TWOMM	-0.00624	0.13302
	0.9721	0.4533
SILTPERC	0.31129	0.29634
	0.0731	*0.0888*
EMBEDD	0.06125	0.05150
	0.7308	0.7724
DISCHMEAN	**-0.41852**	**-0.34176**
	0.0138	**0.0479**
DISCHMAX	**-0.53015**	**-0.40857**
	0.0013	**0.0164**
RAINMEAN	-0.31690	-0.31794
	0.0678	*0.0669*

Response of Dominant-Fish Species.—A two-way model III ANOVA was used to test for differences in densities of *E. spectabile, E. caeruluem,* and *E. flabellare* at the pooled-downstream sites throughout the study since these species (combined) composed 91% of the total-adult individuals collected . The count data for each species (except *E. caeruluem*) were $\log_{10}(x+1)$ transformed to meet the model assumptions.

E. spectabile exhibited no significant differences in density between habitats (ANOVA; p = 0.1375) and sample dates (ANOVA; p = 0.0803) (Figure 33). Subsequently, a contrast confirmed no significant change in the overall density of *E. spectabile* occurred after the start of road construction (ANOVA; p = 0.2119).

The density of *E. caeruluem* was significantly different between sample dates (ANOVA; p = 0.0006); however, no statistical difference existed between habitats (ANOVA; p = 0.8012) throughout the study at pooled-downstream sites (Figure 33). Further, no significant difference in the overall density of *E. caeruluem* existed before versus during construction (ANOVA; p = 0.5754).

E. flabellare exhibited the most significant differences in density of all three species throughout the study. Significant differences were found between sample dates (ANOVA; p < 0.0001), habitats (ANOVA; p = 0.0239), and the interaction effect (ANOVA; p = 0.0100). The density of *E. flabellare* was significantly higher in riffle habitat on five occasions (Figure 34). A significant difference also existed in the overall density of *E. flabellare* before versus during construction (ANOVA; p = 0.001). This difference was evident in a positive trend after the start of road construction (Figure 34).

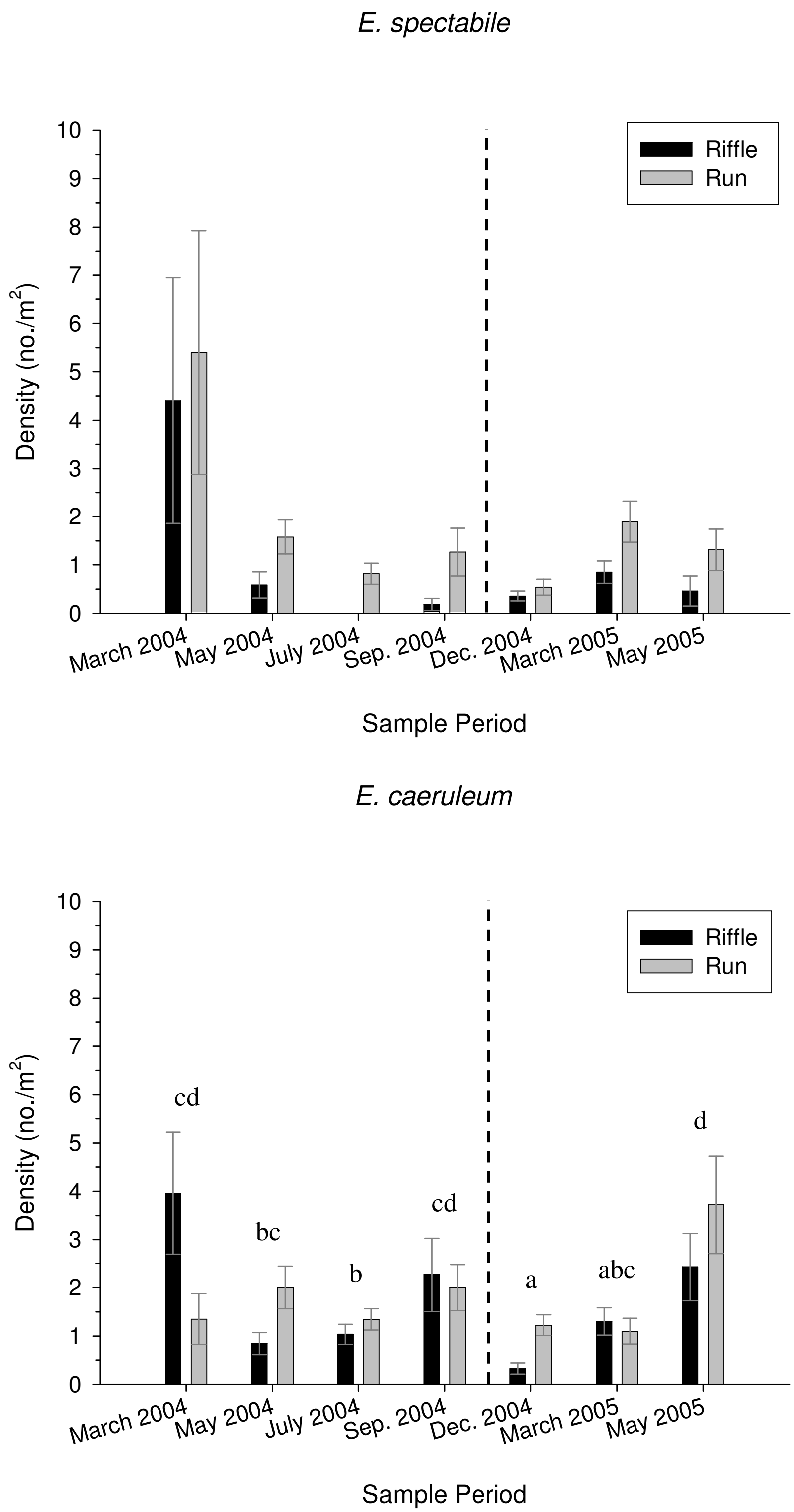


Figure 33.—Mean (± S.E.) density of adult *E. spectabile* (top) and *E. caeruleum* (bottom) for pooled-downstream sites. Sample dates associated with the same letter are not statistically different at $\alpha = 0.05$. The vertical-dashed line denotes the start of road construction.

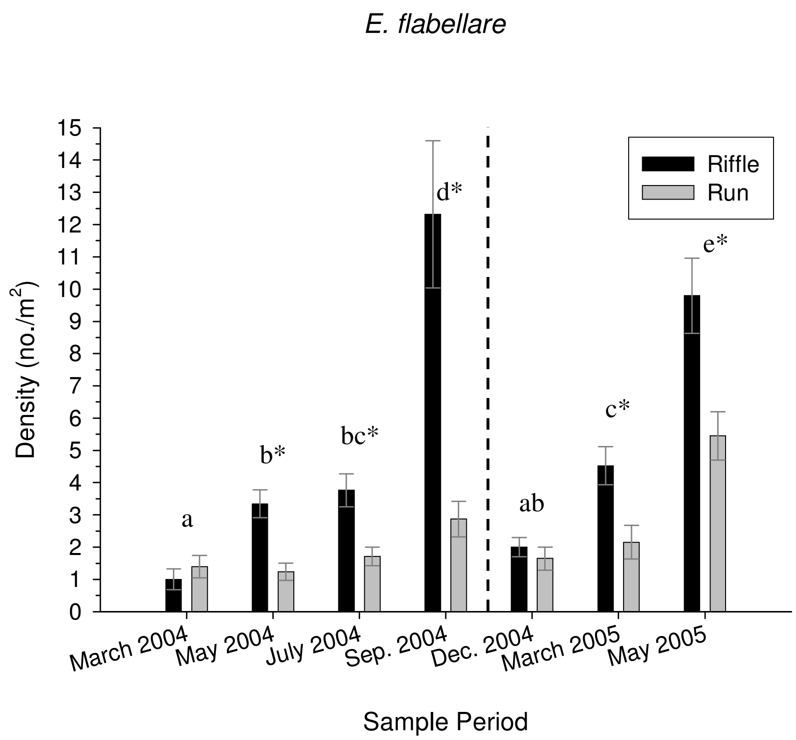
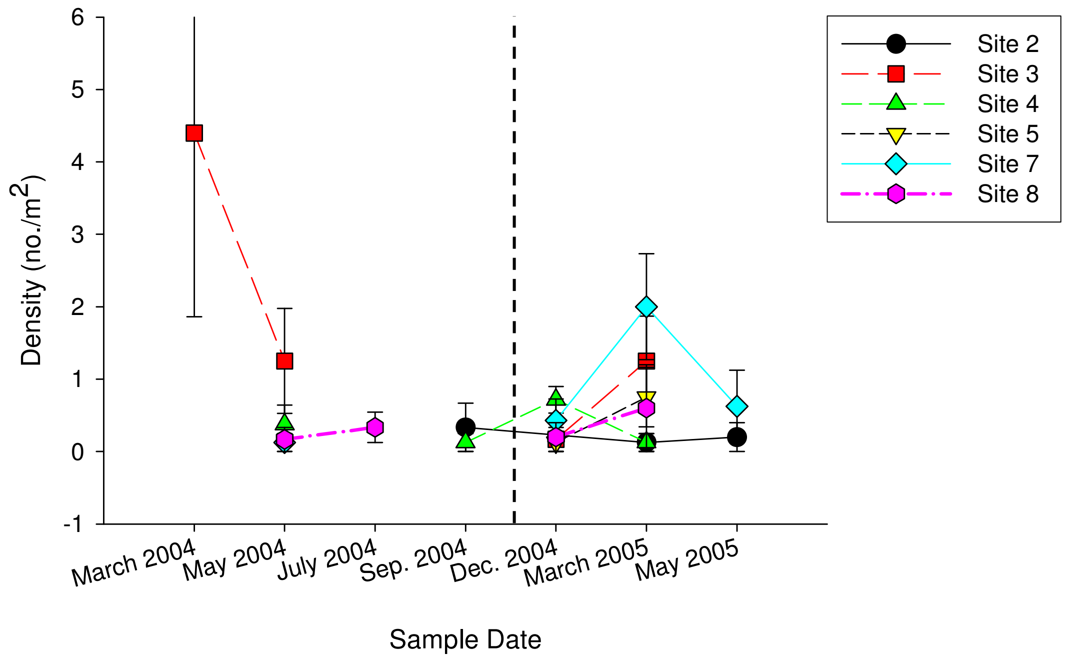


Figure 34.—Mean (± S.E.) density of adult *E. flabellare* for pooled-downstream sites. Sample dates associated with the same letter are not statistically different at $\alpha = 0.05$. The asterisk notes a significant difference between habitats for an individual-sample date. The vertical-dashed line denotes the start of road construction.

Site-Specific Characteristics of Dominant Fish Species.—The density of adult *E. spectabile* did not exhibit any discernable trend among sites in riffle or run habitat throughout the study period (Figure 35). Most sites had a mean density of less than two *E. spectabile* per square meter collected throughout the study, especially in riffle habitat. Moreover, *E. spectabile* were less frequently collected at every site in riffle habitat (Figure 35). In run habitat, site 2 consistently had the lowest-mean density of *E. spectabile* with less than one per square meter collected throughout the study (Figure 35).

Similarly, few consistent trends in the mean density of adult *E. caeruleum* were evident among sites throughout the study (Figure 36). Site 4 was the only site that displayed a detectable tendency with the highest-mean density of *E. caeruleum* (relative to all other sites) in riffle habitat for several-sample dates (Figure 36).

Adult *E. flabellare* were consistently found in higher-mean densities at site 8 (relative to most other sites) in riffle habitat throughout the study; whereas, site 3 commonly displayed the lowest-mean density (Figure 37). Site 4, however, displayed a markedly high-mean density of *E. flabellare* on September 2004 in riffle habitat. In run habitat, site 7 consistently displayed higher-mean densities of *E. flabellare* especially prior to construction (Figure 37). Sites 7 and 8 displayed a distinct increase in mean density of *E. flabellare* after December 2004 in run habitat.



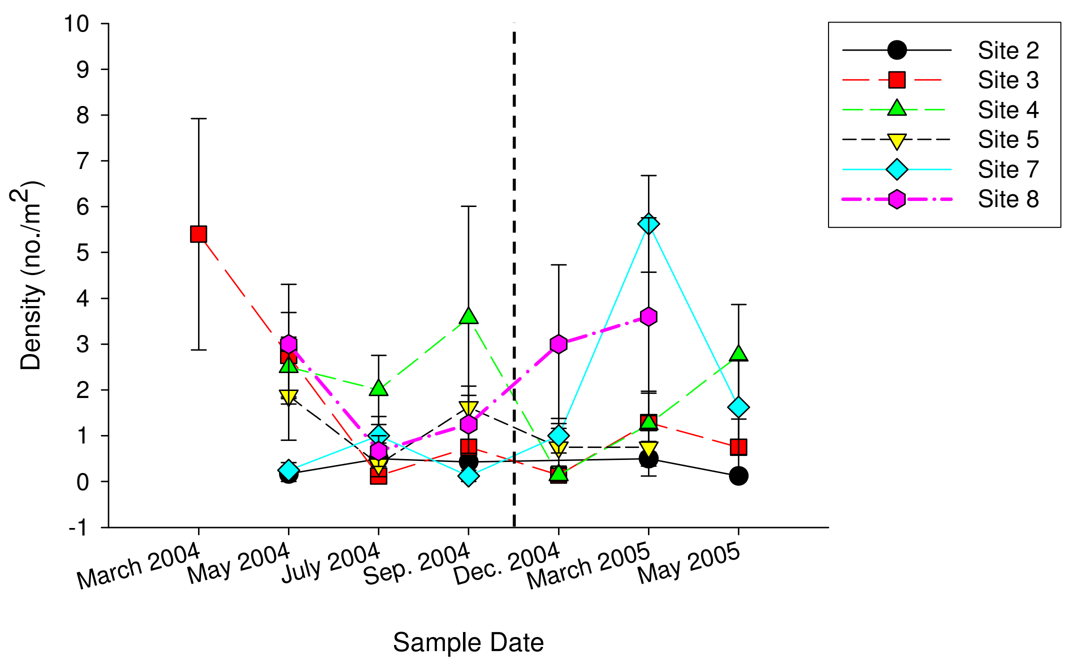
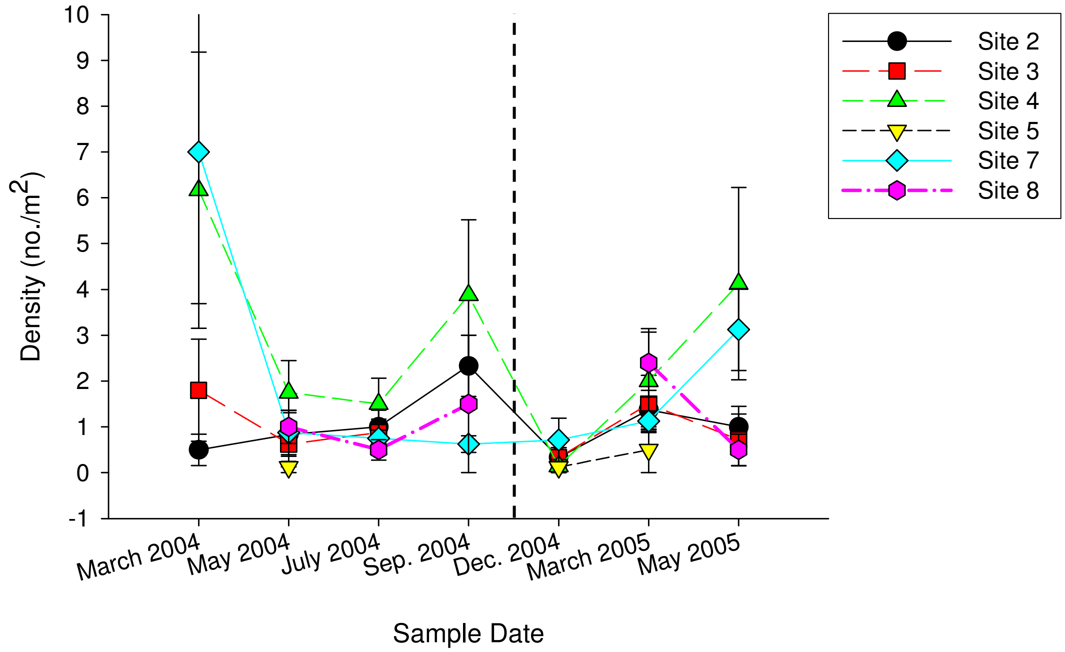


Figure 35.—Mean (±S.E.) density of adult *E. spectabile* for each site in riffle (top) and run (bottom) habitats throughout the study period. The vertical-dashed line denotes the start of road construction.



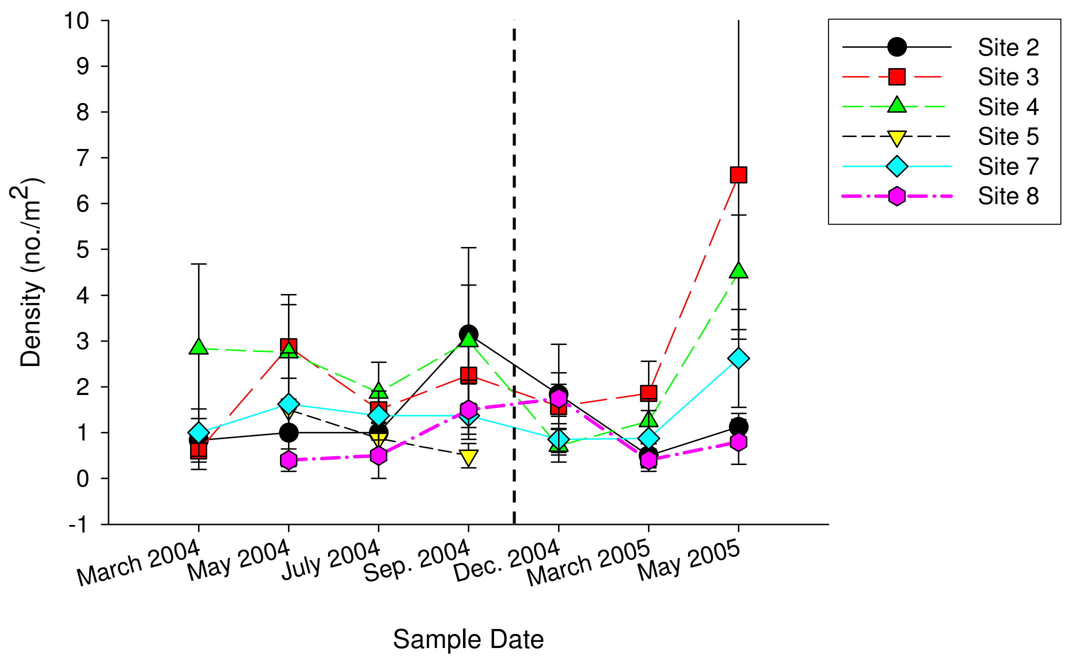


Figure 36.—Mean (±S.E.) density of adult *E. caeruleum* for each site in riffle (top) and run (bottom) habitats throughout the study period. The vertical-dashed line denotes the start of road construction.

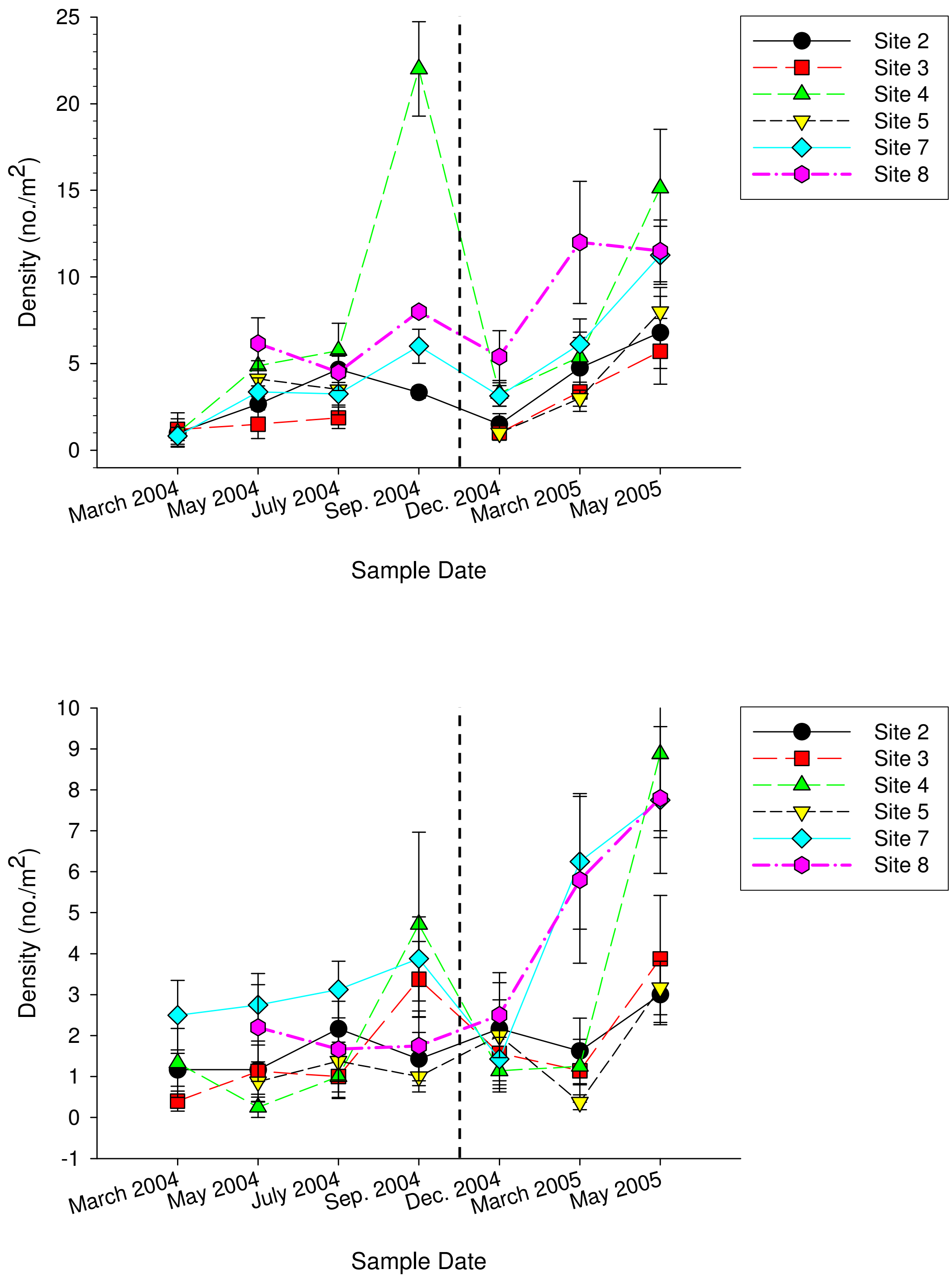


Figure 37.—Mean (±S.E.) density of adult *E. flabellare* for each site in riffle (top) and run (bottom) habitats throughout the study period. The vertical-dashed line denotes the start of road construction.

CHAPTER FOUR

DISCUSSION

Evaluation of Sediment Sampling Methodology

<u>Single-Stage Suspended Sediment Sampler.</u>—The U. S. U-59B single-stage-sediment sampler proved to be a very economical yet effective method of collecting suspended sediment samples during storm events, although a few sources of error existed. Vandalism of these samplers was not a problem at any site; however, during dry-summer months the nozzles of the single-stage samplers often became clogged with wasp nests. This was a temporary problem that resulted in either a reduced-sample volume or no sample at all. No solution to this problem was determined. Another source of error associated with the single-stage sampler is the circulation that can occur when the sampler is submerged for an extended period of time (Subcommittee on Sedimentation 1961). This did occur in several samples throughout the study, however, the sample volume within the bottle served as an indication of this sampling error. If the water surface within the bottle was above the inner end of the exhaust tube, then it was determined that circulation occurred or water entered the bottle due to a dislodged stopper. Samples that did not contain the correct amount of sample volume due to these sources of error were discarded on site and were not taken to the laboratory.

The single-stage sampler consistently provided an accurate measurement of the suspended sediment concentrations during the rising limb of the hydrograph (i.e., sedigraph) as was determined by a direct comparison with the SIGMA automated samplers for many rain events. The primary disadvantage of the single-stage sampler is that it provides little information regarding the duration of suspended sediment.

Deposited Sediment Sampler.—Deposited sediment samples were successfully collected throughout the study using the modified sampler based on the original design by Hedrick et al. (2005). The modification of added-wire mesh on the surface of the sampler effectively retained the artificial gravels inside the sampler. The greatest difficulty in using this sampler was securing each one in the streambed so they did not get flushed away during high flows. The most effective resolution, although not foolproof, was to anchor each sampler to a rebar stake that was driven into the substratum as deeply as possible. This provided a challenge especially at upstream-study sites where bedrock was the dominant-substrate type especially in pools.

In addition to being flushed, the deposited sediment samplers were sometimes buried completely as a result of shifting cobble/gravel. This was especially noticeable at sites 3, 4 and 5 where the substrate frequently shifted due to high flow events. An additional problem existed at site 4 where cattle accessed the stream and trampled many of the samplers. The only solution for these sources of error was to increase the number of samplers per mesohabitat at each site to compensate for those that were either flushed, buried, or trampled.

The deposited sediment samplers did not appear to exhibit a sampling bias based upon the interstitial space measured in each sampler upon collection (see Appendix 4). No correlation existed between sampler volume and the total dry weight of collected sediments, which suggests that the accumulated-deposited sediments were not affected by the interstitial volume of each sampler. This potential bias was not examined in the original study conducted by Hedrick et al. (2005), and did not appear to be an important factor in affecting the accuracy of deposited sediment measurements.

Sediment Dynamics

Suspended Sediment.—The most substantial effect of road construction in the Brush Creek watershed was the overall change in suspended sediment concentration which was 53% greater downstream versus upstream for storm flows during construction activity. A significant increase in suspended sediment due to highway construction is a general consensus reported in the literature (Barton 1977; Carline et al. 2003; Embler and Fletcher 1983; Hainly 1980; Lenat et al. 1981; Reed 1980). More specifically, Carline et al. (2003) reported significant-sediment increases in 55% of rain events that occurred after the start of road construction; whereas, the results of my study showed significant increases in suspended sediment occurred for 44% of the rain events during construction activity at pooled-downstream sites (Table 2).

Elevated-mean TSS concentrations in Brush Creek during pre-construction conditions were present for all seasons compared to other-sediment studies in Missouri streams examined by Doisy and Rabeni (2004). Distinct-seasonal differences in TSS concentrations were found in several Missouri streams that showed the highest and lowest TSS concentrations existed during the spring and fall, respectively (Doisy and Rabeni 2004). The results of my study, however, showed mean-TSS concentrations were significantly higher only during the fall (compared to all other seasons) for the baseline data. In comparison with the seasonal data from a study conducted in Hinkson Creek, Missouri, (Ozark Highlands ecoregion) by Parrish (2000), the mean-TSS concentration during the fall was 23-times greater in Brush Creek. Spring exhibited the second highest mean-TSS concentration in Brush Creek, which was a 3-fold increase (for that season) compared to results of Parrish (2000). The significant increase in mean TSS during the fall in Brush Creek may be attributed to the lack of streamside vegetation during this season and associated riparian functions that serve to filter overland flow (Wilkin and Hebel 1982). Additionally, the increase in significant-rain events associated with fall-weather patterns likely influenced the increase in TSS concentration.

The elevated-TSS concentrations in Brush Creek across seasons (relative to other studies in Missouri) during baseline conditions is likely an indication of long-term-anthropogenic disturbance throughout the watershed associated with various land-use practices. The activities of agriculture, cattle grazing, and timber harvest have extensively degraded or completely removed the riparian corridor in many locations throughout the basin which has resulted in the formation of numerous unstable and actively eroding stream banks. Aerial photography and personal observation throughout the basin confirm this. As a result, extensive amounts of sediment are eroded and transported into the stream during high-flow events as has been demonstrated in other Missouri streams (Burckhardt and Todd 1998). The high proportion (mean > 80%) of inorganic solids found throughout all seasons in Brush Creek confirms the presence of these highly erodable sediments entering Brush Creek.

*Deposited Sediment.—*Increases in deposited sediment measurements at downstream sites in Brush Creek after the start of road construction were expected. However, like other road construction studies (e.g., Carline et al. 2003; Hainly 1980), significant increases in suspended sediment during or after construction did not significantly influence depositional sediment measurements. Furthermore, I expected to find significant differences in deposited sediment dry weight between habitats which were only found on one occasion (Table 5). The reduced variation of deposited sediment dry weight in riffle and pool habitats found at most sites suggest these habitats exhibited more homogenous-sediment profiles. However, significantly higher surface cover of fine sediment was observed in run habitat with low variation found in both mesohabitats sampled. These differences suggest that percent-surface cover may provide better resolution of subtle-habitat differences.

The lack of significant results regarding deposited sediment particles < 2 mm in size provides further evidence that road construction did not have a direct or indirect effect on the composition of the streambed throughout the study. The characteristics of particle sizes did identify fine sand (0.25-0.125 mm) as the dominant-size class found in deposited sediments, which is likely a reflection of the local soil types (Figure 15). Additionally, no clear distinction was evident in the coarse-silt fraction of deposited sediments associated with road construction activity.

Many studies examining the dynamics of deposited sediment failed to report the organic fraction of such sediment. In my study, I showed the organic fraction of deposited sediment in Brush Creek was minimal (overall mean of 4%) throughout the study (Table 7). These results are very similar to a 5-year road construction study conducted by Kreutzweiser et al. (in press) which reported a dominance of inorganic materials (97%) in the deposited sediment of a Canadian headwater stream. The results of my study present evidence that the deposited sediment in Brush Creek provided little (if any) benefit of organic material to benthic fauna before or during road construction, and suggests the origin of these sediments was from various sources of soil erosion previously mentioned.

Physical Habitat Relations

I expected to find a significant association between suspended and deposited sediment measurements in this study. My hypothesis for the lack of significance is that a combination of the discharge and gradient of Brush Creek were adequate to keep most sediment in suspension (i.e., washload) during high flows which resulted in little deposition. Subsequently, this sediment was carried downstream and deposited at some location downstream of site 2, most likely in the Sac River (which flows into Truman Lake), where gradients and discharge were less. This downstream-sediment transport was also reported by Hainly (1980) and Cline et al. (1982) and has been described by Waters (1995) and Gordon et al. (2004). Simply stated, these results show the capacity of Brush Creek to transport the sediment load was not exceeded during this study.

Numerous correlations were found, however, with deposited sediment and physical-habitat measurements (Table 10). The negative association between deposited sediment dry weight and percent-coarse silt is a reflection of the dominance of particle sizes larger than coarse silt (i.e., fine sand). The positive association between percent-surface cover of fine sediment and deposited sediment dry weight was nearly significant ($p = 0.0563$), which suggests that surface cover estimates of fine sediment would be a sufficient surrogate for measuring deposited sediment as was demonstrated by Zweig and Rabeni (2001). This has value since estimating percent-surface cover of fine sediments requires sufficiently less time and effort (i.e., no laboratory time) than using the deposited sediment sampler (or measuring embeddedness) in this study; and the associated struggle of retaining sediment samplers over long periods of time in the streambed is eliminated.

Additionally, the relations between discharge, deposited sediment dry weight, and embeddedness suggest that increased discharge mobilized bedload sediments and increased-substrate embeddedness. Barton (1977) also found that deposited sediment was influenced by discharge. The strong correlation with discharge suggests the deposited sediment sampler used in my study collected primarily bedload sediment rather than sediment that settled out of the washload. This also explains the lack of significant correlations between suspended and deposited sediment measures.

The resolution of sediment dynamics gained from this study will be very useful in aiding the approach of future-sediment monitoring in streams throughout Missouri. The methods used in this study were successful in quantifying both suspended and deposited sediment for a long duration which provides information that was scarcely available for Missouri streams before this study. Needed information was collected regarding the natural variation and distribution of deposited sediment which will help future investigators identify normal and excessive sediment levels in other Ozark highland streams with similar landuse types. Ultimately, the results of this study will aid in the development of sediment-mitigation strategies for potentially impacted watersheds and facilitate the development of TMDL for sediment in Missouri streams.

Benthic Macroinvertebrate Assemblage

<u>*Biomonitoring Metric Response.—*</u>The lack of a notable shift in invertebrate metrics immediately after the start of construction activity compared to the long-term, baseline data for Brush Creek reflects similar, non-significant trends in deposited sediment measurements. This is not unlike other studies that reported little or no effect of

fine-sediment deposition on stream macroinvertebrates (Carline et al. 2003; Culp et al. 1986; Murphy et al. 1981). The only significant differences in invertebrate metrics for both the long-term and present study periods existed between coarse-flow and non-flow habitats specifically in regard to EPT richness. This is likely due to the increased gradient, current velocity, substrate heterogeneity, and overall-water quality of coarse-flow (i.e., riffle and run) habitat required by these sensitive taxa (Merritt and Cummins 1996).

The associations between macroinvertebrate metrics and environmental variables were not consistent between coarse-flow and non-flow habitats. Coarse-flow habitat exhibited the strongest relations between richness metrics and environmental variables, while most non-flow habitat relations with environmental variables were associated with the Biotic Index. Of the deposited sediment measures, percent-coarse silt resulted in the only negative-metric correlation. All other significant correlations were positive. These correlations suggest that deposited sediment did not substantially alter the substrate after the start of construction, and are similar to the weak correlations between metrics and deposited sediment found by Angradi (1999). Only one measure of sediment (in non-flow habitat) was significantly correlated with Shannon diversity in this study which is similar to the non-significant response of this metric to deposited sediment in Missouri Ozark streams studied by Zweig and Rabeni (2001). These results show that perhaps there was little effect of road construction on the biomonitoring metrics evaluated in this study.

*Macroinvertebrate Assemblage Responses.—*Ordination analysis was useful in examining subtle shifts in the composition of the invertebrate assemblage for all sites with samples collected in the spring immediately before and during road construction. Coarse-flow habitat provided the most discernible shift in the assemblage composition (i.e., high-Beta diversity) indicating a greater sensitivity of the riffle assemblage to abiotic differences among sites. The associations with sediment measures and the primary axis (Axis 1) for both habitats (ordinated separately) showed TSS and surface cover of fine sediments were the only variables that had a significant, strong influence on the macroinvertebrate assemblage. Deposited sediment dry weight only appeared to have a significant influence on the invertebrate assemblage in non-flow habitat.

The decline in abundance of two taxa (at downstream sites) known to be sediment intolerant (Zweig and Rabeni 2001) in May 2005 identifies a negative response to a sediment gradient between these two sample dates. Due to the comparison of only one sample before and one during construction, it is tenuous to suggest the response of these taxa was associated with road construction activity. There may have been additional factors such as hydrology that influenced these shifts. According to the results of each ordination, however, if a single mesohabitat must be selected to monitor the response of macroinvertebrates to a perturbation, coarse-flow habitat may provide the greatest ability to discern subtle changes in assemblage composition along a gradient of sediment variables.

Benthic Fish Assemblage

*Biomonitoring Metric Response.—*The most notable differences in the metrics of adult-benthic fishes existed between habitats since a significant time-trend after the start of road construction was not evident. Hydrologic variability of Brush Creek reduced the habitat availability and subsequent metric values in riffle habitat especially during the summer and early fall (i.e., September) when dewatering of riffle habitat was common at all sites (Figure 30). Low Shannon diversity values throughout the study indicated the dominance of *E. spectabile, E. caeruleum,* and *E. flabellare* in the fish assemblage.

Community-metric correlations suggest that Shannon diversity was the most sensitive metric to sediment measures in riffle and run habitats. Most significant relations were negative except for riffle habitat where a positive correlation existed between diversity and deposited sediment dry weight. This deposited sediment relation was strongly negative in run habitat, however, which suggests deposited sediment had a greater influence in this habitat compared to riffles. These results are in contrast to those of Rabeni and Smale (1995) who found no association between Shannon diversity and deposited sediment for fishes sampled only in riffles in small-agricultural streams located in northeast Missouri. This difference is likely attributed to the focus on the benthic-fish assemblage in my study.

*Assemblage Characteristics.—*The notable increase in densities of adult *E. flabellare* in riffle habitat (compared to the other dominant species) can be attributed to the morphological traits (i.e., flexible body and oblique mouth) described by Schlosser and Toth (1984) that allow this species to exploit crevice microhabitat in shallow riffles.

These results also agree with the findings of Rettig (2003) in which adult *E. flabellare* selected fast, shallow habitats in several Ozark headwater streams.

Influences of Cattle Grazing

The lack of a consistent trend throughout the study in sediment measurements and most biomonitoring metrics at the site of intense-cattle disturbance (site 4) suggests instream conditions at that site were not excessively degraded relative to other sites in Brush Creek. However, an examination of invertebrate richness and diversity during the long-term period did indicate reduced water quality and/or habitat suitability at site 4 in coarse-flow habitat only.

These results may be explained by the conditions at this site which have existed for several decades (according to the landowner) and the seasonal access of cattle to the stream. The aquatic assemblage has already been significantly influenced and has likely adapted to the disturbance at this site. Furthermore, cattle were not allowed year-round access to the stream with primary access occurring during the summer and early fall months. During this time, rain events were much less frequent and significant compared to rains during the spring and winter. This may explain the lack of a significant response in sediment measurements simply because very little sediment was transported during this time.

The instability of riffle and run (i.e., coarse-flow) substratum was dominant at this site and others adjacent to it (i.e., sites 3 and 5) where cattle were also present but not allowed instream access. Measures of streambed mobility have recently been examined by Kappesser (2002) and negative associations with aquatic insects were reported by Roy et al. (2003). This likely had an influence on the invertebrate and fish assemblages in Brush Creek and future research is needed in Missouri streams with similar landuses to affirm this relation.

Longitudinal Patterns of Benthic Fishes

Site-specific examinations of biomonitoring metrics for adult-benthic fishes did not reveal a consistent-longitudinal trend in Brush Creek; however, site 2 (the most downstream site) did tend to exhibit the highest species richness and diversity especially in riffle habitat. The lack of decreased-metric values at the most upstream sites may be attributed to presence of an active spring located directly upstream of site 8. This had a noticeable effect on the hydrology of this site during extremely dry periods. Furthermore, the substratum at sites 2 and 7 contained a heterogeneity of large particle sizes (e.g., large cobble and gravel) and higher-gradient riffles relative to all other sites (see Appendix 6 for photographs of each site), which partially explains the increased species richness at these sites in riffle habitat.

Most studies that examined longitudinal trends in fish distribution sampled the complete composition of stream fishes (cf. Horwitz 1978; Kinsolving and Bain 1993; Reash and Berra 1987). The focus on the benthic fish assemblage in my study, however, likely reduced the ability to detect longitudinal patterns in species distribution of fishes in Brush Creek as this was not a primary focus in this study.

Implications for Recovery Efforts of the Niangua Darter

In Brush Creek, the last recorded observations of the federally threatened Niangua darter (*E. nianguae*) were in 1997 (Missouri Department of Conservation, unpublished data) and since that time extensive effort has been made to find this species in Brush Creek with regard to ongoing-monitoring efforts of the MDC. This research provides some suggestions as to causes for the apparent absence of Niangua darters in this Osage River tributary.

First, the presence of disturbance in the Brush Creek watershed due to land use has affected the natural-sediment regime of this stream made evident by the existence of numerous-eroding banks, high-suspended-sediment concentrations (relative to other Ozark streams), and dominance of sand size (inorganic) particles. The results of my study lend further support for a proposed link between pasture land use (with subsequent bank erosion) and reduced availability of substrate particles required by Niangua darters (Mattingly and Galat 2002). Secondly, drought conditions that were frequent during this study caused dewatering of riffle and run habitats at most sites throughout Brush Creek. The lack of available habitat required by this species for extended periods of time has likely contributed to the decline of the Niangua darter.

Associations with the greenside darter (*E. blenniodes*) have been suggested for the Niangua darter since these species are commonly observed together and have similar-habitat requirements (Calfee and Novinger 2003). It is noteworthy that in my study, greenside darters comprised a small fraction (0.8%) of the total-adult individuals collected. This may propose future implications regarding the status of other benthic fishes with similar-substrate preferences of the Niangua darter.

This research showed that effects of road construction were minimal on sediment variables and were not conclusive regarding biotic variables in Brush Creek. This may be attributed to extensive efforts made during road construction to limit sedimentation. Another possible explanation for this minimal impact is the integrity of Brush Creek was already degraded (e.g., pasture and grazing) so road construction was imposed on a fauna that had "pre-adapted" to an altered sediment regime. Therefore, construction impacts were minor and were within the natural range of variability in Brush Creek.

LITERATURE CITED

Alabaster, J. and R. Lloyd. 1982. Water Quality Criteria for Freshwater Fish, 2nd edition. Butterworth Scientific, Boston.

American Public Health Association. 1995. Standard Methods for the Examination of Water and Wastewater, 19th ed. American Public Health Assoc., New York.

Angermeier, P. L., A. P. Wheeler, and A. E. Rosenberger. 2004. A conceptual framework for assessing impacts of roads on aquatic biota. Fisheries 29(12):19-29.

Angradi, T. R. 1999. Fine sediment and macroinvertebrate assemblages in Appalachian streams: a field experiment with biomonitoring applications. Journal of the North American Benthological Society 18(1):49-66.

Barton, B. A. 1977. Short-term effects of highway construction on the limnology of a small stream in southern Ontario. Freshwater Biology 7:99-108.

Bergthold, B. 2001. Macroinvertebrate communities in headwater prairie streams: influence of adjacent land use. Master of Science. University of Missouri, Columbia.

Berkman, H. E., C. F. Rabeni, and T. P. Boyle. 1986. Biomonitors of stream quality in agricultural areas fish vs. invertebrates. Environmental Management 10(3):413-420.

Berkman, H. E. and C. F. Rabeni. 1987. Effect of siltation on stream fish communities. Environmental Biology of Fishes 18:285-294.

Bowlby, J., J. Imhof, R. Biette, and D. Dodge. 1987. Long term impact of highway construction on Mill Creek, Ontario. Ontario Ministry of Natural Resources, Ontario Fisheries Technical Report Series no. 24, Toronto, Ontario.

Boyero, L. 2003. The quantification of local substrate heterogeneity in streams and its significance for macroinvertebrate assemblages. Hydrobiologia 499(1-3):161-168.

Bruton, M. 1985. The effects of suspensoids on fish. Hydrobiologia 125:221-241.

Buchanan, T. J. and W. P. Somers. 1969. Techniques of water-resources investigations of the United States Geological Survey. Book 3, Chapter A8: Discharge measurements at gaging stations. U.S. government printing office, Washington, D.C.

Burckhardt, J. C. and B. L. Todd. 1998. Riparian forest effect on lateral stream channel migration in the glacial till plains. Journal of the American Water Resources Association 34(1):179-184.

Burton, T., R. Turner, and R. Harriss. 1977. The impact of highway construction on a north Florida watershed. Water Resources Bulletin 12:529-538.

Calfee, J. and D. Novinger. 2003. Long-term monitoring efforts for the endangered Niangua darter. The Glade 6(1):4-5.

Carline, R. F., A. Smith, C. Gill, et al. 2003. Effects of bridge and road construction on Spring Creek: Route 26 Transportation Improvements Project. Final Report. Pennsylvania Cooperative Fish and Wildlife Research Unit, Pennsylvania State University.

Cline, L. D., R. A. Short, and J. V. Ward. 1982. The influence of highway construction on the macroinvertebrates and epilithic algae of a high mountain stream. Hydrobiologia 96:149-160.

Cordone, A. J., and D. W. Kelley. 1961. The influences of inorganic sediment on the aquatic life of streams. California Fish and Game 47(2):189-228.

Culp, J. M., F. J. Wrona, and R. W. Davies. 1986. Response of stream benthos and drift to fine sediment deposition versus transport. Canadian Journal of Zoology 64:1345-1351.

Cummins, K. W. and R. W. Merritt. 1996. Ecology and distribution of aquatic insects. Pages 74-86 *in* R. W. Merritt and K. W. Cummins, editor. An introduction to the aquatic insects of North America, third edition. Kendall and Hunt, Dubuque, Iowa.

Dallaire, G. 1976. Controlling erosion and sedimentation at construction sites. Civil Engineering, American Society Civil Engineers 45:73-77.

Doisy, K. E. and C. F. Rabeni. 2004. Effects of suspended sediment on native Missouri fishes: a literature review and synthesis. Missouri Cooperative Fish and Wildlife Research Unit, University of Missouri, Columbia, MO.

Ellis, M. M. 1936. Erosion silt as a factor in aquatic environments. Ecology 17:29-42.

Embler, P. F. and M. O. Fletcher. 1983. Effects of highway construction on stream turbidity and suspended solids: a case study. FHWA/SC-81/3, Federal Highway Administration, Washington, D. C.

Forman, R. T. T. and L. E. Alexander. 1998. Roads and their major ecological effects. Annual Review of Ecology & Systematics 29:207-231.

Forman, R. T. and R. D. Deblinger. 2000. The ecological road-effect zone of a Massachusetts (U.S.A) suburban highway. Conservation Biology 14(1):36-46.

Forman, R. T. 2000. Estimate of the area affected ecologically by the road system in the United States. Conservation Biology 14(1):31-35.

Gauch, H. G., Jr. 1982. Multivariate analysis in community ecology. Cambridge University Press, New York.

Gordon, N. D., T. A. McMahon, et al. 2004. Stream Hydrology: An Introduction for Ecologists. 2nd Edition. John Wiley and Sons, England.

Groshens, T. P. 1997. Brush Creek Basin Inventory and Management Plan. Missouri Department of Conservation, Sedalia, MO.

Hainly, R. A. 1980. The effects of highway construction on sediment discharge into Blockhouse Creek and Steam Valley Run, Pennsylvania. U.S.G.S. Water-Resources Investigations 80-68.

Hedrick, L. B., S. A. Welsh, and J. D. Hedrick. 2005. A new sampler design for measuring sedimentation in streams. North American Journal of Fisheries Management 25(1):238-244.

Hellawell, J. 1977. Biological surveillance and water quality monitoring. Pages 69-88 *in* J. S. Alabaster, editor. Biological monitoring of inland fisheries. Applied Science, London.

Hilsenhoff, W. L. 1987. An improved biotic index of organic stream pollution. Great Lakes Entomology 20:31-39.

Hocutt, C. H. 1981. Fish as indicators of biological integrity. Fisheries 6(6):28-31.

Horwitz, R. J. 1978. Temporal variability patterns and the distributional patterns of stream fishes. Ecological Monographs 48:307-321.

Kappesser, G. B. 2002. A riffle stability index to evaluate sediment loading in streams. Journal of the American Water Resources Association 38(4):1069-1081.

Karr, J. R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6:21-27.

Kinsolving, A. D. and M. B. Bain. 1993. Fish assemblage recovery along a riverine disturbance gradient. Ecological Applications 3(3):531-544.

Klemm, D. J., et al. 2002. Methods development and use of macroinvertebrates as indicators of ecological conditions for streams in the Mid-Atlantic highlands region. Environmental Monitoring and Assessment 78:169-212.

Kreutzweiser, D. P., S. S. Capell, and K. P. Good. In press. Effects of fine sediment inputs from a logging road on stream insect communities: a large-scale experiment approach in a Canadian headwater stream. Canadian Forest Service, Natural Resources Canada, Ontario, Canada.

Lee, K. E., R. M. Goldstein, and P. E. Hanson. 2001. Relation between fish communities and riparian zone conditions at two spatial scales. Journal of the American Water Resources Association 37(6):1465-1473.

Lenat, D. R., D. L. Penrose, and K. W. Eagleson. 1981. Variable effects of sediment addition on stream benthos. Hydrobiologia 79(2):187-194.

Lisle, T. E. 1989. Sediment transport and resulting deposition in spawning gravels, north coastal California. Water Resources Research 25:1303-1319.

Mattingly, H. T. and D. L. Galat. 2002. Distributional Patterns of the Threatened Niangua Darter, *Etheostoma nianguae*, at Three Spatial Scales, with Implications for Species conservation. Copeia 2002(3):573-585.

McCune, B. and M. Medford. 1999. Multivariate Analysis of Ecological Data. Version 4.14, MjM software, Gleneden Beach, Oregon.

Merritt, R. K., and K. W. Cummins. 1996. An introduction to the aquatic insects of North America. (3rd edition). Kendall/Hunt, Dubuque, Iowa.

Meyer, L. D. 1971. Soil erosion by water and upland areas. H. W. Shen, editor. River Mechanics, volume 2. H. W. Shen, Fort Collins, Colorado.

Murphy, M. L., C. P. Hawkins, and N. H. Anderson. 1981. Effects of canopy modification and accumulated sediment on stream communities. Transactions of the American Fisheries Society 110:469-478.

Neter, J., M. H. Kutner, C. J. Nachtsheim, and W. Wasserman. 1996. Applied linear statistical models, Fourth edition. WCB/McGraw-Hill, U.S.A.

Newcombe, C. P., and J. O. Jensen. 1996. Channel suspended sediment and fisheries: a synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management 16(4):693-727.

Newcombe, C. and D. MacDonald. 1991. Effects of suspended sediments on aquatic ecosystems. North American Journal of Fisheries Management 11:72-82.

Nigh, T. A. and W. A. Schroeder. 2002. Atlas of Missouri ecoregions. Missouri Department of Conservation, Jefferson City, Missouri.

Parrish, J. 2000. Temporal variability in the physical, chemical, and biological parameters of Hinkson Creek in response to changes in discharge. Masters of Science. University of Missouri, Columbia, MO.

Peterson, J. T. and C. F. Rabeni. 2001. Evaluating the efficiency of a one-square-meter quadrat sampler for riffle-dwelling fish. North American Journal of Fisheries Management 21:76-85.

Pflieger, W. L. 1997. The fishes of Missouri. Missouri Department of Conservation, Jefferson City, Missouri.

Rabeni, C. F. and M. A. Smale. 1995. Effects of siltation on stream fishes and the potential mitigating role of the buffering riparian zone. Hydrobiologia 303:211-219.

Rabeni, C. F., et al. 1997. Development of regionally based biological criteria for Missouri. Final Report to the Missouri Department of Natural Resources. Missouri Cooperative Fish and Wildlife Research Unit, Columbia, Missouri.

Reash, R. J. and T. M. Berra. 1987. Comparison of fish communities in a clean-water stream and an adjacent polluted stream. The American Midland Naturalist 118(2):301-322.

Reed, L. A. 1980. Suspended sediment discharge in five streams near Harrisburg, PA, before, during, and after highway construction. U.S.G.S. Water-Supply Paper 2072.

Redding, J., C. Schreck, and F. Everest. 1987. Physiological effects on coho salmon and steelhead of exposure to suspended solids. Transactions of the American Fisheries Society 116:737-744.

Rehg, K. J., A. I. Packman, and J. Ren. 2004. Effects of suspended sediment characteristics and bed sediment transport on streambed clogging. Hydrological Processes 19:413-427.

Rettig, A. V. 2003. Associations between benthic fishes and habitat at multiple spatial scales in headwater streams of the Missouri Ozarks. Masters of Science. University of Missouri, Columbia, MO.

Roy, A. H., A. D. Rosemond, D. S. Leigh, et al. 2003. Habitat-specific responses of stream insects to land cover disturbance: biological consequences and monitoring implications. Journal of the North American Benthological Soceity 22(2):292-307.

Sarver, R. and S. McCord. 2001. Taxonomic levels for macroinvertebrate identifications. Missouri Department of Natural Resources, Division of Environmental Quality, Environmental Services Program, Jefferson City, MO.

SAS Institute Inc. 1999. SAS OnlineDoc®, Version 8, Cary, NC: SAS Institute Inc.

Schlosser, I. J. and L. A. Toth. 1984. Niche relationships and population ecology of rainbow (Etheostoma caeruleum) and fantail (E. flabellare) darters in a temporally variable environment. Oikos 42:229-238.

Shannon, C. E. and W. W. Weaver. 1949. The mathematical theory of communication. University of Illinois Press, Urbana.

Stewart, J. S., L. Wang, J. Lyons, J. A. Horwatich, and R. Bannerman. 2001. Influences of watershed, riparian-corridor, and reach-scale characteristics on aquatic biota in agricultural watersheds. Journal of the American Water Resources Association 37(6):1475-1487.

Subcommittee on Sedimentation. Federal Inter-Agency Sedimentation Project. 1961. A study of the methods used in measurement and analysis of sediment loads in streams. Report no. 13:The single-stage sampler for suspended sediment.

Thomas, R. B. 1985. Measuring suspended sediment in small mountain streams. General Technical Report PSW-83. Pacific Southwest Forest and Range Experiment Station, Forest Service, U.S. Department of Agriculture, Berkeley, CA. 9 p.

(USEPA), U. S. E. P. A. 2000. Atlas of America's polluted waters, volume EPA 840-B-00-002. Office of Water, US Environmental Protection Agency, Washington, DC.

(USEPA), U. S. E. P. A. 2005. Water quality inventory for the state of Missouri. Available at http://oaspub.epa.gov/waters/w305b_report.state?p_state=MO.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Science 37:130-137.

Waters, T. 1995. Sediment in streams: sources, biological effects, and control. American Fisheries Society, Monograph 7.

Wellman, J. C., D. Combs, and S. Cook. 2000. Long-term impacts of bridge and culvert construction or replacement on fish communities and sediment characteristics of streams. Journal of Freshwater Ecology 15(3):317-328.

Wesche, T. A., D. W. Reiser, V. R. Hasfurther, W. A. Hubert, and Q. D. Skinner. 1989. New technique for measuring fine sediments in streams. North American Journal of Fisheries Management 9:234-238.

Wilkin, D. C. and S. J. Hebel. 1982. Erosion, redeposition and delivery of sediment to midwestern streams. Water Resources Research 18:1278-1282.

Wood, P. J., and P. D. Armitage. 1997. Biological effects of fine sediment in the lotic environment. Environmental Management 21(2):203-217.

Zimmerman, J. K., H. B. Vondracek, and J. Westra. 2003. Agricultural land use effects on sediment loading and fish assemblages in two Minnesota (USA) watersheds. Environmental Management 32(1):93-105.

Zweig, L. D. 2000. Effects of deposited sediment on stream benthic macroinvertebrate communities. Masters of Science. University of Missouri, Columbia.

Zweig, L. D., and C. F. Rabeni. 2001. Biomonitoring for deposited sediment using benthic invertebrates: a test on 4 Missouri streams. Journal of the North American Benthological Society 20:643-657.

APPENDIX 1: DETAILED LIST OF MATERIALS FOR SEDIMENT SAMPLERS AND WATER LEVEL LOGGER

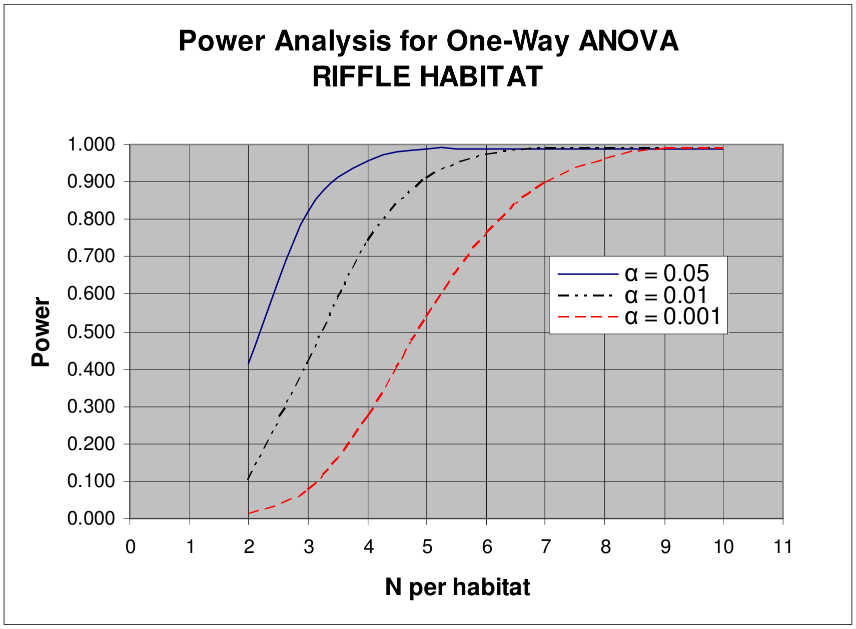
Appendix 1.—A detailed list of materials and supporting information for the suspended sediment sampler, deposited sediment sampler, and water level logger used in this study. An estimate of the cost per sampler (and logger) is provided as of January 2004.

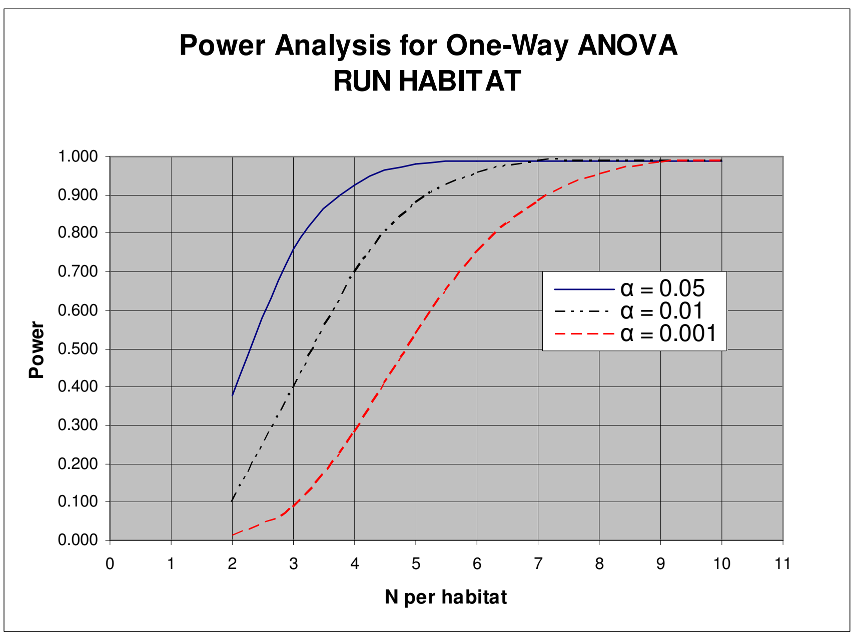
Suspended Sediment Sampler			
Item	**Supplier**	**Part #**	**Price**
Pint-plastic bottle	Rickly Hydrological	405-010	$1.33 /bottle
Two-hole rubber stopper	Fisher Scientific	14140L	$1.77 /stopper
Copper tubing (1/4" I. D.)	any hardware store		$1.50 /sampler
1/2" steel rebar (8 foot)	any hardware store		$3.00
		Estimate of cost per bottle =	**$5.00 /sampler**

Deposited Sediment Sampler			
Item	**Supplier**	**Part #**	**Price**
Sch-40 PVC pipe - 10 ft. section makes 60 samplers	any hardware store		$16.96 /10 ft.
1/2" hardware cloth	any hardware store		N/A
4" plastic knock-out plug	any hardware store		N/A
1/4" staples	any hardware store		N/A
Artifical river rock	any home store		N/A
		Estimate of cost per sampler =	**$1.50 /sampler**

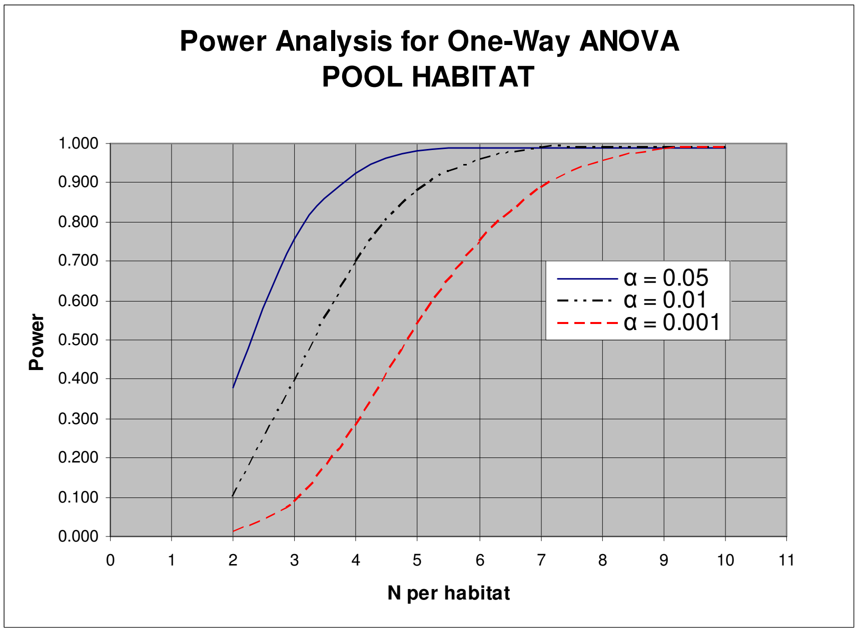
Water Level Logger			
Item	**Supplier**	**Part #**	**Price**
Water level logger - 15 ft. range	Forestry Suppliers	90714	$779
ABS pipe - 2"	any hardware store		$12 /10 ft.
90 degree PVC elbow - 2"	any hardware store		$1.18
Locking well cap	Global Water	N/A	$15
		Estimate of cost for logger used in this study =	**$820 /logger**

APPENDIX 2: POWER ANALYSIS FOR DEPOSITED SEDIMENT SAMPLERS



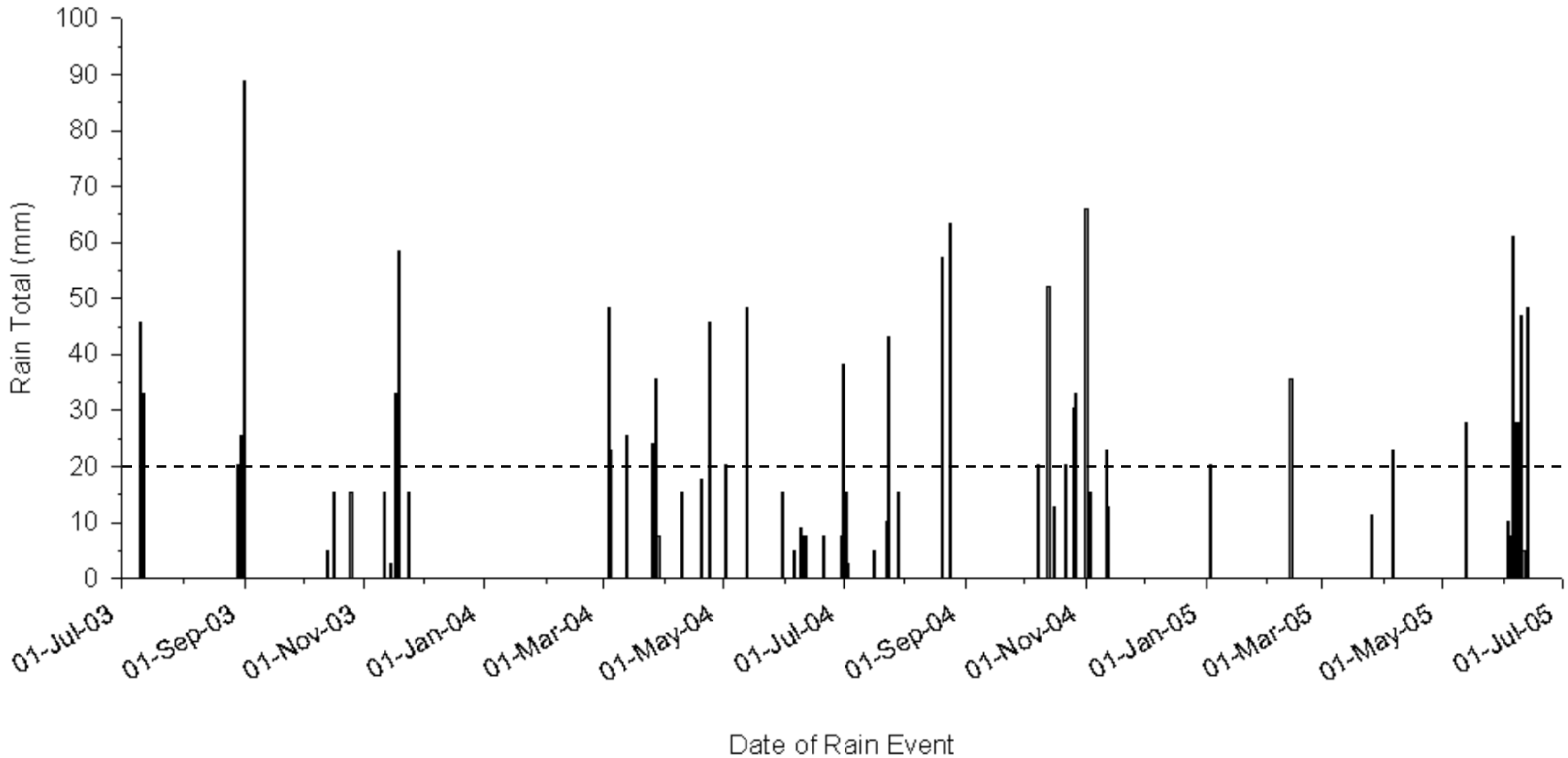


Appendix 2.—Power analysis results for deposited-sediment samplers in riffle, run, and pool habitats. Number of samplers (n) per habitat is represented on the x-axis with the power of a one-way ANOVA on the y-axis at different alpha levels.



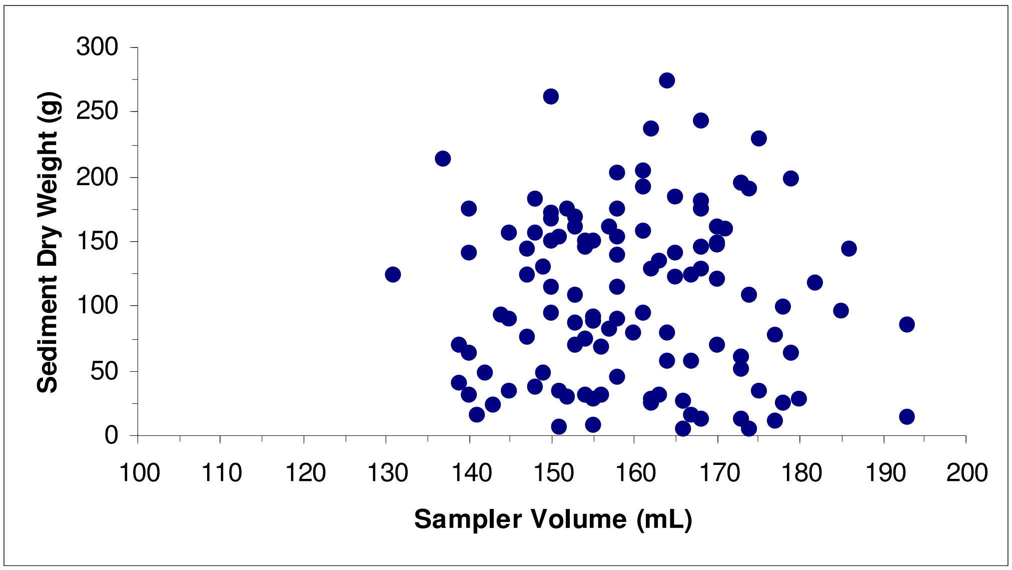
Appendix 2.—continued.

APPENDIX 3: PRECIPITATION TOTALS FOR THE BRUSH CREEK WATERSHED THROUGHOUT THE STUDY



Appendix 3.—The rainfall data for the Brush Creek watershed provided by a local landowner in Humansville, MO. The median rainfall for the entire study period is represented by the horizontal-dashed line.

APPENDIX 4: INTERSTITIAL VOLUME OF DEPOSITED SEDIMENT SAMPLERS



Appendix 4.—Scatterplot of the interstitial (i.e., water) volume versus the total-dry weight of sediments in each sampler for deposited sediment samples collected throughout the study.

APPENDIX 5: DEFINITIONS OF ENVIRONMENTAL VARIABLES USED IN CORRELATION ANALYSIS THROUGHOUT THE STUDY

Appendix 5.—Definitions of the environmental variables used in the correlation analysis of this study. All variables were calculated for each sample date (i.e., date when deposited sediment was collected) at every site where such variables were measured.

Variable Abbreviation	Definition of Variable
DRYWTSED --	The mean-dry weight of all deposited sediment among pooled habitats for each time period between collection dates.
TSS --	The overall mean TSS for pooled rain events that occurred during each time period between deposited sediment sample dates.
SURCOVER --	The mean-percent-surface cover of fine sediments among pooled habitats for each time period between sample dates.
%<TWOMM --	The mean-relative percent of deposited sediment particles <2mm in size among pooled habitats for each time period between sample dates.
SILTPERC --	The mean-relative percent of coarse-silt among pooled habitats for each time period between sample dates.
EMBEDD --	The mode embeddedness rating among pooled habitats for each time period between sample dates.
DISCHMEAN--	The mean discharge in Brush Creek at site 2 for each time period between sample dates.
DISCHMAX --	The maximum discharge in Brush Creek at site 2 that occurred for each time period between sample dates.
PRECIPMEAN-	The mean precipitation that occurred for each time period between sample dates.

APPENDIX 6: PHOTOGRAPHS OF STUDY SITES AND HIGHWAY CONSTRUCTION ACTIVITIES

Appendix 6.—Photographs of site 9 in Panther Creek facing upstream (above) and downstream (below). Highway 13 is visible in the downstream view of site 9. Suspended sediment samplers were the only samplers installed at this site.

Appendix 6.—Photographs of site 8 in Brush Creek at upstream (above) and downstream (below) locations. Both sediment samplers and a water level logger were installed at the upstream location of site 8. The downstream location is immediately upstream of Highway 13. Both photographs were taken facing upstream in Brush Creek.

Appendix 6.—Photographs of site 7 in Brush Creek facing upstream (above) and downstream (below). Both sediment samplers, a water level logger, and an automated sampler were installed at site 7.

Appendix 6.—Photographs of site 6 in Brush Creek facing upstream (above) and downstream (below). Only suspended sediment samplers were installed at site 6. This site is immediately downstream of Highway N and the Humansville, MO, sewage treatment plant.

Appendix 6.—Photographs of the Brush Creek tributary at site 6 facing upstream (above) and at the mouth of the tributary (below). Only suspended sediment samplers were installed at the mouth of this tributary site. This tributary flowed directly downstream of Highway 13.

Appendix 6.—Photographs of site 5 facing upstream (above) and downstream (below). Both sediment samplers were installed at this site. This site had very unstable substrate that shifted throughout the study.

Appendix 6.—Photographs of site 4 facing upstream (above) and downstream (below). Both sediment samplers were installed at this site. The mouth of Panther Creek is at the top-left of the top photo. Local cattle access this site extensively throughout the year. The substrate at this site was highly unstable and shifted throughout the study.

Appendix 6.—Photographs of site 3 facing upstream (above) and downstream (below). Both sediment samplers, a water level logger, and an automated sampler were installed at site 3.

Appendix 6.—Photographs of site 2 facing upstream (above) and downstream (below). Both sediment samplers and a water level logger were installed at site 2. The county road SE 431 bridge is visible in the downstream photograph.

Appendix 6.—Aerial photographs of Highway 13 bridge construction at site 9 (Panther Creek) looking upstream (above) and site 8 (Brush Creek) looking downstream (below).

Appendix 6.—Aerial photograph of Highway N bridge construction at site 6 (above) and the erosion-control structure installed at the mouth of the Brush Creek tributary at site 6 (below). The direction of stream flow is right to left in both photographs. The stream course of the tributary at site 6 is visible in the top photograph.