THE IMPACT OF URBANIZATION ON MACROINVERTEBRATE COMMUNITIES OF CREEKS WITHIN WESTERN GEORGIA

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ABSTRACT

Urbanization is a major cause of stream impairment in the United States, altering stream ecological integrity in a variety of ways. While control of point source pollution has largely improved over the last twenty years, the control of non-point source pollution has proved to be more of a challenge. Urbanization in the area surrounding streams has been linked with elevated levels of sediment, heavy metals, organic matter, and nutrients within streams, as well as with other negative effects. Examining the macroinvertebrate communities within streams has proven to be an effective indicator of the effects of urbanization on water quality. This study used that concept to evaluate the health of nine sites on three tributaries of the Chattahoochee River around and within Columbus, Georgia, for the effects of such urbanization. The results of this study indicated that the sites on the less urbanized Upatoi Creek had the healthiest representation of benthic macroinvertebrates, and consequently they were classified as having good to fair water quality, an attribute that needs to be conserved. In contrast, the lower and upper sites on Standing Boy Creek, which is located in a developing urban area, had fairly poor water quality, and were most in need in remediation efforts. To a lesser extent, the lower and middle sites on Bull Creek, which had notably higher percent urban land use than all other sites, also had somewhat degraded aquatic communities within them as well. Despite the less urbanized Upatoi Creek sites having comparatively superior water quality, no significant correlation was found between percent urbanization and any changes in metric values. Presumably, many factors in addition to urbanization caused the differences in macroinvertebrate populations found between sites. Other factors that

likely acted in addition to percent urban land use to affect the benthic macroinvertebrate communities were percent agricultural land use and an ongoing drought. The ongoing drought in the area appeared to affect biotic values the most, as those values increased as sampling continued throughout the year, indicating that water quality was decreasing. Comparisons between sites were further complicated by physical differences among sites and a lack of a significant gradient in percent land use between the majority of sites. Due to the many factors influencing the water quality within these nine locations, additional biomonitoring efforts are suggested to further specify the exact effects of the increasing urbanization and other factors on streams within the city of Columbus and its suburbs.

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Introduction

Urbanization continues to impair aquatic systems worldwide. As human populations increase and expand, they have dramatically altered streams and other bodies of water globally. Up to 83% of the people in the Americas and Europe are expected to be living in urban or suburban areas by the year 2025 (Sheehan 2001). In the United States alone, metropolitan areas currently occupy 19% of the total land surface and more than 75% of the American population lives in these urbanized areas (Stoel 1999, Mitchell 2001). The United States Environmental Protection Agency (EPA) (2000) has classified over 130,000 kilometers of streams and rivers in the United States as impaired due to urbanization. Urbanization is ranked as the second major cause of stream impairment, falling behind only agriculture, despite the fact that the total amount of area covered by urbanization is minor in comparison to the amount of area covered by agriculture (Paul and Meyer 2001). Thus, urban areas exert a disproportionate effect on water quality that spreads beyond the geographical boundaries of the urbanization (Baer and Pringle 2000). The prospect of increasing urban runoff, and its associated pollution problems, is becoming a greater focus of regulatory agencies.

One obvious pollution problem associated with increasing urbanization is the discharge of sewage and wastes into streams. Less than twenty years ago, these point sources of pollution were the major source of water quality problems in urban streams. There has been a somewhat successful effort to control this nationwide. Today, most developed countries have sewage collection systems and modern treatment processes that have brought direct discharges of these pollutants under more control (Jones and Clark 1987). However, in some areas of the United States, point source pollution may still be a

problem. Treatment systems fail and the permitted discharge limits are exceeded (Paul and Meyer 2001). During the late 70's, urban wastes were often discharged into storm sewers and streams without treatment, and some of these discharges have been difficult to locate and correct. A study in North Carolina in the late 1970's reported that only 56% of the identified discharging facilities were in compliance with final effluent limits (North Carolina Division of Environmental Management 1979). Seven urban streams in North Carolina appeared to have been effected by discharges such as these (Duda *ĕt al*. 1982). When studying the restoration of Strawberry Creek in California, Charbonneau and Resh (1992) found that some of the older sewer lines in that area were deteriorating and managing to infiltrate into adjacent storm sewer lines that were emptying into Strawberry Creek and affecting water quality. Overflows from combined sewers, leaking or broken sanitary sewers, illicit discharge connections, failing septic systems, and sewer systems that are cross connected to storm sewers have all been noted to be the cause of the introduction of raw sewage to urban streams (Duda *et al*. 1982, Faulkner *et al*. 2000). Johnson *et al*. (1999) estimated that a volume of more than 193,000 m^3/year of illicit untreated sewage was discharged into the Rouge River catchments in the Detroit, Michigan area.

While progress has been made in the United States in controlling point-source pollution problems, non-point source pollution from agriculture and urbanization is emerging as an even larger component of the water quality problem. Peterson *et al*. (1985) stated that non-point source pollution was one of the most pervasive, persistent, and diverse water quality problems in the United States. This type of pollution often remains unregulated, and in one study done in North Carolina, was identified as the worst

and most widespread cause of problems to streams (North Carolina Department of Natural Resources and Community Development 1979). The major source of non-point source pollution is urban storm water and surface runoff. Urban storm water runoff has been cited as causing the greatest diversity of pollution to streams, being the most difficult pollution problem to assess, and being the most challenging one to correct (Benke *et al*. 1981).

As an area becomes urbanized, there is an increase in impervious surface area as roads, paved parking lots, and roofs on houses are constructed; this directly affects the amount of pollutants washed into a stream and increases the amount of storm water runoff that drains into waterways (Benke *et al*. 1981, Limberg and Schmidt 1990, Weaver and Garman 1994, McMahon and Cuffney 2000, Paul and Meyer 2001). The impervious surface area does not perform the same function as the natural vegetation in aiding in the purification of the polluted waters before they reach the stream (Benke *et al*. 1981). Schueler (1994) reported that the total runoff volume for a parking lot with an area of 4047 meters^2 (one-acre) was almost sixteen times that calculated for the same area of undeveloped meadows. Paul and Meyer (2001) noted that an impervious surface area of ten to twenty percent often marked the threshold for degradation in urban streams. This increase in surface runoff causes multiple problems that can affect the streams (Benke *et al*. 1981, Limberg and Schmidt 1990, Weaver and Garman 1994).

One of these potential problems is that increased impervious surface area increases the speed and the volume of the surface runoff. In Catalonia, Spain, the runoff-to-rainfall ratio was found to be 50% higher than before extensive urbanization of that area occurred (Sala and Imbar 1992). Hydrographs in developed areas reflect the

increases in volume and speed of runoff by exhibiting a higher number and magnitude of peak discharges (Morisawa 1985, Charbonneau and Resh 1992), an increase in total annual flow (Knight 1979), and an earlier occurrence of peak discharges after rainfall begins (Goudie 1981). These factors also result in an increased volume of flood flows during storm periods, and a decrease in low flow volume during non-storm periods (Klein 1979, Simmons and Reynolds 1982). When there is a decrease in discharge volume during the non-storm periods, available stream habitat is decreased, the stream is at a higher likelihood of drying out, diurnal temperature fluctuations could result, and increased concentration of pollutants due to the lack of dilution could occur (Whipple *et al.* 1981, Simmons and Reynolds 1982). The frequency of floods is also increased due to the deforestation that occurs with urbanization (Whipple *et al.* 1981, Sala and Inbar 1992). At Strawberry Creek in California, the higher storm flows and lower dry weather baseflows changed the hydrology of the creek by accelerating channel downcutting, increasing stream bank erosion, and destroying the natural pool-riffle sequence (Charbonneau and Resh 1992). More severe flooding, accelerated channel erosion, altered stream channel form, and changes in bed composition were also noted by Klein (1979) to occur in response to changes in land use due to urbanization. Even in urban areas that are not paved over, the soil is compacted to the extent that it does not have the high infiltration rates associated with forested areas (Dunne and Leopold 1978). McMahon and Cuffney (2000) clearly demonstrated that percent imperviousness was an accurate predictor of urbanization and its effects on streams.

Associated with increases in storm water runoff in urbanized areas are increases in suspended sediment loads in streams and rivers (Johnson *et al.* 1993). Leopold (1968)

found that the degree to which an area was urbanized had a direct effect on the sediment load in that area. The suspended sediment loads in urbanized areas have often been found to be 10 to 100 times greater than those from forested areas in the same location (Randall *et al.* 1978, Rhoads 1995). This increase in sediment load in urban rivers and streams can have several effects on aquatic populations. Bottom habitat is smothered by extra sedimentation. In addition, this smothering and the limited light penetration reduce food supply, and physiological functions of benthic organisms, such as feeding and reproduction, can be impaired (Mangun 1989). Construction activities in particular have been found to cause sediment loading to the extent of eliminating habitat and interfering with feeding for fish and aquatic invertebrates (Reed 1977). Pitt and Bozeman (1980) found a significant relationship between quality of aquatic communities in urban streams and amounts of suspended sediment, and Johnson *et al.* (1993) found that this increased sediment load altered the benthic macroinvertebrate communities through changes in food availability and utilization.

Streams flowing through developed areas may be subject to other types of human disturbance, which can cause physical alterations. They may be straightened or dredged to increase their aesthetic value or for development purposes. In addition, bridges are built across many of them (Elliott *et al.* 1997). If the riparian vegetation surrounding the stream is altered, the temperature of the stream could change due to the decrease or increase in shading, which would consequently affect autotrophic production. Also, the vegetation in this area is responsible for providing the stream with allochthonous detritus in the form of course particulate matter (Hachmoller *et al.* 1984). Any change in the

composition or availability of this food source would have the potential to echo through the entire food web (*sensu* the River Continuum, Vannote *et al.* 1981).

Many pollutants are carried along with urban runoff directly into lotic systems. Urbanization is linked with elevated levels of heavy metals, nutrients, and organic matter (Klein 1985, Garie and McIntosh 1986, Elliott *et al.* 1997). Atmospheric fallout and washout of air pollutants, road surface and vehicular pollutants, street litter, animal wastes, and lawn and garden chemicals contribute to this problem (Duda *et al.* 1982, Muschak 1990). Lead, zinc, mercury, chromium, copper, manganese, nickel, and cadmium are examples of heavy metals that have been observed in increased concentrations at urban sites (Wilber and Hunter 1979, Porcella and Sorenson 1980, Charbonneau and Resh 1992). Non-point sources of these metals appear to be more common than point sources in urbanized areas (Mason and Sullivan 1998). Some such sources include brake linings, tires, and engine parts (Muschak 1990, Mielke et al. 2000). Frick et al. (1998) found a significant positive correlation between the concentrations of heavy metals in bed sediment in streams and the amount of industrial land use. All of these metals can affect stream life, reaching levels in urban runoff that could potentially cause death of macroinvertebrates (Water Planning Division, 1983). Macroinvertebrates such as mollusks, arthropods, and annelids from urbanized areas have exhibited elevated metal levels in their tissues (Rauch and Morrison 1999, Gundacker 2000). Rauch and Morrison (1999) found that the organisms' responses to metal concentrations included reduced abundances and altered community structures. Even when the levels of metal concentrations are below set standards, they are suspected to causes changes in urban stream communities (Duda *et al.* 1982, Garie and Macintosh 1986).

Toxic organic compounds can reach dangerous concentrations for macroinvertebrate life in urban streams as well (Water Planning Division 1983, Klein 1985). Pesticides are frequently detected at levels above that set in guidelines for the protection of aquatic biota (USGS 1999, Hoffman *et al.* 2000). Surprisingly, the concentrations of many of the organochloride-based insecticides in urban sediments are often higher than those recorded in agricultural areas in the United States, due to high use around homes, gardens, parks, and commercial areas (USGS 1999). Other organic contaminants that are often detected in urban streams in amounts potentially damaging to macroinvertebrates include polychlorinated biphenyls (PCBs), petroleum-based aliphatic hydrocarbons, and polycyclic aromatic hydrocarbons (Moring and Rose 1997, Frick *et al.* 1998).

Elevated levels of nutrients such as phosphorous and nitrogen may result from urbanization, and while they are not usually as directly harmful to aquatic communities as metals or toxic organic compounds, they can alter stream life as well. Phosphorous and nitrogen were found at levels two to ten times greater in urban areas than in forested areas in the same location (Burton *et al.* 1977, Grizzard *et al.* 1978). Fertilizer is a common source of such nutrients, as is wastewater (LaValle 1975). As nutrient levels are elevated, algal growth increases where light is available, and the aquatic food web can be changed (Jones and Clark 1987, Elliott *et al.* 1997). Other pollutants can also be washed into streams along with the urban storm water runoff, but those discussed above are some of the more common and known causes of problems.

Traditionally, the principal method for assessing effects of water quality on streams has been to use a variety of water chemistry tests and compare the results with

standards set by the state or federal regulatory agencies. This method is still used, but in some cases, it may be ineffective in determining the full extent of pollution problems in waters. Karr (1995) lists several reasons why evaluating water quality using chemical criteria is not effective. First, the biological components of the water resources of this nation are in steep decline, indicating that depending upon chemical technology to identify problems has not been effective. Second, the degradation of water systems may not always be caused by chemical contamination, and therefore would not be identified by chemical testing. Third, the laws and regulations that apply to the water resources do not allow for a timely response, particularly because they focus mainly on wastewater control and human cancer risks. Fourth, long-term success in restoring and protecting aquatic systems requires the development of end points that incorporate biological parameters as well as chemical ones. Finally, the biological health of the nation's waters varies geographically, and applying chemical criteria uniformly is not effective. Duda *et al.* (1982) add that some of the water quality criteria and standards in effect now are not scientifically sound or they may not be strict enough to effectively protect the aquatic biota.

A more recent approach to this problem, which may be more effective in many cases, is to judge the quality of the water by monitoring the state of its aquatic communities. This method is termed "biomonitoring" and uses the biological responses of organisms in the water to evaluate changes in the system (Rosenberg and Resh 1993). As Benke *et al.* (1981) wrote, "The ultimate criterion for stream degradation is whether or not a natural community of aquatic organisms is able to exist." If the aquatic populations are in their natural balanced state, then the water quality standards are likely

being met (Duda *et al.* 1982). Yoder (1991) found that assessments using biota accurately identified the presence of human influence almost 50% of the time when that influence was not identified by examining the chemical water quality data available. More specifically, Wang *et al.* (1997) and others (Garie and Macintosh 1986, Elliot *et al.* 1997) found that the biomonitoring method indicated that there were water quality problems in streams even when chemical tests of the water showed no real basis for this. This research supports the idea that if aquatic communities are changed from their natural state, water quality of the stream is likely to blame.

When using a biomonitoring approach to judging water quality, macroinvertebrates have been shown to be the ideal indicator organisms to study the effects of urbanization (Duda *et al.* 1982, Johnson 1993). Aquatic macroinvertebrates are found in almost all types of aquatic systems and in almost all variations of habitats within those systems (Rosenberg and Resh 1993). The relatively long length of their life cycles provides long-term exposure to toxic substances in comparison with other aquatic organisms such as zooplankton. Many benthic macroinvertebrates have only one generation per year, and a few, such as some Megaloptera, Odonata, and Plecoptera, have larval or nymphal aquatic stages that live up to five years. Therefore, they tend to show the effects of long-term water quality and not just instantaneous conditions (Johnson 1993). In addition, most benthic macroinvertebrates spend much of their time in contact with the sediment, which tends to accumulate the excess nutrients and toxins that are responsible for much of the environmental degradation of aquatic systems. Benthic macroinvertebrates have limited mobility and cannot easily move in order to avoid toxic discharges; providing an effective spatial analysis of pollution or disturbance effects

(Rosenberg and Resh 1993, Bode and Novak 1994). Benthic macroinvertebrates bioaccumulate and biomagnify some toxins, such as heavy metals and pesticides (Reice and Wohlenberg 1993). They are a key link in the food webs of aquatic systems since they prey on lower life forms, they help process organic matter, and they are preyed upon by higher life forms such as fish (Duda *et al.* 1982). If their population structures are affected, then the other stream populations, and community and ecosystem structure, should be affected as well.

Invertebrates are known to have various responses to water quality challenges, with some types of invertebrates being very pollution intolerant while others are pollution tolerant. Therefore, pollution can affect macroinvertebrate community structure in a variety of ways. It can cause a change in species composition, productivity, trophic pathways, or species interactions, among other things (Benke *et al.* 1981). Invertebrate families such as Ephemeroptera, Plecoptera, and Trichoptera (the so-called EPT's) are known to be among the most sensitive to pollution as can be seen by the use of the EPT index to assess pollution (Lenat 1988, Lenat and Crawford 1994, Baker and Sharp 1998, Helms *et al.* 2003). A lack of these EPT taxa in streams indicates low water quality. In contrast, some invertebrates actually thrive in certain pollution conditions, and have in some cases developed mechanisms such as specialized blood, respiratory tubes, or other adaptations that allow them to exist in the low dissolved oxygen levels that often occur in polluted streams. A high number of chironomids or oligocheates, especially in combination with a low number of the individuals from more pollution intolerant groups, is a good indication that the stream is impacted by human disturbances (Garie and McIntosh 1986, Fore *et al.* 1996). Determining presence, absence, and abundance of

certain groups of invertebrates in contrast with others can give biologists an idea of the health of the water system, and, in some cases, what type of pollution is causing the differences in community diversity (Lenat 1988, Gibert *et al.* 1995). In some circumstances, the overall abundance of invertebrates in a stream can be unaffected by or actually increase due to certain forms of pollution, such as an excess of inorganic or organic nutrients or sludge deposits. With these types of pollutants, the standing crop of the pollution tolerant groups of invertebrates will grow and dominate, causing the increase in biomass. However, the overall diversity of the community will decrease in response to pollutants such as those, as the sensitive species will die off (Resh and Grodhaus 1983, Jones and Clark 1987). Also, organisms with short life cycles may not be as affected by water pollution as the more long-lived groups (Gibert *et al.* 1994). Therefore, many factors must be taken into account when analyzing the invertebrate communities in streams, lakes, and rivers.

Using invertebrates to assess water quality does have some inherent difficulties. Invertebrates may not respond to all impacts. Some cases have been cited where invertebrate populations were virtually unaffected, but the chemical tests and analysis of plant species indicated the detrimental effects of pollution (Rosenberg and Resh 1993). The responses of invertebrates could also vary due to a combination of stressors acting synergistically or antagonistically. Other natural factors in an environment could cause results similar to the invertebrates' response to pollution, such as the destruction of a food source or substrate through a temperature change or turbidity. Physical differences between sites such as substrate type or current velocity can also affect invertebrate community structure and abundance (Resh and Grodhause 1983). More practical

concerns and disadvantages occur due to the sampling procedures use. Generally a high number of replications are required for precision, which means that it requires large amounts of money and time. Processing samples and identifying invertebrates is also time-consuming (Resh and Grodhaus 1983, Rosenberg and Resh 1993). The taxonomic keys needed to identify the invertebrates are lacking or incomplete for some groups, such as Diptera and Trichoptera. Even when keys are available, identifying invertebrates down to the species level is often difficult and may produce uncertain results (Resh and Grodhaus 1983, Hilsenhoff 1987). Also, while most biologists will agree that studies on the aquatic fauna are essential, they do not all agree on how to analyze and interpret the data from a water quality standpoint (Benke et al. 1981, Norris and Hawkins 2000).

Despite some difficulties, many studies have demonstrated the effectiveness of using biomonitoring of aquatic invertebrate communities to detect the deleterious effects of urbanization on water quality. Benke *et al.* (1981) used this approach to study several different streams in the Atlanta, Georgia area. The chemical tests that he performed did not indicate any major levels of pollution, but he did find a highly significant negative relationship between the degree of urbanization in the areas he studied and the abundance of aquatic invertebrate species in those streams. Duda *et al.* (1982) performed a similar study on streams in North Carolina. For this study, the forested upstream sites of two streams were compared with sites further downstream that were in urbanized areas. They found that the diversity of benthic macroinvertebrates was reduced by 70 to 80% in the urban areas in comparison to the upstream controls. The control sites contained a good mix of Ephemeroptera, Plecoptera, and Trichoptera, but the downstream sites consisted mostly of the worms and midges that are capable of tolerating various types of pollution.

In most of their urbanized study sites, representatives from the families of the Ephemeroptera, Plecoptera, and Trichoptera were not just low in numbers; they were completely absent. Duda *et al* (1982) determined that organic wastes and toxic substances were the cause of the differences in community structure, and consequently discovered the presence of broken and leaking sanitary sewers, small illegal discharges of wastes, and periodic dumping of oil and other pollutants in those areas.

Following Duda's study, Jones and Clark (1987) studied the effects of urbanization on the invertebrate biota in 22 sites in north Virginia. Their results mirrored those of the earlier studies. The relative abundance of Ephemeroptera, Coleoptera, Megaloptera, Plecoptera, and Odonata were negatively correlated with the degree of urbanization. Only dipteran abundance was positively correlated with urbanization. The total number of insects was not significantly affected by urbanization, but diversity and richness were much greater in the less urbanized streams. Certain species of caddisflies, mayflies, and beetles were virtually absent in the streams in areas of moderate to heavy urbanization. In a similar study, Mangun *et al.* (1989) sampled streams in north Virginia and used the Shannon-Weaver index (Shannon and Weaver 1963) to correlate increased urbanization with decreased species diversity in the benthic macroinvertebrate communities. At the more urbanized sites, a large number of individuals of one species were found, but there was little or no representation of other species. At the less urbanized sites, a more even spread in the numbers of individuals was found across several species. In other studies, Plecoptera was the most sensitive group of invertebrates to human influences. Therefore, they were the first group of invertebrates to disappear within a stream when water quality was being affected by pollution (Woodiwiss 1978,

Fore *et al.* 1996). The disappearance of Plecoptera was followed by the loss of Ephemeroptera and then Trichoptera, in that order (Woodiwiss 1978). Mangun also reported this pattern with Plecoptera being absent from all but the most heavily forested areas, and Ephemeroptera being low at the urbanized sites. Trichoptera, which tend to fill the void created by the absence of Plecoptera and Ephemeroptera, were comparatively more numerous at all sites surveyed (Mangun *et al.* 1989).

Lenat and Crawford (1994) studied three streams in North Carolina, one located within a forested area, one in an agricultural area, and one in an urbanized area. Using EPT taxa richness to study the macroinvertebrate communities produced bioclassifications of "good" for the forested site, "fair" for the agricultural site, and "poor" for the urban site. In the urban stream, taxa richness decreased for nine taxonomic groups, and only increased for one group - the tolerant Oligochaeta. In comparison with the forested stream, total taxonomic richness decreased by 52 to 58%, and EPT taxa richness decreased by 76 to 84%. Lenat and Crawford also found that while there were 75 unique taxa at the forested site, there were only nine unique taxa at the urban site, all of which were limited to the groups of Oligochaeta and Diptera. While some differences were found in chemical and physical tests between the urban and forested sites in this regard, Lenat and Crawford did not believe that they were sufficient to account for the differences found in the macroinvertebrate communities, which suggests that there was some unmeasured toxicity.

More recently, Morley and Karr (2002) examined the water quality of urban streams in Puget Sound, Washington. The streams incorporated into this study were evaluated using the benthic invertebrate index of biological integrity (B-IBI), which

includes ten metric values within it to measure the diverse effects of urbanization. In this study, percent urban land cover was strongly associated with decreased B-IBI, indicating that the water quality was affected by urbanization in those areas. As could be expected, the percent impervious area was also correlated with the B-IBI to a lesser extent. In the most urbanized study site, a total of only fifteen taxa were found in the samples, and no representatives from Plecoptera were found. Out of the 15 taxa, only one long-lived taxon was found. Almost 90% of the samples from this site were made up of amphipods, chironomids, and a tolerant mayfly genus. Another stream included in this survey produced its highest B-IBI value at the site with the lowest percentage of urban land cover, while the lowest B-IBI score was linked with the site with the highest amount of urban land cover (Morley and Karr 2002). Wang *et al.* (1997) showed that watersheds with as little as 10 to 20% urban land cover were consistently correlated with lowered biotic integrity scores. Many other studies support the results described above and link changes in invertebrate species richness and diversity with urbanization of streams (Pratt *et al.* 1981, Garie and Macintosh 1986, Elliot *et al.* 1997, Kemp and Spotila 1997, Wang *et al.* 1997, Baker and Sharp 1998, Walsh *et al.* 2001).

In summary, as urbanization increases rapidly in the United States, the health of the stream systems are put more and more at risk. Discharge of sewage; increased surface and storm water runoff; increased sedimentation; high levels of nutrients, organic matter, and other toxins and physical alteration of the habitat are problems associated with urbanization of the surrounding area (Benke *et al.* 1981, Duda *et al.* 1982, Jones and Clark 1987, Johnson *et al.* 1993, Elliott *et al.* 1997). In the past, water quality was primarily tested for via chemical analysis, but, more recently, judging the health of the

water by monitoring the state of its aquatic invertebrate communities has become a useful alternative (Rosenberg and Resh 1993). This approach has been shown to indicate pollution problems when chemical analysis did not, and, therefore, may show pollution effects before chemical tests do (Benke *et al.* 1981, Jones and Clark 1987, Garie and MacIntosh 1986, Lenat and Crawford 1994, Wang *et al.* 1997). Therefore, in urban streams that have been polluted, not only will the diversity of species be decreased, but also the organisms that are known to be sensitive to pollution will be decreased in abundance and in richness.

The objective of the current study is to apply biomonitoring practices to nine sites located on three streams in the Columbus, Georgia, area to determine if decreased diversity or differences within the community composition of the macroinvertebrate populations indicate that they have been affected by the urbanization occurring around them. The expected outcome is that various biological measures used to evaluate the macroinvertebrate communities at these sites will indicate water quality problems at the more urbanized locations. Those same measures should indicate less or no water quality problems at the sites facing less impact from the encroaching urbanization. This study was part of a larger project that examined the potential impacts of wet-weather events upon the Chattahoochee River and its tributaries. In this larger study, not only were the effects of urbanization being studied, but also the effects of other types of land alterations such as agricultural use and impoundments. One of the main goals of this larger project was to assess how land use within Columbus is affecting the source waters for the area's drinking water supplies, as well as measuring the overall health of the Middle Chattahoochee River system. Studying the invertebrate communities of the creeks

included in this study was one of the ways to assist in reaching these goals, as well as accomplishing the goal of understanding specific effects of urbanization on invertebrate life. Such research was necessary to assess current and historical efforts towards solving water quality problems. By analyzing the effects urbanization has on the diversity and community composition of the invertebrate populations at sites on three Chattahoochee River tributaries, this study will evaluate the extent to which the water quality in these streams has been affected.

Study Sites

Macroinvertebrate samples were obtained from three streams for this study - Bull Creek, Upatoi Creek, and Standing Boy Creek (Figure 1). All three are tributaries of the middle reaches of the Chattahoochee River. Each stream was sampled at three locations to represent upper, middle, and lower reaches of each stream. In general, the amount of urbanization surrounding the streams increased as each stream approached confluence with the Chattahoochee River. The exception to this was found at Standing Boy Creek, where the middle site sampled actually had a slightly higher percentage of urbanization in the area surrounding it than the lower site did. Based on GIS data provided for this study by John Olson of Columbus State University (personal communication 1999), the Bull Creek sites overall were subject to the highest amounts of urbanization and its presumed effects. The sites on Standing Boy Creek were within areas with slightly less urbanization, and the sites on Upatoi Creek were in general within areas with the lowest amount of urbanization. Invertebrates were collected from all three sites on each creek during June 1998, October 1998, February 1999, May 1999, and July 1999. All samples were collected from areas in the creek at least 50 meters upstream of the road crossing by which each site was accessed. Informative data concerning land use in the areas surrounding each stream were also provided by John Olson (personal communication 1999). These comparative characteristics are summarized in Table 1.

Figure 1: Map of Study Sites for Middle Chattahoochee Watershed Study
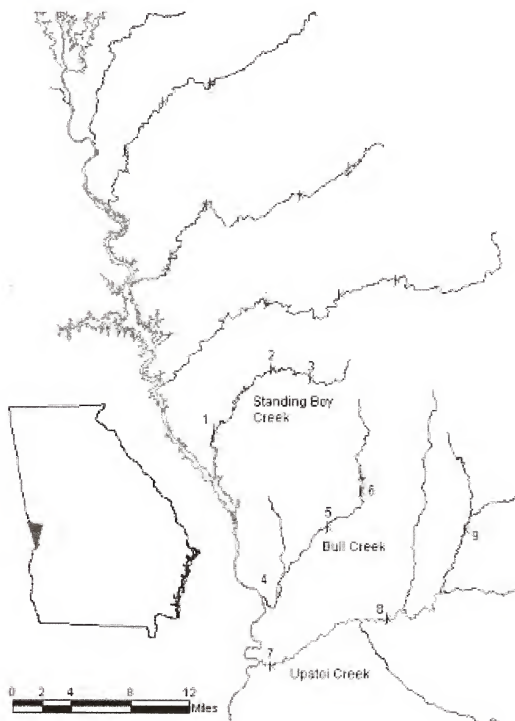
Table 1: Physiographical features and land use patterns for sites on Standing Boy Creek, Upatoi Creek, and Bull Creek

Site	Catchment Area (m^2)	% Urban	% Forest	% Agricultural
Upper Standing Boy	25,855,605	0.63	75.11	16.79
Middle Standing Boy	59,623,148	0.73	84.83	10.37
Lower Standing Boy	118,169,784	0.68	87.05	7.13
Upper Bull Creek	31,193,554	1.21	86.95	11.72
Middle Bull Creek	93,386,354	10.3	79.35	9.40
Lower Bull Creek	170,162,810	28.81	62.85	7.36
Upper Upatoi Creek	99,429,044	0.3	80.64	7.28
Middle Upatoi Creek	884,170,940	0.62	77.93	5.18
Lower Upatoi Creek	1,163,738,576	1.81	79.46	4.57

Standing Boy Creek enters the Chattahoochee at the northernmost site in relation to the other two creeks. Overall, Standing Boy Creek was chosen to represent a stream that does potentially face some water quality problems due to the increasing amount of urbanization occurring in the areas surrounding it. However, due to a lesser amount of urbanization in the areas surrounding this creek, it was expected to show lessened effects in comparison to Bull Creek's lower and middle sites. Standing Boy Creek runs through two counties - the northernmost portion of Muscogee County and the southern portion of Harris County. Both counties are facing intensifying amount of development as the city

of Columbus expands northward and the population of Harris County increases as a result.

The uppermost site sampled on Standing Boy Creek was accessed at the point where the stream flows under US Road 27 in Harris County, GA. Based on the GIS data, 75% of the 26 km^2-area encompassed by this site is forested, with only 0.63% of the surrounding land occupied by urbanized areas. In addition, agricultural uses were highest at this site compared to the others, occupying 16.79% of the area. The agricultural uses potentially could have a further impact on this section of the stream. Data concerning land use patterns for the sites on this creek are detailed in Table 1. The substrate for this portion of the stream was sand and embedded cobble.

The site representing the middle reaches of Standing Boy Creek was located approximately 6 kilometers downstream of the upper site, at the crossing of Fortson Road in Muscogee County, GA. This site covers an area of about 60 km^2, with over 84% of the site being forested. Compared to the previous site, agricultural uses occupied less of the land (slightly over 10%), but the amount of urbanization increased slightly to 0.73%. The substrate at the middle site consisted of sand and medium to small cobble.

The lowest site sampled on Standing Boy Creek was located in Muscogee County, about 9 km downstream of the middle site. It was situated about 4 km upstream of Lake Oliver, and immediately upstream of the smaller Biggers Lake. This location was problematic when sampling as efforts to collect macroinvertebrates were hindered due to waters from Lake Oliver backing up to this portion of the stream, resulting in muddy standing water on all sampling occasions. Therefore, qualitative samples were collected via net only due to lack of current and increased depth of the water, making the

quantitative Hess sample an unviable option. The substrate at this area was largely sand and silt. This portion of Standing Boy Creek was accessed via Biggers road, within 100 meters of the cross section of Biggers and River Road. More than 87% of the 118 km^2-area adjacent to this section of the creek is forested. The amount of surrounding area occupied by urban uses dropped slightly from the previous site to 0.68%. Agricultural use occupies less than 8% of the area.

Bull Creek enters the Chattahoochee River approximately 15 kilometers south of the confluence of the river with Standing Boy Creek. This stream faces the biggest threat from urban runoff as it flows directly through the southern portion of the city of Columbus, Georgia. Urbanization varies from a little over 1% to almost 29% at the most downstream site. Therefore, this stream was chosen to denote a stream that is regularly subject to runoff from the urbanized areas around it. All sites sampled for Bull Creek were located within Muscogee County, Georgia. Specific data concerning land use patterns at each site is listed in Table 1.

The uppermost site on Bull Creek was located at its crossing under US Highway 80. This site encompassed a drainage area of 31 km^2, with almost 87% of that area left in its natural forested state, and 1.2% of it under urban use. In comparison to the two lower sites, this location faces the lowest risk from urban runoff and its effects. However, even at this uppermost site, the amount of urbanization found surrounding Bull Creek clearly increased in comparison from that surrounding Standing Boy Creek. Agricultural uses again occupied a significant percentage of the land in the surrounding areas at 11.72%. The substrate at this part of the creek consisted of sand, gravel, and cobble.

Bull Creek's middle sampling site was accessed about 5 km from the upper site where Woodruff Farm Road passes over the creek. The amount of urbanization in this area increased considerably up to 10.3% in the 93 km^2 surrounding it. Seventy-nine percent of the surrounding land is covered with forest, and 9.4% is used for farming purposes. This site was well within the city limits, but slightly to the east of the main portion of the city. Sand, gravel, and small cobble made up the substrate of this sampling site.

The final and lowest site on Bull Creek was located right at the core of the city of Columbus, directly past where Buena Vista Road and St. Mary's Road intersect and approximately ten kilometers downstream of the middle site. The urbanization in this area is the highest of all sites sampled at almost 29%, while the percentage of forested land dropped to 63%. The drainage area encompassed by this site was 170 km^2. The substrate in this area was sand, gravel, and cobble. This site was visibly affected by the urbanization around it, as a significant amount of litter and foul odors were noted at this site, with glass, tires, and plastic frequently found within the creek bed. Convenience stores and other commercial property are visible immediately around this area.

Upatoi Creek was designated as the stream facing the lowest risk from urban impacts. It enters the Chattahoochee River over 9 km south of Bull Creek. Upatoi Creek forms the boundary between Muscogee and Chattahoochee County for much of its length, but the upper portion of the stream extends into Talbot County as well. This stream is larger than Bull Creek and Standing Boy Creek, a factor that had to be considered during analysis of the collected data. This stream flows through areas with little urbanization, with the exception of most downstream site, which flows under a road

that is an access point to Ft. Benning, an army base directly south of the city of Columbus. As such, this stream served as the "reference" stream, as it faces the fewest challenges to water quality. Most of this stream flows through training areas for Ft. Benning, with few public roads, commercial facilities, or housing found in the surrounding areas. Therefore, the amount of pollution emptying into the creek due to urban runoff should have remained low for much of the length of the stream.

The northernmost sampling site on Upatoi Creek was accessed where US Highway 80 crosses over it. This site was located in Talbot County, near the town of Box Springs, and covered an area of 99 km^2. This site is found in the area with the lowest amount of urbanization surrounding it- only 0.3%. Therefore, it served as the standard for a stream that is subject to only minor urban effects. The size of the stream at this point is more comparable to the other six sites as well, while the two downstream sites of Upatoi were noticeably larger. Eighty-six percent of the surrounding area is forested. During the year the samples were taken however, this site was subjected to the effects of road construction on Highway 80, which likely impacted the macroinvertebrate communities. The substrate at this location consisted of sand, gravel, and medium cobble.

The middle sampling site on Upatoi Creek was located on the Ft. Benning Military Reservation where it flows under "McBride's Bridge" on First Division Road, in close proximity to the intersection of First Division Road and Second Armored Division Road. This site was approximately 29 km downstream of the upper site. The area encompassed by this site was 884 km^2, with 0.62% of the land here dedicated to urban uses. Almost 78% of the area is forested, and a little over 7% of the land dedicated to agricultural use. This site should have faced minimum challenges to water quality from

urbanization, although it may have been susceptible to impacts from military training. The substrate in this area was sand and gravel. The width and depth of this site was markedly increased in comparison to the other locations.

The most downstream site on Upatoi Creek was about 14 km south of the middle site, but still located within Ft. Benning Military Reservation. This site was accessed from Ft. Benning Road, close to where it intersects with Tenth Armored Division Road. The width and depth of this section of Upatoi was again considerably greater than those found at the Standing Boy and Bull Creek sites, as well as the uppermost site of the Upatoi itself. During one sampling effort, the depth was too great for any samples other than net samples to be collected. Distinguishable riffle/run areas and pool/glide areas were not recognizable on most sampling occasions. The amount of urbanization here increased to 1.81%. Seventy-nine percent of the surrounding area was forested and 4.57% under agricultural uses. The substrate consisted of sand with small amounts of gravel.

Methods

Macroinvertebrates were sampled quarterly for approximately a year during June 1998, October 1998, February 1999, May 1999, and July of 1999. Since the study progressed through all the seasons, both "winter" (long-generation) and "summer" (short generation) invertebrates were sampled. This method has been shown to result in somewhat higher taxonomic richness than sampling from a single season or date (Lenat 1988). Both qualitative and quantitative samples were collected when possible, although the quantitative samples on the lower sites on Standing Boy Creek and Upatoi Creek were not collected on several occasions due to either lack of current or depth of the water.

For qualitative samples, kick net samples were taken using a D-ring net. Kick net samples were found to be superior to and produce more consistent results than Surber samplers and artificial substrates in a study done by Hornig and Pollard (1978). One qualitative sample at each site was taken from a riffle/run area, and the other was taken from a pool/glide area. This method was used to result in a variety of organisms from the differing habitats (Lazorchak and Klemm 1997). At the lower sites on both Upatoi Creek and Standing Boy Creek, often there was no series of riffle/run areas interspersed with pool/glide areas due to the depth of the water there. Despite this, kick net samples at these sites were taken using the available habitat.

A 0.1 m^2 Hess bottom sampler was employed to collect the quantitative samples at each site. These samples differed from the qualitative samples described above because area sampled was known and constant between samples. Hauer and Resh (1996) have shown Hess samplers to better represent densities than Surber samplers due to wash around. Bottom samples were collected in the shallow riffle areas at each site. Hynes

(1970) reports that riffle areas supported the most diverse communities because of the substrate variability found there. Four quantitative samples were taken at each site. As noted previously, at the lower sites, there were no riffle run areas available during some of the sampling efforts made at the lower Standing Boy Creek site and the lower Upatoi Creek site. The lack of riffle areas, the lack of a current, and the depth of the water at these sites prevented the Hess sampler from being of use. As a result only qualitative dip net samples were taken at these sites on such occasions.

Samples collected using a D-ring net and a Hess bottom sampler were placed in separate Ziploc bags and preserved using 70% ethanol. All samples were labeled with names of the collectors, date, and sample reference number. Measurements of velocity, depth, and type of substrate were recorded while at each site. Once samples reached the laboratory, macroinvertebrates were sorted by hand from remaining mud and debris. Invertebrates were identified down to the lowest taxonomic level possible using various keys (Merritt and Cummins 1996, Epler 1995, Thorp 1991, Brigham *et al.* 1982, Pennak 1978, Edmunds *et al,* 1976). All chironomid species were mounted on slides with CMC-10 mounting medium in order to facilitate microscopic identification.

Total abundance and richness were calculated at each site and season. Due to large differences among abundances from each site, rarefaction was used to standardize richness for sample size (Krebs 1989). To account for differences among macroinvertebrate species' tolerance to pollution, Hilsenhoff's biotic index (1987) was used. It assigns each species a pollution tolerance value ranging from 0 (intolerant) to 10 (tolerant) (Barbour *et al.* 1999), and therefore was useful to identify sites with high richness and abundance values due to high numbers of these pollution tolerant

invertebrates and not due to superior water quality. Organic pollution and physical disturbances increase the value of Hilsenhoff's index (Hilsenhoff 1987, Barbour *et al.* 1999). Simpson's index was used to measure diversity of macroinvertebrates at each site. This index is primarily influenced by increases and decreases in the abundance of the dominant species within the macroinvertebrate communities (Krebs 1989). The nonparametric Mann-Whitney U Test was employed to detect any significant differences among the diversity index values calculated for each site and season.

The percentage of EPT taxa and individuals was calculated; this percentage should decrease as the effect of urbanization increases (Barbour *et al.* 1999). In addition, percentages of individuals and taxa belonging to family Chironomidae were also calculated. Chironomid metrics are expected to show an opposite effect from EPT metrics, and hence increase directly with increases in urbanization since many Chironomidae are tolerant of certain types of pollution (Garie and McIntosh 1986, Jones and Clark 1987, Fore *et al.* 1996).

Morisita's similarity measure was employed to compare the macroinvertebrate communities from the nine sites. It is relatively unaffected by sample size and is touted as the best overall similarity measure to use in ecological studies (Krebs 1989). Hierarchical, agglomerative cluster analysis using average linkage was utilized to further compare the species composition found at each site. Spearman's rank coefficient was employed to determine if percentages of urbanization at each site or other land use characteristics could be negatively or positively correlated with various metrics and values calculated.

Results

Between June of 1998 and July of 1999, 8137 macroinvertebrates belonging to 284 taxa were collected during sampling efforts. Appendix 1 details the species collected and provides abundances at each of the nine stream sites during the five sampling dates. Highest macroinvertebrate abundance was collected at the lower Bull Creek site, totaling 1657 individuals from 80 species. At the other end of the spectrum, the lower Upatoi Creek site produced the lowest abundance of macroinvertebrates with 158 individuals collected from 54 species. While the middle Standing Boy Creek site did not have the highest abundance of macroinvertebrates collected, it did produce the highest number of species with 121 taxa identified out of the 1297 macroinvertebrates collected. The lowest site on Standing Boy Creek had the lowest richness value with 32 species collected. Abundance and richness values are summarized in Table 2.

Table 2: Abundance and number of taxa collected for each site

Site	Abundance	Richness
Lower Bull Creek	1657	80
Middle Bull Creek	1266	83
Upper Bull Creek	1504	79
Lower Standing Boy Creek	257	32
Middle Standing Boy Creek	1297	121
Upper Standing Boy Creek	604	79
Lower Upatoi Creek	158	54
Middle Upatoi Creek	295	51
Upper Upatoi Creek	1099	114

Sample size differences among the nine sites were extreme, ranging from fewer than 200 to over 1500 macroinvertebrates collected. This indicates that comparing richness values among sites would produce unreliable results, since abundance and richness are positively correlated (Krebs 1998). A rarefaction method was used to standardize all the samples to a common sample size of the same number of individuals. A rarefaction curve was plotted based upon data from the site with the highest abundance (lower Bull Creek). Using this curve, the numbers of invertebrates actually collected at the other eight sites were compared with the number of species expected to be collected at lower Bull Creek had the number of total invertebrates obtained been the same. Ninety-five percent confidence intervals were used to verify whether any differences noted between the actual and expected richness was significant. Figure 2 shows a graphical representation of the rarefaction curve. The rarefaction curve and confidence intervals reveal that almost all sites would have had a significantly higher number of species collected in comparison to the number expected to be collected at the lower Bull Creek site if the sample sizes had been approximately equal. Lower Standing Boy Creek site was the exception to this as the only site to have a significantly lower number of species collected than what would have been expected from the lower Bull Creek site. Also, there was no significant difference found between richness at the lower Bull Creek site and either the middle Upatoi site or the upper Bull Creek site.

Figure 2: Rarefaction curve based upon the lower Bull Creek site

◇ No. Adjusted with Rarefaction
● Actual No. Collected at Standing Boy Creek
▲ Actual No. Collected at Bull Creek
■ Actual No. Collected at Upatoi Creek
✕ Upper 99% Confidence Interval
- Lower 99% Confidence Intervals
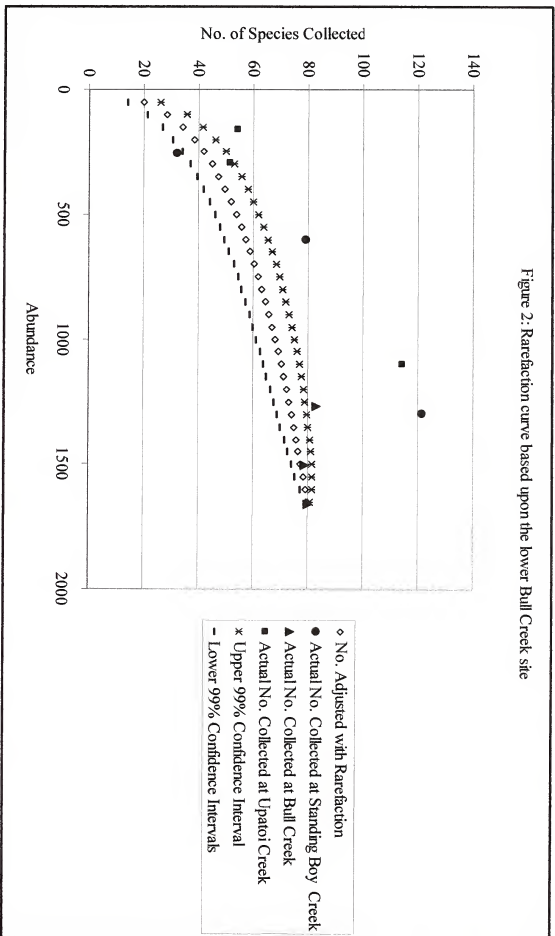
Hilsenhoff's biotic index (Hilsenhoff 1987, Barbour *et al.* 1999) calculations for the combined samples from each site ranged from 4.827 to 7.209. The middle Upatoi Creek site yielded the lowest value (indicating the highest water quality), and the lower Standing Boy Creek site produced the highest value (indicating the lowest water quality). Biotic index values assigned to each species are listed in Appendix 2. An average biotic index value was calculated for each site. These values, along with the corresponding water quality categories designated by Hilsenhoff's classification system (1987), are listed in Table 3. Using this classification system, the water quality at only two sites, middle and lower Upatoi creek, was ranked as "good", indicating that some impairment existed but that the comparatively highest water quality out of the sites occurred at these locations. Most other sites ranked as "fair" according to Hilsenhoff's system, denoting that fairly significant levels of impairment persisted at these sites. Two of the sites on Standing Boy Creek, the lower and upper sites, were categorized as having "fairly poor" water quality. This classification indicates significant impairment at those two sites, as well as verifying that the water quality at those sites was significantly inferior in comparison to the other six sites.

Table 3: Hilsenhoff's biotic index values calculated for each site with the corresponding water quality classifications

Site	Hilsenhoff Biotic Index Value	Classification
Lower Bull Creek	6.237	Fair
Middle Bull Creek	6.128	Fair
Upper Bull Creek	5.966	Fair
Lower Standing Boy Creek	7.209	Fairly Poor
Middle Standing Boy Creek	6.060	Fair
Upper Standing Boy Creek	6.565	Fairly Poor
Lower Upatoi Creek	5.360	Good
Middle Upatoi Creek	4.827	Good
Upper Upatoi Creek	5.839	Fair

In addition to biotic index values, Simpson's diversity indices were calculated for each location. High diversity is an indication of superior community integrity. To some degree, this measure substantiated the results of biotic index calculations, with Upatoi Creek sites having relatively higher diversity within them than did the other sites. Middle Standing Boy Creek site could also be included with those sites, as its diversity index value was identical to the middle Upatoi Creek site's value. Conversely, three Bull Creek sites and the two remaining Standing Boy Creek sites scored toward the lower end of the scale. Index values ranged from 0.893 at the middle Bull Creek site to 0.961 at the lower Upatoi Creek site. Figure 3 shows the diversity values for each site.

Figure 3: Simpson's Diversity Index Values for Each Site
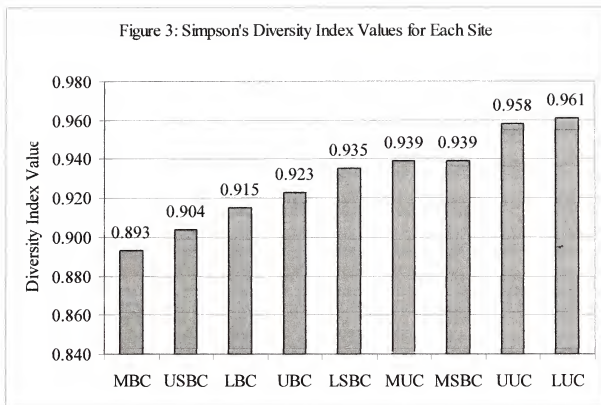
Significance of the differences noted among index values was assessed using the nonparametric Mann Whitney U test. This test indicated that the macroinvertebrate communities at the upper and lower Upatoi Creek sites were significantly more diverse than those communities sampled at the three Bull Creek sites (p=0.0159, Mann Whitney U Statistic = 0). In addition, the middle Upatoi Creek site was also more diverse than the middle Bull Creek site (p=0.0317, Mann Whitney U Statistic = 1). No other differences in diversity index values between sites proved significant.

Composition of macroinvertebrate populations collected was also evaluated to assess ecological integrity at each site. Of all macroinvertebrates collected, dipterans, mainly those belonging to family Chironomid, comprised almost 51% when all samples from all sites were combined. One hundred twenty-one species of Diptera were

collected, with 94 of those being chironomid species. Ephemeroptera made up a little over 15% of the entire sample with 35 species collected, and the 20 species of Trichoptera collected made 13% of the collection. The introduced exotic bivalve, *Corbicula fluminea.* was also very abundant at some sites and comprised almost 13% of the combined samples. No other species of bivalves were collected. Separately, all other macroinvertebrate orders comprised less than 2% of the collection.

Upon examining populations at each site, richness and percent composition of three groups of macroinvertebrates were examined: EPT taxa, plecopteran taxa (included as a component of the EPTs but also considered separately), and chironomid taxa. Abundance and diversity of the EPT taxa should exhibit a negative relationship with impairment. Abundance of the Plecopterans alone was examined because they have been considered the most sensitive group of those included in the EPT index when reacting to environmental stressors (Woodiwiss 1978, Mangun *et al.* 1989, Fore *et al.* 1996). Abundance and richness within the chironomids was measured, as this family is reported to dominate in urban streams with poor water quality (Duda et al. 1982, Lenat and Crawford 1994). The results of these measures are recorded in Table 4.

Table 4: Selected indicator metrics for macroinvertebrate communities from study sites

Site	No. EPT taxa	% EPT individuals	No. Plecop-teran taxa	% Plecop-teran individuals	No. Chir-onomid taxa	% Chironomid individuals
Lower Bull Creek	17	21.25	2	0.24	38	55.76
Middle Bull Creek	18	21.69	0	0	30	40.13
Upper Bull Creek	17	21.52	1	0.07	31	32.71
Lower Standing Boy Creek	4	12.5	0	0	19	79.38
Middle Standing Boy Creek	29	23.97	5	1	43	47.57
Upper Standing Boy Creek	14	17.72	3	0.83	33	67.05
Lower Upatoi Creek	15	27.78	4	3.8	21	39.24
Middle Upatoi Creek	18	35.29	7	11	22	36.27
Upper Upatoi Creek	39	34.21	14	5.64	43	42.77

The number and percent of plecopterans is comparatively high in all three Upatoi Creek sites. Plecoptera were particularly diverse (14 species) in the upper Upatoi Creek site, while the middle Upatoi Creek site produced the highest overall percentage of these macroinvertebrates at 11% of the total community composition collected. In comparison,

the Bull Creek sites and the lower Standing Boy site all either range from having zero to two species of Plecopterans found within the samples, and these invertebrates make up less than 0.3% of the community composition in all four sites. These findings support the low diversity and high biotic index scores for these sites discussed previously.

Examining the EPT taxa within the samples produced less distinctive results, as equally as many EPT taxa existed in the Bull Creek sites as did in the lower and middle Upatoi sites. However, some similarities can be seen. The lowest EPT richness (twelve species) and percentage (over twelve percent) was found at the lower Standing Boy site, which emphasizes again the low biotic integrity found at this site. The site that produced the highest number of EPT species was upper Upatoi Creek site with 39 EPT species identified within its samples. The middle Upatoi Creek site had fewer EPT taxa collected, but had the highest percentage of EPT individuals collected at 35%. Both of these sites have been indicated to have comparatively high community integrity by the previous measures as well. The percent EPT taxa is high for all three Upatoi Creek sites. The percent composition metrics are particularly noteworthy in that they demonstrate relative abundances and thus are not affected by sample size.

Chironomid metrics calculated did not produce any obvious trends, as chironomids comprised a large proportion of the macroinvertebrate communities at all sites. The number of taxa was high for Bull Creek sites as might be expected from the higher amounts of urbanization at most of those sites, but the number of taxa was even higher at the middle Standing Boy and upper Upatoi Creek sites. Chironomids were highly dominant at the lower Standing Boy site, making up almost 80% of all individuals collected, even though only nineteen chironomid species were included in those samples.

Collections from the lower Bull Creek site, middle Standing Boy Creek site, and upper Standing Boy Creek site also were quite high, consisting of greater than 45% Chironomidae in all cases. The lowest percent composition of chironomids was calculated for the upper Bull Creek site with only 32% of the macroinvertebrates collected belonging to this family.

Spearman's rank coefficient test was used to determine whether percent urbanization in the areas surrounding the sampling locations was correlated with any of the various metrics incorporated into this study. Only one metric demonstrated a positive correlation with percent urbanization, denoted by an r^2 value greater than 0.5. That metric was the percent composition of Plecopterans, which decreased as the amount of urbanization rose. A graphical representation of this metric can be seen in Figure 4. However, this correlation was not significant (p= 0.120). Additionally, Spearman's rank correlation test was also used to determine if any of the other stream attributes listed in Table 1 were linked with changes in metric values. Percent agriculture demonstrated a positive relationship with abundance and chironomid richness, and a negative relationship with Simpson's diversity index values. The catchment area was directly related to Simpson's index values as well. However, once again, none of these correlations proved to be significant. The r^2 and p values for each of these tests are listed in Table 5.

Figure 4: Correlation between Percent Urbanization and Percent Plecoptera
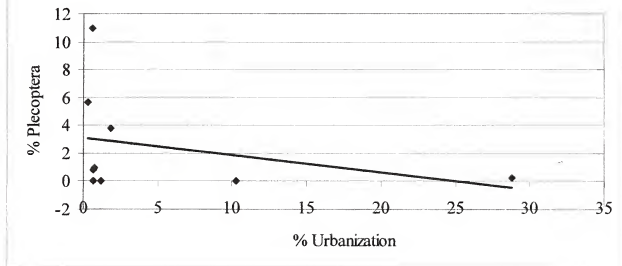
Table 5: Results of Spearman Rank Coefficient tests for determining correlation between the variables listed and percent urbanization, percent forested, percent agriculture, or catchment area.

Variable	Effect of % Urbanization		Effect of % Forested		Effect of % Agriculture		Effect of Area (m^2)	
	rs statistic	two-tailed p-value	rs statistic	two-tailed p-value	rs statistic	two-tailed p-value	rs statistic	two-tailed p-value
Abundance	0.42	0.265	-0.15	0.7001	0.63	0.0671	-0.45	0.2242
Richness	-0.33	0.380	-0.08	0.8305	0.48	0.1942	-0.43˜	0.252
Biotic Index Value	0.23	0.546	0.02	0.9961	0.47	0.2054	-0.43	0.244
Simpson's Index	-0.39	0.300	0.35	0.3537	-0.66	0.0525	0.54	0.1301
% EPT	-0.28	0.460	-0.02	0.9661	-0.23	0.5457	0.18	0.6368
% Plecoptera	-0.55	0.120	-0.25	0.5003	-0.39	0295	0.37	0.3296
% Chironomidae	-0.07	0.865	-0.07	0.8647	0.17	0.6682	-0.17	0.6682
EPT Richness	-0.21	0.587	-0.02	0.9658	0.03	0.9467	-0.04	0.9145
Plecopteran Richness	-0.60	0.086	-0.18	0.6511	-0.26	0.5003	0.22	0.5739
Chironomid Richness	-0.12	0.764	-0.19	0.6198	0.54	0.1301	-0.47	0.2032

Morisita's similarity index values were calculated for the nine sites. The most similar sites were the middle and upper Bull Creek sites (Morisita's similarity coefficient = 0.84), as might be expected from their proximity. The least similarity occurred between the lower Standing Boy site and the middle, lower, and upper Upatoi Creek sites (Morisita's similarity coefficient = 0.08, 0.09, and 0.15 respectively). The remainder of the results were scattered with no apparent pattern. Cluster analysis results were also largely inconclusive. The lower two sites on Upatoi Creek grouped together as would

have been expected, but the other seven sites were not clustered with sites on the same

creek or with other upstream or downstream sites (Figure 5).

Figure 5: Dendogram produced via cluster analysis using agglomerative average linkage clustering of all nine sites.
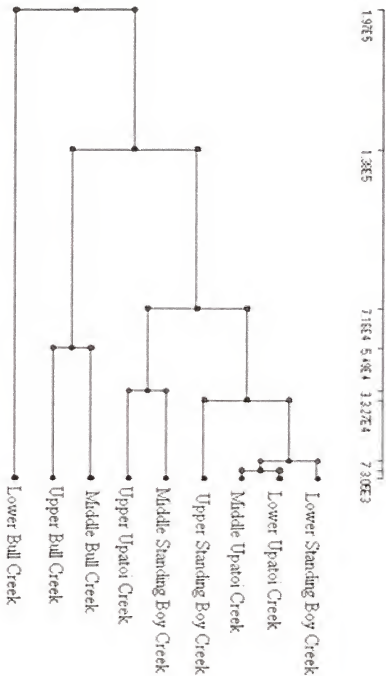
Time of year during which each creek was sampled was also examined as a possible influencing factor on macroinvertebrate communities, based primarily on the fact that rainfall, temperature, and other possible differences between sampling efforts may have affected abundance, richness, or composition. An ongoing drought persisted throughout the year of sampling, and likely resulted in much of the variation seen in the macroinvertebrate communities at each site over the five seasons. The metric values calculated for the combined samples for each season are recorded in Table 6.

Table 6: Summary of metrics calculated for the combined samples for each of the five seasons.

Metrics	June 1998	October 1998	February 1999	May 1999	July 1999
Hilsenhoff's Biotic Index	5.315	6.012	6.314	6.165	6.326
Abundance	1130	1044	1427	3451	1085
Total No. of Taxa	100	122	127	107	89
Simpson's Diversity Index	0.922	0.916	0.932	0.941	0.941
No. of EPT taxa	32	26	31	30	21
% of EPT individuals	63.36	20.3	18	25.04	29.22
No. of Plecopteran taxa	0	4	14	9	2
% of Plecopteran individuals	0	1.05	3.71	1.65	0.28
No. of Chironomid taxa	34	44	50	47	33
% of Chironomid individuals	20	39.75	46.88	58.42	42.67

The most apparent trend noted in the metric scores across the seasons was the positive correlation between biotic index values and sampling time progression (r^2=0.9,

p= 0.0374). The lowest value for the biotic index was calculated for June of 1998, and it increased throughout the progression of sampling periods to its maximum value in the following July. One discrepancy in this pattern did exist, as the value for May dipped lower than the value calculated for February. According to Hilsenhoff's (1987) correlation between these values and water quality, all but the June 1998 sample fell in the "fair" water quality category, indicating fairly significant amounts of organic pollution persisted in the streams. The sample for June 1998 was considered to be of significantly better water quality and hence fell into the category of "good." A graphical representation of this correlation can be seen in Figure 6. When the biotic values for each site were examined separately, in general they followed the same increasing pattern, with the noticeable exception of the lower Upatoi Creek site. As this site had increased discharge in comparison to most other sites, perhaps it was less consistently affected by the drought. Biotic scores for the three sites on Bull Creek, Standing Boy Creek, and Upatoi Creek over the five sampling efforts are graphically represented in Figures 7, 8, and 9 respectively. Other metric values calculated did not demonstrate as clear a pattern.

Figure 6: Correlation between Biotic Index Values and Seasons
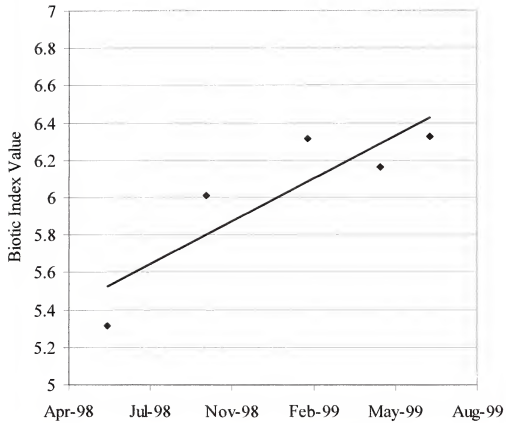
Figure 7: Correlation of Biotic Index Scores with Sampling Season for Bull Creek Sites
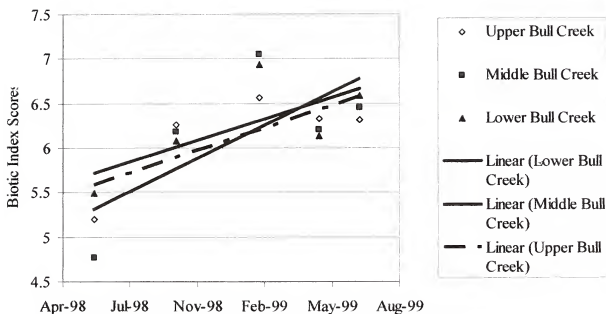
Figure 8: Correlation of Biotic Index Scores with Sampling Season for Standing Boy Creek Sites

Figure 9: Correlation of Biotic Index Scores with Sampling Season for Upatoi Creek Sites
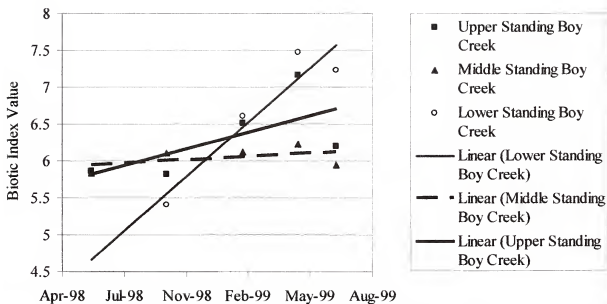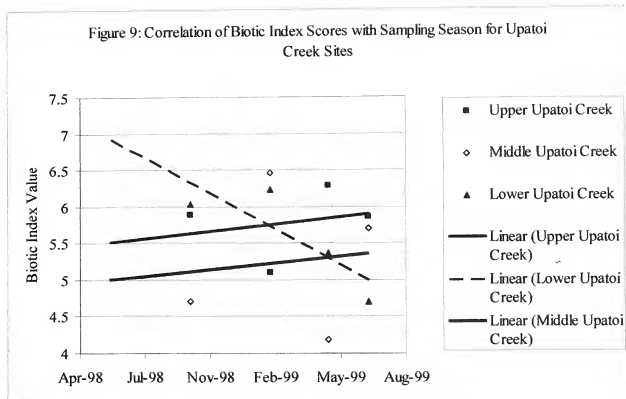
An independent study of chemical parameters in these creeks provided the data recorded in Table 7. This study was performed by Columbus Water Works, but only one site on each creek was sampled, and the time scale during which these measurements were taken did not accurately correspond with the sampling times for this study. Therefore, associating differences in the chemical parameters found at the three creeks with the biological data collected for this study would not provide accurate correlations. However, as a point of interest, many of the chemical and microbial constituents recorded had the highest mean values at the Bull Creek site.

Table 7: Mean concentrations of chemical and microbial constituents at Bull Creek, Standing Boy Creek, and Upatoi Creek

Constituent	Bull Creek	Standing Boy Creek	Upatoi Creek
Chromium (mg/L)	<0.010	<0.010	<0.010
Copper (mg/L)	<0.010	<0.010	<0.010
Iron (mg/L)	2.63	2.53	2.42
Lead (mg/L)	0.016	<0.010	<0.010
Nickel (mg/L)	0.01	<0.010	<0.010
Zinc (mg/L)	0.04	<0.02	0.028
Total Phosphorous ((mg/L)	<0.3	<0.3	<0.3
Ammonia (mg/L)	0.237	0.091	0.075
Total Organic Carbon (mg/L)	7.53	5.19	4.01
Chemical Oxygen Demand (mg/L)	47.85	26.37	42.69
Biological Oxygen Demand (mg/L)	7.09	4.23	3.15
Fecal Coliform Levels (col/100 mL)	93000	2613	4878
E. coli Levels (col/100 mL)	5440	502	1010
Total Suspended Solids (mg/L)	137.45	40.13	62.56

Discussion:

A lack of a significant correlation between percent urbanization and the various water quality metrics used in this study to leaves doubt as to the specific cause of the differences noted among the nine creek sites. Other land use patterns such as percent agriculture and percent forest, as well as the size of the catchment area, also produced no significant relationship with any of the metrics used. Urbanization has been definitively linked to variations among macroinvertebrate communities by an abundance of other research (Benke *et al.* 1981, Duda *et al.* 1982, Jones and Clark 1987, Lenat and Crawford 1994, Baker and Sharp 1998, Walsh *et al.* 2001, Morley and Karr 2002). Therefore, the absence of correlation in the current study should not be interpreted as a lack of any effect of urbanization on ecological integrity at these sites. Instead, the combination of many factors, including urbanization, acting synergistically or antagonistically could be attributed as the cause of the differences detected between stream sites in this study. As Norris and Hawkins (2000) noted, many confounding factors exist when attempting to determine the effects of a gradient of human activities on water quality. Separating effects of these factors from one another is difficult and was not within the scope of this study.

The drought that persisted throughout the year of sampling was one such confounding factor. It seemed to most consistently influence the biotic index scores, which generally increased steadily from the June 1998 sampling period up to the July 1999 sampling period. The earliest sample was the only one that produced a "good" water quality rating according to Hilsenhoff's system (1987): all other sampling efforts resulted in water quality ratings of "fair", indicating decreased water quality. The

reduction in flow that could have resulted from the drought may have concentrated pollutants already existing within the water, meaning that as it progressed, more and more of the sensitive organisms were not able to subsist due to increasing pollution levels. Also, during drought periods, pollutants from vehicle exhaust, street litter, fertilizers, and animal wastes can build up on urban surfaces, only to be suddenly washed into streams during storm event (Baer and Pringle 2000). These circumstances would raise pollution levels rapidly and consequently affect benthic macroinvertebrate community structure. Other biomonitoring studies done concurrently with this one as part of the larger Columbus Waterworks study did indicate that drought was a partial cause of changes noted in the biota and water quality (Gore 2001).

Even with the lack of significant correlation between urbanization and the various metrics used to evaluate the macroinvertebrate communities, some generalizations can be made about the ecological integrity of the study sites. All metrics used indicated that the three Upatoi Creek sites provided a comparatively healthier environment for a diverse array of macroinvertebrates than most other sites. This result was unsurprising as the Upatoi Creek sites were also located in areas of low urbanization, especially the upper and middle sites. The middle and lower Upatoi Creek sites were the only two sites rated as having "good" water quality according to their biotic index scores, which based on Hilsenhoff (1987), indicates that with regards to organic pollution levels and physical disturbances within their waters, these two sites were superior to all other sites. While the upper Upatoi site did not attain this ranking, its biotic index score was just above the threshold. Road construction occurring at this site may have affected the composition of macroinvertebrate communities there. Construction activities have been documented to

significantly increase sedimentation within streams (Reed 1977), which in turn could have caused changes in the type of macroinvertebrates inhabiting the site and resulted in the poorer biotic index score. Despite the differing biotic index classifications, all three sites had highly diverse macroinvertebrate populations living within them, more so than the Bull Creek sites in all cases. These macroinvertebrate populations included many intolerant taxa, most notably a relatively diverse array of plecopteran species. As Plecoptera are known to require clean substrate and high water quality, their continued presence within a stream is often correlated with a lack of or low pollution (Baer and Pringle 2000). They have also been recognized as the first EPT taxa to disappear when pollution issues begin to occur (Mangun *et al.* 1989, Fore *et al.* 1996), so their strong presence here further attests to the healthy ecological integrity at these sites.

Physical differences exist between the lower two Upatoi sites and the other sites, which affected the ability to make truly valid comparisons between all nine sites. The most noticeable differences were the increase in discharge and the difference in substrate at these two sites in contrast to the other seven sites. Such physical factors have been cited as affecting the macroinvertebrate community structure within streams (Resh and Grodhouse 1983), which in turn would have affected the metric values used in this study. Additionally, higher discharges would also result in more effective dilution of any pollutants within the water column, which could be responsible for the higher percentages of pollution intolerant species collected from these two sites. Being located below the fall line, the substrate at these sites was proportionately much sandier than at the other sites, with little cobble or rocks. Low abundances of macroinvertebrates collected from these sites could be accounted for by the change in substrate. Sandy

substrates provide less of a variety of hospitable environments for macroinvertebrates to colonize (Lamberti and Berg 1995). Despite comparatively low richness and abundance values at the lower and middle Upatoi Creek sites, both locations had a high diversity of macroinvertebrates collected during sampling periods.

Higher water quality within sampled reaches of Upatoi Creek was expected, based upon their locations. Much of Upatoi Creek, and the two downstream sites, are located on Fort Benning, a large army installation with a high proportion of training areas that remain forested and unaffected by urbanization. Even the upper site is located in an area of low urban land use and is predominantly forested. As such, much of the land surrounding Upatoi Creek should be exposed to few of the typical impacts of urbanization. While the army training areas are criss-crossed by roads, most roads remain unpaved, resulting in higher infiltration rates. The areas surrounding roads are often forested, decreasing runoff further. The riparian zones are well established in almost all areas visited, and thus should filter out much of the pollutants in runoff as well. Traffic is limited primarily to military vehicles in the training areas, likely minimizing the amount of vehicular pollutants available to wash into the stream. Agricultural land use is also low in comparison to the upper and middle Standing Boy Creek sites. The location of the lower Upatoi Creek site, near one of the entrance gates to Ft. Benning, does put it in more trafficked and vulnerable area. That influence may be offset by a lack of pollution flowing from upstream and the diluting effect of the high volume of water at that site. Therefore, the original assumption that the Upatoi Creek sites would be the least affected has been substantiated by the results of this study. Unfortunately, based on the physical differences, these sites cannot truly be used as a reference to compare with the other sites

due to the inability to isolate the effects of land use from the effects of those physical differences.

Evaluating the water quality of the remaining six sites was less clear. The Standing Boy Creek sites are located along the northern edge of Columbus in an area that is developing. Thus, they were expected to show some of the effects of urbanization, but not as much so as the Bull Creek sites, which are located more within the city. Upon examining the GIS data however, the amount of urbanization at the Standing Boy Creek sites was not much higher than that surrounding the Upatoi Creek sites. In fact, the lower Upatoi Creek site had twice the amount of urbanization within the surrounding area, although the percentage was still quite low. Interestingly enough, the only site on Standing Boy Creek that was judged to have fairly good water quality was the middle site, which had the highest percent urbanization out of the three (although still less than 1%). These results lead to the speculation that urbanization is not the main factor influencing the macroinvertebrate communities at these sites.

The lower and upper Standing Boy Creek sites were the only ones categorized by their biotic index scores as having fairly poor water quality. According to Hilsenhoff (1987), this index is most sensitive to organic and nutrient pollution, suggesting that that type of pollutant may be linked with the decreased ecological integrity found at these two sites. This result was unanticipated since all three sites had low amounts of urbanization within them. However, since these sites were both in areas undergoing relatively new suburban development, perhaps the GIS percentages calculated did not accurately reflect the changes and increasing environmental stress occurring at these sites. Another possibility, particularly at the upper site, is that agricultural land use may have had more

of an effect on the macroinvertebrate communities than urbanization. The percent agriculture at the upper Standing Boy Creek site was twice that of most of the other sites at almost 17%, while percent urbanization remained low at less than 0.7%. This site was highly dominated by chironomids, which have been noted to be numerous at agriculturally impacted sites (Riva-Murray *et al.* 2002). In contrast, the percentage of EPT individuals was lower than that found at all other sites except the lower Standing Boy Creek site. Plecoptera were present but very scarce comprising less than one percent of the total population. This site is located adjacent to US highway 27, which carries many Harris County residents into the city of Columbus on daily commutes. Large amounts of vehicular pollutants resulting from this traffic could have been a contributing factor to the poor ecological integrity found here. This study was not designed to enable separation of the possible effects of agricultural land use from the effects of urbanization.

The lower Standing Boy Creek site seems to be suffering even more from impairment than the upper site. In fact, metrics suggest that this site has the worst ecological integrity out of all nine locations. Macroinvertebrate communities at this site were almost completely dominated by chironomids, with very few EPT taxa and no Plecoptera. The biotic index value was the highest of all sites as well, indicating that the few species collected within this site are mainly those that are tolerant of pollution. The only obvious cause of such decreased water quality at this site is the back up of water from Lake Oliver and Lake Biggers, which resulted in muddy, standing water that cannot support healthy macroinvertebrate communities. The high sediment load and lack of current would result in limited light penetration and low oxygen levels, as well as affecting the macroinvertebrate food supply either directly or via affecting the

mechanisms by which macroinvertebrates feed (Mangun *et al.* 1989). Therefore, it is not surprising that chironomids dominated such waters, as many species are known to be adapted to tolerate low levels of oxygen and high levels of organic pollution (Garie *et al.* 1986, Fore *et al.* 1996). Based on the low percentages of land use, neither urbanization nor agriculture should be high enough to account for the significant amount of stress at this site. Due to the standing water, complete sampling here was not possible. Initially, the incomplete sampling was blamed for the low number of macroinvertebrate species collected at this area. However, after rarefaction methods were used to standardize for the lower sample size, the number of species found at this site was still significantly decreased from expectations for the lower Bull Creek site. Also, metrics such as percentages of EPT taxa, Plecoptera, and Chironomidae should not have been affected by incomplete sampling since they represent relative abundances, yet they still indicate degraded water quality. Therefore, even had the site been sampled as thoroughly as all others, severe impairment would almost certainly still been observed.

When examining the data from Bull Creek sites, the upper site is apparently in fair health, as would be expected by the low amount of urbanization at that site in comparison to the other two sites. It seems to support a diverse macroinvertebrate community consisting of almost equal percentages of EPT and chironomid individuals. However, again at this site, the proportion of agricultural land use is high. The few effects on macroinvertebrate community composition (such as lack of plecopterans) that were noted therefore likely resulted more from impacts of agricultural land uses than impacts from urbanization.

The other two Bull Creek sites had comparatively higher amounts of urbanization in areas surrounding them than at all other study sites. In fact, an argument could be made that based upon the percent urban land use, these two sites should be the only ones in which urban runoff and activities played a significant role in affecting the macroinvertebrate communities. Wang *et al*. (1997) found that watersheds with 20% or more urban land regularly had poor to very poor invertebrate biotic index scores (IBI), indicating significant effects of urbanization on water quality and the macroinvertebrate communities at those sites. Furthermore, results of that study noted some percentage of urban land use between ten and twenty percent as being the point at which biological integrity began to decline dramatically. Both the middle and lower Bull Creek sites consisted of over ten percent urban land use, and, indeed, water quality does seem to be suffering somewhat at these two sites. However, based on the biotic index classifications, the ecological integrity still seems to be better at these Bull Creek sites than at the upper and lower Standing Boy sites. In fact, compared to the presumed high amounts of urban runoff that potentially empties into these two sites, the macroinvertebrate communities seems to be less effected than was initially expected. This is not to say that some important signs of impairment are not evident. Diversity within macroinvertebrate communities at these sites is significantly decreased from what was recorded for Upatoi Creek sites, and the percent of EPT individuals is also lower at most of those sites, with very few to no Plecoptera collected. In addition, macroinvertebrate communities at both sites are highly dominated by Chironomidae. Yet while the water quality is relatively diminished, the difference is not that remarkable considering the much larger differences in percent urbanization. Based on visual assessments alone, the lower Bull Creek site

initially seemed particularly susceptible to urban impacts, as foul odors existed on every sampling occasion, and businesses and houses were situated in close proximity to the stream. Yet, these locations are ranked via their biotic index scores as having fair water quality. Therefore, while these sites do show some signs of being negatively impacted by the amounts of urban land use occurring around them, the water quality is still within the same category as the upper Upatoi Creek site, the middle Standing Boy Creek site, and the upper Bull Creek site based on that metric. One factor that might be responsible for lessening some of the impacts of urban runoff is the developed riparian area surrounding both of these sites. Despite being within the city limits, the riparian areas immediately surrounding both sites consisted of dense vegetation, with little clearing evident within the visual field. This dense vegetation would have filtered out at least some pollutants washed towards the streams during storm events. The riparian zones have been indicated to be a significant buffer from the impacts of urbanization for streams (Paul and Meyer 2001).

When considering the three sites on Bull Creek, one trend in biotic index values was noted that was not apparent at the other two streams. As the water in Bull Creek flowed downstream and through the city of Columbus, the biotic index scores climbed slightly, indicating that the water quality was decreasing. While the differences were not significant enough to place the three sites in different water quality categories, this pattern could be associated with the increasing amounts of urbanization along this stream. A graphical representation of this correlation can be seen in Figure 10. This trend was not demonstrated at the sites on the other two creeks, possibly because the differences in percent urbanized area were so slight. The increase in biotic index scores seen at the Bull

Creek sites could be attributed to the accumulation of urban effects on the stream as the water flows towards the Chattahoochee, or the increase could be the result of a more localized increase in polluted runoff at each area. Two other metrics echoed the same pattern at the Bull Creek sites. Percent Chironomidae rose with increasing urbanization, while percent EPT individuals dropped with increasing urbanization at these sites. These correlations can be seen in Figure 11.

Figure 10: Correlation between Percent Urbanization and Biotic Index Scores for Bull Creek Sites
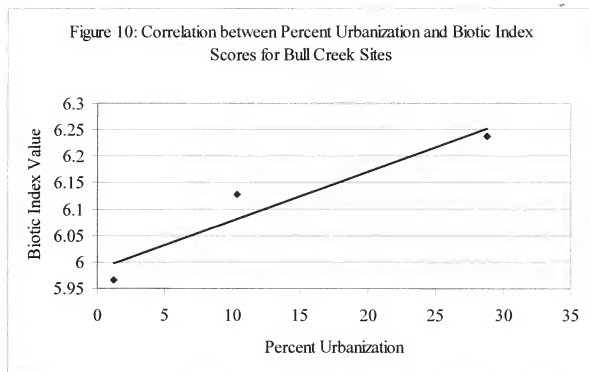
Figure 11: Correlation of Percent Urbanization with Percent EPT and Percent Chironomidae at Bull Creek Sites
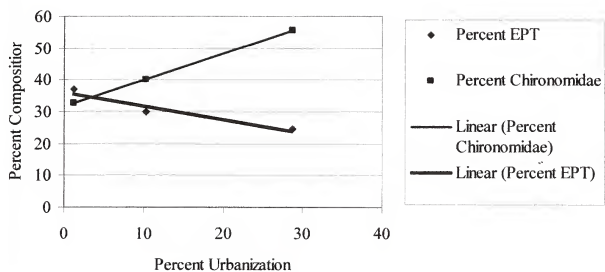
Conclusions

Several changes in the methodology of this study could have provided more insight in identifying possible causes of the differences noted in ecological integrity between sites. First, the study could have been extended over a longer period of time, and sampling efforts could have been intensified. Sampling over a longer period of time would have increased precision and possibly differentiated the effects of urbanization and drought.

This study would have also benefited from choosing sites that were more comparable to each other physically. Thus minimizing other sources of variation among sites might have resulted in a better delineation of these factors. The two sites indicated to have comparatively superior ecological integrity by this study were indeed in areas of relatively low urbanization, but they were also the two sites that differed most from all the others in substrate and discharge. Choosing sites with very limited agricultural land use would have been another method by which to better isolate the effects of urbanization on ecological integrity. Attempts to separate the effects of agriculture from the effects of urbanization resulted in uncertainty as to which, if either, was the primary cause of any declines seen in ecological integrity at certain stream sites. This was particularly obvious in the upper Standing Boy Creek site, as it suffers from significant impairment in comparison to almost all other sites, and has a low level of urbanization and a higher level of agriculture in the area surrounding it.

While all other variation should have been minimized among sites, the study would have benefited from choosing study sites that maximized the differences in percent urbanization. Out of the nine sites chosen here, the GIS data (provided after the study

was well underway as it was not included in the original protocol for this project) showed that seven out of the nine sites had less than two percent urbanization in the surrounding areas. The effects of urbanization have typically been most noticeable at a threshold of at least ten percent urban land use (Wang *et al.* 1998), a criterion that only two of the sites in this study met. This threshold was supported by the results seen at Bull Creek sites. Only at these sites could a pattern be seen in biotic index values, percent Plecoptera, and percent Chironomidae as percent urban land use increased, and these three sites were the only ones that had ten percent or more differences in the amount of urbanization between sites. Similar differences would have likely surfaced on other streams had the sites been located in areas of bigger differences in percent urbanization. All the extraneous variables, coupled with the lack of a notable gradient in percent urbanization between most sites, resulted in the inability to significantly link any single land use characteristic as the cause of poor ecological integrity at certain sites.

In hindsight, further changes in sampling design would have been beneficial. Most notably, the sample sizes could have been standardized before going on to identify the macroinvertebrates, eliminating that source of variation. Barbour et al. (1999) described a method for selecting 200 macroinvertebrates out of larger samples using grids, and thus keeping sample size equal for all collections. In order to address the problem of unequal sample size in this study, rarefaction methods were used to better compare richness values, and percent composition within the macroinvertebrate communities was stressed as those metrics should have been relatively unaffected. However, diversity and biotic index measures were incorporated into the study, yet both

are to some degree a product of the richness at each site and thus are affected by sample size (Krebs 1989). Removing the possible bias of sample size would have been helpful.

Finally, if chemical data been taken in concordance with the times and locations when the biological sampling occurred, it might have been better correlated with changes noted between sites as well and possibly provided stronger insight into the cause of differences among various sites. Most contaminants tested were assessed as higher in Bull Creek than the other two creeks. However, without better concordance between the chemical and biological data collection, no significant results were obtained that related the two. This lack of correspondence highlights one of the criticisms of biomonitoring studies: that often the data collected do not indicate what precisely is to blame for the changes detected in the water quality, only that the change exists (Riva-Murray et al. 2002).

The ongoing research via Columbus State University involving the use of biomonitoring to determine water quality in Georgia streams and rivers should provide insight regarding future studies in this region. Once minimally impacted streams are identified with a variety of diverse physical characteristics, the macroinvertebrate communities within those streams can be sampled to serve as a reference for comparison with the macroinvertebrate communities in more impacted streams.

In summary, there are indications that the Upatoi Creek sites, especially the lower and middle sites, are the least impacted out of the nine sites. These sites are located in areas of low urbanization, supporting the idea that low levels of urbanization are linked with good ecological integrity. Patterns exist in the Bull Creek sites that also suggest that

increasing urbanization is associated with decreasing water quality. However, no significant correlation was found between percent urbanization and any changes in metric values. Indeed, the macroinvertebrate communities of two of the Standing Boy Creek sites are most impacted by pollution, yet these two sites are located in areas of low urbanization. These results should not be interpreted to mean that urban land use is not having any effect on the macroinvertebrate communities within these creeks. Based on a plethora of other studies (Benke *et al.* 1981, Duda *et al.* 1982, Jones and Clark 1987, Lenat and Crawford 1994, Baker and Sharp 1998, Walsh *et al.* 2001, Morley and Karr 2002), urbanization almost certainly is or will have a detrimental effect on water quality and the macroinvertebrate communities within those waters. A logical conclusion from the outcome of this study would be that many factors are affecting the water quality synergistically or antagonistically at examined sites, making separation of those effects impossible based on the sampling protocol. Urban effects are numerous, varied, and diffuse (Chessman and Williams 1999), and in this case, the effects of agriculture and drought at the very least must be added to these effects.

With no significant correlation detected between percent urbanization and the various metrics calculated to determine impairment, this study leaves doubt as to the direction to take to improve the water quality at sites such as the upper and lower Standing Boy Creek sites. Remediation at these sites, as well as at the less impaired lower and middle Bull Creek sites, would benefit from further research to determine the exact causes of stress at these locations. As more regional data are collected, interpreting any further biomonitoring data should become more accurate and more capable of pointing out possible causes of poor ecological integrity, therefore making conservation and

restoration efforts more effective. As much of the pollution problems currently affecting streams in the United States are not from the more readily identifiable point sources, future biomonitoring efforts must have the capacity to precisely target the less easily identified non-point sources of pollution in order to direct any restoration and conservation efforts (Riva-Murray *et al.* 2002). The more accurately the source of pollution is identified, the quicker policies can be set in place to prevent further detrimental effects to water quality and to correct effects that are already present.

Based on the results of this study, restoration efforts are called for at the lower and upper Standing Boy Creek sites. The lower and middle Bull Creek sites should also be considered as being in need of lesser amounts of remediation, as the macroinvertebrate community composition suggests decreasing ecological integrity at these sites as well. Although the middle Standing Boy site, upper Bull creek site, and all Upatoi Creek sites appear to be the least affected overall by surrounding land use, these sites should not be ignored, as wise land use practices are required to maintain such conditions.

With each site, restoration or conservation of the riparian area may be one of the most effective measures to be taken (Morley and Karr 2002). The density of the riparian area encompassing the lower Bull Creek site in particular may be the key that explains why this site isn't as impacted by the high amount of surrounding urbanization as initially expected. Morely and Karr (2002) reported that when overall basin development was low to moderate, maintaining the natural riparian corridors had the potential to maintain current biological conditions or possibly even improve biological integrity in many cases.

Morley and Karr (2002) further recommend combining biological assessment with chemical and physical assessment of stream systems to effectively confirm and

correct the source of degradation to streams and rivers. Baer and Pringle (2000) add that planning, education, and community involvement should be included as well. They point out that increasing population size in urban areas is linked with stream degradation in many cities, but that the same resource, the community, can be harnessed to correct such degradation. Urban stream conservation and restoration relies on the cities' residents to see themselves as a necessary component to maintaining functioning catchments. Riversmart, a national public education campaign directed by River Network, is aimed towards such a goal (River Network 2004). Urban sprawl will inevitably continue, but increasing public awareness, coupled with increases in knowledge about how the multiple effects of land use impact the benthic biota and water quality, could combat further degradation of streams and rivers and provide for more effective restoration and conservation efforts.

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Appendix 1: Macroinvertebrate species list and abundance data by site and date sampled.

Location: Bull Creek Upper

Order	Family	Genus and Species	Jun-98	Oct-98	Feb-99	May-99	Jul-99	Total
Bivalvia	Corbiculidae	*Corbicula fluminea*	8	63	153	24	41	289
Coleoptera	Elmidae	*Ancyronxy variegatus*	3					3
		Dubiraphia sp.		2				2
	Hydrophilidae	*Berosus sp.*		1	4			5
	Psephenidae	*Ectopria thoracica*		1				1
Decapoda	Cambaridae	*Cambarus latimanus*				2		2
Diptera	Ceratopo-gonidae	*Bezzia/Palpomyia sp.*			2			2
	Chironomidae	*Ablabesmyia mallochi*				18		18
		Alotanypus aris	2					2
		Brillia sp.	2					2
		Corynoneura B				1		1
		Crictopus/Orth.sp.	3	1	20	5		29
		Cryptochironomus sp.			1	2	1	4
		Dicrotendipes neomodestus	2	1	6			9
		Diplocladius sp.	1					1
		Eukiefferiella claripennis gr.			5			5
		Hayesomyia sp.	6				1	7
		Hayesomyia senata	7			3		10
		Helopelopia sp.		1				1
		Hudsonimyia karelena gr.	1	2				3
		Micropsectra sp.	1					1
		Phanopsectra obediens gr.				62	1	63
		Phanopsectra	2	4				6

		Phanopsectra puntipes gr.	2	4				6
		Polypedilum convictum	55	11	1	13	1	81
		Polypedilum tritum	1	4	1		2	8
		Procladius sp.	2					2
		Rheocrictopus robacki	5	1	1	7		14
		Rheotanytarsus sp.	7	2	7	81	2	99
		Tanytarsus A	1			1	-	2
		Tanytarsus B			1			1
		Tanytarsus C	9	1				10
		Tanytarsus D		6	2	13		21
		Tanytarsus L	3	1		4		8
		Tanytarsus P		2				2
		Tanytarsus U			5	68		73
		Thienemanniella A	1					1
		Thienemanniella xena			2	1		3
		Thienemannimyia gr. sp.				5		5
	Empididae	*Hemerodromia sp.*			2	1		3
	Simulidae	*Simulium sp.*	2		1		1	4
		Simulium decorum	1		3			4
		Simulium haysi					2	2
		Simulium parnassum			3	9		12
		Simulium podestemi			1			1
	Tipulidae	*Hexatoma sp.*	2	1				3
Ephemer-optera	Baetidae	*Acentrella sp.*	1					1
		Baetis amplus			7	10	2	19
		Baetis brunneicolor	6					6
		Labiobates sp.				3		3
		Paracleodes sp.	1					1
	Caenidae	*Caenis sp.*	3	3				6
	Heptageniidae	*Stenonema*	74	3	6	2		85

		modestum						
		Stenonema smithae	4					4
	Isonychiidae	*Isonychia sp.*	142	1	1	5		149
	Tricorythidae	*Tricorythodes sp.*	45	1		9		55
Gastropoda	Physidae	*Physella gyrina*		2				2
	Planorbidae	*Gyraulus sp.*	1					1
		Planorbella armigera arm.		15	1	1		17
Hemiptera	Gerridae	*Rheumatobates tenuipes*		1				1
		Trepobates inermis	1					1
	Pleidae	*Paraplea sp.*		1				1
	Veliidae	*Rhagovelia obesa*					6	6
Isopoda	Asellidae	*Lirceus lineatus*		1	10	11		22
Lumbri-culida	Lumbriculidae	*Lumbriculus variegatus*	10	2	12		4	28
Megaloptera	Corydalidae	*Corydalus cornutus*	4	1				5
Odonata	Aeshnidae	*Boyeria vinosa*				6		6
	Coena-grionidae	*Argia bipuntulata*		1	3			4
	Gomphidae	*Lanthus vernalis*	1					1
Plecoptera	Perlidae	*Perlesta placida*				1		1
Trichoptera	Hydro-psychidae	*Cheumatopsyche sp.*	82	13	19	53		167
		Diplectrona modesta	1					1
		Hydropsyche frisoni	20		2	14		36
		Hydropsyche scalaris	4	2		7	1	14
	Hydroptilidae	*Hydroptila sp.*	2			4	2	8
	Philo-potamidae	*Chimarra sp.*	1					1
Tubificida	Naididae	*(unable to further identify)*					2	2
		Amphicheata leydigi			1			1
		Ophidonais serpentina	1					1
	Tubificidae	*Branchiura*	1		20		2	23

		sowerbyi						
Total			532	152	303	446	71	1504

Location: Bull Creek Middle

Order	Family	Genus and Species	Jun-98	Oct-98	Feb-99	May-99	Jul-99	Totals
Bivalvia	Corbiculidae	*Corbicula fluminea*	56	71	45	85	39	296
Coleoptera	Elmidae	*Anacyronxy variegatus*		4				4
		Dubiraphia sp.			1			1
		Stenelmis sp.		1				1
		Stenelmis bicarinata					3	3
		Stenelmis hungerfordi	1					1
		Stenelmis miribilis	1					1
	Hydro-philidae	*Berosus sp.*		5	2			7
Decapoda	Cambaridae	*Cambarus latminus*			1			1
Diptera	Chiron-omidae	*Ablabesmyia mallochi*			1	3		4
		Corynoneura B			1			1
		Crictopus/Orthocladius sp.		1	11	69	25	106
		Crictopus bicinctus			31	1	17	49
		Crictopus politus			2		2	4
		Crictopus sylvestris	3					3
		Cryptochironomus sp.				1	1	2
		Dicrotendipes neomodestus			4		1	5
		Hayesomyia sp.			1			1
		Hayesomyia senata			1	6		7
		Hudsonimyia karelena			1	1	3	5
		Larsia sp.				1		1
		Micropsectra sp.	1					1
		Orthocladius annectens	1			5		6
		Phanopsectra obediens gr.				4		4
		Phanopsectra punctipes gr.			1			1
		Polypedilum		31	5	2		38

		convictum gr.					
		Polypedilum tritum			4	4	5	13
		Procladius bellus var. 3			3			3
		Rheocrictopus sp.			1			1
		Rheocrictopus robacki			11	12	6	29
		Rheotanytarsus sp.			10	146	25	181
		Saetheria tylus					1	1
		Tanytarsus sp.			1		1	2
		Tanytarsus A				1		1
		Tanytarsus C				1		1
		Tanytarsus U			3	15		18
		Thienemanniella B				2		2
		Thienemanniella xena			4	1	2	7
		Thienemannimyia gr.	3			5	3	11
	Empididae	*Hemerdromia sp.*	2			2		4
	Simulidae	*Simulium sp.*	3					3
		Simulium parnassum			1	11		12
		Simulium slossonae					1	1
	Tipulidae	*Cryptolabis sp.*			1			1
		Dicranota sp.	1					1
		Ormosia sp.			1			1
		Tipula sp.		1		1		2
Ephemer-optera	Baetidae	*Acentrella sp.*	2			8		10
		Baetis amplus				2		2
		Baetis brunneicolor				2		2
		Paracleodes sp.			1			1
		Psuedocleon sp.	2					2
	Caenidae	*Caenis sp.*	2	1				3
	Hepta-geniidae	*Stenonema sp.*	2					2
		Stenonema modestum	2		3	1		6
	Isonychidae	*Isonychia sp.*	1			4	1	6
		Isonychia arida	2					2
		Isonychia sicca	17					17
	Tricory-	*Tricorythodes sp.*	44	6		4	8	62

	ithidae							
Gastropoda	Physidae	*Physella sp.*	1					1
		Physella gyrina				2		2
	Planorbidae	*Planorbella armigera arm.*		4			1	5
Hemiptera	Belosto-matidae	*Belastoma testaceum*		2				2
	Gerridae	*Neogerris hesione*					1	1
Isopoda	Asellidae	*Lirceus fontinalis*	2					2
Lumbri-culida	Lumbri-culidae	*Lumbriculus variegatus*					6	6
Megaloptera	Corydalidae	*Corydalus cornutus*		1			2	3
Odonata	Aeshnidae	*Boyeria vinosa*	1					1
	Coen-agrionidae	*Argia sp.*		1				1
		Argia bipunctulata			1			1
		Enallagma divagans			1			1
	Gomphidae	*Gomphus aponmyius*	1					1
		Ophiogomphus carolinus		1				1
Trichoptera	Hydro-psychidae	*Ceratopsyche alhedra*	22					22
		Cheumatopsyche sp.	9		6	80	74	169
		Hydropsyche frisoni				14	9	23
		Hydropsyche scalaris	22			13	6	41
		Hydropsyche venularis	1			9		1
	Hydroptilidae	*Hydroptila sp.*					1	10
Tubificida	Naididae	*Dero sp.*					1	1
		Dero digitata			1			1
	Tubificidae	*Aulodrilus limniobius*			1			1
		Branchiura sowerbyi			1		5	6
Totals			205	130	163	518	250	1266

Location: Bull Creek Lower

Order	Family	Genus and Species	Jun-98	Oct-98	Feb-99	May-99	Jul-99	Total
Bivalvia	Corbiculidae	*Corbicula fluminea*	10	7	56		32	188
Coleoptera	Elmidae	*Ancyronyx variegatus*				1		1
		Dubiraphia sp.		1				1
		Stenelmis sp.				5		5
		Stenelmis hungerfordi				3		3
	Hydrophilidae	*Berosus sp.*		1		2	2	5
		Berosus striatus		4				4
	Gyrinidae	*Gyrinus sp.*				3		3
Diptera	Ceratopo-gonidae	*Probezzia sp.*				1		1
	Chironomidae	*Ablabesmyia mallochi*				22		22
		Chironomini III sp.					1	1
		Chironomus sp.				1		1
		Cladotanytarsus sp.				3		3
		Crictopus/Orthocladius sp.	1		7	100	9	117
		Crictopus/Orthocladius bicinctus			49	1		50
		Crictopus/Orthocladius politus			7			7
		Crictopus/Orthocladius sylvestris			1			1
		Cryptochironomus sp.			1	4	1	6
		Dicrotendipes neomodestus		1		6	3	10
		Hayesomyia sp.	1			1		2
		Hayseomyia senata		2	8	4		14
		Hudsonimyia karelena			2	8		10
		Labrudinia sp.				1		1
		Meropelopia sp.		2				2
		Microchironomus sp.				1		1
		Micropsectra sp.		1				1

		Nanocladius rectinervus		2				2
		Natarsia sp.		1				1
		Nilotanypus americanus				1		1
		Orthocladius annectens		1	16			17
		Parametriocnemus lundbecki				2		2
		Phanopsectra obediens gr.				20		20
		Phanopsectra punctipes gr.				5		5
		Polypedilum convictum gr.		42	6	20		68
		Polypedilum tritum			26	272	28	326
		Procladius sp.				1		1
		Procladius bellus var. 1		1				1
		Rheocrictopus robacki			17	58		75
		Rheopelopia sp.		1				1
		Rheotanytarsus sp.		1	2	38	1	42
		Robackia demejerei				8		8
		Tanytarsus C		1	3			4
		Tanytarus D			2	9		11
		Tanytarsus P		1				1
		Tanytarsus U				50		50
		Thienemannimyia gr.sp.	3			30	1	34
		Thienemanniella xena				5		5
	Empididae	*Hemerodromia sp.*	2			3	2	7
	Simulidae	*Simulium decorum*	8					8
		Simulium parnassum				30		30
		Simulium podestemi	1					1
		Simulium slossonae				29		29
	Tipulidae	*Tipula sp.*	2			10		12
Ephemer-optera	Baetidae	*Acentrella sp.*	1					1
		Baetis sp.				1		1
		Baetis amplus	1			30		31
		Paracleodes sp.	1	5			9	15

	Caenidae	*Caenis sp.*				2		2
	Heptageniidae	*Stenonema modestum*				4		4
	Isonychiidae	*Isonychia sp.*				8		8
	Tricorythidae	*Tricorythodes sp.*	26	3		15		44
Gastropoda	Physidae	*Physella sp.*				1		1
		Physella gyrina			1			1
Hirudinae	Glossiphoniidae	*Helobdella triserialis*		1				1
Lumbricula	Lumbriculidae	*Lumbriculus variegatus*	2	2	13			17
Odonata	Aeshnidae	*Boyeria vinosa*				1		1
	Coenagrionidae	*Argia bipunctata*		2				2
		Nahellenia gracilus		1				1
	Corduliidae	*Macromia taeniolata*			1			1
Plecoptera	Perlidae	*Perlesta placida*				2		2
	Pteronarcyidae	*Pteronarcys sp.*				2		2
Trichoptera	Brachycentridae	*Brachycentrus sp.*				7		7
	Hydropsychidae	*Ceratopsyche sp.*	1	1				2
		Cheumatopsyche sp.	22	10	9	164	27	232
		Hydropsyche frisoni	8	2		19	2	31
		Hydropsyche scalaris				18	1	19
	Hydroptilidae	*Hydroptila sp.*			1	2	1	4
	Polycentropodidae	*Polycentropus sp.*				2		2
Tubificidae	Naididae	*Amphicheata americana*					2	2
		Dero digitata					1	1
Totals			90	97	228	1119	123	1657

Location: Standing Boy Creek Upper

Order	Family	Genus and Species	Jun-98	Oct-98	Feb-99	May-99	Jul-99	Totals
Amphipoda	Hyallelidae	*Hyallela azteca*			4			4
Bivalvia	Corbiculidae	*Corbicula fluminea*		7	17	18	12	54
Coleoptera	Dytiscidae	*Hydroporus sp.*			1			1
	Elmidae	*Ancyronyx variegatus*			1	1		2
	Helophoridae	*Helophorus linearis*			1			1
Decapoda	Cambaridae	*Cambarus latimanus*	1		1			2
		Procambarus spiculifer	2			1		3
Diptera	Chironomidae	*Ablabesmyia mallochi*		4	2			6
		Chironomus sp.				24		24
		Corynoneura B		1	5	3	1	10
		Crictopus (Iso.) sp.			1			1
		Crictopus/Orthocladius sp.		2				2
		Crictopus bicinctus			2			2
		Crictopus politus			3		3	6
		Cryptochironomus sp.				10		10
		Dicrotendipes neomodestus		1				1
		Hayesomyia sp.		1				1
		Hayesomyia senata		2	1			3
		Hudsonimyia karelena		3				3
		Hydrobaenus sp.			1			1
		Nanocladius distinctus		2				2
		Paracladopelma sp.				3		3
		Paracladopelma undine			1			1
		Phanopsectra obediens gr.				158		158

Order	Family	Species						Total
		Phanopsectra punctipes gr.			1			1
		Polypedilum convictum gr.	1	30	8	9	2	50
		Polypedilum tritum		7	1	3		11
		Rheocrictopus robacki		1	10	13		24
		Rheopelopia sp.			1			1
		Rheotanytarsus sp.		11	8	1	4	24
		Robackia demeijerei				1		1
		Stictochironomus devinctus			1			1
		Stilocladius sp.			3			3
		Tanytarsus sp.		2				2
		Tanytarus A			3			3
		Tanytarsus C		14	5			19
		Tanytarsus D		12	5		2	19
		Tanytarsus L		3				3
		Tanytarsus U					1	1
		Thienemannimyia gr.			2	5	1	8
	Culicidae	*Wyeomyia sp.*		1				1
	Empididae	*Hemerodromia sp.*	1					1
	Simulidae	*Simulium haysi*					1	1
		Simulium parnassum			2			2
		Simulium slossonae				1		1
		Simulium venustum			2			2
	Tipulidae	*Dicranota sp.*		8				8
		Limonia sp.		2				2
		Tipula sp.		1	1	2		4
Ephem-eroptera	Baetidae	*(could not be identified further)*			1			1
	Heptageniidae	*Stenonema annexum*	1					1
		Stenonema modestum	1	12	6		6	25
		Stenonema terminatum		8				8
	Isonychiidae	*Isonychia sp.*	2	1		1		4

	Tricorythidae	*Tricorythodes sp.*	1					1
Gastropoda	Physidae	*Physa sp.*		2				2
		Physella gyrina			1			1
Hemiptera	(nymph)	*(could not be identified further)*		1				1
	Corixidae	*(nymph)*	1					1
	Gerridae	*Neogerris hesione*					1	1
	Hebridae	*Lipogomphus sp.*		1				1
Isopoda	Asellidae	*Lirceus lineatus*		3				3
Lumbri-culida	Lumbriculidae	*Lumbriculus variegatus*	1	1	1			3
Megaloptera	Corydalidae	*Corydalus cornutus*		1				1
Odonata	Aeshnidae	*Boyeria vinosa*	1			1		2
	Calop-terygidae	*Hetaerina americana*					1	1
	Coena-grionidae	*Enallagma sp.*		1				1
	Gomphidae	*Progomphus sp.*	1					1
		Progomphus obscurus					3	3
Plecoptera	Leuctridae	*Paraleuctra sara*		1				1
	Nemouridae	*Amphinemura delosa*		3				3
	Perlodidae	*Isoperla nana*		1				1
Trichoptera	Hydro-psychidae	*Ceratopsyche sparna*	1					1
		Cheumatopsyche sp.	6	9	2	4	13	34
		Hydropsyche frisoni	2	1			2	5
		Hydropsyche scalaris						0
		Parapsyche cardis	1					1
Tubificida	Naididae	*Dero digitata*		1				1
		Homocheata naidina		1				1
Totals			24	151	117	258	54	604

Location: Standing Boy Middle

Order	Family	Genus and Species	Jun-98	Oct-98	Feb-99	May-99	Jul-99	Totals
Bivalvia	Corbiculidae	*Corbicula fluminea*	22	59	17	35	17	150
Branchio-bdellida	Branchiob-dellidae	*Cambarincola sp.*	5					5
Coleoptera	Elmidae	*Anacyronxy variegatus*			1	1	1	3
		Dubiraphia sp.	1	2	1			4
		Macronychus sp.	2					2
		Macronychus glabratus					1	1
		Stenelmis sp.		2			4	6
	Gyrinidae	*Dineatus sp.*				2	2	4
		Dineatus horni					4	4
	Hydrophilidae	*Sperchopsis tessallatus*		1				1
Collem-bola	Entomobryidae	*Sinella sp.*		2				2
Decapoda	Cambaridae	*Procambarus spiculifer*	7		5	1		13
Diptera	Chironomidae	*(could not be further identified)*	6				1	7
		Ablabesmyai mallochi	2	4	3	2		11
		Ablabesmyia rhampe	6					6
		Apedilum sp.	2					2
		Brillia flavifrons		1				1
		Corynoneura B		2	2	8	1	13
		Corynoneura D			5			5
		Crictopus/Orthocladius sp.	2		3		1	6
		Crictopus politus			2			2
		Crypt-ochironomus sp.	9			11	3	23
		Dicrotendipes neomodestus	2			2		4
		Hayesomyia		2	4	1		7

		senata						
		Hudsonimyia karelena				6	2	8
		Hydrobaenus sp.			4			4
		Krenopelopia sp.		2				2
		Micropsectra sp.	6					6
		Microptendipes sp.	2					2
		Nanocladius sp.	1					1
		Nanocladius distinctus	2					2
		Natarsia sp.				1		1
		Nilotanypus americanus				1		1
		Nilothauma sp.	1					1
		Paratendipes sp.	2					2
		Phanopsectra obediens				23	1	24
		Phanopsectra punctipes	10					10
		Polypedilum convictum gr.	10	23	34	38	12	117
		Polypedilum tritum				4	11	15
		Procladius sp.			2			2
		Orthocladius annectens				1		1
		Rheocrictopus robacki		1	22	9	8	40
		Rheosmittia sp.			4			4
		Rheotanytarsus sp.	22	4	15	47	44	132
		Robackia demeijerei	9		3			12
		Tanytarsus A			4			4
		Tanytarsus C	6	14	3	1		24
		Tanytarsus D			4	2	13	19
		Tanytarsus S				1		1
		Tanytarsus U	1			33	20	54
		Thienemanniella xena			3	3	4	10
		Thienemannimyia		1	1	7	17	26

		gr.						
		Xestochironomus sp.			1			1
		Xylotopus par				2		2
		Zalutschia sp.					2	2
	Culicidae	*Anopheles punctipennis/per.*		1				1
		Anopheles quadrimaculatus		1				1
	Simulidae	*Simulium sp.*		1	9			10
		Simulium harpi					9	9
		Simulium parnassum			1		3	4
		Simulium slossonae				7	18	25
	Stratiomyidae	*Oxycera sp.*			1			1
	Tabanidae	*Chrysops sp.*		1				1
		Haematopota sp.		1				1
	Tanyderidae	*Protoplasa fitchii*		1				1
	Tipulidae	*Cryptolabis sp.*		1				1
		Hexatoma sp.	4	13	3	2	1	23
		Ormosia sp.			1			1
		Polymera sp.	3					3
		Prionocera sp.	1					1
		Tipula sp.	2					2
Ephem-eroptera	Baetidae	*Acerpenna sp.*				2	6	8
		Baetis amplus					3	3
		Baetis intercalaris					1	1
		Heterocleon petersi	3					3
		Paracleodes sp.				2		2
		Procleon sp.				2		2
		Psuedocleon sp.	3					3
	Caenidae	*Brachycercus nitidus*	3					3
		Caenis sp.	1			3		4
	Ephemeridae	*Hexagenia limbata*		3				3
	Ephemerillidae	*Ephemerella sp.*	1					1

	Heptageniidae	*Stenonema carlsoni*	1					1
		Stenonema femoratum	4		1			5
		Stenonema modestum	38	7	68	41	28	182
	Isonychiidae	*Isonychia sp.*			2		1	3
	Tricorythidae	*Tricorythodes sp.*	66	1		5	2	74
Gastro-poda	Lymnaeidae	*Fossaria sp.*		1				1
	Physidae	*Physella sp.*		2				2
Hemiptera	Corixidae	*Trichorixia sexcinta*				1		1
	Veliidae	*Rhagovelia obesa*					1	1
Isopoda	Asellidae	*Asellus obtusus*	1					1
		Lirceus lineatus	1		2	2		5
Lumbri-culida	Lumbriculidae	*Lumbriculus variegatus*			4			4
Megalop-tera	Corydalidae	*Corydalus cornutus*					1	1
Odonata	Aeshnidae	*Boyeria vinosa*					2	2
	Calopterigidae	*Calopteryx maculata*		2				2
	Coenagrionidae	*Argia sp.*			1			1
		Argia tibialis	1					1
	Corduliidae	*Didymops transversa*		4				4
	Gomphidae	*Dromogomphus spinosus*		1				1
		Gomphus germinatus		1				1
		Gomphus lividus				2		2
		Ophiogomphus mainensis		1				1
		Progomphus obscurus		2	3		1	6
Orthoptera	Tridactlylidae	*Ellipes sp.*	1					1
Plecoptera	Perlidae	*(could not be further identified)*			1			1
		Beloneuria sp.					1	1
		Perlesta placida				8		8
		Perlinella ephyre			1	1		2

	Taeniop-terygidae	*Strophopteryx fasciata*			1			1
Trichop-tera	Hydropsychidae	*Cheumatopysche sp.*	2	1		8	23	34
		Hydropsyche decalda	2					2
		Hydropsyche scalaris	2					2
		Parapsyche sp.					1	1
	Hydroptilidae	*Hydroptila sp.*					6	6
	Leptoceridae	*Setodes incerta*	1					1
	Lymnephilidae	*(could not be further identified)*		1				1
	Philopotamidae	*Chimarra sp.*					2	2
Tubificida	Naididae	*Homocheata naidina*			1			1
	Tubificidae	*(could not be further identified)*			1			1
Totals			279	166	246	327	279	1297

Location: Standing Boy Lower

Order	Family	Genus and Species	Oct-98	Feb-99	May-99	Jul-99	Totals
Bivalvia	Corbiculidae	*Corbicula fluminea*		4	1	10	15
Decapoda	Cambaridae	*Procambarus spiculifer*			2	1	3
Diptera	Ceratopogonidae	*Probezzia sp.*		3			3
	Chironomidae	*(could not be identified further)*				-	8
		Ablabesmyia jante				4	4
		Ablabesmyia mallochi	1		4	35	40
		Chironomus sp.		1			1
		Cladopelma sp.				6	6
		Crictopus/Orthocladius sp.				2	2
		Cryptochironomus sp.			8		8
		Cryptotendipes sp.				7	7
		Dicrotendipes A				10	10
		Dicrotendipes neomodestus			5	13	18
		Phanopsectra obediens group			19	3	22
		Phanopsectra punctipes group		2		8	10
		Procladius sp.			6	16	22
		Stictochironomus devinctus				3	3
		Tanytarsus sp.				15	15
		Tanytarsus D			2		2
		Tanytarsus L	1			4	5
		Tanytarsus U			14	6	20
		Thienemannimyia gr.			1		1
	Tipulidae	*Tipula sp.*		1			1
Ephemeroptera	Baetidae	*Paracleodes sp.*				15	15
	Caenidae	*Caenis sp.*				1	1
	Ephemeridae	*Hexagenia limbata*	5			1	6
	Heptageniidae	*Stenonema modestum*				4	4
Gatropoda	Planorbidae	*Planorbella armigera arm.*		1			1
Lepidoptera	Pyralidae	*Crambus sp.*		1			1
Lumbricula	Lumbriculidae	*Lumbriculus variegatus*		1			1

Odonata	Coryduliidae	*Didymops transversa*				1	1
	Gomphidae	*Lanthus sp.*				1	1
Totals			7	14	62	174	257

Location: Upatoi Creek Upper

Order	Family	Genus and Species	Oct-98	Feb-99	May-99	Jul-99	Totals
Bivalvia	Corbiculidae	*Corbicula fluminea*	6	2	27	10	45
Coleoptera	Dryopidae	*Helichus rugosus*	1				1
	Elmidae	*Anacyronyx variegatus*		2			2
		Stenelmis sp.		4	7	4	15
		Stenelmis hungerfordi	2				2
	Gyrinidae	*Dineutus sp.*			8		8
		Gyrinus sp.			1		1
	Hydrophilidae	*Tropisternus sp.*		1			1
Collembola	Isotomidae	*Isotoma sp.*			2		2
Decapoda	Cambaridae	*Cambarus latimanus*		1	2		3
		Procambarus spiculifer	2		5		7
Diptera	Ceratopo-gonidae	*Bezzia/Palpomyia sp.*			6	2	8
	Chironomidae	*(could not be identified further)*	11				11
		Ablabesmyai rhampe gr.	1				1
		Cheatocladius sp.		1			1
		Cladotanytarsus sp.			1		1
		Corynoneura B	8	18	4		30
		Corynoneura D		3			3
		Crictopus/Orth. sp.	7	7	51		65
		Crictopus bicinctus		2			2
		Crictopus politus	1				1
		Crictopus sylvestris		1			1
		Crictopus trifascia gr.			2		2
		Cryptochironomus sp.	1			1	2
		Hayesomyia senata		12			12
		Hudsonimyia sp.		1	6		7
		Hudsonimyia karelena	1		5		6
		Hydrobaenus sp.	1	1			2
		Labrudinia neopilosella	1				1
		Nanocladius sp.	3				3
		Nanocladius balticus	5				5
		Nilotanypus sp.	1				1

		Nilotanypus americanus		1			1
		Nilotanypus fimbriates		1			1
		Paracladopelma loganae			2		2
		Paracrictopus sp.		2	1		3
		Parametriocnemus lundbeckii		10			10
		Phanopsectra obediens gr.			2		2
		Phanopsectra punctipes gr.	1				1
		Polypedilum convictum gr.	41	66	18	3	128
		Polypedilum tritum			2		2
		Rheocrictopus robacki	8	1	3		12
		Rheopelopia sp.			2		2
		Rheotanytarsus sp.	12	4	54	8	78
		Robackia demeijerei				1	1
		Stenochironomus sp.	1				1
		Tanytarsus sp.		1	6		7
		Tanytarsus C	1	3	1		5
		Tanytarsus D	1		5		6
		Tanytarsus J	1				1
		Tanytarsus L	2	1			3
		Tanytarsus U		3	16		19
		Thienemanniella B	2				2
		Thienemanniella xena	2	2			4
		Thienemannimyia gr.	5	3	11	3	22
	Empididae	*Hemerodromia sp.*		1	1		2
	Simulidae	*Simulium sp.*	24				24
		Simulium haysi	9				9
		Simulium parnassum		10			10
		Simulium slossonae				4	4
	Tipulidae	*Hexatoma sp.*	3	2		2	7
		Limonia sp.	1				1
		Ormosia sp.		6	1		7
		Tipula sp.	2	2			4
Ephem-eroptera	Baetidae	*Acerpenna sp.*		3			3
		Baetis sp.	1			3	4

		Baetis amplus	3			3	6
		Baetis brunneicolor		5	31		36
		Centroptilum sp.	1				1
		Heterocleon curiosum	1				1
		Paracleodes sp.			24		24
	Caenidae	*Brahcycercus nitidus*				1	1
		Caenis sp.		2	5		7
	Ephem-erillidae	*Ephemerella inconstans*		9			9
		Ephemerella invaris		22			22
		Serratella deficiens				-1	1
	Heptageniidae	*Stenacron interpunctatum*	1				1
		Stenonema annexum	1				1
		Stenonema femoratum	3				3
		Stenonema modestum	50	19	5	14	88
		Stenonema terminatum	15				15
	Isonychiidae	*Isonychia sp.*		2			2
	Tricorythidae	*Tricorythodes sp.*			67	5	72
Hemiptera	Gelasto-coridae	*Gelastocoris oculatus oculatis*			1		1
	Gerridae	*Metrobates hesperius*	1				1
		Neogerris hesione				1	1
	Veliidae	*Rhagovelia obesa*	2				2
Isopoda	Asellidae	*Lirceus lineatus*	1				1
Lumbri-culida	Lumbriculidae	*Lumbricula variegatus*	1		1	4	6
Megalop-tera	Corydalidae	*Corydalus cornutus*	2				2
Odonata	Aeshnidae	*Boyeria vinosa*			3	1	4
	Gomphidae	*Ophiogomphus mainensis*		1		1	2
		Progomphus obscurus	22	3	3	1	29
Plecoptera	Chloro-perlidae	*Suwallia marginata*			1		1
	Leuctridae	*Paraleuctra sara*		1			1
	Perlidae	*(could not be identified further)*	1				1
		Acroneuria internata			4	1	5
		Acroneuria mela		1			1
		Hansonoperla sp.		1			1

		Perlesta placida		4	7		11
		Perlinella ephyre	4				4
	Perlodidae	*Clioperla sp.*		2			2
		Isoperla bilineata		24			24
		Isoperla holochlora		5			5
		Remenus bilobatus	1				1
	Nemouridae	*Amphineura delosa*		3			3
	Taeniop-terygidea	*Oempteryx contorta*		2			2
Trichoptera	Brechy-centridae	*Brachycentrus sp.*				1	1
	Hydro-psychidae	*Cheumatopysche sp.*	18		15	11	44
		Hydropsyche frisoni	2		4		6
		Hydropsyche scalaris	2				2
	Hydroptilidae	*Hydroptila sp.*			3		3
	Polycentro-podidae	*Neuroclipsis sp.*	1				1
Tubificidae	Naididae	*Amphicheata americana*		1			1
Totals			302	285	426	86	1099

Location: Upatoi Creek Middle

Order	Family	Genus and Species	Oct-98	Feb-99	May-99	Jun-99	Total
Bivalvia	Corbiculidae	*Corbicula fluminea*	1				1
Coleoptera	Elmidae	*Anacyronyx variegatus*			1		1
		Stenelmis sp.	2	1	8	1	12
		Stenelmis bicarinate				3	3
		Stenelmis hungerfordi			2		2
Diptera	Chironomidae	(unable to identify further)	1				1
		Ablabesymia mallochi		2			2
		Crictopus/Orthocladius sp.	1	2			3
		Hudsonimyai karalena			4		4
		Paracrictopus sp.			8		8
		Paratendipes albimanus		1			1
		Phanopsectra obediens group		3			3
		Phanopsectra punctipes group		12			12
		Polypedilum convictum group	1	5	1		7
		Polypedilum halterale group		2			2
		Polypedilum tritum		1	1		2
		Procladius sp.		2			2
		Rheocrictopus robacki		1	11		12
		Rheotanytarsus sp.			18		18
		Robackia claviger			15		15
		Robackia demeijerei	1			2	3
		Stenochironomus sp.			1		1
		Stictochironomus devinctus		3			3
		Tanytarsus sp.		1			1

Order	Family	Species					
		Tanytarsus C			1		1
		Tanytarsus S		5			5
		Thienemannimyia gr.			1		1
	Empididae	*Hemerodromia sp.*			6		6
	Simulidae	*Simulium slossonae*			1		1
	Tipulidae	*Ormosia sp.*		5			5
Ephemerop-tera	Baetidae	*Acerpenna sp.*			2		2
	Baetiscidae	*Baetisca obesa*		1			1
	Caenidae	*Caenis sp.*		1			1
	Ephemeridae	*Hexagenia limbata*		4		-	4
	Heptageniidae	*Stenonema modestum*			1		1
	Isonychiidae	*Isonychia sp.*			2		2
	Tricorythidae	*Tricorythodes sp.*			1		1
Lumbri-culida	Lumbri-culidae	*Lumbriculus variegatus*		9		1	10
Odonata	Aeshnidae	*Boyeria vinosa*				1	1
Plecoptera		(unable to identify further)	4				4
	Perlidae	*Acroneuria carolinensis*			1		1
		Attaneuria ruralis			1		1
		Perlesta placida			2		2
		Perlinella ephyre			16		16
	Perlodidae	*Isoperla orata*		2			2
	Ptero-narcyidae	*Pteronarcys dorsata*			7		7
Trichoptera	Brachy-centridae	*Brachycentrus sp.*			14		14
	Hydro-psychidae	*Cheumatopsyche sp.*		1	46	2	49
		Hydropsyche frisoni			36		36
	Hydroptilidae	*Orthotrichia sp.*			1		1
Tubificidae	Naididae	*Amphicheata leydigi*		1			1
Totals			11	65	209	10	295

Location: Upatoi Creek Lower

Order	Family	Genus and Species	Oct-98	Feb-99	May-99	Jul-99	Totals
Bivalvia	Corbiculidae	*Corbicula fluminea*		1	1		2
Coleoptera	Elmidae	*Anacyronxy variegatus*				1	1
		Dubiraphia sp.	3				3
		Macronychus glabratus				2	2
		Stenelmis sp.			13	7	20
		Stenelmis hungerfordi			4	2	6
		Stenelmis miribilis				2	2
Decapoda	Cambaridae	(unable to identify further)			1		1
	Palaemonidae	*Macrobranchium acanthurus*	1				1
Diptera	Cerato-pogonidae	*Bezzia/Palpomyia sp.*	1	1			2
	Chaoboridae	*Chaeoborus punctipennis*		1			1
	Chironomidae	(unable to identify further)				1	1
		Corynoneura B	1		1		2
		Crictopus/Orthocladius sp.	1		2	3	6
		Hudsonimyai karelena		1			1
		Nanocladius sp.	1				1
		Nilotanypus sp.			1		1
		Nilotanypus americanus	1				1
		Polypedilum convictum group		2			2
		Polypedilum tritum			4		4
		Paracrictopus sp.			3		3
		Parakeifferiella sp.			1		1
		Procladius bellus var. 1	1				1
		Rheocrictopus robackia	1		3		4
		Rheotanytarsus sp.			9		9
		Robackia claviger			3		3
		Robackia demeijerei	1			1	2
		Stictochironomus				1	1

		devinctus					
		Tanytarsus sp.	7				7
		Tanytarsus D			4		4
		Tanytarsus U			1		1
		Thienemannimyia group	1		6		7
	Simulidae	*Simulium sp.*				1	1
	Tipulidae	*Hexatoma sp.*	1				1
		Tipula sp.			1		1
Ephemerop-tera	Baetidae	*Baetis sp.*			2		2
		Baetis brunneicolor	1				1
	Heptageniidae	*Stenonema integrum*			1		1
		Stenonema modestum			1		1
	Isonychidae	*Isonychia sp.*			3	3	6
	Neoephemeridae	*Neoephemera purpurea*	1				1
	Tricoryithidae	*Tricorythodes sp.*			2		2
Hemiptera	Veliidae	*Rhagovelia obesa*	1				1
Megaloptera	Corydalidae	*Corydalus cornutus*				2	2
Odonata	Corduliidae	*Macromia taeniolata*	1		1		2
	Gomphidae	*Dromogomphus spinosus*	2				2
Plecoptera		(unable to identify further)	1				1
	Perlidae	*Beloneuria sp.*				1	1
		Hansonoperla sp.			3		3
		Perlesta placida			1		1
Trichoptera	Brachycentridae	*Brachycentrus sp.*			1		1
	Hydropsychidae	*Cheumatopysche sp.*			6	4	10
		Hydropsyche frisoni			6	7	13
	Hydroptilidae	*Ochrotrichia sp.*			1		1
Totals			28	6	86	38	158

Appendix 2: Macroinvertebrate species list with total abundances for all sites combined and modified Hisenhoff biotic index values (Hilsenhoff 1987, Barbour et al. 1999)

Order	Family	Genus and Species	Number	Index Value
Amphipoda	Hyallellidae	*Hyallela azteca*	4	7.9
Bivalvia	Corbiculidae	*Corbicula fluminea*	1040	6.3
Branchiobdellida	Branchio-bdellidae	*Cambarinocola sp.*	5	6
Coleoptera	Dryopidae	*Helichus rugosus*	1	5.4
	Dytiscidae	*Hydroporus sp.*	1	8.9
	Elmidae	*Anacyronyx variegatus*	17	6.9
		Dubiraphia sp.	11	6.4
		Macronychus sp.	2	4.7
		Macronychus glabratus	3	4.7
		Stenelmis sp.	59	5.4
		Stenelmis bicarinata	6	5.4
		Stenelmis hungerfordi	14	5
		Stenelmis miribilis	3	5.4
	Gyrinidae	*Dineutus sp.*	12	5.5
		Dineutus horni	4	5.5
	Gyrinidae	*Gyrinus sp.*	4	6.3
	Helophoridae	*Heloporus linearis*	1	7.9
	Hydrophilidae	*Berosus sp.*	17	8.6
		Berosus striatus	4	8.6
		Sperchopsis tessallatus	1	6.5
		Tropisternus sp.	1	9.8
	Psephenidae	*Ectopria thoraacica*	1	4.3
Collembola	Entomobryidae	*Sinella sp.*	2	10
	Isotomidae	*Isotoma sp.*	2	10
Decapoda	Cambaridae	(unable to identify further)	1	8.1
		Cambarus latimanus	8	8.1
		Procambarus spiculifer	26	9.5
	Palaemonidae	*Macrobranchium acanthurus*	1	4
Diptera	Cerato-pogonidae	*Bezzia/Palpomyia sp.*	12	6
		Probezzia sp.	4	6
	Chaoboridae	*Cheaeoborus punctipennis*	1	8.5

		punctipennis		
	Chironomidae	(unable to identify further)	28	6
		Ablabesmyia jante	4	7.1
		Ablabesmyia mallochi	103	7.6
		Ablabesmyia rhampe	7	7
		Alotanypus aris	2	6
		Apedilum sp.	2	6
		Brillia sp.	2	5.2
		Brillia flavifrons	1	5.2
		Cheatocladius sp.	1	6
		Chironomini III sp.	1	6
		Chironomus sp.	26	9.8
		Cladopelma sp.	6	2.5
		Cladotanytarsus sp.	4	3.7
		Corynoneura B	57	6.2
		Cornynoneura D	8	6.2
		Crictopus (Isocladius) sp.	1	7
		Crictopus/Orthocladius sp.	336	7
		Crictopus bicinctus	103	8.7
		Crictopus politus	20	7
		Crictopus sylvestris	5	10
		Crictopus trifascia group	2	7
		Cryptochironomus sp.	55	8
		Cryptotendipes sp.	7	6.1
		Dicrotendipes A	10	7.9
		Dicrotendipes neomodestus	47	8.3
		Diplocladius sp.	1	7.7
		Eukiefferiella claripennis group	5	5.7
		Hayesomyia sp.	11	4.6
		Hayesomyia senata	53	4.6
		Helopelopia sp.	1	6
		Hudsonimyia sp.	7	6
		Hudsonimyia karelena group	40	6
		Hydrobaenus sp.	7	9.6
		Krenopelopiasp.	2	6
		Labrudinia sp.	1	3.8

		Labrudinia neopilosella	1	7
		Larsia sp.	1	8.3
		Meropelopia sp.	2	7
		Microchironomus sp.	1	6
		Micropsectra sp.	9	1.4
		Microptendipes sp.	2	6.2
		Nanocladius sp.	5	7.2
		Nanocladius balticus	5	7.2
		Nanocladius distinctus	4	7.2
		Nanocladius rectinervis	2	7.2
		Natarsia sp.	2	10
		Nilotanypus sp.	2	4
		Nilotanypus americanus	4	4
		Nilotanypus fimbriates	1	4
		Nilothauma sp.	1	5.5
		Orthocladius annectens	24	6
		Paracladopelma sp.	3	6.4
		Paracladopelma loganae	2	6.4
		Paracladopelma undine	1	5.2
		Paracrictopus sp.	14	6
		Parakeifferiella A	1	5.9
		Parametriocnemus lundbecki	12	3.7
		Paratendipes sp.	2	5.3
		Paratendipes albimanus	1	5.3
		Phanopsectra obediens group	296	7
		Phanopsectra punctipes group	46	6.8
		Polypedilum convictum group	491	5.3
		Polypedilum halterale group	2	7.2
		Polypedilum tritum	381	6
		Procladius sp.	29	9.3
		Procladius bellus var. 1	2	9.3
		Procladius bellus var. 3	3	9.3
		Rheocrictopus robacki	211	7.7
		Rheopelopia sp.	4	6
		Rheosmittia sp.	4	6
		Rheotanytarsus sp.	583	6.4

		Robackia claviger	18	2.4
		Robackia demeijerei	27	4.3
		Saetheria tylus	1	8.1
		Stenochironomus sp.	2	6.4
		Stictochironomus devinctus	8	6.7
		Stilocladius sp.	3	6
		Tanytarsus sp.	34	6.7
		Tanytarus A	10	6.7
		Tanytarsus B	1	6.7
		Tanytarsus C	64	6.7
		Tanytarsus D	82	6.7
		Tanytarsus J	1	6.7
		Tanytarsus L	19	6.7
		Tanytarsus P	3	6.7
		Tanytarsus S	6	6.7
		Tanytarsus U	236	6.7
		Thienemanniella A	1	6
		Thienemanniella B	4	6
		Thienemanniella xena	29	6
		Thienemannimyia group	115	6
		Xestochironomus sp.	1	6
		Xylotopus par	2	6.6
		Zalutschia sp.	2	7
	Culicidae	*Anopheles punctipennis*	1	9.1
		Anopheles quadrimacula	1	9.1
		Wyeomyia sp.	1	8
	Empididae	*Hemerodromia sp.*	22	6
	Simulidae	*Simulium sp.*	42	4.4
		Simulium decorum	12	4.4
		Simulium harpi	9	4.4
		Simulium haysi	12	4.4
		Simulium parnassum	70	4.4
		Simulium podestemi	2	4.4
		Simulium slossonae	61	4.4
		Simulium venustum	2	7.4
	Stratiomyidae	*Oxycera sp.*	1	8
	Tabanidae	*Chrysops sp.*	1	7.3
		Haemotopota sp.	1	8
	Tanyderidae	*Protoplasa fitchii*	1	5
	Tipulidae	*Cryptolabis sp.*	2	3

		Dicranota sp.	9	0
		Hexatoma sp.	34	4.7
		Limonia sp.	3	10
		Ormosia sp.	14	6.5
		Polymera sp.	3	4.7
		Prionocera sp.	1	4
		Tipula sp.	26	7.7
Ephemeroptera	Baetidae	(unable to identify further)	1	4
		Acentrella sp.	12	4
		Acerpenna sp.	13	4
		Baetis sp.	7	6
		Beatis amplus	61	6
		Baetis brunneicolor	45	6
		Baetis intercalaris	1	5.8
		Centropilum sp.	1	6.3
		Heterocleon curiosum	1	3.6
		Heterocleon petersi	3	3.6
		Labiobates sp.	3	6
		Paracleodes sp.	58	8.7
		Procleon sp.	2	5
		Psuedocleon sp.	5	4.4
	Baetiscidae	*Baetisca obesa*	1	4
	Caenidae	*Brachycercus nitidus*	4	3.5
		Caenis sp.	24	7.6
	Ephemeridae	*Hexagenia limbata*	13	4.7
	Ephemerillidae	*Ephemerella sp.*	1	2.9
		Ephemerella inconstans	9	2.9
		Ephemerella invaris	22	2.2
		Serratella deficiens	1	2.7
	Heptageniidae	*Stenacron interpunctatum*	1	7.1
		Stenonema annexum	2	5.8
		Stenonema carlsoni	1	2.1
		Stenonema femoratum	8	7.5
		Stenonema integrum	1	5.5
		Stenonema modestum	399	5.8
		Stenonema smithae	4	3
		Stenonema terminatum	23	4.5
	Isonychiidae	*Isonychia sp.*	180	3.8
		Isonychia arida	2	3.8

		Isonychia sicca	17	3.8
	Neo-ephemeridae	*Neoephemera purpurae*	1	2.1
	Tricorythodae	*Tricorythodes sp.*	311	5.4
Gastropoda	Lymnaeidae	*Fossaria sp.*	1	6
	Physidae	*Physa sp.*	2	8
		Physella sp.	4	9.1
		Physella gyrina	6	9.1
	Planorbellidae	*Gyraulus sp.*	1	8
		Planorbella armigera armigera	23	6
Hemiptera	(nymph)	(unable to identify further)	1	7
	Belostomatidae	*Belastoma testaceum*	2	9.8
	Corixidae	(unable to identify further)	1	9
		Trichorixa sexcincta	1	5
	Gelastocoridae	*Gelastocoris oculatus oculatus*	1	5
	Gerridae	*Metrobates hesperius*	1	5
		Neogerris hesione	3	5
		Rheumatobates tenuipes	1	5
		Trepobates inermis	1	5
	Hebridae	*Lipogomphus sp.*	1	5
	Pleidae	*Paraplea sp.*	1	6
	Veliidae	*Rhagovelia obesa*	10	6
Hirudinae	Glossi-phoniidae	*Helobdella triserialis*	1	8.9
Isopoda	Asellidae	*Asellus obtusus*	1	9.4
		Lirceus fontinalis	2	7.7
		Lirceus lineatus	31	7.7
Lepidoptera	Pyralidae	*Crambus sp.*	1	5
Lumbriculida	Lumbriculidae	*Lumbriculus variegatus*	75	7.3
Megaloptera	Corydalidae	*Corydalus cornutus*	14	5.6
Odonata	Aeshnidae	*Boyeria vinosa*	17	6.3
	Calopterigidae	*Calopteryx maculata*	2	8.3
		Hataerina americana	1	6.2
	Coenagrionidae	*Argia sp.*	2	6
		Argia bipunctulata	7	6
		Argia tibialis	1	6
		Enallagma sp.	1	9

		Enallagma divagans	1	9
		Nahellenia gracilius	1	9
	Corduliidae	*Didymops transversa*	5	5
		Macromia taeniolata	3	6.7
	Gomphidae	*Dromogomphus spinosus*	3	6.3
		Gomphus aponymius	1	5
		Gomphus germinatus	1	5
		Gomphus lividus	2	5
		Lanthus sp.	1	2.7
		Lanthus vernalis	1	2.7
		Ophiogomphus carolinus	1	6.2
		Ophiogomphus mainensis	3	6.2
		Progomphus sp.	1	8.7
		Progomphus obscurus	38	8.7
Orthoptera	Tridactlylidae	*Ellipes sp.*	1	5
Plecoptera		(unable to identify further)	5	3
	Chloroperlidae	*Suwallia marginata*	1	0
	Leuctridae	*Paraleuctra sara*	2	0
	Nemouridae	*Amphineura delosa*	6	3.4
	Perlidae	(unable to identify further)	2	1
		Acroneuria carolininsis	1	0
		Acroneuria internata	5	2.2
		Acroneuria mela	1	0.9
		Attaneuria ruralis	1	1
		Beloneuria sp.	2	0
		Hansonoperla sp.	4	1
		Perlesta placida	25	4.9
		Perlinella ephyre	22	0
	Perlodidae	*Clioperla sp.*	2	4.8
		Isoperla bilineata	24	5.5
		Isoperla holochlora	5	0
		Isoperla nana	1	2
		Isoperla orata	2	0
		Remenus bilobatus	1	0.3
	Pteronarcyidae	*Pteronarcys sp.*	2	1.7
		Pteronarcys dorsata	7	1.8
	Taenioptery-gidea	*Oempteryx contorta*	2	2

		Stratophopteryx faxciata	1	2.5
Trichoptera	Brachy-centridae	*Brachycentrus sp.*	23	2.2
	Hydro-psychidae	*Ceratopsyche sp.*	2	1
		Ceratopsyche alhedra	22	0
		Ceratopsyche spama	1	3.2
		Cheumatopsyche sp.	739	6.6
		Diplectrona modesta	1	2.2
		Hydropsyche decalda	2	4.1
		Hydropsyche frisoni	149	1.8
		Hydropsyche scalaris	79	3
		Hydropsyche venularis	1	5.3
		Parapsyche sp.	1	0
		Parapsyche cardis	1	0
	Hydroptilidae	*Hydroptila sp.*	31	6.2
		Ochrotrichia sp.	1	7.2
		Orthotrichia sp.	1	7
	Leptoceridae	*Setodes incerta*	1	0.9
	Lymnephilidae	(unable to identify further)	1	4
	Philopotamidae	*Chimarra sp.*	3	2.8
	Poly-centropodidae	*Neuroclipsis sp.*	1	4.4
		Polycentropus sp.	2	3.5
Tubificidae	Naididae	(unable to identify further)	2	9
		Amphicheata americana	3	9
		Amphicheata leydigi	2	9
		Dero sp.	1	10
		Dero digitata	3	10
		Homocheata naidina	2	9
		Ophidonais serpentina	1	9
	Tubificidae	(unable to identify further)	1	10
		Aulodrilus limniobius	1	5.2
		Branchiura sowerbyi	29	8.4
Totals			8137	