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Relationships Between Macroinvertebrate Communities and Environmental Characteristics of Headwater Streams in Central Louisiana

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RELATIONSHIPS BETWEEN MACROINVERTEBRATE COMMUNITIES AND ENVIRONMENTAL CHARACTERISTICS OF HEADWATER STREAMS IN CENTRAL LOUISIANA

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for my family

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ABSTRACT

Headwater streams are an integral part of any watershed system because they strongly influence the physical, chemical, and biological components of downstream reaches. Little information exists about macroinvertebrate community structure, spatiotemporal variation, or their relationships with environmental factors in low-gradient headwater streams of the subtropical coastal plain region in the Southern US. These headwater streams are typically slow moving, capable of accumulating large amounts of organic material, and often become intermittent during the dry season. Research is needed to understand the effects of these unique characteristics on stream health and ecology. This study aimed to determine aquatic macroinvertebrate community structure, identify spatial and seasonal patterns, and investigate relationships between the macroinvertebrate community and environmental variables in seasonally hypoxic, first- and second-order streams with varying flow permanence in a lowland subtropical watershed located in central Louisiana, USA. Eleven monitoring locations throughout the watershed were sampled twice over one year for macroinvertebrates and physicochemical parameters including velocity, wetted area, nutrients, and dissolved oxygen (DO). Aquatic benthic macroinvertebrates were sampled within a 160-m stream reach with a modified core sampler that was specially designed for the low-gradient system comprised. Seasonal and spatial differences between water quality characteristics, individual taxa, and biological metrics were determined. Correlation analysis detected seasonal differences in environmental variables that were related to abundances of individual taxa. Spring indicated by a positive correlation with total suspended solids and negatively with temperature and nitrate was positively associated with crustaceans and negatively associated with chironomids. Most notably, the burrowing mayfly, *Hexagenia,* was positively correlated to DO levels. Many of the metrics,

including percent of Ephemeroptera, Plecoptera, Trichoptera (EPT) taxa, differed between sites with varying DO levels and flow permanence. Surprisingly, analysis of variance did not detect seasonal differences among the metrics. This study is one of the first comprehensive assessment on macroinvertebrate communities with detailed hydrologic and water quality measurements in the headwaters of a low-gradient, subtropical watershed. The study supports the importance of recognizing stream permanence in water quality assessments. In addition, the determination of useful metrics for low-gradient, headwater streams are suggested for future research.

INTRODUCTION

In 1999, the Flat Creek watershed in central Louisiana was classified by the state as impaired because it failed to meet the water quality criteria set for fish and wildlife propagation (LDEQ 2004). The suspected causes of impairment include high total dissolved solids (TDS) and low dissolved oxygen (DO) concentrations, both of which may have resulted from human activities, natural background conditions, or a combination of both. These criteria are based on national standards that may not be applicable to the South Central Plains Ecoregion (SCPE, Omernik 1987), which includes the Flat Creek watershed in central Louisiana.

Stream health can be characterized by chemical, physical, and biological properties of the water body. An integrated monitoring program, which includes biological assessments, can provide a more detailed picture of a stream ecosystem compared to chemistry-based monitoring programs that may reveal only a limited amount of information and possibly miss anthropogenic impacts (Karr and Yoder 2004). Many state water quality monitoring programs have integrated chemical, physical, and biological characteristics. However, a few states, such as Louisiana, depend solely on physicochemical parameters and some biological indicators, including fecal coliform and other microbes, for water quality assessment, and fail to include macroinvertebrates and fishes. An integrated assessment using biological indicators of water quality may be particularly useful in determining the biological integrity of headwater streams, such as those in the Flat Creek watershed.

Headwater streams are important in structuring the macroinvertebrate community in downstream reaches (Vannote et al. 1980, Richardson and Danehy 2007). These streams have a unique hydrology that is often driven by precipitation and influenced by local geomorphological features (Rosgen 1994). It has been reported that these streams comprise over 50% of total

stream length in the contiguous U.S., including Hawaii, with approximately half of the headwater stream lengths being intermittent or ephemeral (Nadeau and Rains 2007). Because of their influence on the rest of the stream network, much work has been done to describe and understand their relationships and ecological significance on downstream receiving waters. In recent years headwater stream systems have gained special attention due to a political debate over the inclusion of these streams as "protected waters" under the Clean Water Act (Nadeau and Rains 2007), and researchers have increased efforts to examine the contribution of these systems to the ecological function of downstream reaches (Meyer et al. 2007). The Flat Creek watershed offers an opportunity to explore the macroinvertebrate community assemblage and function along a hydrological gradient in a headwater system.

Biological indicators of water quality can include all living components of an aquatic ecosystem such as fish, aquatic insects, or algae. Physical and chemical water quality parameters are point-in-time characterizations, whereas biotic community composition reflects long-term adaptations to environmental conditions. Providing a comprehensive measure of aquatic health, biological assessments are increasingly being used to determine the impacts on water quality of land use practices such as forestry (Vowell 2001) and agriculture (Genito et al. 2002). Aquatic macroinvertebrates are organisms large enough to see with an unaided eye, such as insect larvae, snails, and crustaceans that live all or part of their life in water. Macroinvertebrates are commonly used as indicators of water quality conditions because they are ubiquitous and relatively easy to collect, and unlike chemical parameters, they are typically indicative of long-term changes.

Over the past several decades, research has been conducted to develop biological assessments based on aquatic macroinvertebrates in many geographical regions and under

various climatic conditions (Lenat 1988, Barbour et al. 1999). Most of the research has focused on macroinvertebrates in perennial streams, although some was conducted in intermittent streams with an emphasis on drought effect in semi-arid regions (e.g., Stanley et al. 1994, Boulton and Stanley 1995, Boulton 2003). Some research concerning freshwater macroinvertebrate communities has been conducted in Louisiana (Stewart et al. 1976, Sloey 1992, DeWalt 1995, Drury and Kelso 2000, Alley 2004, Kaller and Kelso 2006, 2007). However, none of these studies investigated spatial and temporal variation of macroinvertebrates at a watershed scale and under varied flow conditions. In general, there is a knowledge gap about aquatic macroinvertebrate assemblages, their spatiotemporal variation, and their relationship with environmental conditions such as stream permanence and water quality in low-gradient, humid, subtropical headwaters.

Streams within the central SCPE have been described as sluggish streams inhabited by low DO-tolerant fauna (DeWalt 1995). In a study on stream DO conditions in this region, Ice and Sugden (2003) found that 81% of the streams sampled in northern Louisiana during the summer were below the national standard (< 5 mg L^{-1}). Based on a 1-year intensive monitoring of DO in the Flat Creek watershed, Mason et al. (2007) recently reported that all headwater streams in the watershed showed DO levels below 4 mg L^{-1} during much of the year (March – November). Most of these streams had organic substrates with low, intermittent, or no flow during summer. These unique systems have made water quality compliance difficult for regulatory agencies, because it is currently unclear whether or not these conditions are wholly natural or have an anthropogenic component.

This study aims to characterize the aquatic macroinvertebrate community of headwater streams, identify spatial and seasonal patterns within the macroinvertebrate community, and

examine the association between environmental variables and the macroinvertebrate community in the Flat Creek watershed. The study is part of an interdisciplinary project that addresses headwater stream hydrology, chemistry, and aquatic ecology. This work focuses on the latter aspect, while research focused on the first two components are being completed by Saksa (2007) and BryantMason (in progress). Results from their work have provided the opportunity to systematically investigate the macroinvertebrate communities of these headwater streams with supporting physicochemical data.

LITERATURE REVIEW

Biological Indicators of Stream Health

After the Industrial Revolution in the early 19th century, human health concerns prompted the development of biological monitoring. Kolkwitz and Marsson (1909) recognized the use of aquatic biota as indicators of water pollution in lotic systems in Europe, and subsequently developed the Saprobien system. This paramount research gave a detailed outline of flora and fauna associated with distinct pollution gradients, and initiated the use of aquatic organisms as indicators of stream health conditions. The century-old system is still used today in Europe (Cairns and Pratt 1993), and has led to the development of biomonitoring tools used in North America. In the 1950's, aquatic ecology research in the United States began to focus on protecting streams from sewage and industrial wastes, and again certain aquatic biota were found to be associated with different pollution gradients (Gaufin and Tarzwell 1952, Patrick 1953, Gaufin and Tarzwell 1956, Bartsch and Ingram 1959). The usefulness of aquatic organisms as indicators of water quality and environmental conditions has become more widely recognized.

Later on, lists of aquatic species that were tolerant of pollution had been developed by many researchers. Gaufin and Tarzwell (1952) described the use of aquatic insects as indicators of stream health and discussed pollution tolerant and intolerant species, as well as the use of relative abundance as opposed to just presence/absence of a species. The main objective of their research was to identify indicators of water quality and to achieve a better definition of indicator organisms. Additionally, they stressed the importance of using biological indicators over water chemistry. However, a discrepancy existed as to which organisms were absolutely indicative of pollution (Gaufin and Tarzwell 1956). This led to another shift in the research from an individual focus to a holistic assessment of the macroinvertebrate community (Goodnight 1973).

When water quality criteria are solely based on chemical and physical conditions of a stream, many limitations exist. The most apparent weakness of a chemistry-based water quality survey is that the data typically reflects the stream condition at a particular point-in-time and therefore may not be indicative of the long-term condition (Wilhm and Dorris 1968). Additionally problems exist with establishing chemical criteria based on toxicity to aquatic organisms because of the potential interactions of numerous toxic compounds and aquatic species. A numerically-based criteria founded on community diversity was proposed by Wilhm and Dorris (1968). Their research assessed the receiving waters of treated sewage effluent and reinforced that the abundance and diversity of aquatic organisms are related to a stream's physicochemical composition and are indicative of stream health. Wilhm and Dorris's community diversity indices took into account the number of individuals and the number of species, whereas other metrics only accounted for the presence and absence of certain species. This index for assessing water quality has been implemented and developed over the years (e.g., Fore et al. 1996).

The passage of the Clean Water Act in 1972 has directed national water quality policy towards the goal of restoring and maintaining the chemical, physical and biological integrity of the nation's waters. Biological integrity has been defined as " the ability of an aquatic ecosystem, to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitats of a region" (Karr and Dudley 1981). Although the term has resulted in much debate as to how to measure such integrity, it has built a general consensus for integrating biological assessment in water quality monitoring programs. In an attempt to maintain biological integrity of water bodies, it became necessary to devise methods to measure community diversity and

function. The first attempt to use biota as indicators of biological integrity was by Hocutt (1981) using fish. This was followed by Hughes et al. (1982) and resulted in a one year project, the Ohio Stream Regionalization Project, which utilized fish and macroinvertebrates to assess aquatic ecosystem health.

Macroinvertebrates as Indicators of Water Quality

Ecological function of freshwater systems is closely linked to the structure of the biological communities they support (Cummins 1974). Within these communities, benthic macroinvertebrates fill lower and intermediate trophic levels and perform important ecosystem roles (Wallace and Webster 1996). Many aquatic macroinvertebrates are functionally adapted to utilize resources associated with the bottom of channels (Cummins 1973, 1974). Local stream bed sediments, flow regimes (e.g., velocity, discharge, flood, duration, etc.), and water quality characteristics (e.g., nutrients, temperature, organic matter, etc.) are important habitat components for macroinvertebrates. During the past two decades, both quantitative and qualitative biological assessments intended for aquatic macroinvertebrates have emerged (Lenat 1988, Karr 1991, Lenat 1993, Norris and Thoms 1999).

Using macroinvertebrates as indicators of water quality is often preferred over other organisms (e.g., fish, periphyton) in biological assessments, one of the major reasons is that macroinvertebrates are ubiquitous and inhabit all aquatic systems, ensuring that investigators will find these organisms. Semi-aquatic and aquatic insects are a major source of food for fishes, insectivorous birds, bats, riparian vertebrates, and terrestrial arthropods (Henschel et al. 2001, Seidman and Zabel 2001, Kato et al. 2003, Fukui et al. 2006). Such functional versatility and ecological importance is a result of a multitude of adaptations that have evolved in benthic

invertebrates in response to the complex, spatially and temporally dynamic nature of running water habitats (Heino et al. 2005).

Also, when compared to fishes (i.e., long lived) and periphyton (i.e., short lived), macroinvertebrate life spans may be an ideal range. They live long enough to detect changes in water quality over time but also have a short enough life span to produce multiple generations for long-term investigations. Unlike highly mobile fish, macroinvertebrates typically stay in the same area and are less likely to move away from polluted areas. Therefore, information on factors affecting benthic macroinvertebrates is not only vital for basic ecological understanding, but also serves as a reference for monitoring, restoring and maintaining the quality of stream ecosystems (Rosenberg and Resh 1993, Palmer et al. 1997).

Furthermore, identification of benthic macroinvertebrates is relatively simple and straightforward when compared to other organisms, such as periphyton. Though macroinvertebrate sampling and laboratory identification can be labor intensive, methods have been developed to make the procedures more time-efficient and cost-effective. For instance, the U.S. EPA's Rapid Bioassessment Protocol (Barbour et al. 1999) was developed in the 1980's and refined during the 1990's to provide cost-effective and scientifically valid procedures for biological field surveys in streams and wadeable rivers. Additionally, benthic macroinvertebrates are commonly used in bioassessments because of an increasing amount of research describing the relationships between certain groups of macroinvertebrates and physicochemical water quality parameters, providing useful information at the individual and community level.

Louisiana has not developed criteria nor implemented a biological monitoring program, though many other states have utilized benthic macroinvertebrates to determine stream health (USEPA 2002, A. Hendricks at Louisiana Department of Environmental Quality, personal

communication, August 23, 2006). However, work to develop biological criteria for fish is currently being undertaken by researchers at Louisiana State University (LDEQ 2006). Because macroinvertebrates can be indicative of water quality, development and implementation of a biological monitoring program with macroinvertebrates would also be useful in determining the health of Louisiana's unique waterways. The first step towards this goal is to determine the relationships of macroinvertebrate structure with stream environmental characteristics and water quality parameters.

There are some disadvantages of using macroinvertebrates as biological indicators of water quality. Some benthic macroinvertebrates may not respond to changes in water quality, such as increases in nutrients. Collection of samples at different times of year would also pose a problem because many benthic organisms inhabit the stream in various seasons. Additionally, some macroinvertebrates drift to different waters which may also be problematic.

Importance of Headwater Streams for Macroinvertebrates

The beginning reaches of a river are the headwaters, which comprise a network of 1^{st} to 2^{rd} order streams with perennial or ephemeral flow regimes and small drainage areas. Across the landscape, headwater streams drain over half the area of a typical drainage network (Horton 1945) and are known as the ultimate sources of water, nutrients and sediments to downstream fluvial networks. Therefore, headwater streams provide unique habitat and play an important role in structuring the biological communities in the larger downstream reaches (Vannote et al. 1980, Meyer et al. 2007). The River Continuum Concept suggests that physical and chemical processes in headwaters structure biological communities downstream (Vannote et al. 1980), which is based on a concept of dynamic equilibrium. At a large spatial scale from upstream to downstream, wetted width increases and the riparian canopy changes from closed to open. A

closed canopy leads to reduced light availability, limiting primary productivity, and promoting a macroinvertebrate community designed to take advantage of allochthonous inputs of detritus (i.e., shredders, collector-gatherers). Research on macroinvertebrates at the community level has shown that functional feeding groups reflect stream size (Heino et al. 2005), and that species richness increases with stream order (Paller et al. 2006), which is in agreement with the River Continuum Concept (Vannote et al. 1980). Headwaters are a major source of detrital inputs, providing food for downstream communities, and are clearly important to overall stream health, warranting investigation of the role of aquatic benthic macroinvertebrates (Wipfli et al. 2007).

Research has indicated that headwater and intermittent streams provide critical habitat for many aquatic insects, reflected by the high diversity of macroinvertebrates found in many headwater streams throughout the United States (Meyer et al. 2007). For instance, in North Carolina headwaters not shown on standard topographical maps, 51 different families of macroinvertebrates were documented (Meyer et al. 2007). Intermittent streams (i.e., dry in summer) in Oregon were found to have over 200 aquatic and semi-aquatic insects species (Meyer et al. 2007). Undoubtedly, headwater streams and intermittent streams are important in providing food and habitat for aquatic insects.

Very few studies have been conducted in headwater streams of Louisiana on macroinvertebrates. In north-central Louisiana, Morse and Barr (1990) found 43 species of trichopterans comprising 5 endemics. In western Louisiana, Williams et al. (2005) observed similar assemblages among drainages in the headwater streams, which included 70 different families of macroinvertebrates. These two studies indicate the importance of headwater streams for macroinvertebrates, because these stream systems provide habitat for a diverse population of aquatic macroinvertebrates as well as for native species. Due to differences in channel

geomorphology and hydrologic regimes between these two areas, a direct comparison of findings is not possible. However, these results seem to be comparable with headwater streams from other southeastern states. A study conducted in first-order streams of Alabama found that intermittent and permanent streams have similar benthic fauna (Feminella 1996). Whereas in Florida headwater streams, invertebrate species composition, diversity, and density differed, which was attributed to seasonal variations and drought conditions (Cowell et al. 2004). Likewise, in the west Louisiana study, Williams et al. (2005) also found seasonality to affect the structure of macroinvertebrate assemblages.

Research in other regions of the country, such as the Pacific Northwest, appears to produce similar results. For instance, a study in Oregon found differences in headwater macroinvertebrate assemblages (Herlihy et al. 2005) and in Washington, Haggerty et al. (2002) found low taxa richness and macroinvertebrate densities. Cole et al. (2003) reported that headwater streams in a coastal mountain range in Oregon supported a rich community of taxa as well as taxa endemic to these areas. While headwaters do provide important habitat for macroinvertebrate communities, as documented in the southeast, there is considerable variation in assemblage size throughout the United States. This variation may be attributed to differences in various headwater conditions, sampling techniques, and study objectives.

Spatial and Temporal Dynamics of Macroinvertebrates

Aquatic macroinvertebrates are found throughout most water bodies in the world during most times of the year. The spatial range for macroinvertebrates varies from specific habitat locations, such as substrate, to an entire watershed (Minshall 1988). Their temporal scales range from seconds to years, but most research typically ranges from days to multiple years (Minshall 1988). Extensive research has been conducted on exploration of spatial (e.g., Ramirez et al.

2006, Hansen and Closs 2007, Johnson et al. 2007, Martel et al. 2007) and temporal (e.g., Beche et al. 2006, Ramirez et al. 2006, Sporka et al. 2006, Kratzer and Batzer 2007) influences on aquatic macroinvertebrates.

Studies on spatial variability of macroinvertebrates have been conducted at numerous scales from basin-wide to microhabitat. Some studies have highlighted the importance of catchment scale variables (Allan and Johnson 1997, Townsend et al. 2003), while others have focused on reach (Richards et al. 1997, Roy et al. 2003) or microhabitat (Brosse et al. 2003, Hansen and Closs 2007) level variables. Ramirez et al. (2006) examined spatial patterns among physicochemical and aquatic insects within a basin. They found physicochemical characteristics differed among streams, but could not detect differences among insect assemblages. Conversely, Hansen and Closs (2007) studied long-term patterns of macroinvertebrate drift over a small spatial scale (i.e., riffles). Their research found differences in drift density, but not taxonomic diversity, and concluded drift density consistently varied between riffles. Studies conducted at multiple spatial scales have found conflicting results. Martel et al. (2007) noted species composition changed along increasing spatial gradients, whereas Johnson et al. (2007) found similar responses of invertebrate assemblages at different spatial scales, ranging from local to regional. Environmental variables across multiple spatial scales may be interdependent, resulting in complex linkages between stream biota and their environment.

Macroinvertebrates may be influenced by seasonal patterns, which will likely influence metrics used in biological monitoring. In tropical lowland streams, Ramirez et al. (2006) observed that seasonal patterns in rainfall influenced insect assemblages and stream physicochemistry, with discharge and pH most affected. Conversely, in a blackwater swamp in Georgia, researchers did not find temporal variations in macroinvertebrate communities (Kratzer

and Batzer 2007). The different outcomes from these two studies are likely due to the former being conducted in streams and the latter in a homogenous swamp system. In streams with irregular flow, such as European metarhithral mountain streams, metrics differed significantly between months (Sporka et al. 2006) and in Mediterranean-climate streams, abundance and taxonomic composition were highly seasonal (Beche et al. 2006).

Spatial variations at a small scale (i.e., lower-order watershed) are inherently connected to seasonal changes in precipitation and temperature. The headwaters within a lower-order watershed often reflect seasonal influences as a change in flow permanence. For instance, during the wet season small headwater streams may be continually flowing but as precipitation tapers off, stream channels become intermittent or completely dry. Seasonally controlled hydrological processes influence macroinvertebrate assemblages under spatially variable conditions.

Association between Macroinvertebrates and DO

DO concentrations of headwater streams can be a result of stream conditions including temperature, flow, and nutrient levels in the streams. The first directly controls dissolubility of oxygen in waters, while the latter two indirectly affect DO levels through stimulating supply and consumption of oxygen in streams. Therefore, the amount of DO in low-gradient, eutrophic water bodies may change largely over a 24-hour period due to photosynthesis and respiration processes (Dodds 2002).

Macroinvertebrates respond to shifting water chemistry by drifting to more optimal locations (Connolly et al. 2004). However, if conditions persist for a long period of time (e.g., weeks or months) then mortality may occur. Macroinvertebrates depend on DO for cellular respiration, and many organisms have developed respiratory mechanisms capable of obtaining DO at varying concentrations (Eriksen et al. 1996). The evolution of tolerances of individual

macroinvertebrates has led to a classification scheme of an organisms' level of tolerance (i.e., "tolerant" or "intolerant" of low DO; or "facultative" being capable of living in either setting). Tolerance values for aquatic macroinvertebrates for streams with organic enrichment was pioneered by Hilsenhoff (1987) and has been revised for southern stream systems (Lenat 1993).

Knowledge of differing macroinvertebrate morphology is important in explaining why certain organisms can thrive at particular DO levels. For instance, Ephemeroptera have closed tracheal systems with external gills and are typically associated with higher levels of DO (Eriksen et al. 1996), whereas taxa adapted for low oxygenated waters, including dipterans, coleopterans, or hemipterans have open respiratory systems or respiratory pigment utilizing mechanisms such as air stores, hemoglobin, or atmospheric breathing (Eriksen et al. 1996). Observations of these distinct morphological characteristics have helped structure the use of indicator organisms for water quality.

Several studies have been conducted on the relationships between aquatic macroinvertebrates and DO levels in the southern United States (Davis et al. 2003, Cowell et al. 2004, Bednarek and Hart 2005, Kaller and Kelso 2007) and in other geographical regions (Parr and Mason 2003, Chapman et al. 2004, Connolly et al. 2004, Ndaruga et al. 2004). A study of coastal plain streams in Georgia (Davis et al. 2003) found an increase in tolerant taxa during naturally stressed summer conditions of intermittent and no-flow streams. Summer stream conditions in this region typically showed low DO, and macroinvertebrate taxa adapted to these conditions. In a study of Florida streams, Cowell et al. (2004) reported seasonal differences in meiofauna with higher densities in late summer and winter, and lower densities in spring and summer during which low DO was prevalent. Additionally, they found the reclaimed streams to have low DO with low species richness and diversity, than impacted streams. These studies

employed different analytical approaches, whereby the former distinguished indicators based on differences in DO levels at an individual level (i.e., tolerant and intolerant) and the latter attempted to use community characteristics (i.e., diversity, richness, and density) to determine differences between sites. It should be noted that these two studies did not seek to determine relationships between DO and macroinvertebrates exclusively; rather these were simply part of the observed results of an overall analysis.

Research by Kaller and Kelso (2007) sought to determine relationships between macroinvertebrate communities and DO in some western Louisiana streams. Results of their study found higher total abundance and higher taxa richness in low DO stream sections, while Shannon-Wiener diversity was reported to be higher in the stream with higher DO. Results from a study on the response of macroinvertebrate communities after dam improvement in Tennessee by Bednarek and Hart (2005) found an increase in total abundance under low DO conditions which is in agreement with Kaller and Kelso (2007). However, under higher DO conditions, the Tennessee researchers found an increase in taxa richness, which differs from the Louisiana study. The opposing responses of total abundance under differing DO levels in these two studies suggests that this metric may not be appropriate for oxygen depleted systems. Bednarek and Hart (2005) noted the importance of understanding biological interactions of the different taxa. They attributed the increase of total abundance during low DO conditions to the availability of suitable habitat for the low-biomass chironomids, while the increase in DO led to favorable conditions for larger-bodied, intolerant organisms such as ephermeropterans, plecopterans, and trichopterans.

Studies on the relationship between DO and benthic macroinvertebrates have been conducted in many other geographical regions (Parr and Mason 2003, Chapman et al. 2004,

Connolly et al. 2004, Ndaruga et al. 2004), as well as under both field and experimental conditions. In a study on eutrophic lowland rivers in England, Parr and Mason (2003) observed an increase in low DO tolerant organisms in ponded areas. Overall the researchers found that macroinvertebrate assemblages were negatively influenced by drought conditions (i.e., low flow). Research on the impact of water quality to macroinvertebrate communities in Kenyan tropical streams found an increase in invertebrate densities with decreasing DO concentrations (Ndaruga et al. 2004). In a study on the relationship between DO concentration and the abundance of macroinvertebrate respiratory groups in swamp-river systems in Uganda, Chapman et al. (2004) reported that atmospheric breathers (e.g., dipterans, hemipterans, gastropods) were negatively correlated with DO while tracheal gill breathers (e.g., ephemeropterans, plecopterans, and certain coleopteran larvae) showed a positive. These researchers suggested that DO concentrations were predictive of abundance of dominant respiratory modes.

Connolly et al. (2004) conducted a study of tropical streams in Australia specifically aimed to identify the hypoxia tolerance of freshwater macroinvertebrates with artificial mesocosms. The researchers used assemblages from both upland and lowland regions to identify DO tolerance (by survival), drift, and emergence. Results from their study showed no significant differences between the two assemblages. Additionally, they found that macroinvertebrates were tolerant of hypoxic (25-35% DO saturation) conditions but intolerant of anoxic (10-20% DO saturation) conditions. An increase in drift for all taxa under the anoxic treatment was observed, implying that assemblages may not be solely determined by DO concentrations but perhaps by changing conditions.

Overall, the studies introduced above demonstrate that DO concentrations can influence macroinvertebrate assemblages, and that total abundance may respond differently under varying

DO concentrations. The literature also infers that geomorphological processes such as velocity and available habitat (i.e., stream wetted area) are indicative of macroinvertebrate community structure. However, most these studies have been conducted in perennial streams. Little is known about aquatic macroinvertebrate structure, their spatiotemporal variation and the relationships with environmental conditions such as stream permanence and water quality in low-gradient, humid subtropical headwaters. Headwater streams have been gaining more attention as their role in structuring downstream reaches is realized, and macroinvertebrates have been recognized as an important component in these systems. Such information is needed to guide resources managers and forest practitioners in decision making for headwater protection.

METHODS

Study Area

The Flat Creek watershed covers 365 km^2 of the northern portion of the South Central Plains Ecoregion (SCPE), located in Winn Parish, central Louisiana (Figure 3.1). Average annual climatic data was obtained from the National Climatic Data Center (NCDC) Winnfield 2W station, south of the study area (NCDC 2002). From 1971-2000, the average annual temperature in the area was 17.9 ºC, ranging from 7.2 ºC in January to 27.5 ºC in July. Long term average annual precipitation was 1508 mm with a low of 91 mm and a high of 158 mm. During this 17-month study, temperature and precipitation was measured locally with a HOBO weather station (4 Channel MicroStation, #H21-002; Onset Computer Corp., Bourne, MA). The average temperature was 16 ºC ranging from 7 ºC in January 2007 to 28 ºC in August 2006 (Figure 3.2). Total precipitation was 1632 mm with monthly totals ranging from 24 mm in June 2006 to 312 mm in October 2006 (Figure 3.3).

The watershed is predominately forested and managed for wood production with a secondary land use for livestock grazing (Figure 3.4). Upland forests are typically comprised of loblolly pine (*Pinus taeda*), whereas hardwoods such as magnolia (*Magnolia grandafolia*), sweet gum (*Liquidambar styriciflua*), and bald cypress (*Taxodium distichum*) are more common in riparian zones. Composition of the soil is mainly hydrous over loamy, covering more than three quarters of the entire area, with clayey soil comprising the remainder of the watershed, predominantly in the riparian zones. Streams in the northern portion of the SCPE differ from southern SCPE streams in their physicochemistry and aquatic macroinvertebrate communities (DeWalt 1995). Historically, streams in central Louisiana have been described as ranging from sand and silt bearing creeks to sluggish or still water bayous (Viosca 1933). These particular

reaches have been described as sluggish streams inhabited by low DO-tolerant fauna (DeWalt 1995), and it has been observed that stoneflies (order: Plecoptera) are absent (Stewart et al. 1976). Many of the streams are hypoxic (< 3 mg L^{-1}) for part of the year, which may be considered the natural condition of these systems due to their low flow and high organic matter load (Ice and Sugden 2003).

Figure 3.1. Location of the Flat Creek watershed in central Louisiana, USA.

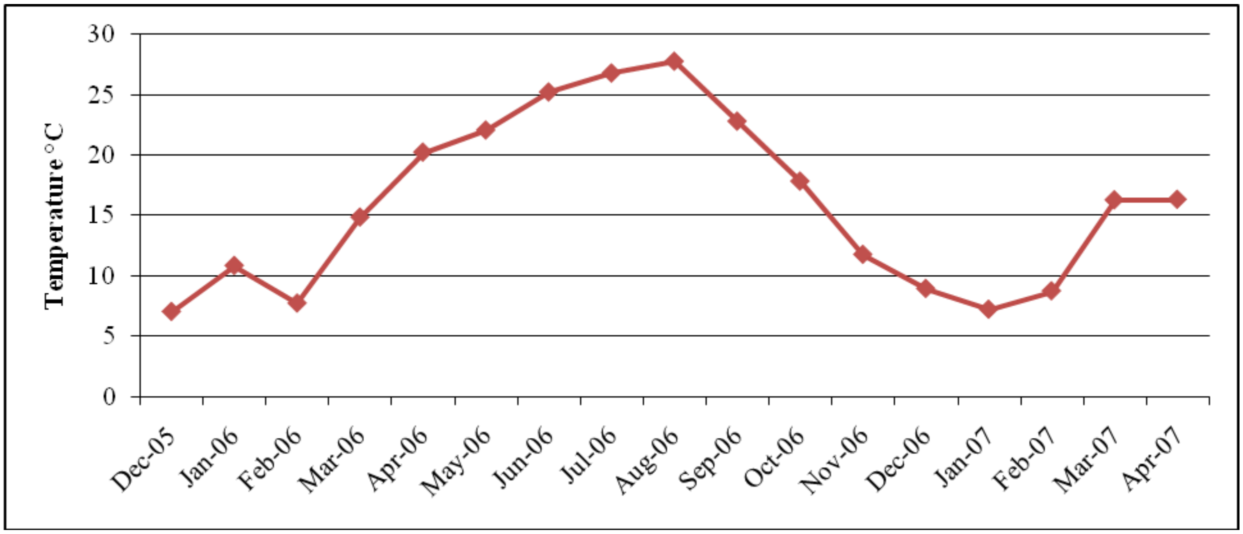

Figure 3.2. Average monthly temperature over 17 months obtained from the HOBO weather station located in the Flat Creek watershed of central Louisiana.

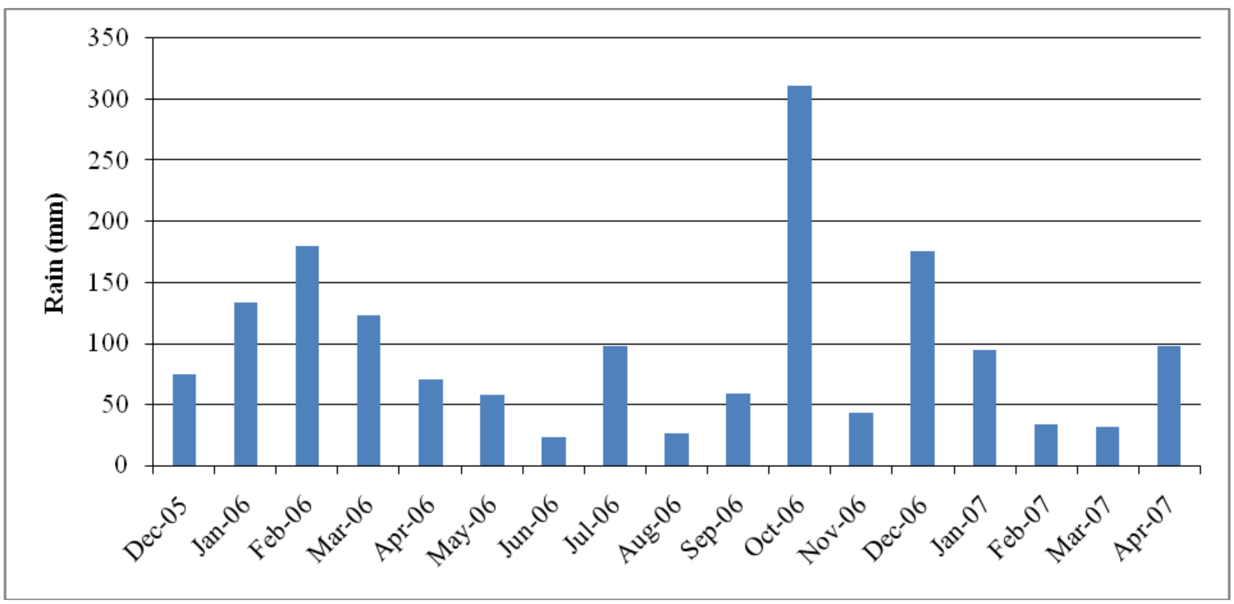

Figure 3.3. Monthly precipitation over 17 months obtained from the HOBO weather station located in the Flat Creek watershed of central Louisiana.

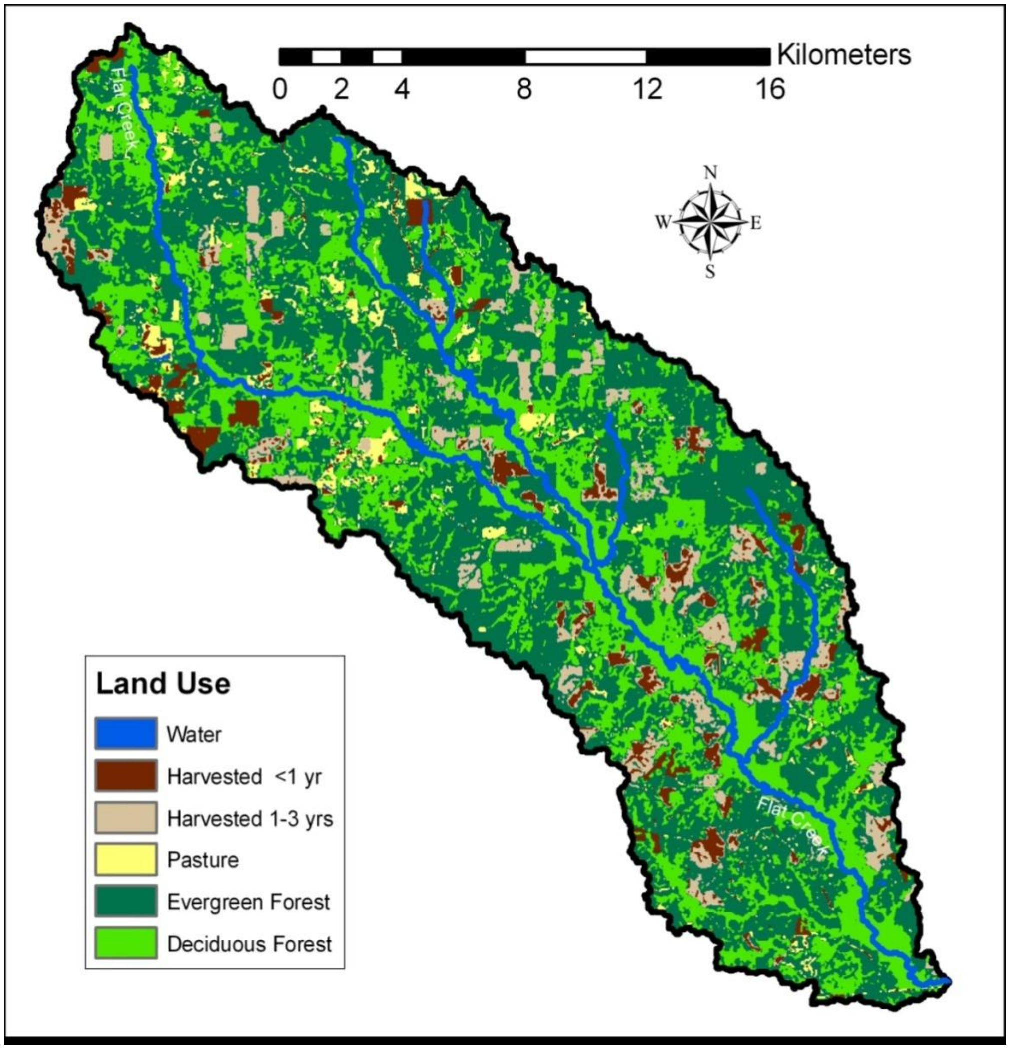

Figure 3.4. Land-use composition of the Flat Creek watershed in central Louisiana.

Study Design and Monitoring Locations

All monitoring sites for this study were located on 1st and 2nd order streams with variable drainage areas throughout the Flat Creek watershed (Figure 3.5 and Table 3.1). Five of the 11 monitoring locations became intermittent during the summer months and two were completely dry. Two monitoring sites (I1, I2) were located on Spring Creek, a tributary of Turkey Creek. Four sites (I3 – I6) were located on Turkey Creek upstream of Spring Creek, and one site (E2) below. One site (E1) was located along Flat Creek, upstream from its junction with Turkey

Creek. One site (E3) was located along Fish Creek, and two sites were along Big Creek, with both creeks draining into Flat Creek (Figure 3.5). This selection of sampling sites was based on a paired watershed approach with additional monitoring locations to assess the watershed-scale effect.

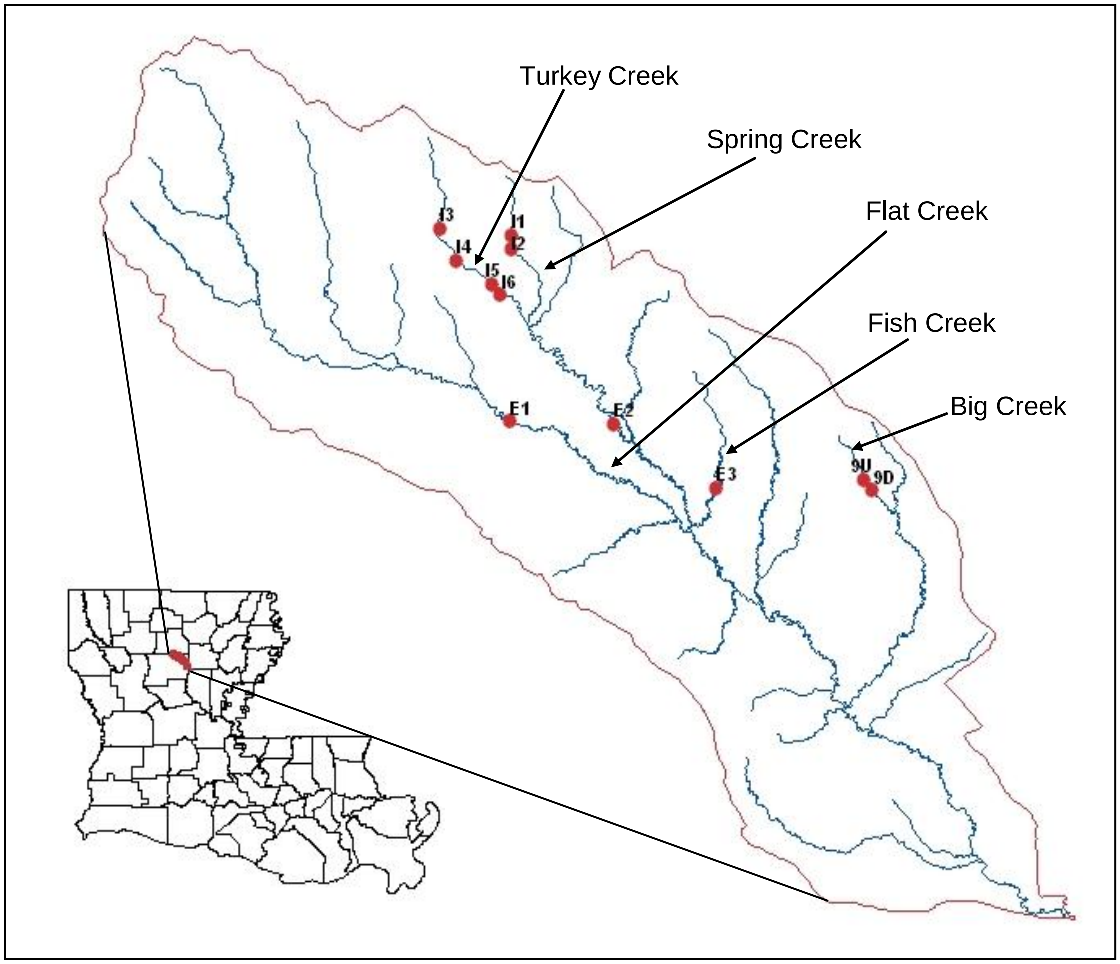

Figure 3.5. Location of macroinvertebrate sampling sites in the Flat Creek watershed in central Louisiana.

Table 3.1. Location and physical characteristics of monitoring sites in Flat Creek watershed.

Site	Latitude	Longitude	Elevation above MSL (m)	Drainage Area (km^2)	Stream Order
I1	32º 04’ 51”	92º 27’ 38”	54.4	3.0	1
I2	32º 04’ 50”	92º 25’ 34”	50.9	3.6	1
I3	32º 03’ 35”	92º 23’ 35”	54.1	12.4	1
I4	32º 00’ 56”	92º 21’ 58”	50.8	14.3	1
I5	32º 00’ 30”	92º 20’ 12”	48.3	17.8	1
I6	32º 08’ 21”	92º 27’ 36”	47.4	18.3	1
E1	32º 08’ 06”	92º 27’ 36”	38.6	109.6	2
E2	32º 08’ 30”	92º 29’ 01”	38.9	45.1	2
E3	32º 03’ 35”	92º 23’ 35”	37.2	6.1	1
9U	32º 03’ 56”	92º 20’ 46”	50.3	2.1	1
9D	32º 03’ 39”	92º 20’ 34”	52.0	3.4	1

Sites I1 and I2 were located on Spring Creek east of Parish Line Road. Site I1 was located approximately 0.5-km upstream of I2 and the two sites differed considerably. I1 had high, steep banks and a narrow channel with shallow base flow, while I2 had less incision and a wider U-shaped channel with deeper base flow. Average wetted width at I1 and I2 was 2.2 m and 4.1 m ranging from 1 - 4 m and 3 -10 m respectively, in spring and late summer.

Sites I3 and I4 were located on the upper portion of Turkey Creek approximately 1.2 km apart and west of Parish Line Road. The banks at both sites had a gentle slope and a channel size similar to I2. Average wetted width at sites I3 and I4 was 3.8 m and 4.7 m, and ranged from 1 - 8 m and 2.5 - 8 m, respectively. Turkey Creek had moderate beaver activity with higher activity apparent on the lower reaches as indicated by beaver dams, deeper channels, and slower velocities.

Sites I5 and I6 were also located on Turkey Creek approximately 0.5 km apart downstream of LA Hwy 499 site. I5 was located near the downstream side of the highway bridge. The channel in the reach was deeper and wider with steeper banks compared to sites I3 and I4 located upstream. The wetted width ranged from 6 - 8 m with an average of 6.5 m at both locations during the spring and summer.

Site E1 was located on Flat Creek above the confluence with Turkey Creek, and upstream of the LA Hwy 126 bridge. The wetted width ranged from 1 - 11 m with an average width of 6.3 m. Cypress trees and associated root structures were found in sections of this reach.

Site E2 was located on Turkey Creek approximately 5.6 km down stream of site I6 upstream of the LA Hwy 126 bridge. This site differed considerably from the sites located upstream, with a wetted width that ranged from approximately 1 - 5.5 m with an average of 3.5 m in spring and summer. The depth at base flow was typically low compared to upstream sites.

Site E3 was located on Fish Creek (also known as Spring Creek) upstream of the intersection of Lonehill Church Road and Dulany Road. This stream was narrower than the others mentioned above, with an average wetted width of 3.1 m ranging from 2 - 5 m. There was very little visual stream flow, banks were steep with moderate undercutting, and leaf packs dominated the substrate, with fine sediments found underneath.

At all locations the substrate was comprised of leaf packs and woody debris with silt or sand. Stream velocity was typically low at all sites except I1 and E2, where it tended to exceed 2 cm s^{-1}. All sites were well shaded by riparian vegetation except I3, which had less cover than other sites (Table 3.2). The physical attributes such as flow permanence, wetted width, and wetted area of sites changed from spring to late summer (Table 3.2). At most sites in spring, the

sampling reach was entirely wet, whereas in late summer many sites became intermittent or dry (e.g., Figures 3.6 and 3.7).

Table 3.2 Flow permanence and physical characteristics of monitoring sites taken in April and August 2006 in Flat Creek watershed.

	Flow Permanence		Mean Wetted Width (m)		Mean Wetted Area (m^2)		Mean Vegetation Cover (%)	
Site	**Spring**	**Late Summer**	**Spring**	**Late Summer**	**Spring**	**Late Summer**	**Spring**	**Late Summer**
9U	Intermittent	Dry	1.86	na	1.40	na	83	na
9D	Intermittent	Dry	1.01	na	0.52	na	94	na
E1	Continual	Intermittent	8.81	2.60	27.09	6.13	68	87
E2	Continual	Intermittent	4.31	1.87	6.55	2.08	80	88
E3	Continual	Intermittent	3.16	1.74	3.70	7.43	74	96
I1	Continual	Intermittent	2.45	1.45	2.23	0.94	72	96
I2	Continual	Continual	4.60	3.66	9.08	7.40	70	94
I3	Continual	Intermittent	4.59	2.26	8.07	6.09	52	51
I4	Continual	Continual	5.62	3.27	11.61	12.20	73	89
I5	Continual	Continual	7.49	6.03	19.84	16.07	81	84
I6	Continual	Continual	6.45	6.39	14.61	19.62	71	70

Figure 3.6. Channel with continual flow at site E2 in early spring in the Flat Creek watershed, central Louisiana.

Figure 3.7. Intermittent channel at site E2 in late summer in the Flat Creek watershed, central Louisiana.

Macroinvertebrate Sampling

Benthic macroinvertebrate samples and habitat characteristics were collected at each of the 11 monitoring locations in April and August 2006. Eight transects were selected for each monitoring site with a random number generator in MS Excel for a 160-m reach. Each transect (sampling location) was at least 6 meters apart. Transects sampled in April were not sampled in August.

Because of low current velocities (average < 6 cm s^{-1}) and substrates that were comprised mostly of small woody debris and leaf packs, it was necessary to design an apparatus that could effectively gather a representative benthic sample. A rectangular core sampler (0.25 m by 0.55 m) with an open top and bottom was made from 0.3 cm thick aluminum (Figure 3.8), with a total sampling area of 0.1375 m^2.

Figure 3.8. Side view of modified core sampler used for macroinvertebrate sampling of low-gradient streams in Flat Creek watershed.

Eight benthic samples were collected at each monitoring site along a 160-m reach. The samples were collected by driving the core sampler at least 10 cm into the substrate and removing the top 2.5 cm (Figure 3.9). After substrate removal, the area within the sampler was swept with an aquarium net for 1 minute. The entire sample was then preserved in 95% ethanol, and transported to the laboratory at Louisiana State University for sorting, identification and enumeration. A total of 139 samples were collected in April and August of 2006. Eighty-four samples were collected from 11 sites in the spring and 55 samples were collected from 9 sites in late summer. Sites 9D and 9U were not sampled in late summer because the stream bed was entirely dry.

Figure 3.9. Collection of benthic macroinvertebrate sample using the modified core sampler in the Flat Creek watershed. Substrate was removed prior to netting within the apparatus.

Taxonomic Identification

Samples were dyed with Rose Bengal to aid in separating organisms from detritus. To rid samples of sand and silt particles while retaining small organisms, samples were rinsed with tap water in a 500-µm sieve (No. 35) prior to sorting. Once sorted, specimens were identified to lowest possible taxon, usually family but occasionally genus. Specimens were identified with the aid of a stereo microscope (Wild Leitz M3 microscope; Wild Heerbrugg Ltd, Heerbrugg, Switzerland) and several guides and reference keys (Merritt and Cummins 1996, Smith 2001, Thorp and Covich 2001, Voshell 2002). All identified organisms were preserved in a 10 ml glass vial with 95% ethanol.

Water Quality and Habitat Measurements

Environmental variables, such as water quality and habitat characteristics, were collected at each site. *In situ* water samples were taken with a YSI 556 multiprobe (YSI Inc., Yellow Springs, OH) in April and August of 2006 at each site. Measured water quality parameters included temperature (°C), DO (as % saturation and in mg L^{-1}), specific conductance (µS), and pH. The DO probe was not working in April 2006. The average DO values were calculated with March and May 2006 DO values and reasonably reflect the expected values for April. Table 3.3 contains the average DO values calculated for April 2006. Grab samples were taken and analyzed for nitrate, nitrite, total phosphorus, total suspended solids, and total solids at the Department of Agricultural Chemistry at Louisiana State University AgCenter. Total dissolved solids (TDS) was calculated as the difference between total solids (TS) and total suspended solids (TSS).

Various physical habitat measurements were taken during each sampling event. Wetted width and depth were recorded and velocity measurements were taken with a SonTek

FlowTracker (SonTek/YSI, Inc., San Diego, CA) at each sampling location. Wetted area and width to depth ratio were calculated for each transect, and with each depth measurement, substrate type was recorded as one of the following: leaf pack, woody debris, silt, sand, or other (e.g., roots, thick hydrophilic vegetation). Instream cover, and bank measurements were determined according to methods described by Lazorchak et al. (1998).

Table 3.3 summarizes the survey results for fourteen physical habitat and water quality variables used in a canonical correlation analysis. These physical habitat parameters include the average values of a reach for wetted width, wetted area, velocity, and stream cover. The water quality parameters include pH, DO, temperature, conductivity, TSS, TDS, TS, total phosphorus, nitrate, and nitrite.

Table 3.3. Average physical habitat and water quality values which comprised the environmental variables used in canonical correlation analysis. Number on top is from spring and number below is from late summer.

	Sites										
Environmental Variable	9D	9U	E1	E2	E3	I1	I2	I3	I4	I5	I6
Wetted Width	1.86	1.01	8.81	4.31	3.16	2.45	4.60	4.59	5.62	7.49	6.45
	-	-	2.60	1.87	1.74	1.45	3.66	2.26	3.27	6.03	6.39
Wetted Area	1.40	0.52	27.09	6.55	3.70	2.23	9.08	8.07	11.61	19.84	14.61
	-	-	6.13	2.08	7.43	0.94	7.40	6.09	12.20	16.07	19.62
Velocity	0.14	0.29	4.89	4.07	0.24	2.06	0.27	0.79	-0.04	0.48	0.05
	-	-	-0.11	0.06	0.00	0.00	0.02	0.00	0.13	-0.13	-0.07
Vegetation Cover	82.74	94.48	67.85	79.79	73.91	71.90	70.23	52.39	72.99	80.53	71.14
	-	-	86.77	88.23	95.55	96.07	93.56	50.78	88.58	84.19	70.41
Temperature	15.63	-	19.49	18.33	16.56	17.96	19.20	18.61	17.71	17.32	17.36
	-	-	29.65	-	26.27	24.23	26.17	-	24.17	29.28	25.09
Conductivity	70	-	92	90	84	58	61	101	99	99	101
	-	-	142	-	125	76	90	-	138	123	115
DO	3.56	-	3.18	5.09	2.87	5.27	1.64	2.70	1.72	1.27	0.89
	-	-	6.46	-	3.79	2.06	0.81	-	1.20	4.06	0.55
pH	5.25	-	5.73	5.72	5.26	6.34	6.19	6.40	6.35	6.07	6.21
	-	-	6.62	-	6.52	5.55	5.34	-	5.71	6.16	6.19
TSS	35.90	-	28.70	25.35	22.90	25.50	23.40	23.50	28.30	57.10	22.40
	-	-	21.30	-	33.55	6.10	2.50	-	9.01	5.01	24.30
TS	156	-	138	123	134	108	118	139	137	191	155
	-	-	119	-	154.	113	130	-	126	115	150
TDS	120.10	-	109.30	97.65	111.10	82.50	94.60	115.50	108.70	133.90	132.60
	-	-	97.70	-	120.95	106.90	127.50	-	116.99	109.99	125.70
Total Phosphorus	0.06	-	0.09	0.06	0.03	0.02	0.06	0.06	0.07	0.25	0.06
	-	-	0.05	-	0.06	0.03	0.04	-	0.06	0.15	0.09
Nitrite	0.04	-	0.05	0.05	0.00	0.03	0.02	0.03	0.06	0.05	0.06
	-	-	0.10	-	0.04	0.07	0.08	-	0.09	0.02	0.05
Nitrate	0.11	-	0.11	0.35	0.11	0.11	0.45	0.11	0.11	0.11	0.40
	-	-	0.50	-	0.52	0.27	0.33	-	0.60	0.91	1.12

Biological Metrics

Metrics are measures used to numerically describe an assemblage of organisms either as individuals or as a community. For this study, sixteen metrics were calculated for each macroinvertebrate sample (Table 3.4), based on commonly accepted indexes and prevalent species observed in the watershed.

Total abundance was calculated as the total number of individuals per a sampling area of 0.1375 m^2 and measures overall variety of the assemblage. Taxa richness was calculated as the total number of taxa at the family level and gives an indication of the variety of organisms found in a stream. Shannon-Weiner's index of diversity was calculated because it is commonly used (Magurran 1988) and was calculated as:

$$H' = -\sum p_i \ln p_i$$

where p_i is the proportion of individuals found in the ith species and ln is natural logarithm.

Evenness for Shannon's index was calculated as:

$$E = H' / H_{max}$$

which ranges from 0 to 1, where a value of 1 would mean all species are equally abundant.

Other metrics that were used to describe macroinvertebrate communities were also based on a particular taxon or group of taxa. Percent dominant taxa (family level) was calculated as the proportion of the most abundant taxa to the total number of individuals. This metric is used to describe the dominance of the most abundant taxon. A common pollution-sensitive metric, percent EPT, was calculated as the proportion of taxa from orders Ephemeroptera, Plecoptera, and Trichoptera to total individuals. This metric is commonly used across the U.S. as an indicator of stream health because these taxa are found to be abundant in fast flowing waters with high DO levels. Percents Diptera, Chironomidae, Odonata, Gastropoda, Pelecypoda,

Crustacea, Amphipoda, and Isopoda were all calculated as the proportion of the specific taxa to total individuals. These eight metrics were chosen because researchers in other southeastern states have found them to be useful (Barbour et al. 1996, Davis et al. 2003).

Table 3.4. Benthic macroinvertebrate metrics calculated for each sample. Bold indicates metrics used in analysis of variance.

Metric	Description
Total Abundance	Total number of individual macroinvertebrates collected in a sample
Taxonomic Richness	Total number of individual taxa at the family level
Shannon-Weiner Index	Measure of Diversity
Evenness	Describes how equally abundant the species are
Percent Dominant Taxa	Percent of organisms in sample that is the single most abundant taxon
Percent EPT	Percent of individuals from the orders Ephemeroptera, Plecoptera, and Trichoptera
Percent Diptera	Percent of individuals from the order Diptera
Percent Chironomidae	Percent of individuals from the family Chironomidae
Percent Odonata	Percent of individuals from the order Odonata
Percent Gastropoda	Percent of individuals from the class Gastropoda
Percent Pelecypoda	Percent of individuals from the class Pelecypoda
Percent Crustacea	Percent of individuals from the phylum Crustacea
Percent Amphipoda	Percent of individuals from the order Amphipoda
Percent Isopoda	Percent of individuals from the order Isopoda

Statistical Analyses

Exploratory Analysis

In order to explore relationships between the environmental variables and the aquatic macroinvertebrate taxa, two multivariate datasets were constructed. The first data set included the common macroinvertebrates. Rare taxa were removed and included taxa that comprised less than one percent of the total individuals. Total individuals excluded Chironomidae, Ceratopogonidae, and Asellidae because they were extremely abundant. Twenty-five common taxa remained after removing rare taxa. Fourteen environmental characteristics, including water quality and physical habitat measurements, made up the second dataset. Canonical correlation analysis (PROC CANCORR, SAS, version 9.1, SAS Institute Inc., Cary, NC, USA) was used to explore relationships between biological and environmental variables (James and McCulloch 1990). Due to the large sample size (n=139), canonical variates were interpreted if ≥ 0.4 (Stevens 2001).

Overall Spatial and Seasonal Patterns

To determine spatial and seasonal patterns, a two-way ANOVA was used to test for significant differences among fifteen of the sixteen metrics between seasons and sites (PROC MIXED, SAS, version 9.1, SAS Institute Inc., Cary, NC, USA). Percent dominant taxon was excluded because of potential overlap with other metrics. Only nine of the eleven sites were included in the ANOVA because sites 9D and 9U were not sampled in late summer. Thirteen out of fifteen tested metrics were log(x+1) transformed (x= metric value) to improve normality. A critical value based on the Dunn-Sidak method (Sokal and Rohlf 1995) was calculated because more than one ANOVA was performed on the metrics, increasing the chance of making a Type I error. The value was determined with the equation:

$$\alpha' = 1 - 1(1 - \alpha)^{1/k}$$

where:

k = number of tests

$\alpha = 0.05$

α' = new alpha value

Tukey's post-hoc test for all pairwise comparisons was used to determine: 1) significant differences between particular sites within the same season and; 2) site differences between spring and late summer. The critical value for the post-hoc test remained at 0.05.

Perennial versus Intermittent Streams

Nine of the sites sampled in spring and late summer held differing amounts of water in their channels and some sites are influenced by beaver activity, which created a pooling effect. Other sites lacked this influence and tended to hold less water in their channels. In spring, all sites had water throughout the sampling reach, whereas in late summer only four sites were wet throughout the reach. Based on these observations, the nine sites were partitioned as perennial (n=4) and intermittent (n=5). *A priori* contrasts in a one-way ANOVA were used to determine significant differences between perennial and intermittent streams for seven metrics. Metric comparisons were determined individually for each season.

DO and Benthic Community

The State's current criterion for DO is 5.0 mg L^{-1}, however, seasonal criteria have been proposed (3.0 mg L^{-1} for June – October and 5.0 mg L^{-1} for November – May) (LDEQ 2001). In order to determine the relationship of these metrics to the proposed criteria, sites were divided as above and below the DO criteria. For spring samples, two sites were above 5.0 mg L^{-1}, and seven sites fell below. For late summer, three sites were above 3.0 mg L^{-1} and four sites below. *A priori*

contrasts were used in a one-way ANOVA (PROC MIXED, SAS, version 9.1, SAS Institute Inc., Cary, NC, USA) to determine if metrics differed between sites with differing DO levels for each season.

RESULTS

Benthic Community Description

Abundance and Richness

A total of 25,467 aquatic macroinvertebrates were enumerated from eleven monitoring locations in spring and nine locations in late summer of 2006 (Appendix A). The average abundance of benthic macroinvertebrates in spring was 716, ranging from 355 – 1707, and in late summer average abundance was 2006, ranging from 62 - 4222. Total abundance was dominated by Diptera (48.8%) and Crustacea (37.7%) in spring and Diptera (93%) in late summer with Chironomidae being the dominate dipteran (Table 4.1).

Table 4.1. Seasonal macroinvertebrate abundance, richness, and dominance by site.

Season	Site	Total Abundance	Taxa Richness	Dominant Taxa (%)
Spring	9D	180	10	Culicidae (38.3)
	9U	301	7	Culicidae (40.5)
	E1	636	24	Chironomidae (30.6)
	E2	602	15	Chironomidae (38.4)
	E3	355	18	Chironomidae (27.2)
	I1	390	24	Chironomidae (54.1)
	I2	1311	20	Asellidae (45.1)
	I3	841	22	Ceratopogonidae (53.3)
	I4	645	16	Chironomidae (37.3)
	I5	906	22	Asellidae (45.2)
	I6	1707	20	Asellidae (43.6)
Late Summer	9D	-	-	NA
	9U	-	-	NA
	E1	2064	22	Chironomidae (62.5)
	E2	562	11	Chironomidae (54.6)
	E3	773	15	Chironomidae (70.0)
	I1	62	14	Chironomidae (29.0)
	I2	1850	21	Chironomidae (61.8)
	I3	3339	14	Ceratopogonidae (48.3)
	I4	2372	15	Chironomidae (77.8)
	I5	2813	23	Chironomidae (90.4)
	I6	4210	27	Chironomidae (84.3)

Biological Indices

In total, 55 families were collected from 18 orders, with some families including multiple genera. Spring samples had an average taxonomic richness of 18, ranging from 7 – 24, and late summer average richness was 18, ranging from 11 – 27 (Table 4.1). Total abundance was higher in late summer than in spring at six of the nine sites, while taxonomic richness was similar between seasons at all sites except one (Figures 4.1 and 4.2).

Shannon-Wiener diversity index decreased at six out of the nine sampled sites in late summer (Figures 4.3). Excluding site E3, all sites with a decrease in diversity in late summer also exhibited increases in macroinvertebrate abundance, with no major changes in taxonomic richness. This decrease in diversity was due to an increase in dipterans which doubled or tripled at most sites from spring to late summer samples. All metrics are located in Appendix B.

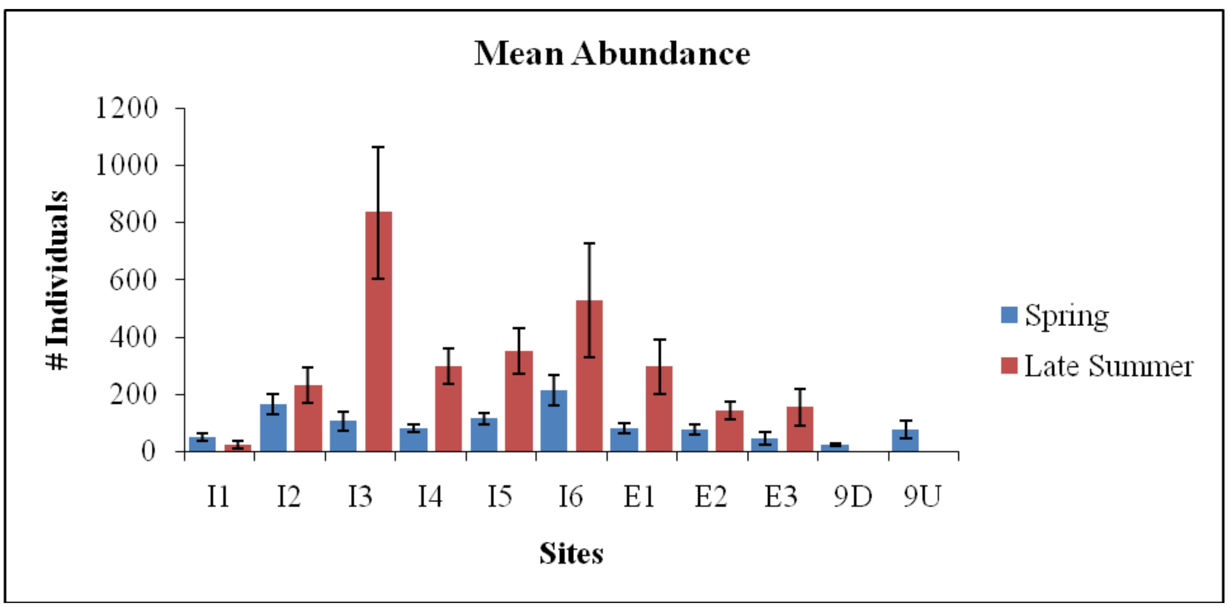

Figure 4.1. Mean abundance (± 1SE) at 11 monitoring locations in the Flat Creek watershed in spring and late summer 2006.

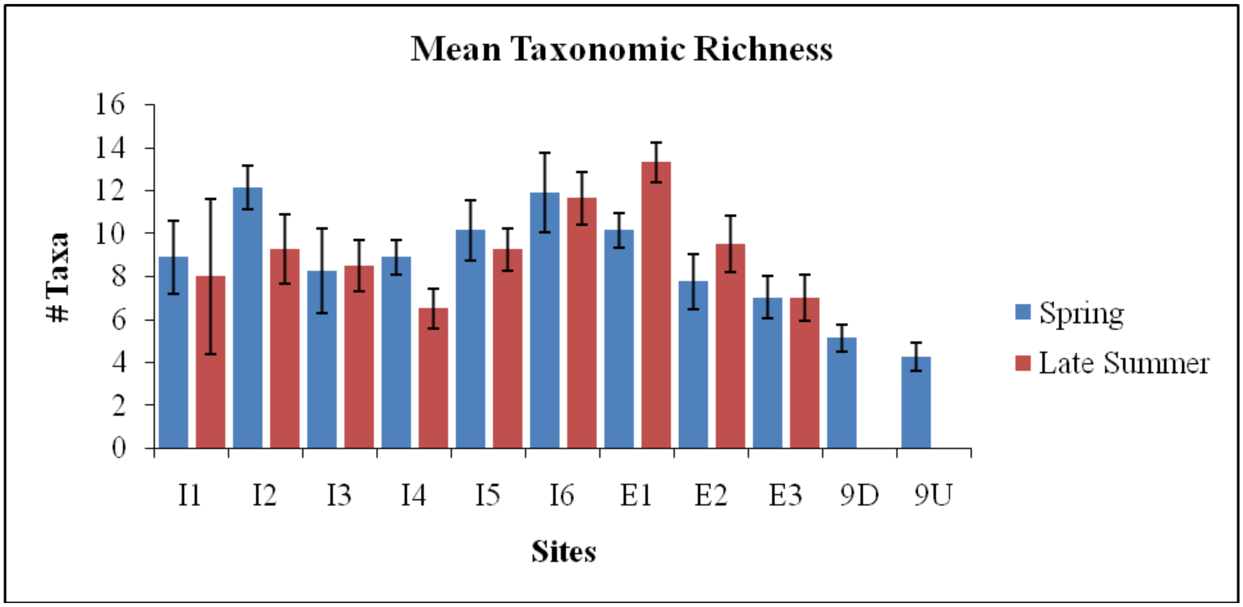

Figure 4.2. Mean taxonomic richness (± 1SE) at 11 monitoring locations in the Flat Creek watershed in spring and late summer 2006.

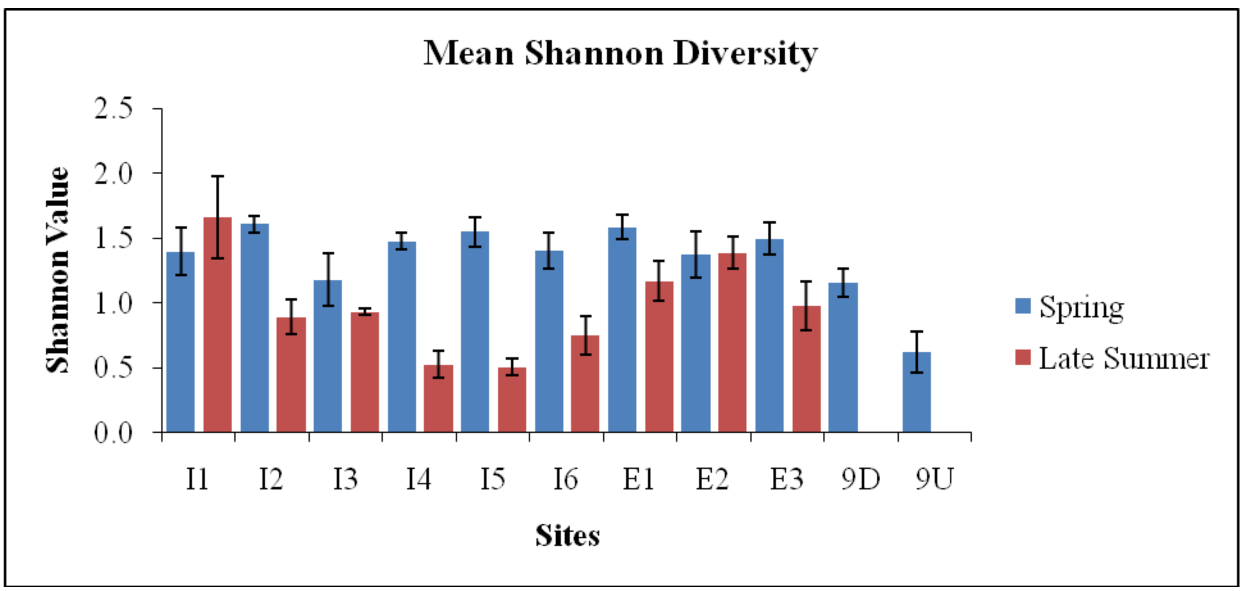

Figure 4.3. Mean Shannon-Wiener diversity (± 1SE) at 11 monitoring locations in the Flat Creek watershed in spring and late summer 2006.

Functional Feeding Groups

Functional feeding groups (FFG) were determined for the 25 most common taxa (Table 4.2) based on classification schemes by Merritt and Cummins (1996) , Smith (2001), and Thorp and Covich (2001). Collectors include organisms that filter and gather. Because some taxa have more than one FFG categorization making classification difficult, only the 25 most common taxa were used. Of these common taxa, 67% were collectors and 21% were predators (Figure 4.4).

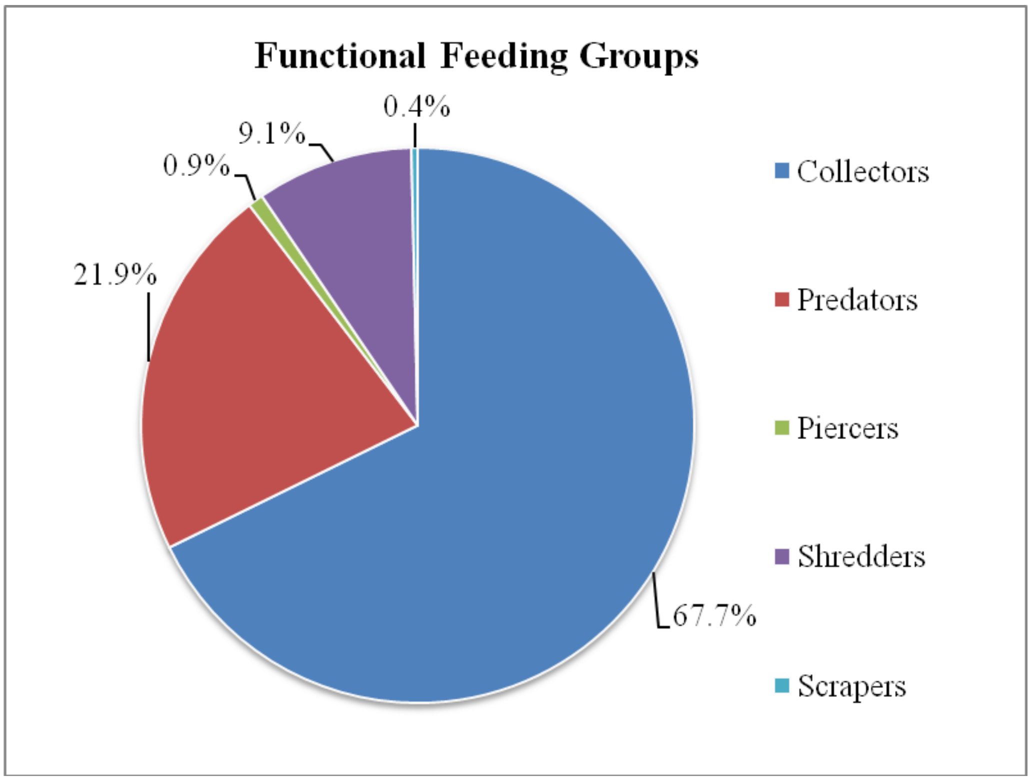

Figure 4.4. Functional feeding group composition of common taxa.

Table 4.2. Functional feeding group classification for the 25 most common taxa collected throughout the Flat Creek watershed during spring and late summer 2006.

Taxa	**Functional Feeding Group**
Lumbricidae	Collectors
Hirudinea	Predators
Hydrachnidia	Predators
Amphipoda	Collectors
Synurella	Collectors
Hyalella	Collectors
Asellidae	Shredder
(Palaemonidae) *Palaemontes*	Collectors
Dytiscidae	Predators
Oreodytes	Predators
Ceratopogonidae	Predators
(Ceratopogonidae) *Serromyia*	Predators
Chaoboridae	Predators
Chironomidae	Collectors*
Culicidae	Collectors
(Culicidae) *Aedes*	Collectors
(Culicidae) *Culex*	Collectors
Caenidae	Collectors
(Ephemeridae) *Hexagenia*	Collectors
Corixidae	Piercers*
(Sialidae) *Sialis*	Predators
(Libellulidae) *Pachydiplax*	Predators
Ancylidae	Scrapers
Pelecypoda	Collectors
Sphaeriidae	Collectors

*May also be predators

Environmental Variables and the Macroinvertebrate Community

Canonical correlation analysis described relationships between community structure and environmental variables, including season (Tables 4.3 and 4.4). The first four canonical variates were highly significant (all $p < 0.0001$) and explained 75% of the cumulative variance. The first variate was associated with spring (lower temperature, lower nitrate, and higher TSS) and was positively related to *Hyalella*, *Synurella* and Asellidae, and was negatively related to Chironomidae, Chaboridae, Hydrachnidia, and Sialidae. The second variate was related to pH and was positively related to Caenidae. The third variate was positively associated with velocity, DO, Lumbricidae, and *Hexagenia* and negatively associated with *Hyalella*. The fourth variate was positively associated with wetted area and wetted width, and was positively related to Amphipoda and Hirudinea.

Seasonal Variation

Shannon-Weiner diversity, evenness, and percents Diptera, Chironomidae, Isopoda, and Crustacea failed to meet the assumption of normality based on Shapiro-Wilks test and therefore are not considered in further analyses. Results from the two-way ANOVA showed no seasonal difference for any metrics in spring and late summer (Table 4.5). The seasonal examination of functional feeding groups showed a 28% increase in collectors from spring to late summer (Figure 4.5).

Table 4.3. Canonical variates (CV1 – CV4) for environmental parameters. Canonical variates in bold indicates significant at >0.4.

	CV1	CV2	CV3	CV4
Season	**0.8696**	-0.0681	0.2572	0.0541
Width	0.1677	-0.0147	-0.1923	**0.6520**
Area	-0.0846	0.0073	-0.2386	**0.5880**
Velocity	0.1201	-0.3428	**0.6535**	0.3370
Cover	-0.2880	0.0981	-0.0242	-0.2900
Temp	**-0.8633**	0.2224	-0.1121	0.0175
Specific Conductance	**-0.7136**	0.1310	-0.2340	0.2218
DO	-0.2742	0.0480	**0.6769**	-0.1576
pH	-0.1470	**0.5090**	-0.2625	0.3036
TSS	**0.5087**	0.1030	-0.0934	0.2777
TS	0.3665	-0.1040	**-0.4735**	0.3424
TDS	0.0724	-0.2433	**-0.6014**	0.2455
TP	0.0615	0.0776	**-0.4668**	0.4315
Nitrite	-0.4150	0.2134	-0.0076	0.2688
Nitrate	**-0.7149**	0.1431	-0.2809	0.3093

Table 4.4. Canonical variates (CV1 – CV4) for taxa. Canonical variates in bold indicates significant at >0.4.

	CV1	CV2	CV3	CV4
Lumbricidae	0.1015	-0.1639	**0.4604**	0.3186
Hydrachnidia	**-0.5459**	0.3758	-0.1649	0.2667
Amphipoda	0.3460	-0.0258	-0.0306	**0.4750**
Synurella	**0.4515**	-0.0655	0.1966	0.3113
Hyalella	**0.4560**	0.3639	**-0.4783**	0.2655
Oreodytes	0.2473	-0.0977	0.0690	-0.1255
Dytiscidae	0.0469	0.0422	0.1506	0.1898
Palaemontetes	-0.1507	0.2157	-0.0710	-0.1651
Serromyia	0.1704	0.4035	0.0124	-0.0648
Ceratopogonidae	-0.0986	0.0474	-0.1911	0.3155
Chaoboridae	**-0.5054**	0.0650	-0.2458	-0.1012
Chironomidae	**-0.6006**	0.1690	-0.1710	0.3806
Culicidae	0.2595	-0.0166	-0.1029	-0.0720
Aedes	0.2116	0.0322	-0.1927	0.0733
Culex	0.3588	0.1154	-0.1581	-0.0370
Caenidae	-0.2184	**0.8339**	0.3067	-0.0610
Hexagenia	0.0000	-0.1445	**0.4533**	0.3684
Corixidae	0.0408	0.2423	0.1740	-0.0828
Asellidae	**0.6741**	0.3563	-0.0616	0.3390
Sialis	**-0.5753**	0.2534	-0.1810	0.0209
Pachydiplax	-0.2040	-0.0040	-0.1668	0.1504
Ancylidae	-0.3707	0.1238	0.0217	-0.1685
Hirudinea	0.0863	0.0889	-0.2674	**0.5193**
Pelecypoda	0.0689	0.3251	0.2565	0.0134
Sphaeridae	0.2913	-0.2437	0.1071	0.0410

Table 4.5. ANOVA results for biological metrics were not different between spring and late summer 2006. Critical value determined with the Dunn-Sidak method ($p \leq 0.0057$).

Metric	Seasonal Difference?	F value, P value
Total Abundance	No	F=2.45, p=0.0178
Taxonomic Richness	No	F=0.96, p=0.4691
Percent EPT	No	F=0.61, p=0.6943
Percent Amphipoda	No	F=1.04, p=0.4008
Percent Odonata	No	F=1.11, p=0.36
Percent Gastropoda	No	F=1.05, p=0.4127
Percent Pelecypoda	No	F=3.06, p=0.0079

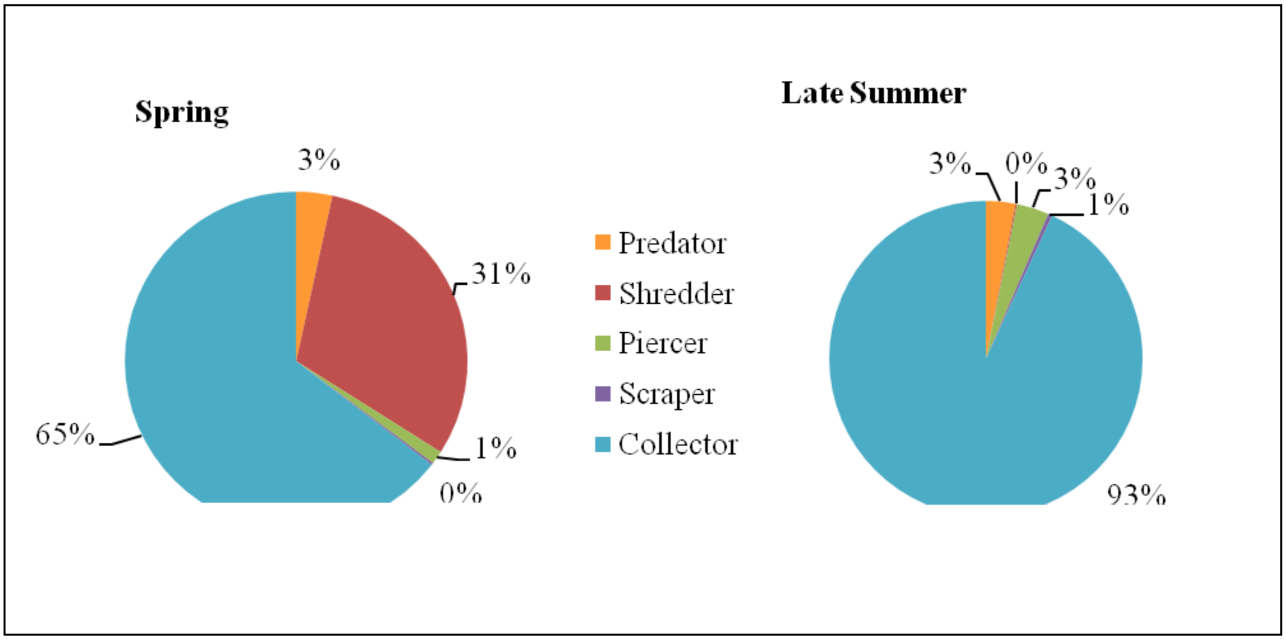

Figure 4.5. Functional feeding groups for spring and late summer. Collectors include filterers and gatherers.

Spatial Variation

Intermittent and Perennial Differences

A priori contrasts showed significant differences between intermittent and perennial streams for four biotic metrics in spring and four metrics in late summer (Table 4.6). Differences between intermittent and perennial stream sites were found for total abundance, percent EPT, and percent Pelecypoda in both seasons (Figure 4.6). Significant differences were found for taxonomic richness in spring, and percent Gastropoda in late summer. The numbers of EPT and Pelecypoda taxa were greater at intermittent than perennial sites and conversely, total abundance and the number of chironomid taxa were greater at perennial sites (Figure 4.7). Additionally, chironomids were more abundant at perennial sites and their increase contributed largely to the abundance metric (Figure 4.7). Total abundance differed between stream types during both seasons with more individuals at perennial sites than intermittent sites. Differences in stream permanence were most evident along Turkey Creek where the upper- and lower-most sites (I3 and E2) were intermittent and the three sites in between (I4, I5 and I6) were perennial.

Table 4.6. Results from ANOVA *a priori* contrast of perennial and intermittent streams for metrics from spring and late summer showing critical value ≤ 0.05 in bold.

Metric	Spring	Late Summer
Total Abundance	**F=16.78, p=0.0001**	**F=5.81, p=0.02**
Taxonomic Richness	**F=6.51, p=0.0132**	F=0.01, p=0.9138
Percent EPT	**F=11.88, p=0.0019**	**F=7.27, p=0.0225**
Percent Amphipoda	F=0.45, p=0.5079	F=1.44, p=0.2532
Percent Odonata	F=0.20, p=0.6645	F=0.71, p=0.4059
Percent Gastropoda	F=1.21, p=0.2805	**F=4.8, p=0.0419**
Percent Pelecypoda	**F=8.08, p=0.0068**	**F=28.93, p=0.0001**

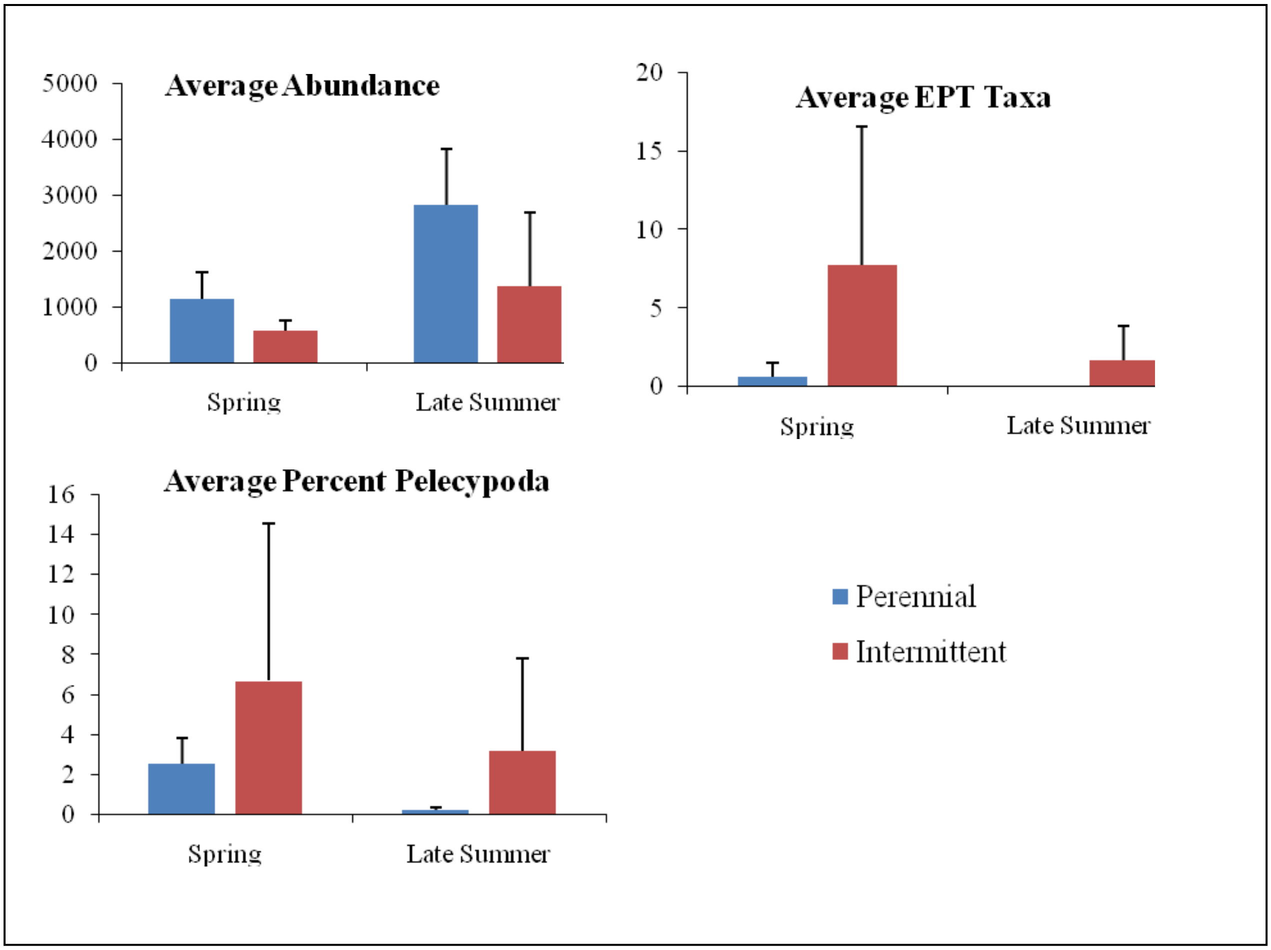

Figure 4.6. Metrics that differed between stream types during both seasons. Average values for Abundance, Percent EPT, and percent Pelecypoda with standard error bars.

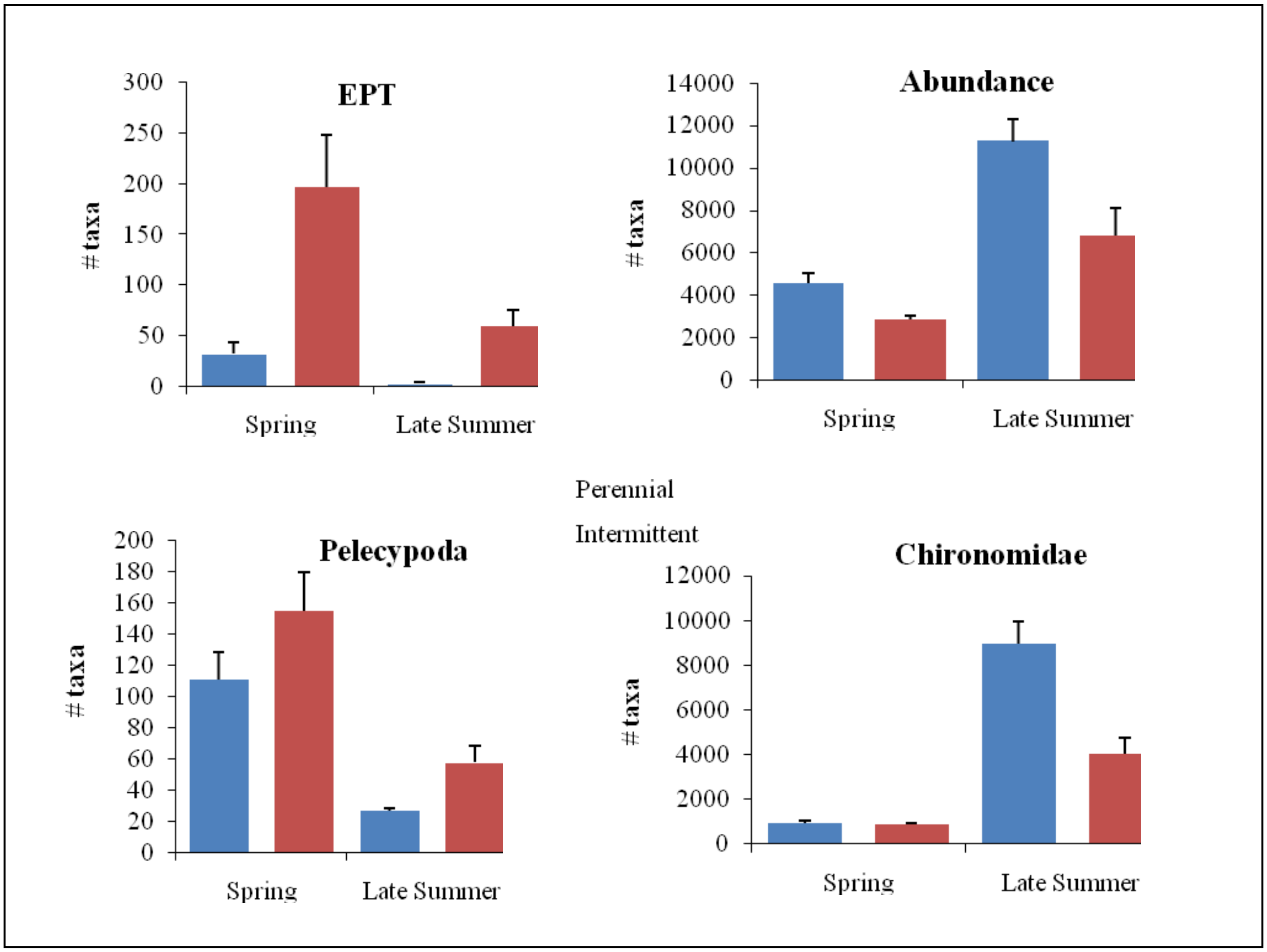

Figure 4.7. Number of EPT taxa, individuals (abundance), Pelecypoda, and Chironomidae with 2 standard error bars at perennial and intermittent sites in spring and late summer in the Flat Creek watershed during 2006.

DO Levels and Metrics

Comparison of sites based on proposed DO standards were significant for three biotic metrics in spring and two metrics in late summer (Table 4.7). Percent Gastropoda was significantly different in spring. The six other metrics did not differ between the partitioned DO levels for each season. Total abundance was significantly different in spring, whereas percent EPT was significantly different in both spring and late summer (Figure 4.8).

Table 4.7. ANOVA results showing differences between biological metrics and sites partitioned by proposed DO values for spring and late summer. Bold indicates critical value ≤ 0.05.

Metric	Spring	Late Summer
Total Abundance	**F=6.54, p=0.013**	**F=3.99, p=0.0518**
Taxonomic Richness	F=1.75, p=0.1910	F=0.95, p=0.3347
Percent EPT	**F=33.82, p=0.0001**	**F=15.66, p=0.0027**
Percent Amphipoda	F=0.92, p=0.3422	F=0.03, p=0.8645
Percent Odonata	F=0.41, p=0.5331	F=1.65, p=0.2095
Percent Gastropoda	**F=5.40, p=0.0276**	F=0.12, p=0.7341
Percent Pelecypoda	F=0.00, p=0.9974	F=0.04, p=0.8349

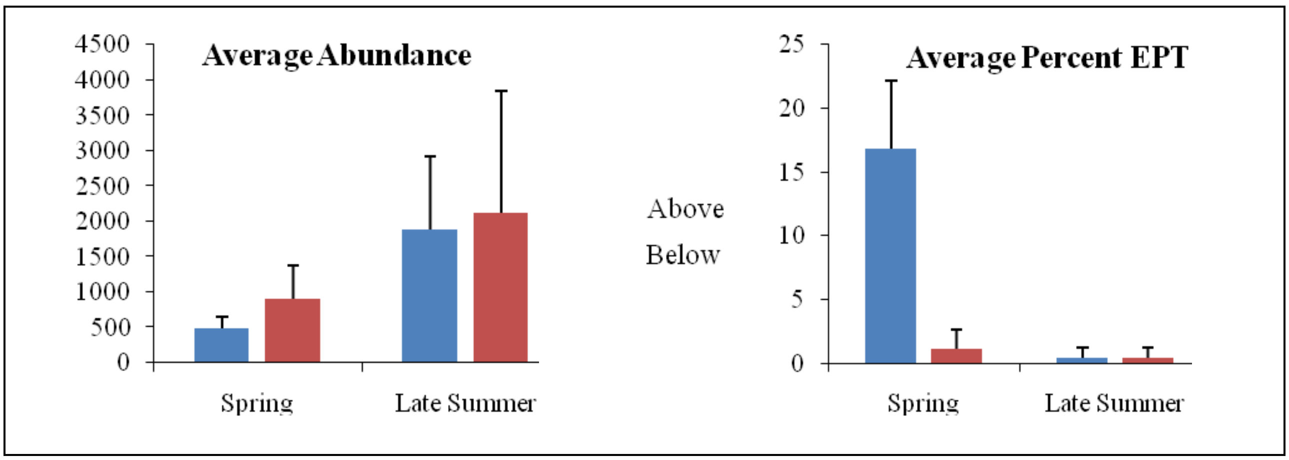

Figure 4.8. Biotic metrics that were significantly different at sites above and below LDEQ's proposed seasonal DO levels. Average values for abundance and percent EPT with standard error bars.

The ephemeropteran, *Hexagenia*, was positively correlated with DO (Tables 4.3 and 4.4). These ephemeropterans were found only at one site, E2, which had a DO level ≥ 4.4 mg L^{-1} and ≥ 48 % saturation throughout the year (Figure 4.9), whereas most other intermittent sites (with the exception of E1) fell below the proposed standard of 3.0 mg L^{-1} between June and October (Figure 4.10).

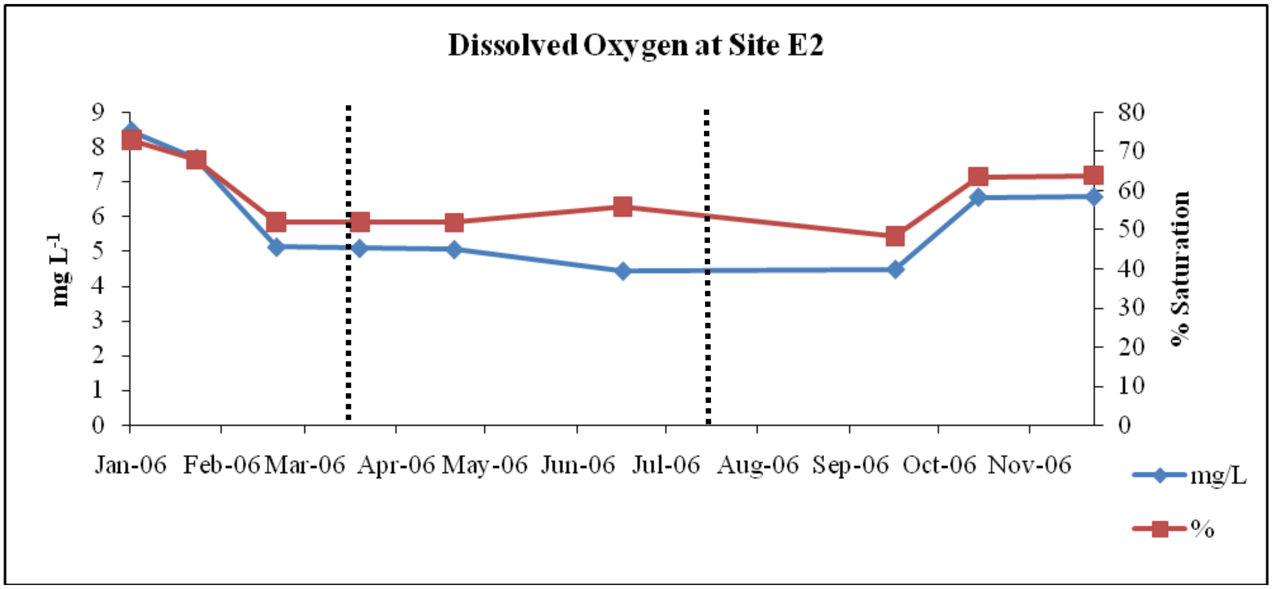

Figure 4.9. Year long *in situ* DO values for site E2 with the most *Hexagenia* individuals. Dashed vertical lines indicate macroinvertebrate sampling dates.

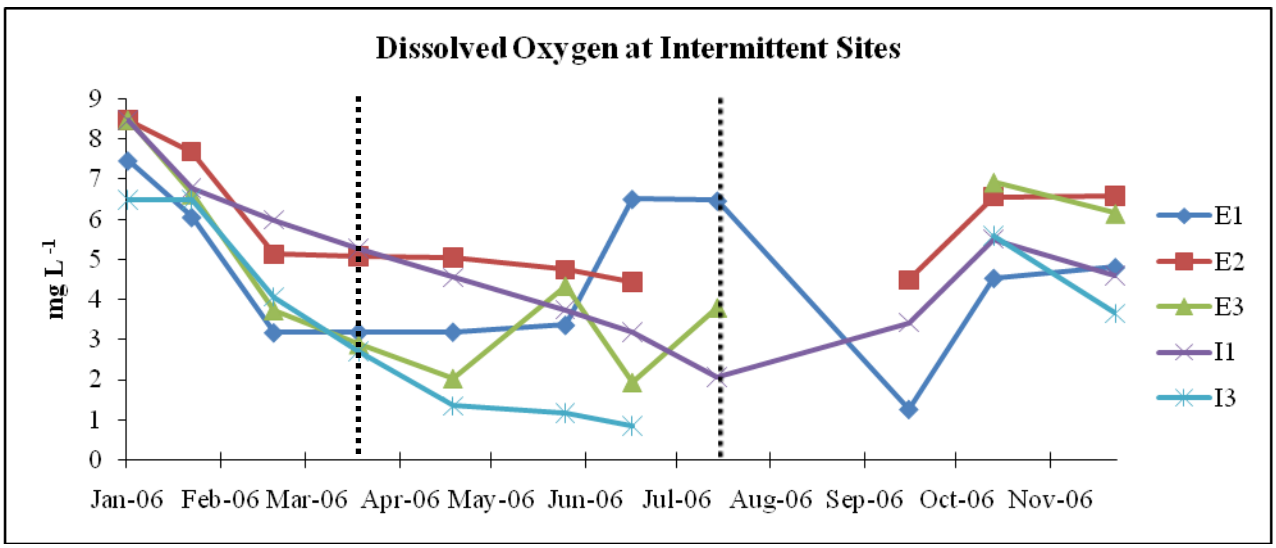

Figure 4.10. Year long *in situ* DO values for all intermittent sites. At sites E2, E3, and I3, values were missing in August through October due to lack of water. Dashed vertical lines indicate macroinvertebrate sampling dates.

DISCUSSION

Watershed-Wide Characteristics of Macroinvertebrates

The most common taxa found in the watershed were typically generalist taxa tolerant of low DO conditions that have adapted to the unique subtropical lowland streams of central Louisiana. Macroinvertebrate assemblages are usually described by their feeding strategies, which commonly reflect water quality characteristics. Throughout the watershed, filtering- and gathering- collectors, which were most common (Figure 5.1) feed on organic matter suspended in the water column and on fine organic detritus (Merritt and Cummins 1996, Camargo et al. 2004). Feeding strategies of these organisms reflect the slow-moving, high organic matter stream characteristics throughout the Flat Creek watershed. The 28% increase of collectors in late summer was mostly due to the decrease in shredders. The lack of shredders in later summer was possibly because of the decrease in allochthonous material during that time of year. The number of predators stayed the same between seasons, whereas piercers increased in late summer. The increase of those organisms was due to the increased availability of food. In all, the macroinvertebrates collected in this study reflected the water quality characteristics in these low-gradient, seasonally hypoxic, headwater streams.

There was considerable variation in macroinvertebrate abundance throughout the Flat Creek watershed (Figure 5.2) and the variation was not consistent among the sites between seasons. The Coefficient of Variation (CV) for abundance in spring was highest at E3 and lowest at I4 whereas in late summer sites I1 and I6 had the highest CV and E2 had the lowest CV. Macroinvertebrate abundance is highly variable within a stream and the sample size in this study (n=8) was chosen to decrease variation, however, the CV for abundance was ≥ 44% at all sites during both seasons. The variation between seasons was expected because life history

characteristics, such as aquatic stages, voltinism, and potential for diapause, differ between organisms. Likewise, differences in CV between sites within a season was also anticipated because of the heterogeneous nature of stream habitats that is typically reflected by the biota (Heino et al. 2004). In agreement with this study, Sporka et al. (2006) found high variation among metrics including abundance between seasons and attributed it to seasonal life cycles of the benthic taxa.

No clear relationship between macroinvertebrate community structure and stream position was found in this study. Although the river continuum concept (Vannote et al. 1980) demonstrates a stream as a series of longitudinal, physical gradients and corresponding biological changes, the concept has attracted many critics, mainly for its lack of generality (Cushing et al. 1995). Research has increasingly recognized that small headwater streams physically differ from downstream reaches. Some researchers have recently proposed that the uppermost headwaters of a stream differ structurally in terms of stream hydraulics and morphology, which sequentially influences habitat characteristics and resident biota (Gooderham et al. 2007). In the Flat Creek watershed, beaver and woody debris dams were prevalent throughout the stream network, particularly along Turkey Creek, and had modified the stream flow and sediment transport (Saksa et al. 2007).

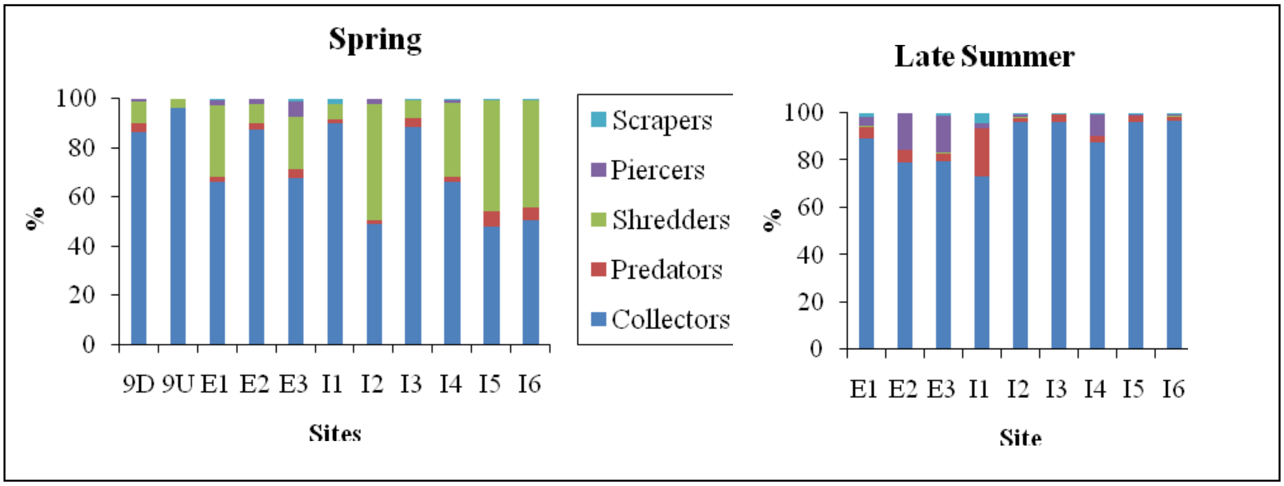

Figure 5.1. Functional feeding groups by site in spring and late summer. Collectors include gatherers and filterers.

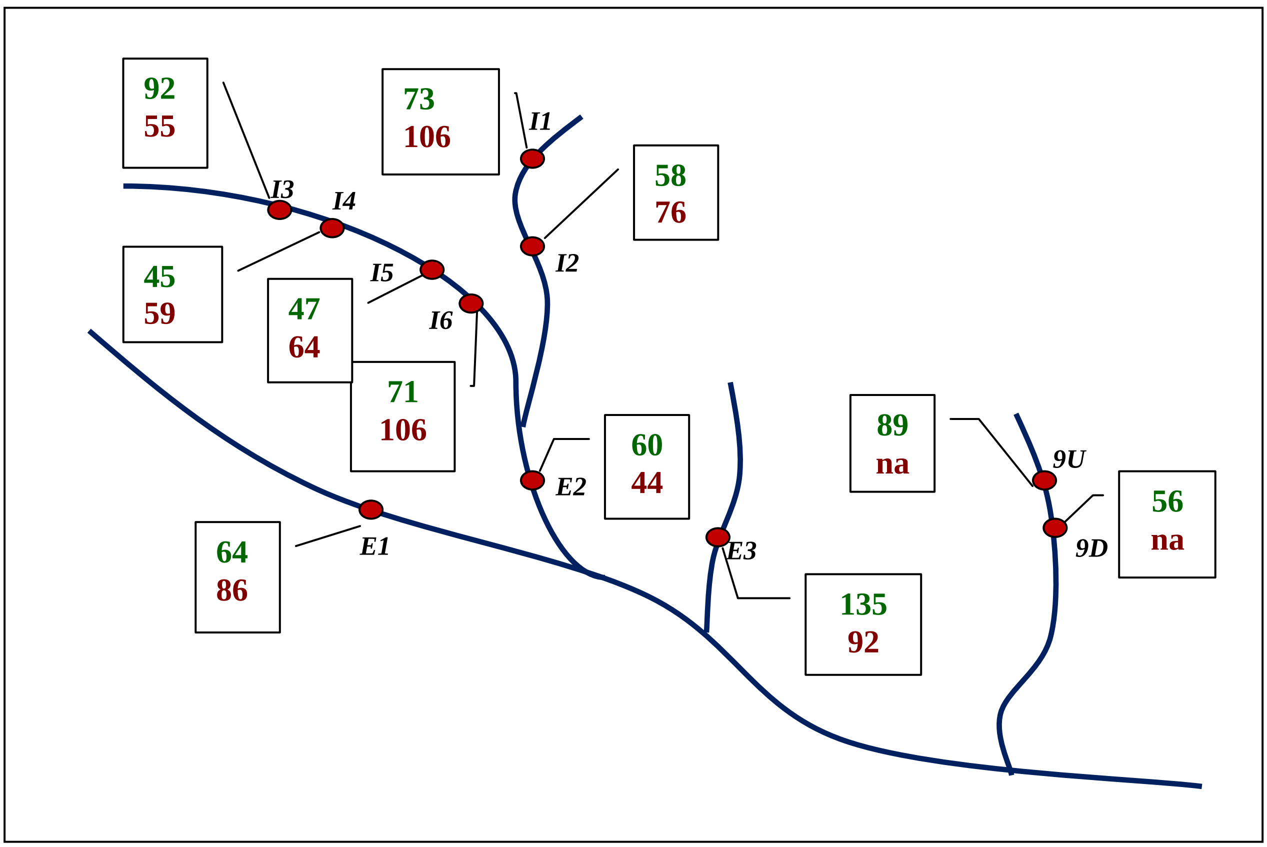

Figure 5.2. Schematic of Flat Creek watershed showing eleven monitoring locations with associated coefficient of variation (%) for abundance. For each site, the green percent value on top indicates spring and red value below indicates late summer.

Seasonal Variation of Macroinvertebrates

A strong seasonal difference reflected by the biotic metrics was expected based on previous research (Lenat 1993, Sporka et al. 2006). However, despite the apparent physicochemical differences among seasons in the Flat Creek watershed, none of the biotic metrics differed significantly between spring and late summer. This is likely a result of both biological attributes of the macroinvertebrates and physicochemical changes within the stream. It is important to recognize that the life history characteristics of resident macroinvertebrates are shaped by the chemical and hydrological processes, within a stream, thus separating biological and physicochemical influences from one another is not practical.

Streams in the Flat Creek watershed were sampled during periods of high and flow, thus a significant change in the macroinvertebrate metrics was expected, however none of the tested metrics differed between the two sampling periods. Interestingly overall community structure did differ between the two seasons. In spring, a period of high flow, the number of *Hyalella* (Amphipoda) and Asellidae (Isopoda) were high, whereas chironomidae abundance was low. These results are similar to Davis et al. (2003), who observed an increase in isopods and crustaceans, and a decrease in dipterans during periods of high flow.

Other studies have shown seasonal physicochemical influences on macroinvertebrate metrics. In European mountain streams, macroinvertebrate community structure as a whole was affected by seasonality and macroinvertebrate metrics differed significantly between months (Sporka et al. 2006). Although overall seasonal changes in subtropical regions are not as drastic as temperate mountain environments, seasonal differences, including differences in ambient temperature and precipitation, were observed in the Flat Creek watershed.

The seasonal influences on macroinvertebrate metrics are also caused by distinct biological differences among taxa (Beche et al. 2006). Although not statistically significant, a decrease in Shannon-Wiener diversity index was observed at five sites in late summer. An increase in abundance coincided with the decrease in diversity. The lower diversity may be explained by biological differences in the macroinvertebrate taxa such as developmental characteristics and life history patterns, both of which have been found to differ in headwater streams (Meyer et al. 2007). The observed increase in abundance from spring to late summer was likely driven by the 43% increase of dipterans in late summer. Similar to the observations in Flat Creek, fall macroinvertebrate abundance has been found to increase in headwater streams of central Florida (Cowell et al. 2004) and in western Louisiana, differences in species composition have been attributed to seasonal variations (Williams et al. 2005). The increase of dipterans in the Flat Creek watershed likely influenced the Shannon-Wiener diversity index because chironomids typically dominate freshwater environments (Merritt and Cummins 1996). The percents Crustacea (including Isopoda and Amphipoda), Odonata, Gastropoda, Pelecyopoda, and EPT were observed to decrease considerably in late summer, which is also attributed to dipteran increase. In late summer, DO and wetted stream area was lower. Aquatic insects, including chironomids, have adapted to low DO conditions as depicted by cutaneous respiration and respiratory pigments (Eriksen et al. 1996).

Although many biotic metrics are available for use, their effectiveness varies by geographical region. For example, Davis et al. (2003) found that EPT, commonly used in high altitude, cold-water streams, did not appropriately represent Georgia’s Gulf Coastal Plain region. This was attributed to the low number of EPT individuals found in the study streams, and the authors concluded that the biotic metrics including percents Diptera, Isopoda, and Crustacea

would reflect the macroinvertebrate community of that region. Similar to the findings of Davis et al. (2003), EPT taxa were sparse in the Flat Creek watershed and dipteran, isopod, and crustaceans were denser in the Flat Creek watershed.

Three possible explanations exist for the lack of seasonal influence on the macroinvertebrate metrics in the Flat Creek watershed. First, it is possible that physicochemical differences between sampling periods were not of sufficient magnitude to influence the metrics. In addition, selected metrics may not have appropriately reflected the macroinvertebrates biological traits, such as reproduction, life cycle duration, and respiration mode. In my study, season did not influence the selected metrics and was not an important factor. Finally, this research was also limited to only two sampling periods over one year, which may not have been enough time to conclusively determine the importance of season on the tested metrics.

Impacts of Flow Permanence on Macroinvertebrates

The influence of stream flow permanence on macroinvertebrate community dynamics was reflected by differences in biotic metric scores. Intermittent sites were not all located in first order streams, suggesting that site specific differences, such as drainage size, position on the landscape, shallow groundwater level, or beaver activity, were important factors contributing to flow variability. Many perennial sites, impacted by beaver activity, had an increase in abundance, and researchers have shown that pooling caused by beaver dams is an important factor affecting macroinvertebrate assemblages (McDowell and Naiman 1986). The spatial differences found in the metrics based on stream permanence are attributed to site specific influences such as elevation changes.

The increase of abundance at perennial sites may have been an artifact of the sampling technique or reflectance of the difference in available habitat in the two stream types. Other

researchers have reported a decrease in invertebrate abundance associated with a decrease in wetted width (Dewson et al. 2007). In my study, the average wetted width was higher at perennial sites than at intermittent sites, supporting the concept that an increase in available habitat results in an increase in macroinvertebrate abundance. Similar to abundance, the percent of EPT taxa and Pelecypoda also differed between intermittent and perennial sites in both seasons. In contrast to abundance, however, the numbers of EPT taxa and Pelecypoda were higher in intermittent streams. Other researchers in the southeast have found EPT taxa abundance to decrease in intermittent streams (Feminella 1996, Davis et al. 2003). However, these studies were conducted in Alabama upland streams, and 2nd to 3rd order streams in areas draining agricultural fields that were larger and physically different, which makes direct comparisons difficult. The increase in EPT taxa found in this study is attributed to the abundance of the burrowing mayfly, H*exagenia*, which was most abundant at site E2, an intermittent stream reach.

Two metrics, taxonomic richness, and percent gastropod, differed between stream types in spring (richness only) and late summer (diversity and gastropod). Research in upland streams of Alabama indicated that the benthic fauna was similar among streams with differing flow permanence, and resolved that assemblages differences were reflected by spatial and temporal variation in stream flow (Feminella 1996). Additionally, year-to-year variations in assemblages within a stream were similar to differences in assemblages among streams with contrasting flow permanence within a given year. In the Flat Creek watershed, spring macroinvertebrate diversity was comparable among streams but not in late summer, whereas the opposite trend was observed for taxonomic richness. A possible explanation for the difference of taxonomic richness between stream types in spring, and not late summer, is the increased frequency of natural disturbance (i.e., storm events) in spring months. Site E2, an intermittent site, had the lowest number of taxa

in spring. Research conducted on intermittent prairie streams found taxonomic richness was lower at sites with a high harshness index, which was based on flow regime and surface water connectivity (Fritz and Dodds 2005). It is interesting to note that differences in taxonomic richness among sites were not significant in late summer throughout the Flat Creek watershed. This finding may be attributed to the overriding influence of drought and the lack of hydrological disturbance resulting from dry season conditional typical of that time of year.

Low DO and Macroinvertebrate Metrics

The Flat Creek macroinvertebrate community was comprised of many taxa adapted for low oxygenated waters. Dipterans have specific respiratory adaptations for low DO conditions, and chironomids, with cutaneous respiration and specialized hemoglobin, are typical of streams with low DO levels and high organic matter content. Asellid isopods were dominant at some sites in spring but were scarce in late summer, which could have been due to the ability of some asellid species to burrow in the substrate during times of drought (Smith 2001). Overall, the dominant taxa, including chironomids and asellids, found throughout the watershed reflect the low DO water quality characteristics and hydrological seasonal changes.

Total macroinvertebrate abundance and percent of EPT taxa were significantly different between high and low DO sites for both seasons. Similar results were reported from a study on macroinvertebrate assemblages and DO in southwest Louisiana, where total abundance and diversity of macroinvertebrates increased with lower DO levels (Kaller and Kelso 2007). In the Flat Creek watershed, total abundance was observed to increase at sites with lower DO, which is attributed to the increased abundance of the low-DO tolerant, chironomids. Percent EPT taxa did differ between high and low DO, which is notable because of the low number of Ephemeropterans, Plecopterans, and Trichopterans found in the samples in comparison to other

taxa. Only one Trichopteran and six Plecopterans were found throughout the watershed, although 285 Ephemeropterans were identified in the benthic sample.

On observations from this research, total macroinvertebrate abundance as a metric in aquatic bioassessments should be used with caution. Chironomids dominated the taxa found throughout the watershed and are known to be ubiquitous in stream systems. These organisms are adapted to low oxygenated water bodies which would explain their extreme abundance, and the higher macroinvertebrate abundance metric in summer was due to the frequency of this generalist taxon. Also at perennial sites a greater area was sampled because these streams were deeper. This may have lead to collecting more individual organisms at those locations. Thus, due to taxonomic homogeneity and artifacts from the sampling procedure, the use of total abundance as a viable metric to describe macroinvertebrate community structure in the Flat Creek watershed may be ineffective.

Gastropod abundance also differed in spring between sites above and below 5.0 mg L^{-1}, which warrants investigation of this metric as a useful bioassement tool. The relative proportion of prosobranchs (gastropods that use gills to obtain oxygen) and pulmonates (gastropods that have a pulmonary cavity for respiration) is indicative of DO conditions. More prosobranchs indicate sufficient DO whereas more pulmonates indicate low DO problems. Sites above the proposed spring-season DO level of 5.0 mg L^{-1} yielded 31 prosobranchs, whereas one site with less than 5.0 mg L^{-1} had 36 prosobranchs. By further investigation of the percent gastropod metric it is clear that this metric too should be used with caution. For future assessments, a more useful metric would be one that uses the proportion of pulmonates to prosobranchs. Although it remains questionable whether percent of gastropod taxa would be a useful metric for Flat Creek

based on these findings, others have recommended using it in bioassessments (Camargo et al. 2004).

Macroinvertebrates Descriptive of Environmental Variables

Correlation analyses showed spring related to many individual macroinvertebrate taxa whereas analysis of variance did not detect seasonally significant differences among the metrics. An inverse relationship between seasons and metrics was expected because of differences in temperature, specific conduntance, nitrate, and TSS. The increase in TSS and decrease in nitrate concentrations are due to higher spring precipitation levels. Research in headwater streams has shown that low precipitation and drying resulted in high nitrogen concentrations (Nakashima and Yamada 2005), whereas in the Flat Creek watershed, increases of TSS are attributed to increased precipitation (Saksa 2007).

A high abundance of chironomids in spring was observed. TSS was negatively correlated with chironomid abundance, which is unexpected because chironomids are collector-filterers. Asellidae isopod abundance was found to positively correlate with spring, which would be expected because more allochthonous detritus would be present at that time of year due to increased runoff. Additionally, a positive relationship was found between the burrowing mayfly *Hexagenia* (Ephemeropteran) and DO level. These individuals were identified only at site E2, which had a DO level at or above 4.4 mg L^{-1} throughout the year. The lack of this mayfly genus at other sites is attributed to their sensitivity to hypoxic conditions. In two studies of the nymph *Hexagenia limbata*, one found chronic oxygen levels below 7.0 mg L^{-1} to negatively influence their survival and size (Winter et al. 1996) and the other found hypoxic conditions in the littoral zone of lakes decreased their abundance (Rasmussen 1988). Although this particular species was not identified in the Flat Creek watershed, it does suggest that the chronic low DO conditions

found throughout the watershed influenced the presence, and abundance of the hexaganeid mayflies. Researchers in the Great Lakes region have found *Hexagenia* to be a useful indicator of recovering hypoxic conditions (Edsall 2001, Edsall et al. 2005) and developing a metric based on this genus might be useful for determining stream health in subtropical, low-gradient stream systems.

The generalist macroinvertebrate taxa found throughout the Flat Creek watershed is indicative of water quality characteristic high in organic matter and low in DO. An oxygen sensitive taxa was correlated with higher levels of DO. From this research, it is not possible to conclusively determine which metrics would be optimal in describing the benthic macroinvertebrate community of central Louisiana's slow-moving, oxygen-depleted streams. However, based on this study, metrics that would be useful in low-gradient subtropical streams in Louisiana include percent chironomidae, percent gastropoda (ideally the proportion of pulmonates to prosobranchs), and percent Ephemeroptera (or percent *Hexagenia*). Although it was not possible to test percent chironomids with *a priori* contrast, they should be considered as a viable biotic metric because of their dominance throughout the watershed and their use by other researchers in southeastern streams (Davis et al. 2003, Camargo et al. 2004, Herlihy et al. 2005). In addition, the development of a metric based on the burrowing mayfly *Hexagenia* would be useful for streams with seasonal hypoxic conditions because it has been useful elsewhere (Winter et al. 1996, Edsall 2001, Edsall et al. 2005). Due to the lack of Plecopterans and Trichopterans, the use of percent EPT taxa seems to be impractical. Ultimately a combination of these DO-sensitive and –insensitive metrics would be most beneficial in determining stream health in these low-gradient stream systems. More research is needed throughout the state to determine which macroinvertebrate metrics would be beneficial.

CONCLUSIONS

This study investigated spatial and seasonal patterns of 18 macroinvertebrate metrics and examined associations between the macroinvertebrate community and environmental variables in a seasonally hypoxic headwater streams in a low-gradient, subtropical watershed. The study achieved its primary objectives and made three important findings. First, a positive relationship existed between DO and the burrowing mayfly, *Hexagenia*. Second, most of the metrics were not significantly different between seasons but spatial variations were reflected by many of the metrics when sites were partitioned by stream permanence and DO level. Third, there was no clear trend in the macroinvertebrate structure and stream order. Localized flow conditions affected macroinvertebrates more than the stream position in this watershed.

The positive relationship between the burrowing mayfly, *Hexagenia*, and DO in an intermittent headwater stream suggests the importance of this taxon as an indicator of water quality for the low-gradient, headwater streams in the South Central Plains Ecoregion (SCPE). This mayfly was common to a stream which held a DO level ≥ 4.4 mg L^{-1} over a one year period. My research shows that a common DO-sensitive individual was common to one stream naturally disturbed by decreased flow permanence with a DO level not meeting current state regulations of 5.0 mg L^{-1} during the summer months. It is likely that *Hexagenia* was absent from the other intermittent sites because those sites typically had DO levels less than 4.4 mg L^{-1} throughout the summer months. This suggests that DO criteria less than 5.0 mg L^{-1} may be sufficient for intermittent headwater streams in these stream systems. Additionally, the use of EPT taxa in biological assessments of stream health should not be used in the SCPE because of the lack of plecopterans and trichopterans, but the observed abundance of ephemeropterans at several sites throughout the Flat Creek watershed suggest the usefulness of this taxon alone. Future

investigations into the use of Ephemeroptera as a metric would be useful for water quality managers developing macroinvertebrate bioassessment metrics in Louisiana.

Spatial differences between many of the metrics were significant when streams were partitioned by flow permanence and DO level. The distinction between intermittent and perennial streams was more important in determining differences between the metrics than season. This study indicates that distinguishing between stream permanence is important when conducting biological assessments based on finding differences among some metrics between varying stream types. Resource managers may consider using benthic macroinvertebrates in stream assessments of intermittent streams in lieu of physicochemical assessments, especially when water samples are unavailable due to non-continual stream flow.

In conclusion, this study demonstrates the importance of considering flow permanence when using benthic macroinvertebrates as indicators of water quality in headwater areas. It highlights the potential use of particular metrics for future bioassessments in low-gradient subtropical headwater streams. Although the study was conducted in a watershed that broadly represents the climate, geology, topography, and vegetation cover for the region, further research is needed in order to verify these findings and to conclusively determine appropriate macroinvertebrate metrics applicable to the SCPE in Louisiana.

LITERATURE CITED

Allan, J. D., and L. B. Johnson. 1997. Catchment-scale analysis of aquatic ecosystems. Freshwater Biology 37:107-111.

Alley, V. E. 2004. Analysis of benthic macroinvertebrate associations in low gradient streams of the Kisatchie National Forest (Central Louisiana, USA). MS Thesis. University of Louisiana, Monroe, LA.

Barbour, M. T., J. Gerritsen, G. E. Griffith, R. Frydenborg, E. McCarron, J. S. White, and M. L. Bastian. 1996. A framework for biological criteria for Florida streams using benthic macroinvertebrates. Journal of the North American Benthological Society 15:185-211.

Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish, 2nd ed. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water, Washington, D.C.

Bartsch, A. F., and W. M. Ingram. 1959. Stream life and the pollution environment. Public Works 90:104-110.

Beche, L. A., E. P. McElravy, and V. H. Resh. 2006. Long-term seasonal variation in the biological traits of benthic-macroinvertebrates in two Mediterranean-climate streams in California, USA. Freshwater Biology 51:56-75.

Bednarek, A. T., and D. D. Hart. 2005. Modifying dam operations to restore rivers: ecological responses to Tennessee river dam mitigation. Ecological Applications 15:997-1008.

Boulton, A. J. 2003. Parallels and contrasts in the effects of drought on stream macroinvertebrate assemblages. Freshwater Biology 48:1173-1185.

Boulton, A. J., and E. H. Stanley. 1995. Hyporheic processes during flooding and drying in a Sonoran Desert stream. II: Faunal dynamics. Archiv Fur Hydrobiologie 134:27-52.

Brosse, S., C. J. Arbuckle, and C. R. Townsend. 2003. Habitat scale and biodiversity: influence of catchment, stream reach and bedform scales on local invertebrate diversity. Biodiversity and Conservation 12:2057-2075.

BryantMason, A. In Progress. Spatial and temporal dynamics of stream water chemistry in a headwater catchment of Central Louisiana. MS Thesis. Louisiana State University, Baton Rouge, LA.

Cairns, J., and J. R. Pratt. 1993. A history of biological monitoring using benthic macroinvertebrates. In D. M. Rosenberg and V. H. Resh, editors. Freshwater biomonitoring and benthic macroinvertebrates. Chapman & Hall, New York. p 10-27.

Camargo, J. A., A. Alonso, and M. De La Puente. 2004. Multimetric assessment of nutrient enrichment in impounded rivers based on benthic macroinvertebrates. Environmental Monitoring and Assessment 96:223-249.

Chapman, L. J., K. R. Schneider, C. Apodaca, and C. A. Chapman. 2004. Respiratory ecology of macroinvertebrates in a swamp-river system of east Africa. Biotropica 36:572-585.

Cole, M. B., K. R. Russell, and T. J. Mabee. 2003. Relation of headwater macroinvertebrate communities to in-stream and adjacent stand characteristics in managed second-growth forests of the Oregon Coast Range mountains. Canadian Journal of Forest Research 33:1433-1443.

Connolly, N. M., M. R. Crossland, and R. G. Pearson. 2004. Effect of low DO on survival, emergence, and drift of tropical stream macroinvertebrates. Journal of the North American Benthological Society 23:251-270.

Cowell, B. C., A. H. Remley, and D. M. Lynch. 2004. Seasonal changes in the distribution and abundance of benthic invertebrates in six headwater streams in central Florida. Hydrobiologia 522:99-115.

Cummins, K. W. 1973. Trophic relations of aquatic insects. Annual Review of Entomology 18:183-206.

Cummins, K. W. 1974. Structure and function of stream ecosystems. Bioscience 24:631-641.

Cushing, C. E., K. W. Cummins, and G. W. Minshall, editors. 1995. Ecosystems of the world 22: river and stream ecosystems. Elsevier, Amsterdam. 817 p.

Davis, S., S. W. Golladay, G. Vellidis, and C. M. Pringle. 2003. Macroinvertebrate biomonitoring in intermittent coastal plain streams impacted by animal agriculture. Journal of Environmental Quality 32:1036-1043.

DeWalt, R. E. 1995. Biological communities of reference streams in the South Central Plains and Upper Mississippi Alluvial Plains Ecoregions of Louisiana. Louisiana Department of Environmental Quality Office of Water Resources, Baton Rouge, LA.

Dewson, Z. S., A. B. W. James, and R. G. Death. 2007. A review of the consequences of decreased flow for instream habitat and macroinvertebrates. Journal of the North American Benthological Society 26:401-415.

Dodds, W. K. 2002. Freshwater ecology: concepts and environmental applications. Academic Press, San Diego. 569 p.

Drury, D. M., and W. E. Kelso. 2000. Invertebrate colonization of woody debris in coastal plain streams. Hydrobiologia 434:63-72.

Edsall, T. A. 2001. Burrowing mayflies (Hexagenia) as indicators of ecosystem health. Aquatic Ecosystem Health and Management 4:283-292.

Edsall, T. A., M. T. Bur, and O. T. Gorman. 2005. Burrowing mayflies as indicators of ecosystem health: status of poplulations in Western Lake Erie, Saginaw Bay, and Green Bay. Aquatic Ecosystem Health and Management 8:106-116.

Eriksen, C. H., G. A. Lamberti, and V. H. Resh. 1996. Aquatic insect respiration. In R. W. Merritt and K. W. Cummins, editors. An introduction to the aquatic insects of North America. Kendall Hunt Publishing Dubuque. p 29-40.

Feminella, J. W. 1996. Comparison of benthic macroinvertebrate assemblages in small streams along a gradient of flow permanence. Journal of the North American Benthological Society 15:651-669.

Fore, L. S., J. R. Karr, and R. W. Wisseman. 1996. Assessing invertebrate responses to human activities: Evaluating alternative approaches. Journal of the North American Benthological Society 15:212-231.

Fritz, K. M., and W. K. Dodds. 2005. Harshness: characterisation of intermittent stream habitat over space and time. Marine and Freshwater Research 56:13-23.

Fukui, D., M. Murakami, S. Nakano, and T. Aoi. 2006. Effect of emergent aquatic insects on bat foraging in a riparian forest. Journal of Animal Ecology 75:1252-1258.

Gaufin, A. R., and C. M. Tarzwell. 1952. Aquatic invertebrates as indicators of stream pollution. Public Health Reports 67:57-64.

Gaufin, A. R., and C. M. Tarzwell. 1956. Aquatic macro-invertebrate communities as indicators of organic pollution in Lytle Creek. Sewage and Industrial Wastes 28:906-924.

Genito, D., W. J. Gburek, and A. N. Sharpley. 2002. Response of stream macroinvertebrates to agricultural land cover in a small watershed. Journal of Freshwater Ecology 17:109-119.

Gooderham, J. P. R., L. A. Barmuta, and P. E. Davies. 2007. Upstream heterogeneous zones: small stream systems structured by a lack of competence? Journal of The North American Benthological Society 26:365-374.

Goodnight, C. J. 1973. Use of aquatic macroinvertebrates as indicators of stream pollution. Transactions of the American Microscopical Society 92:1-13.

Haggerty, S. M., D. P. Batzer, and C. R. Jackson. 2002. Macroinvertebrate assemblages in perennial headwater streams of the Coastal Mountain range of Washington, USA. Hydrobiologia 479:143-154.

Hansen, E. A., and G. P. Closs. 2007. Temporal consistency in the long-term spatial distribution of macroinvertebrate drift along a stream reach. Hydrobiologia 575:361-371.

Heino, J., P. Louhi, and T. Muotka. 2004. Identifying the scales of variability in stream macroinvertebrate abundance, functional composition and assemblage structure. Freshwater Biology 49:1230-1239.

Heino, J., J. Parviainen, R. Paavola, M. Jehle, P. Louhi, and T. Muotka. 2005. Characterizing macroinvertebrate assemblage structure in relation to stream size and tributary position. Hydrobiologia 539:121-130.

Henschel, J. R., D. Mahsberg, and H. Stumpf. 2001. Allochthonous aquatic insects increase predation and decrease herbivory in river shore food webs. Oikos 93:429-438.

Herlihy, A. T., W. J. Gerth, J. Li, and J. L. Banks. 2005. Macroinvertebrate community response to natural and forest harvest gradients in western Oregon headwater streams. Freshwater Biology 50:905-919.

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

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

Horton, R. E. 1945. Erosional development of streams and their drainage basins: hydrophysical approach to quantatative morphology. Geological Society of America Bulletin 56:275-370.

Hughes, R. M., J. H. Gakstatter, M. A. Shirazi, and J. M. Omernik. 1982. An approach for determining biological integrity in flowing waters. In, In place resource inventories: principles and practices, A National Workshop. Paper presented at the workshop August 9-14, 1981. Society of American Foresters, Bethesda, Maryland. p 877-888.

Ice, G., and B. Sugden. 2003. Summer DO concentrations in forested streams of northern Louisiana. Southern Journal of Applied Forestry 27:92-99.

James, F. C., and C. E. McCulloch. 1990. Multivariate analysis in ecology and systematics: panacea or Pandora's Box. Annual Review of Ecology and Systematics 21:129-166.

Johnson, R. K., M. T. Furse, D. Hering, and L. Sandin. 2007. Ecological relationships between stream communities and spatial scale: implications for designing catchment-level monitoring programmes. Freshwater Biology 52:939-958.

Kaller, M. D., and W. E. Kelso. 2006. Effects of a small-scale clearing on habitat and macroinvertebrates of a coastal bottomland stream in Louisiana. Southwestern Naturalist 51:143-151.

Kaller, M. D., and W. E. Kelso. 2007. Association of macroinvertebrate assemblages with DO concentration and wood surface area in selected subtropical streams of the southeastern USA. Aquatic Ecology 41:95-110.

Karr, J. R. 1991. Biological Integrity: A long-neglected aspect of water-resource management. Ecological Applications 1:66-84.

Karr, J. R., and D. R. Dudley. 1981. Ecological perspective on water-quality goals. Environmental Management 5:55-68.

Karr, J. R., and C. O. Yoder. 2004. Biological assessment and criteria improve total maximum daily load decision making. Journal of Environmental Engineering-ASCE 130:594-604.

Kato, C., T. Iwata, S. Nakano, and D. Kishi. 2003. Dynamics of aquatic insect flux affects distribution of riparian web-building spiders. Oikos 103:113-120.

Kolkwitz, R., and M. Marsson. 1909. Ecology of animal saprobia. In L. E. Kemp, W. M. Ingram, and K. M. Mackenthum, editors. Biology of water pollution. Federal Water Pollution Control Administration, Washington, DC. p 85-95.

Kratzer, E. B., and D. P. Batzer. 2007. Spatial and temporal variation in aquatic macroinvertebrates in the Okefenokee Swamp, Georgia, USA. Wetlands 27:127-140.

Lazorchak, J. M., D. J. Klemm, and D. V. Peck, editors. 1998. Environmental monitoring and assessment program - surface waters: field operations and methods for measuring ecological condition of wadeable streams. EPA/620/R-94/004. U.S. Environmental Protection Agency, Washington, D.C.

LDEQ. 2001. Flat Creek watershed TMDL for oxygen demand Louisiana Department of Environmental Quality, Baton Rouge, LA.

LDEQ. 2004. State of Louisiana water quality management plan. Water quality inventory integrated report (section 305(b) and 303(d) reports). Louisiana Department of Environmental Quality, Baton Rouge, LA.

LDEQ. 2006. Developing nutrient criteria for Louisiana. Louisiana Department of Environmental Quality. Baton Rouge, LA.

Lenat, D. R. 1988. Water quality assessment of streams using a qualitative collection method for benthic macroinvertebrates. Journal of the North American Benthological Society 7:222-233.

Lenat, D. R. 1993. A biotic index for the Southeastern United States: derivation and list of tolerance values, with criteria for assigning water-quality ratings. Journal of the North American Benthological Society 12:279-290.

Magurran, A. E. 1988. Ecological diversity and its measurements. Princeton University Press, Princeton. 179 p.

Martel, N., M. A. Rodriguez, and P. Berube. 2007. Multi-scale analysis of responses of stream macrobenthos to forestry activities and environmental context. Freshwater Biology 52:85-97.

Mason, A., Y. J. Xu, P. Saksa, A. Viosca, J. M. Grace, J. Beebe, and R. Stich. 2007. Streamflow and nutrient dependence of temperature effects on DO in loworder forest streams. Pages 374-380 in Watershed Management to Meet Water Quality Standards and TMDLS (Total Maximum Daily Load). ASABE Publication Number 701P0207.

McDowell, D. M., and R. J. Naiman. 1986. Structure and function of a benthic invertebrate stream community as influenced by beaver (*Castor-canadensis*). Oecologia 68:481-489.

Merritt, R. W., and K. W. Cummins. 1996. An introduction to the aquatic insects of North America. 3rd ed. Kendall Hunt Publishing, Dubuque.

Meyer, J. L., D. L. Strayer, J. B. Wallace, S. L. Eggert, G. S. Helfman, and N. E. Leonard. 2007. The contribution of headwater streams to biodiversity in river networks. Journal of the American Water Resources Association 43:86-103.

Minshall, G. W. 1988. Stream ecosystem theory: a global perspective. Journal of the North American Benthological Society 7:263-288.

Morse, J. C., and C. B. Barr. 1990. Unusual caddisfly (Trichoptera) fauna of Schoolhouse-Springs, Louisiana, with description of a new species of *Diplectrona* (Hydrosychidae). Proceedings of the Entomological Society of Washington 92:58-65.

Nadeau, T. L., and M. C. Rains. 2007. Hydrological connectivity between headwater streams and downstream waters: How science can inform policy. Journal of the American Water Resources Association 43:118-133.

Nakashima, S., and Y. Yamada. 2005. Temporal-spatial distributions of high nitrogen concentrations in headwater areas of regions with low precipitation. Limnology 6:53-60.

NCDC. 2002. Climatography of the United States No. 81: 16 Louisiana. National Climatic Data Center, National Oceanic and Atmospheric Administration, Asheville, NC.

Ndaruga, A. M., G. G. Ndiritu, N. N. Gichuki, and W. N. Wamicha. 2004. Impact of water quality on macroinvertebrate assemblages along a tropical stream in Kenya. African Journal of Ecology 42:208-216.

Norris, R. H., and M. C. Thoms. 1999. What is river health? Freshwater Biology 41:197-209.

Omernik, J. M. 1987. Ecoregions of the conterminous United States. Annals of the Association of American Geographers 77:118-125.

Paller, M. H., W. L. Specht, and S. A. Dyer. 2006. Effects of stream size on taxa richness and other commonly used benthic bioassessment metrics. Hydrobiologia 568:309-316.

Palmer, M. A., R. F. Ambrose, and N. L. Poff. 1997. Ecological theory and community restoration ecology. Restoration Ecology 5:291-300.

Parr, L. B., and C. F. Mason. 2003. Long-term trends in water quality and their impact on macroinvertebrate assemblages in eutrophic lowland rivers. Water Research 37:2969-2979.

Patrick, R. 1953. Aquatic organisms as an aid in solving waste disposal problems. Sewage and Industrial Wastes 25:210-214.

Ramirez, A., C. M. Pringle, and M. Douglas. 2006. Temporal and spatial patterns in stream physicochemistry and insect assemblages in tropical lowland streams. Journal of the North American Benthological Society 25:108-125.

Rasmussen, J. B. 1988. Habitat requirements of burrowing mayflies (Ephemeridae: *Hexagenia*) in lakes, with special reference to the effects of eutrophication Journal of the North American Benthological Society 7:51-64.

Richards, C., R. J. Haro, L. B. Johnson, and G. E. Host. 1997. Catchment and reach-scale properties as indicators of macroinvertebrate species traits. Freshwater Biology 37:219-230.

Richardson, J. S., and R. J. Danehy. 2007. A synthesis of the ecology of headwater streams and their riparian zones in temperate forests. Forest Science 53:131-147.

Rosenberg, D. M., and V. H. Resh. 1993. Freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, New York.

Rosgen, D. L. 1994. A classification of natural rivers. Catena 22:169-199.

Roy, A. H., A. D. Rosemond, M. J. Paul, D. S. Leigh, and J. B. Wallace. 2003. Stream macroinvertebrate response to catchment urbanisation (Georgia, USA). Freshwater Biology 48:329-346.

Saksa, P. 2007. The hydrology and sediment transport of low-gradient, forested headwater streams. MS Thesis. Louisiana State University, Baton Rouge, LA.

Saksa, P., Y. J. Xu, A. Mason, and A. Viosca. 2007. The influence of forested headwater stream hydrology on target TMDLS.in Watershed Management to Meet Water Quality Standards and TMDLS (Total Maximum Daily Load) ASABE Publication Number 701P0207.

Seidman, V. M., and C. J. Zabel. 2001. Bat activity along intermittent streams in Northwestern California. Journal of Mammalogy 82:738-747.

Sloey, D. J. 1992. Effects of physicochemistry and forest management practices on benthic macroinvertebrates in small streams of Central Louisiana. MS Thesis. Louisiana State University, Baton Rouge, LA.

Smith, D. G. 2001. Pennak's freshwater invertebrates of the United States: Porifera to Crustacea. John Wiley & Sons, Inc New York.

Sokal, R. R., and F. J. Rohlf. 1995. Biometry: the principles and practice of statistics in biological research. 3rd ed. W.H. Freeman and Company, New York.

Sporka, F., H. E. Vlek, E. Bulankova, and I. Krno. 2006. Influence of seasonal variation on bioassessment of streams using macroinvertebrates. Hydrobiologia 566:543-555.

Stanley, E. H., D. L. Buschman, A. J. Boulton, N. B. Grimm, and S. G. Fisher. 1994. Invertebrate resistance and resilience to intermittency in a desert stream. American Midland Naturalist 131:288-300.

Stevens, J. 2001. Applied multivariate statistics for the social sciences. 4th ed. Lawrence Erlbaum Associates, Mahwah.

Stewart, K. W., B. P. Stark, and T. G. Higgins. 1976. The stoneflies (Plecoptera) of Louisiana. Great Basin Naturalist 36:366 - 384.

Thorp, J. H., and A. P. Covich (eds). 2001. Ecology and classification of North American Freshwater Invertebrates. 2nd ed. Academic Press, San Diego.

Townsend, C. R., S. Doledec, R. Norris, K. Peacock, and C. Arbuckle. 2003. The influence of scale and geography on relationships between stream community composition and landscape variables: description and prediction. Freshwater Biology 48:768-785.

USEPA. 2002. Summary of biological assessment programs and biocriteria development for states, tribes, territories, and interstate commissions: streams and wadeable rivers. EPA-822-R-02-048. U.S. Environmental Protection Agency.

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

Viosca, P. 1933. Louisiana Out-of-Doors: A handbook and guide. Author, New Orleans. 187 p.

Voshell, J. R. 2002. A guide to common freshwater invertebrates of North America. The McDonald & Woodward Publishing Company, Blacksburg.

Vowell, J. L. 2001. Using stream bioassessment to monitor best management practice effectiveness. Forest Ecology and Management 143:237-244.

Wallace, J. B., and J. R. Webster. 1996. The role of macroinvertebrates in stream ecosystem function. Annual Review of Entomology 41:115-139.

Wilhm, J. L., and T. C. Dorris. 1968. Biological parameters for water quality criteria. Bioscience 18:477-481.

Williams, L. R., T. H. Bonner, J. D. Hudson, M. G. Williams, T. R. Leavy, and C. S. Williams. 2005. Interactive effects of environmental variability and military training on stream biota of three headwater drainages in western Louisiana. Transactions of the American Fisheries Society 134:192-206.

Winter, A., J. J. H. Ciborowski, and T. B. Reynoldson. 1996. Effects of chronic hypoxia and reduced temperature on survival and growth of burrowing mayflies, *Hexagenia limbata* (Ephemeroptera: Ephemeridae). Canadian Journal of Fisheries and Aquatic Sciences 53:1565-1571.

Wipfli, M. S., J. S. Richardson, and R. J. Naiman. 2007. Ecological linkages between headwaters and downstream ecosystems: Transport of organic matter, invertebrates, and wood down headwater channels. Journal of the American Water Resources Association 43:72-85.

APPENDIX A

LIST OF MACROINVERTEBRATE TAXA

Spring 2006

	9D	9U	E1	E2	E3	I1	I2	I3	I4	I5	I6
Lumbricidae	1	2	20	17	0	0	0	2	0	5	0
Acariformes	0	0	0	0	0	0	0	0	0	0	0
Hydrachnidia	0	0	1	0	1	0	5	1	3	6	4
Amphipoda	2	7	14	24	2	1	13	29	4	17	56
Crangonyctidae *Synurella*	17	68	107	55	13	11	16	45	19	33	121
Hyalellidae *Hyalella*	0	0	0	0	1	0	78	11	15	99	39
Coleoptera	1	0	3	0	0	0	1	0	0	0	2
Chrysomelidae *Pyrrhalta*	0	0	1	0	0	0	0	0	0	0	0
Curculionidae	0	1	2	0	0	0	0	0	0	0	0
Dytiscidae *Coptotmus*	0	0	1	0	0	0	0	0	0	0	0
Dytiscidae *Derovatellus*	0	0	0	0	0	0	0	0	0	0	0
Dytiscidae *Desmopachria*	0	0	27	0	0	0	0	0	0	0	0
Dytiscidae *Oreodytes*	1	0	2	3	10	3	1	7	3	0	14
Dytiscidae Undetermined	5	0	4	4	1	1	3	2	1	0	9
Elmidae	0	0	0	0	1	0	1	0	0	1	0
Gyrinidae *Dineutus*	0	0	0	0	0	1	0	0	0	0	0
Halipidae *Peltodytes*	0	0	1	5	0	0	1	1	1	0	5
Halipidae	0	0	0	0	0	0	0	0	0	0	0

Hydrochidae *Hydrochus*	0	0	1	0	0	0	0	0	0	0	0
Hydrophiloidea	0	0	0	0	0	0	1	0	0	0	0
Ptiliidae	0	0	0	0	1	0	0	0	0	0	0
Salpingidae	0	0	0	0	0	0	0	0	0	0	0
Scirtidae *Scirtes*	0	0	0	0	0	0	0	0	0	0	7
Scirtidae Unidentified	0	0	0	0	0	0	0	0	0	0	19
Tenebrionidae	0	0	0	0	0	1	0	0	0	0	0
Collembola	1	1	1	0	2	0	1	1	1	3	6
Decopoda	0	0	0	0	0	0	0	0	0	0	0
Cambaridae	5	12	0	0	1	2	2	0	0	0	2
Palaemonidae *Palaemontetes*	0	0	0	1	0	4	1	3	0	5	0
Diptera	0	2	1	2	4	2	3	0	0	4	2
Ceratopogonidae *Serromyia*	0	0	0	0	0	0	126	0	0	0	0
Ceratopogonidae Undetermined	0	1	28	47	16	21	152	433	38	39	188
Chaoboridae *Eucorthra*	0	0	0	0	0	0	0	0	0	1	0
Chaoboridae Undetermined	0	0	0	0	0	0	0	0	0	0	1
Chironomidae	32	3	188	222	95	210	138	178	239	180	382
Corethrellidae *Corethrella*	0	0	0	0	0	0	1	0	0	0	0
Culicidae Undetermined	22	766	0	0	11	0	2	0	31	7	3
Culicidae *Aedes*	5	27	0	0	0	0	0	1	5	6	3
Culicidae *Anopheles*	0	0	0	0	0	0	9	0	3	0	1
Culicidae *Culex*	67	211	1	0	6	0	11	6	53	10	19

Dixidae	0	0	0	0	0	0	1	0	0	0	0
Psychodidae	0	0	0	0	0	1	0	0	0	0	0
Tabanidae	0	0	1	0	0	0	0	2	0	1	0
Tipulidae	1	4	0	0	1	9	0	0	1	2	1
Ephemeroptera	0	0	0	6	12	5	4	0	0	1	3
Baetidae	0	0	1	0	0	4	4	0	0	0	0
Caenidae *Caenis*	0	0	0	0	0	33	1	4	0	3	0
Caenidae Unidentified	0	0	0	0	0	0	14	0	0	0	0
Ephemerellidae *Eurylophella*	0	0	0	0	0	0	2	0	0	0	0
Ephemerellidae Undetermined	0	0	0	0	0	0	0	0	0	0	0
Ephemeridae *Hexagenia*	0	0	0	115	0	0	0	0	0	0	0
Ephemeridae Undetermined	0	0	1	1	2	1	0	0	0	0	0
Heptageniidae *Stenacron*	0	0	0	0	0	7	0	0	0	0	0
Hemiptera	0	0	0	0	0	0	0	0	0	0	0
Corixidae	2	0	14	11	20	0	26	4	9	2	4
Gerridae	0	0	0	0	0	0	0	0	0	0	0
Naucoridae	0	0	0	0	0	0	0	0	0	0	0
Pleidae *Neoplea*	0	0	0	0	0	0	0	0	0	0	0
Pleidae Undetermined	0	0	0	0	0	0	0	0	0	0	0
Saldidae	0	0	0	0	0	0	0	0	0	1	0
Homoptera	0	1	6	0	3	1	0	1	0	3	5
Isopoda	0	0	0	0	0	0	0	0	0	0	0

Asellidae	15	108	167	46	67	18	582	61	190	396	718
Lepidoptera	0	0	0	0	0	1	1	0	1	0	1
Cosmopterigidae	0	0	0	0	0	0	0	0	0	0	0
Pyralidae *Crambus*	0	0	0	0	1	0	0	0	0	0	0
Megaloptera	0	0	0	0	0	0	0	0	0	0	0
Corydalidae *Chauliodes*	0	0	0	0	0	0	0	0	0	0	0
Corydalidae Unidentified	0	0	0	0	0	0	0	0	0	0	0
Sialidae *Sialis*	0	0	0	0	0	0	1	5	5	5	2
Neuroptera	0	0	0	0	0	0	0	0	0	0	0
Sisyridae *Sisyra*	0	0	1	0	0	0	0	0	0	0	0
Sisyridae Undetermined	0	0	0	0	0	1	0	0	0	0	0
Odonata	0	0	1	0	1	0	0	0	0	0	0
Anisoptera	0	0	0	0	0	0	0	0	0	0	0
Corduliidae *Epitheca*	0	0	0	0	0	0	0	0	0	0	0
Corduliidae *Somatochlora*	0	0	0	0	0	0	0	1	0	0	0
Corduliidae Undetermined	0	0	0	0	0	0	0	0	0	0	0
Libellulidae *Libellula*	0	0	3	1	5	0	0	0	0	3	1
Libellulidae *Macrothemis*	0	0	0	0	0	0	0	0	0	1	2
Libellulidae *Miathyria*	0	0	0	0	0	0	0	1	0	0	0
Libellulidae *Pachydiplax*	0	0	0	0	0	0	0	0	0	1	24
Libellulidae *Perithemis*	0	0	0	0	0	0	0	0	0	0	0
Libellulidae Undetermined	0	0	0	0	0	0	0	0	0	0	2

Corduliidae/ Libellulidae	0	0	0	0	0	0	0	0	0	0	0
Gomphidae *Dromogomphus*	0	0	1	0	0	1	0	2	0	0	0
Gomphidae *Gomphus*	0	0	0	0	0	1	0	0	0	0	0
Gomphidae Undetermined	0	0	0	0	0	1	0	0	0	0	0
Zygoptera	0	0	0	0	0	0	0	0	0	0	0
Coenagrionidae *Amphiagrion*	0	0	0	0	0	0	0	3	0	0	0
Coenagrionidae Undetermined	0	0	0	0	0	0	0	0	0	0	1
Plecoptera	0	0	0	0	0	0	0	0	0	0	0
Perlidae *Perlesta*	0	0	1	2	1	1	0	0	0	0	0
Trichoptera	0	0	0	0	0	0	0	0	0	0	0
Limnephilidae	0	0	0	0	0	0	0	0	0	0	0
Branchiopoda	2	3	7	7	4	0	8	6	8	8	8
Copepoda	8	2	8	4	6	4	8	5	8	8	8
Gastopoda	0	0	0	0	0	0	0	0	0	4	0
Ancylidae	0	0	1	0	4	6	0	1	2	1	1
Physidae *Physa*	0	0	6	1	0	0	2	0	4	1	0
Physidae Undetermined	0	0	0	0	0	2	7	0	1	0	4
Planorbidae	0	1	2	0	0	2	0	1	0	1	3
Valvatidae *Valvata*	0	0	0	0	0	0	0	0	0	2	0
Viviparidae *Campeloma*	0	0	0	0	0	4	32	0	0	0	0
Viviparidae *Viviparus*	0	0	0	0	0	26	4	0	0	0	0
Viviparidae Undetermined	0	0	0	0	0	1	0	0	0	0	0

Hirudinea	0	0	4	10	0	0	5	13	2	45	35
Nematomorpha	P	P	P	P	P	P	P	P	P	P	P
Oligochaeta	P	P	P	P	P	P	P	P	P	P	P
Pelecypoda	0	0	6	20	55	1	47	7	6	16	2
Unionoida	0	0	0	0	0	0	0	0	0	0	0
Unionidae	0	0	0	1	0	1	0	3	1	0	0
Veneroida	0	0	0	0	0	0	0	0	0	0	0
Sphaeriidae *Pisidium*	0	0	0	0	0	0	7	0	0	0	0
Sphaeriidae Undetermined	3	3	17	9	18	5	5	12	7	11	15

A = absent
P = present

Fall 2006

	9D	9U	E1	E2	E3	I1	I2	I3	I4	I5	I6
Lumbricidae	.	.	0	0	1	0	0	0	0	0	0
Acariformes	.	.	0	0	0	0	0	0	0	0	0
Hydrachnidia	.	.	32	20	0	8	2	72	1	10	43
Amphipoda	.	.	0	0	0	0	1	0	1	0	4
Crangonyctidae *Synurella*	.	.	1	0	0	0	3	0	0	0	1
Hyalellidae *Hyalella*	.	.	0	0	0	0	10	3	3	2	6
Coleoptera	.	.	0	0	0	0	2	6	0	0	3
Chrysomelidae *Pyrrhalta*	.	.	0	0	0	0	0	0	0	0	0
Curculionidae	.	.	0	0	0	0	0	0	1	0	0
Dytiscidae *Coptotmus*	.	.	0	0	0	0	0	0	0	0	0
Dytiscidae *Derovatellus*	.	.	0	0	0	0	4	0	0	0	0
Dytiscidae *Desmopachria*	.	.	0	0	0	0	0	0	0	0	0
Dytiscidae *Oreodytes*	.	.	0	0	0	0	0	0	0	0	0
Dytiscidae Undetermined	.	.	4	0	23	0	0	0	1	0	3
Elmidae	.	.	0	0	0	0	1	0	0	0	0
Gyrinidae *Dineutus*	.	.	0	0	0	0	0	0	0	1	0
Halipidae *Peltodytes*	.	.	1	0	0	0	0	0	0	0	1
Halipidae	.	.	0	0	0	0	0	0	0	0	0
Hydrochidae *Hydrochus*	.	.	0	0	0	0	0	0	0	1	3

Hydrophiloidea	.	.	0	0	0	0	0	0	0	1	0
Ptiliidae	.	.	0	0	0	0	0	0	0	0	0
Salpingidae	.	.	0	0	0	0	0	0	0	0	0
Scirtidae *Scirtes*	.	.	0	0	0	0	0	0	0	0	0
Scirtidae Unidentified	.	.	0	0	0	0	6	1	0	5	5
Tenebrionidae	.	.	0	0	0	0	0	0	0	0	0
Collembola	.	.	0	1	0	0	2	0	1	0	3
Decopoda	.	.	0	0	0	0	0	0	0	0	0
Cambaridae	.	.	1	3	6	2	1	0	0	0	0
Palaemonidae *Palaemontetes*	.	.	10	6	5	1	0	7	4	1	0
Diptera	.	.	0	0	0	0	2	12	2	0	1
Ceratopogonidae *Serromyia*	.	.	0	0	0	0	0	0	8	0	0
Ceratopogonidae Undetermined	.	.	24	81	38	6	568	1606	204	111	448
Chaoboridae *Eucorthra*	.	.	0	0	0	0	0	0	0	0	0
Chaoboridae Undetermined	.	.	1	44	106	0	0	2	222	7	42
Chironomidae	.	.	1302	306	539	18	1141	1546	1844	2539	3547
Corethrellidae *Corethrella*	.	.	0	0	0	0	0	0	0	1	0
Culicidae Undetermined	.	.	0	0	1	0	2	0	0	1	2
Culicidae *Aedes*	.	.	0	0	0	0	0	0	0	0	0
Culicidae *Anopheles*	.	.	0	0	0	1	0	0	0	0	0
Culicidae *Culex*	.	.	1	0	0	0	0	0	0	4	8
Dixidae	.	.	0	0	0	0	0	0	0	0	0

Psychodidae	.	.	0	0	0	0	0	0	0	0	0
Tabanidae	.	.	15	0	0	0	0	6	0	0	3
Tipulidae	.	.	0	0	2	0	1	0	0	0	1
Ephemeroptera	.	.	0	0	0	0	2	1	0	0	0
Baetidae	.	.	0	0	0	0	0	0	0	0	0
Caenidae *Caenis*	.	.	0	0	0	0	0	0	0	0	0
Caenidae Unidentified	.	.	24	25	0	1	0	0	0	0	0
Ephemerellidae *Eurylophella*	.	.	0	0	0	0	0	0	0	0	0
Ephemerellidae Undetermined	.	.	0	0	0	0	0	0	0	0	0
Ephemeridae *Hexagenia*	.	.	0	5	0	0	0	0	0	0	0
Ephemeridae Undetermined	.	.	0	0	0	0	0	0	0	0	0
Heptageniidae *Stenacron*	.	.	0	0	0	0	0	0	0	0	0
Hemiptera	.	.	0	0	0	0	0	1	0	0	4
Corixidae	.	.	23	41	10	1	25	4	6	1	6
Gerridae	.	.	0	0	3	1	0	0	0	2	0
Naucoridae	.	.	0	0	0	0	1	1	0	0	0
Pleidae *Neoplea*	.	.	0	0	0	0	0	0	0	0	5
Pleidae Undetermined	.	.	0	0	0	0	0	0	0	1	4
Saldidae	.	.	0	0	0	0	1	0	0	0	0
Homoptera	.	.	6	2	3	4	0	1	0	1	0
Isopoda	.	.	0	0	0	0	0	0	0	0	0
Asellidae	.	.	16	0	6	0	3	0	0	1	2

Lepidoptera	.	.	1	0	0	0	0	0	0	0	1
Cosmopterigidae	.	.	0	0	0	0	0	0	0	1	0
Pyralidae *Crambus*	.	.	0	0	0	0	0	0	0	0	0
Megaloptera	.	.	0	0	0	0	0	0	0	0	0
Corydalidae *Chauliodes*	.	.	0	0	0	0	0	0	0	0	1
Corydalidae Unidentified	.	.	0	0	0	0	0	0	0	0	0
Sialidae *Sialis*	.	.	33	8	0	1	11	18	51	56	20
Neuroptera	.	.	0	0	0	0	0	0	0	0	0
Sisyridae *Sisyra*	.	.	0	0	0	0	0	0	0	0	0
Sisyridae Undetermined	.	.	0	0	0	0	0	0	0	0	1
Odonata	.	.	0	0	0	0	0	1	0	0	0
Anisoptera	.	.	0	0	0	0	0	0	0	0	3
Corduliidae *Epitheca*	.	.	4	0	0	0	0	0	0	0	0
Corduliidae *Somatochlora*	.	.	0	0	0	0	0	0	0	0	0
Corduliidae Undetermined	.	.	0	0	0	0	1	0	0	0	0
Libellulidae *Libellula*	.	.	6	0	0	0	8	0	3	6	0
Libellulidae *Macrothemis*	.	.	0	0	0	0	0	0	0	0	0
Libellulidae *Miathyria*	.	.	1	0	0	0	0	0	0	0	0
Libellulidae *Pachydiplax*	.	.	6	0	0	0	19	35	5	8	3
Libellulidae *Perithemis*	.	.	0	0	0	0	0	1	0	5	7
Libellulidae Undetermined	.	.	4	0	0	0	0	12	0	4	6
Corduliidae/ Libellulidae	.	.	4	1	0	0	0	0	0	3	0

Gomphidae *Dromogomphus*	.	.	0	0	0	0	0	0	0	0	0
Gomphidae *Gomphus*	.	.	2	0	0	0	0	0	0	0	0
Gomphidae Undetermined	.	.	4	0	0	0	0	0	1	0	0
Zygoptera	.	.	0	0	0	0	1	0	0	0	2
Coenagrionidae *Amphiagrion*	.	.	0	0	0	0	0	0	0	0	0
Coenagrionidae Undetermined	.	.	0	0	0	0	3	0	0	0	2
Plecoptera	.	.	0	0	0	0	0	0	0	0	1
Perlidae *Perlesta*	.	.	0	0	0	0	0	0	0	0	0
Trichoptera	.	.	0	0	0	0	0	0	0	0	0
Limnephilidae	.	.	1	0	0	0	0	0	0	0	0
Branchiopoda	.	.	6	0	2	0	2	3	4	7	0
Copepoda	.	.	4	3	0	0	3	4	3	3	6
Gastopoda	.	.	0	12	0	0	1	0	0	0	0
Ancylidae	.	.	35	0	9	2	6	2	5	16	2
Physidae *Physa*	.	.	0	0	0	0	0	0	0	0	0
Physidae Undetermined	.	.	0	0	0	0	1	0	0	1	4
Planorbidae	.	.	5	0	1	3	0	1	1	8	2
Valvatidae *Valvata*	.	.	0	0	0	0	0	0	0	0	0
Viviparidae *Campeloma*	.	.	0	0	0	0	0	0	0	0	0
Viviparidae *Viviparus*	.	.	0	0	0	0	0	0	0	0	0
Viviparidae Undetermined	.	.	0	0	0	6	6	0	0	0	0
Hirudinea	.	.	8	0	0	0	7	0	2	5	9

Nematomorpha	.	.	A	A	A	P	P	P	P	P	P
Oligochaeta	.	.	P	A	P	P	P	P	P	P	P
Pelecypoda	.	.	24	7	16	0	8	0	4	6	8
Unionoida	.	.	0	0	0	0	0	0	0	0	0
Unionidae	.	.	0	0	0	0	0	0	0	0	0
Veneroida	.	.	0	0	0	0	0	0	0	0	0
Sphaeriidae *Pisidium*	.	.	0	0	0	0	0	0	0	0	0
Sphaeriidae Undetermined	.	.	0	0	1	7	0	0	1	0	0

A = absent
P = present

APPENDIX B

LIST OF MACROINVERTEBRATE METRICS

	Site													
Spring 2006 Metrics	9D	9U	E1	E2	E3	I1	I2	I3	I4	I5	I6	Average	Min	Max
Total Abundance	180	301	636	602	355	390	1311	841	645	906	1707	716	180	1707
Taxa Richness	10	7	24	15	18	24	20	22	16	22	20	18	7	24
Shannon-Weiner Diversity	1.5	0.6	1.9	1.8	2.0	1.8	1.7	1.5	1.7	1.6	1.7	1.6	0.6	2.0
Simpon's Index (1-D)	0.7	0.2	0.8	0.8	0.8	0.7	0.7	0.6	0.7	0.7	0.7	0.7	0.24	0.82
Evenness	0.6	0.3	0.6	0.7	0.7	0.6	0.6	0.5	0.6	0.5	0.6	0.6	0.3	0.7
Margalef Diversity	1.7	1.1	3.6	2.2	2.9	3.9	2.7	3.1	2.3	3.1	2.6	2.7	1.1	3.9
Percent Dominant	38.3	40.5	30.6	38.4	27.2	54.1	45.1	53.3	37.3	45.2	43.6	41.2	27.2	54.1
Percent EPT	0.0	0.0	0.5	20.6	4.2	13.1	1.9	0.5	0.0	0.4	0.2	3.8	0.0	20.6
Percent Odonata	0.0	0.0	0.8	0.2	1.7	0.8	0.0	0.8	0.0	0.6	1.8	0.6	0.0	1.8
Percent Diptera	70.6	87.0	34.4	45.0	37.5	62.3	33.8	73.7	57.4	27.7	35.1	51.3	27.7	87.0
Percent Chironomidae	13.3	0.0	29.6	35.2	26.8	53.8	10.5	21.2	37.1	20.0	22.4	24.5	0.0	53.8
Percent Crustacea	21.7	14.0	45.3	20.9	23.4	7.7	52.8	17.7	35.3	44.6	54.2	30.7	7.7	54.2
Percent Isopoda	8.3	3.7	26.3	7.6	18.9	4.6	44.4	7.3	29.5	43.7	42.1	21.5	3.7	44.4
Percent Amphipoda	8.3	6.6	19.0	13.1	4.5	3.1	8.2	10.1	5.9	14.7	12.7	9.7	3.1	19.0
Percent Gastropoda	0.0	0.0	1.4	0.2	1.1	10.5	3.4	0.2	1.1	1.0	0.5	1.8	0.0	10.5
Percent Pelecypoda	1.7	0.7	3.6	5.0	20.6	1.8	4.0	2.6	2.2	3.0	1.0	4.2	0.7	20.6

Fall 2006 Metrics	Site													
	9D	9U	E1	E2	E3	I1	I2	I3	I4	I5	I6	Average	Min	Max
Total Abundance	-	-	2064	562	773	62	1850	3339	2372	2813	4222	2006	62	4222
Taxa Richness	-	-	22	11	15	14	21	14	15	23	27	18	11	27
Shannon-Weiner Diversity	-	-	1.0	1.5	1.1	2.1	1.0	0.9	0.8	0.5	0.6	1.1	0.5	2.1
Simpon’s Index (1-D)	-	-	0.4	0.6	0.5	0.8	0.5	0.5	0.4	0.2	0.3	0.5	0.17	0.84
Evenness	-	-	0.3	1	0	0.8	0	0	0.3	0.2	0.2	0.4	0.2	0.8
Margalef Diversity	-	-	2.8	2	2	2.1	3	2	1.8	2.8	3.1	2.3	1.6	3.1
Percent Dominant	-	-	62.5	54.6	70.0	29.0	61.8	48.3	77.8	90.4	84.3	64.3	29.0	90.4
Percent EPT	-	-	1.4	5.3	0.0	1.6	0.1	0.0	0.0	0.0	0.0	0.9	0.0	5.3
Percent Odonata	-	-	1.8	0.2	0.0	0.0	1.7	1.5	0.4	0.9	0.5	0.8	0.0	1.8
Percent Diptera	-	-	85.4	76.7	88.7	40.3	92.6	95.0	96.1	94.7	95.9	85.1	40.3	96.1
Percent Chironomidae	-	-	78.6	54.4	69.7	32.3	61.7	46.3	77.7	85.3	84.0	65.6	32.3	85.3
Percent Crustacea	-	-	1.4	1.6	2.2	4.8	1.0	0.3	0.1	0.1	0.3	1.3	0.1	4.8
Percent Isopoda	-	-	0.8	0.0	0.8	0.0	0.2	0.0	0.0	0.0	0.0	0.2	0.0	0.8
Percent Amphipoda	-	-	0.0	0.0	0.0	0.0	0.8	0.1	0.2	0.1	0.3	0.2	0.0	0.8
Percent Gastropoda	-	-	1.9	2.1	1.3	17.7	0.8	0.1	0.3	0.9	0.2	2.8	0.1	17.7
Percent Pelecypoda	-	-	1.3	1.2	2.2	11.3	0.4	0.0	0.2	0.2	0.2	1.9	0.0	11.3

VITA

Adrienne was born and raised in the New Orleans area. She received her Bachelor of Science in conservation biology from the College of Santa Fe and subsequently began a career as a field biologist in the Southwest. While working on a native fish project in Arizona, she discovered her interest in freshwater ecology and subsequently worked with the state of Colorado conducting stream assessments. After spending eight years away from Louisiana, she came to Baton Rouge to pursue a master's degree in natural resources. She plans to continue working on water quality related issues in Louisiana.