MACROINVERTEBRATE ASSEMBLAGE COMPOSITION ALONG A LONGITUDINAL MULTIPLE-LAND-USE GRADIENT IN A MIDWESTERN STREAM

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by

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

MACROINVERTEBRATE ASSEMBLAGE COMPOSITION ALONG A LONGITUDINAL MULTIPLE-LAND-USE GRADIENT IN A MIDWESTERN STREAM.

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a candidate for the degree of Master of Science

and hereby certify that, in their opinion, it is worthy of acceptance.

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DEDICATION

I would like to dedicate this thesis to my wife, Alexis M. Malone, who has been a constant and indispensable source of inspiration and encouragement. And to my parents, Pat and Jerry, who have given me love and support throughout my life.

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I would like to acknowledge my advisor, Dr. Jason Hubbart, who encouraged me and gave me the freedom to design my own project and to go beyond the scope of my research assistantship. His insights and experience were always helpful in interpreting the complexities of my data. Working with Dr. Hubbart facilitated my growth as a student and scientist as well as a person.

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ABSTRACT

Land use changes often alter hydrogeomorphology and hydrologic flow regimes, which can adversely impact stream aquatic biota. Hinkson Creek Watershed (HCW, 231km^2) located in Boone County, central Missouri, is comprised of 50% agriculture, 26% forest, and 20% urban (City of Columbia; 2010 population, 108,500) land use types. Five replicated study sites were established to investigate land use effects on macroinvertebrate assemblages using a nested-scale experimental watershed study design. Water temperature, suspended sediment, chlorophyll-a, pH, chloride, discharge, physical habitat and benthic macroinvertebrates were monitored at each site. Missouri Biotic Index was higher ($P < 0.05$) in urban sites (6.77) compared to rural sites (6.26). Percentage of fine substrate increased 328% from the headwaters to the lower reaches. Submerged woody rootmats were 78% smaller in the lower reaches of the stream. Average winter Chloride concentrations were 126% higher in urban reaches of the stream compared to rural reaches (116.6 mg/L and 51.5 mg/L respectively). Mean suspended sediment particle size decreased with stream length ($P < 0.05$) from 108.2µm to 66.6µm. Results indicate that the influence of disturbance regimes associated with local hydrogeomorphology may be as important in structuring benthic community composition as anthropogenic effects associated with agriculture and urbanization.

CHAPTER I

INTRODUCTION

LAND USE EFFECTS ON STREAMS

Land use, including agriculture, forest harvesting, and urbanization, can have profound impacts on receiving water bodies (Allen 2004). Impacts can include alteration of flow, sediment, thermal regimes, stream geomorphology, aquatic and riparian habitat, the addition of pollutants and nutrients, and a reduction of aquatic species richness and diversity (Allen et al. 2004). Agriculture can result in excess nutrient loading that can lead to eutrophication and anoxia (Morgan et al. 2006). In addition, habitat degradation associated with riparian forest clearing, channel straightening and sedimentation, is often present in agricultural streams, and can lead to substantially degraded macroinvertebrate community assemblages (Heatherly et al. 2007, Stone et al. 2006). Aquatic invertebrate communities can be affected by forest harvesting activities, which often leads to soil erosion and sedimentation of streams (Mallik et al. 2011) and increases in stream temperature (Reid et al. 2010). Similarly, urban development can lead to increased sedimentation and habitat degradation (Boothe and Jackson 1997), changes in riparian vegetation (White and Greer 2006), and increases in hydrologic disturbance (Coleman et al. 2011).

Increases in suspended sediment concentration can reduce light penetration through the water column, and cause respiratory impairment of fish and insects with external gill structures (Waters 1995, Doisy and Rabeni 2004). Reduction in light penetration can impact fish and macroinvertebrates that rely on sight for feeding (Doisy and Rabeni 2004). Sediment increases water temperature through increased absorption of solar radiation, and reduces primary productivity through a reduction in photosynthesis. Sediment also acts as a transport mechanism for nutrients and pollutants (Doisy and Rabeni 2004). Increased sediment deposition can degrade aquatic habitat by filling interstitial spaces between rocks that provides habitat to many types of macroinvertebrates, thereby smothering macrophytes and algae that supply the base of aquatic food webs (Rabeni et al. 2005, Zweig and Rabeni 2001).

Increases of stream water temperature stress aquatic organisms by reducing the dissolved oxygen concentration of the water, by disrupting the timing of thermal cues that trigger development stages, or by causing direct mortality (Hynes 1970, Resh and Rosenberg 1984, Wang and Kanehl 2003, Hester and Doyle 2011). In addition, the removal of riparian vegetation and channelization of streams can disrupt stream water temperature regimes, causing further disruption or mortality to aquatic organisms (Resh and Rosenberg 1984).

In addition to land use, geomorphology plays a critical role in the structure and function of streams in relation to habitat patch structure and hydraulic conditions (Montgomery 1999). At the reach scale, local geomorphic features influence patch dynamics and habitat complexity (Montgomery 1999). Local geomorphology is influenced by slope and sinuosity of the stream channel, which can vary based on the position of a reach longitudinally within a stream (Church 2002). Headwater streams tend to exhibit steeper slopes and straighter channel form and are typically characterized by coarser sediment (Church 2002). Larger streams with well-developed floodplains are more prone to lateral meandering and have shallower gradients, and finer sediment (Church 2002). Streams often exhibit corresponding longitudinal patterns of fish and macroinvertebrates assemblage structure associated with the progression of the stream from headwaters to mouth (Vannote et al. 1980). However, longitudinal progressions are not always continuous and localized geomorphic features are understood to exact greater influence over community composition at the scale of a single reach, riffle, or pool (Montgomery 1999, Walters et al. 2003).

Specific Effects of Urbanization

Urban development impacts the environment in a variety of ways, including reduction of fish and wildlife habitat, increased impervious surface area, introduction of exotic species and disruption of natural ecosystem processes (Alberti 2005). Low levels of urbanization, with as little as 10% impervious surface area were shown to result in detectable changes in aquatic community composition (Booth and Jackson 1997). Stream flow regime alterations resulting from urbanization can include increased sediment load, a flashier hydrograph characterized by higher peak discharge and lower base flow, elevated water temperature, higher nutrient loading, increased algal biomass, and the addition of petroleum products, pesticides and other pollutants (Paul and Meyer 2001, Walsh et al. 2005). The combined effect of these alterations is referred to as “The Urban Stream Syndrome” (Walsh et al. 2005). Urban induced impacts of natural hydrologic flow regimes and additions of excess nutrients and anthropogenic pollutants can impair fish, macroinvertebrates, algae, and macrophytes communities, reducing species richness and diversity (Paul and Meyer 2001, Walsh et al. 2005). Increases in impervious surfaces in urban areas can result in reduced infiltration and subsurface flow and increased surface runoff (Booth and Jackson 1997). Decreased infiltration can cause stream levels to rise faster during runoff events and can reduce sources of base flow from groundwater (Booth and Jackson 1997, Paul and Meyer 2001). Higher stream velocity may increase bank erosion and streambed incision and result in declines of habitat diversity (Booth and Jackson 1997, Walsh et al. 2005). Water flowing over surfaces such as roads, sidewalks, parking lots and rooftops, can also contribute to increasing water temperature. Hot pavement and other surfaces conduct heat to rainwater flowing in contact with it and the water carries that energy to the stream (Walsh et al. 2005, Paul and Meyer 2001).

Alterations to stream geomorphology have been associated with urban development including channel straightening (Booth and Jackson 1997), bank erosion (Trimble 1997), and streambed incision (Booth and Jackson 1997). However, these observations are not universal. Nelson et al. (2006) studied aerial photographs of Dead Run Creek near Baltimore, MD and found very little change in channel geomorphology since the 1930s. Similarly, Kang and Marston (2006) found local geologic conditions to be more important than urbanization in maintaining stable stream geomorphology. The physical integrity of stream channels has been identified as an important factor regulating the diversity and composition of benthic communities (Death and Winterbourn 1995) and can play a critical role in the degradation of invertebrate populations in urban streams (Asmus et al. 2009, Coleman et al. 2011). Studies evaluating stream restoration projects in urban areas found that the creation of habitat structures and meanders does not consistently improve macroinvertebrate health when the hydrologic regime remains unchanged (Tullos et al. 2009, Violin et al. 2011).

The magnitude of impacts from urbanization at a given location is dependent, in part, on the natural geomorphology of the stream and antecedent environmental conditions prior to urban development. Cuffney et al. (2010) studied invertebrate communities across urban gradients in nine cities in the US and found a stronger response to urbanization in areas where the antecedent land cover was forest or grassland as opposed to agriculture. The authors concluded that the effects of past agricultural activity can create degraded reference conditions (Cuffney et al. 2010). Soil erosion associated with poor agricultural practices and forest clearing, which often precedes agricultural activity, can contribute significant amounts of sediment to streams (Allen 2004, Jackson et al. 2005). The legacy effects of land uses such as deforestation or agriculture can have significant long term effects on macroinvertebrate communities that can persist long after that land use has ceased or has been replaced by another type of land use (Harding 2000, Harding et al. 2005, Greenwood et al. 2012).

BENTHIC MACROINVERTEBRATES AS BIOINDICATORS

Because benthic macroinvertebrates are known to respond to a wide range of environmental stressors and are highly diverse and widely distributed, they are commonly used as bioindicators of aquatic ecosystem status (Rosenberg and Resh 1993, Barbour et al. 1999). For example, certain taxonomic groups, such as the orders Ephemeroptera, Tricoptera, and Plecoptera, are known to be more sensitive than other organisms to pollution and disturbance (Hilsenhoff 1997, Lenat 1993, Barbour et al. 1999, Hynes 1970). By measuring diversity and relative abundance of species that are intolerant and tolerant to perturbation at a given stream site, scientists can indirectly assess the health status of a stream (Rosenberg and Resh 1993, Barbour et al. 1999). Metrics used to compare stream sites can include ratios of pollution intolerant organisms to tolerant organisms, percentage of the dominant taxon, percentages or richness of certain groups of taxa (e.g. Ephemeroptera, Plecoptera, and Tricoptera (EPT), and comparisons of different functional feeding groups (Merritt et al 2008, Barbour et al. 1999, Rabeni et al. 1997).

The average pollution tolerance level of a population of invertebrates is typically expressed using a biotic index (BI) that can be used to compare different sites and streams (Hilsenhoff 1987, Lenat 1993, Rabeni et al. 1997). A number of biotic indices exist in different regions, but most are based primarily on the work of Hilsenhoff (1987) and later work by Lenat (1993). Hilsenhoff derived tolerance values from a study of 53 Wisconsin streams in which the degree of organic and nutrient pollution had been assessed a priori using physical and chemical parameters (Hilsenhoff 1987). Taxa were assigned values from 0-10 based on their presence and abundance in streams with differing levels of pollution, with lower values indicating that those taxa were only found in relatively unpolluted streams (Hilsenhoff 1987). Lenat (1993) used similar methods to develop the North Carolina Biotic Index (NCBI), although accommodating a wider range of pollution or disturbance types. Higher BI values result from greater percentages of tolerant organisms and indicate greater levels of disturbance or degradation (Hilsenhoff 1987, Lenat 1993, Rabeni et al. 1997).

Declines in invertebrate communities associated with urbanization are well documented, and have been shown to be related to landscape or land use at various spatial scales (Walsh et al. 2001, Roy et al. 2003a, Stepinuck et al. 2002, Cuffney et al. 2010). However, most investigations do not identify the specific physical processes that affect macroinvertebrate communities. Numerous studies have used macroinvertebrates as bioindicators to assess the effects of urbanization on stream health. Urban streams were shown to generally have a higher Biotic Index than rural or reference streams, indicating degraded conditions (Walsh et al. 2001, Stepenuck et al. 2002, Roy et al. 2003a, Alberti 2005, Voelz et al. 2006, Cuffney et al. 2010). Walsh et al. (2001) concluded that total taxa richness and EPT richness declined as urbanization increased and that declines were correlated with increased impervious surface and more degraded riparian zones. Roy et al. (2003a) showed that sites with 10-15% or more urban land cover showed consistently high BI values. In that study, higher scores of the Invertebrate Community Index, a multi-metric index developed by the Ohio Environmental Protection Agency (1989), were best explained ($r^2 = 0.78$, $P = <0.001$) by larger average riffle particle size, lower specific conductivity, and higher variability in stream substrate particle size. Cuffney et al. (2010) synthesized results from nine cities across the United States and found consistent declines of macroinvertebrate diversity and richness measures to urbanization.

Water quality data associated with macroinvertebrate studies are often collected at intermittent intervals, such as monthly (Walsh et al. 2001, Roy et al. 2005, Voelz et al. 2006) or seasonally (Cuffney et al. 2010). Monthly and seasonal temporal resolution may not adequately capture the variability present in certain water quality parameters of flashy urban streams. Instead of direct measurements, some studies used measures of urban intensity such as percent urban area (Roy et al. 2003a), percent impervious surface (Walsh et al. 2001), area of impervious surface connected to stream channel (Stepinuck et al. 2002), road density, population density or some combination (Cuffney et al. 2010) to correlate with macroinvertebrate metrics.

Biomonitoring with Trait-Based Metrics

In recent decades, examining aquatic fauna life-history or functional traits to assess stream health has gained popularity (Poff 1997, Statzner and Beche 2010, Webb et al. 2010). By examining physical or behavioral traits that show the most change between reference and study stream conditions, inference can be made about the physical environmental conditions that result in changes in trait composition (Poff 1997, Statzner and Beche 2010). Studies comparing the effectiveness of functional trait approaches and structural identity approaches concluded that traits-based analysis can provide a more consistent response to environmental stressors than conventional approaches which focus on multimetric scores (Dolodec et al. 2003, Pollard and Yuan 2010). Relative abundance of traits were found to have very little variation among natural streams across relatively large geographical areas, making them a suitable bioassessment tool for developing attainment criteria or regulatory standards for aquatic health (Statzner and Beche 2010). Trait-based studies have been aided by the development of large electronic databases describing trait affinities of many taxa across large geographic regions, primarily Europe and North America (Statzner et al. 2007, Usseglio-Poletera et al. 2000, Vieira et al. 2006). This increase in available information makes study results more accessible to researchers and land managers (Statzner and Beche 2010).

CASE STUDY: HINKSON CREEK WATERSHED

Hinkson Creek is located in Boone County, central Missouri, USA. The Missouri Department of Natural Resources (MDNR) placed Hinkson Creek on the state’s list of impaired streams under section 303(d) of the Clean Water Act (CWA) in 1998. The cause of impairment was listed as unknown (EPA 2011). Subsequent bioassessment studies of Hinkson Creek indicated that sections of the creek were not fully supporting of aquatic life, but water quality analyses and follow-up studies were unable to determine a specific cause of impairment (MDNR 2002, 2004, 2005, 2006).

Previous Investigations

Macroinvertebrate assessments in Hinkson Creek were conducted from 2001 to 2006 (EPA 2011). During that time a total of 11 sites were sampled during 7 distinct seasons, resulting in 34 samples collected throughout the stream (See Table 1, Figure 1) (EPA 2011). In all of these studies, four indicator metrics were used to compare macroinvertebrate communities among sites, including 1) Taxa Richness (TR), which is the total number of taxa, 2) EPT taxa richness (EPT), 3) Missouri Biotic Index (BI), and Shannon's Diversity Index (SDI) (MDNR 2002, 2004, 2005, 2006). The SDI is a measure of community diversity that takes into account both the number of taxa and evenness of distribution of taxa. The values derived from the four metrics were compared to the values of regional reference streams and assigned a score according to the degree to which they exceeded, matched, or fell below the reference stream values (MDNR 2002, 2004, 2005, 2006). The scores, with possible values ranging from 1 to 5, were added together for each site to calculate a Stream Condition Index (SCI) value with a possible range of 4 to 20. The SCI is used to determine if a stream reach is considered fully supporting (score of 20-16), partially supporting (14-10), or not supporting (<10) of aquatic life. During five of the seven sampling seasons macroinvertebrate samples were collected from Bonne Femme Creek, a stream approximately 10 km to the south of Hinkson Creek, within a predominately rural watershed. Bonne Femme Creek was chosen to serve as a "control" stream (MDNR 2002, 2004, 2006). Water quality indices were also collected during macroinvertebrate sampling visits. Variables measured included stream temperature, dissolved oxygen (DO), conductivity, pH, turbidity, chloride concentration, total phosphorus (TP), ammonia nitrogen (NH3), nitrate/nitrite nitrogen (NO2+NO3) and total kjeldahl nitrogen (TKN) (MDNR 2002, 2004, 2005, 2006).

Table 1. Locations and seasons of macroinvertebrate sampling conducted by the Missouri Department of Natural Resources in Hinkson Creek, Missouri, USA

Site Name	Fall 2001	Spring 2002	Fall 2003	Spring 2004	Spring 2005	Fall 2005	Spring 2006
Rogers Rd.	X	X					
Hinkson Creek Rd.	X	X	X	X	X	X	
Hwy 63 Connector				X			
Walnut Street	X	X	X	X	X	X	
Broadway St.			X	X	X	X	
Capen Park	X	X					
Rock Quarry Rd.	X	X					
Recreation Dr.					X	X	
Forum Blvd.	X	X					X
Twin Lakes	X	X					X
Scott Rd.	X	X					X

The results of the Hinkson bioassessments indicated that 13 of the 30 sampling events had a SCI score of 14 to 12 indicating conditions which are partially supportive of aquatic life (EPA 2011). Almost all of the partial-supporting values were from sites downstream of the Interstate-70 bridge within the section of the stream listed as impaired. The exceptions to that trend occurred during the fall 2001 sampling season in which the three most upstream reaches received SCI scores of 12, attributed to unusually low flows (MDNR 2002). Each of the four macroinvertebrate metrics used in the data analysis indicated better stream quality in the rural portion of the stream and degraded conditions in the urban portion, during at least one or more sampling seasons. However results were not always consistent. For example during the spring of 2004, the sampling site near Broadway Blvd had the same taxa richness as Hinkson Creek Rd, the site used as a reference site (MDNR 2004). There were fewer sensitive EPT taxa at the Broadway site and a greater variety of pollution tolerant species of Oligochaeta (aquatic worms), and Chironomidae (midges), which resulted in a higher BI value (MDNR 2004). In general, EPT richness was consistently lower in urban reaches relative to rural reaches with the least inter-annual variation of the four primary metrics. With the exception of fall 2001, none of the rural sites had less than 15 EPT taxa, and none of the urban sites had more than 13 (MDNR 2002, 2004, 2005, 2006).

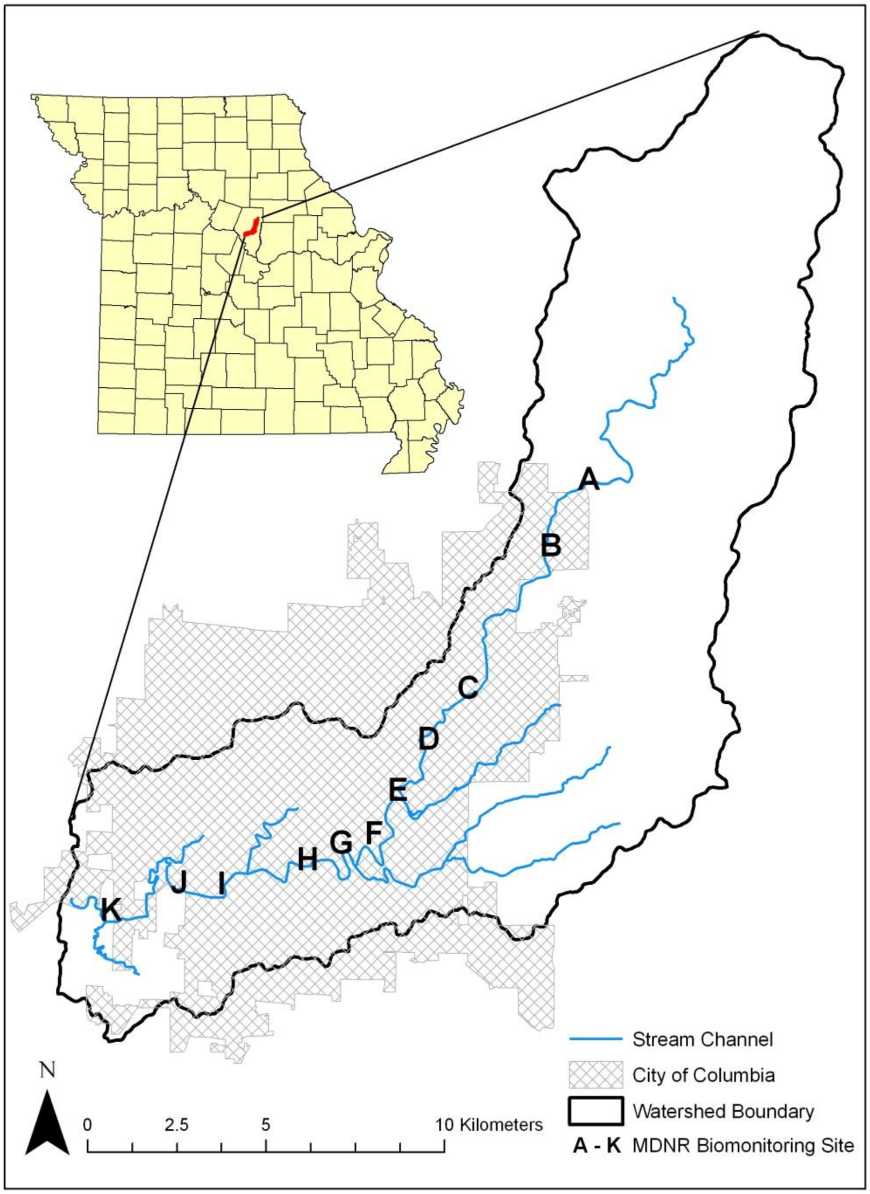

Figure 1. Map of Hinkson Creek watershed, Missouri, USA, showing the City of Columbia and the locations of biomonitoring conducted by the Missouri Department of Natural Resources (MDNR) between 2001 to 2006. Letters signify individual sites as follows: A = Rogers Rd., B = Hinkson Creek Rd., C = HW 63 Connector, D = Walnut Ave., E = Broadway Blvd., F = Capen Park, G = Rock Quarry Rd., H = Hinkson Recreation Field, I = Forum Blvd., J = Twin Lakes Park, K = Scott Blvd.

Compared to Bonne Femme Creek sites, rural Hinkson Creek sites had similar or better metric scores, while urban Hinkson Creek sites generally had lower scores indicating impairment in the urban portion of the stream. With the exception of fall 2001, average taxa richness in Bonne Femme was 4% lower than rural Hinkson Creek and 7% higher than urban Hinkson Creek sites. The EPT richness metric in Bonne Femme was on average 17% lower than rural Hinkson Creek and 54% higher than urban Hinkson Creek. The Biotic Index in Bonne Femme was 8% and 9% lower than rural and urban Hinkson Creek sites respectively (MDNR 2002, 2004, 2006). In fall 2001, taxa richness in Bonne Femme was 27% and 13% higher than rural and urban Hinkson Creek sites respectively. EPT richness was 26% and 12% higher in Bonne Femme relative to rural and urban Hinkson Creek sites respectively. Biotic Index values in Bonne Femme in fall 2001 were less than 1% lower than rural Hinkson sites, but were 5% higher than urban sites. Shannon diversity was 23% and 15% lower in rural and urban Hinkson Creek sites relative to Bonne Femme in 2001.

Turbidity was generally higher in Hinkson Creek relative to Bonne Femme Creek at almost all Hinkson Creek sites during all seasons. Chloride levels were higher in the urban portion of Hinkson Creek compared to Bonne Femme Creek, as was conductivity, particularly in the spring, attributed to deicing road salts (MDNR 2002, 2004, 2006). Chloride concentrations in Bonne Femme Creek ranged from 10.1 to <0.05 mg/l in the fall and from 24.5 to 10.4 mg/l in the spring. Chloride in Hinkson Creek during those same years ranged from 36.7 to 7.7 mg/l and from 26.7 to 17.2 mg/l in rural sites in the fall and spring respectively. Chloride in urban Hinkson Creek sites ranged from 46.7 to

10.8 mg/l in the fall and from 64.3 to 23.5 mg/l during the spring. Nutrient levels were generally comparable between streams with the exception of TKN, which was consistently higher in Hinkson.

Toxicity levels, deposited sediment, and dissolved oxygen levels were also investigated. Isolated samples of stormwater from between I-70 and Broadway Blvd (Figure 1) were toxic with insecticides and herbicides, waste oil and other petroleum by-products, heavy metals, and an industrial solvent (MDNR 2004). Stormwater testing in spring 2005 yielded similar results. However, base flow samples did not indicate toxicity and sampling during fall 2005 and spring 2006 did not find any samples to be toxic, from either base flow or stormwater (MDNR 2005, 2006). Results from dissolved oxygen monitoring, conducted during an 8 week period between July 28 and September 21, 2005, showed that low levels of DO were associated with dry periods of low flow and relatively high water temperatures (MDNR 2006). A study of deposited sediment showed that the lower urban reaches of Hinkson Creek had 10 to 64 percent higher fine sediment (particles <2mm) relative to the rural portions of the creek (MDNR 2004).

Previous studies conducted in Hinkson Creek detected an apparent impact to the aquatic biota in the urban areas (e.g. lower taxa richness, fewer sensitive organisms). Several possible stressors were implicated including sediment, chloride, and other substances carried by stormwater runoff. However, these studies stopped short of quantifying the relative importance of the various stressors, and did not yield a mechanistic explanation for the differences observed in the macroinvertebrate communities. Thus, further research was warranted to improve understanding of those mechanistic relationships.

STUDY OBJECTIVES

The overall objectives of the following research were to identify potential mechanistic relationships between macroinvertebrate community composition and the physical environment in which they live, and how these relationships are altered by the effects of urban land use. Sub-objectives included *A)* identifying differences in macroinvertebrate assemblages between rural and urban portions of the stream, in terms of taxonomic and functional trait composition, *B)* identify differences in water quality and quantity as well as differences in the stream channel environments of the rural and urban portions of the stream, and *C)* quantify relationships between the biological assemblage measures and physical variables which may be responsible for observed differences in biotic communities.

HYPOTHESES

1.) Invertebrate communities in urban stream sites will be less taxonomically diverse than rural sites.

    H_o: Taxonomic Diversity in Urban sites = Taxonomic Diversity in Rural sites

    H_a: Taxonomic Diversity in Urban sites ≠ Taxonomic Diversity in Rural sites

2.) Functional trait composition of invertebrate communities will be different in urban sites compared to rural sites.

H_o: Trait Composition in Urban sites =Trait Composition in Rural sites

H_a: Trait Composition in Urban sites ≠ Trait Composition in Rural sites

3.) Physical stream conditions between rural and urban stream sites differ.

H_o: μ(*variable x*) at rural sites = μ(*variable x*) at urban sites

H_a: μ(*variable x*) at rural sites ≠ μ(*variable x*) at urban sites

4.) Differences in invertebrate trait composition will be explained by differences in at least one physical environmental variable.

H_o: No significant regression relationships exist.

H_a: Significant regression relationships do exist.

CHAPTER II

METHODS AND MATERIALS

STUDY WATERSHED

Hinkson Creek is located in the Lower Missouri-Moreau River basin (HUC 10300102) in central Missouri and is approximately 42 km long. The Hinkson Creek Watershed (HCW) spans 231 km^2 (MDNR 2006). The 10-year average annual temperature (2000-2010) recorded at the University of Missouri Sanborn Field climate monitoring site (located within the HCW), was 13.3 °C, with an average temperature for the fall and winter (October through March) of 5.16ºC and an average temperature during the spring and summer (April through September) of 21.33ºC (Sanborn Field, Missouri Agricultural Experiment Station). The 10-year average (2000-2010) annual precipitation measured at Sanborn Field was 1108.6 mm, with an average during the fall and winter of 390.8 mm and an average during the spring and summer of 717.8 mm. Soils in the upper reaches of the watershed are Keswick-Hatton-Winnegan Association, characterized as loamy till with a well-developed clay pan. Soils in the lower portion of the watershed are of the Weller-Bardley-Clinkenbeard Association and are characterized as thin cherty clay and silty to sandy clay (USDA 2001). Land use in the upper watershed is mostly agricultural and forested and becomes increasingly urban further downstream (Table 2). Land use was determined using ArcGIS 9.3 (ESRI, Redlands CA) from the 2005

National Land Cover Dataset. Land use and location of study reaches are shown in Figure 2.

Nested-Scale Experimental Watershed Study

In November of 2008, permanent gauging stations were established at 5 points along Hinkson Creek using a nested scale experimental watershed study design (Figure 2) (Hubbart et al. 2010). The objectives of studies conducted using the experimental watershed study design included analyses of suspended sediment (Hubbart and Freeman 2010, Hubbart and Gebo 2010, Freeman 2011), monitoring and modeling the flow regime (Scollan 2011) and investigations of nutrient loading (articles in prep) among other studies. Hubbart and Freeman (2010) collected and analyzed water samples for particle size class distribution during March, 2010. They identified a sharp increase in the concentration of fine particles in an urban stream comparing pre and post-precipitation event conditions. They identified a 450% increase in the concentration of the smallest particle size class (2.06 µm). Thus, with a doubling of streamflow (1.4 m^3/s to 2.9 m^3/s), the concentration of fine sediment was more than quadrupled. This relationship can be attributed to a number of natural in-stream or overland processes, however, urban influence was indicated to potentially impact their results. Freeman (2011) found that suspended sediment concentrations appeared greater at urban sites relative to rural sites but that the difference was not statistically significant. His results also showed smaller mean particle sizes of the suspended sediment at urban sites which was attributed to both in-stream weathering processes and land use (Freeman 2011). In other research conducted in the HCW, Huang (2012) conducted a study of streambank erosion and found that bank erosion can contribute as much as 67% of suspended sediment material.

At each of the five nested study sites, an automated hydroclimate station was installed that monitors a full suite of climate variables including stream stage and temperature (Table 2). A study reach was established upstream of each hydroclimate station for the purpose of collecting macroinvertebrates and assessing habitat. The length of each reach was 20 times the average of 10 measurements of the wetted width (MDNR 2002, Rabeni et al. 1999). Where climate stations were located near bridges, the downstream end of the study reach was offset 30m upstream of the bridge to avoid localized effects of shading and channel constriction. The study reach at Site 4 was bisected by a pair of bridges. At that location the portion of the stream under the bridges was excluded and the study reach resumed 30m upstream of the bridges. Characteristics of each reach and drainage area are listed in Table 2.

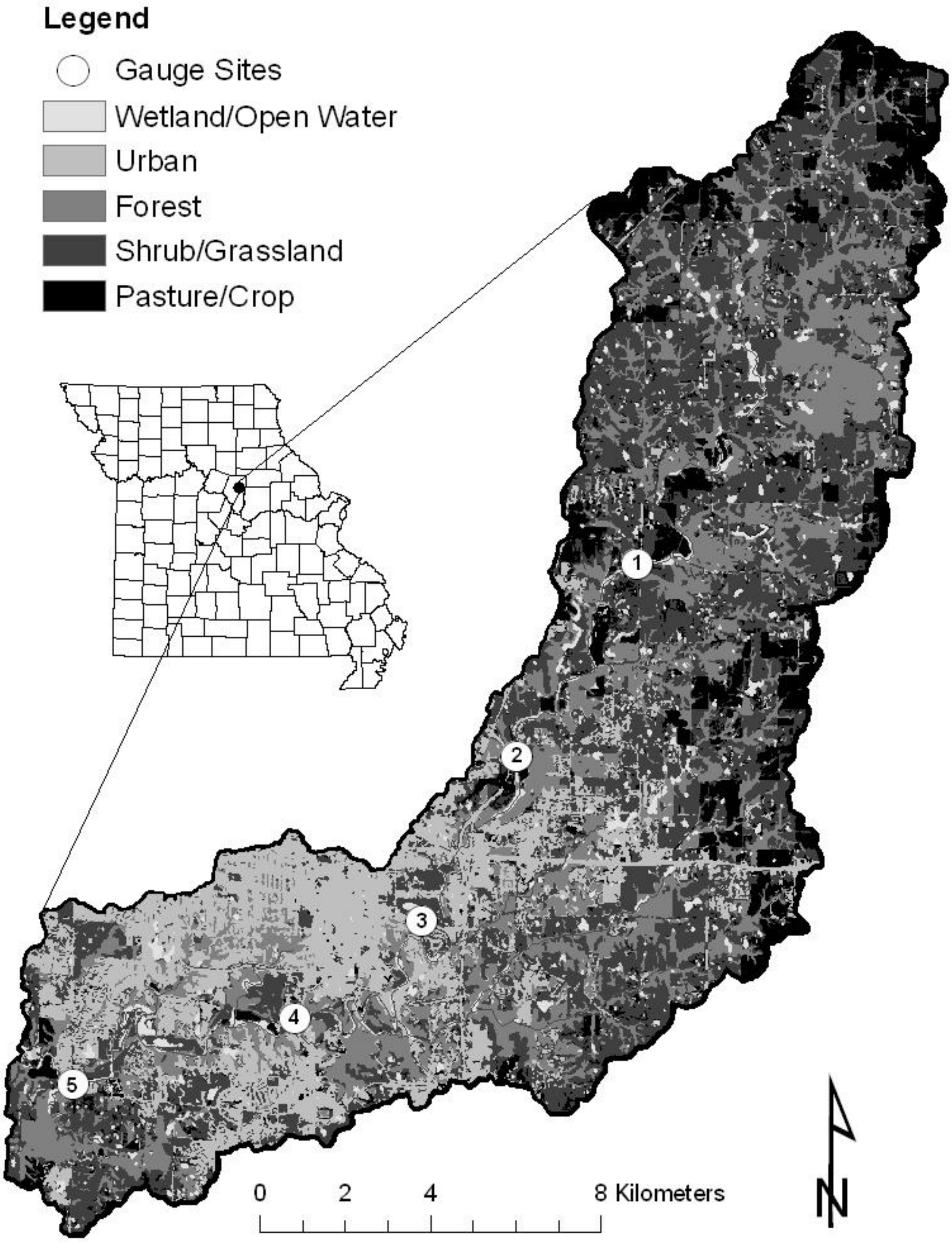

Figure 2. Map of Hinkson Creek Watershed, Missouri, USA, showing land cover and location of five hydroclimate stations and study reaches.

Table 2. Descriptive characteristics of five study reaches and variables sensed by climate stations located at each reach in Hinkson Creek, Missouri, USA.

Variable	**Site 1**	**Site 2**	**Site 3**	**Site 4**	**Site 5**
Study Reach Characteristics					
Drainage Area (km^2)	77	101	114	180	206
Reach Length (m)	244	320	268	369	282
Stream Width (m)	12.2	16.0	13.4	18.4	14.1
Slope	0.0025	0.0023	0.0025	0.0009	0.0006
Sinuosity	1.08	1.13	1.43	1.11	1.06
Percent Urban Land Cover	5	6	11	16	23
Percent Forest Land Cover	36	36	36	36	34
Percent Pasture/Crop Land Cover	55	54	49	44	39
Variables Sensed at all Climate Stations		**Sensor Model**			
Air Temperature		Campbell Scientific, Inc. Model HMP45C			
Relative Humidity		Campbell Scientific, Inc. Model HMP45C			
Water Temperature		Campbell Scientific, Inc. Model 107			
Precipitation		Texas Environmental, Inc. TE525 Tipping Bucket			
Stream Stage		Accubar© Constant Flow Bubble Gauge 56-0133			

CLIMATE

Climate measurements were recorded on an hourly basis using a Campbell Scientific, Inc. datalogger (Model CS1000). The variables and sensors used in this study are listed in Table 2. Site 4 was co-located with a United States Geological Survey (USGS) stream gauging station (USGS-06910230) that had collected data intermittently since 1966. At all sites stage was sensed using an Accubar© Constant Flow Bubbler (Sutron, Inc.). Stage values from the 4 sites were converted to discharge using Velocity-Area rating curves developed separately for each site. Each rating curve consisted of 32 cross-sectional measurements collected over a period of 2 years. Discharge from Site 4 was based on Velocity-Area rating curves developed by the USGS.

FLOW ANALYSIS

A wide variety of metrics have been used in the literature to describe hydrologic regimes and quantify associated ecologically relevant components (Olden and Poff 2003). The metrics used in this study were primarily based on those presented by Steuer et al. (2010), which were shown to be important to the composition of macroinvertebrate communities. The metrics generally follow the classifications described by Olden and Poff (2003) including metrics describing the magnitude of average conditions, magnitude of low flow conditions, magnitude of high flow conditions, frequency of small flow pulses, frequency of high flow events, duration of low flow events, duration of high flow events, and the rate of change, or “flashiness.” A total of 23 metrics were selected (Table 3). Detailed descriptions can be found in Olden and Poff (2003), McMahon et al. (2003), and Steuer et al. (2010).

Table 3. List of hydrologic condition metrics used in data analysis of hourly discharge data from Hinkson Creek, Missouri, USA.

Index Category	Metric	Units	Definition
Magnitude of average flow condition	Mean	m^3/s	Mean discharge for study period
	Median	m^3/s	Median discharge for study period
	Skewness	unitless	Skewness of hourly discharge values
	cv	unitless	Coefficient of variation of hourly values
	Med/DA	m^3/s/km^2	Median discharge divided by drainage area
Magnitude of low flow	pct_5a	m^3/s	5th percentile value of discharge
	pct_5n	unitless	5th percentile divided by median value
Magnitude of high flow condition	pct_99a	m^3/s	95th percentile value of discharge
	pct_99n	unitless	95th percentile divided by median value
	max_totfall/DA	unitless	Maximum change in value during one falling period, divided by drainage area
Frequency of small flow pulses	periodr1	events/month	Frequency of rising events where the total rise is greater than or equal to the median rise
	periodr1/DA	events/month/km^2	Periodr1 divided by drainage area
Frequency of high flow events	periodr9	events/month	Frequency of rising events where the total rise is greater than or equal to 9 times the median rise
	periodr9/DA	events/month/km^2	Periodr9 divided by drainage area
Duration of low flow events	mxl_25	hours	Duration of longest pulse less than the 25th percentile value of discharge
	mdl_25	hours	Median duration of pulses less than the 25th percentile value of discharge
Duration of high flow events	mxh_95	hours	Duration of longest pulse greater than the 95th percentile value of discharge
	mdh_95	hours	Median duration of pulses greater than the 95th percentile value of discharge
Rate of change	cumm_chg	m^3/s	Sum of the absolute value of the total rise and fall values
	cumm_chg/DA	m^3/s/km^2	Cumm_chg divided by drainage area
	Houry_pctchg	unitless	Sum of the absolute value of the percent change in hourly discharge values
	rb_flash	unitless	Version of the Richard-Baker Flashiness Index (Baker et al. 2004), sum of absolute value of total rise and fall values divided by sum of hourly flow values

Hydrologic condition metrics were calculated for the winter (Oct. 2010 – Mar. 2011) and summer (Apr. 2011 – Sept 2011) study periods using hourly discharge data. During the summer from approximately May 25 to September 9 2011, site 5 was inundated by backwatering as a result of record flooding on the Missouri River approximately 20 km downstream. As a result, discharge could not be accurately calculated from the developed rating curve for site 5 during that time. Therefore, Site 5 is omitted from the flow analysis for that season.

WATER QUALITY

Dissolved oxygen (mg/L), Chloride (mg/L), and pH were measured using a YSI Quattro Pro© (YSI Inc., Yellow Springs, OH) water quality probe four times a week during the period from October 1, 2010 to February 15, 2011, and increased to daily measurements for the remainder of the study period. Missing daily values from the early portion of the study period were estimated using linear interpolation. Grab samples were collected at the downstream end of each reach and transported to the lab for analysis of suspended sediment and Chlorophyll-a concentrations.

Two-liter grab samples were collected daily and analyzed for suspended sediment concentration and particle size distribution using a Laser In-Situ Scattering and Transmissometry (LISST-Streamside, Sequoia Scientific, Inc.) laser diffraction instrument. The LISST-Streamside is designed for monitoring sediment in shallow rivers, streams, and ponds. The device is capable of estimating particle sizes ranging from silt to very fine sand, 2.0 to 500 µm (Hubbart and Freeman, 2010; Hubbart and Gebo, 2010).

Laser diffraction instruments measure optical scattering of light over a wide range of angles, providing a multiparameter measurement corresponding to a wide range of particle sizes (Agrawal and Pottsmith 2000; Hubbart and Freeman, 2010). Total concentration is equivalent to the total volume of particles. Mean particle size was computed by the ratio of total particle area to total particle volume (Hubbart and Freeman, 2010). The LISST Streamside partitions sediment less than 500 µm in to 32 particle size classes, thereby allowing for a detailed analysis of concentrations of finer sediments. The measurements obtained using laser diffraction instruments are expressed as volumetric concentrations (µl/L), as opposed to mass concentrations. The sensor does not distinguish between organic and inorganic sediments (Agrawal and Pottsmith 2000). The particle density of sediments must be known in order to convert between volumetric and gravimetric measurements. Particle density analysis was beyond the scope of this research. Gravimetric suspended sediment concentration (SSC) was estimated using 1L grab samples collected 4 times a week. Suspended sediment was partitioned into sand (>53 µm), silt (53-1.5 µm) and clay (1.5-0.7 µm) using a standard filtration method (ASTM D3977-97). Results are expressed as mass per volume of water (mg/l).

Chlorophyll-a is produced by all photosynthetic organisms and is a useful indirect measure of primary productivity in aquatic ecosystems (Vollenweider 1974). Approximately every two weeks from April 2011 to September 2011, two separate water samples were collected in clean 1000 ml plastic jars during base flow. Samples were placed in a cooler and kept in the dark during transport to the laboratory.

In the lab, each individual 1000 ml water sample was filtered through a 47 mm diameter glass fiber filter, with a pore size of 0.7 microns (μm). If the filter clogged before the entire sample was filtered, the volume filtered was recorded and the rest of the sample discarded. The volume of sample filtered was input into Equation 1 (see below). The filters were kept frozen until extraction (ISO 1992).

To extract the pigments from the filter, each filter was placed in a separate 10ml test tube with 8ml of 95% Ethanol (ETOH). The test tubes were heated to 75°C for 5 minutes and then left in an opaque container to cool for 24 hours (Sartory and Grobbelaar 1986). Once cooled, 3 ml of ETOH was extracted with a pipette, put into a clean test tube and analyzed in a Hach DR3800 spectrophotometer. Absorbency was measured with a wavelength of 665 nm, the absorption frequency of chlorophyll, and 750 nm, to correct for turbidity (ISO 1992). The extract was then acidified using hydrochloric acid (HCl), and measured again at 665 nm and 750 nm (ISO 1992). Phaeopigment, a byproduct of chlorophyll breakdown, absorbs light at nearly the same frequency as Chlorophyll-a, therefore the sample must be acidified to destroy the chlorophyll and the remaining absorbency is subtracted from the original measurement (ISO 1992). The concentration of Chlorophyll-a is found using the following calculation (ISO 1992):

$$P_c = (A - A_a) \times 29.6 \times \frac{V_e}{V_s \times d} \qquad \text{(Eq. 1)}$$

Where $A = A_{665} - A_{750}$ and is equal to the absorbance value of the sample before acidification, $A_a = A_{665} - A_{750}$ and is the absorbance value after acidification, V_e is the volume (mL) of ETOH used in the extraction, V_s is the volume of sample (L) filtered, d is the path length of the spectrophotometer, and P_c is the concentration of Chlorophyll-a expressed in µg/L (ISO 1992).

HABITAT

The streambed is the physical space inhabited by many invertebrate organisms. Alterations to grain composition can have significant impacts on invertebrate community structure (Waters 1995). To investigate the effects of substrate composition on macroinvertebrate communities, substrate samples were collected from the location of each macroinvertebrate composite sampling location. Substrate composition was sampled using a McNeil bulk core sampler (McNeil 1960). Substrate samples associated with root mat locations were collected from the streambed directly below each rootmat. If the stream bed at any rootmat location consisted of bedrock or clay-pan, no substrate sample was collected, and the presence of such condition was recorded.

The McNeil core sampler consists of a 10 cm diameter stainless steel tube mounted in the bottom of a larger barrel which serves as a stilling well (McNeil 1960). The tube was inserted 15 cm into the streambed, supplying a sampled volume of approximately 1178.1 cm^3. The substrate was scooped out of the tube and placed in the reservoir of the McNeil sampler, and then bagged for transport to the lab. After excavation, much of the fine sediments remained in suspension within the stilling well. Neglecting to account for suspended sediment can lead to biased estimates of fine sediment composition (Bundt and Abt 2001). To estimate the amount of fine sediment in suspension, the total volume of water within the stilling well was measured and a 500ml sample was collected and processed according to the gravimetric suspended sediment analysis described previously (ASTM D3799-97). In the laboratory, sediment cores were placed in a drying oven at 105°C for 48 hours. Dried sediment samples were then sieved according to standard soil sieving methods to separate the material into gravel (>2mm), sand (2mm-53µm) and fine (<53µm) particle classes (ASTM 2009). Substrate composition was expressed as percent by weight for each size class.

A few previous bioassessment protocols included procedures for recording the presence or absence of rootmat habitat (e.g., Barbour et al. 1999). However, no widely accepted standardized method exists to quantify the volume or density of available woody rootmats. Because root habitat is targeted specifically in the macroinvertebrate collection procedure used for this study, a procedure was developed to estimate the size and biomass density of individual rootmats. Within each reach, six separate contiguous areas of roots deemed to be most representative, were selected for sampling. At Site 3 where very few submerged woody roots were present three woody rootmats were selected, and six sections of macrophyte roots were sampled. The height, width and depth of each sampled rootmat were measured in situ using a measuring tape, to obtain an estimate of volume. Then, a 15 cm by 15 cm square quadrat was placed over the approximate center of the mat and all of the material within the square was cut out and placed in a plastic bag for transport. In the laboratory, biomass density was estimated following established procedures for estimating terrestrial vegetation biomass (Chiariello et al. 1989). Each sample was placed in a pre-weighed paper bag, weighed, then placed in a drying oven at 70°C and reweighed until the weight remained constant to within <1% of the total sample weight for approximately four hours (Chiariello et al. 1989). Total drying time was approximately two days. Biomass density was expressed as (g/cm^3). It should be noted that the term rootmat density as used in this study, refers to the mass of roots per unit of volume of physical space, rather than a count of roots per unit area.

Canopy Measurements

Average canopy cover at the reach scale was measured using a transect running longitudinally through the center of each stream the entire length of the reach following the method of Platts et al. (1987) designed for 4th order streams. At 11 equidistant points spaced 10% of the reach length apart along the transect, four densiometer readings were collected; one each at the right and left bank, and in the center of the stream facing upstream and downstream (Platts et al. 1987). Following Kaufman et al (2008), a convex densiometer (Lemmon 1957) was used, which had been modified to prevent overlap from measurements taken close together. This modification consists of drawing a V on the face of the densiometer with the vertex pointing towards the viewer such that 17 line intersections exist within the V (Mulvey et al 1992). The presence or absence of canopy at each of the 17 line intersections is recorded and canopy cover is determined as the percentage of points covered by canopy (Kaufmann et al 2008).

Canopy was also measured at each macroinvertebrate sampling station using a convex densiometer (Lemmon 1957). Measurements are taken facing in each of the cardinal directions, and the four measurements are averaged to obtain the percent cover at each station (Lemmon 1957). Results are reported for each habitat type separately.

MACROINVERTEBRATE SAMPLE COLLECTION

Macroinvertebrate samples were collected using a 500-µm mesh rectangular dip net according to the macroinvertebrate stream assessment protocol used by MDNR (MDNR 2010a). Samples were collected separately from three habitat types: riffles, characterized by water flowing over coarse substrate, depositional areas, characterized by still or slow-moving water over smaller particles, and submerged woody rootmats. Samples consisted of composites comprised of up to six locations (hereinafter referred to as stations) from each habitat type distributed throughout the reach (MDNR 2010a). In some cases where the amount of habitat was insufficient to collect samples from 6 stations, all of the available habitat within the reach was sampled. Riffles were sampled with a 9” x 18” rectangular kick net (500 micron mesh) by disturbing the substrate approximately 1.5 m upstream of the net to a depth of at least 15 cm and an area of approximately 1 m^2 so that the current would carry any dislodged organisms into the net. Depositional areas were kick net sampled by disturbing the stream bed in an area of approximately 1 m^2 while sweeping the net back and forth near the feet through the suspended sediment. Rootmats were sampled from areas of streambank of approximately 1m length with good quality submerged fibrous roots. Where rootmats were sparse and spread out, multiple smaller areas of bank were sampled until the total length of bank sampled was approximately 1 m. In the fall, there were no submerged woody rootmats present at Site 3 so the roots of a stand of emergent macrophytes, primarily Water Willow (*Justicia americana*), were sampled instead. The contents of the net after each collection effort were emptied into a large plastic tub, after which larger rocks and leaves were rinsed and removed. Excess water was drained off through a fine mesh net and the remaining material was preserved in a 10% formalin solution until processing (MDNR 2010a).

In order to obtain an estimate of within-site variance of macroinvertebrate assemblage metrics, duplicate samples were collected from one habitat type from one urban site and one rural site. Due to limitations in the amount of available habitat for all habitat types at all sites, it was not possible to collect an entire, three-habitat duplicate sample from any one site, or to collect a duplicate of one habitat from all sites. During spring sample collection, a duplicate sample was collected from riffle habitats at rural Site 1 and urban Site 3. In the fall, lower stream flow resulted in less available riffle area and thus, a duplicate depositional sample was collected from rural Site 1 and urban Site 4. Collection stations from each set of duplicates, were taken in pairs, adjacent to, but not overlapping each other.

MACROINVERTEBRATE LABORATORY PROCESSING

Macroinvertebrate samples were subsampled in the laboratory using a gridded tray from which grid squares were randomly selected until a target number of organisms (+/- 10%) were collected or the entire sample was processed (MDNR 2010a). The target number for riffle samples, which can exhibit relatively higher diversity and variation among samples, is 600 organisms (MDNR 2010a). For depositional and rootmat habitats the target is 300 organisms (MDNR 2010a, McCord et al. 2007). All organisms were identified to the lowest possible level, usually genus or species. Terrestrial organisms, zooplankton and Oligochaeta worms of the family Naididae were not processed as specified by the protocol (MDNR 2010a). After the target number of organisms was reached, remaining material was inspected for any large and/or rare organisms. A large organism is a specimen that is larger than other specimens of the same taxa previously found in that sample, and which may be in better condition and may be identified to a lower taxonomic level (MDNR 2010a). A rare organism is an individual of any taxa that had not previously been removed from other grids in that sample (MDNR 2010a). Organisms found by this method are included in richness counts but are not included in other population metrics.

MACROINVERTEBRATE DATA ANALYSIS

Four primary metrics were derived from macroinvertebrate population data following the procedure used by MDNR (2010a) and Sarver et al. (1999). Metrics included Taxa richness (Taxa), EPT richness (EPT), Missouri Biotic Index (BI) and Shannon Diversity (SD) (Sarver et al. 1999). These four metrics are then used to calculate the Missouri Stream Condition Index (SCI). The SCI is a unitless score which can range from 4 to 20 and is equal to the sum of the scores of each of the four primary metrics. Those metrics are scored based on how the measured values compare to the range of values collected in reference streams from the same ecological drainage unit (Sarver

1999). For Taxa, EPT, and SD, which increase with increasing stream condition, they are given a score of 5 if the value is greater than the 75th percentile value of the reference streams, a score of 3 is given if the value is less than or equal to the 75th percentile and greater than or equal to the 50th percentile, and any value below that receives a score of 1. For BI, which decreases with increasing stream conditions, a score of 5 is given if it is below the 25th percentile of the reference streams, a 3 if it is between or equal to the 75th and 50th percentile, and a 1 if it is higher than the median (Sarver 1999). Thus the SCI is not an arbitrary value but is a comparison to reference conditions in the same region (Sarver 1999). The ranges for reference streams in the Ozark/Moreau/Loutre drainage are listed in Table 4.

Table 4. Components of the Missouri Stream Condition Index (SCI) and reference values used for assigning scores. Interpretation of SCI score criteria are listed below.

	Spring			Fall		
SCI Component Score:	**5**	**3**	**1**	**5**	**3**	**1**
Taxa Richness	>69	69 - 35	<35	>71	71 - 35	<35
EPT Richness	>15	15 - 7	<7	>13	13 - 7	<7
Missouri Biotic Index	<6.5	6.5 - 8.3	>8.3	<6.9	6.9 – 8.5	>8.5
Shannon Diversity	>2.78	2.78 – 1.39	<1.39	>3.17	3.17 – 1.59	<1.59

Interpretation	Stream Condition Index Score
Fully Biologically Supporting	16 – 20
Partially Biologically Supporting	10 – 14
Non-Biologically Supporting	4 – 8

Taxa Richness and EPT richness are simple counts of the number of all taxa present or the number of Ephemeroptera, Plecoptera, and Tricoptera taxa, respectively, present in a sample. The Missouri Biotic Index is a regional biotic index based on those developed by Hilsenhoff (1987) and Lenat (1993) and regional biotic indices used in other states. Values for individual taxa range from 0-10 (MDNR 2010b), and the average Biotic Index (BI) value for a site is computed as:

$$BI = \sum_{i=1}^{n} \frac{T_i X_i}{n} \qquad \text{(Eq. 2)}$$

Where X_i is the number of individuals within species i, T_i is the tolerance value for that species, and n is the number of organisms in the sample (Sarver et al. 1999).

Shannon’s diversity index was used to calculate the diversity of each site as a whole, and the diversity within each habitat individually. The formula used for Shannon’s diversity index (H) is given as:

$$H = -\sum_{i=1}^{S} p_i \ln p_i \qquad \text{(Eq. 3)}$$

Where S represents the number of species present in the assemblage and p_i is the relative abundance of each species (Shannon and Weaver 1963). Taken by itself, this measure represents the α-diversity, the diversity within an assemblage (Whittaker 1972), of the site or habitat it describes. It is also of interest to calculate the β-diversity, which is the diversity among assemblages within a particular landscape, and γ-diversity, the total diversity of the entire landscape (Whittaker 1972). These components of diversity are related using the exponential of Shannon diversity (D) by the function (Jost 2007):

$$D_\beta = D_\gamma / D_a \qquad \text{(Eq. 4)}$$

In this equation, D_α is the weighted average α-diversity of all assemblages being considered and is found by the following formula (Jost 2007):

$$D_\alpha = \exp\left(-w_1 \sum_{i=1}^{S} p_{i1} \ln p_{i1} - w_2 \sum_{i=1}^{S} p_{i2} \ln p_{i1} \ldots - w_N \sum_{i=1}^{S} p_{iN} \ln p_{iN}\right) \qquad \text{(Eq. 5)}$$

Here the weights (w_j) are the relative size of each sampled population. Gamma diversity is found as (Jost 2007):

$$D_\gamma = exp\left[-\sum_{i=1}^{S}(w_1 p_{i1} + \cdots + w_N p_{iN})\ln(w_1 p_{i1} + \cdots + w_N p_{iN})\right] \qquad \text{(Eq. 6)}$$

Using these methods, β will range from 1.0 if all assemblages are identical to the exponent of the Shannon diversity of the weights themselves if all assemblages are entirely unique.

In addition, community similarity was assessed using the Jaccard similarity index (Jaccard 1900). This index computes pair-wise similarity between two sites based on presence or absence of species and is found from the following formula:

$$L = \frac{a}{a+b+c} \qquad \text{(Eq. 7)}$$

Where L is the Jaccard index value, *a* is the number of shared species, *b* is the number of unique species in the first assemblage and *c* is the number of unique species in the second assemblage.

A number of ecological, morphological, and life history traits, categorized following Poff et al. (2006), were selected to describe the functional composition of the macroinvertebrate communities. Table 5 lists the variables used, definition of the trait, and the expected response to the kinds of disturbances associated with urbanization (i. e. sedimentation, habitat degradation, toxicants, etc.). Trait categories were assigned to each taxon at the genus level or higher from data and descriptions derived primarily from Poff et al. (2006), Vierra et al. (2006) and Merritt et al. (2008). Body size (Size) and respiration (Resp) were shown to be related to pollution stress or toxicity (Carlisle and Hawkins 2008, Statzner and Beche 2010). Attachment to substrate (Atch), armoring (Armr), and body shape (Shpe) were shown to be related to hydrologic disturbance or channel instability (Statzner and Beche 2010). Fast developing organisms (Devl) and bi- or multivoltinism (Volt) were linked to more frequent disturbance regimes in general (Tullos et al. 2009). Rheophily (Rheo), locomotion habit (Habi), and trophic habit (Trop) are related to habitat use and the stream food web (Merritt and Cummins 1996, Poff et al. 2006). Trait composition is expressed as percent for each category and all traits are treated as independent (e.g. Carlisle and Hawkins 2008, Tullos et al. 2009, Walters 2011).

In addition to macroinvertebrate data collected during the current research, data from previous sampling conducted by MDNR (MDNR 2002, 2004, 2006) were analyzed using the same methodology. Those data are available online and were accessed on January, 20 2011 from the Missouri Department of Natural Resources Biological Assessment Database (MDNR 2011).

Table 5. Definitions of trait variables used in analysis of macroinvertebrate assemblages and the expected response to the types of disturbance associated with urbanization.

Variable	Definition of Trait	Expected Response
Taxa	Taxa richness	Decrease
EPT	Ephemeroptera, Plecoptera, and Trichoptera taxa richness	Decrease
BI	Missouri Biotic Index of pollution tolerance	Increase
SD	Shannon Diversity Index	Decrease
SCI	Stream Condition Index, a multimetric index	Decrease
Volt	Bi- or multivoltine (>1 generation / year)	Increase
Devl	Fast development	Increase
Resp1	Respiration through tegument	Decrease
Resp2	Respiration through gills	Increase
Resp3	Aerial respiration (plastron, spiracle)	Increase
Size1	Small body size (< 9 mm)	Decrease
Size2	Medium body size (9 – 16 mm)	Decrease
Size3	Large body size (> 16 mm)	Increase
Atch	Attachment to substrate (sessile)	Decrease
Armr	Soft bodied, no armoring	Decrease
Shpe	Streamlined shape	Decrease
Rheo1	Depositional habitat only	decrease
Rheo2	Depositional and erosional habitat	Increase
Rheo3	Rheophile, erosional habitat only	Decrease
Habi1	Burrower	Increase
Habi2	Climber	Decrease
Habi3	Sprawler	Decrease
Habi4	Clinger	Decrease
Habi5	Swimmer	Increase
Trop1	Collector - gatherer	Increase
Trop2	Collector – filterer	Decrease
Trop3	Herbivore (scraper, piercer, shredder)	Increase
Trop4	Predator (piercer and engulfer)	Decrease
Trop5	Shredder (detritivore)	Decrease

STATISTICAL ANALYSIS

Descriptive statistics of all variables were calculated using the Statistical Analysis package in Excel (Microsoft Inc., Redmond WA, 2010). Descriptive statistics include mean, maximum, minimum, and standard deviation.

Climate, water quality, and habitat variables were analyzed on a seasonal basis as the winter season (October 2010 through March 2011) preceding the spring macroinvertebrate collection period, and the summer season (April 2011 through September 2011) preceding the fall collection period. Seasons correspond to a six month period preceding each macroinvertebrate sampling period. Analysis of variance (ANOVA) was used to test for differences between individual sites. Where significant differences ($\alpha < .05$) were found, post-hoc pair-wise multiple comparison tests were conducted using the Tukey method (Zar 2010). Analysis of variance and post hoc procedures were conducted using Origin$^{\copyright}$ 8.0 software (OriginLab Corporation, Northhampton MA, 2011).

Macroinvertebrate metrics and trait relative abundance comparisons were made between rural sites (Sites 1 and 2), and urban sites (Sites 3, 4 and 5). Nonparametric analysis of variance (Kruskal-Wallis) was used to test for significant differences ($\alpha < 0.05$) between urban and rural sites and between individual habitats. Kruskal-Wallis was performed using SAS (SAS Inc., Cary NC). Those metrics found to be significantly different were used for a regression analysis (SAS, PROC REG) to identify relationships with physical variables. Only significant relationships are reported ($\alpha = 0.05$). All graphs were created using Origin© software.

CHAPTER III

RESULTS

CLIMATE

Precipitation during the period of study (water year 2011) was 31% lower relative to the historic annual average at Sanborn Field for the 10 years prior to the study period (i.e. Oct. 1, 2000 – Sept. 30, 2010), which was 1071.5 mm. Average climate from the five gauging stations is shown in Figure 3 and descriptive statistics are listed in Table 6. Total annual precipitation measured during the study period ranged from 660.1 mm at HCW Site 5 to 810.8mm at HCW Site 4 with a mean of 740.4 mm (Table 6).

Daily maximum air temperature during the study period ranged from 31.6 °C at Site 1 to 33.3 °C at Site 3 with a mean of 32.4 °C (Table 6). Daily minimum air temperatures during the study period ranged from -19.7 °C at Site 1 to -16.3 °C at Site 3, with a mean of -18.1 °C (Table 6). Average water temperature ranged from 5.6 °C at Site 1 and 5.8 °C at Site 5 during the winter season and between 21.2 at Site 1 and 22.3 at Site 2 in the summer season. There were no statistically significant differences between mean air temperatures or mean water temperatures detected between sites in either season (ANOVA $p > 0.05$).

Daily mean relative humidity ranged from 68.1 % at Site 3 to 75.0 % at Site 1, with a mean for all five stations of 72.0 % (Table 6). Daily mean solar radiation (important stream water heating variable) ranged from 10.7 MJ/m^2 at Site 1 to 14.5 MJ/m^2 at Site 2 with a mean of 13.1 MJ/m^2 (Table 6).

Table 6. Summary of climate observations during the 2011 water year (Oct. 2010 – Sept. 2011) for five hydroclimate stations in the Hinkson Creek Watershed, Missouri, U.S.A

Climate Data	**Statistic**	**Site 1**	**Site 2**	**Site 3**	**Site 4**	**Site 5**
Winter (October 2010 – March 2011)						
Precipitation (mm)	Total	197.9	248.9	283.5	291.8	256.3
Air Temperature (°C)	Mean	2.8	3.4	3.9	3.6	3.6
	Max	8.4	8.4	8.3	8.2	8.4
	Min	-19.7	-19.1	-16.3	-17.0	-18.1
	St. Dev.	20.3	20.7	21.2	20.4	21.2
Water Temperature (°C)	Mean	5.6	5.7	5.7	5.7	5.8
	Max	17.4	18.4	18.1	17.7	17.5
	Min	0.0	-0.1	-0.1	-0.1	-0.1
	St. Dev.	5.4	5.7	5.7	5.6	5.6
Relative Humidity (%)	Mean	75.4	73.6	69.6	73.1	72.8
	Max	97.7	98.0	97.8	95.1	99.4
	Min	38.9	37.6	34.4	42.9	32.9
	St. Dev.	11.0	11.4	11.9	10.4	11.5
Summer (April 2011 – September 2011)						
Precipitation (mm)	Total	473.2	513.3	514.1	518.9	403.9
Air Temperature (°C)	Mean	20.2	21.0	21.6	21.1	21.1
	Max	6.4	6.5	6.6	6.5	6.6
	Min	6.3	6.7	7.1	7.1	6.9
	St. Dev.	31.6	32.3	33.4	32.3	32.3
Water Temperature (°C)	Mean	21.2	22.3	21.8	21.8	21.4
	Max	29.3	31.4	30.3	30.8	29.1
	Min	8.7	9.4	9.3	8.3	9.2
	St. Dev.	5.4	5.8	5.7	5.5	5.1
Relative Humidity (%)	Mean	74.6	71.9	66.7	70.8	71.4
	Max	93.9	93.1	90.8	90.5	91.3
	Min	40.5	36.7	32.3	36.2	35.4
	St. Dev.	9.7	10.3	10.7	9.6	9.8

DISCHARGE

Discharge statistics and hydrologic condition metrics are listed in Tables 7 and 8. During the winter study period, mean discharge ranged from 0.57 m^3/s at Site 1 to 2.24 m^3/s at Site 5. During the summer, mean discharge ranged from 0.41 m^3/s at Site 1 to 1.35 m^3/s at Site 4. Discharge could not be calculated at Site 5 during the summer due to persistent backwatering from prolonged flooding of the Missouri River, downstream (see Methods). Median discharge ranged from 0.03 m^3/s at Site 1 to 0.41 m^3/s at Site 5 during the winter (Table 7) and from 0.02 m^3/s at Site 1 to 0.27 m^3/s at Site 4 during the summer (Table 8). Skewness and coefficient of variation (CV), both measures of variation in discharge values, were highest in the headwater sites decreasing with stream distance. Skewness ranged from 9.32 at Site 1 to 5.66 at Site 5 in the winter and from 23.19 at Site 1 to 9.25 at Site 4 in the summer. CV ranged from 4.22 at Site 2 to 2.78 at Site 4 in the winter and from 6.65 at Site 1 to 3.23 at Site 4 in the summer.

The magnitude of high flows (Pct_99a) increased from 11.72 m^3/s at Site 1 to 34.36 m^3/s at Site 5 in the winter and 7.01 m^3/s at Site 1 to 19.64 m^3/s at Site 4 in the summer. However, the magnitude of high flow relative to median discharge (Pct_99n) decreased from 452.98 at Site 1 to 83.92 at Site 5 in the winter, and from 286.52 at Site 1 to 73.06 at Site 4 in the summer. The frequency of small pulses (Periodr1) was slightly higher in the headwater reaches, ranging from 133.83 events/month at Site 1 to 119.67 events/month at Site 5 in the winter, but was highest in the mid reaches in the summer ranging from 92.00 events/month at Site 2 to 50.33 events/month at Site 1. Frequency of high flow events ranged from 111.17 events/month at Site 1 to 37.00 events/month at Site

4 in the winter, but was highest at Site 3 in the summer with an average of 47.67 events/month ranging to 28.33 events/month at Site 4 in the summer. Median duration of high flow events ranged from 18.5 hours at Site 1 to 23 hours at Site 5 in the winter and from 17.5 hours at Site 1 to 14 hours at Site 3 in the summer. The duration of low flow events did not show a clear longitudinal trend during either season. Values of the Richard-Baker flashiness index (RB_Flash) were highest in the headwater reaches, ranging from 0.13 at Site 2 to 0.06 at Site 5 in the winter and from 0.21 at Site 1 to 0.12 at Site 4 in the summer.

Table 7. Hydrologic condition metrics derived from discharge data recorded during the winter at five gauging stations in Hinkson Creek, Missouri, USA. All metrics are based on hourly flow data. Detailed descriptions of metrics can be found in Steuer et al. (2011).

Winter (October 2010 - March 2011)						
Index Category	Metric	Site 1	Site 2	Site 3	Site 4	Site 5
Magnitude of average flow condition	Mean	0.57	0.81	1.20	1.76	2.24
	Median	0.03	0.14	0.20	0.30	0.41
	Skewness	9.32	8.87	7.30	6.42	5.66
	cv	4.15	4.22	3.40	2.78	2.97
	Med/DA	0.00034	0.00139	0.00175	0.00165	0.00199
Magnitude of low flow	pct_5n	0.01	0.12	0.09	0.14	0.18
	pct_5a	0.48	0.87	0.44	0.47	0.43
Magnitude of high flow condition	pct_99a	11.72	17.46	21.04	26.19	34.36
	pct_99n	452.98	124.57	105.28	88.16	83.92
	max_totfall/DA	0.10	0.11	0.07	0.08	0.02
Frequency of small flow pulses	periodr1	133.83	134.00	117.00	124.00	119.67
	periodr1/DA	1.74	1.33	1.03	0.69	0.58
Frequency of high flow events	periodr9	111.17	54.50	58.67	37.00	51.67
	periodr9/DA	1.44	0.54	0.51	0.21	0.25
Duration of low flow events	mxl_25	236	240	321	199	264
	mdl_25	7	12	38	4.5	9
Duration of high flow events	mxh_95	96	94	89	87	74
	mdh_95	18.5	19	19.5	22	23
Rate of change	cumm_chg	302.30	447.86	529.96	655.23	563.23
	cumm_chg/DA	3.93	4.43	4.65	3.64	2.73
	Hourly_pctchg	437.33	105.63	151.70	172.76	102.25
	rb_flash	0.12	0.13	0.10	0.09	0.06

Table 8. Hydrologic condition metrics derived from discharge data recorded in the summer at four gauging stations in Hinkson Creek, Missouri, USA. All metrics are based on hourly flow data. Detailed descriptions of metrics can be found in Steuer et al. (2011).

		Summer (April - September 2011)			
Index Category	Metric	Site 1	Site 2	Site 3	Site 4
Magnitude of average flow condition	Mean	0.41	0.57	0.85	1.35
	Median	0.02	0.13	0.11	0.27
	Skewness	23.19	16.21	11.71	9.25
	cv	6.65	5.56	4.29	3.23
	Med/DA	0.00032	0.00127	0.00099	0.00149
Magnitude of low flow	pct_5n	0.01	0.10	0.04	0.01
	pct_5a	0.47	0.77	0.37	0.05
Magnitude of high flow condition	pct_99a	7.01	10.48	15.64	19.64
	pct_99n	286.52	81.45	138.51	73.06
	max_totfall/DA	0.41	0.26	0.19	0.09
Frequency of small flow pulses	periodr1	50.33	92.00	91.50	66.67
	periodr1/DA	0.65	0.91	0.80	0.37
Frequency of high flow events	periodr9	37.67	34.00	47.67	28.33
	periodr9/DA	0.49	0.34	0.42	0.16
Duration of low flow events	mxl_25	630	255	55	227
	mdl_25	8	13	15.5	35
Duration of high flow events	mxh_95	47	50	50	53
	mdh_95	17.5	17	14	16.5
Rate of change	cumm_chg	375.61	477.05	600.01	693.75
	cumm_chg/DA	4.88	4.72	5.26	3.85
	Hourly_pctchg	159.53	90.45	167.56	237.84
	rb_flash	0.21	0.19	0.16	0.12

WATER QUALITY

Average daily pH ranged from 7.98 at Site 5 to 8.27 at Site 3 during the winter period and from 7.66 at Site 5 and 7.98 at Site 3 during the summer period (Table 9). There were significant differences in mean pH between sites during both seasons ($p < 0.05$). In the winter, pH at Site 3 was higher than all other sites, Site 1 was higher than Sites 2 and 4, and Site 2 was higher than Site 5. During the summer, pH at Site 3 was higher than Sites 1, 4, and 5, and Site 5 was lower than all other sites. Average dissolved oxygen (DO) concentrations ranged from 13.93 mg/L at Site 1 to 11.94 mg/L at Site 5 during the winter and from 8.10 mg/L at Site 3 to 6.14 mg/L at Site 5 (Table 9). During the winter study period, DO at Site 5 was significantly lower than all other sites ($P <.001$). During the summer period, DO concentrations at Sites 1 and 5 were both significantly lower than Sites 3, 4, and 5 ($P <.001$), but not from each other. Chloride concentrations during the winter were significantly higher ($P < 0.001$) in all three urban sites relative to the rural sites. Average concentrations during that period ranged from 50.2 mg/L at Site 1 to 127.9 mg/L at Site 5 (Table 8). During the summer period, a different pattern emerged with the lowest average concentration being 14.9 mg/L at Site 1, the highest average concentration being 55.2 at Site 2 and the concentration decreasing from Site 2 to Site 5 (Table 9). Suspended Chlorophyll-a concentrations ranged from 1.6 µg/l at Site #2 in April and Site #3 in September, to 109.5 µg/l at Site #5 in July (Table 9). In general, Chlorophyll-a concentrations were higher at the most upstream site and the most downstream site.

Table 9. Descriptive statistics of water quality measurements recorded in Hinkson Creek, Missouri, USA during water year 2011. Chloride, dissolved oxygen, and pH measurements made using a YSI Quattro© probe. Chlorophyll-a estimated from grab samples collected during base flow approximately every two weeks between April and September 2011 (n=12 samples per site).

Variable	**Statistic**	**Site #1**	**Site #2**	**Site #3**	**Site #4**	**Site #5**
October 2010 – March 2011						
Chloride Concentration (mg/L)	Mean	50.2	52.7	106.5	115.2	128.0
	Maximum	247.7	199.8	887	736.9	941.4
	Minimum	5.3	0	1.1	0	1
	St. Dev.	42.4	36.0	145.9	133.3	165.9
Dissolved Oxygen (mg/L)	Mean	13.9	13.3	13.9	13.0	11.9
	Maximum	27.5	21.4	23.5	19.3	18.4
	Minimum	6.7	6.6	7.1	6.5	4.9
	St. Dev.	4.2	3.5	3.6	3.4	3.7
pH	Mean	8.2	8.2	8.3	8.1	8.0
	Maximum	8.9	9.4	11.5	10.3	10.9
	Minimum	6.6	7.6	7.6	6.6	7.4
	St. Dev.	0.3	0.2	0.4	0.3	0.3
April 2011 – September 2011						
Chloride Concentration (mg/L)	Mean	14.9	55.2	44.1	43.1	33.4
	Maximum	39.6	244.6	154.0	103.8	80.1
	Minimum	0	0	0	0	0
	St. Dev.	8.0	52.3	31.1	22.9	14.4
Dissolved Oxygen (mg/L)	Mean	6.8	8.1	8.2	8.0	6.1
	Maximum	12.7	12.7	13.2	12.1	11.1
	Minimum	0.8	0.9	0.8	1.0	1.1
	St. Dev.	2.6	2.4	2.7	2.4	2.3
pH	Mean	7.8	7.9	8.0	7.8	7.7
	Maximum	8.9	8.6	8.8	8.5	9.3
	Minimum	6.5	6.4	6.8	6.5	6.5
	St. Dev.	0.5	0.4	0.4	0.3	0.4
Chlorophyll-a (mg/L)	Mean	23.8	10.9	7.0	9.3	40.9
	Maximum	63.9	26.9	17.2	26.2	109.5
	Minimum	2.4	1.6	1.6	1.8	4.0
	St. Dev.	20.0	7.9	5.6	7.8	36.1

Average total suspended sediment concentration measured using the LISST-Streamside ranged from 109.62 µl/L at Site 1 to 117.26 µl/L at Site 5 during the winter season, and from 73.36 µl/L at Site 3 to 88.11 at Site 5 during the summer season (Table 9). The mean particle size varied significantly, from 148.95 µm at Site 1 to 101.46 µm at Site 5 in the Winter and 71.78 µm at Site 1 to 31.37 µm at Site 5 during the Summer sampling period (Table 10). Average total suspended sediment concentrations measured using the gravimetric method ranged from 32.35 mg/L at Site 4 to 41.02 mg/L at Site 5 in the Winter sampling season and from 36.97 mg/L at Site 2 to 53.60 mg/L at Site 5 in the Summer (Table 10). There were no significant differences observed between sites (ANOVA, $\alpha = 0.05$) in terms of total sediment concentration or with each of the sand, silt, or clay size class sediment concentrations measured using the gravimetric method.

Table 10. Total concentration of suspended sediment and mean particle size measured using a LISST-Streamside laser particle diffraction analyzer and total suspended sediment concentration (SSC) measured using gravimetric sediment analysis from water samples collected in WY 2011 from Hinkson Creek, Missouri, USA.

	October – March			**April – September**		
Site 1	**Total Conc. (μl/L)**	**Mean size (μm)**	**SSC (mg/L)**	**Total Conc (μl/L)**	**Mean size (μm)**	**SSC (mg/L)**
Average	109.62	144.56	37.2	81.84	54.94	43.6
Min	0.01	2.06	0.3	2.00	8.17	2.2
Max	1920.76	356.79	1240.0	1410.27	319.87	1226.2
Std Dev	217.63	119.80	134.3	161.70	84.07	127.6
Site 2						
Average	109.61	148.95	37.2	79.33	66.45	37.0
Min	0.00	2.06	0.5	1.23	6.77	1.5
Max	2641.95	356.79	1174.0	1253.57	311.85	936.7
Std Dev	271.76	119.05	124.9	197.08	73.01	109.1
Site 3						
Average	99.80	146.66	35.4	73.36	58.76	38.3
Min	0.00	2.06	0.6	0.20	3.93	1.1
Max	1901.02	356.79	938.0	3076.12	318.38	696.9
Std Dev	242.55	113.44	105.1	269.30	63.48	93.9
Site 4						
Average	107.24	138.03	32.4	77.71	46.32	43.0
Min	0.01	2.06	0.6	2.16	8.90	2.9
Max	2031.15	356.79	844.0	1462.26	332.59	857.9
Std Dev	295.83	114.86	97.1	198.27	54.72	104.9
Site 5						
Average	88.11	101.46	41.0	117.26	31.37	53.6
Min	0.55	4.39	1.7	6.43	7.22	4.2
Max	1779.87	336.11	936.0	3562.04	289.86	685.2
Std Dev	208.04	94.94	107.8	341.87	37.60	92.7

HABITAT

Substrate composition varied between sites and between habitat types. Coarse particles (>2mm) were the dominant substrate in riffles with relative composition ranging from 80.9% at Site 3 to 55.7% at Site 5 (Table 11). Percent coarse material in depositional areas ranged from 28.4% at Site 2 to 11.1% at Site 5 but there were no statistically significant differences detected between sites ($P > 0.05$). Samples collected along the banks near rootmats were highly variable, ranging from 44.1% at Site 4 to 0.9% at Site 5 (Figure 8). Sand was the dominant substrate in depositional samples, ranging from 86.2% at Site 1 to 71.0% at Site 2, with no significant differences ($P > 0.05$) between sites. In riffles, percent sand ranged from 38.7% at Site 5 to 17.9% at Site 3. Percent sand was also highly variable in rootmat samples and ranged from 81.5% at site 1 to 48.2% at site 4. Site 5 had the highest percentage of fine sediment in all habitats. In riffles, percent of fine substrate at Site 5 was significantly higher than all other sites ($P < 0.001$). Riffle substrate ranged from 0.6% fine sediment at Site 4 to 5.2% at Site 5. In depositional areas, percent fine sediment at Site 5 was significantly higher than Sites 1, 2, and 3 ($P < 0.001$). Depositional areas ranged from 0.4% fine sediment at Site 1 to 4.1% at Site 5 (Table 11). Percent fine substrate at Site 5 rootmats was significantly higher than that at Sites 1 and 4 ($P = 0.004$). Fine sediment in rootmat samples ranged from 5.6% at Site 1 to 20.3% at Site 5. It should be noted that at Site 2, 4 out of 6 rootmat samples were unable to be collected due to the streambed at those locations being comprised of either bedrock, or exposed claypan. In addition, no samples were collected near rootmats of macrophytes samples in the fall at Site 3 due to the substrate consisting of limestone bedrock, and large cobble.

Table 11. Results of sieve analysis of streambed substrate bulk core samples collected in 2011 from macroinvertebrate collection stations at five sites in Hinkson Creek, Missouri, USA. All values are expressed as percent by weight.

Habitat Area	**Statistic**	**Site 1**	**Site 2**	**Site 3**	**Site 4**	**Site 5**
			Fine Sediment			
Riffle Stations	Mean	1.0	0.7	1.2	0.6	5.2
	Maximum	1.6	1.9	6.6	1.4	11.9
	Minimum	0.7	0.2	0.1	0.3	2.2
	Std. Dev.	0.3	0.6	2.2	0.4	3.4
Depositional Stations	Mean	0.4	0.6	1.6	2.0	4.1
	Maximum	0.5	1.2	4.0	4.5	7.8
	Minimum	0.2	0.3	0.5	0.8	1.0
	Std. Dev.	0.1	0.4	1.2	1.3	2.7
Rootmat Stations	Mean	5.6	14.5	9.6	7.6	20.3
	Maximum	13.0	16.1	18.3	15.7	29.1
	Minimum	0.9	12.8	1.5	0.9	16.9
	Std. Dev.	4.4	2.3	8.4	5.2	4.6
			Sand			
Riffle Stations	Mean	27.0	21.3	17.9	31.0	38.7
	Maximum	35.6	25.8	45.7	39.9	65.0
	Minimum	17.5	15.8	2.5	18.3	19.3
	Std. Dev.	6.3	3.3	13.2	7.3	15.8
Depositional Stations	Mean	86.2	71.0	85.5	84.9	84.1
	Maximum	94.7	86.2	98.7	96.1	98.7
	Minimum	77.7	48.8	46.0	73.2	62.0
	Std. Dev.	6.4	13.7	19.2	9.0	13.3
Rootmat Stations	Mean	81.5	48.6	68.8	48.2	78.6
	Maximum	94.7	55.0	89.0	65.0	82.5
	Minimum	40.0	42.2	39.0	24.2	70.8
	Std. Dev.	20.6	9.1	26.3	14.9	4.1
			Coarse Substrate			
Riffle Stations	Mean	72.0	77.9	80.9	68.4	55.7
	Maximum	81.8	83.7	97.3	81.1	75.5
	Minimum	63.2	73.8	47.6	59.5	25.5
	Std. Dev.	6.4	3.1	15.1	7.3	16.7
Depositional Stations	Mean	13.4	28.4	12.9	13.1	11.1
	Maximum	21.9	50.9	53.5	25.4	32.5
	Minimum	5.0	13.5	0.1	1.6	0.3
	Std. Dev.	6.4	13.7	19.8	9.5	13.2
Rootmat stations	Mean	12.9	36.6	19.9	44.1	0.9
	Maximum	52.1	41.5	59.4	74.9	2.4
	Minimum	0.1	31.7	0.0	24.1	0.1
	Std. Dev.	19.6	6.9	34.2	18.3	0.8

Average volume of rootmats declined steadily with distance downstream. However, average density was not significantly different ($P = 0.38$) between sites (Figure 9). Average volume of rootmats ranged from 58,256.3 cm^3 at Site 1 to 591.4 cm^3 at Site 3. Average biomass of rootmats ranged from 435.3 g at Site 1 to 3.5 g at site 3. Macrophytes at Site 3 had the lowest density (3.6 mg/cm^3) while Site 5 had the highest density (10.4 mg/cm^3) though there were no significant differences between sites ($P>0.05$).

RIPARIAN CANOPY

Canopy cover was not significantly different ($P > 0.05$) between study sites. Rootmat locations had the highest percentage of canopy cover except for Macrophyte rootmat stations at Site 3, which were significantly lower ($P < 0.001$) than all others. Canopy above depositional areas was higher than canopy over riffles at all sites (Table 12). There were no significant differences ($P > 0.05$) among sites in either depositional or riffle habitats.

Table 12. Canopy cover (%) measured using a spherical densiometer in five reaches in Hinkson Creek, Missouri, USA. Data were collected along a longitudinal reach transect, and at each macroinvertbrate sampling location in three habitats. Macrophyte rootmat stations at Site 3, shown in parentheses, had significantly lower canopy cover than other rootmat stations ($P < 0.001$).

Habitat Area	**Statistic**	**Site 1**	**Site 2**	**Site 3**	**Site 4**	**Site 5**
Whole Reach Transect	Mean	69.2	67.1	52.6	70.4	59.3
	Maximum	93.1	83.2	69.4	93.9	88.9
	Minimum	50.4	42.9	39.5	54.3	11.7
	Std. Dev.	15.3	12.9	10.6	14.4	27.5
Riffle Stations	Mean	60.3	55.6	37.5	54.1	51.3
	Maximum	84.4	93.2	64.9	82.3	75.0
	Minimum	37.9	15.5	10.3	24.9	37.9
	Std. Dev.	18.6	32.1	20.2	21.0	11.7
Depositional Stations	Mean	68.8	79.0	56.1	55.1	72.2
	Maximum	85.7	94.3	93.0	94.8	91.7
	Minimum	41.8	53.5	21.7	22.8	11.3
	Std. Dev.	16.5	15.6	31.4	27.5	28.5
Rootmat Stations	Mean	88.0	92.2	77.9 (47.6)	94.6	87.1
	Maximum	95.32	95.6	95.1 (63.3)	99.7	95.1
	Minimum	77.4	88.8	46.4 (33.2)	88.3	77.4
	Std. Dev.	7.8	2.4	27.3 (13.3)	4.5	7.0

MACROINVERTEBRATE COMMUNITIES

The most abundant macroinvertebrate family at all sites during the spring sampling period was Chironimidae (Diptera), which also occupied the majority of the top five genera at all sites. Table 12 lists the top five families found at each site. *Stenelmis* (Coleoptera: Elmidae), ranked among the top five genera at Sites 1 through 4, *Caenis* (Ephemeroptera: Caenidae) was among the top genera at Sites 1 through 3, and Tubificidae (Oligochaeta) was among the top taxa at Site 4. In the fall, Caenidae was the dominant family at Sites 1, while Chironimidae was most abundant at Sites 2, 3 and 5 (Table 13). Elmidae and Tubificidae were the most abundant families at Site 4, from the

1st and 2nd replicates respectively. In each season, there were a number of taxa that were unique to one site or to the urban or rural groups of sites. In the spring, there were 7, 9, 7, 8, and 6 taxa unique to sites 1 through 5 respectively, 4 taxa unique to rural sites as a group, and 11 taxa only found in urban sites. In the fall, there were 13, 13, 7, 3, and 7 taxa unique to sites 1 through 5 respectively, 6 unique to rural sites, and 8 unique to urban sites.

Table 13. Top five most abundant families in macroinvertebrate samples collected during 2011 from Hinkson Creek, Missouri, USA.

Site 1	Site 2	Site 3	Site 4	Site 5
		Spring		
Chironomidae	Chironomidae	Chironomidae	Chironomidae	Chironomidae
Elmidae	Caenidae	Elmidae	Elmidae	Tubificidae
Caenidae	Elmidae	Caenidae	Tubificidae	Caenidae
Hydropsychidae	Simuliidae	Tubificidae	Caenidae	Elmidae
Philopotamidae	Philopotamidae	Baetidae	Hydropsychidae	Stratiomyidae
		Fall		
Caenidae	Chironomidae	Chironomidae	Tubificidae	Chironomidae
Chironomidae	Caenidae	Physidae	Elmidae	Tubificidae
Planorbidae	Elmidae	Baetidae	Chironomidae	Coenagrionidae
Elmidae	Planorbidae	Planorbidae	Coenagrionidae	Baetidae
Physidae	Coenagrionidae	Elmidae	Caenidae	Caenidae

Taxa richness in the spring was highest at Site 3, where the two replicates had 82 and 84 taxa respectively (Table 14). Site 4 had the lowest taxa richness with 73. Spring EPT richness was highest at Site 1 where the two replicates had 21 and 19 EPT taxa respectively. The lowest EPT richness was recorded at Site 5 with 10 EPT taxa. Biotic index values in the spring ranged from 6.17 at Site 1 to 6.99 at Sites 4 and 5 (Table 14). Shannon diversity index values ranged from 3.27 at Site 4 to 3.43 at Site 5. Taxa richness from the fall samples ranged from 91 at Site 2 to 68 at Site 4. EPT richness was lowest at Site 1 in the fall with 11 taxa and highest at Site 2 with 25 taxa. Biotic index ranged significantly from 6.52 at Site 2 to 7.58 at Site 5, while the Shannon diversity index ranged from 2.94 at Site 1 to 3.48 at Site 5 (Table 14).

Table 14. Metric scores from macroinvertebrate samples collected in 2011 from Hinkson Creek, Missouri, USA. Metrics include taxa richness (Taxa), Ephemeroptera, Trichoptera, and Plecoptera taxa richness (EPT), Missouri biotic index (BI), Shannon diversity (SD), and Stream Condition Index (SCI), a composite index of the four previous metrics. Whole reach samples consist of samples collected from riffle, depositional, and rootmat habitat areas. Where replicates were collected, standard deviations (±) are in parentheses.

Season:	Spring					Fall				
Site:	**1**	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Whole Reach										
Taxa	80 (1.4)	80	83 (1.4)	73	74	88.5 (2.1)	91	87	67.5 (.7)	79
EPT	20 (1.4)	16	17.5 (.7)	13	10	11.5 (.7)	25	18	12 (0)	13
BI	6.20 (.04)	6.32	6.34 (.05)	6.99	6.99	6.87 (.01)	6.53	7.09	6.97 (.1)	7.58
SD	3.35 (.04)	3.37	3.35 (.06)	3.27	3.43	3.0 (.08)	3.45	3.6	3.03 (.1)	3.48
SCI	20 (0)	18	20 (0)	14	14	16 (0)	20	18	13 (1.4)	16
Riffle Habitats										
Taxa	50 (2.8)	51	54 (1.4)	43	47	63	53	60	47	47
EPT	14 (1.4)	13	13.5 (.7)	9	6	9	17	15	12	11
BI	5.59 (.03)	5.89	5.97 (.1)	6.91	6.99	7.28	5.68	6.47	6.11	7.34
SD	2.94 (.09)	2.93	2.77 (.1)	2.75	3.05	2.71	3.10	3.05	2.73	2.93
Depositional Habitats										
Taxa	27	30	33	26	37	29.5 (.7)	28	32	11.5 (.7)	32
EPT	4	3	5	2	5	4 (1.4)	6	5	0 (0)	5
BI	6.95	6.57	7.21	7.16	7.31	7.29 (0)	7.36	7.51	9.05(.05)	8.15
SD	1.66	2.58	2.84	2.38	2.9	1.93 (.26)	1.86	3.01	0.45 (.2)	2.57
Rootmat Habitats										
Taxa	53	53	52	49	45	34	45	39	36	41
EPT	12	11	12	11	9	4	10	5	4	4
BI	6.86	7.01	6.7	7.07	6.83	5.53	7.43	7.92	7.35	7.55
SD	2.98	2.81	3.12	2.95	3.0	2.35	2.8	2.67	2.62	3.0

Beta Diversity and Compositional Similarity

Table 15 shows partitioned diversity components of A) different habitats within individual sites and of B) individual habitats across all sites. In the first case, maximum D_β is approximately between 4.8 and 5.0, while in the latter case maximum D_β is approximately between 2.5 and 2.8. In case A of diversity among habitats within individual sites, values of D_β from the fall samples are higher than in the spring with the exception of Site 1. Spring D_β is on average 30% lower than the maximum, while fall D_β is 34% lower than the maximum. In case B of diversity among sites, D_β is noticeably lower than the maximum values. D_β values are on average, 29% lower in the spring than in the fall (Table 15).

Table 15. Alpha, beta, and gamma components of diversity from macroinvertebrate samples collected during 2011 at five sites in Hinkson Creek, Missouri, USA.

			Spring					**Autumn**		
A)	**Site 1**	**Site 2**	**Site 3**	**Site 4**	**Site 5**	**Site 1**	**Site 2**	**Site 3**	**Site 4**	**Site 5**
D_α	14.49	17.08	18.05	15.50	20.22	11.12	14.80	19.09	9.85	17.80
D_β	1.96	1.70	1.46	1.92	1.52	1.78	2.11	1.91	2.10	1.85
D_γ	28.31	29.07	26.20	29.80	30.74	19.82	31.19	36.46	20.56	32.91
$D_{\beta\text{-max}}$	2.74	2.60	2.51	2.50	2.72	2.82	2.82	2.83	2.65	2.78

B)	**Riffle**	**Depositional**	**Rootmat**	**Whole Reach**	**Riffle**	**Depositional**	**Rootmat**	**Whole Reach**
D_α	17.88	10.85	19.25	28.49	18.38	7.52	14.52	27.27
D_β	1.33	1.42	1.28	1.24	1.92	1.89	1.96	1.70
D_γ	23.82	15.44	24.54	35.34	35.28	14.16	28.42	46.25
$D_{\beta\text{-max}}$	5.00	4.80	4.98	4.99	4.99	4.88	4.99	4.99

Table 16 shows Jaccard similarity for pairwise comparisons between sites for individual habitats and for each reach as a whole during the spring. These results show an apparent difference between urban and rural sites wherein Sites 1 and 2 had greater Jaccard similarity to one another than they were to any other sites. The exception to this trend occurs in rootmat habitats where little variation in similarities occurs among any of the pairwise comparisons (Table 16). In both riffles and the reach as a whole, sites tended to be increasingly dissimilar with increasing longitudinal distance (Table 16). The longitudinal trend is apparent in the fall samples as well, as shown in Table 17, although less pronounced due to Site 1 being relatively more dissimilar than most other sites, particularly in riffle samples. In addition, depositional samples from Site 4 appear to be relatively dissimilar to all other sites. In general, similarity was lower in the fall relative to the spring (Tables 16 and 17).

Table 18 shows similarity between different habitats in each reach. In the spring samples from Sites 1, 2 and 5, riffle samples were most similar to depositional samples, while at Sites 3 and 4, riffles were most similar to rootmat samples. In the fall samples riffle composition was most similar to rootmats at Sites 2, 3, and 4, while Site 1 riffle and depositional samples were most similar, and Site 5 depositional samples were most similar to rootmat samples. In general similarity values were lower in the fall relative to the spring (Table 18).

Table 16. Jaccard similarity index values from pairwise comparisons between sites using macroinvertebrate samples collected during the spring of 2011 from Hinkson Creek, Missouri, USA.

			Riffle			
	Site 1a	**Site 1b**	**Site 2**	**Site 3a**	**Site 3b**	**Site 4**
Site 1b	0.60					
Site 2	0.58	0.63				
Site 3a	0.49	0.53	0.52			
Site 3b	0.50	0.52	0.53	0.57		
Site 4	0.42	0.48	0.47	0.47	0.48	
Site 5	0.48	0.47	0.48	0.55	0.52	0.48
			Depositional			
	Site 1	**Site 2**	**Site 3**	**Site 4**		
Site 2	0.50					
Site 3	0.36	0.28				
Site 4	0.30	0.31	0.28			
Site 5	0.36	0.31	0.42	0.29		
			Rootmat			
	Site 1	**Site 2**	**Site 3**	**Site 4**		
Site 2	0.44					
Site 3	0.42	0.47				
Site 4	0.43	0.46	0.50			
Site 5	0.44	0.48	0.47	0.44		
			Whole Reach			
	Site 1a	**Site 1b**	**Site 2**	**Site 3a**	**Site 3b**	**Site 4**
Site 1b	0.86					
Site 2	0.60	0.63				
Site 3a	0.54	0.55	0.57			
Site 3b	0.52	0.54	0.56	0.84		
Site 4	0.45	0.50	0.55	0.55	0.52	
Site 5	0.50	0.51	0.52	0.58	0.55	0.49

Table 17. Jaccard similarity index values from pairwise comparisons between sites using macroinvertebrate samples collected during the fall of 2011 from Hinkson Creek, Missouri, USA.

Riffle

	Site 1	**Site 2**	**Site 3**	**Site 4**
Site 2	0.29			
Site 3	0.34	0.47		
Site 4	0.25	0.37	0.49	
Site 5	0.31	0.30	0.45	0.42

Depositional

	Site 1a	**Site 1b**	**Site 2**	**Site 3**	**Site 4a**	**Site 4b**
Site 1b	0.50					
Site 2	0.39	0.35				
Site 3	0.45	0.44	0.26			
Site 4a	0.18	0.26	0.15	0.22		
Site 4b	0.21	0.23	0.06	0.23	0.44	
Site 5	0.33	0.35	0.23	0.50	0.19	0.20

Rootmat

	Site 1	**Site 2**	**Site 3**	**Site 4**
Site 2	0.26			
Site 3	0.22	0.34		
Site 4	0.21	0.36	0.37	
Site 5	0.23	0.29	0.27	0.34

Whole Reach

	Site 1a	**Site 1b**	**Site 2**	**Site 3**	**Site 4a**	**Site 4b**
Site 1b	0.93					
Site 2	0.39	0.41				
Site 3	0.39	0.41	0.55			
Site 4a	0.34	0.34	0.45	0.51		
Site 4b	0.32	0.33	0.42	0.49	0.90	
Site 5	0.40	0.40	0.39	0.49	0.46	0.44

Table 18. Jaccard similarity index values from pairwise comparisons between habitats within individual stream reaches using macroinvertebrate samples collected during the spring of 2011 from Hinkson Creek, Missouri, USA.

Spring				Fall			
Site 1							
	Riffle 1	Riffle 2	Dep.		Riffle	Dep. 1	Dep. 2
Riffle 2	0.60			Dep. 1	0.31		
Dep.	0.41	0.37		Dep. 2	0.25	0.50	
Rootmat	0.33	0.30	0.29	Rootmat	0.20	0.16	0.09
Site 2							
	Riffle	Dep.			Riffle	Dep.	
Dep.	0.37			Dep.	0.14		
Rootmat	0.34	0.27		Rootmat	0.21	0.20	
Site 3							
	Riffle 1	Riffle 2	Dep.		Riffle	Dep.	
Riffle 2	0.57			Dep.	0.24		
Dep.	0.35	0.36		Rootmat	0.29	0.27	
Rootmat	0.43	0.36	0.26				
Site 4							
	Riffle	Dep			Riffle	Dep. 1	Dep. 2
Dep.	0.28			Dep. 1	0.09		
Rootmat	0.30	0.29		Dep. 2	0.12	0.44	
				Rootmat	0.30	0.12	0.12
Site 5							
	Riffle	Dep.			Riffle	Dep.	
Dep.	0.50			Dep.	0.30		
Rootmat	0.33	0.32		Rootmat	0.24	0.31	

ANOVA and Functional Composition

Analysis of variance of spring macroinvertebrate samples showed that relative abundance of 10 different trait metrics were significantly different between urban and rural site groups ($P < 0.05$, listed in Table 19). In the fall, there were seven trait states which were significantly different ($P < 0.05$, listed in Table 19). When all samples collected in Hinkson Creek between 2001 and 2011 were pooled, there were nine traits which were significantly different between site groups in the spring and 18 traits which were significantly different in the fall. Significant P-values are listed in Table 19 according to which habitats they occurred in, and whether metric values were higher in the urban or rural portion of the stream.

Table 19. Results from analysis of variance (ANOVA) of macroinvertebrate metrics from samples collected in Hinkson Creek, Missouri USA. P-values are listed under the habitat in which significant (α = 0.05) differences were detected. Plus (+) or minus (-) symbols denote whether metric values increase or decrease, respectively, in urban sites relative to rural sites.

	Water Year 2011							
	Spring (October 2010 – March 2011)				Fall (April 2011 – September 2011)			
Metric	Reach	Riffle	Dep	Rootmat	Reach	Riffle	Dep	Rootmat
BI	0.0311 +		0.0337 +		0.0263 +		0.0491 +	
Volt		0.0297 +		0.0388 -				
Resp1							0.0123 +	
Resp2							0.0082 -	
Size1	0.0093 -		0.0081 -		0.0016 -		0.0003 -	
Size2		0.0118 -						
Size3	0.0002 +	0.0057 +	0.0008 +		0.0004 +		<0.0001 +	0.0021 +
Atch		0.0408 -						
Rheo2	0.0322 +						0.0321 +	
Rheo3	0.0402 -							
Habi1		0.0361 +						
Habi3							0.0007 -	
Habi5						0.046 +		
Trop1		0.0018 +	0.0255 -					
Trop4			0.0187 +					
	All Sampling Periods (2001 – 2011)							
	Spring (October 2010 – March 2011)				Fall (April 2011 – September 2011)			
Metric	Reach	Riffle	Dep.	Rootmat	Reach	Riffle	Dep.	Rootmat
Volt					0.0005 -	0.0072 -	0.0213 -	
Devl					0.0013 +		0.0059 +	
Resp1	0.0244 +		0.0004 +				0.049 +	
Resp2			0.0016 -		0.0477 -			
Size1	0.0173 -		0.0002 -		0.016 -			
Size2					0.0183 +	0.0233 +		
Size3	0.0012 +		<0.0001 +		0.0135 +		0.017 +	
Atch				0.0069 -		0.009 +	0.0341 -	
Shpe						0.0202+		
Rheo1			0.026 -		0.0107 -	0.0047 -	0.0146 -	
Rheo2			0.0251 +				0.0216 +	
Rheo3	0.0008 -			0.0013 -	0.0313 +	0.0074 +		
Habi1					0.0193 +		0.002 +	
Habi2				0.0351 -				
Habi3					0.0032 -	0.0093 -	0.0151 -	
Habi5					0.0464 +	0.0014 +		
Trop2					<0.0001 +	0.0006 +		
Trop4					0.0253 +			

REGRESSION ANALYSIS

Regression analysis of spring samples showed that all significant traits were strongly related to one or more of the flow metrics used in this study. Results of regression analysis are shown in Table 20. All 10 macroinvertebrate measures tested were related to at least one measure of the magnitude of average flows in at least one habitat (Table 20). With metrics derived from the reach totals from the spring samples, the strongest relationships between biotic index and flow were positive relationships to mean discharge (R^2 = 0.89) and median duration of high flow events (R^2 = 0.96). Size3 was negatively related to the coefficient of variation of flow (R^2 = 0.91). Rheo2 was negatively related to the frequency of small pulses (R^2 = 0.92) while Rheo3 was positively related (R^2 = 0.94). However Rheo3 was most strongly related positively to skewness (R^2 = 0.99). The percentage of organisms that display permanent attachment to the substrate was positively related to Skew (R^2 = 0.984) and CV (R^2 = 0.96). Size3 was negatively related to CV (R^2 = 0.834). The lower numbers of large organisms was accompanied by an increase in small and medium bodied organisms, although the difference between rural and urban Site1 was not significant. Size 2 was most strongly related positively to periodr1_DA (R^2 = 0.992) and median flow values (R^2 = 0.961). Percent burrowers was negatively related to periodr1_DA (R^2 = 0.966) and positively related to median discharge (R^2 = 0.962). Trop1 was negatively related to CV (R^2 = 0.971). Multivoltanism was strongly positively related to the magnitude of high flows (pct_99a, R^2 = 0.911). There were only five macroinvertebrate metrics from depositional habitats which were significantly related to any flow measures. Biotic Index was only significantly related to a measure of magnitude of low flows (pct_5n, $R^2 = 0.862$). Trop1 was most strongly related negatively to Median/DA ($R^2 = 0.929$). Trop4 was strongly positively related to and pct_99a ($R^2 = 0.883$). Size 1 was positively related ($R^2 = 0.989$) and Size3 negatively related ($R^2 = 0.993$) to periodr1. In Rootmats, Volt was positively related Totfall/DA ($R^2 = 0.856$) and Periodr1 ($R^2 = 0.83$).

In the fall, average biotic index at the reach scale was not significantly related to any of the flow metrics. Size3 was positively related to mean discharge ($R^2 = 0.909$) and mdl_25 ($R^2 = 0.96$) while Size1 was negatively related to those flow metrics ($R^2 = 0.962$ and 0.919 respectively) as well as RB_Flash ($R^2 = 0.937$). In Riffle samples, percent swimmers was only related to mdh_95 ($R^2 = 0.993$). In depositional samples, mdl_25 showed the strongest relationships with most significant macroinvertebrate metrics. Mdl_25 was positively related to BI ($R^2 = 0.971$), Resp1 ($R^2 = 0.973$), and Size3 ($R^2 = 0.971$) and negatively related to Resp2 ($R^2 = 0.972$), and Size1 ($R^2 = 0.976$). Rheo2 was also positively related to Mdl_25 ($R^2 = 0.957$), but was most strongly positively correlated to Mean discharge ($R^2 = 0.96$).

Table 20. Regression results from analysis of macroinvertebrate, flow, water quality and habitat data collected during the winter and spring seasons (Oct. 2010-Mar. 2011) in Hinkson Creek, Missouri, USA. Variables and significant ($P < 0.05$) R^2 values from the strongest relationships with each invertebrate metric are shown. A * denotes $P < 0.01$ and ** denotes $P < 0.001$. Symbols (+) and (-) denote whether relationship is positive or negative. Definitions of invertebrate variables (Table 5) and flow variables (Table 3) are found in the Methods.

Invertebrate Metric	**Flow Metrics**	**Water Quality Measures**	**Habitat Measures**
		Whole Reach	
BI	Mdh_95 (+) 0.962*	-	-
Size1	-	-	-
Size3	CV (-) 0.908	-	-
Rheo2	Periodr1 (-) 0.915	-	Rootmat Volume (-) 0.937*
Rheo3	Skew (+) 0.994**	-	Rootmat Volume (+) 0.825
		Riffle	
Volt	Pct_99a (+) 0.911	-	-
Size2	Periodr1/DA (+) 0.992**	Avg. Chloride (mg/L) (-) 0.85	-
Size3	CV (-) 0.834	-	-
Atch	Skew (+) 0.984**	Avg. Chloride (mg/L) (-) 0.981*	-
Habi1	Periodr1/DA (-) 0.967*	Avg. Chloride (mg/L) (+) 0.782	-
Trop1	CV (-) 0.971*	Avg. Chloride (mg/L) (+) 0.834	-
		Depositional	
BI	Pct_5n (-) 0.862	-	-
Size1	Periodr1 (+) 0.99**	Maximum Chloride (mg/L) (-) 0.921*	-
Size3	Periodr1 (-) 0.993**	Maximum Chloride (mg/L) (+) 0.977*	-
Trop1	Med/DA (-) 0.929*	Avg. Chloride (mg/L) (-) 0.773	-
Trop4	Pct_99a (+) 0.883	Avg. Chloride (mg/L) (+) 0.826	% Fine substrate (+) 0.859
		Rootmat	
Volt	Totfall/DA (+) 0.856	Maximum Chloride (mg/L) (-) 0.824	-

There were no significant relationships between macroinvertebrate metrics and water quality parameters at the level of the entire reach during either season. However relationships were evident when each habitat was considered individually. From riffle samples collected in the spring, average chloride concentration was negatively related to Atch (R^2 = 0.981), and Size2 (R^2 = 0.85), and positively related to Habi1 (R^2 = 0.782), and Trop1 (R^2 = 0.834). From depositional samples, Trop 1 was positively related to average chloride concentration (R^2 = 0.773) and maximum pH (R^2 = 0.781). Trop4 was positively related to average (R^2 = 0.826) and maximum (R^2 = 0.806) chloride concentration. Size1 was strongly negatively related to Cl_Max (R^2 = 0.921) and pH_Max (R^2 =.0983), while Size3 was positively related to those parameters (R^2 = 0.977 and 0.938 respectively). In rootmat samples, Volt3 was negatively related to Cl_Max (R^2 = 0.823).

In the fall, there were no significant relationships between macroinvertebrate trait relative abundances and water quality in riffle or depositional samples (Table 21). In rootmat samples, Size3 positively related to the minimum DO concentration (R^2 = 0.786) and negatively related to suspended sediment mean particle size (R^2 = 0.98).

With respect to habitat variables, in spring samples at the whole reach level, Rheo2 was negatively related (R^2 = 0.937) and Rheo3 was positively related (R^2 = 0.825) to average rootmat volume. In depositional samples, Trop4 was positively related to percent fine substrate (R^2 = 0.859). There was no significant relationship between riffles or rootmat samples in the spring sampling period. In the fall, the only significant relationship was from rootmat samples where Size3 was negatively related to rootmat volume ($R^2 = 0.85$).

Table 21. Regression results from analysis of macroinvertebrate, flow, water quality and habitat data collected during the summer and autumn seasons (Apr.-Sept. 2011) in Hinkson Creek, Missouri, USA. Variables and significant ($P < 0.05$) R^2 values from the strongest relationships with each invertebrate metric are shown. A * denotes $P < 0.01$ and ** denotes $P < 0.001$. Symbols (+) and (-) denote whether relationship is positive or negative. Definitions of invertebrate variables (Table 5) and flow variables (Table 3) are found in the Methods.

Invertebrate Metric	**Flow Metrics**	**Water Quality Measures**	**Habitat Measures**
		Whole Reach	
BI	-	-	% Coarse Substrate (-) 0.871
Size1	Mean (-) 0.962	-	-
Size3	Mdl_25 (+) 0.96	-	-
		Riffle	
Habi5	Mdh_95 (-) .993*	-	-
		Depositional	
BI	Mdl_25 (+) 0.971	-	-
Resp1	Mdl_25 (-) 0.973	-	-
Resp2	Mdl_25 (-) 0.972	-	-
Size1	Mdl_25 (-) 0.976	-	-
Size3	Mdl_25 (+) 0.971	-	-
Rheo2	Mean (+) 0.961	-	-
Habi3	Pct_99a (-) 0.931	-	-
		Rootmat	
Size3	Mean (+) 0.989*	Mean Particle Size (-) 0.98*	Rootmat Volume (-) 0.85

CHAPTER IV

DISCUSSION

CLIMATE

Total average precipitation in the HCW during water year (WY) 2011 (740.4 mm) was only 67% of the ten-year WY average of 1108.6 mm. The greatest difference between the current WY and the ten year average was during the winter period when total precipitation was 34% lower. During the winter, precipitation was 32% lower than the ten-year average. Average air temperature during WY 2011 (12.2 °C) was 8.3% cooler than the ten year average (13.3 °C) with the greatest difference observed during the winter when temperatures were 34% lower than average. Average summer temperature (21.0 °C) was 1.4% lower than average (12.3 °C). There were differences between precipitation, air temperature, and water temperature recorded at each site; however, differences were not significantly different ($P < 0.05$) in any of those variables. In general, average air and water temperatures and total precipitation were highest in the mid reaches, closest to the urban center (Table 5). Figure 3 shows average daily climate in the HCW.

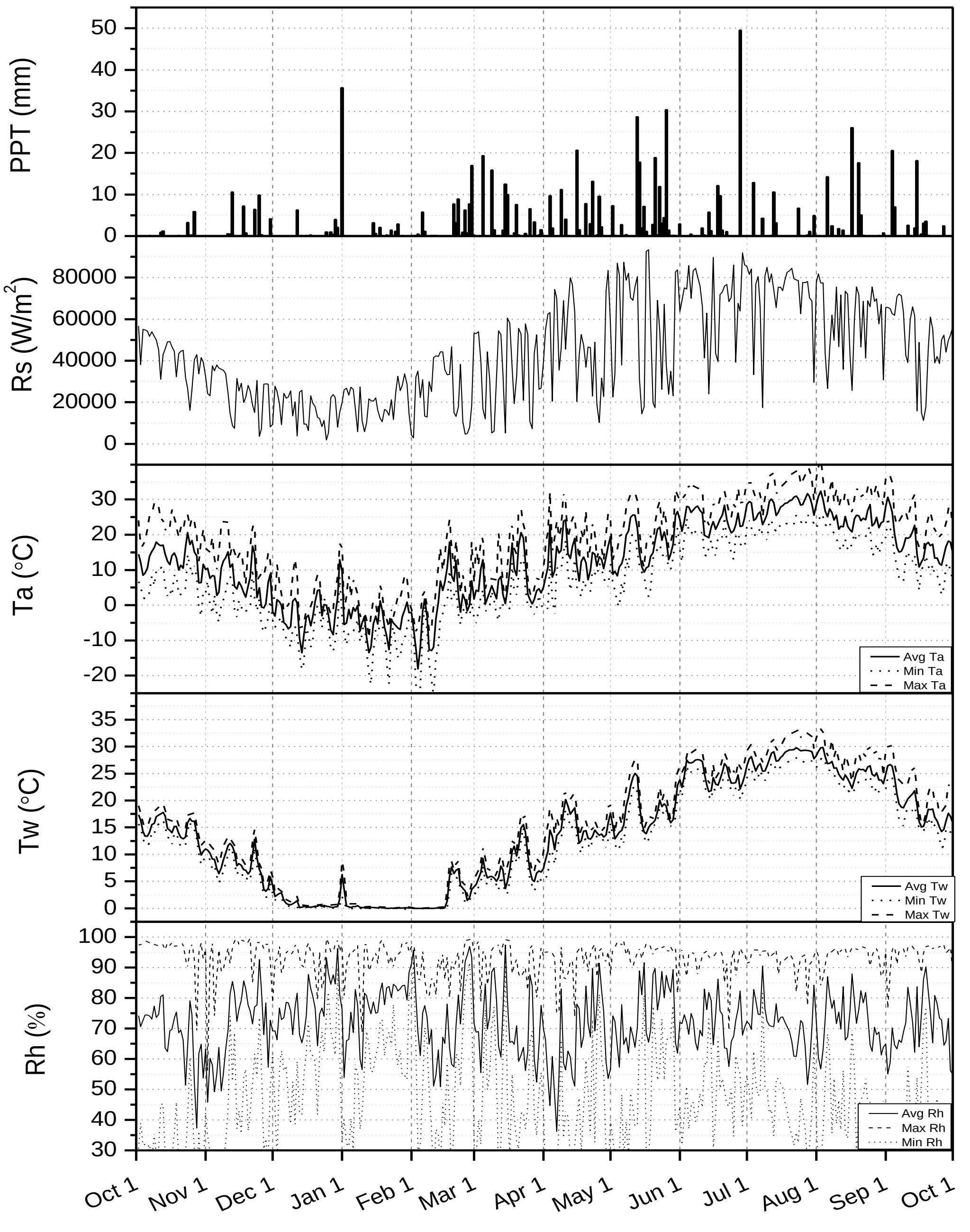

Figure 3. Average daily climate measurements from 5 climate stations for water year 2011 in Hinkson Creek Watershed, Missouri, USA. PPT = precipitation, Rs = shortwave solar radiation, Ta = air temperature, Tw = water temperature, Rh = relative humidity.

DISCHARGE

Figure 4 shows discharge at each site during water year 2011. An analysis of hydrologic condition metrics suggests that reaches in the rural headwaters were flashier than urban sites lower in the watershed. For instance, headwater reaches exhibited higher peak flows relative to base flow (pct_99n) during the current study. Discharge increased (Mean, Median, Pct_99a) in magnitude from the headwaters to the lower reaches, as did the duration of high flows (mdh_95), while frequency (Periodr9) of high flow events decreased (Figure 5). In the spring mean discharge at Site 5 was 292% higher than Site 1 while Pct_99a was 193% higher at Site 5 relative to Site 1. The measure of peak flow relative to base flow (Pct_99n) was 440% higher at Site 1 than at Site 5. Measures of flashiness of stream flow (RB_Flash, Cumm_chg/DA) decreased with stream length as well. RB_Flash was 100% higher at Site 1 than at Site 5 during the spring. In the fall, discharge statistics could not be calculated for site 5 due to the backwatering phenomenon mentioned previously, but RB_Flash at Site 4 was 79% lower than Site 1. The frequency of high flows (Periodr9) at Site 1 was 200% and 115% higher than Sites 4 and 5 respectively in the spring and 33% higher than Site 4 in the fall.

An analysis of the time series of hydrologic metrics derived from discharge at Site 4 (USGS gauging station) showed that there were no significant increases or decreases in any metric over the 45 year time period analyzed, with the exception of mean discharge, which showed an increasing trend (P = 0.03) (Figure 6). However, it should be noted that the increasing trend in mean discharge was largely influenced by two wetter than average years (2008 and 2010) that follow a 16 year period (1992 to 2007) of missing data when

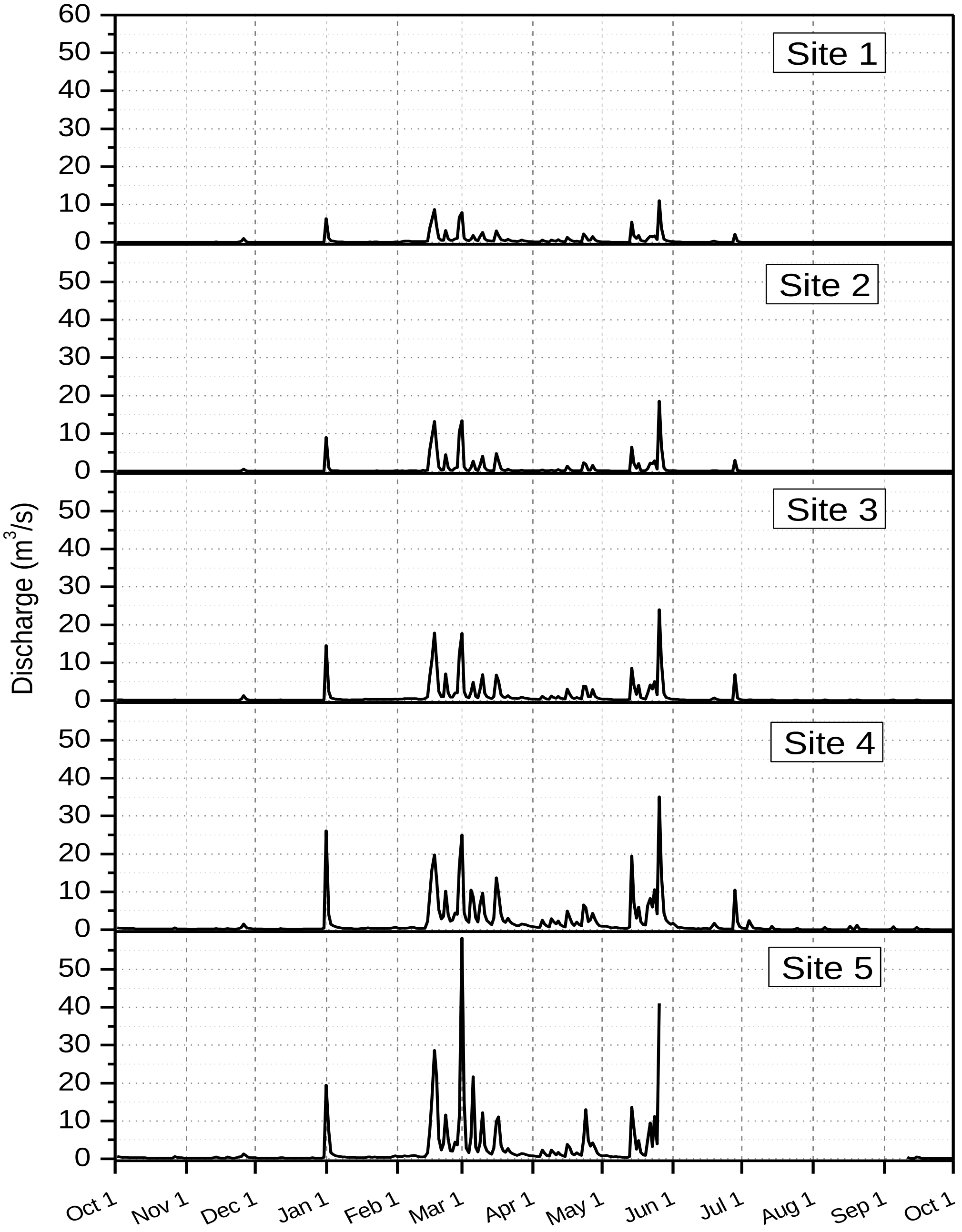

Figure 4. Discharge measured during water year 2011 at five stations located along Hinkson Creek, Missouri, USA. Data gap at Site 5 from May 25 to Sept. 5 is due to the influence of Missouri River backwatering.

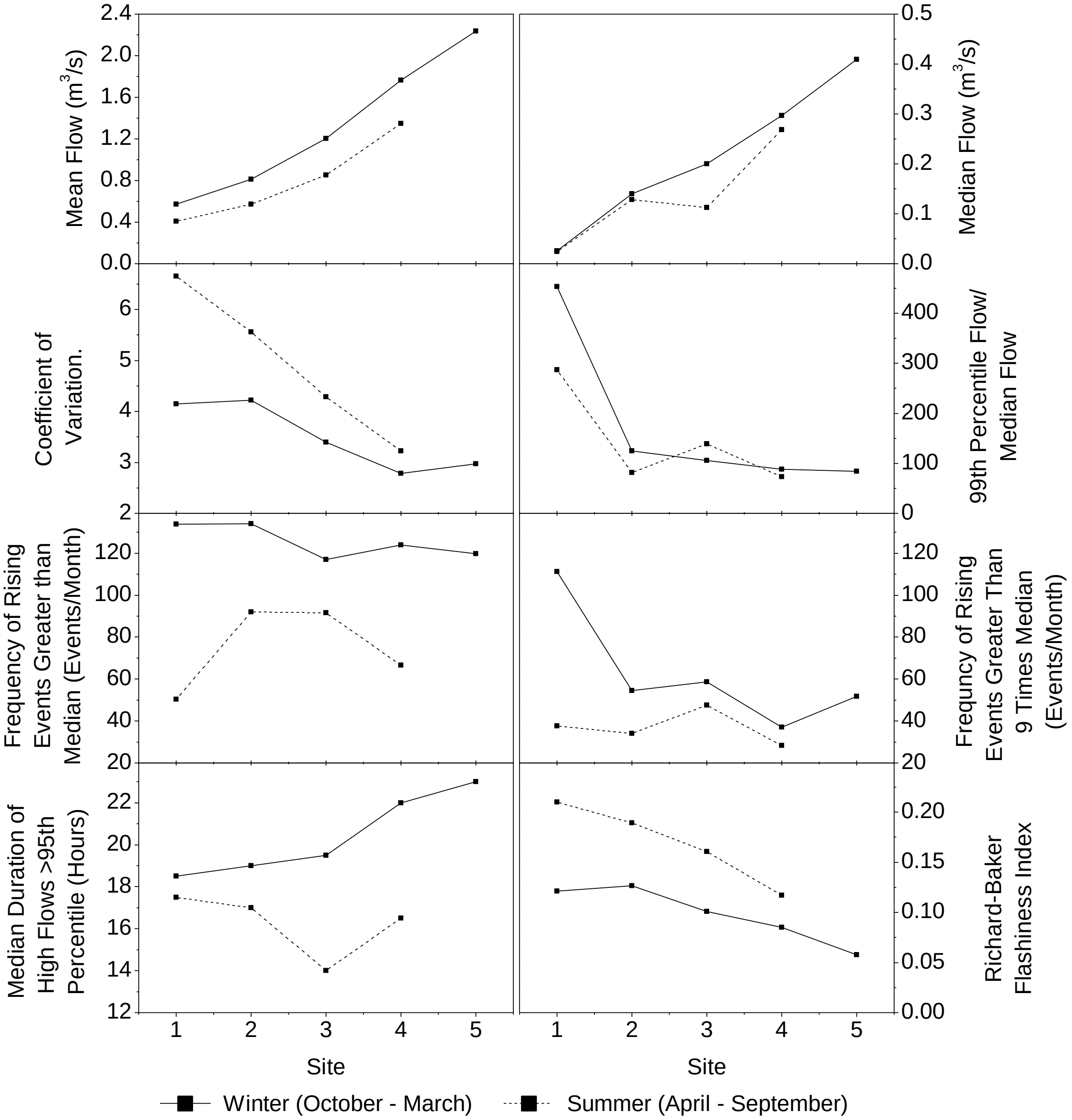

Figure 5. Hydrologic condition metrics derived from hourly discharge recorded during two seasons of water year 2011 at 5 sites on Hinkson Creek, Missouri, USA.

the gauging station was non-operational. When these two “outlier” years are removed, the increase was no longer significant (P = 0.77) (Figure 6). Increases in water yield have been found to occur following forest vegetation removal as a result of decreased evapotranspiration (Hubbart et al. 2007) and is also associated with urban development (Douglas 1976, White and Greer 2006). Thus the finding of an increasing trend in mean discharge cannot be completely ignored and continued monitoring is necessary. It should be noted that median discharge, which reflects baseflow conditions, has not changed significantly over the last 45 years, nor has the magnitude of high flows relative to median discharge (pct_99n).

Hydrologic metrics derived from discharge data collected by Stine (1910) fall within the range of values in recent years, which suggests the flow regime may have been relatively stable for at least a century, however it is difficult to say this with certainty with only one year of data from that time period. If the flow regime has not changed significantly in a century, then it is unlikely that urbanization, most of which has occurred since the beginning of the 1900s was the harbinger of the majority of any land use induced changes. It may be that the current flow regime is largely a legacy of deforestation and agricultural activity which preceded urban development in Hinkson Creek Watershed.

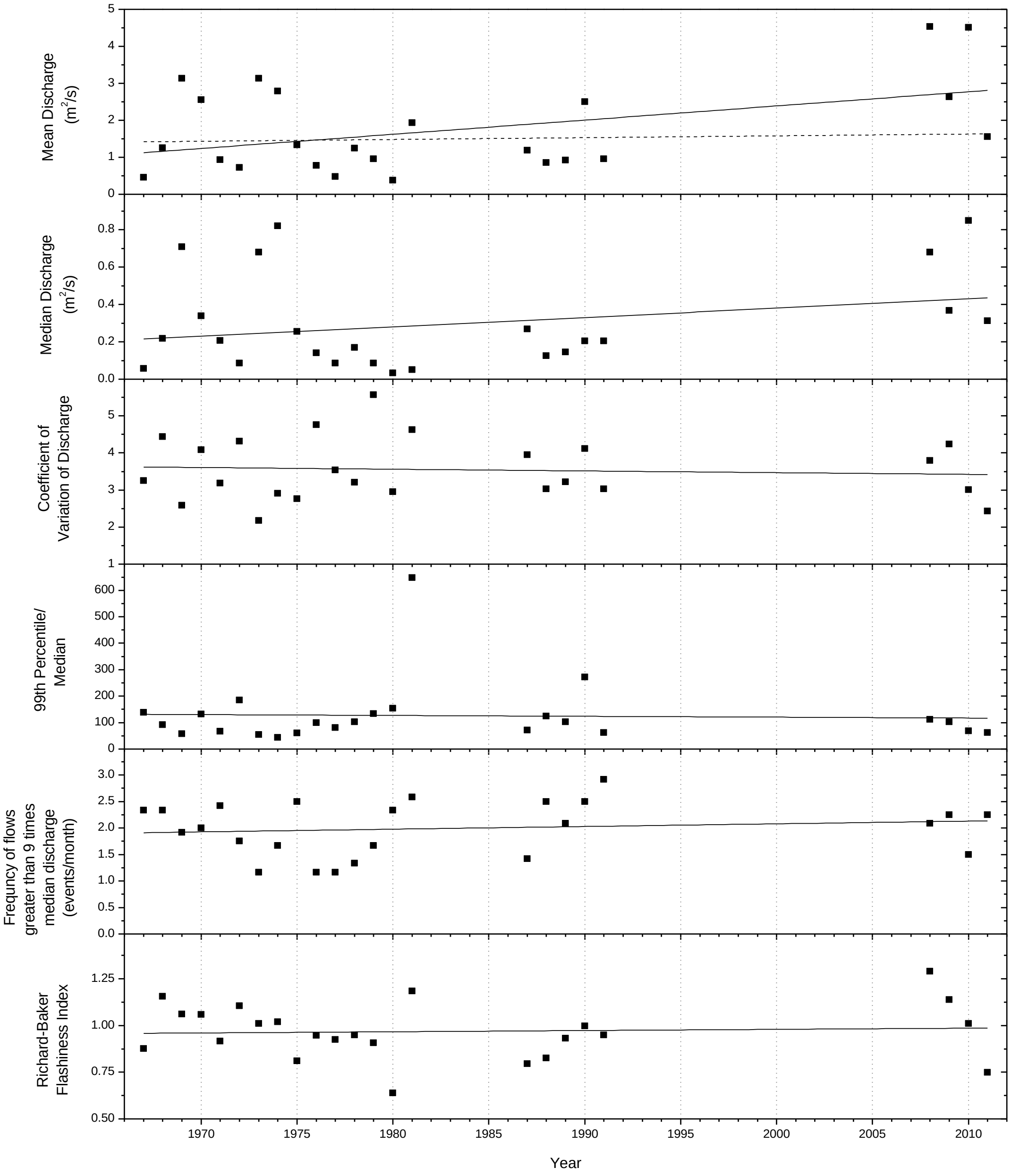

Figure 6. Time series analysis of selected hydrologic condition metrics derived from discharge data collected at a USGS gauging station (#06910230) in Hinkson Creek, Missouri, USA. Dashed line represents mean discharge with 2008 and 2010 values removed. All linear trendlines are not significant ($p > 0.05$) except Mean discharge inclusive of all years of data.

Results indicate that macroinvertebrate assemblages in the upper and lower reaches of Hinkson Creek may represent different species compositions adapted to the local conditions. Stream reaches in the lower portion of the watershed contain higher volumes of discharge and a substrate composed of smaller material than the headwaters. These conditions result in a streambed which is more likely to become mobilized (i.e. resuspended) during runoff events (Church 2002). Hydrologic disturbance and channel scouring have been noted as important sources of stress to aquatic organisms in urban streams (Coleman et al. 2011). Reaches in the headwater portion of the stream exhibited a higher range between peak flows and baseflows, and are also subject to drought stress, becoming intermittent in particularly dry summers. If the headwater reaches represented a refuge of biodiversity, and urbanization acted as a disturbance which filtered out more sensitive taxa, then species compositions in the urban reaches might be expected to be a less tolerant subset of headwater assemblages (Poff 1997). The presence of an equal or greater number of taxa unique to the urban reaches suggests that assemblages are not just a more tolerant subset of assemblages in the headwaters, but are of a different composition adapted to the conditions in the lower reaches of the stream. It is also possible that the lower baseflow in the upper half of the stream contributes to differences in assemblage composition by limiting dispersal and mobility of organisms (Boulton 2003, Dewson et al. 2007). Median discharge at Site 1 was 0.03 m^3/s in the spring and 0.02 m^3/s in the fall with a minimum value of zero during both seasons. Median discharge at Sites 4 and 5 was 900% and 1267% higher respectively, than Site 1 during the spring, and 1250% higher at Site 4 during the fall. Drier reaches of Hinkson Creek would not have enough flow to facilitate drift, which is an important mechanism of dispersal for many organisms (Resh and Rosenberg 1984), thus limiting downstream recruitment of species found in the upper reaches.

WATER QUALITY

Chloride concentrations were on average, 126% higher in urban reaches (Table 9) in the spring and at times, exceeded the EPA 230 mg/L chronic limit for chloride, defined as the four-day average concentration (USEPA 1988). In January 2011, the four day average chloride concentration was greater than 230 mg/L for a period of seven days at Site 3 and five days at Sites 4 and 5. In February, the chronic limit was exceeded for periods of 12, 14, and 15 days at Sites 3, 4 and 5 respectively. In addition, the acute chloride toxicity criterion (> 860 mg/L) was exceeded on one day at Site 3 and two days at Site 5. These results corroborate previous findings of elevated levels of chloride in Hinkson Creek and various tributaries (MDNR 2002, 2004, 2006, Allert et al. 2012). Average chloride levels from the current study are within the ranges measured during baseflow during previous years of data collection. Winter and spring baseflow chloride concentrations ranged from 19.6 (Dec. 2004) to 333 mg/l (Dec. 2005) in urban sites and from 17.2 (Mar. 2004) to 217 mg/l (Oct. 2005) in rural Hinkson Creek sites (MDNR 2002, 2004, 2005, 2006). Summer and fall baseflow chloride concentrations ranged from 10.8 (Sept. 2003) to 53.7 mg/l (Aug. 2005) in urban sites and from 7.7 (Sept. 2001) to 75.4 mg/l (Aug. 2005) in rural sites. None of the values measured during baseflow or storm events in main stem Hinkson Creek during previous studies were as high as the highest concentrations measured in the current study (941.4 mg/l, Site 5, 2/7/2011), however concentrations measured from urban drainage outfalls in 2004 were as high as 22,800 mg/l (MDNR 2004). Recently, Allert et al. (2012) found levels of chloride in water from an urban tributary to Hinkson Creek during periods of snow melt to be toxic to *Ceriodaphnia dubia* in laboratory tests. In recent years Chloride has been recognized as an important pollutant in urban streams in northern latitudes (Kaushal et al. 2005, Daley et al. 2009). In the current study, maximum chloride concentrations in the winter were significantly related to a decrease in small bodied organisms ($R^2 = 0.92$) and an increase in large bodied organisms ($R^2 = 0.97$). Smaller organisms are expected to be more sensitive to pollution due to their higher surface area to volume ratio (Statzner and Beche 2010). Further monitoring of chloride is warranted to better understand the role it plays in the stream environment.

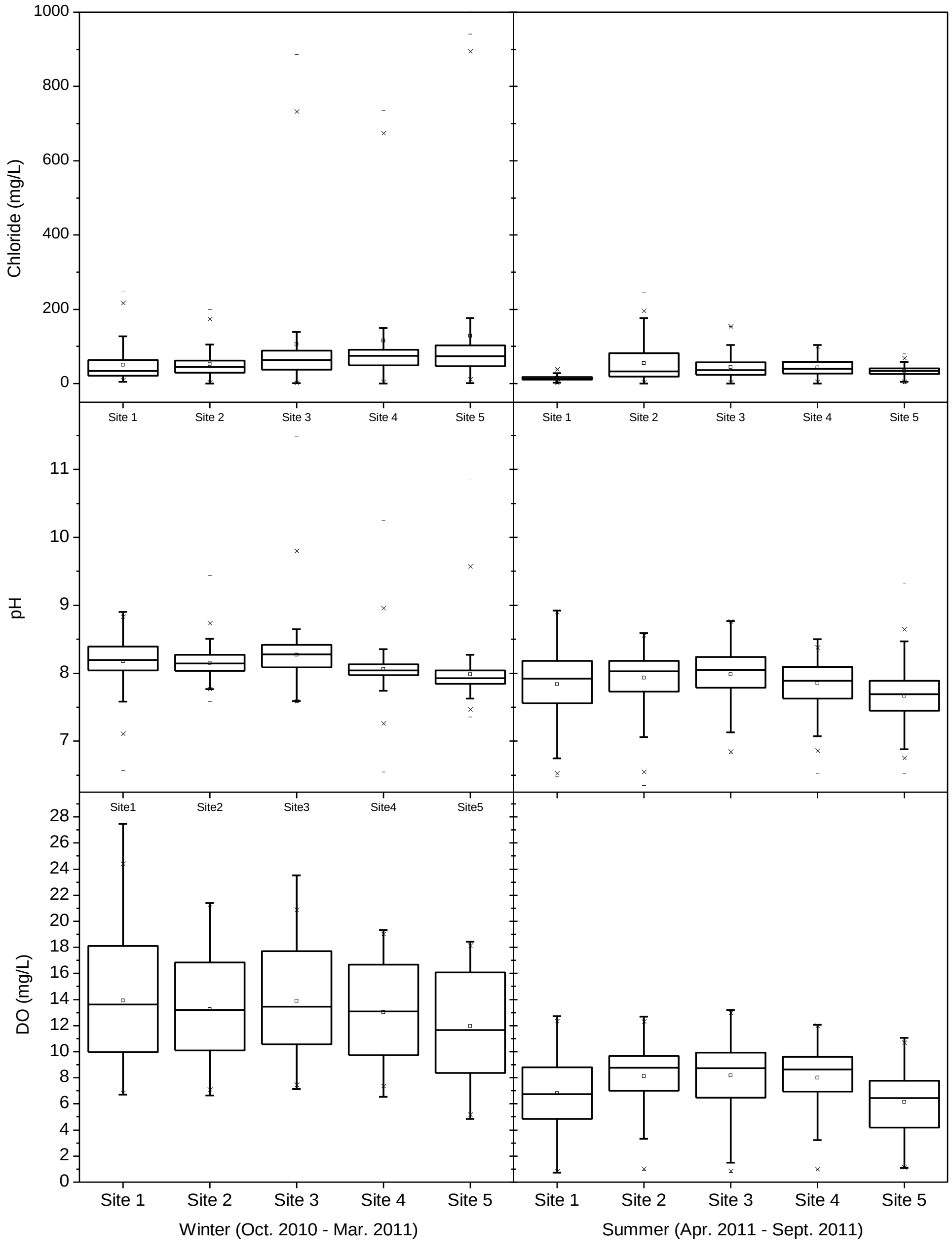

Figure 7. Box plots of chloride concentration, pH, and dissolved oxygen (DO) measured in Hinkson Creek, Missouri, USA.

Other water quality variables which were found to be significantly different ($P < 0.05$) between sites included pH, dissolved oxygen, and mean particle size of suspended sediment. Mean pH values showed relatively little variation between sites (Figure 7). The lowest values for mean pH, recorded at Site 5 were 8.0 in the winter and 7.7 in the summer and were only 3.8% and 3.9% lower than the highest mean values, recorded at Site 3, in the winter and summer respectively (Table 8). These figures are within hospitable ranges for most organisms (Smith 2001, Giller and Malmquist 1998). The pH values measured in the current study are also similar to those measured during previous studies of Hinkson Creek, which ranged from 6.9 (Jan. 2004) to 8.8 (March 2002) over all measurements (MDNR 2002, 2004, 2005, 2006). The finding of significant differences (ANOVA, $P < 0.05$) is likely also attributable to the very small amount of variation within sites; standard deviations were less than ±0.5 (Table 9, Figure 7).

Dissolved oxygen may play a role in structuring macroinvertebrate assemblages, but the pattern of average values suggests that differences between sites cannot be attributed to the influence of urbanization. The lowest DO readings were recorded at Site 1 and Site 5 (Figure 7), and are likely the result of higher amounts of respiration associated with algal growth (Mason 2002). Dissolved oxygen concentrations were lower during the summer (Figure 7) when solar radiation and photosynthesis are the highest. Mean DO concentrations at Sites 1 and 5 were 21% and 34% lower respectively, than Site 3 which had the highest mean concentration of 8.2 mg/L. Average DO levels in the Summer could be explained by average Chlorophyll-a concentrations ($R^2 = 0.958$, $P = 0.003$). In addition, water at sites 1 and 5 was relatively still during the summer, creating conditions favorable to algal growth (Dewson et al. 2007). Site 1 had very low flows and consisted of a series of relatively still, isolated pools, while Site 5 had backwatering from flooding on the Missouri River. The distribution of dissolved oxygen levels is similar to results from dissolved oxygen data collected by MDNR in the fall of 2001, another relatively dry year, in which values in the two headwater sites were lowest (5.8 and 6.8 mg/l), followed by the most downstream sites (7.0 mg/l), while sites in the middle reaches were highest (9.2 mg/l, Rock Quarry Rd.) (MDNR 2002). When data loggers were deployed in the stream to measure DO between July 28, 2005 and September 21, 2005, recorded concentrations ranged from 2 to 12 mg/l with the lowest concentrations occurring during periods of low precipitation when flow was reduced and water temperatures were the warmest (approximately 25 to 30 °C) (MDNR 2006).

Previous chlorophyll data do not exist with which to compare the current results from most of the study sites, with the exception of Site 4. Perkins (1995) measured suspended Chlorophyll-a concentrations from March 1 to May 22, 1989 at a spot located just upstream from the current Site 4 reach. Perkins (1995) reported concentrations during baseflow conditions ranged from 14.0 μg/l to 1.8 μg/l which are similar to values recorded during April and May 2011,which ranged from 10.6 to 2.0 μg/l at Site 4, and at from 15.9 to 1.6 μg/l all sites as a whole. Parris (2000) measured suspended chlorophyll-a concentrations at the same site as Perkins (1995) between February 6, 1995 and January 31, 1996 and recorded values ranging from 0.2 to 58 μg/l. Average baseflow concentrations were 2, 9, and 1 μg/l in the spring, summer and fall respectively (Parris 2000). The average chlorophyll-a concentration at Site 4 during the current study was 9.3

μg/l. Samples collected during the current study in the equivalent summer time period defined by Parris (May 7 to September 8) averaged 12.9 μg/l (n = 7), which is 30% higher than the 1995 values. The difference in average concentrations may be attributable to the difference in sampling frequencies between the current study (twice monthly) and Parris (2000, daily). Differences in chlorophyll-a concentration may also be related to differences in average discharge which were 62% lower in 2011 (1.55 m^3/s) relative to 1995 (4 m^3/s) (Parris 2000).

Suspended sediment concentrations from the current study are similar to those observed by Fraley et al (2009) in an urbanizing watershed in Pennsylvania. Sediment concentrations in stormwater samples ranged from 20 mg/L to 850 mg/L in that study (Fraley et al 2009). Diehl and Wolfe (2010) measured suspended sediment in two biological reference streams in Tennessee and recorded values which ranged from 2 mg/L during low flows up to 2000 mg/L during high flow events. Total volumetric suspended sediment concentrations during the winter study period were on average 16.5% higher than total concentrations measured during the summer study period (Table 9). However, total gravimetric suspended sediment concentrations increased 17.6% on average, between the winter and summer study periods. These opposing trends of increasing mass and decreasing volume suggest that suspended sediment particle density was higher during the summer. Lower particle density during the winter may be explained by seasonal inputs of organic matter carried to the stream by surface runoff during winter and spring runoff events (Allan and Castillo 2007). The finding of smaller mean particle sizes of suspended sediment in the lower reaches of Hinkson Creek supports previous findings by Freeman (2011). Freeman (2011) found no significant differences in sediment concentration between the monitoring sites but mean particle size was significantly smaller in the lower, urban sites. Differences in particle size were attributed to a combination of finer sediment originating from urban land cover, and in-stream erosion of sediment particles. Another explanation for smaller particles could be a decrease in stream competence as the slope of the channel decreases in lower reaches (Church 2002). As the slope decreases (Table 2), the energy of the stream decreases, and the ability to transport larger particles is reduced (Church 2002). These portions of the stream would also experience greater sediment deposition, which is reflected by the increase in fine sediment in the substrates of both riffles and pools (Figure 8). The increase in fine sediment in the lower reaches of the stream was accompanied by significantly greater percentages of burrowing organisms and collector-gatherers detected in riffles in the spring.

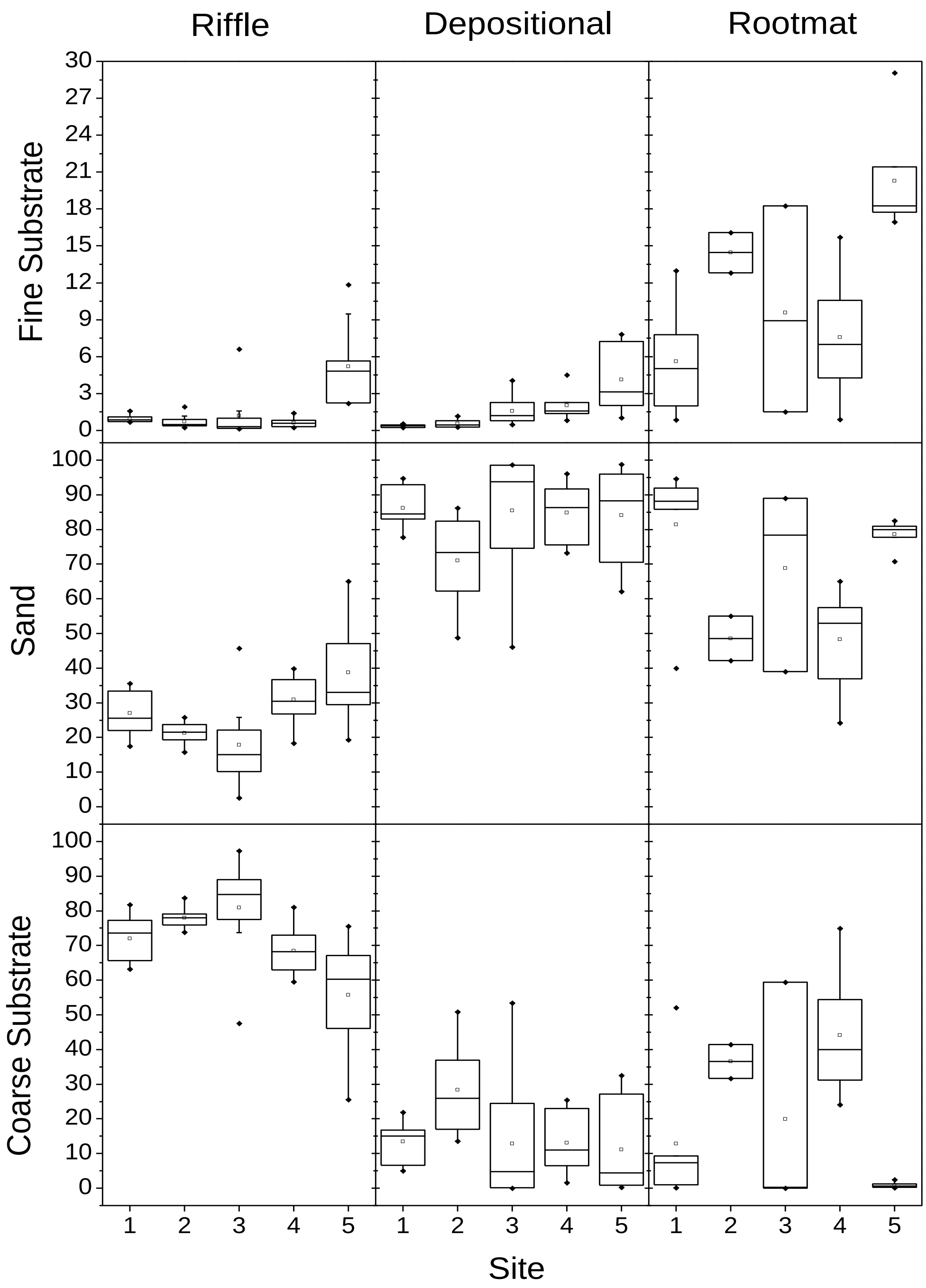

Figure 8. Box plots showing percentages of substrate composition from bulk core samples collected from 5 macroinvertbrate study reaches in Hinkson Creek, Missouri, USA in 2011.

Southwell (1911) commented on the high amount of suspended sediment in Hinkson Creek during high flow events; however he did not quantify sediment concentration. That comment however, implies that high concentrations of suspended sediment have been present in this stream for over a century. Thus, the current suspended sediment regime may be a legacy of early deforestation and agricultural land use in the watershed. In a description of Missouri soils, Baver (1935) reported that topsoils in Boone County and elsewhere in central Missouri had been eroded by as much as one-half of pre-settlement depths. Upland soil erosion in agricultural watersheds leads to downstream deposition in floodplains and as bedload (Jackson et al. 2005). It is estimated that the effects of excess sediment supply can persist for millennia as floodplain streambanks erode and the sediment is transported as suspended sediment and bedload (Jackson et al. 2005). Land-use related sediment inputs likely preceded most of the urban development in this watershed. Ultimately, it is likely that historic forest clearing and agricultural activity, since the mid-1800s, created a set of stressors from which aquatic biota have not recovered (Harding et al. 1998) and which today persist in the context of urban development. More research is needed to quantify the relative effects of current and historic land use on stream ecosystems.

HABITAT

A notable difference between upstream and downstream reaches in the current work was in the quantity and quality of woody rootmat habitat (Figure 9). Rootmats were much smaller in lower reaches, and tended to be composed of coarser roots with fewer fine roots. There are several possible explanations for the decline in rootmat quality and availability. One possible explanation could be that rootmats are degraded in the lower reaches as a result of scouring and abrasion during high flow events. Wood (1999) noted that rootmats tend to degrade over time after being exposed by bank erosion, resulting in a reduction of fine rootlets. Geomorphology may also play a role as the lower reaches of the stream tend to be more incised, and have been since the early 1900s (Tarr 1924). Greater incision may lead to the normal wetted portion of the channel being below the rooting depth of trees higher up on the bank (Hupp 1999). Toledo and Kauffman (2001) compared roots of riparian vegetation in incised and unincised streams and found that total mean root biomass was approximately twice as great in unincised streams. They also found that root biomass decreased more rapidly with depth from the soils surface in incised channels relative to unincised channels (Toledo and Kauffman 2001).

Despite a 98% reduction of rootmat volume between the Site 1 and Site 5 reaches during the current study (Figure 9), macroinvertebrate assemblages in these habitats showed the fewest numbers of significant differences between urban and rural reaches, in both the 2011 samples and in the pooled set of samples from all years. This finding is counter to that found by Wood (1999) who studied the structural composition of submerged rootmats and the associated insect communities, and found that more structurally diverse rootmats contained higher taxa richness and abundance. In most sites in the current study, Taxa and EPT richness were as high or higher in rootmats as they were in riffles, which suggests that rootmats are a very important refuge for biodiversity, even in areas where the rootmats are smaller and more degraded. Wood and Sites (2002) found no significant differences in taxa richness or diversity between riffle, pool, and rootmat mesohabitats, but found that insect abundance was higher in rootmats. The macroinvertebrate sampling methods used in the current study do not measure true abundance so it is possible that an existing difference in abundances was not detected. Roy et al. (2003b) also found that invertebrate richness and diversity in streambank habitats increased in areas where riffles were impacted by sediment. The authors of that study concluded that streambank habitat, which includes rootmats, acts as a refuge for facultative species able to migrate from other habitats as a response to disturbance (Roy et al 2003b). Sudduth and Meyer (2006) studied the ecological responses to urban stream restoration using bioengineering techniques such as geotextile fabrics and vegetation plantings. They found that invertebrate abundance and biomass were higher on organic habitats, such as roots and wood, than on inorganic habitats such as rock sand and mud (Sudduth and Meyer 2006). The authors concluded that there was a positive response to the increase in organic habitats following restoration activity but noted that in all the urban sites, altered hydrology and water quality resulting from urban land use limited the success of such small-scale projects (Sudduth and Meyer 2006). Given that rootmats appear to be a particularly important habitat, a great deal of research is warranted to explore the role that rootmats play in stream ecosystems.

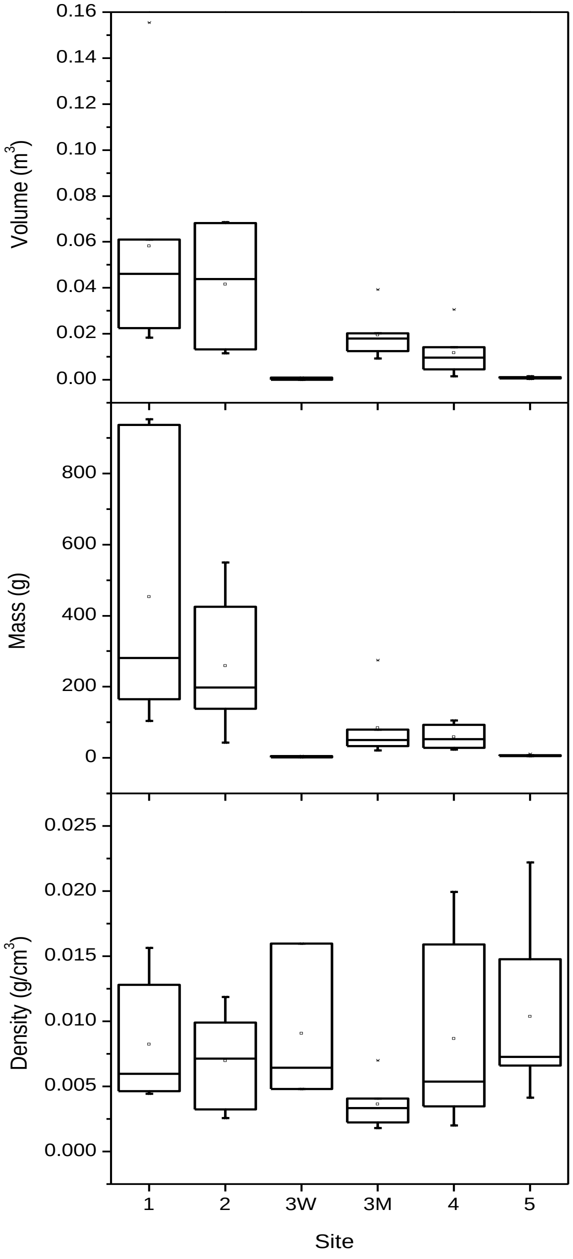

Figure 9. Box plots of woody rootmat volume, weight, and biomass density from samples collected in five study reaches in Hinkson Creek, Missouri, USA. 3W and 3M denote woody and macrophyte roots respectively, collected from Site 3.

MACROINVERTEBRATE COMMUNITY COMPOSITION

Analysis of macroinvertebrate assemblages in Hinkson Creek show differences between sites which can be attributed to both urban stressors and differences associated with longitudinal hydrogeochemical gradients of the creek. In general, the response of conventional macroinvertebrate metrics to urbanization was mixed, with few significant differences between urban and rural groups of sites. In the spring, the maximum Taxa richness was 83 at Site 3, while the minimum was 73 at Site 4, both of which are in the urban portion of the stream (Table 14). In the fall, Site 1 had the lowest EPT richness, which was 117% lower than Site 2, which had the highest EPT richness. Site 1 also had the lowest Shannon diversity of 3.0, which was 20% lower than Site 3, which had the highest, with 3.6. Reduced values of diversity and EPT richness values at Site 1 in the fall were consistent with the effects of drought stress caused by the reduction of available submerged habitat (Dewson et al. 2007). In a review of literature related to the biotic effects of decreased flow, Dewson et al. (2007) found a majority of studies reported a decrease in invertebrate density and richness following a low-flow disturbance.

Figures 10 through 13 show taxonomic composition at the Class and Order level during each season. All of the samples collected during the spring are dominated by the class Insecta, followed by Oligochaeta (Figure 10). Greater diversity is apparent in the fall, as evidenced by increased percentages of Gastropoda and Turbellaria, although Insecta remains dominant in most samples (Figure 11). One exception to this is the depositional habitat sample from Site 4 which is composed of 93% Oligochaeta. Diptera is the dominant Order at all sites during the spring (Figure 12). Samples in the spring show greater diversity at the Order level with increasing relative abundance of Ephemeroptera, particularly in riffles and in depositional samples from Sites 1 and 2. Rootmat communities in the fall contain a greater relative abundance of Basommatophera and Odonata (Table 13).

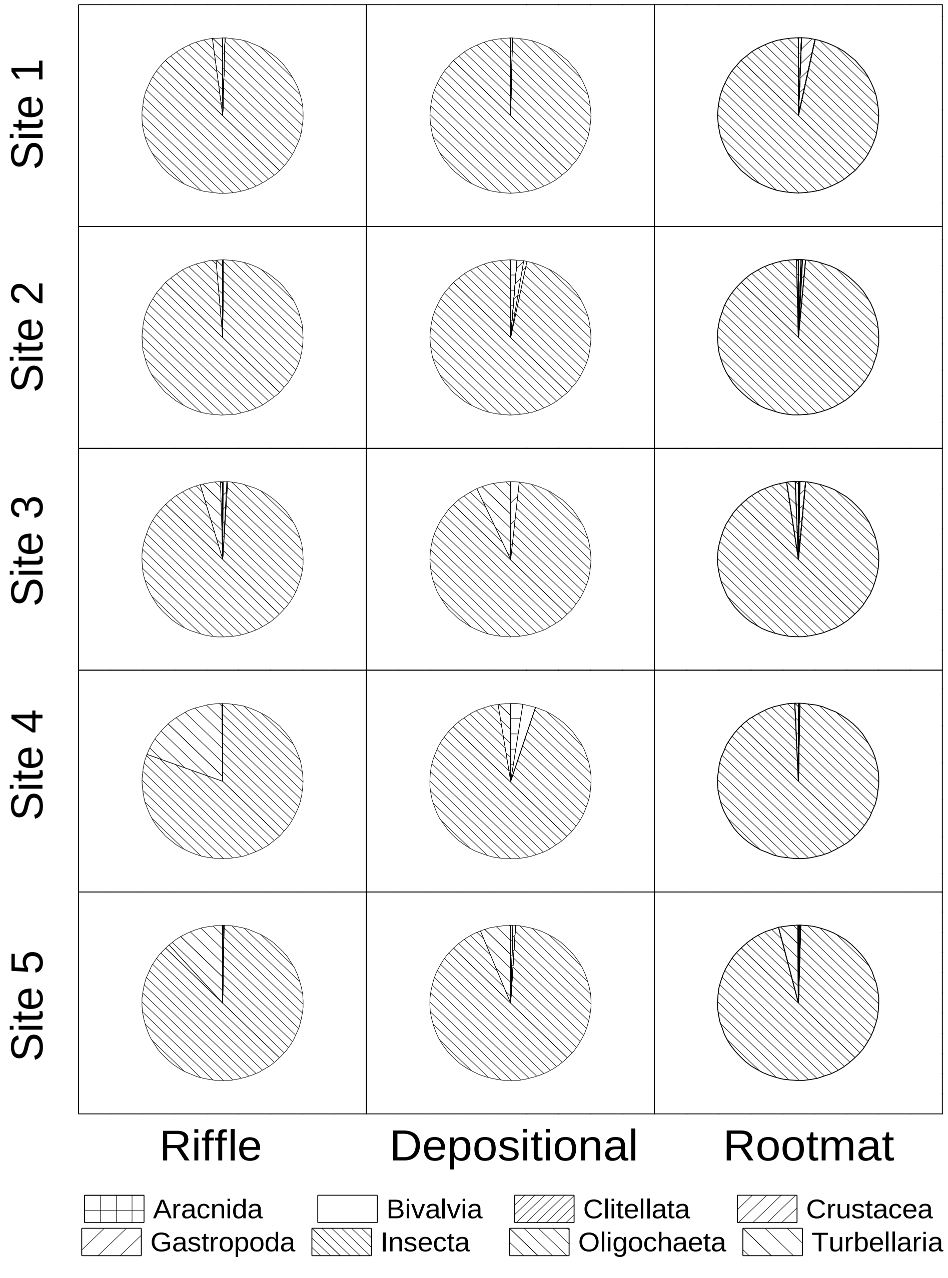

Figure 10. Relative abundance of macroinvertebrate organisms at the Class level from samples collected in spring of 2011 in Hinkson Creek, Missouri, USA.

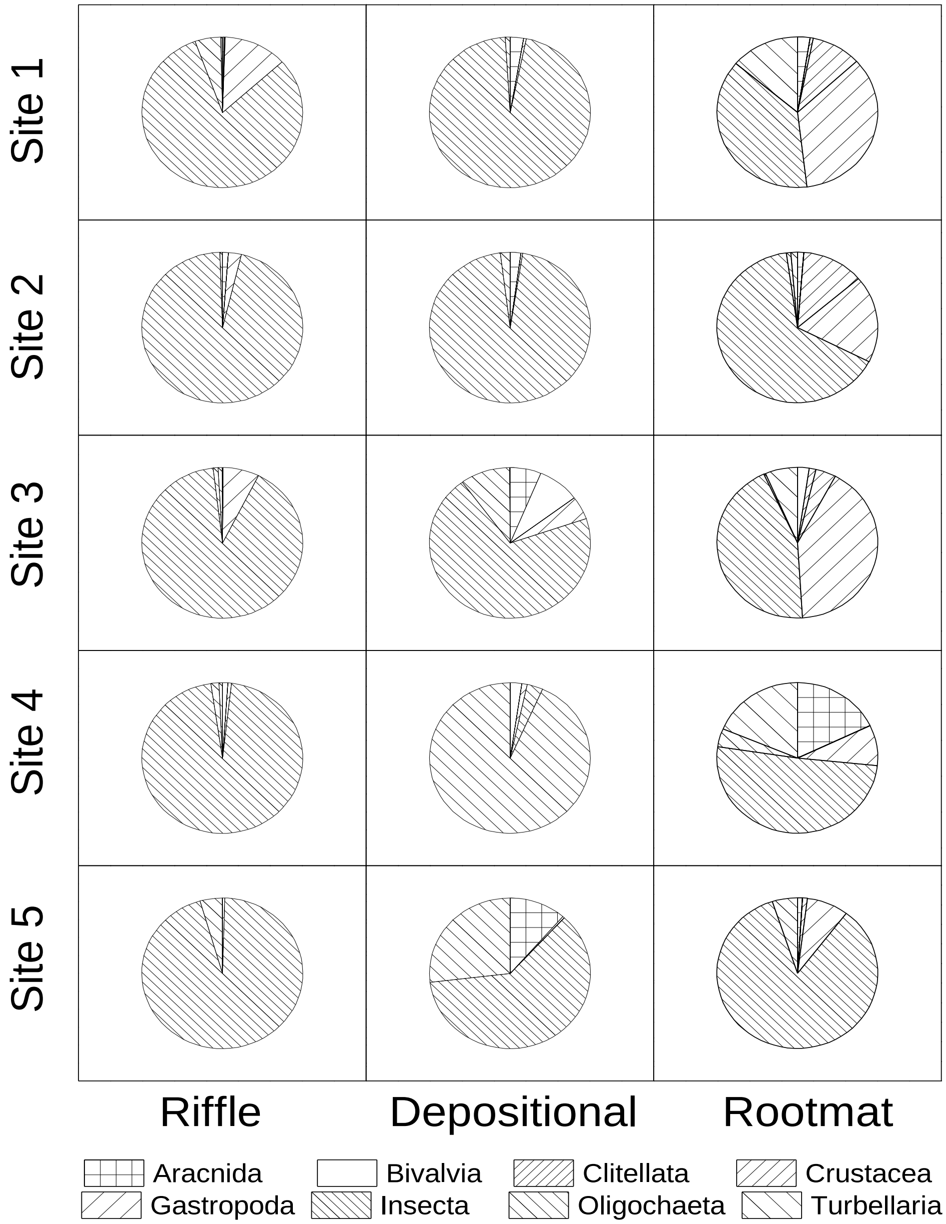

Figure 11. Relative abundance of macroinvertebrate organisms at the Class level from samples collected in the fall of 2011 in Hinkson Creek, Missouri, USA.

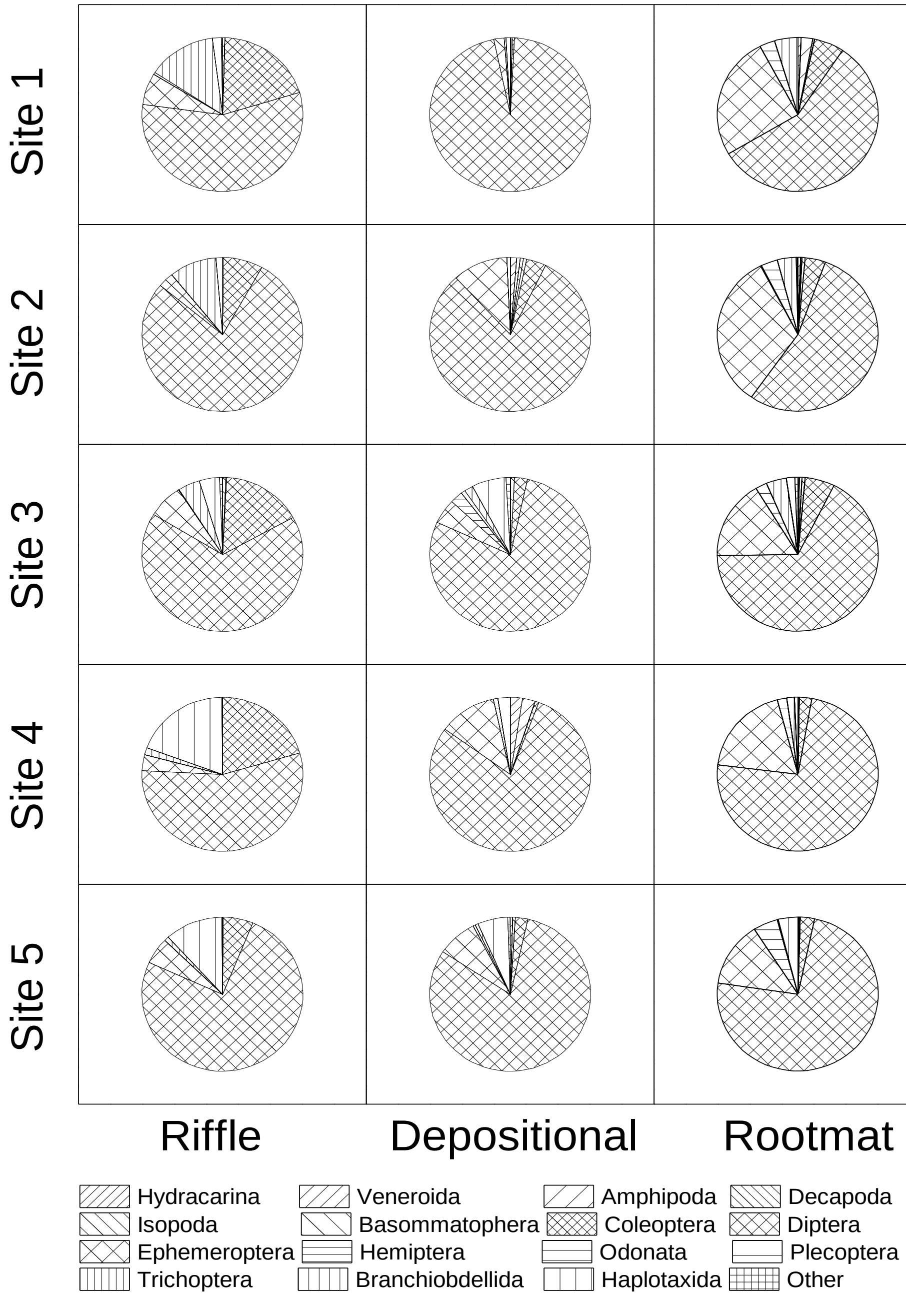

Figure 12. Relative abundance of macroinvertebrate Orders from samples collected in the spring of 2011 in Hinkson Creek, Missouri, USA.

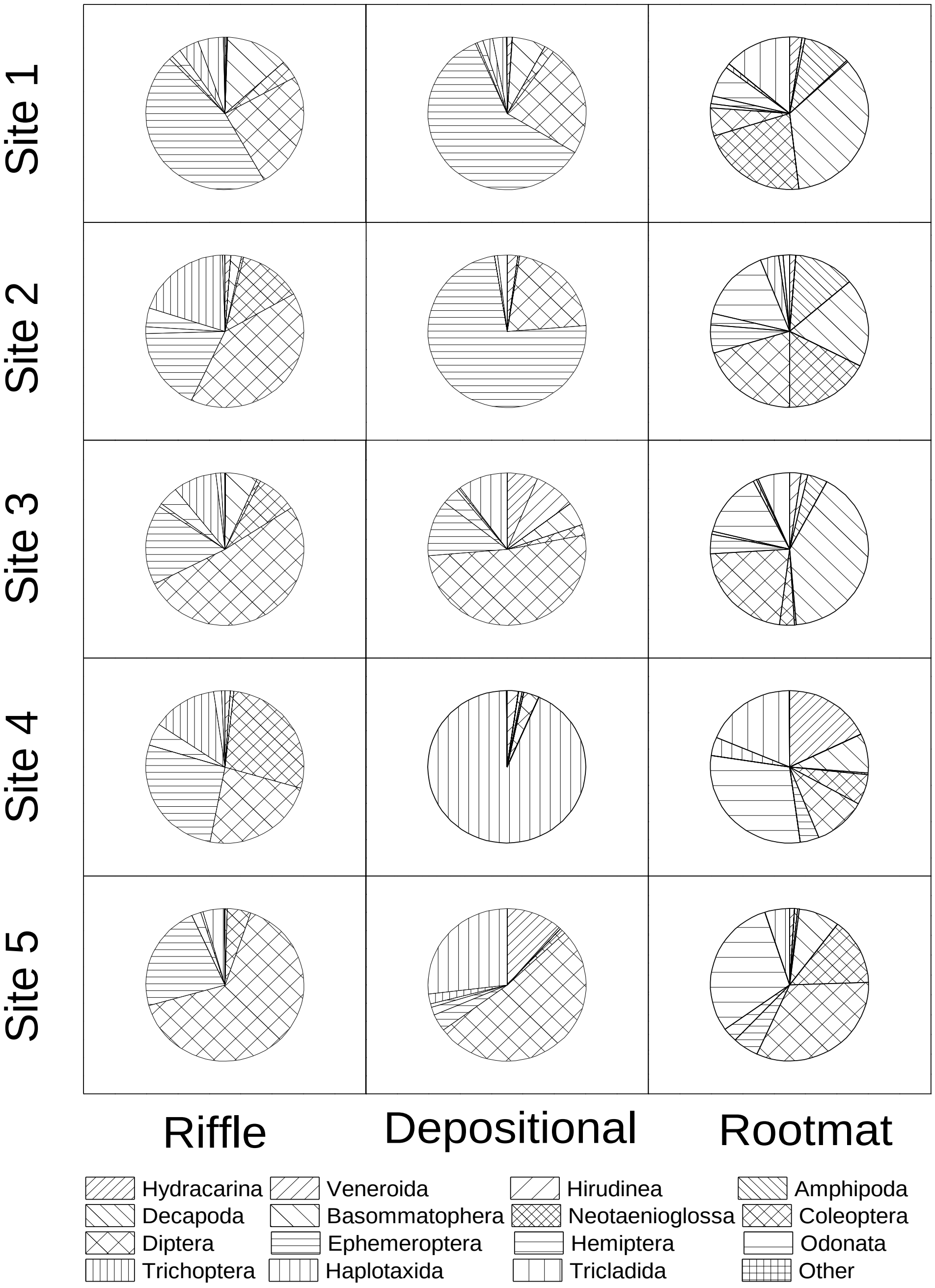

Figure 13. Relative abundance of macroinvertebrate Orders from samples collected in the fall of 2011 in Hinkson Creek, Missouri, USA.

The results of macroinvertebrate sampling in the fall of 2001 by MDNR are similar to the results of the current study from fall 2011. During 2001, Taxa richness at the two most upstream sites was only 63, which was lower than all other sites (MDNR 2002). In the current study Taxa richness at Site 1 and Taxa and EPT richness at Site 2 were higher than in urban sites. Shannon Diversity Index values did not show a consistent pattern in either season. Biotic Index was the only metric with values that were statistically significant between urban and rural sites and was 6.8% higher at urban sites in the spring and 5.9% higher in the fall. Stream Condition Index scores tended to be lower in urban reaches, with the exception of Site 3, which scored consistently high in both seasons. In the spring Sites 4 and 5 were considered partially biologically supporting based on the SCI criteria. In the fall, Site 4 was the only site to score below the fully supporting category. These results are consistent with previous bioassessments conducted in Hinkson Creek (MDNR 2002, 2004, 2006). Previous bioassessments found that metric scores generally indicated greater richness in the rural reaches although sites in the urban portions occasionally were similar to the upstream reference reaches (MDNR 2002, 2004, 2006). The lack of significant differences between metric values in urban and rural sites suggests a relatively weak or inconsistent signal from urban land-use related stressors. This supports the findings of Cuffney et al. (2010) who found that cities that developed on previously agricultural lands, where agriculture was still present in surrounding rural areas, show relatively weak responses to urbanization in macroinvertebrate metrics. The lack of clear differences between urban and rural site groups in the current study contrasts with the findings of Herringshaw et al. (2011) who sampled macroinvertebrates in a mixed land use watershed in Iowa and found that streams were degraded in both agricultural and urban affected areas but that urban land use did impact the stream beyond the level of agricultural lands. Within individual habitat types, metric responses in the current study were often inconsistent with the urban stream syndrome (Walsh et al. 2005). Within depositional samples in the spring, Taxa richness was highest at Site 5, which was 37% higher than Site 1. Similarly, Sites 3 and 5 had the highest EPT richness in spring depositional samples (Table 12). In depositional samples from the fall, Sites 3 and 5 both had 32 taxa which were 8.5% and 14.2% higher than Sites 1 and 2 respectively. Rootmats at sites 3 and 5 had the highest SD in the spring with 3.12 and 3.0 respectively which was only slightly higher than Site 1 which had an SD of 2.98, however in the fall, Site 5 had and SD of 3.0 which was 27.7% higher than Site 1.

There are seasonal differences evident in macroinvertebrate assemblages. Analysis of the pooled set of samples from all sampling seasons showed a greater number of significantly different trait metrics during the fall. This may be a result of the loss of connectivity between sites due to low flows (Boulton et al. 2003, Dewson et al. 2007). Several traits which were significantly different in fall samples are consistent with the effects of intermittency and reduced riffle habitat in headwater reaches. For instance, samples collected in the fall from riffles in the lower reaches, where the stream was perennial, had higher percentages of obligate riffle dwellers (P = 0.0074). However, spring sample results indicated that headwater streams had significantly more riffle obligates (P = 0.0008). Garcia-Roger et al. (2010) sampled intermittent Mediterranean streams and found macroinvertebrate assemblages during dry periods had a higher proportion of organisms indicative of pool habitats, even in riffles. Further, riffles in lower reaches had higher percentages of permanently attached organisms (P = 0.009). Attachment would be a detrimental trait in intermittent reaches where riffles might be dry for portions of the year. Likewise, lower reaches had significantly higher numbers of filter-feeders (Figure 14), which rely on flowing water to obtain fine organic matter (Merrit and Cummins 1996).

The effects of climatic conditions on streamflow and macroinvertebrate communities has implications for future biomonitoring. Samples from the fall during the study period as well as during the fall of 2001 (MDNR 2002) showed a pattern in which headwater sites have lower Taxa and EPT richness, diversity and SCI score, while streams in the lower portion of the watershed have higher richness and diversity and higher SCI scores. During times of lower than normal precipitation, biotic communities in the headwaters are stressed from reduced area of submerged habitat while sites lower in the stream benefit from a lack of high-discharge scouring events. Given this trend, it is reasonable to predict that in water year 2012, during which precipitation was 33% lower than the 2001-2010 average (Sanborn Field, University of Missouri data), macroinvertebrate taxa richness and diversity would be similar to the current year. Water year 2010 was a relatively wet year during which precipitation was 33% higher than the average (Sanborn Field data). If macroinvertebrates had been collected during that time period, sites at the lower end of the watershed may have had a lower SCI score as a result of disturbance from high scouring flows.

It is difficult to determine what effect the backwatering at Site 5 had on macroinvertebrates during this study. Site 5 had the highest Shannon diversity (Table 14) which indicates that the backwatering may have had a positive effect on macroinvertebrates; however that site also had the highest Biotic Index, meaning that the assemblage, while diverse, was composed of more tolerant taxa. Site 5 also had the highest Jaccard similarity between habitats of any site during the fall (Table 18). Site 5 had the 2nd lowest beta diversity (only Site 1 was lower), further indicating a degree of homogeneity between habitats. During this period of backwatering water was slow moving and hydraulic forces would have been minimal. At the streambed, the entire channel would have been under similar influences with little differentiation between riffles and pools. The water level receded only a couple weeks prior to the fall sample collection, so it is likely the effects of the backwatering were still present. Other studies have shown that backwater areas in and around tributaries can have greater macroinvertebrate densities than the main river channel (Sheaffer and Nickum 1986), and serve as important spawning and feeding habitat (Nunn et al 2010) and may act as refuges during floods (Koizumi et al 2012). These studies focused on areas that are seasonally (Nunn et al 2010) or perennially (Sheaffer and Nickum 1986) affected by backwatering. More research is needed to explore the effects of backwatering on streams such as Hinkson Creek which are normally lotic environments and are only intermittently affected by backwatering. The phenomenon of backwatering was observed at Site 5 during previous research on Hinkson Creek (Scollan 2011), which raises the question of how frequently these events occur in this portion of the stream.

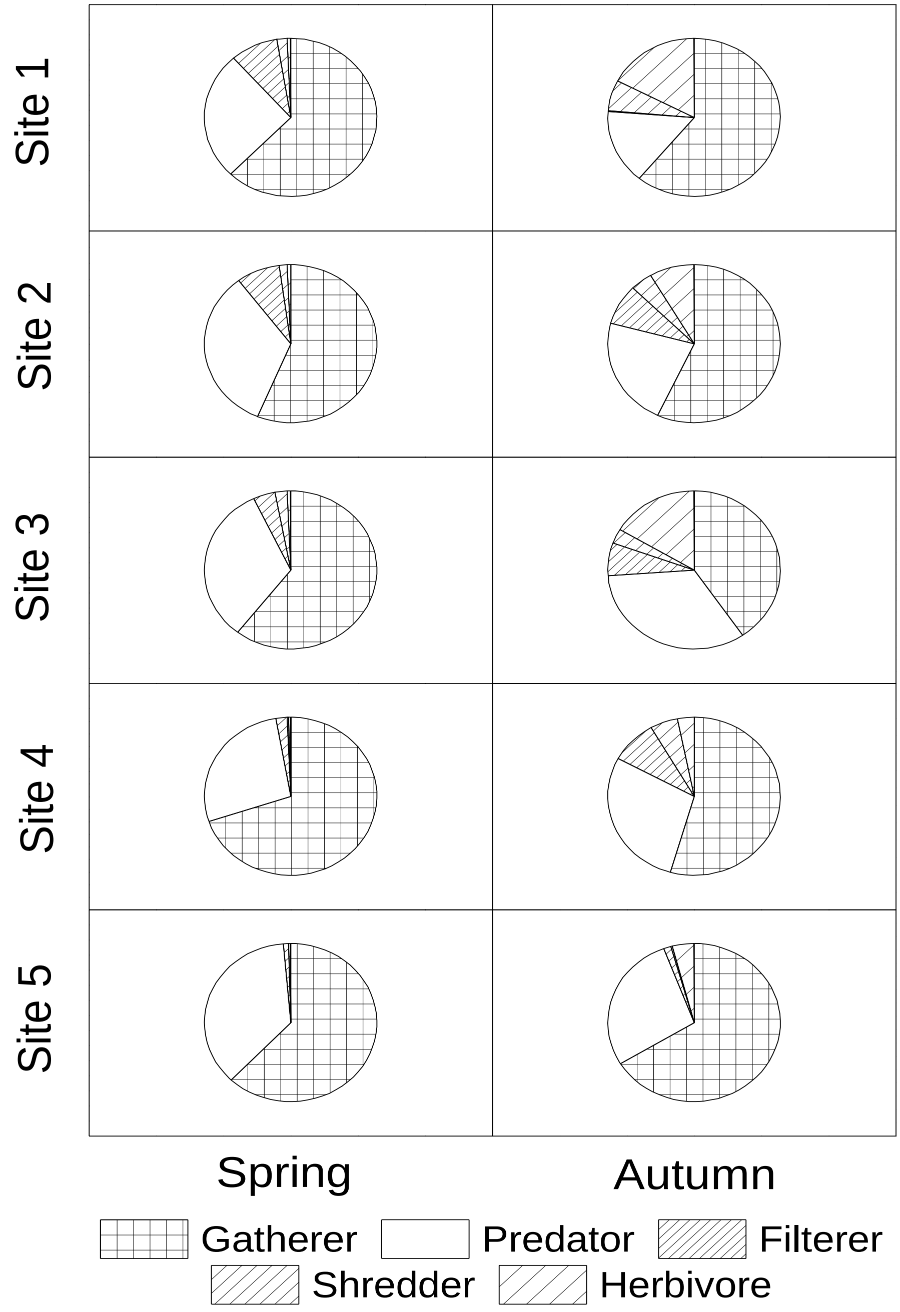

Figure 14. Relative abundances of functional feeding groups (FFG) from macroinvertebrate samples collected in Hinkson Creek, Missouri, USA.

Figure 14 shows a trend of decreasing percent shredders with distance from the headwaters during the spring and similarly decreasing percentages of shredders and herbivores in the fall. These trends are consistent with patterns predicted by the River Continuum Concept (Vannote et al 1980). Analysis of the pooled samples showed 40% and 108% increase, in the spring and fall respectively in the percentage of large bodied organisms between the rural and urban site groups. Those figures are similar to, but less pronounced than the 369% and 335% increases in large bodied organisms, in the spring and fall respectively, between the rural and urban sites. Relative abundance of gill-breathing organisms was 12.6% lower in urban reaches in the spring, which, along with differences in body size, further suggests stress related to water quality (Statzner and Beche 2010).

COMPARISON TO PREVIOUS BIOASSESSMENTS

Relative to results from previous bioassessments performed by MDNR between 2001 and 2006, the results of the current study fall generally in the same ranges of metric values. Figure 15 shows box plots of metric values derived from the earlier MDNR studies as well as the current study. Taxa richness observed in rural sites during the most recent samples from fall 2011 is higher than urban sites from both study periods as well as rural samples from the previous years (Figure 15). Taxa richness was similar in all groups during the spring. Average EPT richness was generally higher in rural sites in the spring but was similar across all groups in fall samples (Figure 15). Interestingly, in all years and in both seasons, Shannon diversity index values were higher in urban sites than in rural sites (Figure 15). This pattern is counter to the findings of most other studies of urban streams (Walsh et al. 2005, Cuffney et al. 2010).

The bioassessment reports published by MDNR identified a number of potential stressors including sedimentation and habitat degradation, chloride and other pollutants, and dissolved oxygen (MDNR 2002, 2004, 2006). The use of daily water quality measurements and the trait-based analysis used in the current study allowed for an improved understanding of these potential stressors. For instance, the finding of higher relative abundance of large bodied organisms in the urban sites, which was correlated with concentrations of chloride, suggests that chloride may, in fact, be acting as a stressor in this environment. Conversely, dissolved oxygen was not correlated with respiration or body size, which suggests that it did not exact a strong negative impact on macroinvertebrate populations. Several of the traits which showed significant trends, such as an increase in percent burrowers and percent gatherers in riffles suggest an impact of excess fine sediment. Future work in the watershed should include a bedload study to better understand changes in streambed composition related to stream length and land-use, and effects on aquatic habitat. In addition, the current study identified a correlation between several macroinvertebrate traits and measures of flow suggesting that differences in flow regime between upstream and downstream reaches are affecting the structure of macroinvertebrate assemblages. The use of hourly discharge data collected at five different points along the stream made it possible to quantify differences in the flow regime at each site. Flow regime has been shown to be important in structuring macroinvertebrate populations (Steuer et al. 2010, Coleman et al. 2011). However, many studies of macroinvertebrates in urban environments do not collect discharge data (Meyer et al. 2005, Roy et al. 2003a, Stepenuck et al. 2002, Walsh et al. 2001) or such data is only collected periodically throughout the year (Herringshaw et al. 2011).

Similarly, many studies do not include detailed water quality data. These data are often collected monthly (Walsh et al. 2001, Roy et al. 2003a, Chadwick 2006, Voelz et al. 2005), seasonally (Cuffney et al. 2010), or not at all (Stepenuck et al. 2002). Daily water quality data used in the current study captured a broad range of values in the measured parameters throughout the entire year.

This study is the first known to include a quantitative assessment of submerged woody root habitat. While several studies have investigated the effects of roots on stream bank stability (Decker and Hubble 2008, Wynne et al. 2004), the importance of woody roots as habitat has not been investigated. Future research is greatly warranted to better understand the role of woody roots in aquatic ecosystem stability and health.

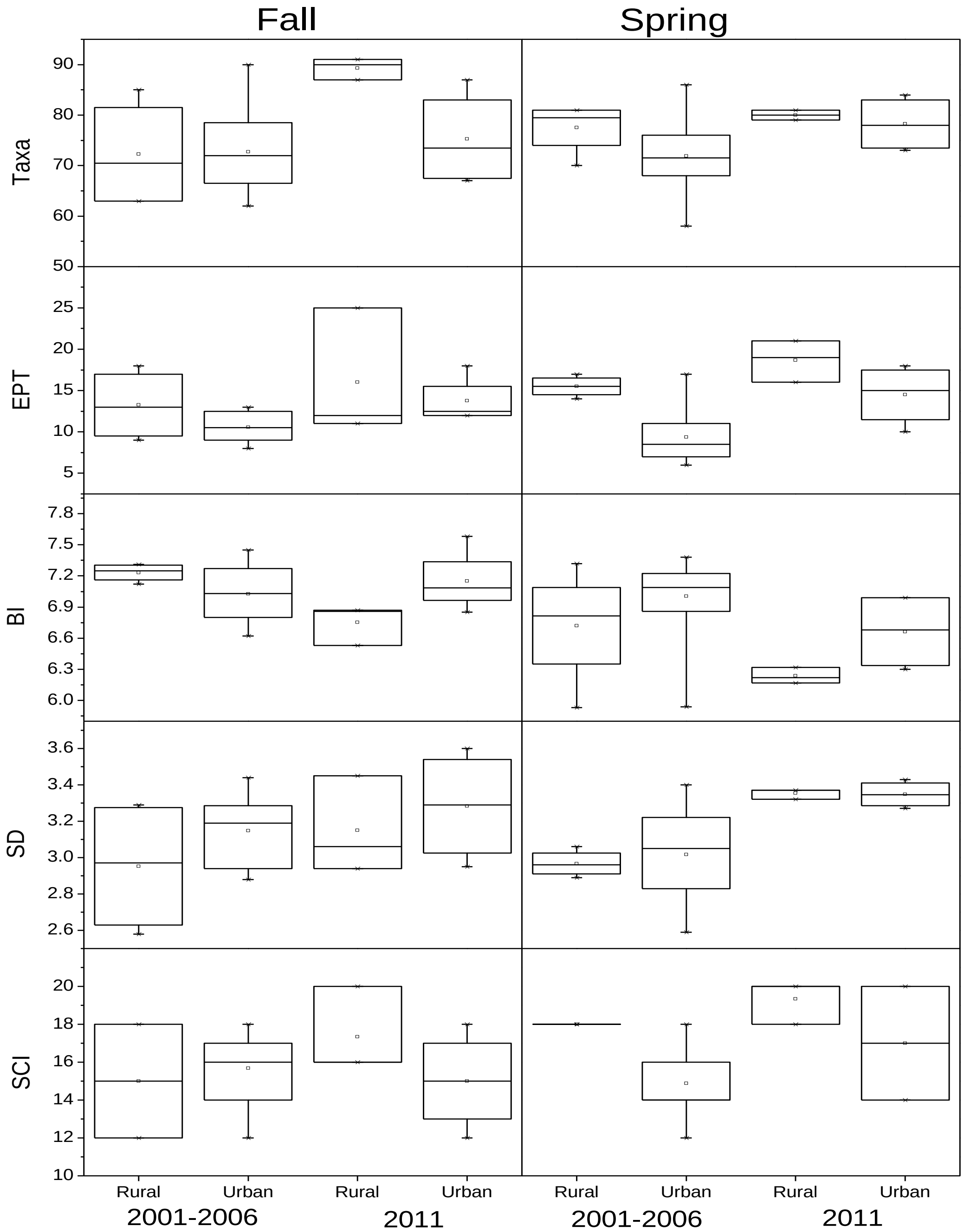

Figure 15. Metric values from macroinvertbrate samples collected in Hinkson Creek, Missouri, USA. Taxa = taxa richness, EPT = Ephemeroptera, Plecoptera, and Tricoptera taxa richness, BI = Biotic Index, SD = Shannon diversity index, SCI = Stream Condition Index.

STUDY LIMITATIONS AND RECOMMENDATIONS

The current research advances understanding of factors contributing the distribution of macroinvertebrates in Hinkson Creek. However, several limitations were evident in this study. Only five sites were used which resulted in small sample sizes (i.e. site replication) for use with regression analysis. Backwatering at Site 5 lead to the inability to calculate discharge for a large portion of the summer, which reduced the available sample size for regressions with flow variables. It is difficult to determine what effect the backwatering had on benthic macroinvertebrate community composition. The lack of significant regression relationships between habitat variables and macroinvertebrate metrics despite noticeable differences in habitat may be a result of the small sample sizes as well. Only linear models were used in regression analysis. It may be that the relationship between habitat and macroinvertebrates is non-linear; however the small sample sizes precluded the addition of higher order terms or the use of multiple explanatory variables or interaction terms in regression models.

Additional years of macroinvertebrate data consistently collected from the same sites will result in improved longitudinal comparisons by reducing the effects of outliers during any individual season (Jackson and Fureder 2006). In addition, continued monitoring of macroinvertebrates, water flow, and quality will provide the ability to detect changes resulting from any future land use changes or land management activities (Smith et al. 2011).

Further investigations of physical habitat and stream geomorphology are greatly needed to better quantify the effects of these factors on macroinvertebrate communities.

A thorough on-the-ground physical site assessment and analysis should be completed. The method used in this study to quantify woody rootmat availability was effective at estimating rootmat volume, biomass, and density. However, volume was a better indicator of habitat availability and quality than rootmat density as measured using this method. Woody rootmat density was not significantly different between sites even though rootmats were much smaller in the lower reaches. Removing the density estimation portion from future studies would save time, and energy, and would not be destructive of habitat. Alternatively, other researchers studying the hydraulic and structural effects of roots on stream banks have measured root length density (RLD), which is the length of roots per volume of soil and is closely related to root surface area (Wynn et al 2004, Pollen-Bankhead and Simon 2010). This was done using photographic image analysis software and images of excavated roots (Wynn et al 2004, Pollen-Bankhead and Simon 2010). Wood (1999) used similar photographic analysis software to measure several descriptors of rootmat complexity including root volume, length and surface area, of roots in multiple diameter size classes. Future research should incorporate a measure of root surface area as that may be more important to macroinvertebrate use (Sudduth and Meyer 2006).

The McNeil bulk core sample method was an effective technique for quantifying streambed fine substrate. However, the diameter of the core sampler (10 cm) limits the ability to quantify larger cobble substrate particles. Larger particles (> 10 cm) seemed to be more prevalent in the upper reaches, which could influence the roughness of the streambed and the distribution of invertebrates (Brooks et al. 2005). Brooks et al. (2005)

studied macroinvertebrate assemblages in various hydraulic habitats within riffles and found surface roughness to be the most important variable in explaining macroinvertebrate richness and abundance. The inclusion of an alternative method of substrate measurement, such as a pebble-count (Wohlman 1954), would improve the ability to detect and compare larger substrate. Analysis of trait based metrics suggests that these indicators responded better than conventional metrics to differences in the stream environment between sites. With the exception of Biotic Index, none of the conventional metrics detected any significant differences between urban and rural sites. The utility of trait based approaches has been demonstrated in various environments (Richards et al. 1997, Doledec et al. 2007, Tullos et al. 2009). Future biomonitoring efforts would benefit from the inclusion of measures of functional composition and trait-based metrics as well as conventional metrics.

Future research in Hinkson Creek will benefit from continued cooperation between University researchers and state agencies. Such collaboration will prevent duplication of efforts and will promote greater exchange of ideas and development of novel approaches to dealing with complex ecological questions. The benefits of and need for this kind of interdisciplinary research has been recognized by multiple researchers working in urban ecosystems (Picket et al. 2008, Wenger et al. 2009, Hubbart et al. 2010). This study benefitted from an interdisciplinary approach which combined conventional bioassessment methods with detailed time-series hydrological and water quality data, and more quantitative habitat measures than are typically employed in bioassessment procedures. Additionally, a trait-based approach to macroinvertebrate data analysis led to the identification of possible mechanistic relationships between stream physiochemical properties and macroinvertebrate assemblage composition.

This study also benefited from the availability of several years of data, collected between 2001 and 2006, by the Missouri Department of Natural Resources. Long-term studies of macroinvertebrates, which include repeated sampling from the same sites, are rare but some have suggested a high degree of year to year variability in macroinvertebrate populations (Jackson and Fureder 2006, Mazor et al. 2009). This variation has implications for biomonitoring when biological criteria are used to determine compliance with legal mandates. Currently, the Total Maximum Daily Load (TMDL) established for Hinkson Creek includes a set of narrative criteria which must be met to maintain compliance (USEPA 2011). One criterion listed applies to aquatic life: "Waters shall be free from physical, chemical or hydrologic changes that would impair the natural biological community." The definition of impairment stipulated in this criteria should be better quantified such that normal temporal variation which may result from stochastic environmental processes such as drought or flooding, do not cause the stream to be deemed non-compliant when otherwise it would not be. Furthermore, the absence of any macroinvertebrate data from the time period preceding agricultural and urban development prevents current researches from describing the "natural biological community" (USEPA 2011) that would have existed in Hinkson Creek at that time.

CHAPTER V

CONCLUSIONS

The work presented in this study advances understanding of the factors affecting stream macroinvertebrate assemblages in a multiple land-use watershed of the Midwestern USA. The objectives of this study were to quantify differences between macroinvertebrate assemblages and between hydrologic, water quality, and habitat parameters in five sites along longitudinal and land-use gradients, and to quantify the relationships between invertebrate assemblages and the physical environment.

The study was conducted from October 1, 2010 to September 30, 2011. Mean discharge increased 293% between Site 1 and Site 5 in the spring. The ratio of peak flow magnitude to base flow (Pct_99n) was 432% higher in Site 1, in the headwaters, than site 5, nearest the mouth of the stream. The frequency of high flows (Periodr9) decreased 53% from Site 1 to Site 5. The duration of high flows increased 23% over the same distance. Analysis of historic discharge from Site 4 shows that none of the hydrologic condition metrics had significantly increased or decreased over the last 45 years. However, it remains unknown how the flow regime in other parts of the stream were affected during that time period.

Chloride concentration increased 126% between the rural sites and the urban sites during the winter study period (October 2010 through March 2011). During the summer study period (April through September 2011) dissolved oxygen was significantly lower

(P = <0.001) at Sites 1 and 5 which had average concentrations of 6.8 mg/L and 6.1 mg/L respectively. DO concentrations were significantly correlated with Chlorophyll-a concentrations (R^2 = 0.94, P = 0.004) which ranged from an average of 40.9 mg/L at Site 5 to 7.0 mg/L at Site 3. Algal productivity was likely influenced by the presence of isolated relatively stagnant pools caused by low stream flow at Site 1 (Dewson et al. 2007), and by backwatering caused by flooding on the Missouri River at Site 5. Significant differences in average pH were present (P = <0.001). However, average values were within ranges considered hospitable to most organisms and ranged from 8.0 at Site 5 to 8.3 at Site 3 in the winter and between 7.7 at Site 5 to 8.0 at Site 3 in the summer. Suspended sediment concentrations were not found to vary significantly between sites when measured using either the gravimetric of volumetric methods employed in this study. Mean particle size decreased 30% between Site 1 and Site 5 during the winter and 43% during the summer. These findings agree with those of Freeman (2011) and Hubbart (2012) who also found smaller mean particle sizes in urban sites in Hinkson Creek. There are several possible explanations for the decrease in mean particle size observed during this study. Decreasing stream competence, related to the slope of the channel may result in the transport of smaller particles (Church 2002). Alternatively, previous work in this watershed has suggested that urban areas in the lower portion of the watershed are supplying finer sediments during runoff events (Hubbart 2012). More research is needed to characterize sediment compositions from various sources (i.e. land use types) in the watershed.

Fine substrate increased 332% along the length of the stream between Site 1 and Site 5. This increase in fine substrate was significantly correlated to mean particle size of suspended sediment ($R^2 = 0.926$, $P = 0.005$). Rootmat volume decreased 76% on average between rural and urban sites, while canopy cover was not significantly different between sites. Additional research is needed to explore the factors which influence rootmat size, abundance, and quality.

Macroinvertebrate assemblages in the lower reaches tended to be more pollution tolerant as evidenced by significantly higher Biotic Index values, which ranged from 6.2 at Site 1 to 6.99 at Sites 4 and 5 in the spring and from 6.53 at Site 2 to 7.58 at Site 5 in the fall. Species composition was dominated by the family Chironimidae at almost all sites. Other dominant families included Caenidae, Tubificidae, and Elmidae. In the spring there were 15 taxa which were unique to one or both of the rural sites and 32 taxa which were unique to one or more of the urban sites. In the fall there were 32 taxa and 25 taxa unique to rural and urban sites respectively.

Analysis of the functional composition of macroinvertebrate assemblages suggests that sites in the lower, urban portion of the stream are adapted to a different habitat, characterized by a higher percentage of fine sediment, higher magnitude of peak flows and perennially flowing riffles, as well as a more frequent disturbance regime, and are influenced by stress related to hydrologic disturbance and water quality.

Time series analysis of hydrologic condition metrics indicated that the flow regime has not changed significantly in the past 45 years. A variety of historical sources (Stine 1911, Tarr 1924, Davis 1928, Baver 1935) confirm that the condition of Hinkson

Creek has remained relatively unchanged over the course of the last century during which the majority of the current urban area was developed. Early forest clearing and agricultural land use which preceded urban development are likely responsible for the initial degradation of the stream environment (Harding et al. 1998). Research in other watersheds suggests that the legacy of degradation resulting from historical deforestation and agricultural land use can persist for long periods of time (Harding et al. 1998), possibly even millennia (Jackson et al. 2005). It is possible that it may take a relatively long period of time for any future management in Hinkson Creek watershed to yield results in changes of stream geomorphology or sediment regime, thus raising questions about the reasonableness of the current TMDL requirements coupled to increasing urbanization. Channel restoration activities which involve directly restoring channel geomorphology in other streams have not shown corresponding improvements in macroinvertebrate community diversity or richness (Tullos et al. 2009, Violin et al. 2011, Louhi et al. 2011).

Current and future watershed management activities, and the criteria used to assess the effectiveness of those activities, should take into account the long-term impacts of historical land use. Ward (1989) described lotic systems as four-dimensional systems in which the temporal scale of rebound from a disturbance is related to the spatial scale of the disturbance. Thus, a long recovery period would be expected from land-use-related disturbances which occur across an entire catchment. Future bioassessments conducted on Hinkson Creek should take into account the differences in geomorphology and drainage between the upper and lower reaches of the stream when selecting reference streams for comparison of macroinvertebrate communities. Reaches in the headwater portion of Hinkson creek are steeper, with coarser substrate composition and are subject to intermittent flow during dry periods, while lower reaches have a greater amount of fine sediment and are perennial.

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APPENDIX A

MACROINVERTEBRATE DATA FROM SPRING 2011

Date	Taxa	Riffle	Dep	Root	L&R Rif	L&R Dep	L&R Root	Total
Site 1. Rogers Road, Replicate #1								
3/19/2011	Lumbriculidae	3	0	0	0	0	0	3
3/19/2011	Stenelmis	97	1	2	0	0	0	100
3/19/2011	Hydracarina	6	0	2	0	0	0	8
3/19/2011	Ceratopogoninae	2	3	0	0	0	0	5
3/19/2011	Helicopsyche	1	0	0	0	0	0	1
3/19/2011	Silvius	2	0	0	0	0	0	2
3/19/2011	Tipula	1	0	0	0	0	1	1
3/19/2011	Chimarra	32	1	0	0	0	0	33
3/19/2011	Hyalella azteca	1	1	7	0	0	0	9
3/19/2011	Hydroptila	1	0	0	0	0	0	1
3/19/2011	Simulium	18	0	0	0	0	0	18
3/19/2011	Stenonema femoratum	14	0	2	0	0	0	16
3/19/2011	Acerpenna	19	0	0	0	0	0	19
3/19/2011	Cheumatopsyche	16	1	2	0	0	0	19
3/19/2011	Hydropsychidae	40	0	1	0	0	0	41
3/19/2011	Hydropsyche	16	0	0	0	0	0	16
3/19/2011	Dubiraphia	1	0	10	0	0	0	11
3/19/2011	Baetidae	3	0	0	0	0	0	3
3/19/2011	Caenis	17	4	27	0	0	0	48
3/19/2011	Perlidae	1	0	0	0	0	0	1
3/19/2011	Rhyacophila	1	0	0	0	0	0	1
3/19/2011	Stenacron	14	0	2	0	0	0	16
3/19/2011	Caenis latipennis	11	1	43	0	0	0	55
3/19/2011	Allocapnia	1	0	0	0	0	0	1
3/19/2011	Clinocera	7	2	0	0	0	0	9
3/19/2011	Hemerodromia	1	0	0	0	0	0	1

3/19/2011	Orconectes	0	0	1	1	0	0	1
3/19/2011	Branchiobdellida	0	0	0	1	0	0	0
3/19/2011	Calopteryx	0	1	0	0	0	1	1
3/19/2011	Diptera	0	1	0	0	0	0	1
3/19/2011	Empididae	0	1	0	0	0	0	1
3/19/2011	Scirtidae	0	0	1	0	0	0	1
3/19/2011	Argia	0	0	2	0	0	0	2
3/19/2011	Coenagrionidae	0	0	1	0	0	0	1
3/19/2011	Enallagma	0	0	5	0	0	0	5
3/19/2011	Helichus fastigiatus	0	0	3	0	0	0	3
3/19/2011	Heptageniidae	0	0	1	0	0	0	1
3/19/2011	Centroptilum	0	0	6	0	0	0	6
3/19/2011	Triaenodes	0	0	10	0	0	0	10
3/19/2011	Limnephilidae	0	0	1	0	0	0	1
3/19/2011	Unionidae	0	0	0	0	0	1	0
3/19/2011	Crangonyx	0	0	0	0	0	1	0
3/19/2011	Sialis	0	0	0	0	0	1	0
3/19/2011	Basiaeschna janata	0	0	0	0	0	1	0
3/19/2011	Choroterpes	0	0	0	0	0	1	0
3/19/2011	Somatochlora	0	0	0	0	0	1	0
3/19/2011	Dromogomphus	0	0	0	0	0	1	0
3/19/2011	Tipulidae	0	0	0	0	0	1	0
3/19/2011	Planorbidae	0	0	0	0	0	1	0
3/19/2011	Pycnopsyche	0	0	0	0	0	1	0
3/19/2011	Thienemannimyia grp.	21	1	47	0	0	0	69
3/19/2011	Orthocladiinae	1	2	1	0	0	0	4
3/19/2011	Parametriocnemus	31	0	1	0	0	0	32
3/19/2011	Cricotopus/Orthocladius	50	4	8	0	0	0	62
3/19/2011	Diplocladius	2	0	0	0	0	0	2
3/19/2011	Polypedilum convictum grp.	130	1	4	0	0	0	135
3/19/2011	Hydrobaenus	4	2	5	0	0	0	11
3/19/2011	Rheotanytarsus	19	1	7	0	0	0	27
3/19/2011	Phaenopsectra	1	0	3	0	0	0	4
3/19/2011	Polypedilum scalaenum grp.	17	2	2	0	0	0	21
3/19/2011	Stictochironomus	2	15	0	0	0	0	17
3/19/2011	Polypedilum halterale grp.	1	153	1	0	0	0	155
3/19/2011	Thienemanniella	9	0	4	0	0	0	13

3/19/2011	Paratanytarsus	1	1	15	0	0	0	17
3/19/2011	Cladotanytarsus	4	15	0	0	0	0	19
3/19/2011	Cryptochironomus	3	1	0	0	0	0	4
3/19/2011	Tubificidae	14	0	0	0	0	0	14
3/19/2011	Enchytraeidae	2	0	0	0	0	0	2
3/19/2011	Dicrotendipes	1	1	0	0	0	0	2
3/19/2011	Tanytarsus	16	8	28	0	0	0	52
3/19/2011	Saetheria	2	14	1	0	0	0	17
3/19/2011	Tvetenia	1	0	0	0	0	0	1
3/19/2011	Corynoneura	1	0	0	0	0	0	1
3/19/2011	Eukiefferiella	1	0	0	0	0	0	1
3/19/2011	Microtendipes	0	0	1	0	0	0	1
3/19/2011	Chironominae	0	5	2	0	0	0	7
3/19/2011	Polypedilum illinoense grp.	0	0	4	0	0	0	4
3/19/2011	Zavrelimyia	0	0	1	0	0	0	1
3/19/2011	Tanypodinae	0	0	2	0	0	0	2
3/19/2011	Ablabesmyia	0	0	8	0	0	0	8
3/19/2011	Nanocladius	0	0	23	0	0	0	23
Site 1. Rogers Road, Replicate #2								
3/19/2011	Hexatoma	5	0	0	0	0	0	5
3/19/2011	Tipula	1	0	0	0	0	1	1
3/19/2011	Stenelmis	130	0	2	0	0	0	132
3/19/2011	Chimarra	28	1	0	0	0	0	29
3/19/2011	Ceratopogoninae	1	3	0	0	0	0	4
3/19/2011	Chaoborus	1	0	0	0	0	0	1
3/19/2011	Helicopsyche	4	0	0	0	0	0	4
3/19/2011	Acerpenna	13	0	0	0	0	0	13
3/19/2011	Stenonema femoratum	4	0	2	0	0	0	6
3/19/2011	Baetidae	2	0	0	0	0	0	2
3/19/2011	Stenacron	5	0	2	0	0	0	7
3/19/2011	Hydropsyche	5	0	0	0	0	0	5
3/19/2011	Cheumatopsyche	16	1	2	0	0	0	19
3/19/2011	Hydropsychidae	30	0	1	0	0	0	31
3/19/2011	Simuliidae	6	0	0	0	0	0	6
3/19/2011	Silvius	5	0	0	0	0	0	5
3/19/2011	Simulium	5	0	0	0	0	0	5

3/19/2011	Alloperla	3	0	0	0	0	0	3
3/19/2011	Perlidae	1	0	0	0	0	0	1
3/19/2011	Hemerodromia	2	0	0	0	0	0	2
3/19/2011	Clinocera	4	2	0	0	0	0	6
3/19/2011	Caenis latipennis	3	1	43	0	0	0	47
3/19/2011	Caenis	16	4	27	0	0	0	47
3/19/2011	Calopteryx	0	1	0	0	0	2	1
3/19/2011	Diptera	0	1	0	0	0	0	1
3/19/2011	Empididae	0	1	0	0	0	0	1
3/19/2011	Hyalella azteca	0	1	7	0	0	0	8
3/19/2011	Hydracarina	0	0	2	0	0	0	2
3/19/2011	Dubiraphia	0	0	10	0	0	0	10
3/19/2011	Orconectes	0	0	1	0	0	0	1
3/19/2011	Scirtidae	0	0	1	0	0	0	1
3/19/2011	Argia	0	0	2	0	0	0	2
3/19/2011	Coenagrionidae	0	0	1	0	0	0	1
3/19/2011	Enallagma	0	0	5	0	0	0	5
3/19/2011	Helichus fastigiatus	0	0	3	0	0	0	3
3/19/2011	Heptageniidae	0	0	1	0	0	0	1
3/19/2011	Centroptilum	0	0	6	0	0	0	6
3/19/2011	Triaenodes	0	0	10	0	0	0	10
3/19/2011	Limnephilidae	0	0	1	0	0	0	1
3/19/2011	Unionidae	0	0	0	0	0	1	0
3/19/2011	Crangonyx	0	0	0	0	0	1	0
3/19/2011	Sialis	0	0	0	0	0	1	0
3/19/2011	Basiaeschna janata	0	0	0	0	0	1	0
3/19/2011	Choroterpes	0	0	0	0	0	1	0
3/19/2011	Somatochlora	0	0	0	0	0	1	0
3/19/2011	Dromogomphus	0	0	0	0	0	1	0
3/19/2011	Tipulidae	0	0	0	0	0	1	0
3/19/2011	Planorbidae	0	0	0	0	0	1	0
3/19/2011	Pycnopsyche	0	0	0	0	0	1	0
3/19/2011	Thienemannimyia grp.	14	1	47	0	0	0	62
3/19/2011	Orthocladiinae	1	2	1	0	0	0	4
3/19/2011	Cricotopus/Orthocladius	51	4	8	0	0	0	63
3/19/2011	Parametriocnemus	9	0	1	0	0	0	10
3/19/2011	Microtendipes	1	0	1	0	0	0	2

3/19/2011	Eukiefferiella	4	0	0	0	0	0	4
3/19/2011	Hydrobaenus	4	2	5	0	0	0	11
3/19/2011	Nanocladius	1	0	23	0	0	0	24
3/19/2011	Polypedilum convictum grp.	126	1	4	0	0	0	131
3/19/2011	Polypedilum scalaenum grp.	30	2	2	0	0	0	34
3/19/2011	Chironominae	2	5	2	0	0	0	9
3/19/2011	Tanytarsus	12	8	28	0	0	0	48
3/19/2011	Thienemanniella	8	0	4	0	0	0	12
3/19/2011	Paraphaenocladius	1	0	0	0	0	0	1
3/19/2011	Rheotanytarsus	8	1	7	0	0	0	16
3/19/2011	Stictochironomus	4	15	0	0	0	0	19
3/19/2011	Cladotanytarsus	11	15	0	0	0	0	26
3/19/2011	Saetheria	10	14	1	0	0	0	25
3/19/2011	Dicrotendipes	1	1	0	0	0	0	2
3/19/2011	Polypedilum illinoense grp.	2	0	4	0	0	0	6
3/19/2011	Corynoneura	5	0	0	0	0	0	5
3/19/2011	Tubificidae	5	0	0	0	0	0	5
3/19/2011	Enchytraeidae	2	0	0	0	0	0	2
3/19/2011	Cryptochironomus	7	1	0	0	0	0	8
3/19/2011	Polypedilum halterale grp.	0	153	1	0	0	0	154
3/19/2011	Paratanytarsus	0	1	15	0	0	0	16
3/19/2011	Zavrelimyia	0	0	1	0	0	0	1
3/19/2011	Tanypodinae	1	0	2	0	0	0	3
3/19/2011	Ablabesmyia	0	0	8	0	0	0	8
3/19/2011	Phaenopsectra	0	0	3	0	0	0	3
Site 2 Mexico Gravel Road								
5/22/2011	Polycentropus	1	0	0	0	0	1	1
5/22/2011	Rhyacophila	1	0	0	0	0	0	1
5/22/2011	Hemerodromia	1	0	0	0	0	0	1
5/22/2011	Simulium	4	1	1	0	0	0	6
5/22/2011	Baetidae	1	0	0	0	0	1	1
5/22/2011	Acerpenna	1	0	0	0	0	0	1
5/22/2011	Hydropsychidae	6	0	0	0	0	0	6
5/22/2011	Hydropsyche	2	0	0	0	0	0	2
5/22/2011	Cheumatopsyche	14	0	1	0	0	0	15
5/22/2011	Caenis	11	8	66	0	0	0	85

5/22/2011	Caenis latipennis	7	9	41	0	0	0	57
5/22/2011	Empididae	1	1	0	0	0	0	2
5/22/2011	Clinocera	9	1	0	0	0	0	10
5/22/2011	Simuliidae	28	0	0	0	0	0	28
5/22/2011	Stenonema femoratum	11	0	0	0	0	1	11
5/22/2011	Stenacron	9	0	0	0	0	1	9
5/22/2011	Helicopsyche	2	1	0	0	0	0	3
5/22/2011	Dubiraphia	1	0	9	0	0	0	10
5/22/2011	Hydracarina	1	2	2	0	0	0	5
5/22/2011	Ceratopogoninae	1	0	0	0	0	0	1
5/22/2011	Hexatoma	5	0	0	0	0	0	5
5/22/2011	Chimarra	34	0	0	0	0	1	34
5/22/2011	Stenelmis	40	5	4	0	0	0	49
5/22/2011	Tipula	2	0	0	0	0	0	2
5/22/2011	Chrysops	0	0	0	1	0	0	0
5/22/2011	Ferrissia	0	0	2	1	0	0	2
5/22/2011	Planorbidae	0	1	0	0	0	0	1
5/22/2011	Caecidotea (hypogean)	0	1	0	0	0	0	1
5/22/2011	Crangonyx	0	1	0	0	0	0	1
5/22/2011	Peltodytes	0	1	0	0	0	1	1
5/22/2011	Orconectes	0	0	1	0	0	0	1
5/22/2011	Branchiobdellida	0	0	1	0	0	0	1
5/22/2011	Argia	0	0	1	0	0	0	1
5/22/2011	Hagenius brevistylus	0	0	1	0	0	0	1
5/22/2011	Choroterpes	0	0	1	0	0	1	1
5/22/2011	Calopteryx	0	0	3	0	0	0	3
5/22/2011	Sphaeriidae	0	0	1	0	0	0	1
5/22/2011	Macronychus glabratus	0	0	1	0	0	0	1
5/22/2011	Enallagma	0	0	4	0	0	1	4
5/22/2011	Coenagrionidae	0	0	2	0	0	0	2
5/22/2011	Triaenodes	0	0	12	0	0	0	12
5/22/2011	Centroptilum	0	0	2	0	0	0	2
5/22/2011	Dromogomphus	0	0	0	0	0	1	0
5/22/2011	Lirceus	0	0	0	0	0	1	0
5/22/2011	Hyalella azteca	0	0	0	0	0	1	0
5/22/2011	Axarus	1	0	0	0	0	0	1
5/22/2011	Tanypodinae	2	1	0	0	0	0	3

5/22/2011	Thienemanniella	3	0	5	0	0	0	8
5/22/2011	Cricotopus/Orthocladius	87	4	9	0	0	0	100
5/22/2011	Cricotopus bicinctus	4	0	3	0	0	0	7
5/22/2011	Dicrotendipes	3	2	2	0	0	0	7
5/22/2011	Polypedilum convictum grp.	156	2	1	0	0	0	159
5/22/2011	Parametriocnemus	2	0	0	0	0	0	2
5/22/2011	Paratanytarsus	1	3	18	0	0	0	22
5/22/2011	Saetheria	9	3	0	0	0	0	12
5/22/2011	Cryptochironomus	7	1	1	0	0	0	9
5/22/2011	Paraphaenocladius	1	1	0	0	0	0	2
5/22/2011	Rheotanytarsus	22	0	9	0	0	0	31
5/22/2011	Stictochironomus	4	18	0	0	0	0	22
5/22/2011	Polypedilum scalaenum grp.	26	4	1	0	0	0	31
5/22/2011	Paratendipes	3	0	1	0	0	0	4
5/22/2011	Demicryptochironomus	4	0	0	0	0	0	4
5/22/2011	Polypedilum illinoense grp.	6	0	3	0	0	0	9
5/22/2011	Corynoneura	2	0	4	0	0	0	6
5/22/2011	Hydrobaenus	2	5	6	0	0	0	13
5/22/2011	Tubificidae	8	0	1	0	0	0	9
5/22/2011	Cladotanytarsus	14	16	1	0	0	0	31
5/22/2011	Eukiefferiella	16	0	0	0	0	0	16
5/22/2011	Tanytarsus	28	7	47	0	0	0	82
5/22/2011	Thienemannimyia grp.	23	2	24	0	0	0	49
5/22/2011	Polypedilum halterale grp.	0	48	0	0	0	0	48
5/22/2011	Pseudochironomus	0	1	0	0	0	0	1
5/22/2011	Chironominae	0	1	1	0	0	0	2
5/22/2011	Ablabesmyia	0	0	1	0	0	0	1
5/22/2011	Zavrelimyia	0	0	1	0	0	0	1
5/22/2011	Nanocladius	0	0	39	0	0	0	39
5/22/2011	Phaenopsectra	0	0	1	0	0	0	1
5/22/2011	Labrundinia	0	0	1	0	0	0	1
5/22/2011	Orthocladiinae	0	0	4	0	0	0	4
5/22/2011	Guttipelopia	0	1	0	0	0	0	1
Site 3. Broadway Blvd, Replicate #1								
3/19/2011	Argia	1	0	2	0	0	0	3
3/19/2011	Stenelmis	98	3	16	0	0	0	117

3/19/2011	Lirceus	8	0	2	0	0	0	10
3/19/2011	Stenonema femoratum	29	1	4	0	0	0	34
3/19/2011	Stenacron	11	0	1	0	0	0	12
3/19/2011	Choroterpes	1	0	1	0	0	0	2
3/19/2011	Rhyacophila	3	0	0	0	0	0	3
3/19/2011	Chimarra	2	0	0	0	0	1	2
3/19/2011	Planariidae	3		2	0	0	0	5
3/19/2011	Ceratopogoninae	1	0	1	0	0	0	2
3/19/2011	Acerpenna	22	2	9	0	0	0	33
3/19/2011	Simulium	8	0	0	0	0	0	8
3/19/2011	Clinocera	2	0	1	0	0	0	3
3/19/2011	Simuliidae	2	0	0	0	0	0	2
3/19/2011	Baetidae	1	0	2	0	0	0	3
3/19/2011	Hydropsychidae	11	0	2	0	0	0	13
3/19/2011	Piscicolidae	1	0	0	0	0	0	1
3/19/2011	Menetus	1	0	0	0	0	0	1
3/19/2011	Hydropsyche	1	0	0	0	0	0	1
3/19/2011	Caenis	13	0	22	0	0	0	35
3/19/2011	Caenis latipennis	10	4	24	0	0	0	38
3/19/2011	Hydroptila	2	1	0	0	0	0	3
3/19/2011	Diptera	1	1	0	0	0	0	2
3/19/2011	Cheumatopsyche	14	0	4	0	0	0	18
3/19/2011	Tipulidae	1	0	0	0	0	0	1
3/19/2011	Zealeuctra	1	0	0	0	0	0	1
3/19/2011	Lymnaea (Fossaria)	0	0	0	1	0	0	0
3/19/2011	Tipula	0	0	2	0	0	0	2
3/19/2011	Sphaeriidae	0	3	0	0	0	0	3
3/19/2011	Hydrobiidae	0	1	0	0	0	0	1
3/19/2011	Physa	0	1	0	0	0	0	1
3/19/2011	Hexatoma	0	1	0	0	0	0	1
3/19/2011	Polycentropodidae	0	1	0	0	0	0	1
3/19/2011	Hydracarina	0	0	1	0	0	0	1
3/19/2011	Dubiraphia	0	0	4	0	0	0	4
3/19/2011	Calopteryx	0	0	3	0	0	0	3
3/19/2011	Hetaerina	0	0	1	0	0	0	1
3/19/2011	Enallagma	0	0	2	0	0	0	2
3/19/2011	Triaenodes	0	0	6	0	0	0	6

3/19/2011	Pycnopsyche	0	0	2	0	0	0	2
3/19/2011	Hyalella azteca	0	0	1	0	0	0	1
3/19/2011	Crangonyx	0	0	1	0	0	0	1
3/19/2011	Orconectes	0	0	0	0	0	1	0
3/19/2011	Erpobdellidae	0	0	0	0	0	1	0
3/19/2011	Chrysops	0	0	0	0	0	1	0
3/19/2011	Lumbricina	0	0	2	0	0	0	2
3/19/2011	Branchiura sowerbyi	0	0	1	0	0	0	1
3/19/2011	Thienemannimyia grp.	34	3	55	0	0	0	92
3/19/2011	Zavrelimyia	1	1	0	0	0	0	2
3/19/2011	Tribelos	1	1	0	0	0	0	2
3/19/2011	Cricotopus/Orthocladius	46	3	15	0	0	0	64
3/19/2011	Parametriocnemus	28	0	1	0	0	0	29
3/19/2011	Cricotopus bicinctus	5	1	6	0	0	0	12
3/19/2011	Phaenopsectra	1	0	1	0	0	0	2
3/19/2011	Orthocladiinae	3	2	0	0	0	0	5
3/19/2011	Paraphaenocladius	9	0	0	0	0	0	9
3/19/2011	Polypedilum convictum grp.	161	1	3	0	0	0	165
3/19/2011	Nanocladius	2	0	32	0	0	0	34
3/19/2011	Eukiefferiella	5	0	0	0	0	0	5
3/19/2011	Tvetenia	2	0	0	0	0	0	2
3/19/2011	Rheotanytarsus	5	0	2	0	0	0	7
3/19/2011	Polypedilum scalaenum grp.	9	10	0	0	0	0	19
3/19/2011	Tanypodinae	1	0	0	0	0	0	1
3/19/2011	Cryptochironomus	3	3	1	0	0	0	7
3/19/2011	Stictochironomus	2	14	0	0	0	0	16
3/19/2011	Paratendipes	1	1	0	0	0	0	2
3/19/2011	Saetheria	6	0	0	0	0	0	6
3/19/2011	Thienemanniella	2	1	22	0	0	0	25
3/19/2011	Hydrobaenus	1	4	7	0	0	0	12
3/19/2011	Corynoneura	1	1	9	0	0	0	11
3/19/2011	Limnodrilus hoffmeisteri	5	0	2	0	0	0	7
3/19/2011	Tubificidae	27	8	1	0	0	0	36
3/19/2011	Enchytraeidae	2	0	0	0	0	0	2
3/19/2011	Tanytarsus	4	7	25	0	0	0	36
3/19/2011	Polypedilum illinoense grp.	1	1	7	0	0	0	9
3/19/2011	Dicrotendipes	0	1	1	0	0	0	2

3/19/2011	Polypedilum halterale grp.	0	29	0	0	0	0	29
3/19/2011	Cladotanytarsus	0	1	5	0	0	0	6
3/19/2011	Paratanytarsus	0	2	30	0	0	0	32
3/19/2011	Smittia	0	1	0	0	0	0	1
3/19/2011	Chironomus	0	0	1	0	0	0	1
3/19/2011	Labrundinia	0	0	3	0	0	0	3
Site 3. Broadway Blvd, Replicate #2								
3/19/2011	Stenelmis	93	3	16	0	0	0	112
3/19/2011	Tipula	1	0	2	0	0	0	3
3/19/2011	Rhyacophila	2	0	0	0	0	0	2
3/19/2011	Planariidae	2	0	2	0	0	0	4
3/19/2011	Chimarra	3	0	0	0	0	1	3
3/19/2011	Stenonema femoratum	35	1	4	0	0	0	40
3/19/2011	Heptageniidae	2	0	0	0	0	0	2
3/19/2011	Coenagrionidae	1	0	0	0	0	0	1
3/19/2011	Chaoborus	1	0	0	0	0	0	1
3/19/2011	Caenis	13	0	22	0	0	0	35
3/19/2011	Simuliidae	8	0	0	0	0	0	8
3/19/2011	Diptera	2	1	0	0	0	0	3
3/19/2011	Clinocera	2	0	1	0	0	0	3
3/19/2011	Baetidae	4	0	2	0	0	0	6
3/19/2011	Acerpenna	18	2	9	0	0	0	29
3/19/2011	Caenis latipennis	5	4	24	0	0	0	33
3/19/2011	Ferrissia	2	0	0	0	0	0	2
3/19/2011	Hydroptila	1	1	0	0	0	0	2
3/19/2011	Piscicolidae	1	0	0	0	0	0	1
3/19/2011	Cheumatopsyche	10	0	4	0	0	0	14
3/19/2011	Hydropsyche	1	0	0	0	0	0	1
3/19/2011	Hydropsychidae	4	0	2	0	0	0	6
3/19/2011	Limnephilidae	1	0	0	0	0	0	1
3/19/2011	Simulium	12	0	0	0	0	0	12
3/19/2011	Sphaeriidae	0	3	0	1	0	0	3
3/19/2011	Hydrobiidae	0	1	0	0	0	0	1
3/19/2011	Physa	0	1	0	0	0	0	1
3/19/2011	Hexatoma	0	1	0	0	0	0	1
3/19/2011	Polycentropodidae	0	1	0	0	0	0	1

3/19/2011	Ceratopogoninae	0	0	1	0	0	0	1
3/19/2011	Choroterpes	0	0	1	0	0	0	1
3/19/2011	Lirceus	0	0	2	0	0	0	2
3/19/2011	Hydracarina	0	0	1	0	0	0	1
3/19/2011	Dubiraphia	0	0	4	0	0	0	4
3/19/2011	Calopteryx	0	0	3	0	0	0	3
3/19/2011	Argia	0	0	2	0	0	0	2
3/19/2011	Hetaerina	0	0	1	0	0	0	1
3/19/2011	Enallagma	0	0	2	0	0	0	2
3/19/2011	Triaenodes	0	0	6	0	0	0	6
3/19/2011	Stenacron	0	0	1	0	0	0	1
3/19/2011	Pycnopsyche	0	0	2	0	0	0	2
3/19/2011	Hyalella azteca	0	0	1	0	0	0	1
3/19/2011	Crangonyx	0	0	1	0	0	0	1
3/19/2011	Orconectes	0	0	0	0	0	1	0
3/19/2011	Erpobdellidae	0	0	0	0	0	1	0
3/19/2011	Chrysops	0	0	0	0	0	1	0
3/19/2011	Lumbricina	0	0	2	0	0	0	2
3/19/2011	Branchiura sowerbyi	0	0	1	0	0	0	1
3/19/2011	Natarsia	1	0	0	0	0	0	1
3/19/2011	Thienemannimyia grp.	46	3	55	0	0	0	104
3/19/2011	Cricotopus/Orthocladius	30	3	15	0	0	0	48
3/19/2011	Hydrobaenus	11	4	7	0	0	0	22
3/19/2011	Dicrotendipes	4	1	1	0	0	0	6
3/19/2011	Diplocladius	1	0	0	0	0	0	1
3/19/2011	Cricotopus bicinctus	3	1	6	0	0	0	10
3/19/2011	Parametriocnemus	7	0	1	0	0	0	8
3/19/2011	Paraphaenocladius	3	0	0	0	0	0	3
3/19/2011	Nanocladius	1	0	32	0	0	0	33
3/19/2011	Polypedilum convictum grp.	206	1	3	0	0	0	210
3/19/2011	Cryptochironomus	7	3	1	0	0	0	11
3/19/2011	Eukiefferiella	1	0	0	0	0	0	1
3/19/2011	Eukiefferiella brevicalcar grp.	1	0	0	0	0	0	1
3/19/2011	Rheocricotopus	1	0	0	0	0	0	1
3/19/2011	Paratendipes	5	1	0	0	0	0	6
3/19/2011	Stictochironomus	3	14	0	0	0	0	17
3/19/2011	Polypedilum halterale grp.	2	29	0	0	0	0	31

3/19/2011	Tanytarsus	6	7	25	0	0	0	38
3/19/2011	Thienemanniella	4	1	22	0	0	0	27
3/19/2011	Saetheria	26	0	0	0	0	0	26
3/19/2011	Corynoneura	1	1	9	0	0	0	11
3/19/2011	Polypedilum scalaenum grp.	24	10	0	0	0	0	34
3/19/2011	Cladotanytarsus	1	1	5	0	0	0	7
3/19/2011	Orthocladiinae	2	2	0	0	0	0	4
3/19/2011	Limnodrilus hoffmeisteri	6	0	2	0	0	0	8
3/19/2011	Tubificidae	11	8	1	0	0	0	20
3/19/2011	Rheotanytarsus	1	0	2	0	0	0	3
3/19/2011	Zavrelimyia	0	1	0	0	0	0	1
3/19/2011	Tribelos	0	1	0	0	0	0	1
3/19/2011	Paratanytarsus	0	2	30	0	0	0	32
3/19/2011	Polypedilum illinoense grp.	0	1	7	0	0	0	8
3/19/2011	Smittia	0	1	0	0	0	0	1
3/19/2011	Chironomus	0	0	1	0	0	0	1
3/19/2011	Labrundinia	0	0	3	0	0	0	3
3/19/2011	Phaenopsectra	0	0	1	0	0	0	1
Site 4. Reactor Field, MU Campus								
3/21/2011	Stenelmis	118	4	6	0	0	0	128
3/21/2011	Helicopsyche	1	0	0	0	0	0	1
3/21/2011	Stenonema femoratum	48	0	8	0	0	0	56
3/21/2011	Stenacron	10	0	2	0	0	0	12
3/21/2011	Planariidae	1	0	0	0	0	0	1
3/21/2011	Ectopria	1	0	0	0	0	0	1
3/21/2011	Acerpenna	3	0	1	0	0	0	4
3/21/2011	Clinocera	5	0	0	0	0	0	5
3/21/2011	Caenis	14	8	29	0	0	0	51
3/21/2011	Caenis latipennis	3	5	27	0	0	0	35
3/21/2011	Hydropsychidae	1	0	1	0	0	0	2
3/21/2011	Cheumatopsyche	7	0	3	0	0	0	10
3/21/2011	Simulium	5	0	0	0	0	0	5
3/21/2011	Simuliidae	2	0	0	0	0	0	2
3/21/2011	Rhyacophila	0	0	0	1	0	0	0
3/21/2011	Silvius	0	0	0	2	0	0	0
3/21/2011	Corbicula	0	3	0	0	0	0	3

3/21/2011	Hagenius brevistylus	0	1	0	0	0	0	1
3/21/2011	Stratiomyidae	0	1	0	0	0	0	1
3/21/2011	Hydracarina	0	3	0	0	0	0	3
3/21/2011	Dubiraphia	0	1	1	0	0	0	2
3/21/2011	Berosus	0	0	1	0	0	0	1
3/21/2011	Triaenodes	0	0	1	0	0	0	1
3/21/2011	Calopteryx	0	0	1	0	0	0	1
3/21/2011	Lirceus	0	0	1	0	0	0	1
3/21/2011	Enallagma	0	0	1	0	0	0	1
3/21/2011	Coenagrionidae	0	0	2	0	0	0	2
3/21/2011	Argia	0	0	2	0	0	0	2
3/21/2011	Centroptilum	0	0	2	0	0	0	2
3/21/2011	Orconectes	0	0	0	0	0	1	0
3/21/2011	Choroterpes	0	0	0	0	0	1	0
3/21/2011	Basiaeschna janata	0	0	0	0	0	1	0
3/21/2011	Hexatoma	0	0	0	0	0	1	0
3/21/2011	Tipula	0	0	0	0	0	1	0
3/21/2011	Chrysops	0	0	0	0	0	1	0
3/21/2011	Polycentropodidae	0	0	0	0	0	1	0
3/21/2011	Heteroceridae	0	0	0	0	0	1	0
3/21/2011	Lumbricina	1	0	0	0	0	0	1
3/21/2011	Branchiura sowerbyi	1	0	0	0	0	0	1
3/21/2011	Stictochironomus	2	4	2	0	0	0	8
3/21/2011	Chironomus	1	0	0	0	0	0	1
3/21/2011	Cricotopus/Orthocladius	36	2	8	0	0	0	46
3/21/2011	Thienemannimyia grp.	15	2	57	0	0	0	74
3/21/2011	Cricotopus bicinctus	1	1	8	0	0	0	10
3/21/2011	Tanypodinae	1	0	0	0	0	0	1
3/21/2011	Paratendipes	3	0	0	0	0	0	3
3/21/2011	Stenochironomus	1	0	0	0	0	0	1
3/21/2011	Parametriocnemus	5	0	0	0	0	0	5
3/21/2011	Microtendipes	1	0	0	0	0	0	1
3/21/2011	Polypedilum convictum grp.	89	0	5	0	0	0	94
3/21/2011	Polypedilum scalaenum grp.	30	6	0	0	0	0	36
3/21/2011	Saetheria	46	2	0	0	0	0	48
3/21/2011	Hydrobaenus	4	4	17	0	0	0	25
3/21/2011	Cladotanytarsus	19	2	3	0	0	0	24

3/21/2011	Polypedilum halterale grp.	1	51	0	0	0	0	52
3/21/2011	Nanocladius	1	0	16	0	0	0	17
3/21/2011	Thienemanniella	1	0	9	0	0	0	10
3/21/2011	Corynoneura	2	0	9	0	0	0	11
3/21/2011	Rheotanytarsus	5	0	5	0	0	0	10
3/21/2011	Tubificidae	68	1	2	0	0	0	71
3/21/2011	Limnodrilus hoffmeisteri	49	1	2	0	0	0	52
3/21/2011	Limnodrilus claparedianus	2	1	0	0	0	0	3
3/21/2011	Cryptochironomus	14	0	4	0	0	0	18
3/21/2011	Tanytarsus	10	13	39	0	0	0	62
3/21/2011	Glyptotendipes	0	1	1	0	0	0	2
3/21/2011	Dicrotendipes	0	1	2	0	0	0	3
3/21/2011	Paratanytarsus	0	3	35	0	0	0	38
3/21/2011	Polypedilum illinoense grp.	0	2	6	0	0	0	8
3/21/2011	Stempellinella	0	1	1	0	0	0	2
3/21/2011	Ablabesmyia	0	0	4	0	0	0	4
3/21/2011	Phaenopsectra	0	0	2	0	0	0	2
3/21/2011	Paraphaenocladius	0	0	1	0	0	0	1
3/21/2011	Orthocladiinae	0	0	2	0	0	0	2
Site 5. Scott Blvd								
3/22/2011	Stenelmis	37	4	0	0	0	0	41
3/22/2011	Chimarra	1	0	0	0	0	0	1
3/22/2011	Diptera	1	2	0	0	0	0	3
3/22/2011	Stenonema femoratum	22	1	6	0	0	0	29
3/22/2011	Lirceus	1	1	0	0	0	0	2
3/22/2011	Piscicolidae	1	0	0	0	0	0	1
3/22/2011	Ceratopogoninae	1	1	1	0	0	0	3
3/22/2011	Acerpenna	16	1	19	0	0	0	36
3/22/2011	Cheumatopsyche	5	1	0	0	0	1	6
3/22/2011	Clinocera	1	0	0	0	0	0	1
3/22/2011	Simuliidae	1	0	0	0	0	0	1
3/22/2011	Simulium	2	0	0	0	0	0	2
3/22/2011	Tipulidae	1	0	0	0	0	0	1
3/22/2011	Caenis	14	13	25	0	0	0	52
3/22/2011	Caenis latipennis	5	2	6	0	0	0	13
3/22/2011	Stenelmis sexlineata	5	0	1	0	0	0	6

3/22/2011	Dubiraphia	1	0	5	0	0	0	6
3/22/2011	Argia	0	1	2	0	0	0	3
3/22/2011	Hydracarina	0	1	0	0	0	0	1
3/22/2011	Lepidoptera	0	1	0	0	0	0	1
3/22/2011	Neoporus	0	1	0	0	0	0	1
3/22/2011	Stenacron	0	0	6	0	0	0	6
3/22/2011	Branchiobdellida	0	0	15	0	0	0	15
3/22/2011	Choroterpes	0	0	1	0	0	0	1
3/22/2011	Calopteryx	0	0	4	0	0	0	4
3/22/2011	Hetaerina	0	0	1	0	0	0	1
3/22/2011	Orconectes	0	0	1	0	0	0	1
3/22/2011	Somatochlora	0	0	1	0	0	0	1
3/22/2011	Enallagma	0	0	12	0	0	0	12
3/22/2011	Centroptilum	0	0	2	0	0	0	2
3/22/2011	Nectopsyche	0	0	1	0	0	0	1
3/22/2011	Macromia	0	0	0	0	0	1	0
3/22/2011	Orconectes virilis	0	0	0	0	0	1	0
3/22/2011	Belostoma	0	0	0	0	0	1	0
3/22/2011	Hyalella azteca	0	0	0	0	0	1	0
3/22/2011	Crangonyx	0	0	0	0	0	1	0
3/22/2011	Stictochironomus	23	19	0	0	0	0	42
3/22/2011	Polypedilum fallax grp.	1	0	0	0	0	0	1
3/22/2011	Thienemannimyia grp.	27	0	76	0	0	0	103
3/22/2011	Natarsia	1	1	0	0	0	0	2
3/22/2011	Cricotopus/Orthocladius	58	4	24	0	0	0	86
3/22/2011	Dicrotendipes	6	2	5	0	0	0	13
3/22/2011	Parametriocnemus	1	0	0	0	0	0	1
3/22/2011	Polypedilum illinoense grp.	8	7	8	0	0	0	23
3/22/2011	Cricotopus bicinctus	5	0	22	0	0	0	27
3/22/2011	Polypedilum halterale grp.	14	31	1	0	0	0	46
3/22/2011	Saetheria	15	0	0	0	0	0	15
3/22/2011	Polypedilum scalaenum grp.	51	21	2	0	0	0	74
3/22/2011	Phaenopsectra	2	0	4	0	0	0	6
3/22/2011	Tanytarsus	20	34	32	0	0	0	86
3/22/2011	Rheotanytarsus	6	0	1	0	0	0	7
3/22/2011	Paratanytarsus	2	4	32	0	0	0	38
3/22/2011	Eukiefferiella	2	0	0	0	0	0	2

3/22/2011	Cryptochironomus	72	3	0	0	0	0	75
3/22/2011	Chironomus	2	3	0	0	0	0	5
3/22/2011	Hydrobaenus	19	5	11	0	0	0	35
3/22/2011	Polypedilum convictum grp.	87	1	3	0	0	0	91
3/22/2011	Orthocladiinae	2	0	1	0	0	0	3
3/22/2011	Paratendipes	3	1	0	0	0	0	4
3/22/2011	Cladotanytarsus	13	3	0	0	0	0	16
3/22/2011	Microtendipes	1	0	0	0	0	0	1
3/22/2011	Thienemanniella	1	1	16	0	0	0	18
3/22/2011	Limnodrilus hoffmeisteri	17	3	0	0	0	0	20
3/22/2011	Tubificidae	52	8	0	0	0	0	60
3/22/2011	Enchytraeidae	3	1	0	0	0	0	4
3/22/2011	Limnodrilus claparedianus	1	0	0	0	0	0	1
3/22/2011	Nanocladius	0	3	7	0	0	0	10
3/22/2011	Paralauterborniella	0	2	0	0	0	0	2
3/22/2011	Corynoneura	0	4	12	0	0	0	16
3/22/2011	Tanypodinae	0	1	0	0	0	0	1
3/22/2011	Ablabesmyia	0	5	15	0	0	0	20
3/22/2011	Zavrelimyia	0	0	2	0	0	0	2
3/22/2011	Labrundinia	0	0	3	0	0	0	3
3/22/2011	Rheocricotopus	0	0	1	0	0	0	1

MACROINVERTEBRATE DATA FROM FALL 2011.

Date	Taxa	Riffle	Dep	Root	L&R Rif	L&R Dep	L&R Rm	Total
Site 1. Rogers Road, Replicate #1								
9/20/2011	Helicopsyche	24	0	0	0	0	0	24
9/20/2011	Physa	53	1	14	0	0	0	68
9/20/2011	Sphaeriidae	2	0	4	0	1	0	6
9/20/2011	Ferrissia	23	0	0	0	0	0	23
9/20/2011	Ceratopogoninae	5	11	0	0	0	0	16
9/20/2011	Diptera	2	0	0	0	0	0	2
9/20/2011	Stenelmis	14	0	1	0	0	0	15
9/20/2011	Dubiraphia	4	1	68	0	0	0	73
9/20/2011	Stenonema femoratum	18	0	0	0	0	0	18
9/20/2011	Hydracarina	2	6	8	0	0	0	16
9/20/2011	Enallagma	3	0	15	0	0	0	18
9/20/2011	Helichus lithophilus	1	0	0	0	0	0	1
9/20/2011	Corixidae	3	0	0	0	0	0	3
9/20/2011	Planorbidae	8	2	78	0	0	0	88
9/20/2011	Stenacron	5	0	0	0	0	0	5
9/20/2011	Gomphidae	1	0	0	0	0	0	1
9/20/2011	Forcipomyiinae	1	0	0	0	0	0	1
9/20/2011	Caenis latipennis	210	183	0	0	0	0	393
9/20/2011	Calopteryx	1	0	0	0	0	0	1
9/20/2011	Caenis	88	79	1	0	0	0	168
9/20/2011	Somatochlora	1	0	0	0	0	0	1
9/20/2011	Polycentropodidae	1	0	0	0	0	0	1
9/20/2011	Macromia	1	0	0	0	0	0	1
9/20/2011	Oecetis	1	0	1	0	0	0	2
9/20/2011	Hydroptilidae	1	0	0	0	0	0	1
9/20/2011	Orconectes	1	0	0	0	0	0	1
9/20/2011	Planariidae	2	0	44	0	0	0	46
9/20/2011	Baetidae	2	2	0	0	0	0	4
9/20/2011	Corduliinae	5	0	0	0	0	0	5
9/20/2011	Argia	1	0	3	0	0	0	4
9/20/2011	Tipulidae	1	0	0	0	0	0	1

9/20/2011	Psychodidae	1	0	0	0	0	0	1
9/20/2011	Paracymus	1	0	0	0	0	0	1
9/20/2011	Libellula	1	0	0	0	0	0	1
9/20/2011	Hydrobiidae	0	0	0	1	0	0	0
9/20/2011	Belostoma	0	0	0	1	0	0	0
9/20/2011	Hexatoma	0	0	0	1	0	0	0
9/20/2011	Perithemis	0	1	0	1	0	0	1
9/20/2011	Libellulidae	0	1	2	0	0	0	3
9/20/2011	Chaoborus	0	0	0	0	1	0	0
9/20/2011	Hyalella azteca	0	0	31	0	0	0	31
9/20/2011	Paraleptophlebia	0	0	2	0	0	0	2
9/20/2011	Glossiphoniidae	0	0	2	0	0	0	2
9/20/2011	Anopheles	0	0	1	0	0	0	1
9/20/2011	Coenagrionidae	0	0	1	0	0	0	1
9/20/2011	Lioporeus triangularis	0	0	1	0	0	0	1
9/20/2011	Orconectes virilis	0	0	1	0	0	0	1
9/20/2011	Micromenetus	0	0	19	0	0	0	19
9/20/2011	Gerridae	0	0	1	0	0	0	1
9/20/2011	Triaenodes	0	0	2	0	0	0	2
9/20/2011	Ranatra kirkaldyi	0	0	0	0	0	1	0
9/20/2011	Nasiaeschna pentacantha	0	0	0	0	0	1	0
9/20/2011	Gyretes	0	0	0	0	0	1	0
9/20/2011	Arigomphus	0	0	0	0	0	1	0
9/20/2011	Basiaeschna janata	0	0	0	0	0	1	0
9/20/2011	Branchiura sowerbyi	21	0	0	0	0	0	21
9/20/2011	Tanypodinae	5	0	0	0	0	0	5
9/20/2011	Dicrotendipes	7	4	0	0	0	0	11
9/20/2011	Polypedilum illinoense grp.	8	1	1	0	0	0	10
9/20/2011	Tanytarsus	31	5	0	0	0	0	36
9/20/2011	Procladius	47	3	0	0	0	0	50
9/20/2011	Microtendipes	2	0	0	0	0	0	2
9/20/2011	Paratendipes	3	0	0	0	0	0	3
9/20/2011	Tribelos	1	0	0	0	0	0	1
9/20/2011	Larsia	2	0	0	0	0	0	2
9/20/2011	Cryptotendipes	2	3	0	0	0	0	5
9/20/2011	Chironomus	6	1	0	0	0	0	7
9/20/2011	Labrundinia	1	3	4	0	0	0	8

9/20/2011	Parachironomus	2	0	6	0	0	0	8
9/20/2011	Glyptotendipes	3	0	4	0	0	0	7
9/20/2011	Corynoneura	1	1	0	0	0	0	2
9/20/2011	Paralauterborniella	1	0	0	0	0	0	1
9/20/2011	Stempellinella	1	2	0	0	0	0	3
9/20/2011	Tubificidae	13	1	0	0	0	0	14
9/20/2011	Polypedilum scalaenum grp.	2	0	0	0	0	0	2
9/20/2011	Ablabesmyia	7	2	0	0	0	0	9
9/20/2011	Thienemannimyia grp.	1	0	0	0	0	0	1
9/20/2011	Stenochironomus	1	0	0	0	0	0	1
9/20/2011	Kiefferulus	1	0	1	0	0	0	2
9/20/2011	Cladotanytarsus	1	7	0	0	0	0	8
9/20/2011	Pseudochironomus	0	2	0	0	0	0	2
9/20/2011	Polypedilum halterale grp.	0	31	0	0	0	0	31
9/20/2011	Stictochironomus	0	3	0	0	0	0	3
9/20/2011	Chironominae	0	1	1	0	0	0	2
9/20/2011	Cryptochironomus	0	2	0	0	0	0	2
9/20/2011	Orthocladiinae	0	1	0	0	0	0	1
9/20/2011	Zavreliella	0	0	1	0	0	0	1
Site 1. Rogers Road, Replicate #2								
9/20/2011	Helicopsyche	24	0	0	0	0	0	24
9/20/2011	Physa	53	0	14	0	0	0	67
9/20/2011	Sphaeriidae	2	0	4	0	0	0	6
9/20/2011	Ferrissia	23	1	0	0	0	0	24
9/20/2011	Ceratopogoninae	5	7	0	0	0	0	12
9/20/2011	Diptera	2	0	0	0	0	0	2
9/20/2011	Stenelmis	14	0	1	0	0	0	15
9/20/2011	Dubiraphia	4	0	68	0	0	0	72
9/20/2011	Stenonema femoratum	18	0	0	0	0	0	18
9/20/2011	Hydracarina	2	12	8	0	0	0	22
9/20/2011	Enallagma	3	0	15	0	0	0	18
9/20/2011	Helichus lithophilus	1	0	0	0	0	0	1
9/20/2011	Corixidae	3	0	0	0	1	0	3
9/20/2011	Planorbidae	8	0	78	0	0	0	86
9/20/2011	Stenacron	5	0	0	0	0	0	5
9/20/2011	Gomphidae	1	0	0	0	0	0	1

9/20/2011	Forcipomyiinae	1	0	0	0	0	0	1
9/20/2011	Caenis latipennis	210	132	0	0	0	0	342
9/20/2011	Calopteryx	1	0	0	0	0	0	1
9/20/2011	Caenis	88	72	1	0	0	0	161
9/20/2011	Somatochlora	1	0	0	0	0	0	1
9/20/2011	Polycentropodidae	1	0	0	0	0	0	1
9/20/2011	Macromia	1	0	0	0	0	0	1
9/20/2011	Oecetis	1	2	1	0	0	0	4
9/20/2011	Hydroptilidae	1	0	0	0	0	0	1
9/20/2011	Orconectes	1	0	0	0	0	0	1
9/20/2011	Planariidae	2	0	44	0	0	0	46
9/20/2011	Baetidae	2	3	0	0	0	0	5
9/20/2011	Corduliinae	5	0	0	0	0	0	5
9/20/2011	Argia	1	0	3	0	0	0	4
9/20/2011	Tipulidae	1	0	0	0	0	0	1
9/20/2011	Psychodidae	1	0	0	0	0	0	1
9/20/2011	Paracymus	1	0	0	0	0	0	1
9/20/2011	Libellula	1	0	0	0	0	0	1
9/20/2011	Hydrobiidae	0	0	0	1	0	0	0
9/20/2011	Belostoma	0	0	0	1	0	0	0
9/20/2011	Hexatoma	0	0	0	1	0	0	0
9/20/2011	Perithemis	0	2	0	1	0	0	2
9/20/2011	Libellulidae	0	0	2	0	0	0	2
9/20/2011	Chaoborus	0	1	0	0	0	0	1
9/20/2011	Centroptilum	0	3	0	0	0	0	3
9/20/2011	Hyalella azteca	0	0	31	0	0	0	31
9/20/2011	Hagenius brevistylus	0	0	0	0	1	0	0
9/20/2011	Paraleptophlebia	0	0	2	0	0	0	2
9/20/2011	Glossiphoniidae	0	0	2	0	0	0	2
9/20/2011	Anopheles	0	0	1	0	0	0	1
9/20/2011	Coenagrionidae	0	0	1	0	0	0	1
9/20/2011	Lioporeus triangularis	0	0	1	0	0	0	1
9/20/2011	Orconectes virilis	0	0	1	0	0	0	1
9/20/2011	Micromenetus	0	0	19	0	0	0	19
9/20/2011	Gerridae	0	0	1	0	0	0	1
9/20/2011	Triaenodes	0	0	2	0	0	0	2
9/20/2011	Ranatra kirkaldyi	0	0	0	0	0	1	0

9/20/2011	Nasiaeschna pentacantha	0	0	0	0	0	1	0
9/20/2011	Gyretes	0	0	0	0	0	1	0
9/20/2011	Arigomphus	0	0	0	0	0	1	0
9/20/2011	Basiaeschna janata	0	0	0	0	0	1	0
9/20/2011	Branchiura sowerbyi	21	2	0	0	0	0	23
9/20/2011	Tanypodinae	5	0	0	0	0	0	5
9/20/2011	Dicrotendipes	7	16	0	0	0	0	23
9/20/2011	Polypedilum illinoense grp.	8	0	1	0	0	0	9
9/20/2011	Tanytarsus	31	12	0	0	0	0	43
9/20/2011	Procladius	47	4	0	0	0	0	51
9/20/2011	Microtendipes	2	0	0	0	0	0	2
9/20/2011	Paratendipes	3	0	0	0	0	0	3
9/20/2011	Tribelos	1	0	0	0	0	0	1
9/20/2011	Larsia	2	0	0	0	0	0	2
9/20/2011	Cryptotendipes	2	0	0	0	0	0	2
9/20/2011	Chironomus	6	3	0	0	0	0	9
9/20/2011	Labrundinia	1	1	4	0	0	0	6
9/20/2011	Parachironomus	2	0	6	0	0	0	8
9/20/2011	Glyptotendipes	3	0	4	0	0	0	7
9/20/2011	Corynoneura	1	0	0	0	0	0	1
9/20/2011	Paralauterborniella	1	0	0	0	0	0	1
9/20/2011	Stempellinella	1	4	0	0	0	0	5
9/20/2011	Tubificidae	13	1	0	0	0	0	14
9/20/2011	Polypedilum scalaenum grp.	2	0	0	0	0	0	2
9/20/2011	Ablabesmyia	7	4	0	0	0	0	11
9/20/2011	Thienemannimyia grp.	1	0	0	0	0	0	1
9/20/2011	Stenochironomus	1	0	0	0	0	0	1
9/20/2011	Kiefferulus	1	0	1	0	0	0	2
9/20/2011	Cladotanytarsus	1	4	0	0	0	0	5
9/20/2011	Pseudochironomus	0	8	0	0	0	0	8
9/20/2011	Polypedilum halterale grp.	0	22	0	0	0	0	22
9/20/2011	Stictochironomus	0	1	0	0	0	0	1
9/20/2011	Chironominae	0	2	1	0	0	0	3
9/20/2011	Cryptochironomus	0	4	0	0	0	0	4
9/20/2011	Nanocladius	0	2	0	0	0	0	2
9/20/2011	Limnodrilus hoffmeisteri	0	2	0	0	0	0	2
9/20/2011	Zavreliella	0	0	1	0	0	0	1

Site 2. Mexico Gravel Road								
9/21/2011	Caenis	33	74	4	0	0	0	111
9/21/2011	Caenis latipennis	26	152	2	0	0	0	180
9/21/2011	Chimarra	60	0	0	0	0	0	60
9/21/2011	Stenelmis	80	0	1	0	0	0	81
9/21/2011	Tricorythodes	46	0	0	0	0	0	46
9/21/2011	Hydracarina	8	7	4	0	0	0	19
9/21/2011	Physa	51	1	34	0	0	0	86
9/21/2011	Argia	23	0	9	0	0	0	32
9/21/2011	Ferrissia	14	0	7	0	0	0	21
9/21/2011	Helicopsyche	16	0	0	0	0	0	16
9/21/2011	Hetaerina	3	0	0	0	0	0	3
9/21/2011	Acerpenna	5	0	0	0	0	0	5
9/21/2011	Sphaeriidae	8	0	8	0	1	0	16
9/21/2011	Forcipomyiinae	3	0	0	0	0	0	3
9/21/2011	Stenonema femoratum	1	2	2	0	0	0	5
9/21/2011	Empididae	1	0	0	0	0	0	1
9/21/2011	Silvius	18	0	0	0	0	0	18
9/21/2011	Baetis	1	0	0	0	0	0	1
9/21/2011	Baetidae	4	0	0	0	0	0	4
9/21/2011	Rhagovelia	2	0	0	0	0	0	2
9/21/2011	Ceratopsyche	4	0	0	0	0	0	4
9/21/2011	Oecetis	1	0	2	0	0	0	3
9/21/2011	Hydropsychidae	23	0	0	0	0	0	23
9/21/2011	Helichus lithophilus	1	0	0	0	0	0	1
9/21/2011	Hemerodromia	1	0	0	0	0	0	1
9/21/2011	Hexatoma	3	0	0	0	0	0	3
9/21/2011	Microvelia	1	0	0	0	0	0	1
9/21/2011	Stenacron	2	0	0	0	0	0	2
9/21/2011	Hydroptila	5	0	0	0	0	0	5
9/21/2011	Erioptera	1	0	0	0	0	0	1
9/21/2011	Chrysops	3	0	0	0	0	0	3
9/21/2011	Hydrobiidae	3	1	0	0	0	0	4
9/21/2011	Enochrus	2	0	0	0	0	0	2
9/21/2011	Cheumatopsyche	11	0	0	0	0	0	11
9/21/2011	Ceratopsyche morosa grp.	6	0	0	0	0	0	6

9/21/2011	Hydropsyche	2	0	0	0	0	0	2
9/21/2011	Belostoma	0	0	0	1	0	0	0
9/21/2011	Progomphus obscurus	0	0	0	1	0	0	0
9/21/2011	Orconectes virilis	0	0	0	1	0	0	0
9/21/2011	Ceratopogoninae	0	10	1	0	0	0	11
9/21/2011	Oxyethira	0	1	0	0	0	0	1
9/21/2011	Centroptilum	0	1	4	0	0	0	5
9/21/2011	Tipulidae	0	1	0	0	0	0	1
9/21/2011	Cyrnellus	0	1	0	0	0	0	1
9/21/2011	Enallagma	0	0	37	0	1	0	37
9/21/2011	Hyalella azteca	0	0	40	0	0	0	40
9/21/2011	Dubiraphia	0	0	48	0	0	0	48
9/21/2011	Paraleptophlebia	0	0	5	0	0	0	5
9/21/2011	Planariidae	0	0	4	0	0	0	4
9/21/2011	Macronychus glabratus	0	0	7	0	0	0	7
9/21/2011	Triaenodes	0	0	8	0	0	0	8
9/21/2011	Erythemis	0	0	1	0	0	0	1
9/21/2011	Heptageniidae	0	0	4	0	0	0	4
9/21/2011	Hydroptilidae	0	0	1	0	0	0	1
9/21/2011	Menetus	0	0	53	0	0	0	53
9/21/2011	Gomphidae	0	0	1	0	0	0	1
9/21/2011	Leptoceridae	0	0	1	0	0	0	1
9/21/2011	Libellulidae	0	0	1	0	0	0	1
9/21/2011	Macromia	0	0	0	0	0	1	0
9/21/2011	Basiaeschna janata	0	0	0	0	0	1	0
9/21/2011	Dromogomphus	0	0	0	0	0	1	0
9/21/2011	Tetragoneuria	0	0	0	0	0	1	0
9/21/2011	Orconectes	0	0	0	0	0	1	0
9/21/2011	Branchiura sowerbyi	1	0	1	0	0	0	2
9/21/2011	Demicryptochironomus	1	0	0	0	0	0	1
9/21/2011	Tanypodinae	2	0	4	0	0	0	6
9/21/2011	Dicrotendipes	5	1	1	0	0	0	7
9/21/2011	Polypedilum scalaenum grp.	1	0	0	0	0	0	1
9/21/2011	Procladius	1	4	0	0	0	0	5
9/21/2011	Rheotanytarsus	16	0	0	0	0	0	16
9/21/2011	Chironomus	2	5	0	0	0	0	7
9/21/2011	Thienemannimyia grp.	53	0	5	0	0	0	58

9/21/2011	Cricotopus/Orthocladius	4	0	1	0	0	0	5
9/21/2011	Tanytarsus	23	7	1	0	0	0	31
9/21/2011	Polypedilum convictum grp.	69	0	2	0	0	0	71
9/21/2011	Tubificidae	1	6	2	0	0	0	9
9/21/2011	Limnodrilus hoffmeisteri	1	0	0	0	0	0	1
9/21/2011	Stictochironomus	0	2	0	0	0	0	2
9/21/2011	Tanypus	0	1	0	0	0	0	1
9/21/2011	Cryptotendipes	0	2	0	0	0	0	2
9/21/2011	Parakiefferiella	0	4	0	0	0	0	4
9/21/2011	Pseudochironomus	0	1	0	0	0	0	1
9/21/2011	Stempellinella	0	7	0	0	0	0	7
9/21/2011	Labrundinia	0	1	9	0	0	0	10
9/21/2011	Cladotanytarsus	0	5	1	0	0	0	6
9/21/2011	Cryptochironomus	0	1	0	0	0	0	1
9/21/2011	Polypedilum halterale grp.	0	29	0	0	0	0	29
9/21/2011	Clinotanypus	0	0	1	0	0	0	1
9/21/2011	Nanocladius	0	0	1	0	0	0	1
9/21/2011	Polypedilum illinoense grp.	0	0	2	0	0	0	2
9/21/2011	Paratanytarsus	0	0	1	0	0	0	1
Site 3. Broadway Blvd								
9/22/2011	Argia	19	0	1	0	0	0	20
9/22/2011	Stenelmis	45	0	0	0	0	0	45
9/22/2011	Hydracarina	1	20	0	0	0	0	21
9/22/2011	Stenonema femoratum	7	0	0	0	0	0	7
9/22/2011	Berosus	1	0	1	0	0	0	2
9/22/2011	Tricorythodes	4	0	0	0	0	0	4
9/22/2011	Physa	29	12	62	0	0	0	103
9/22/2011	Hydrobiidae	5	0	1	0	0	0	6
9/22/2011	Ferrissia	10	4	36	0	0	0	50
9/22/2011	Planariidae	5	0	19	0	0	0	24
9/22/2011	Helicopsyche	2	0	0	0	0	0	2
9/22/2011	Hetaerina	6	1	0	0	0	0	7
9/22/2011	Chimarra	1	0	0	0	0	0	1
9/22/2011	Forcipomyiinae	4	0	0	0	0	0	4
9/22/2011	Cheumatopsyche	26	0	0	0	0	0	26
9/22/2011	Hydropsychidae	10	0	0	0	0	0	10

9/22/2011	Ceratopsyche slossonae	3	0	0	0	0	0	3
9/22/2011	Rhagovelia	2	0	0	0	0	0	2
9/22/2011	Hydropsyche	2	0	0	0	0	0	2
9/22/2011	Microvelia	2	0	0	0	0	0	2
9/22/2011	Stenacron	1	1	0	0	0	0	2
9/22/2011	Helichus fastigiatus	1	0	0	0	0	0	1
9/22/2011	Hydroptila	9	1	1	0	0	0	11
9/22/2011	Simulium	6	0	0	0	0	0	6
9/22/2011	Enochrus	3	0	0	0	0	0	3
9/22/2011	Hexatoma	3	0	0	0	0	0	3
9/22/2011	Silvius	1	0	0	0	0	1	1
9/22/2011	Caenis latipennis	6	21	6	0	0	0	33
9/22/2011	Caenis	3	11	5	0	0	0	19
9/22/2011	Dixidae	1	0	0	0	0	0	1
9/22/2011	Baetis	72	0	0	0	0	0	72
9/22/2011	Baetidae	18	0	0	0	0	0	18
9/22/2011	Hydroptilidae	1	0	0	0	0	0	1
9/22/2011	Chrysops	0	0	0	1	0	0	0
9/22/2011	Erioptera	0	0	0	1	0	0	0
9/22/2011	Sphaeriidae	0	11	2	1	0	0	13
9/22/2011	Corbicula	0	27	7	1	0	0	34
9/22/2011	Orconectes virilis	0	0	0	1	0	0	0
9/22/2011	Dubiraphia	0	6	8	0	0	0	14
9/22/2011	Ceratopogoninae	1	10	1	0	0	0	12
9/22/2011	Progomphus obscurus	0	1	0	0	0	0	1
9/22/2011	Centroptilum	0	4	0	0	0	0	4
9/22/2011	Enallagma	0	0	39	0	0	0	39
9/22/2011	Hyalella azteca	0	0	12	0	0	0	12
9/22/2011	Glossiphoniidae	0	0	4	0	0	0	4
9/22/2011	Oecetis	0	0	1	0	0	0	1
9/22/2011	Menetus	0	0	21	0	0	0	21
9/22/2011	Caenis amica	0	0	1	0	0	0	1
9/22/2011	Erpobdellidae	0	0	0	0	0	1	0
9/22/2011	Basiaeschna janata	0	0	0	0	0	1	0
9/22/2011	Perithemis	0	0	0	0	0	1	0
9/22/2011	Somatochlora	0	0	0	0	0	1	0
9/22/2011	Dromogomphus	0	0	0	0	0	1	0

9/22/2011	Hagenius brevistylus	0	0	0	0	0	1	0
9/22/2011	Branchiura sowerbyi	0	4	0	0	0	0	4
9/22/2011	Polypedilum convictum grp.	48	0	0	0	0	0	48
9/22/2011	Polypedilum illinoense grp.	23	0	7	0	0	0	30
9/22/2011	Cryptochironomus	4	12	0	0	0	0	16
9/22/2011	Larsia	1	0	1	0	0	0	2
9/22/2011	Cricotopus bicinctus	7	0	1	0	0	0	8
9/22/2011	Nilotanypus	1	0	0	0	0	0	1
9/22/2011	Paratendipes	1	0	0	0	0	0	1
9/22/2011	Tanypodinae	3	0	3	0	0	0	6
9/22/2011	Nilothauma	1	0	0	0	0	0	1
9/22/2011	Corynoneura	1	0	0	0	0	0	1
9/22/2011	Thienemanniella	1	0	0	0	0	0	1
9/22/2011	Cladotanytarsus	1	4	0	0	0	0	5
9/22/2011	Chironomus	1	37	0	0	0	0	38
9/22/2011	Dicrotendipes	5	10	8	0	0	0	23
9/22/2011	Polypedilum scalaenum grp.	2	0	0	0	0	0	2
9/22/2011	Rheotanytarsus	4	0	0	0	0	0	4
9/22/2011	Ablabesmyia	3	3	4	0	0	0	10
9/22/2011	Thienemannimyia grp.	28	0	1	0	0	0	29
9/22/2011	Cricotopus/Orthocladius	31	2	3	0	0	0	36
9/22/2011	Tanytarsus	123	6	29	0	0	0	158
9/22/2011	Tubificidae	6	24	1	0	0	0	31
9/22/2011	Cryptotendipes	0	29	0	0	0	0	29
9/22/2011	Tanypus	0	2	0	0	0	0	2
9/22/2011	Pseudochironomus	0	1	0	0	0	0	1
9/22/2011	Parakiefferiella	0	3	0	0	0	0	3
9/22/2011	Chironominae	0	2	0	0	0	0	2
9/22/2011	Stempellinella	0	2	0	0	0	0	2
9/22/2011	Limnodrilus hoffmeisteri	0	4	0	0	0	0	4
9/22/2011	Procladius	0	13	2	0	0	0	15
9/22/2011	Polypedilum halterale grp.	0	30	0	0	0	0	30
9/22/2011	Phaenopsectra	0	0	1	0	0	0	1
9/22/2011	Labrundinia	0	0	3	0	0	0	3
Site 4. Reactor Field, Replicate #1								
9/23/2011	Stenelmis	168	0	5	0	0	0	173

9/23/2011	Argia	27	0	30	0	0	0	57
9/23/2011	Physa	3	0	8	0	0	0	11
9/23/2011	Ceratopogoninae	1	0	2	0	0	0	3
9/23/2011	Tricorythodes	68	0	1	0	0	0	69
9/23/2011	Corbicula	7	3	0	0	0	0	10
9/23/2011	Planariidae	4	0	52	0	0	0	56
9/23/2011	Chimarra	18	0	0	0	0	0	18
9/23/2011	Acerpenna	17	0	1	0	0	0	18
9/23/2011	Ferrissia	1	2	13	0	0	0	16
9/23/2011	Calopteryx	1	0	0	0	0	0	1
9/23/2011	Berosus	1	0	0	0	0	0	1
9/23/2011	Heptageniidae	1	0	0	0	0	0	1
9/23/2011	Stenonema femoratum	1	0	0	0	0	0	1
9/23/2011	Hemerodromia	1	0	0	0	0	0	1
9/23/2011	Cheumatopsyche	47	0	0	0	0	0	47
9/23/2011	Hexatoma	2	0	0	0	0	0	2
9/23/2011	Hydropsychidae	16	0	0	0	0	0	16
9/23/2011	Hydroptilidae	1	0	0	0	0	0	1
9/23/2011	Silvius	4	0	0	0	0	0	4
9/23/2011	Simulium	3	0	0	0	0	0	3
9/23/2011	Caenis latipennis	47	0	4	0	0	0	51
9/23/2011	Caenis	21	0	5	0	0	0	26
9/23/2011	Calopterygidae	1	0	0	0	0	0	1
9/23/2011	Enochrus	1	0	0	0	0	0	1
9/23/2011	Baetis	5	0	0	0	0	0	5
9/23/2011	Baetidae	5	0	0	0	0	0	5
9/23/2011	Chrysops	0	0	0	1	0	0	0
9/23/2011	Progomphus obscurus	0	0	0	1	0	0	0
9/23/2011	Orconectes	0	0	0	1	0	0	0
9/23/2011	Empididae	0	1	0	0	0	0	1
9/23/2011	Hydrobiidae	0	1	1	0	0	0	2
9/23/2011	Perithemis	0	0	0	0	1	0	0
9/23/2011	Hydracarina	0	0	50	0	0	0	50
9/23/2011	Dubiraphia	0	0	13	0	0	0	13
9/23/2011	Hetaerina	0	0	1	0	0	0	1
9/23/2011	Enallagma	0	0	43	0	0	0	43
9/23/2011	Planorbidae	0	0	2	0	0	0	2

9/23/2011	Epicordulia	0	0	2	0	0	0	2
9/23/2011	Coenagrionidae	0	0	4	0	0	0	4
9/23/2011	Tipula	0	0	1	0	0	0	1
9/23/2011	Macromia	0	0	2	0	0	0	2
9/23/2011	Sphaeriidae	0	0	0	0	0	1	0
9/23/2011	Belostoma	0	0	0	0	0	1	0
9/23/2011	Dromogomphus	0	0	0	0	0	1	0
9/23/2011	Branchiura sowerbyi	1	1	1	0	0	0	3
9/23/2011	Diptera	0	1	0	0	0	0	1
9/23/2011	Polypedilum scalaenum grp.	1	0	0	0	0	0	1
9/23/2011	Polypedilum convictum grp.	10	0	0	0	0	0	10
9/23/2011	Cricotopus/Orthocladius	46	0	2	0	0	0	48
9/23/2011	Cricotopus bicinctus	23	0	0	0	0	0	23
9/23/2011	Tanypodinae	3	0	1	0	0	0	4
9/23/2011	Labrundinia	1	0	6	0	0	0	7
9/23/2011	Parametriocnemus	1	0	0	0	0	0	1
9/23/2011	Nilotanypus	1	0	1	0	0	0	2
9/23/2011	Tubificidae	8	128	10	0	0	0	146
9/23/2011	Limnodrilus hoffmeisteri	1	3	0	0	0	0	4
9/23/2011	Tanytarsus	13	0	1	0	0	0	14
9/23/2011	Polypedilum illinoense grp.	6	0	2	0	0	0	8
9/23/2011	Cryptochironomus	2	0	0	0	0	0	2
9/23/2011	Rheotanytarsus	1	0	0	0	0	0	1
9/23/2011	Thienemannimyia grp.	24	0	7	0	0	0	31
9/23/2011	Polypedilum halterale grp.	1	0	0	0	0	0	1
9/23/2011	Dicrotendipes	0	1	1	0	0	0	2
9/23/2011	Procladius	0	2	1	0	0	0	3
9/23/2011	Chironomus	0	1	0	0	0	0	1
9/23/2011	Ablabesmyia	0	0	5	0	0	0	5
9/23/2011	Nanocladius	0	0	1	0	0	0	1
Site 4. Reactor Field, Replicate #2								
9/23/2011	Stenelmis	168	0	5	0	0	0	173
9/23/2011	Argia	27	0	30	0	0	0	57
9/23/2011	Physa	3	0	8	0	0	0	11
9/23/2011	Ceratopogoninae	1	0	2	0	0	0	3
9/23/2011	Tricorythodes	68	0	1	0	0	0	69

9/23/2011	Corbicula	7	7	0	0	0	0	14
9/23/2011	Planariidae	4	0	52	0	0	0	56
9/23/2011	Chimarra	18	0	0	0	0	0	18
9/23/2011	Acerpenna	17	0	1	0	0	0	18
9/23/2011	Ferrissia	1	0	13	0	1	0	14
9/23/2011	Calopteryx	1	0	0	0	0	0	1
9/23/2011	Berosus	1	0	0	0	0	0	1
9/23/2011	Heptageniidae	1	0	0	0	0	0	1
9/23/2011	Stenonema femoratum	1	0	0	0	0	0	1
9/23/2011	Hemerodromia	1	0	0	0	0	0	1
9/23/2011	Cheumatopsyche	47	0	0	0	0	0	47
9/23/2011	Hexatoma	2	0	0	0	0	0	2
9/23/2011	Hydropsychidae	16	0	0	0	0	0	16
9/23/2011	Hydroptilidae	1	0	0	0	0	0	1
9/23/2011	Silvius	4	0	0	0	0	0	4
9/23/2011	Simulium	3	0	0	0	0	0	3
9/23/2011	Caenis latipennis	47	0	4	0	0	0	51
9/23/2011	Caenis	21	0	5	0	0	0	26
9/23/2011	Calopterygidae	1	0	0	0	0	0	1
9/23/2011	Enochrus	1	0	0	0	0	0	1
9/23/2011	Baetis	5	0	0	0	0	0	5
9/23/2011	Baetidae	5	0	0	0	0	0	5
9/23/2011	Chrysops	0	0	0	1	0	0	0
9/23/2011	Progomphus obscurus	0	0	0	1	0	0	0
9/23/2011	Orconectes	0	0	0	1	0	0	0
9/23/2011	Hydrobiidae	0	0	1	0	0	0	1
9/23/2011	Perithemis	0	0	0	0	1	0	0
9/23/2011	Libellulidae	0	0	0	0	1	0	0
9/23/2011	Hydracarina	0	0	50	0	0	0	50
9/23/2011	Dubiraphia	0	0	13	0	0	0	13
9/23/2011	Hetaerina	0	0	1	0	0	0	1
9/23/2011	Enallagma	0	0	43	0	0	0	43
9/23/2011	Planorbidae	0	0	2	0	0	0	2
9/23/2011	Epicordulia	0	0	2	0	0	0	2
9/23/2011	Coenagrionidae	0	0	4	0	0	0	4
9/23/2011	Tipula	0	0	1	0	0	0	1
9/23/2011	Macromia	0	0	2	0	0	0	2

9/23/2011	Sphaeriidae	0	0	0	0	0	1	0
9/23/2011	Belostoma	0	0	0	0	0	1	0
9/23/2011	Dromogomphus	0	0	0	0	0	1	0
9/23/2011	Branchiura sowerbyi	1	1	1	0	0	0	3
9/23/2011	Polypedilum scalaenum grp.	1	0	0	0	0	0	1
9/23/2011	Polypedilum convictum grp.	10	0	0	0	0	0	10
9/23/2011	Cricotopus/Orthocladius	46	1	2	0	0	0	49
9/23/2011	Cricotopus bicinctus	23	0	0	0	0	0	23
9/23/2011	Tanypodinae	3	0	1	0	0	0	4
9/23/2011	Labrundinia	1	0	6	0	0	0	7
9/23/2011	Parametriocnemus	1	0	0	0	0	0	1
9/23/2011	Nilotanypus	1	0	1	0	0	0	2
9/23/2011	Tubificidae	8	244	10	0	0	0	262
9/23/2011	Limnodrilus hoffmeisteri	1	1	0	0	0	0	2
9/23/2011	Tanytarsus	13	0	1	0	0	0	14
9/23/2011	Polypedilum illinoense grp.	6	0	2	0	0	0	8
9/23/2011	Cryptochironomus	2	0	0	0	0	0	2
9/23/2011	Rheotanytarsus	1	0	0	0	0	0	1
9/23/2011	Thienemannimyia grp.	24	0	7	0	0	0	31
9/23/2011	Polypedilum halterale grp.	1	0	0	0	0	0	1
9/23/2011	Dicrotendipes	0	0	1	0	0	0	1
9/23/2011	Procladius	0	3	1	0	0	0	4
9/23/2011	Stictochironomus	0	1	0	0	0	0	1
9/23/2011	Ablabesmyia	0	1	5	0	0	0	6
9/23/2011	Nanocladius	0	0	1	0	0	0	1
Site 5. Scott Blvd								
9/24/2011	Argia	11	1	64	0	0	0	76
9/24/2011	Ferrissia	2	0	10	0	0	0	12
9/24/2011	Stenacron	19	1	6	0	0	0	26
9/24/2011	Stenonema femoratum	10	0	0	0	0	0	10
9/24/2011	Acerpenna	41	0	0	0	0	0	41
9/24/2011	Heptageniidae	1	0	1	0	0	0	2
9/24/2011	Baetis	24	0	0	0	0	0	24
9/24/2011	Baetidae	19	0	0	0	0	0	19
9/24/2011	Caenis	8	2	5	0	0	0	15
9/24/2011	Caenis latipennis	38	7	10	0	0	0	55

9/24/2011	Simuliidae	4	0	0	0	0	0	4
9/24/2011	Simulium	6	0	0	0	0	0	6
9/24/2011	Centroptilum	1	0	0	0	0	0	1
9/24/2011	Hydropsychidae	2	0	0	0	0	0	2
9/24/2011	Cheumatopsyche	1	0	0	0	0	0	1
9/24/2011	Planorbidae	1	0	7	0	0	0	8
9/24/2011	Enochrus	3	0	0	0	0	0	3
9/24/2011	Paracymus	2	0	0	0	0	0	2
9/24/2011	Stenelmis	4	0	0	0	0	0	4
9/24/2011	Sphaeriidae	0	2	0	0	0	0	2
9/24/2011	Hydracarina	0	28	3	0	0	0	31
9/24/2011	Dubiraphia	0	2	25	0	0	0	27
9/24/2011	Hydroptila	0	1	0	0	0	0	1
9/24/2011	Corbicula	0	1	0	0	0	0	1
9/24/2011	Erioptera	1	2	0	0	0	0	3
9/24/2011	Dromogomphus	0	1	0	0	0	0	1
9/24/2011	Oecetis	0	4	0	0	0	0	4
9/24/2011	Rheumatobates	0	1	0	0	0	0	1
9/24/2011	Corixidae	0	1	0	0	0	0	1
9/24/2011	Copelatus	0	0	1	0	0	0	1
9/24/2011	Hydrochus	0	0	9	0	0	0	9
9/24/2011	Trepobates	0	0	2	0	0	0	2
9/24/2011	Orconectes	0	0	1	0	0	0	1
9/24/2011	Microvelia	0	0	2	0	0	0	2
9/24/2011	Mesovelia	0	0	2	0	0	0	2
9/24/2011	Epicordulia	0	0	4	0	0	0	4
9/24/2011	Menetus	0	0	6	0	0	0	6
9/24/2011	Physa	0	0	2	0	0	0	2
9/24/2011	Basiaeschna janata	0	0	1	0	0	0	1
9/24/2011	Ceratopogoninae	0	0	2	0	0	0	2
9/24/2011	Neoplea	0	0	5	0	0	0	5
9/24/2011	Macronychus glabratus	0	0	1	0	0	0	1
9/24/2011	Enallagma	0	0	17	0	0	0	17
9/24/2011	Nasiaeschna pentacantha	0	0	1	0	0	0	1
9/24/2011	Hyalella azteca	0	0	2	0	0	0	2
9/24/2011	Coenagrionidae	0	0	2	0	0	0	2
9/24/2011	Belostoma	0	0	0	0	0	1	0

9/24/2011	Branchiura sowerbyi	0	5	10	0	0	0	15
9/24/2011	Polypedilum illinoense grp.	156	1	5	0	0	0	162
9/24/2011	Dicrotendipes	18	2	22	0	0	0	42
9/24/2011	Polypedilum convictum grp.	28	0	0	0	0	0	28
9/24/2011	Cricotopus bicinctus	10	0	0	0	0	0	10
9/24/2011	Glyptotendipes	24	4	32	0	0	0	60
9/24/2011	Tribelos	1	1	4	0	0	0	6
9/24/2011	Cryptochironomus	3	16	1	0	0	0	20
9/24/2011	Larsia	2	0	0	0	0	0	2
9/24/2011	Thienemanniella	11	0	0	0	0	0	11
9/24/2011	Corynoneura	7	0	0	0	0	0	7
9/24/2011	Parachironomus	1	0	2	0	0	0	3
9/24/2011	Chironominae	2	2	0	0	0	0	4
9/24/2011	Tanypodinae	2	0	0	0	0	0	2
9/24/2011	Labrundinia	1	0	0	0	0	0	1
9/24/2011	Orthocladiinae	1	0	0	0	0	0	1
9/24/2011	Nilotanypus	1	0	0	0	0	0	1
9/24/2011	Lumbriculidae	1	0	0	0	0	0	1
9/24/2011	Tubificidae	24	58	5	0	0	0	87
9/24/2011	Limnodrilus hoffmeisteri	1	3	0	0	0	0	4
9/24/2011	Chironomus	4	32	0	0	0	0	36
9/24/2011	Cladotanytarsus	2	2	0	0	0	0	4
9/24/2011	Polypedilum scalaenum grp.	5	0	0	0	0	0	5
9/24/2011	Rheotanytarsus	3	0	1	0	0	0	4
9/24/2011	Ablabesmyia	20	3	3	0	0	0	26
9/24/2011	Thienemannimyia grp.	4	1	0	0	0	0	5
9/24/2011	Cricotopus/Orthocladius	5	1	0	0	0	0	6
9/24/2011	Tanytarsus	30	5	16	0	0	0	51
9/24/2011	Cryptotendipes	0	4	0	0	0	0	4
9/24/2011	Polypedilum halterale grp.	0	15	1	0	0	0	16
9/24/2011	Procladius	0	38	3	0	0	0	41
9/24/2011	Paratanytarsus	0	0	4	0	0	0	4