AQUATIC MACROINVERTEBRATE ASSEMBLAGES IN SOUTHWEST GEORGIA HEADWATER STREAMS

By

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This document is dedicated to my parents.

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AQUATIC MACROINVERTEBRATE ASSEMBLAGES IN SOUTHWEST
GEORGIA HEADWATER STREAMS

By

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Headwater streams account for a significant portion of channel length in a stream network and strongly influence hydrological, water quality, and biological attributes downstream. Little biological monitoring or assessment has been conducted in headwater watersheds, especially in the Southeast coastal plain. Biological assessments must have a standard, or reference condition, against which potentially impacted sites can be compared. The objective of this study was to compare aquatic macroinvertebrate assemblages in four headwater streams as part of the Dry Creek Long-term Watershed Study being conducted by multiple partners. Four headwater streams (designated A, B, C, D) in the Dry Creek watershed of the Southlands Forest of International Paper were selected for this study. Benthic macroinvertebrates were sampled in streams during December 2001, February and December 2002, and February 2003 within fixed distance sample reaches. Macroinvertebrates were identified to the lowest taxonomic level and results were used in biotic indices. Data analysis included using repeated measures

ANOVA to identify differences in macroinvertebrate assemblages due to sampling period, position (upstream vs. downstream), and between streams. Stepwise regressions were used to correlate differences in hydrology and water chemistry to relate with stream differences. ANOVA results for abundance, total taxa, Ephemeroptera Plecoptera Trichoptera (EPT) taxa, Georgia Adopt-A-Stream (AAS) index indicated differences in macroinvertebrate assemblages due to sampling period, with lower values for December 2001 relative to February 2003. Abundance, total taxa, EPT taxa, Georgia Environmental Protection Division (EPD) index, Georgia AAS index, and percent Elmidae displayed significant differences due to stream with comparisons between streams for EPT taxa, Georgia EPD index, and Georgia AAS index resulting in stream A having significantly lower values than stream C. Significant predictors in regressions were average daily flow and specific conductance for selected macroinvertebrate metrics. Natural variability in hydrology, interannual and stream to stream was significantly different even within sub-watersheds of a small catchment, which suggested that hydrology is an important environmental factor influencing stream ecology and should be considered in macroinvertebrate studies. Of all metrics examined in this study, abundance, EPT taxa, total taxa, GA AAS index, and GA EPD index detected differences in macroinvertebrates due to time and stream, and therefore best described differences in the macroinvertebrate assemblage. Differences in the macroinvertebrate assemblages between streams A and C, but not between A and B or C and D, support the overall Dry Creek Long-term Watershed Study design and suggest that A and D would be appropriate reference streams for B and C, respectively.

CHAPTER 1
INTRODUCTION AND OBJECTIVES

Introduction

Headwater Streams

First or second order streams comprise approximately 95% of all streams and represent 73% of total channel length in North America (Leopold et al. 1992). For example, the Chattooga River watershed is characterized by 59% of total stream length being first order streams (Hansen 2001). Guidelines for protecting water quality from anthropogenic activities are usually applied to streams designated as perennial ('blue line') or intermittent (dashed line) on United States Geological Survey topographic maps. For the Chattooga River watershed, only approximately 20% of total stream length (1st-7th order) was represented as perennial streams on 1:24,000 topographic maps with essentially none of the intermittent or ephemeral streams identified (Hansen 2001). When compared with larger aquatic systems, the small size but large numbers of headwater streams have led to underestimation of their functions within a watershed and subsequently inadequate management (Gomi et al. 2002). To ensure adequate protection of water quality and aquatic habitats through the use of Best Management Practices for various land management activities, land managers must recognize the location and importance of headwater streams (Hansen, 2001).

The "edge" to "interior," or perimeter to area ratio (P/A), influences the importance of individual input to a habitat (e.g., watershed to stream) whether the input is natural (e.g., litterfall) or anthropogenic (e.g., nutrients from fertilizer). Headwater streams have

a high P/A ratio compared to larger rivers, and as such, are more influenced by their stream-land interface (Polis et al. 1997). Recognition of this connection between streams and the surrounding landscape (Hynes 1975, Vannote et al. 1980) and between headwater streams and downstream systems (Vannote et al. 1980) has guided recent studies related to headwater streams and their functions.

Headwater streams have many functions important to downstream systems (Meyer et al. 2003). Such streams are often strongly influenced by riparian vegetation, which contributes allochthonous detritus and limits autochthonous primary production by shading (Vannote et al. 1980). Wallace et al. (1997) showed through a leaf litter exclusion study that terrestrial-aquatic linkages in headwater streams influenced diversity and productivity. Headwater streams store, transform (Webster et al. 1999), and export (Wallace et al. 1991) organic matter and nutrients. Invertebrates and diatoms (Allan 1995) are an important energy source for downstream ecosystems. For example, in forested headwater streams of southeastern Alaska, invertebrates and coarse organic detritus are exported downstream year-round (Wipfli and Gregovich 2002). These systems also retain (Dieterich and Anderson 1998) and export sediment (Zimmerman and Church 2001). Headwater streams maintain streamflow by supplying a stable source of water to downstream systems through outflows from hillslopes, channel storage, riparian wetlands (Gomi et al. 2002) and groundwater recharge (Meyer et al. 2003).

Aquatic Fauna

The southeastern United States harbors a rich and diverse aquatic fauna that is threatened by development, habitat fragmentation, chemical pollution, and exotic species introductions. The Southeast contains approximately 40% of the aquatic insect species found in North America (Morse et al. 1997); however, the rich fauna in the Southeast is

poorly known, especially invertebrates (Folkerts 1997). Diversity of aquatic invertebrates is high in the Gulf Coastal plain (Felley 1992). However, in the Apalachicola-Chattahoochee-Flint (ACF) river basin, there is limited information on the number and distribution of invertebrate species, except for checklists of specific taxa in select portions of the basin (Couch et al. 1996). Due to their geographical isolation, headwater streams may support species genetically isolated from those downstream (Gomi, et al. 2002), so these systems may be important for maintaining local and regional biodiversity.

Spatial significance, connection to the landscape, and many functions that maintain downstream ecosystems highlight the need for monitoring and assessing headwater streams. A great deal of research on macroinvertebrates and headwater streams has been conducted in Montane regions, such as the southern Appalachians at the U. S. Forest Service's Coweeta Hydrologic Laboratory in North Carolina, and to a lesser extent in the White Mountains at the Hubbard Brook Experimental Forest in New Hampshire (Stone and Wallace 1998, Whiles and Wallace 1995, Noel et al. 1986). Research has been conducted in lower gradient Coastal Plain systems, including in fourth-order streams and two rivers in Georgia (Benke et al. 1984, Wallace and Benke 1984) and in low order streams of southeastern Virginia (e.g., Kedzierski and Smock 2001, Wright and Smock 2001, Smock et al. 1989). Generally, researchers and government agencies have given little attention to wadeable streams of the coastal plain (Maxted et al. 2000). Therefore, additional information is needed to build on the work previously conducted in the ACF river basin (Muenz 2004, Davis et al. 2003, Gregory 1996) and further characterize the unique aquatic macroinvertebrate assemblages of headwater streams.

Benthic Macroinvertebrates

Benthic macroinvertebrates are often used in biological assessments because: 1) they are found in many types of aquatic habitats, 2) the variety of species that can be monitored offer a range of responses to environmental changes, 3) they do not migrate widely compared to other groups like fish, so they indicate conditions (Barbour et al. 1999), 4) their long life cycles allow temporal assessments, and 5) individual species' tolerance to pollution have been established (Rosenberg and Resh, 1993). As a result, benthic macroinvertebrates are well suited for continuous monitoring of streams, which enables analysis of continuous and intermittent discharges, single or multiple pollutants, and cumulative effects of pollutants (Rosenberg and Resh, 1993). However, using macroinvertebrates in bioassessment also has a number of potential disadvantages: 1) macroinvertebrates do not respond to all impacts, 2) they can be affected by natural stressors and disturbances such as drought (Feminella 1996), 3) they display seasonal variation (Linke et al. 1999), which can present constraints for timing of sampling and comparing samples, and 4) their drift behavior can be problematic if the intent is to detect localized pollution effects (Rosenberg and Resh 1993). This spatial and temporal variability must be accounted for in the sampling design (Hershey and Lamberti 2001).

Analyzing macroinvertebrates for bioassessment can present challenges. Taxonomy for some groups such as the Chironomidae and Oligochaeta requires specialized training. Also, quantitative sampling requires high numbers of samples for precision, and sample processing and identification are time consuming and expensive, although rapid bioassessment methods can reduce these concerns (Rosenberg and Resh, 1993).

Biomonitoring

In the late 1980's, several state water resource monitoring programs were combined and expanded by the United States Environmental Protection Agency (USEPA) and state biologists to create Rapid Bioassessment Protocols (RBPs) as cost-effective biological survey techniques (Barbour et al. 1999). RBPs use an integrated approach to assess waterbody condition by comparing biotic, water quality, and habitat measures with reference conditions. The latter can be empirically defined through historical data, modeling/extrapolation, and/or actual reference sites, but it is best determined by monitoring sites that represent natural ranges of variation, i.e. minimally disturbed with respect to biological condition, water quality, and habitat (Gibson et al. 1996 as cited in Barbour et al. 1999). As a result, biomonitoring programs have attempted to describe reference conditions from a wide range of sites rather than relying on one or two reference sites that could only be used for site specific comparisons.

Field experiments in biomonitoring typically measure abundances and/or other characteristics of macroinvertebrates at different sites or times, each with associated environmental conditions. Detected differences in macroinvertebrates are attributed to differences in environmental conditions of that site or sample time (Cooper and Barmuta, 2000). Biomonitoring programs and field experiments in biomonitoring must have a standard, or reference condition, by which other sites are evaluated. This provides a baseline (Karr and Chu 1999) for comparison. Reynoldson et al. (1997) defines reference condition as "the condition that is representative of a group of minimally disturbed sites organized by selected physical, chemical, and biological characteristics." A challenge in biological assessment is finding separate sites with similar physical, chemical and biological characteristics to serve as a reference. Field studies have sought to decrease

this variability by comparing an upstream site to an impacted downstream area (Kedzierski and Smock 2001), comparing nearby streams draining watersheds with different land use or treatments (Davis et al. 2003, Richards and Minshall 1992, Gurtz and Wallace 1984), comparing before and after a disturbance, or a combination of before and after disturbance with watershed comparisons (Stone and Wallace 1998, Wallace et al. 1996). Other studies have combined two of these approaches into a Before-After-Control-Impact (BACI) design (Rosario and Resh, 2000).

Pre-treatment or baseline data are essential for characterizing responses to management (Hershey and Lamberti 2001, Karr and Chu 1999, Reynoldson et al. 1997), and robust baseline data can lessen the potential of natural variation obscuring identification of a treatment effect. In this study, a two-year pre-treatment characterization of benthic macroinvertebrates in adjacent watersheds and lower and upper reaches of four streams in the Dry Creek watershed of southwestern Georgia was undertaken to determine natural variation in macroinvertebrate assemblages.

Objectives

The objective of this research was to compare aquatic macroinvertebrate assemblages in four headwater streams prior to an experimental evaluation of forestry best management practices.

Major questions that were addressed included:

1. What are the macroinvertebrate assemblages in eight sample reaches (two per study watershed)?
2. Are the assemblages (upstream vs. downstream, between watersheds, across time) similar in population attributes, richness, composition, and functional feeding group composition?
3. If assemblages in reaches are not similar, what are the primary environmental differences between reaches that may be responsible?

4. How do different state assessment protocols score the biological condition of the eight sample reaches?

CHAPTER 2
SITE DESCRIPTION

Study Site

The study was located in southwestern Georgia approximately 16 km south of Bainbridge in the Coastal Plain physiographic province (Figure 2-1) specifically on the boundary of two physiographic districts, the Tifton Upland and Dougherty Plain (Figure 2-2). The steeply sloping Pelham Escarpment also forms the boundary or surface-water divide between the Flint River basin to the west and the Ochlockonee River basin to the east (Couch et al. 1996). Streams originating from the Pelham Escarpment are characterized by perennial headwaters that downstream become intermittent or drain directly into the Flint River. This transitional area is characterized by bluffs and deep ravines that create cool microclimates supporting rare plant species with northern affinities (Wharton 1978 as cited in Entrekin et al. 1999).

Figure 2-1. Location of study site in relation to physiographic regions (modified from US Forest Service 1969).

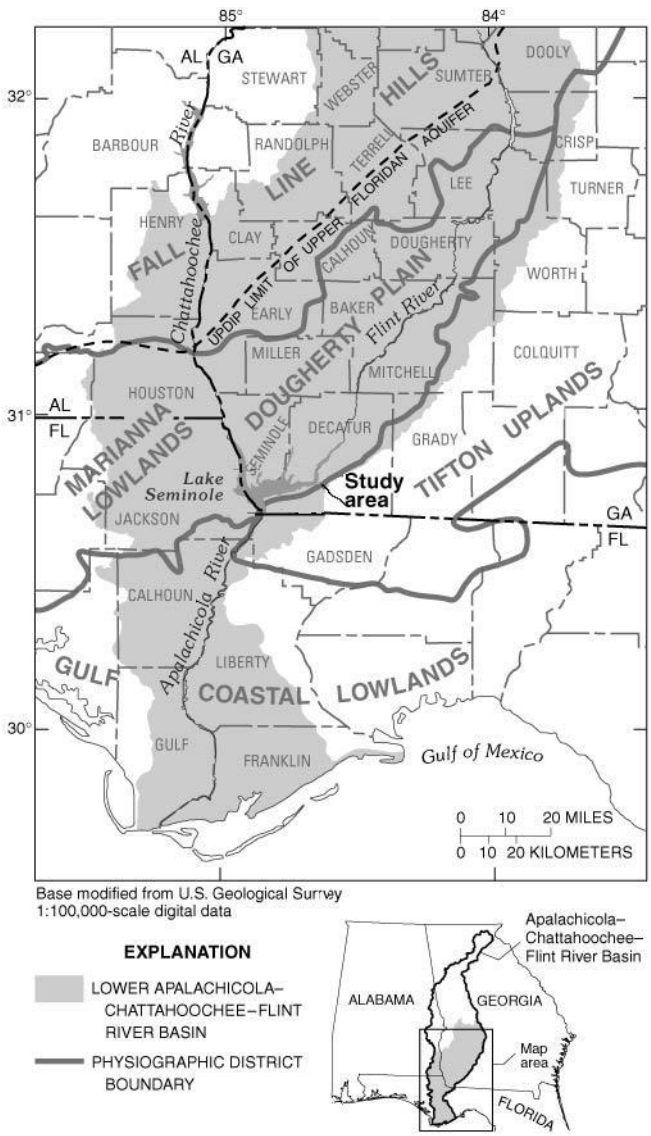

Figure 2-2. Location of study area in relation to physiographic districts (modified from Torak 2003).

The study site is located in the Dry Creek watershed, which discharges to the Flint River approximately 22.5 km up from the Jim Woodruff Dam of Lake Seminole. The Flint River is part of the larger lower Apalachicola-Chattahoochee-Flint River (ACF) Basin. Late Eocene Ocala Limestone extends throughout this 17,600 km^2 river basin. Oligocene Suwannee Limestone extends 26 km up the Flint River impoundment arm (Torak 2003). Soils of the study sites are dominated by Ultisols, with the riparian area

comprised of the Chiefland and Esto series, classified as well drained fine sands over clay loams. The slopes are Eustis series soils, which are loamy sands over sandy loams and classified as somewhat excessively well drained. The upland soils are comprised of Wagram, Norfolk, Lakeland, Orangeburg, and Lucy, which are generally well drained loamy sands over sandy clay loams, with the exception of the Lakeland Unit, which has a sandy texture throughout and is characterized as excessively well drained (International Paper 1980).

Streams draining the four study watersheds, A, B, C, and D (Figure 2-3) comprise part of the headwaters of Dry Creek.

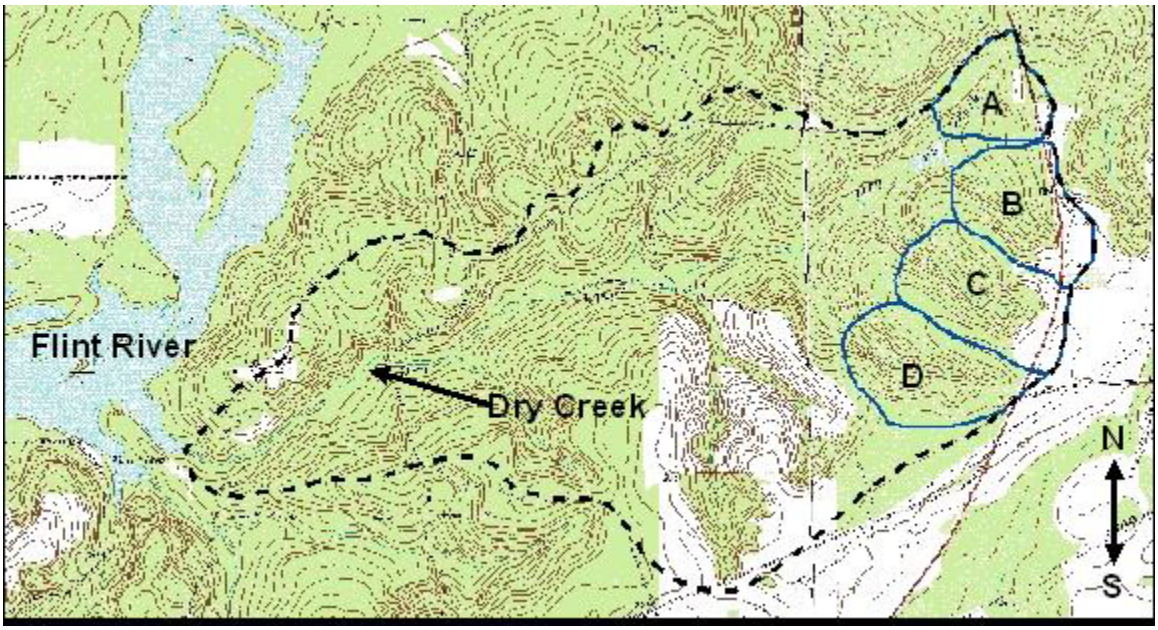

Figure 2-3. Topographic map of Dry Creek watershed and location of four headwater watersheds (A-D).

Surface water flow in the ACF basin is lowest from September to November and peaks during January to April due to higher rainfall and decreased evapotranspiration (Couch et al. 1996). Streams and rivers in the Coastal Plain receive substantial amounts of groundwater because they are typically deeply incised into underlying aquifers (Couch

et al. 1996). The study streams are first order, groundwater-influenced, low to medium gradient, and have sand-dominated substrate. In-stream habitat includes coarse woody debris, undercut banks, leaf packs, and fine roots. The four study watersheds average 39 ha, 1.5 L/s in average annual discharge, and 457 m in channel length (Summer et al. 2003 and Summer unpublished data). Watersheds A and B have gentle slopes and broader, meandering channels, whereas the remaining watersheds, C and D, have steeper slopes with well defined stream channels.

Vegetation

The overstory, midstory, and understory vegetation in riparian, midslope, and upslope areas of watersheds A, B, C, and D are generally similar with a few exceptions. The species dominating the overstory in riparian areas were: Nyssa biflora, Liriodendron tulipifera, Pinus glabra, Magnolia virginiana, Fagus grandifolia, Liquidambar styraciflua, Quercus nigra, and Quercus michauxii. Magnolia grandiflora was found more frequently in watersheds C and D (International Paper unpublished data). The upland of each watershed was dominated by Pinus taeda, which was established at varying times by hand planting. The midstory of all watersheds was generally composed of Carpinus caroliniana, Ostrya virginiana, Acer rubrum, Acer barbatum, and Oxydendrum arboretum. Magnolia pyramidata occurred in riparian areas and midslopes of watersheds C and D. The understory composition and coverage varied from watersheds A and B dominated by riparian wetland species to watersheds C and D with understory similar to that of watersheds A and B, but less abundant. This is likely due to the mixed mesic hardwood forest type and drier soil conditions of watersheds C and D. Typical shrub species of the understory were Ilex coriacea, Myrica cerifera, Rhododendron canescens, Viburnum nudum, Alnus serrulata, Ilex glabra, and Ilex opaca. Herbaceous species of

the understory in watersheds A and B included Boehmeria cylindrica, Woodwardia virginica, Woodwardia areolata, Panicum sp., Carex sp., Cyperus sp., Juncus effusus, and Smilax laurifolia. Typical herbaceous species in the understory of watersheds C and D were Arundinaria gigantea, Leucothoe axillaris, Smilax pumila, and Mitchella repens (International Paper unpublished data).

Climate

Climate of the region is characterized by warm, humid summers, and mild winters. Temperatures in January, the coldest month of the year, range from an average maximum of 16.3^{o}C and a minimum of 2.8 ^{o}C. July is the hottest month of the year with an average maximum temperature of 33.5 ^{o}C and minimum of 21.5 ^{o}C (SERCC 2004). Mean annual precipitation is 1412 mm. June has the highest mean rainfall (152.1 mm) and October lowest (77.5 mm) (SERCC 2004). Summer rains are usually short, with high intensity events giving way to low intensity frontal events from late fall to early spring. Due to proximity of the Gulf of Mexico, heavy rainfall associated with hurricanes and tropical storms in late summer is not unusual. Drought conditions occurred during 1998-2002 and resulted in an accumulated rainfall deficit of 711-1270 mm in some southwestern Georgia areas (Pam Knox, Assistant Georgia State Climatologist, oral communication as cited in Warner and Norton 2003).

Site History

Starting with small-scale disturbance by Native Americans who used fire to manage pinelands and prepare land for cultivation, the forest in many parts of the ACF river basin has been affected by human activity. This continued through European settlement, with pre-and post-Civil War agriculture, and now the area is primarily characterized by second growth stands and acreages of planted pine (Couch et al. 1996).

Site history specific to the study area and when noted, the specific study sites, is as follows.

Table 2-1. Site history events for the study site and watersheds.

	Watersheds					
	*SA	A	B	C	D	Site History Event
Date:						
1837	X					Site settled by Munnerlyn family
						Cattle grazing and sharecropped for cotton, corn, peanuts, and flax
1925	X					Portable sawmill operations
						Riparian areas not likely harvested
1937	X					Managed as a hunting preserve
1957	X					Property aquired by International Paper
1968				X	X	Uplands of C (south side) and D (north side) hand planted with loblolly pine
1969			X	X		Uplands of B (south side) and C (north side) hand planted with loblolly pine
1986		X				5.67 ha portion was hand planted
1987		X				5.67 ha portion had herbicide applied by a skidder to control herbaceous vegetation
1988						
1989		X	X			Uplands of A and B (northern half) hand planted with loblolly pine
1990			X	X		Uplands of B (south side) and C (northern portion) were control burned
1991						
1992			X	X		Uplands of B (south side) and C (northern portion) were control burned
				X	X	Uplands of C (south side) and D (north side) were control burned
					X	Uplands of D (south side) were control burned
1993						
1994		X				5.67 ha portion was control burned
1995				X	X	Uplands of C (south side) and D (north side) were control burned
1996		X				5.67 ha portion was control burned
			X	X		Uplands of B (south side) and C (northern portion) were thinned
				X	X	Uplands of C (south side) and D (north side) were thinned
1997						
1998						
1999						
2000		X				5.67 ha portion was control burned
		X	X			Aerial herbicide application and control burn for uplands of A and B (northern half)
2001		X				5.67 ha portion was thinned

*SA = activities occurred in study area

In 1837, the land known then as the Fowltown tract was bought and settled by Charles Lewis Munnerlyn, originally from Georgetown, South Carolina. The 1,349 ha property was pineland at the time of purchase, which was thought to be of little value except for cattle grazing (International Paper 1997). In the 1830's and 1840's the Munnerlyn slaves cleared all debris (i.e. moss, limbs, and leaves) out of the streams and spread this over the fields. This scattering of debris and another technique known as "cow pinning", which consisted of allowing large herds of cattle to rest in the fields, was

used to enrich the soils. This practice is thought to have occurred in the study streams and watersheds (J. Wingate, personal communication, 17 August 2004). In 1864, Munnerlyn's only son, Charles James, was appointed by Jefferson Davis as a Major with command of the First Battalion, Florida Special Cavalry. This battalion, later named the Cow Cavalry, was organized to collect and drive cattle from Florida to help supply the Confederate Army. Munnerlyn's superior officer noted that he had operated his own large plantation with great success (Taylor 1986). The study area is thought to have been used as a resting and grazing place by Munnerlyn for cattle herds being moved from Florida to Columbus Georgia to supply Confederate troops (J. Wingate, personal communication, 17 August 2004). During the years of Munnerlyn family ownership after the war, the property was sharecropped for cotton, corn, peanuts, and flax (Table 2-1) (J. Wingate, personal communication, 17 August 2004).

There is evidence of damming in an upstream portion of watershed C. This site was dammed, and a water "ram" was constructed downstream. This ram supplied water to a house located upslope from the site, which is thought to be the first house built in 1822 or 1823 in Decatur County (J. Wingate, personal communication, 17 August 2004). It is not known when this dam was installed, or how long it existed.

In 1925, the property was sold to a partnership of Ludwick Gaissert, O.M. Peden, H.R. Garrett, and W.R. Layson. This partnership dissolved, and H.R. Garrett became sole owner and set up a lumber mill site. His operation had portable saws and moved from site to site on the property as needed (Table 2-1). Garrett apparently did not use oxen to move harvested timber, and this has led to the assumption that Garrett predominantly cut timber on the ridges and likely did not harvest riparian areas at the

bottom of steep slopes that are found in the study area (J. Wingate, personal communication, 17 August 2004). Due to better opportunities or possibly because most saleable timber had been cut, Garrett sold the property.

Richard Tift, a prominent land speculator, and Herbert L. Stoddard, generally recognized as the "father" of prescribed burning in the South and for his work with bobwhite quail, began buying options for parcels of land along the Flint River. After approximately a year and a half, Tift and Stoddard acquired options for 28 parcels of land comprising 10,522 ha. In 1937, Houghton P. Metcalf, a wealthy industrialist from Providence, Rhode Island, bought the parcels and named his estate Southlands. Metcalf authorized removal of the remainder of Munnerlyn's cattle herds and wild hogs from the property. The property on the east side of the Flint River, where the study site is located, was managed as a hunting preserve (Table 2-1) and was specifically managed to attract quail, dove and turkeys. The property on the west side of the river had a hog farm and was planted in corn and peanuts. Metcalf was interested in reforestation and wanted to preserve the natural beauty of the property. In 1947, the property was sold to Southern Kraft Timberland Corporation, a division of International Paper Company (International Paper 1997).

On 14 November 1957, the property was dedicated as a research center due, in part, that all four major southern pine species (i.e. loblolly, longleaf, shortleaf, and slash) naturally grew on site (Table 2-1). Also, the property had diverse terrain with upland loblolly sites on the east side of the river, sandy longleaf sites on the west side of the river, river swamps, and bottomland hardwoods (International Paper 1997).

Detailed information on each stand in the study was obtained from International Paper stand inventory data (Table 2-1). Trees in riparian areas of each of the four study watersheds were aged by a timber cruise in 1987. Increment cores were taken on 2 or more trees per plot in the natural pine/hardwood stands. An establishment date was estimated as 1935 (D. Morgan, personal communication, 3 August 2004), although the trees in riparian areas are thought to be older due to the manner of harvesting employed by H.R. Garrett (J. Wingate, personal communication, 17 August 2004). Riparian areas in all four watersheds, (A through D), (Figure 2.3) were not subject to any silvicultural activities due to International Paper policy of maintaining forested buffers. A 5.67 ha portion of watershed A, north of the stream and south of the main road, was hand planted in 1986, and herbicide was applied by a skidder in 1987 to control herbaceous vegetation. This area was control burned in 1994, 1996, and 2000 and thinned in March 2001. The majority of the uplands of watershed A and the northern half of watershed B were hand planted with loblolly pine in 1989. An aerial herbicide application and a control burn were completed in 2000. The uplands on the south side of watershed B and the northern portion of watershed C's uplands were established in 1969 by hand planting of loblolly pine. Control burns were completed in 1990 and 1992, then the area was thinned in 1996. Uplands on the south side of watershed C and the north side of watershed D were hand planted with loblolly pine in 1968. Control burns were completed in 1992 and 1995, and the area was thinned in 1996. Uplands on the south side of the stream in watershed D were allowed to naturally regenerate in 1950. A control burn of this area was completed in 1992.

A private landowner owns 24% of watershed B and 18% of watershed C (Figure 2.4). A cattle farm was operated in these areas from 1950 through 1994. Hogs were also kept on the property. Streams of watersheds B and C were the water source for the cattle and hogs. In 1968, pines were selectively removed from riparian areas, but hardwoods were left (C. Lynn, personal communication, 17 August 2004).

CHAPTER 3
METHODS

Overview of Study

Four headwater streams and their watersheds (A, B, C, and D) were selected for study (Figure 3-1). The overall Dry Creek Study (Streamside Management Zone Effectiveness on Hydrology, Water Quality, and Aquatic Habitats in Southwestern Georgia Headwater Streams) design includes elements of before and after, upstream/downstream, and paired watersheds experimental designs.

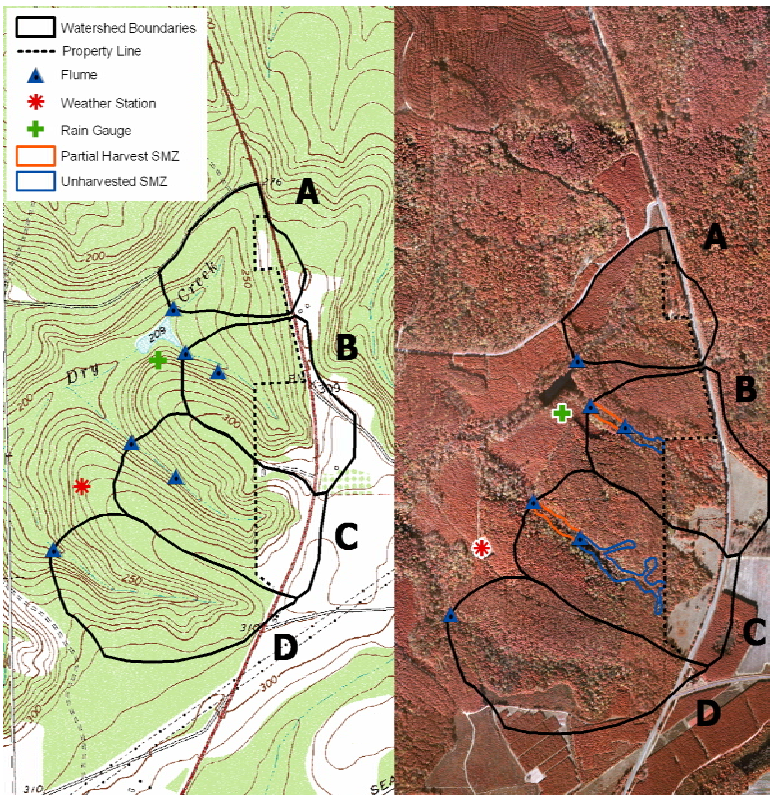

Figure 3-1. Topographic map and aerial photograph of location of four headwater watersheds (A-D).

This study compared upstream vs. downstream, between watersheds, and across time to evaluate macroinvertebrate assemblages and primary environmental variables in the four adjacent study streams. This study established the pre-harvest condition and natural variability of the macroinvertebrate assemblage. This information will be used for post-harvest comparisons in the overall Dry Creek Study, being conducted by multiple partners.

Data Collection

Physical Measurements

Eight fixed-distance sample reaches, two per watershed, were established 30.8 m upstream of flumes (Figure 3-2).

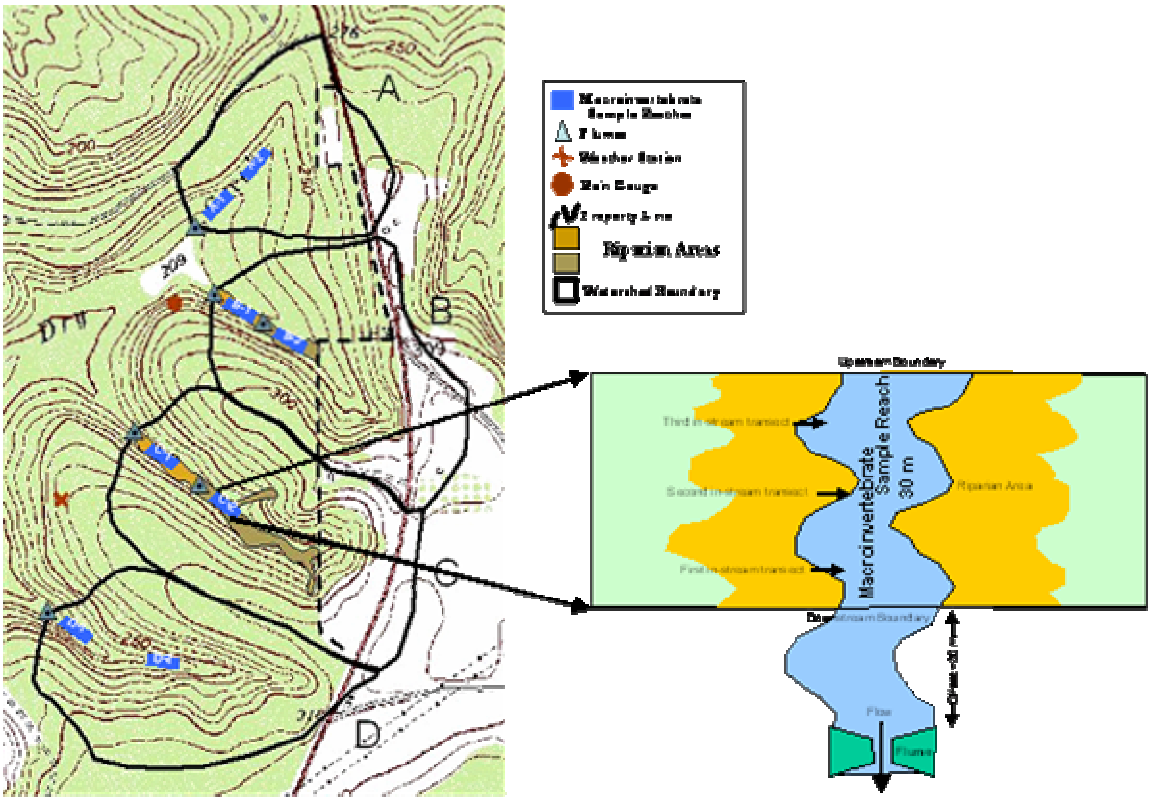

Figure 3-2. Topographic map of location of four headwater watersheds (A-D) with eight sample reaches and schematic of an individual sample reach.

Three transects were established perpendicular to the stream within each reach to serve as in-stream data collection points for physical measurements including channel cross-sections, canopy cover, and percent cover of in-stream habitat. At each transect, percent cover of in-stream habitat was determined by extending a tape across the active channel and recording the length of each habitat type (e.g., sand, small woody debris, roots, leaf pack, gravel). These lengths were converted into percent cover, which was used to define the major habitat types to be sampled for macroinvertebrates. In August 2002, pictures of the canopy were taken at each transect with a digital camera fitted with an 180° hemispherical fisheye lens.

A survey of habitat unit and channel characteristics was conducted longitudinally within established macroinvertebrate sample reaches. A 50 m fiberglass tape was placed in the thalweg of the stream, then divisions between each habitat unit type were determined and physical characteristics were recorded. Unit types included riffle, run, glide, pool, backwater pool, step, and undercut bank. A backwater pool was defined as slower and deeper than a glide, but did not possess characteristics of a pool, such as evidence of scouring, deposition, and having a deep and shallow section (i.e. measurable residual pool depth). For each unit type, a unit end (length), channel width (active channel), and maximum water depth were recorded. For step and pool unit types, a step height and residual pool depth were taken. Primary obstructions (e.g., wood, roots), their length, and diameter, were recorded when the object was primarily responsible for pool formation. A tally of functional and non-functional wood greater than 10 cm in diameter was taken. Functionality was based on the role wood played for changing morphology.

Texture of the streambed (e.g., sand, silty-sand) was estimated by soil feel and appearance in each sample reach.

Environmental Measurements

Sixteen leaf litter traps (surface area of 0.26 m^2 each) were positioned within the riparian area: six along the streambank, six 10 m from the stream, and four 20 m from the stream (Figure 3-3). Litter samples were collected monthly, dried at 60°C for 24 hours, separated into pine and hardwood foliage, woody debris, and mast, and weighed.

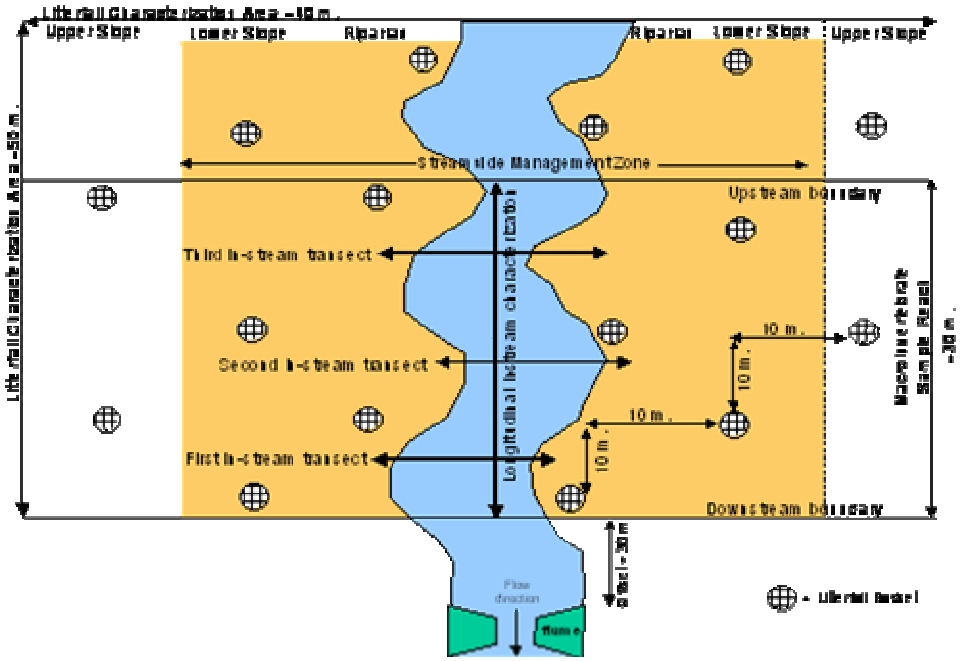

Figure 3-3. Schematic of a representative sample reach with layout of litterfall traps.

Within each stream reach, ten randomly selected locations were sampled for periphyton and macrophytes in June, July, and August. Following the method of Tett et al. (1978), two petri dishes (17.34 cm^2) were inserted into the sediment at each sampling location. Chlorophyll a concentrations of periphyton in the sediment sample were measured using an acetone extraction procedure (American Public Health Association, 1995) followed by colorimetric analysis. The contents of the second petri dish were dried

at 60°C, weighed, burned at 500°C, and reweighed for ash-free dry weight determination. Macrophytes were sampled by cutting all vegetation at the sediment surface that existed within a 0.25 m^2 quadrat. Macrophyte samples were rinsed and dried at 60°C (Kedierski and Smock, 2001). Dry weight was determined for each sample.

Chemical and Hydrological Measurements

Water temperature was measured from October 2001 through December 2003 with an Onset HOBO ® temperature logger (Pocasset, MA), which was programmed to measure temperature every 15 minutes. Stream flow, water chemistry, and meteorological measurements have been collected by other investigators as part of the Dry Creek Study, and these data were available for use in this study. Stream stage and discharge was recorded every 15 minutes by Isco Model 4320 Bubbler Flow Meters at six sites: one in the stream at the outlet of watersheds A, B, C, D and one in the upstream portion of watersheds B and C (Summer 2003). Monthly in-situ measurements for dissolved oxygen, specific conductance, temperature, pH, and turbidity were made at eight sites (two per watershed) with portable meters. Grab samples were taken from a midstream location and analyzed for inorganic nitrogen, inorganic phosphorus, and ammonium (Jones et al. 2003).

Macroinvertebrates

Benthic macroinvertebrates were collected within established sample reaches (Figure 3-2) with a 500-µm-mesh D-frame net (0.3 m wide) in December 2001, February and December 2002, and February 2003 using a multi-habitat sampling procedure (Barbour et al. 1999). This procedure was tested by the Mid-Atlantic Coastal Streams Workgroup and the Florida Department of Environmental Protection and deemed a scientifically valid sampling technique for low-gradient streams (Barbour et al. 1999).

Fall and winter sampling periods were chosen because these seasons are prior to the emergence of most species and larvae are generally easier to identify because of their larger size. Within each reach, 20 sampling sweeps (i.e. disturbing habitat for 0.5 m) were made through major habitat types such as sand, woody debris, fine roots, and leaf packs. This resulted in approximately 3.1 m^2 of habitat sampled. The duration of sampling in each reach was timed to maintain a consistent sampling effort for all reaches. Material collected from each sampling sweep was deposited in a 19 L bucket. Material in the bucket was rapidly stirred to suspend organisms and poured into the 500-µm-mesh D-frame sampling net. Material caught in the net was placed in a 4 L glass jar and preserved with 70% ethyl alcohol in the field. Rose Bengal biological stain was added to each sample in the laboratory. All samples were processed by washing organic debris (leaves and woody debris) with water into a 500-µm-mesh sieve. Invertebrates were handpicked from the sieve contents and identified to genus or species (when possible), under a low power (<50x) dissecting microscope (Richardson 2003, Gelhaus 2002, Pescador, Rasmussen, Richard 2000, Epler 1996, Merritt and Cummins 1996, Pescador, Rasmussen, Harris 1995, Peckarsky et al. 1990, Brigham, Brigham, and Gnilka, 1982). Genus or species-level taxonomy has been found to yield the greatest benefits for biological monitoring studies especially when results could influence management decisions (Lenat and Resh 2001). Ephemeroptera, Plecoptera, and Trichoptera generic identifications were verified, and species identifications were made when possible by M.L. Pescador and A.K. Rasmussen. Dytiscidae species identifications were made by Bill Wolfe. All Chironomidae samples were sent to a consultant, Pennington and Associates, for identification, generally to species. Larval Chironomidae were cleared with cold 10%

KOH for 24 hours, then temporary slide mounts were made using glycerin. Permanent slide mounts were made in CMC mounting media for voucher specimens. Identifications were made to genus under a dissecting microscope using Merritt and Cummins (1996). Further identification to species was made using a compound microscope (Epler 2001). Functional feeding group and habitat/behavior designations were determined using Merritt and Cummins (1996). Oligochaeta, Gastropoda, and Bivaliva were enumerated but not identified beyond Class and were not included in metric calculations or data analysis.

Data Analysis

Physical Measurements

Percent cover of in-stream habitats was summarized for each site. Length multiplied by average width measurements for each channel unit was used to calculate the area of each channel unit. The percentage of the total area occupied by each channel unit was calculated by: % of Area = area of channel unit type / total area of reach x 100 (Bisson, and Montgomery, 1996).

A digital camera was used to convert the hemispherical images of canopy cover into bitmaps, which were then analyzed using Gap Light Analyzer© software (Frazer and Canham, 1999). This software transformed pixel intensities into sky and non-sky classes, then these data were used to estimate percent canopy cover. A tally of functional and non-functional wood > 10 cm in diameter was summarized for each site.

Environmental Measurements

Seasonal average dry-weight data were calculated from the six litterfall traps positioned along the stream bank for each site. Repeated measures analysis of variance (ANOVA) (SPSS Inc., Chicago IL) was used to determine whether there was a

significant effect due to position (upstream versus downstream, df =1,6), season (fall, winter, spring, summer, df =3,18), or the interaction of position and season (df =3,18). A second repeated measures ANOVA was run to determine effects due to season (fall, winter, spring, summer, df =3,12), stream (A, B, C, D, df = 3, 4), or the interaction of season and stream (df =9,12). The sample size for each stream was 2, which limited the power of the across stream comparisons. Periphyton chlorophyll a concentrations, periphyton ash-free dry-weight, and macrophyte dry-weight were analyzed using one way ANOVA (alpha = 0.05), which tested the equality of site means. One way ANOVA was used because the samples for each site were 10 independent random samples. Differences between time periods were not examined because samples were collected in one summer season, June, July, and August of 2003. Fisher's multiple comparison procedure was used for significant ANOVA results, which generated confidence intervals for all pairwise differences between site means (individual error rate = 0.05) (Minitab Inc., State College PA).

Chemical and Hydrological Measurements

Monthly dissolved oxygen, specific conductance, temperature, pH, turbidity, inorganic nitrogen, inorganic phosphorus, and ammonium values were summarized as averages for six months and three months prior to each macroinvertebrate sample. Average daily flow (Liters/second) was converted into minimum, average, and maximum daily flow for each month. Average zero-flow days were calculated and further summarized as averages for six months and three months prior to each macroinvertebrate sample.

Macroinvertebrates

Data from each site were used to develop numerical metrics to describe the macroinvertebrate assemblages of the study streams. Abundance and percent dominant taxon (i.e. dominance of the single most abundant taxon) were tallied and calculated to characterize the population in each stream. Taxa richness, EPT taxa, and number of Chironomidae taxa were calculated to determine richness. Taxa richness was calculated as the number of unique taxa at the family, genus, or species level. For example, if a sample contained 5 Libellulidae, 4 Gomphidae, 5 *Gomphus*, 5 *Progomphus*, and 1 *Boyeria vinosa*, this would result in 4 taxa with 20 individuals. EPT taxa were calculated as the number of unique taxa at the genus or species level. Number of Chironomidae taxa was calculated as the number of unique taxa at the genus or species level. Percent Diptera and percent Chironomidae were also calculated for each stream. Percent filter feeders and number of clinger taxa provided information on partitioning feeding strategies and habitat preference of insects in the assemblages.

Percent Elmidae was calculated as an experimental metric because percent Elmidae was found useful in describing perennial streams within Georgia's Fall Line Hills District (Muenz 2004), which is located in the adjacent physiographic district to the study site (Figure 2.2). Elmids prefer swifter parts of streams such as oxygen rich riffles (Merritt and Cummins 1996). Also, this family of beetles were described by Epler (1996) to be the most truly aquatic of Florida water beetles because the larvae possess gills and adults utilize a plastron (covering of fine dense hydrofuge setae that holds a layer of air where gases can be exchanged), which enables them to remain submerged. Most other aquatic beetles must go to the surface. These characteristics of the Elmidae make them the best candidates of the aquatic beetles as indicators of water quality (Epler 1996).

Data from each site were used to develop numerical metrics which were used to calculate Florida Department of Environmental Protection's Stream Condition Index (SCI) (FDEP, 2004) and Georgia Department of Natural Resources' (DNR) Freshwater Macroinvertebrate Biological Assessment (GA DNR, 2002). Oligochaeta, Gastropoda, and Bivaliva were excluded from analyses. For Florida's SCI, raw data from each site were sub-sampled for 100 individuals using Microsoft Excel©'s random number generator. Metric values for total taxa, Ephemeroptera taxa, Trichoptera taxa, long-lived taxa, percent filter feeders, number of clinger taxa, number of Chironomidae taxa, percent Tanytarsini, sensitive taxa, and percent very-tolerant were calculated and converted into a metric score ranging from 0 to 10 using formulae contained in Table 3-1.

Table 3-1. Stream Condition Index metric scoring formulae.

SCI metric	Northeast	Panhandle	Peninsula
Total taxa	10 * (X–16)/26	10 * (X–16)/33	10 * (X–16)/25
Ephemeroptera taxa	10 * X /3.5	10 * X /6	10 * X /5
Trichoptera taxa	10 * X /6.5	10 * X /7	10 * X /7
% Filterer	10 * (X–1)/41	10 * (X–1)/44	10 * (X–1)/39
Long-lived taxa	10 * X /3	10 * X /5	10 * X /4
Clinger taxa	10 * X /9	10 * X /15.5	10 * X /8
% Dominance	10 – (10 * [(X–10)/44])	10 – (10 * [(X–10)/33])	10 – (10 * [(X–10)/44])
% Tanytarsini	10 * [ln(X + 1) /3.3]	10 * [ln(X + 1) /3.3]	10 * [ln(X + 1) /3.3]
Sensitive taxa	10 * X /11	10 * X /19	10 * X /9
% Very tolerant	10 – (10 * [ln(X + 1)/4.4])	10 – (10 * [ln(X + 1)/3.6])	10 – (10 * [ln(X + 1)/4.1])
(FDEP, 2004)			

Metric scores were summed and divided by a correction factor. The SCI category (good, fair, poor, very poor) for each site followed ranges provided in the index (Table 3-2).

Table 3-2. Category names, ranges of values for Stream Condition Index, and typical biological conditions.

SCI category	SCI range	Example Description
1 sample		
Good	[73–100]	Similar to natural conditions, up to 10% loss of taxa expected
Fair	[46–73)	Significantly different from natural conditions; 20–30% loss of Ephemeroptera, Trichoptera and long-lived taxa; 40% loss of clinger and sensitive taxa; percentage of very tolerant individuals doubles
Poor	[19–46)	Very different from natural conditions; 30% loss of total taxa; Ephemeroptera, Trichoptera, long-lived, clinger and sensitive taxa uncommon or rare; Filterer and Tanytarsini individuals decline by half; 25% of individuals are very tolerant
Very poor	[0–19)	Extremely degraded; 50% loss of expected taxa; Ephemeroptera, Trichoptera, long-lived, clinger, and sensitive taxa missing or rare; 60% of individuals are very tolerant

(FDEP, 2004)

For Georgia's Biological Assessment, the data from each site was used to calculate the following metrics: taxa richness, number of Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa, number of Chironomidae taxa, percent contribution of dominant taxon, percent Diptera, Florida Index, and percent filterers. These metrics measure the richness, composition, tolerance/intolerance, and feeding strategies of the assemblage. Each metric value was converted into a score, and scores were summed. Georgia DNR reference stream data scores from the Southeastern Plains Ecoregion, Tifton Upland sub-ecoregion were averaged and compared to study sites. Percent comparability of study to reference sites were calculated (study site score/reference site score x 100) to determine ecological condition (Table 3-3). A second calculation of Georgia's Biological assessment was performed for a 200 individual sub-sample according to GA DNR's Standard Operating Procedures (GA DNR, 2002). Raw data from each site were sub-

sampled for 200 individuals using Microsoft Excel's random number generator, and scores were generated as outlined above.

Table 3-3. Sample Ecological Condition Worksheet.

METRIC:	Study Site Data:	Ref Site:	SOUTHEASTERN PLAINS (65) SCORE RANGES 5	3	1	0	Study Site	Ref Site Score
Taxa Richness (Total # Taxa)			>30	30-16	<16			
EPT Index			>6	6-4	<4			
# Chironomidae Taxa			>8	8-5	<5			
% Contribution of Dominant Taxon			<23	23-61	>61			
% Diptera			-----	<51	>50			
Florida Index			>15	15-8	<8			
% Filterers			>11	11-6	<6			
Total Habitat Score:			>89%	89-75%	74-60%	<59%		
Total Points Earned							0	0
% of Reference Site								
Ecological Condition:								
Very Good	Comparable to best situation expected; species with endangered, threatened, or							> 82%
Good	Balanced community with sensitive species present.							81-64%
Fair	Expected species absent or in low abundance; few sensitive species present.							63-48%
Poor	Low species richness, with tolerant species predominant, sensitive species absent.							47-35%
Very Poor	Expected species absent, only tolerant organisms present, few or no EPT taxa.							< 35%

(GA DNR 2002)

Using the Georgia Biological Assessment method, study sites were compared to each other to assess year to year, between watersheds, and upstream vs. downstream percent comparability. An ecological condition score was not assigned to these comparisons.

Georgia Adopt-A-Stream volunteer monitoring assessment methods were applied to the macroinvertebrate data. A presence/absence count of sensitive, somewhat sensitive, and tolerant insects, crustaceans, aquatic worms, gastropods, and bivalves were made for each site. All categories were summed and multiplied by a factor of 3 for sensitive, 2 for somewhat sensitive, and 1 for tolerant. The result for each category was summed for the total index value, and a water quality ranking (excellent, good, fair, and poor) was assigned (GA DNR 2000).

All percent metrics were arcsine transformed and abundance was log transformed prior to statistical analysis. Repeated measures ANOVA (SPSS Inc., Chicago IL) was

used to determine any significant effect due to position (upstream versus downstream, df =1,6), time (Dec 01, Feb 02, Dec 02, Feb 03, df =3,18), or interaction of position and time (df =3,18) for all macroinvertebrate metrics and index values. Within and between subjects means and standard error were calculated as part of the repeated measures ANOVA procedure (SPSS Inc., Chicago IL). When significant ($P<0.05$) differences were detected due to time, repeated contrasts (i.e. comparison of adjacent levels) was used to compare time periods. A second repeated measures ANOVA was run when position was insignificant ($P>0.05$) for determining effects due to time (Dec 01, Feb 02, Dec 02, Feb 03, df =3,12), stream (A, B, C, D, df = 3, 4) or interaction of time and stream (df =9,12). The power of this analysis was limited because the sample size for each stream was two. When ANOVA resulted in significant ($P<0.05$) differences between means, a pairwise multiple comparison test (Tukey's honestly significant difference (HSD) test, alpha = 0.05) was made between means of a factor.

Principal components analysis (PCA) was used as an exploratory analysis (Golladay and Battle 2002, Karr and Wisseman 1996) to visualize broad trends in the chemical data, hydrological data, and macroinvertebrate measures among sites. This technique reduces a data set with many variables into a smaller number of composite variables (axes) and indicates covariation among variables with a set of primary axes. Options for PCA included Euclidean distance and cutoff r^2 (0.2) (McCune and Mefford 1995). For each PCA (i.e. chemical, hydrological, and macroinvertebrates), the number of axes included in the analysis was determined using broken-stick eigenvalues. For the PCA graph, each stream was assigned a unique symbol and further identified with a position number (1-downstream, 2-upstream) and a time number (1-Dec 01, 2-Feb 02, 3-

Dec 02, and 4-Feb 03). Interpretation of PCA results assumed that most variation is explained by variables furthest from the origin. Positively correlated variables are close together, whereas negatively correlated variables are located at opposite ends of the axis. Data points close together are more similar, and those far apart are dissimilar.

Stepwise multiple regressions (Alpha-to-Enter: 0.05, Alpha-to-Remove: 0.05) (Minitab Inc., State College PA) were used to quantify relationships between predictor variables (abundance, EPT taxa, total taxa, Georgia EPD index, Georgia AAS index, percent Elmidae, percent dominant taxa, and percent filtering collectors), with response variables (DO, pH, water temperature, specific conductance, inorganic nitrogen, inorganic phosphorus, ammonium, turbidity, maximum daily flow, average daily flow, minimum daily flow). Abundance, EPT taxa, total taxa, Georgia EPD index, Georgia AAS index, and percent Elmidae were selected as response variables because of significant ANOVA results. Five models were run for each response variable: 1) all samples pooled (n=31), 2) December 2001 samples only (n=7), 3) February 2002 samples only (n=8), 4) December 2002 samples only (n=8), 5) February 2003 samples only (n=8). This procedure was run three times with mean values for water chemistry and hydrologic variables from one month, three months, and six months prior to macroinvertebrate sample collection.

CHAPTER 4
RESULTS

Physical Measurements

Habitat units in the study streams included sand (and other fine sediment), leaves, roots, undercut bank, gravel, and small woody debris (SWD, <10 cm in diameter). The percent coverage of these in-stream habitat types varied more among watersheds than within the same stream. Sand and leaf habitat was most abundant in watershed A. Site A-1 contained more undercut bank habitat, whereas A-2 had a greater percentage of roots (Figure 4-1). Sand and leaves were dominant habitats in watershed B, with root habitat also represented. Site B-2 had gravel and undercut bank habitats represented, while site B-1 did not (Figure 4-1). Watershed C, Sites C-1 and C-2 contained sand, leaf, root, and SWD habitats. Coarse bed material was observed at C-1, but did not occur in a measured transect. Sites in watershed D (Figure 4-1) were characterized by sand, leaf, gravel and SWD habitats, with D-1 also having root habitats. Although not measured in this survey, exposed areas of limestone were present in the streambed of watershed D.

Channel units in the study streams included backwater pool, glide, pool, riffle, run, and step. The percent aerial coverage of these channel units varied among watersheds and within the same stream. Site A-1 had a much greater percentage of run (49%) than A-2 (13%), but A-2 had 30% more backwater pool area than A-1 (Figure 4-2). Sites B-1 and B-2 (Figure 4-2) had similar areas of glide, pool, and run, but B-2 had a small area of riffle, and B-1 had a small area of backwater pool.

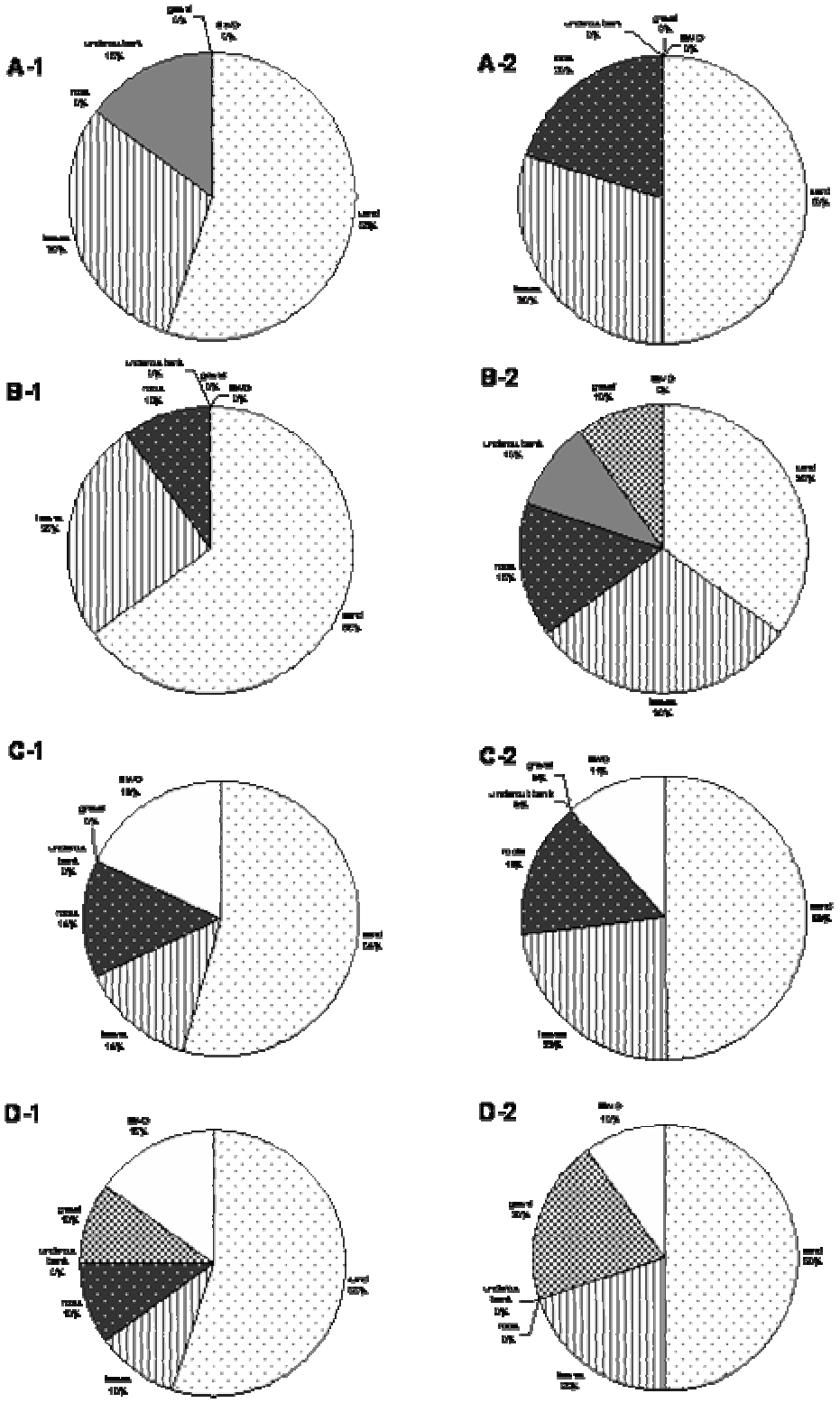

Figure 4-1. Percent coverage of in-stream habitat units for each sampling site (A-1 through D-2).

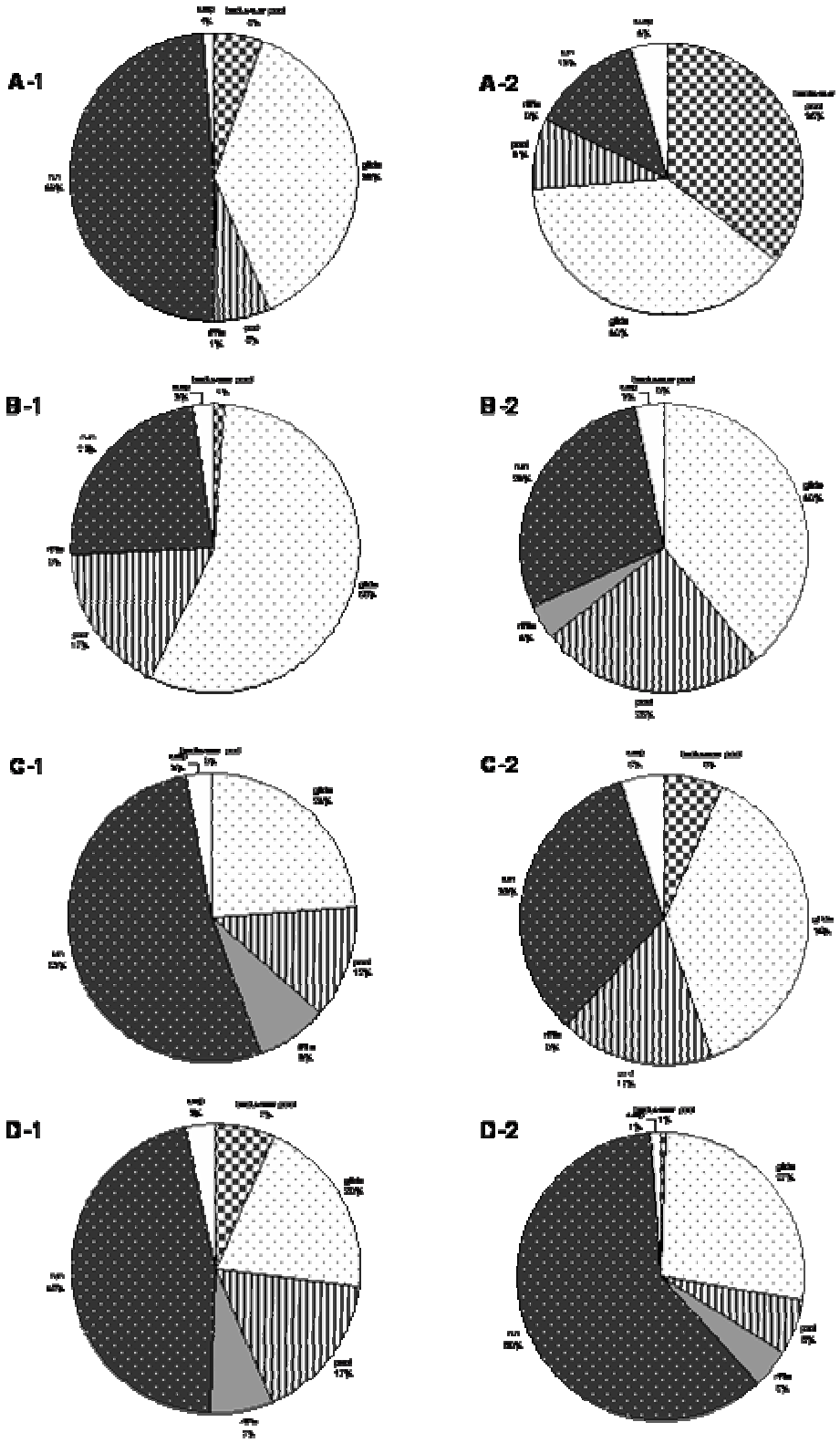

Figure 4-2. Percent coverage of in-stream channel units for each sampling site (A-1 through D-2).

Sites within watershed C were dominated by runs and glides at C-1 and C-2, respectively. C-1 had areas with riffles, whereas C-2 had backwater pools (Figure 4-2). Sites within watershed D were the most similar in channel unit area, with all unit types represented. Runs dominated the stream, with glide areas of secondary importance. Riffles were present at both D-1 and D-2 (Figure 4-2).

Average canopy for all sites combined was 85% with maximum canopy cover of 87% at site B-1 and minimum (83%) at C-2. Canopy cover varied within sites, but percentages were similar among sites (Figure 4-3).

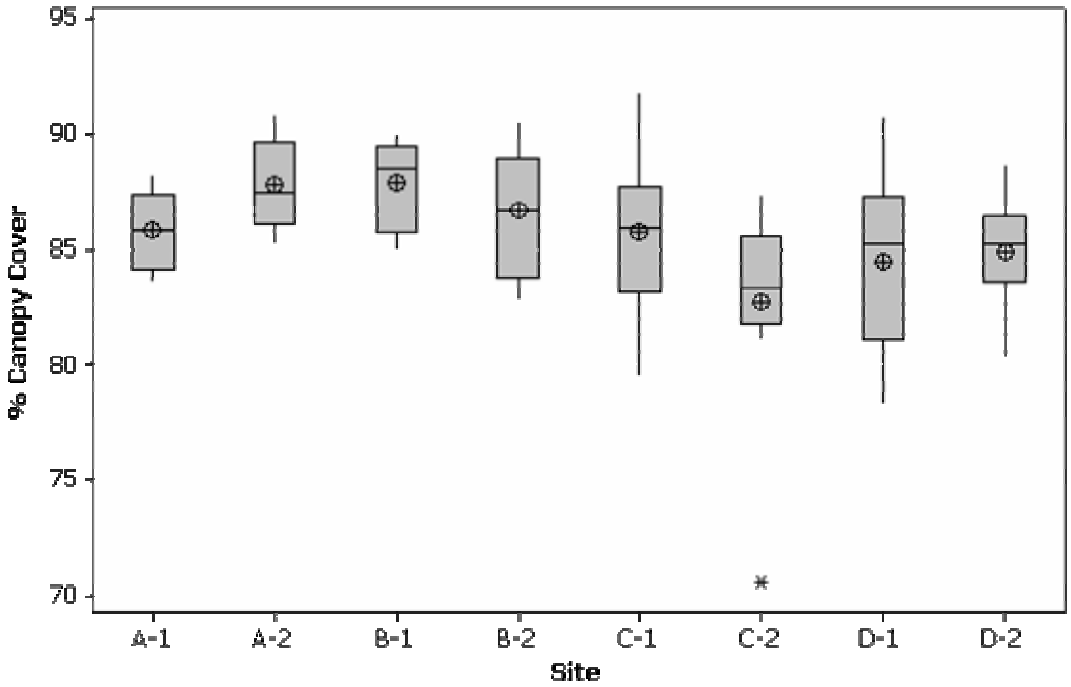

Figure 4-3. Percent canopy cover for each sampling site (A-1 through D-2) as defined by GLA software. Data represented in box (interquartile range) and whiskers plot with median (horizontal line), mean (circle), and outliers (star).

Amounts of large woody debris (LWD) varied greatly among and within watersheds with no readily apparent patterns (Table 4-1). A-1 and B-1 had the greatest number of total LWD. B-2, D-1, and D-2 had lowest total numbers of LWD.

Table 4-1. Tally of large woody debris (>10cm diameter).

	A1	A2	B1	B2	C1	C2	D1	D2
Functional	8	1	3	0	4	5	0	1
Non-functional	2	3	7	1	1	1	2	0

Environmental Measurements

Repeated measures ANOVA for litterfall indicated significant (P<0.001) differences between the four sampling periods (time). There was no significant effect (P=0.92) due to sampling position (upstream/downstream) or the interaction between position and time (P=0.62). There were no significant (P=0.50) differences among sites. Most litterfall in all streams occurred from September through January. Hardwood leaves comprised the greatest proportion of litterfall during this time period (Figure 4-4).

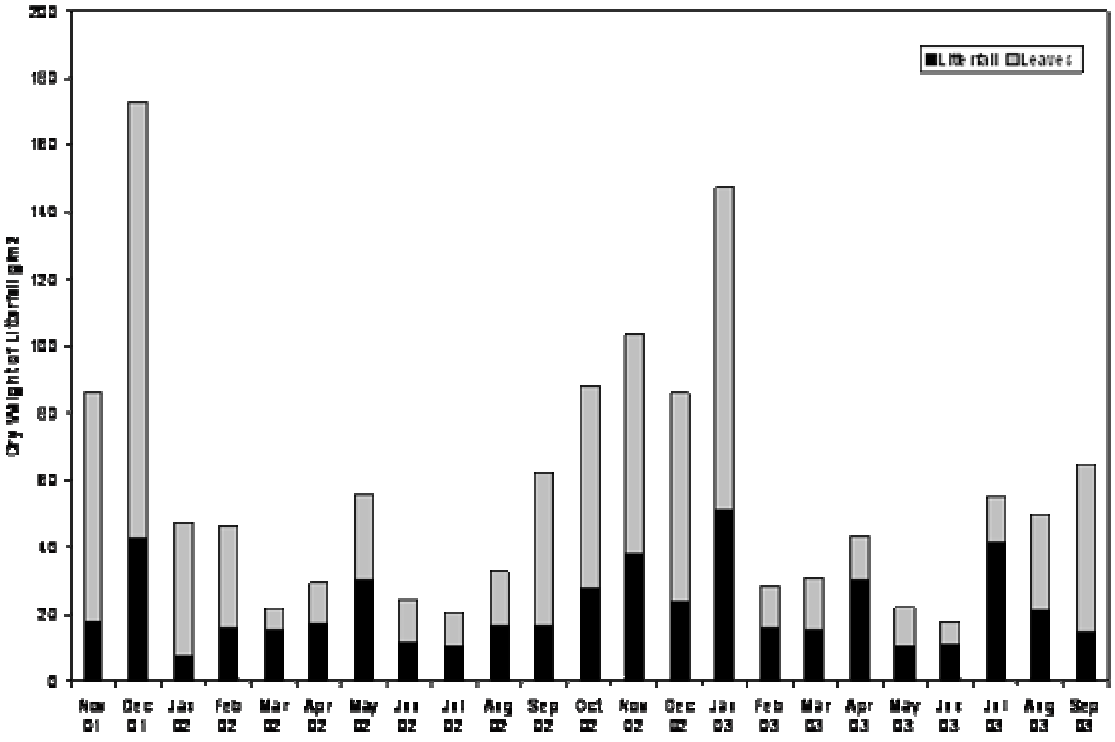

Figure 4-4. Average dry weight of total litterfall (hardwood leaves, pine, woody debris, and mast) across sites with proportion of total litterfall as leaves (hardwood leaves) in grey.

There were no significant differences among sites for periphyton chlorophyll a mean concentrations in July (P=0.49) and August (P=0.59) samples (Table 4-2). June 2003 samples are not reported due to an analytical error.

Table 4-2. Mean periphyton chlorophyll *a* and dry weight. Mean macrophyte dry weight.

	Periphyton						Macrophytes		
	Chlorophyll *a* (mg/m^2)			Dry Weight (g/m^2)			Dry Weight (g/m^2)		
	June 03	July 03	August 03	June 03	July 03	August 03	June 03	July 03	August 03
A-1	--	5.39	9.28	389.37	525.33	423.20	4.21	1.82	0.14
A-2	--	6.12	5.08	345.60	613.72	354.63	6.36	13.28	3.06
B-1	--	2.58	8.78	216.93	69.37	284.18	6.42	2.95	6.98
B-2	--	17.17	3.89	175.13	216.92	264.31	18.01	8.35	10.24
C-1	--	4.47	3.23	102.96	110.27	100.61	0.28	0.04	0.71
C-2	--	6.24	10.43	115.31	147.89	194.29	2.88	4.20	10.46
D-1	--	5.62	3.16	21.55	59.07	75.15	0.07	0.04	2.54
D-2	--	10.16	6.74	28.24	96.95	110.32	0.00	0.00	0.00
P value	--	0.490	0.59	<0.000	<0.000	<0.000	0.079	0.005	0.066

Periphyton ash free dry weight was significantly different (P<0.001) among sites for June, July and August sampling dates. Generally, sites within the same watershed were similar, and mean weight decreased from watershed A to D (Figure 4-5). Macrophyte dry weights among sites were not significantly different for June (P=0.07) and August (P=0.06) samples, but were different (P=0.005) between July 03 samples (Figure 4-6). A1, B1, C1, D1 and D2's mean biomass was very low (<3 g/m^2) and were not significantly different, whereas A2, B2, and C2 had greater biomass of macrophytes (Figure 4-6 and Table 4-2).

Chemical and Hydrological Measurements

Average daily-flow for the study period (July 2001-February 2003) was highest in watershed C (2.66 L/s), followed by B (2.16 L/s), D (1.29 L/s), and A (1.02 L/s). The number of days out of 638 days with zero flow was 161 (25%) for watershed A, 6 (1%) for watershed B, 2 (0.3%) for watershed C, and 206 (32%) for watershed D.

Monthly mean water temperature displayed little variation among sites (Figure 4-7) with lowest values in January and February and highest from June through September.

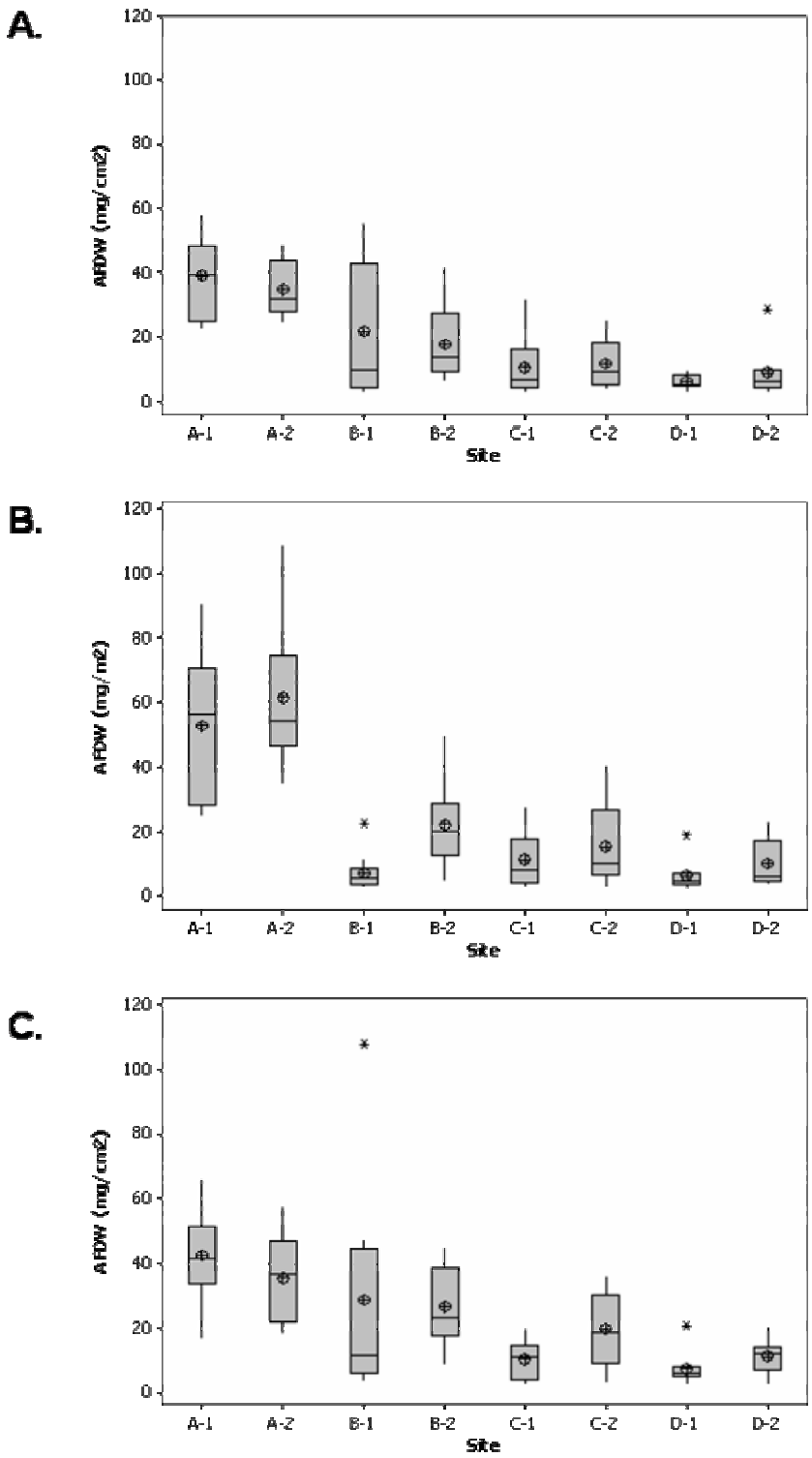

Figure 4-5. Periphyton ash free dry weight. A) June 2003, B) July 2003, C) August 2003. Data represented in box (interquartile range) and whiskers plot with median (horizontal line), mean (circle), and outliers (star).

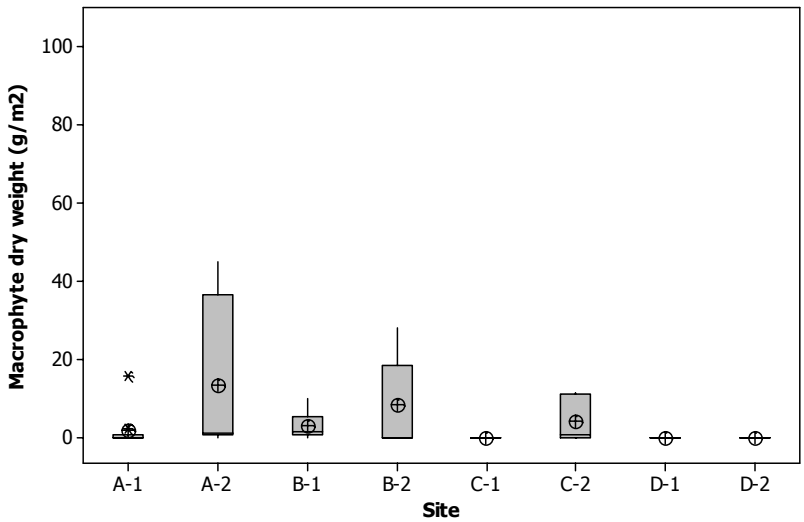

Figure 4-6. Macrophyte dry weight (July 2003). Data represented in box (interquartile range) and whiskers plot with median (horizontal line), mean (circle), and outliers (star).

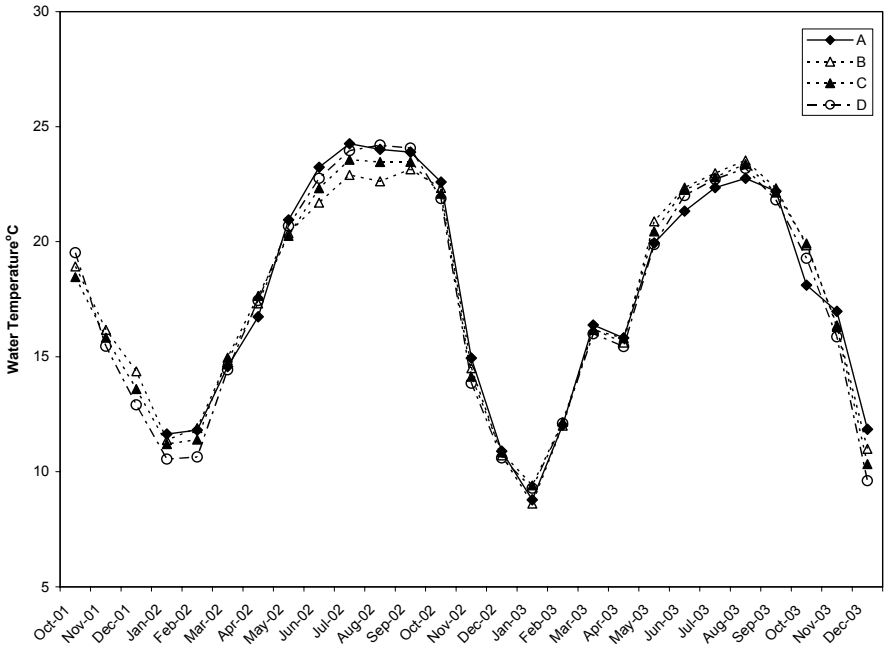

Figure 4-7. Monthly mean water temperature.

Dissolved oxygen values across sites ranged from 0.270 mg/L in September 2001 at A2 to 10.5 mg/L in January 2002 at D1 (Table 4-3). Average dissolved oxygen was lowest in watershed A (Figure 4-8).

Table 4-3. Mean, minimum, and maximum in-situ water chemistry.

Mean (min-max)	A-1	A-2	B-1	B-2	C-1	C-2	D-1	D-2
Temperature °C	18.1 (10.2-23.9)	18.1 (9.3-24.4)	18.3 (10.4-24.4)	18.6 (10.5-24.5)	18.3 (11.1-24.4)	18.7 (11.5-25.2)	18.5 (10.7-24.5)	18.5 (11.9-24.0)
DO mg/L	4.11 (0.98-8.78)	3.99 (0.27-7.60)	4.79 (.97-8.17)	5.01 (0.62-7.86)	6.64 (1.59-9.68)	6.26 (1.91-9.60)	6.69 (2.20-10.56)	5.29 (1.56-9.25)
pH	4.9 (3.5-5.8)	4.5 (3.3-5.6)	5.8 (4.1-6.9)	5.9 (4.3-6.6)	6.7 (5.1-8.0)	6.6 (5.1-7.5)	6.9 (5.1-7.8)	6.7 (5.1-7.3)
Turbidity NTU	2.14 (0.35-7.70)	1.54 (0.00-7.20)	6.13 (2.20-19.00)	6.92 (1.90-52.10)	6.77 (2.30-16.00)	8.48 (2.40-63.00)	4.97 (2.60-15.00)	4.78 (0.55-28.00)
Specific Conductance mS/cm	39.2 (23.0-70.8)	31.7 (24.0-80.2)	106.5 (25.8-272.0)	95.6 (26.0-360.0)	98.9 (25.4-157.8)	86.4 (26.6-154.2)	100.1 (50.8-166.2)	90.6 (25.4-179.4)
NH_4 ug/L	0.011 (0.000-0.097)	0.007 (0.000-0.025)	0.032 (0.000-0.150)	0.037 (0.000-0.180)	0.005 (0.000-0.032)	0.020 (0.000-0.071)	0.006 (0.000-0.023)	0.006 (0.000-0.066)
Inorganic N ug/L	0.002 (0.000-0.018)	0.001 (0.000-0.007)	0.412 (0.000-1.245)	0.850 (0.000-2.419)	0.872 (0.029-1.414)	1.188 (0.031-1.785)	0.010 (0.000-0.219)	0.020 (0.000-0.222)
Inorganic P ug/L	0.003 (0.001-0.005)	0.006 (0.000-0.081)	0.004 (0.000-0.014)	0.003 (0.000-0.011)	0.006 (0.003-0.012)	0.004 (0.002-0.007)	0.035 (0.015-0.073)	0.053 (0.012-0.130)

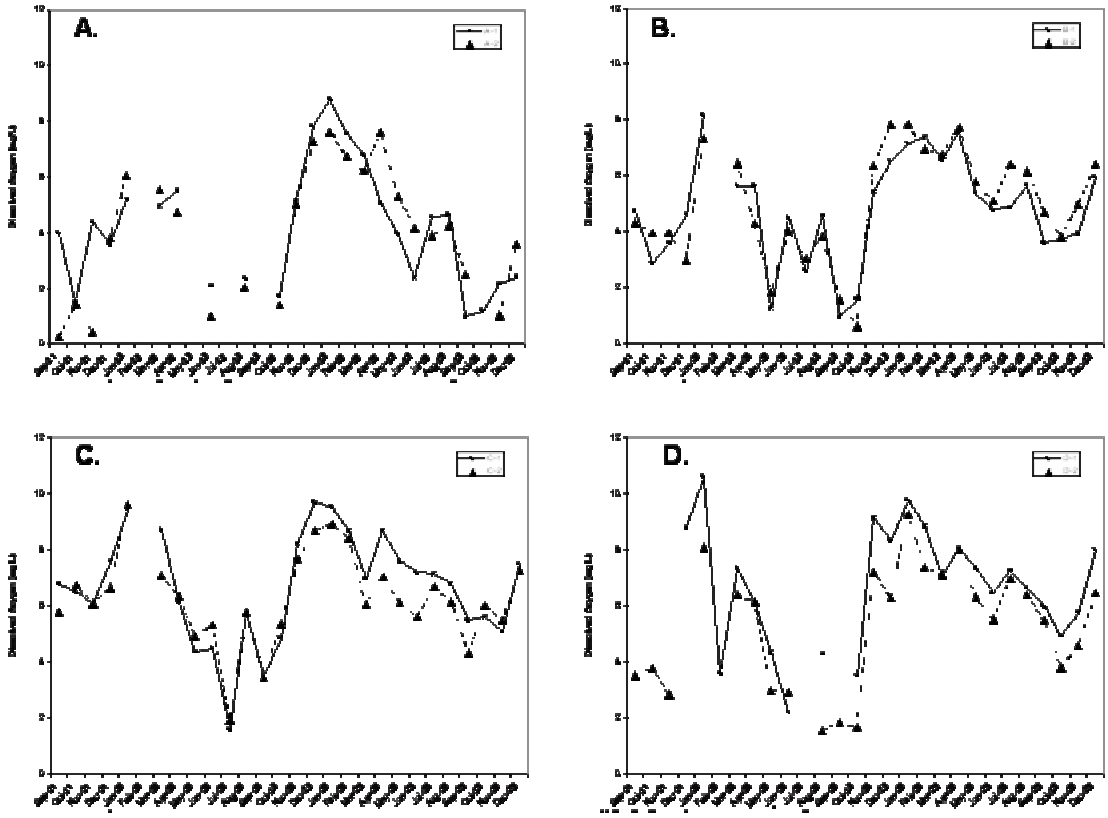

Figure 4-8. Monthly mean dissolved oxygen. A) Site A-1 and A-2. B) Site B-1 and B-2. C) Site C-1 and C-2. D) Site D-1 and D-2. * indicates no data due to equipment malfunction. ** indicates no data due to a no flow period.

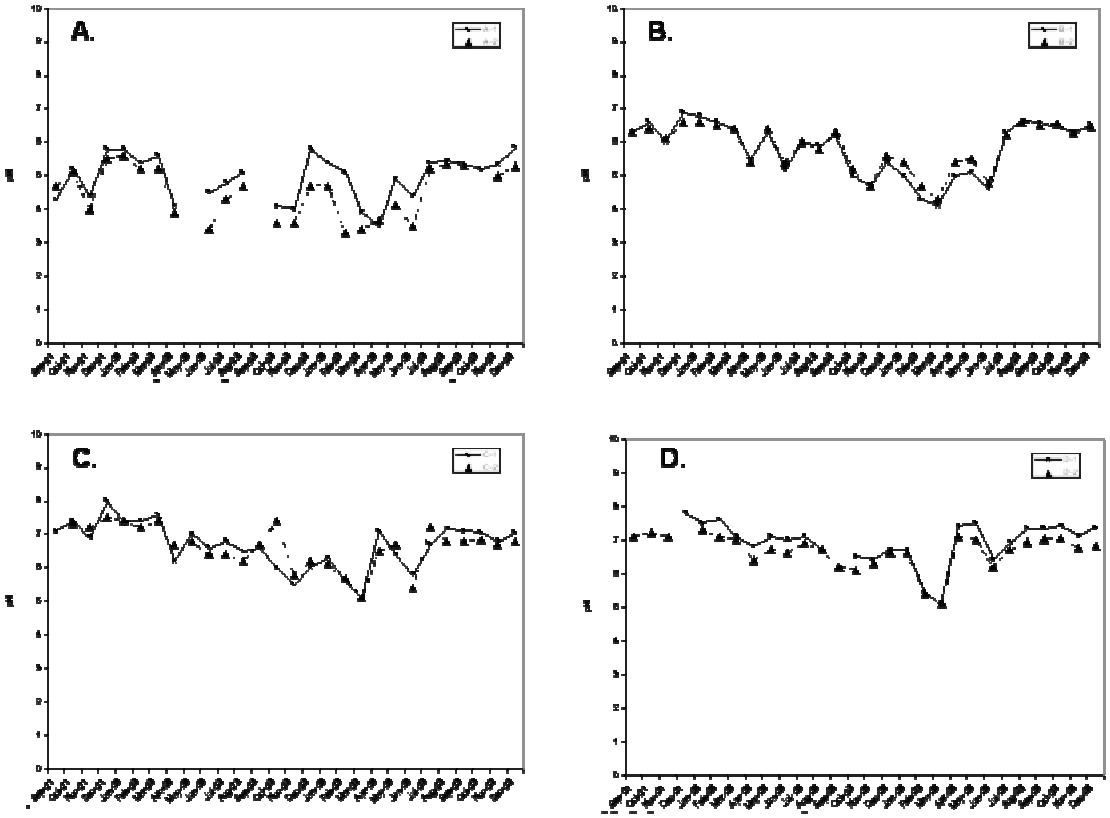

Figure 4-9. Monthly mean pH. A) Site A-1 and A-2. B) Site B-1 and B-2. C) Site C-1 and C-2. D) Site D-1 and D-2. * indicates no data due to equipment malfunction. ** indicates no data due to a no flow period.

pH ranged from 3.3 recorded in February 2003 in A2 to 8.0 in December 2001 at C1 (Table 4-3, Figure 4-9). Mean pH increased from watershed A to D (Table 4-3). Average turbidity was lowest in watershed A, followed by watershed D, with watersheds B and C having highest values. Overall, mean turbidity at all sites was very low (<10 NTU) (Table 4-3). Mean specific conductance was lowest in watershed A, with watersheds B, C, and D having highest mean values (Table 4-3).

Inorganic nitrogen in watersheds A and D was consistently very low with average concentrations < 0.05 mg/L (Table 4-3). Watersheds B and C had average concentrations an order of magnitude higher. An additional monitoring site at the upstream boundary (B-B, C-B) between the study site and an adjoining landowner for watersheds B and C

had consistently higher concentrations of inorganic N (Figure 4-10). Inorganic phosphorus average concentrations for watersheds A, B, and C were <0.0065 mg/L; however, concentrations in watershed D were an order of magnitude higher (Table 4-3).

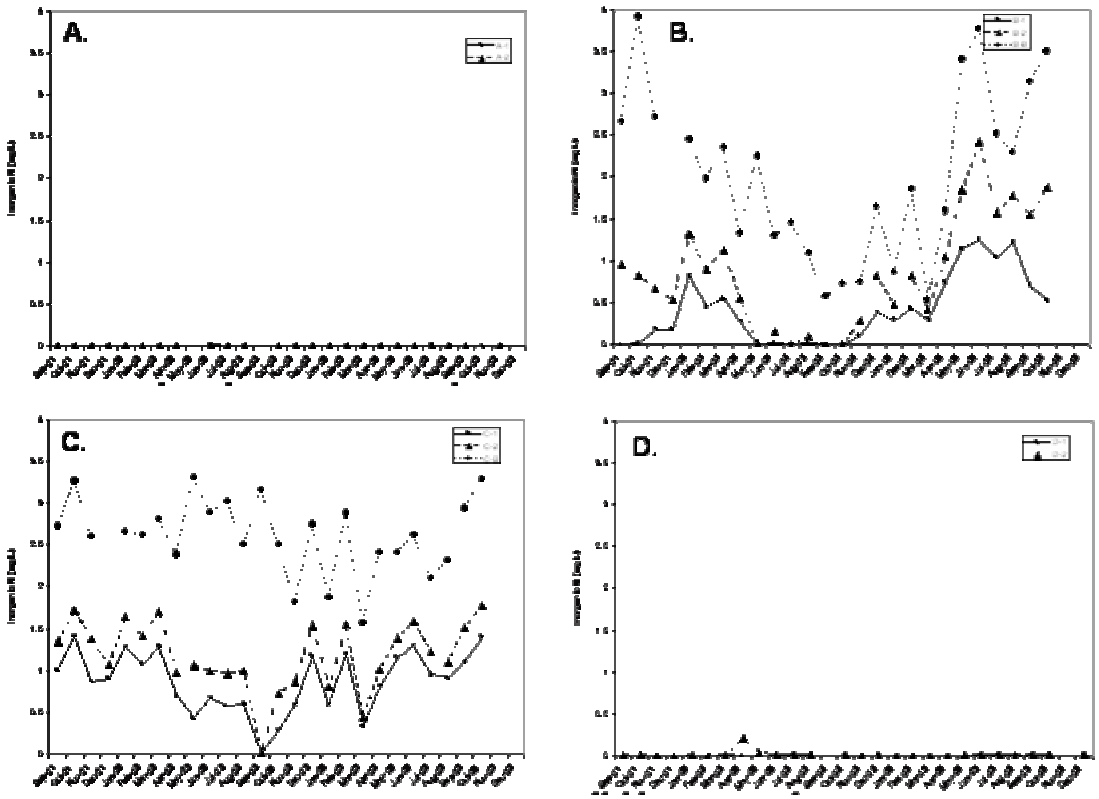

Figure 4-10. Monthly mean inorganic nitrogen. A) Site A-1 and A-2. B) Site B-1 and B-2. C) Site C-1 and C-2. D) Site D-1 and D-2. * indicates no data due to equipment malfunction. ** indicates no data due to a no flow period.

The first two axes of the water chemistry PCA explained 56% of the variation (Figure 4-11), but inclusion of the third axis explained an additional 20% of the variation (Figure 4-12). Turbidity and specific conductance were positively correlated with Axis 1 (r^2 = 0.689 and 0.718, respectively). Dissolved oxygen was negatively correlated with Axis 2 (r^2 = 0.809). Inorganic phosphorus was positively correlated with Axis 3 (r^2 = 0.773). Watershed A was characterized by lower turbidity, specific conductance, pH, and dissolved oxygen. Watershed D was characterized by higher values of inorganic

phosphorus. The PCA did not separate watershed B and C based on water chemistry variables.

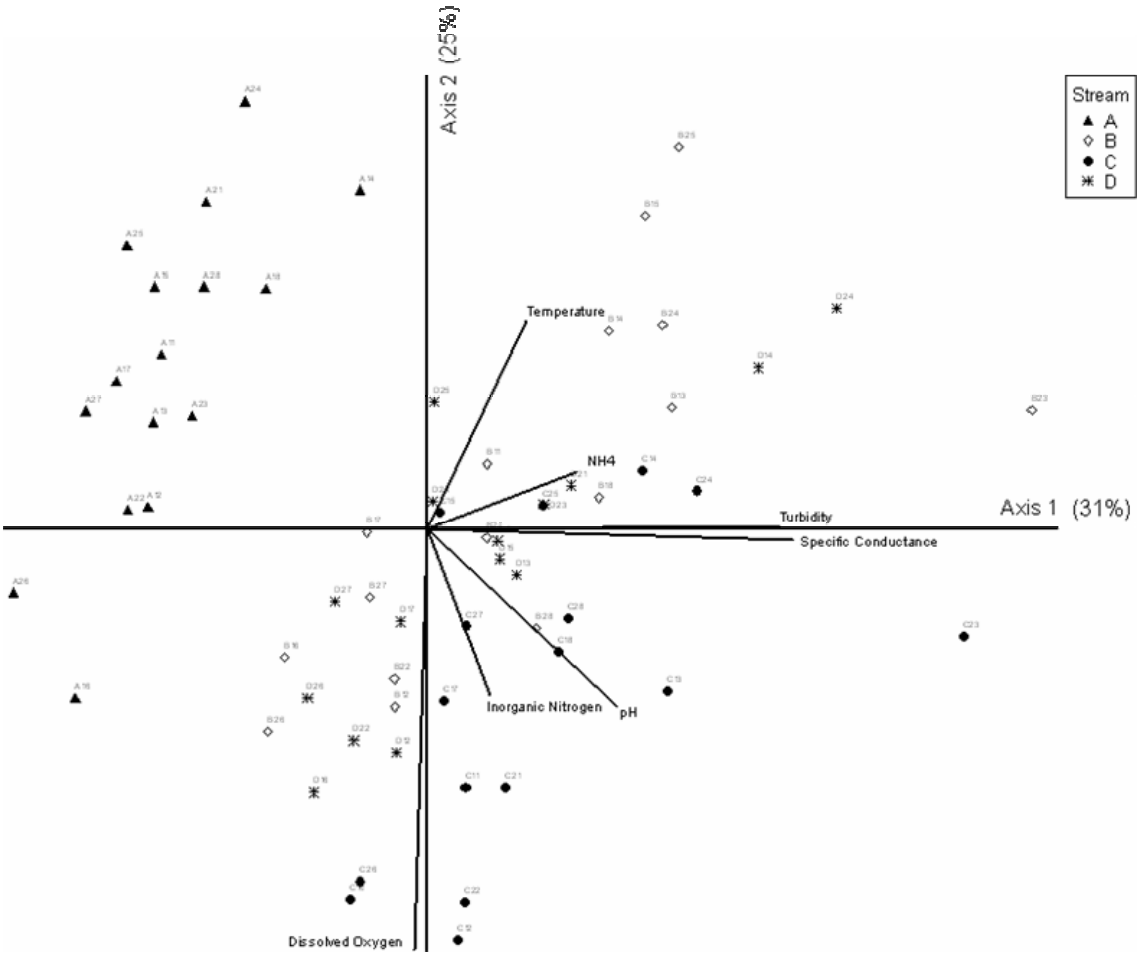

Figure 4-11. First and second axes of the principal components analysis (PCA) for in-situ water chemistry data at all sites from September 2001-December 2003.

Macroinvertebrates

A total of 17,034 individuals representing 126 taxa were collected from the four streams during the study (Appendix). Overall, dipterans were the most diverse insect order with 63 taxa, 44 of which were in the family Chironomidae. Coleoptera contributed 23 taxa, Trichoptera 12, Ephemeroptera 11, Odonata 9, Plecoptera 4 and Megaloptera 2.

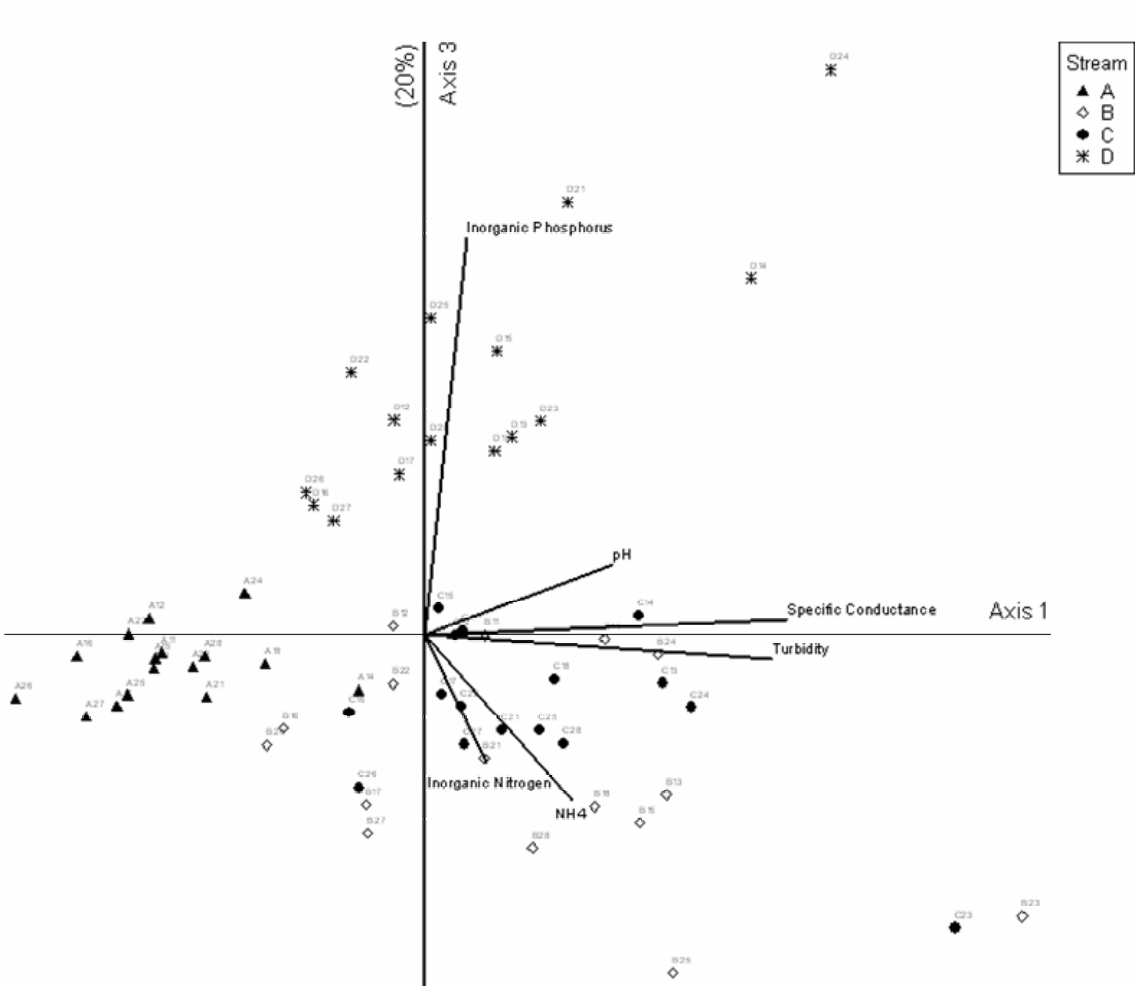

Figure 4-12. First and third axes of the principal components analysis (PCA) for in-situ water chemistry data at all sites from September 2001-December 2003.

Dipterans dominated assemblages in all streams, comprising >60% of total abundance. The composition of orders in watersheds B and C were most similar with Diptera comprising 80% and 82% of individuals, and other orders at 20% and 18%, respectively. Watershed A had < 1% of total individuals from the most sensitive orders, Ephemeroptera, Plecoptera, and Trichoptera (EPT), while watershed D had the most at 23%. Amphipods and Isopods comprised 24% and 10% of total individuals, respectively, in watershed A; whereas watersheds B, C, and D all had < 2% representation from these orders (Figure 4-13).

The following taxa considered sensitive to human disturbance in Florida (Fore 2003, as cited in FDEP 2004) were collected: *Crangonyx, Microtendipes, Parametriocnemus, Rheocricotopus, Tribelos juncundum, Acerpenna pygmaea, Ephemerella, Eurylophella, Stenonema, Habrophlebiodes, Leptophlebia, Caecidotea, Amphinemura, Ciloperla, Perlesta, Allocapnia, Triaenodes*, and *Chimarra* (Appendix A). These 18 taxa comprised 14 % of total taxa in the streams. *Cordulegaster sayi*, which has been identified as a rare and vulnerable odonate of the southeastern Piedmont and Coastal Plain (Morse et al. 1997), was collected in watersheds C and D.

The following taxa were collected only at one site, with those considered sensitive to human disturbance in Florida (Fore 2003, as cited in FDEP 2004) designated with (s). *Tribelos fuscicorne, Pseudosmitta sp*., and *Smittia sp*. were only collected in watershed A. The isopod *Caecidotea* (s) was abundant only at site A2; however, one individual was also found at A1, B1, and D1. *Paracladopelma sp., Odontomesa fulva, Rhyacophia carolina,* and *Polycentropus* were unique to watershed B. Watershed C had the greatest number of site specific taxa: *Brillia flavifrons, Baetis intercalaris, Eurylophella doris* (s), *Hexagenia, Cheumatopsyche, Triaenodes* (s), and *Agarodes*. *Rheocricotopus tuberculatus* (s), *Eukiefferiella claripennis, Zavrelia sp., Molanna blenda*, and *Allocapnia* (s) were found only at watershed D.

Abundance

Mean abundance per sample for all sites across all sampling periods was 436 individuals, and individual values ranged from a maximum at C2 (1896) in February 2003 to a minimum at B1 (19) in December 2001 (Figure 4-14).

Invertebrate abundance generally increased at each site across time. For example, abundance at B2 increased from 86 in December 2001 to 202 in February 2002, to 370 in

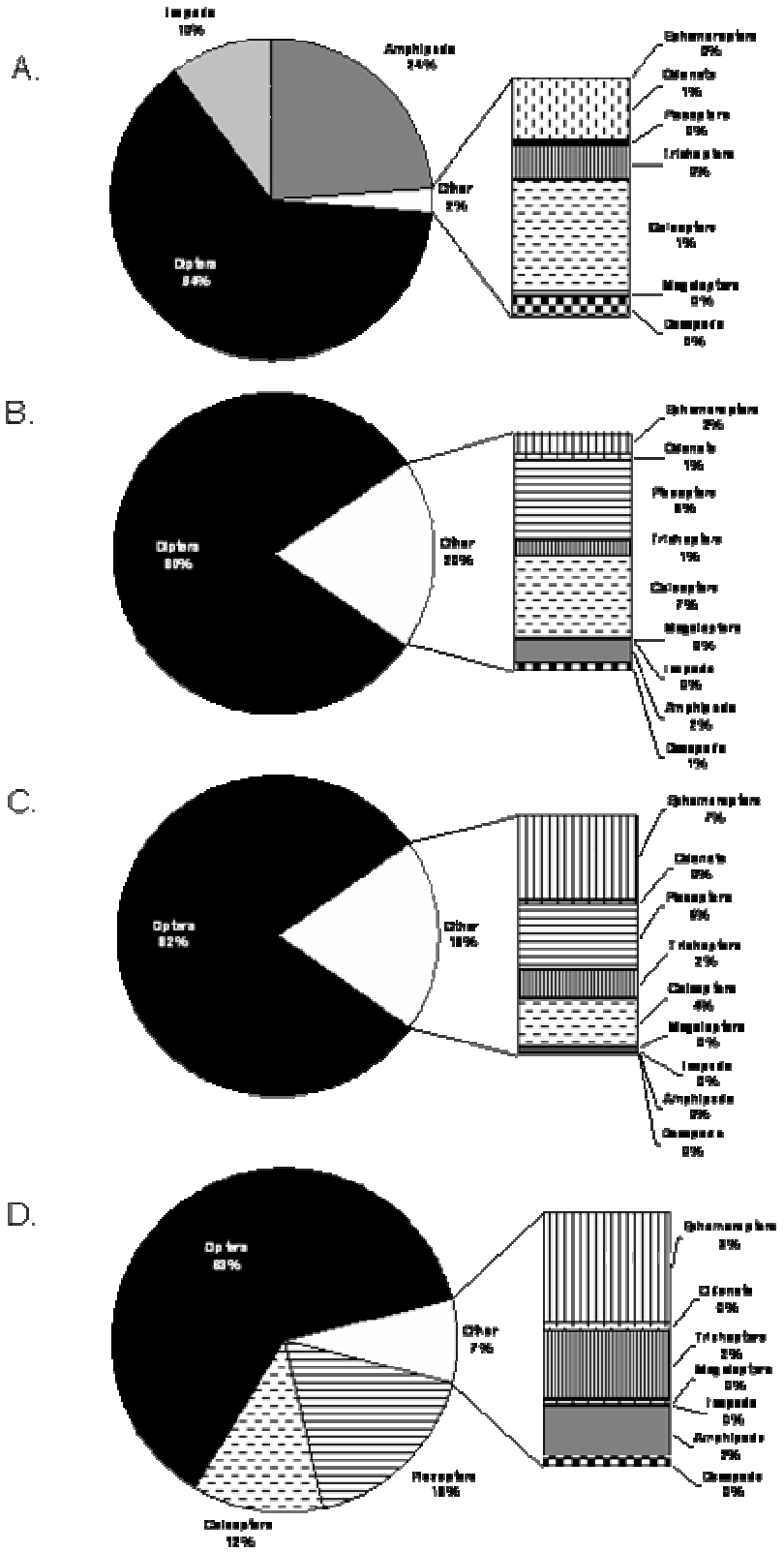

Figure 4-13. Partitioning of total abundance by invertebrate orders in study streams over the entire study period. A) Stream A. B) Stream B. C) Stream C. D) Stream D.

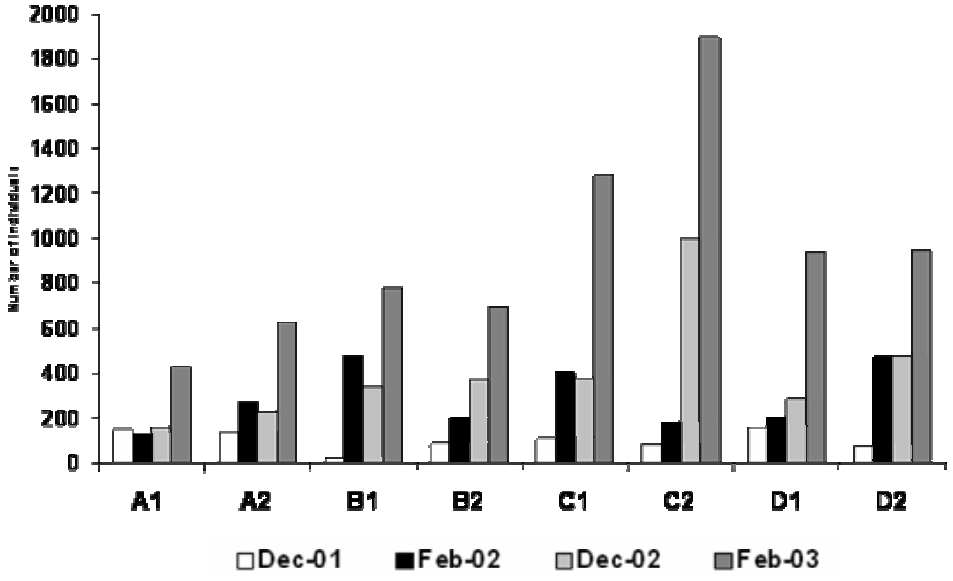

Figure 4-14. Macroinvertebrate abundance (total number of individuals) for upstream and downstream sites of each stream (A-D) over the entire study period.

December 2002, and finally 693 in February 2003. Repeated measures ANOVA (Table 4-4) indicated significant (P<0.001) differences between the four sampling periods.

There was no significant effect due to sampling position, i.e. upstream/downstream (P=0.38) or the interaction between position and time (P=0.72). The December 2001 sample had significantly lower abundance than February 2002 (P=0.03). December 2002 and February 2002 samples were not significantly different. December 2002 had significantly lower abundance than February 2003 (P<0.001) (Figure 4-15). Differences among streams were marginally significant (P=0.05) (Table 4-5). Highest mean abundance was in watershed C followed by D, B, and A (Figure 4-16).

Dominant Taxa

The dominant taxon (i.e. percent dominance of the single most abundant taxon) varied across sampling periods and between sites. Single taxon dominance ranged from 54% at D2 in February 2002, to 8% at C2 in December 2001 (Figure 4-17). 21 of the 32

Table 4-4. Repeated measures analyses for effects of time (Dec 01, Feb 02, Dec 02, Feb 03) and position (upstream vs. downstream) on macroinvertebrate metrics. Significant results are highlighted.

Metric	Source of variation	df	MS	F	p
Abundance					
	Between subjects				
	Position	1	0.013	0.892	0.38
	error	6	0.014		
	Within subjects				
	Time	1.638	2.460	21.167	<0.001
	Position*Time	1.638	0.03212	0.276	0.72
	error (time)	9.831	0.116		
% Dominant Taxon					
	Between subjects				
	Position	1	0.000	0.056	0.82
	error	6	0.006		
	Within subjects				
	Time	3	0.011	0.579	0.630
	Position*Time	3	0.006	0.301	0.820
	error (time)	18	0.020		
Total Taxa					
	Between subjects				
	Position	1	0.945	0.049	0.83
	error	6	19.299		
	Within subjects				
	Time	3	380.948	10.663	<0.001
	Position*Time	3	15.115	0.423	0.73
	error (time)	18	35.726		
EPT Taxa					
	Between subjects				
	Position	1	3.125	0.221	0.65
	error	6	14.141		
	Within subjects				
	Time	3	33.458	7.86	0.001
	Position*Time	3	4.333	1.018	0.40
	error (time)	18	4.257		
Chironomidae Taxa					
	Between subjects				
	Position	1	0.195	0.091	0.77
	error	6	2.154		
	Within subjects				
	Time	3	30.198	2.852	0.06
	Position*Time	3	19.531	1.845	0.17
	error (time)	18	10.587		
% Chironomidae					
	Between subjects				
	Position	1	0.002	0.359	0.57
	error	6	0.006		
	Within subjects				
	Time	3	0.044	2.059	0.14
	Position*Time	3	0.126	5.921	0.005
	error (time)	18	0.021		
% Diptera					
	Between subjects				
	Position	1	0.017	0.839	0.39
	error	6	0.021		
	Within subjects				
	Time	3	0.008	0.27	0.84
	Position*Time	3	0.01	0.352	0.78
	error (time)	18	0.028		

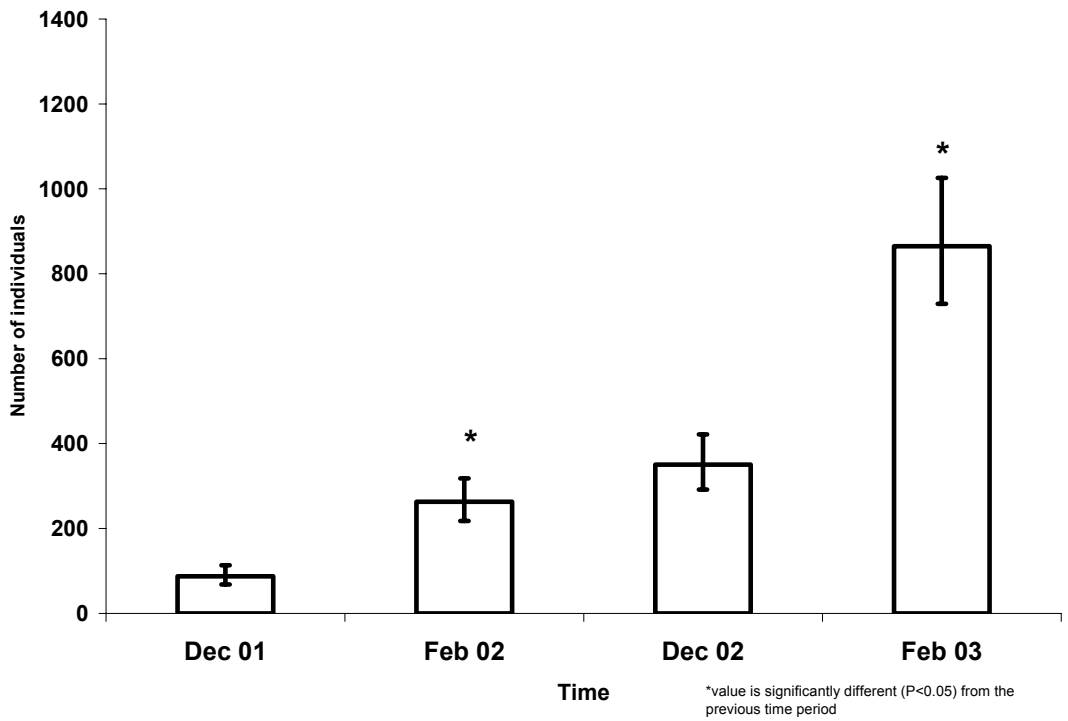

Figure 4-15. Mean macroinvertebrate abundance individual sampling periods with standard error and repeated contrast results (alpha = 0.05). Letters above bars indicate statistical groupings.

samples were dominated by Chironomidae with 7 of the 21 dominated by *Parametriocnemus*, a taxon sensitive to disturbance in Florida (Fore 2003, as cited in FDEP 2004). In the February 2002 samples, 5 of 8 sites were dominated by the predator *Conchapelopia sp*.. Repeated measures ANOVA found no significant effects due to time of sampling, position, or an interaction between position and time (Table 4-4). No significant differences were detected between streams (Table 4-5).

Total Taxa

Overall taxa richness displayed no clear trend between sites, but the December 2001 sampling period had the lowest values. D1 had the highest taxa richness (42) in December 2002, and B1 had the lowest (7) in December 2001 (Figure 4-18).

Table 4-5. Repeated measures analyses for effects of time (Dec 01, Feb 02, Dec 02, Feb 03) and stream (A, B, C, D) on macroinvertebrate metrics. Significant results are highlighted.

Metric	Source of variation	df	MS	F	p
Abundance					
Between subjects					
	Stream	3	0.027	6.407	0.05
	error	4	0.004		
Within subjects					
	Time	3	1.344	26.272	<0.001
	Time*Stream	9	0.065	1.263	0.34
	error (time)	12	0.051		
% Dominant Taxon					
Between subjects					
	Stream	3	0.006	1.544	0.33
	error	4	0.004		
Within subjects					
	Time	3	0.011	1.312	0.31
	Time*Stream	9	0.03	3.425	0.02
	error (time)	12	0.009		
Total Taxa					
Between subjects					
	Stream	3	35.258	12.858	0.01
	error	4	2.742		
Within subjects					
	Time	1.665	686.477	9.186	0.010
	Time*Stream	4.994	38.199	0.511	0.76
	error (time)	6.659	74.728		
EPT Taxa					
Between subjects					
	Stream	3	25.052	7.821	0.03
	error	4	3.203		
Within subjects					
	Time	3	33.458	6.015	0.01
	Time*Stream	9	2.542	0.457	0.87
	error (time)	12	5.562		
Chironomidae Taxa					
Between subjects					
	Stream	3	0.987	0.389	0.76
	error	4	2.539		
Within subjects					
	Time	1.94	46.698	2.072	0.19
	Time*Stream	5.82	12.763	0.566	0.74
	error (time)	7.76	22.535		
% Chironomidae					
Between subjects					
	Stream	3	0.005	1.172	0.42
	error	4	0.005		
Within subjects					
	Time	3	0.044	0.968	0.44
	Time*Stream	9	0.024	0.535	0.82
	error (time)	12	0.045		
% Diptera					
Between subjects					
	Stream	3	0.031	2.537	0.19
	error	4	0.012		
Within subjects					
	Time	1.598	0.014	0.418	0.63
	Time*Stream	4.793	0.066	1.939	0.21
	error (time)	6.391	0.034		

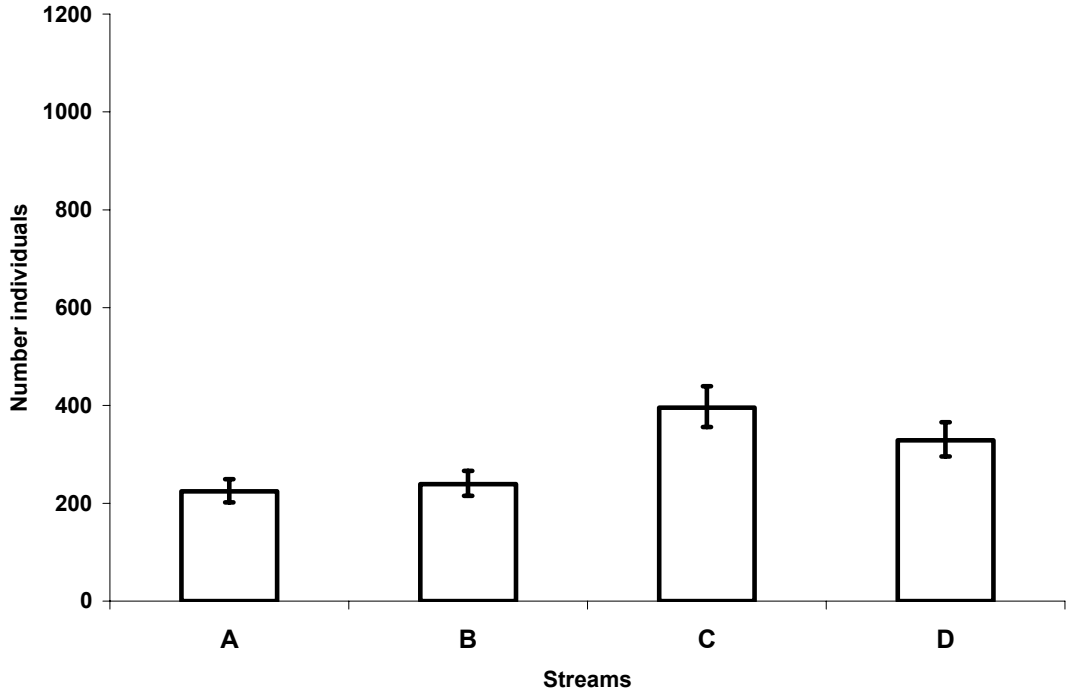

Figure 4-16. Means for macroinvertebrate abundance for all sites within each stream for all time periods with standard error.

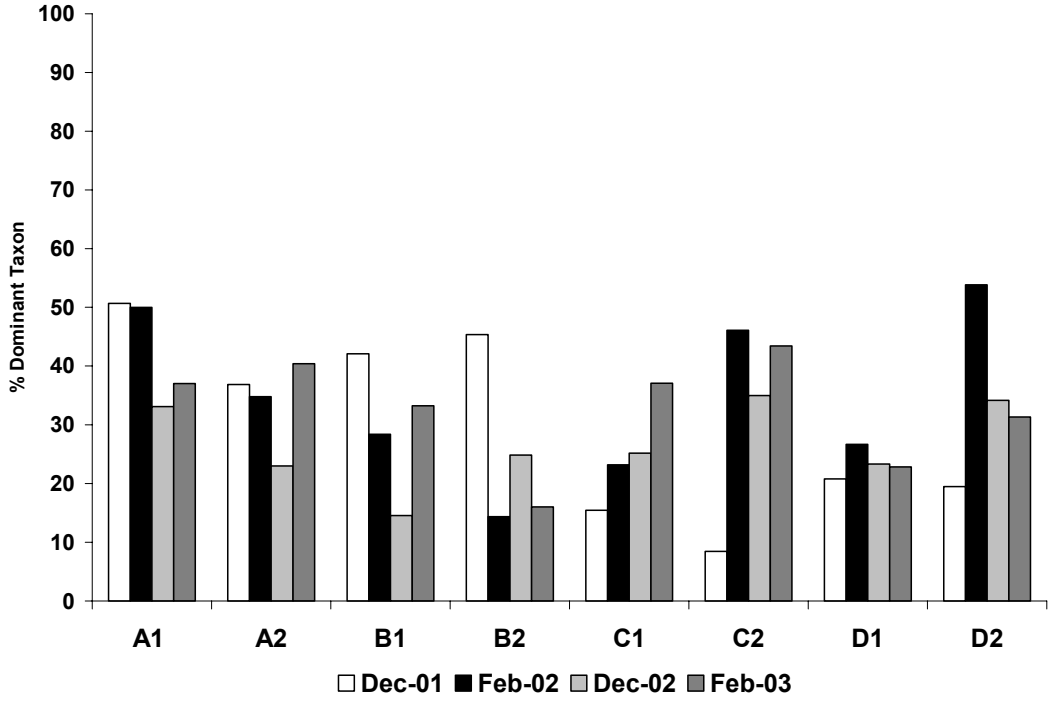

Figure 4-17. Percent dominant taxon for upstream and downstream sites of each stream (A-D) over the entire study period.

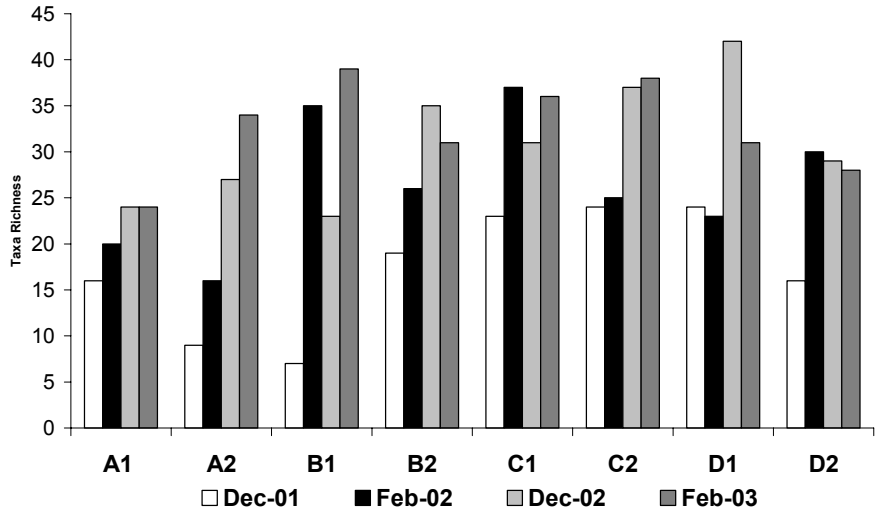

Figure 4-18. Taxa richness for upstream and downstream sites of each stream (A-D) over the entire study period.

Significant (P<0.001) differences between four sampling periods (time) were detected by repeated measures ANOVA (Table 4-4). There was no significant effect due to sampling position (upstream/downstream) or the interaction between position and time. The December 2001 sample had significantly lower taxa richness than February 2002 (P=0.03). The remaining sampling periods were not significantly different (Figure 4-19). Differences (P=0.01) were detected between streams (Table 4-5); however, multiple comparisons between streams were not significant. Means showed highest total taxa in watershed C followed by D, B, then A (Figure 4-20).

EPT Taxa

The number of EPT taxa (Ephemeroptera, Trichoptera, and Plecoptera taxa) was consistently highest at C1. Both sites in watershed A had the lowest number of EPT taxa (Figure 3.22). Repeated measures ANOVA (Table 4-4) indicated significant (P=0.001) differences between the four sampling periods (time), but there was no significant effect due to sampling position (upstream/downstream) or the interaction between position and

time. Differences between the December 2001 and February 2002 sample were marginally significant (P=0.1). The December 2002 sample had significantly lower EPT taxa than February 2003 (P=0.002). The remaining sampling periods were not significantly different (Figure 4-22). Differences (P=0.03) were detected between streams (Table 4-5), with A having significantly lower EPT taxa than C (P=0.03) (Figure 4-23).

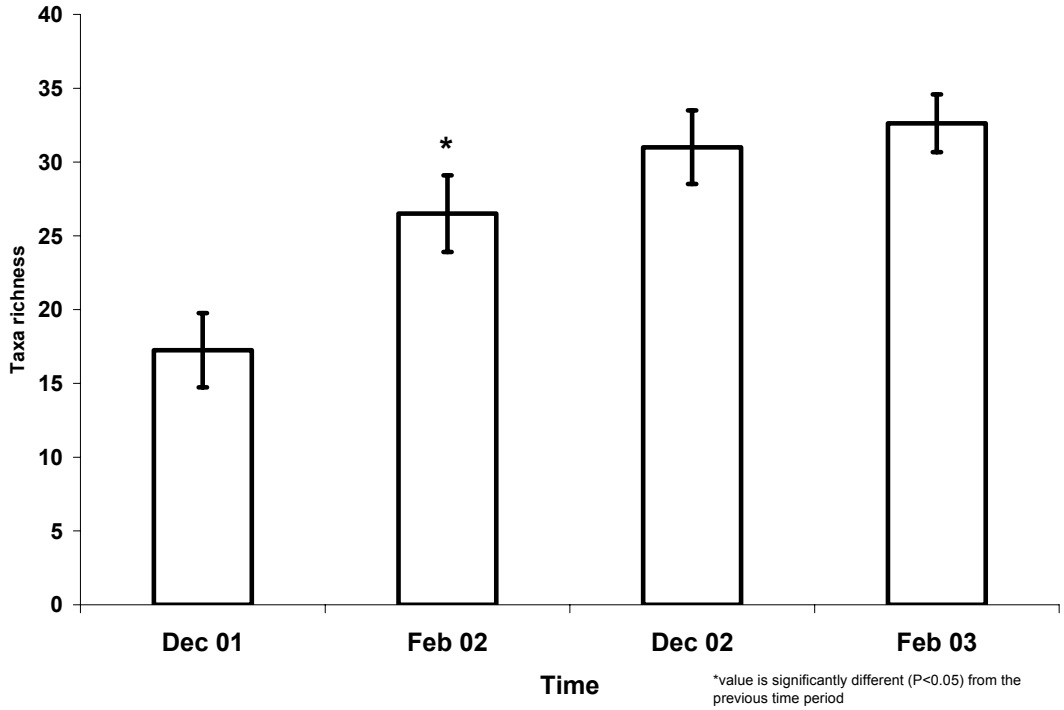

Figure 4-19. Mean taxa richness for individual sampling periods with standard error and repeated contrast results (alpha = 0.05). Letters above bars indicate statistical groupings.

Chironomidae Taxa

A total of 44 species of Chironomidae were identified from four subfamilies (Chironominae, Orthocladiinae, Prodiamesinae, Tanypodinae). The subfamilies were represented at all sites with the exception of the Prodiamesinae (Figure 4-24), which was represented by a single species, *Odontomesa fulva*, and was only found at site B1 on one occasion. B2 had the lowest Chironomidae taxa richness in December 2001 (3), and D2

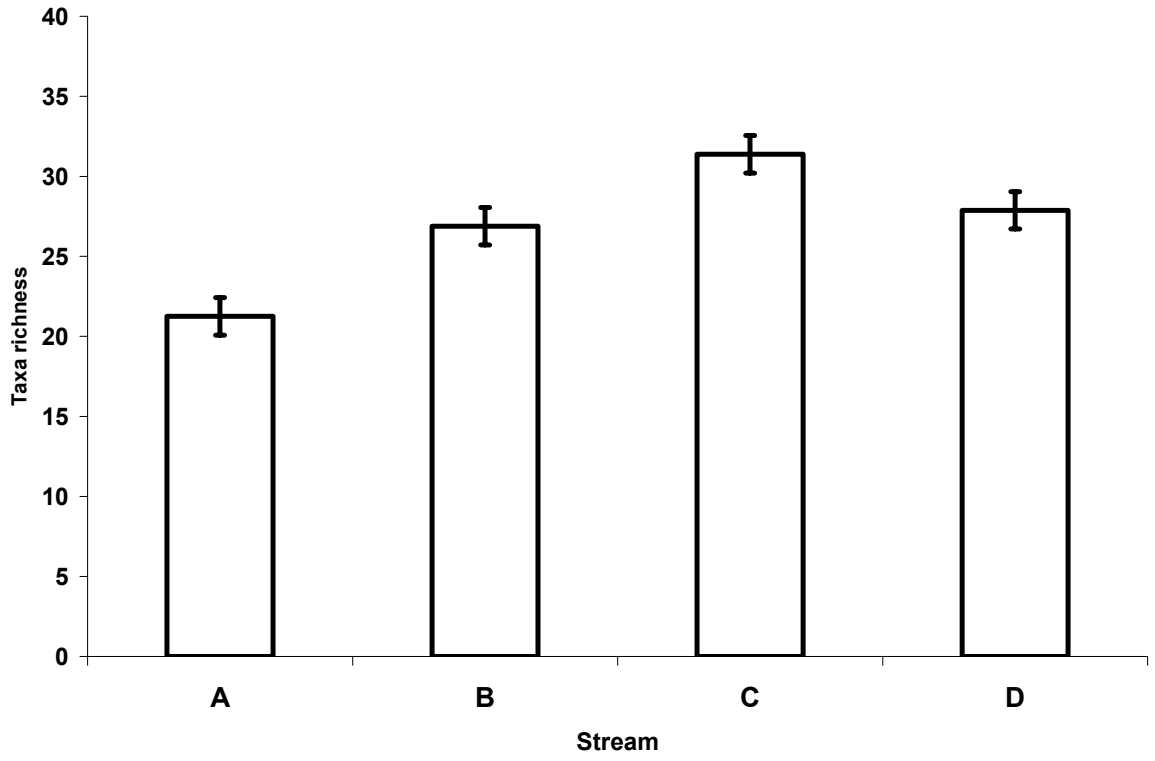

Figure 4-20. Taxa richness for each stream for all time periods with standard error.

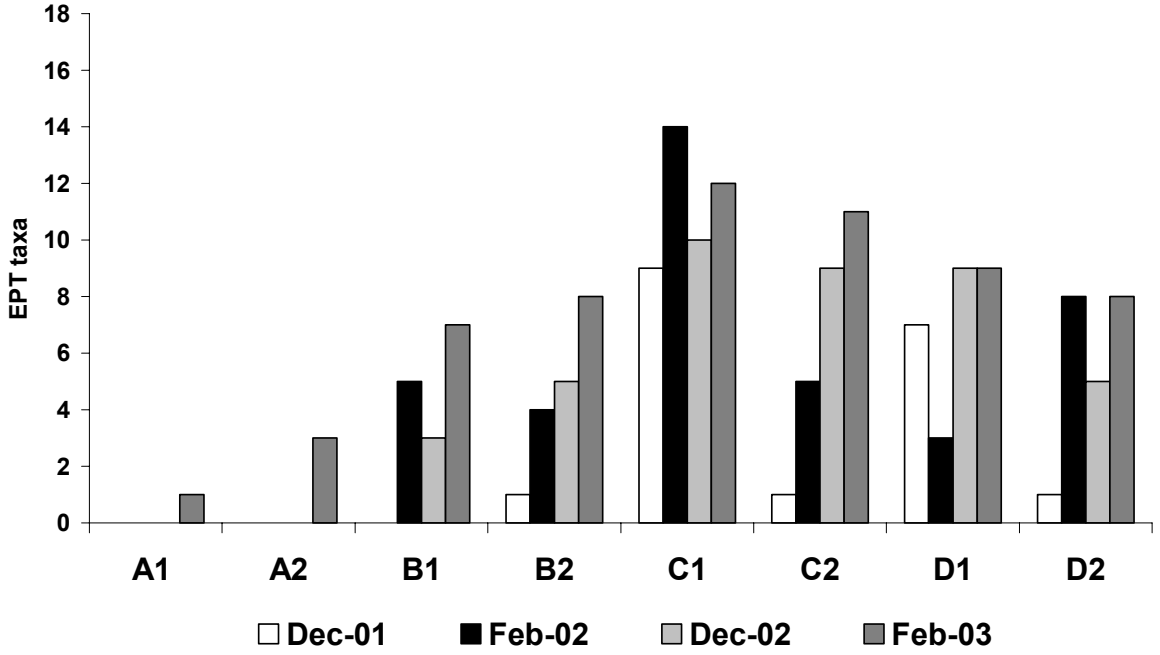

Figure 4-21. Total Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa for upstream and downstream sites of individual streams (A-D) over the entire study period.

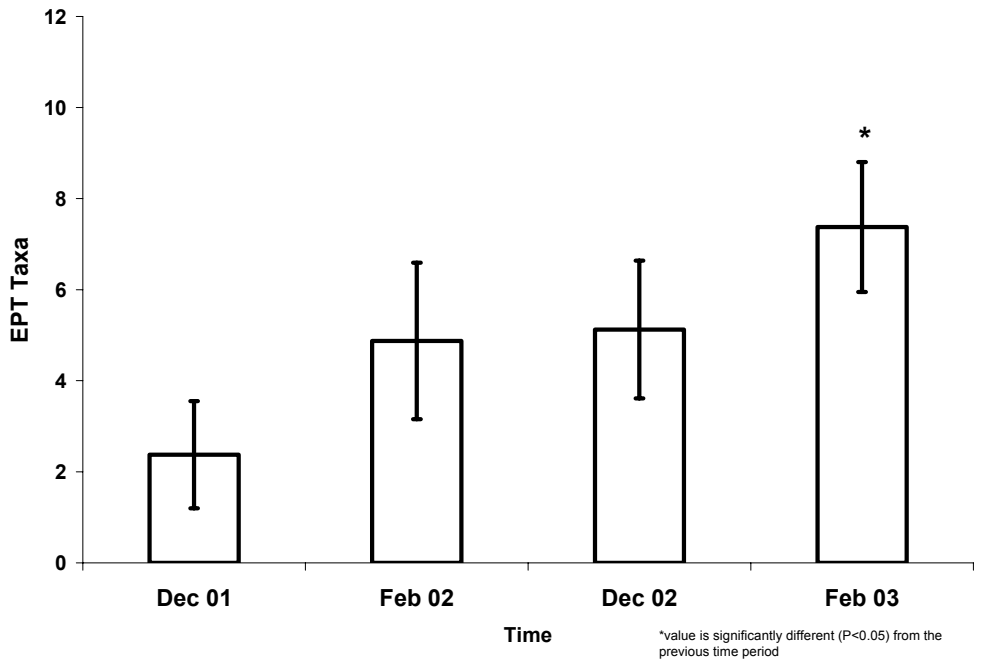

Figure 4-22. Mean Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa within each time period with standard error and repeated contrast results (alpha = 0.05) across time. Letters above bars indicate statistical groupings.

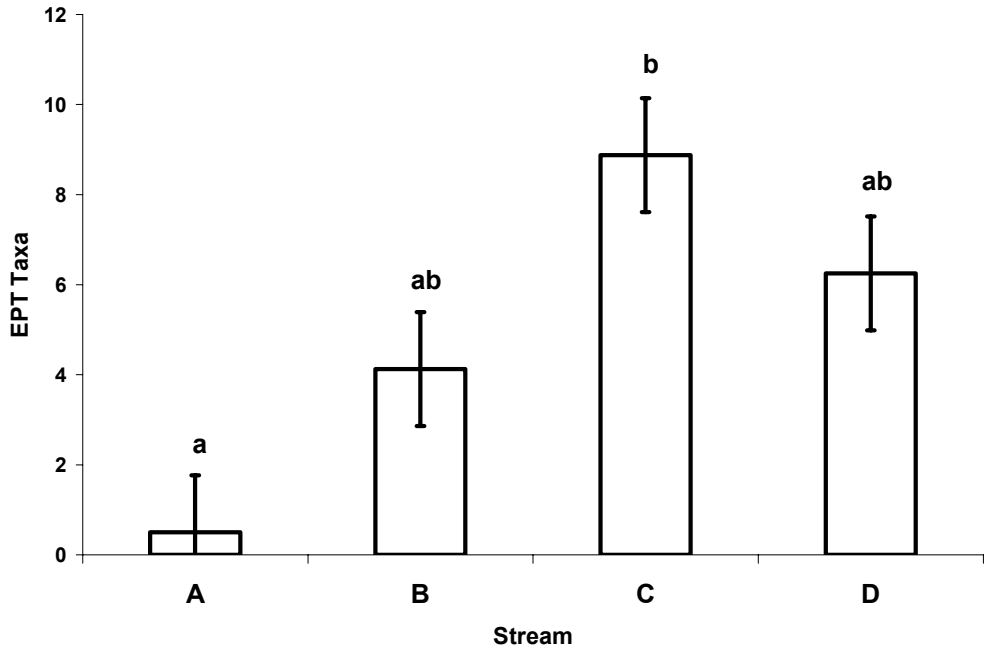

Figure 4-23. Mean Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa for all sites within each stream for all time periods and pairwise multiple comparison test results (Tukey's honestly significant difference (HSD) test, alpha = 0.05). Letters above bars indicate statistical groupings.

had the greatest (17) in December 2002 (Figure 4-25). *Tanytarsus sp., Tribelos sp., Parametriocnemus sp., Conchapelopia sp.*, and *Zavrelimyia sp*. occurred at all sampling sites. Repeated measures ANOVA detected no differences due to time, position, or site (Table 4-4 and 4-5).

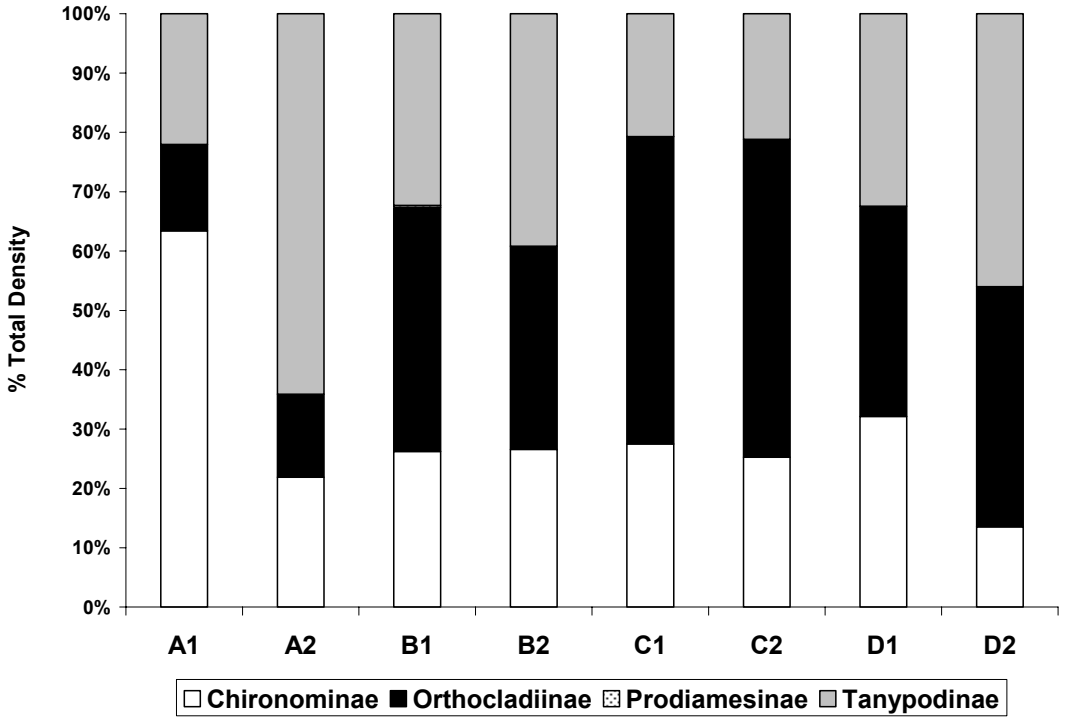

Figure 4-24. Mean subfamily composition of Chironomidae in individual streams (A-D) over the entire sampling period.

Percent Chironomidae

The lowest and highest percentage of a sample composed of Chironomidae was found, respectively, at site D2 with 19% in December 2002 and 82% in December 2001 (Figure 4-26). Seventy-two percent of samples contained >50% Chironomidae. There were no significant effects due to time of sampling, sampling position, or interaction between position and time (Table 4-4 and 4-5).

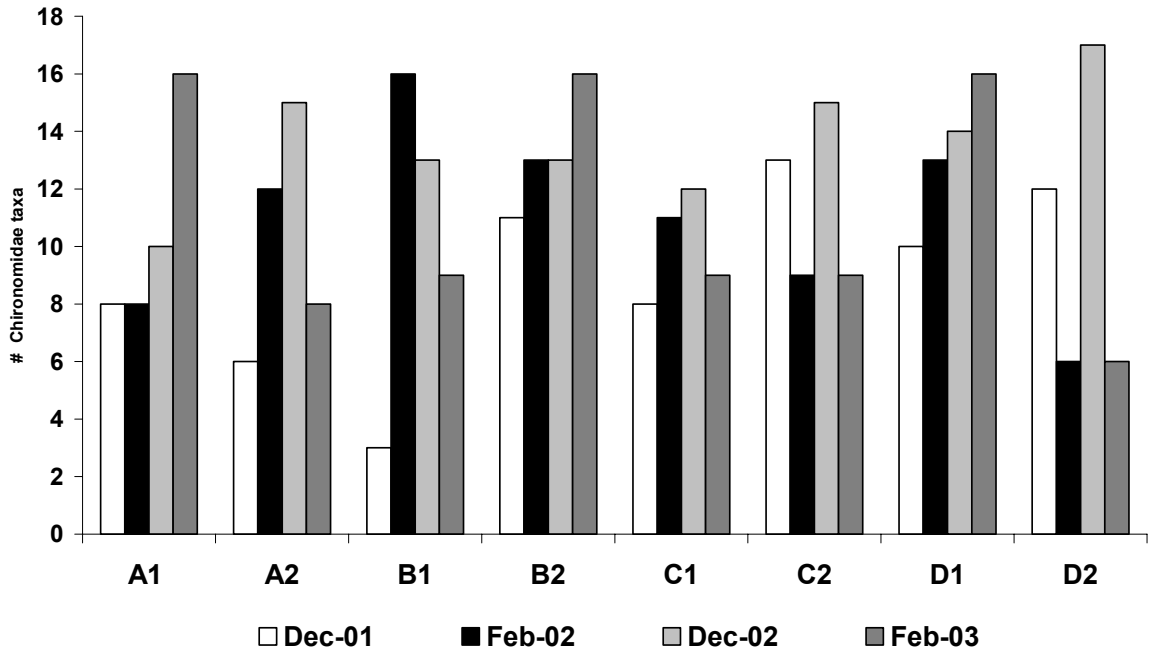

Figure 4-25. Number of Chironomidae taxa for upstream and downstream sites of individual streams (A-D) over the entire study period.

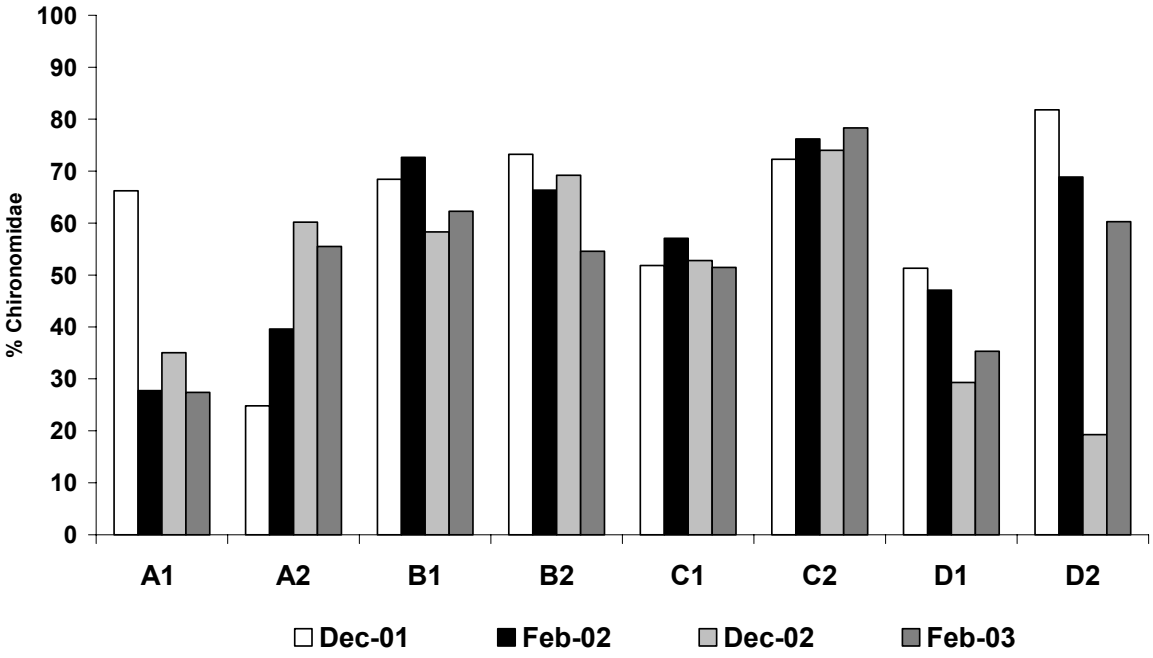

Figure 4-26. Percentage of total abundance contributed by Chironomidae for upstream and downstream sites of individual streams (A-D) over the entire study period.

Percent Diptera

Eighty-four percent of samples had >50% Diptera, and 47% of samples had >75% Diptera. Minimum and maximum values for percent Diptera were December 2001 at A2 (29%) and D2 (96%) (Figure 4-27) Repeated measures ANOVA detected no differences due to time, position, or site (Table 4-4 and 4-5).

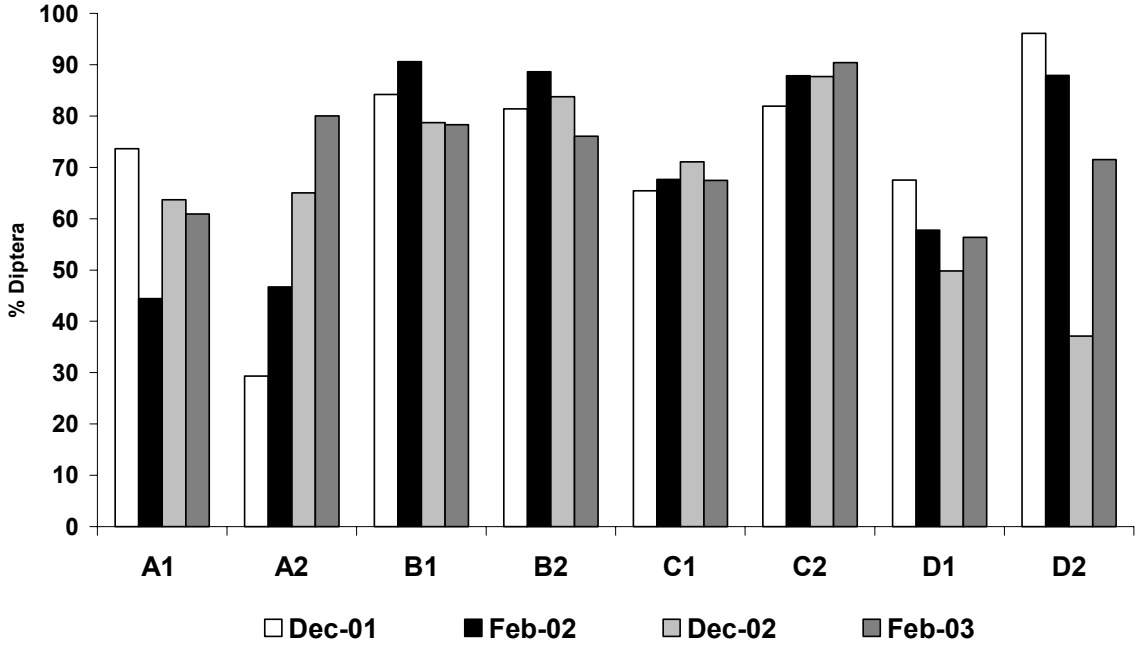

Figure 4-27. Percentage of total abundance contributed by Diptera for upstream and downstream sites of individual streams (A-D) over the entire study period.

Percent Elmidae

All sites had <20% of the family Elmidae, which are a family of beetles which prefer swifter parts of streams such as oxygen rich riffles (Merritt and Cummins 1996). Sites in A and B had no occurrences of elmids in the December 2001 and February 2002 sampling period with site A1 having no occurrences in any sample (Figure 4-28).

Repeated measures ANOVA detected no significant differences due to position (Table 4-6), but significant differences due to time (P<0.001), site (P=0.003), and an interaction between time and site (P=0.002) (Table 4-7).

Figure 4-29 illustrates the interaction because the lines are not parallel, implying that the effect of site upon percent Elmidae depends upon the time period examined. This interaction seems to be the result of the December 2001 sampling period having highest mean percent Elmidae at C, where the remaining sampling periods have highest mean percent Elmidae at D. The December 2002 sampling period had the highest mean percent Elmidae compared to the remaining sampling periods (Figure 4-29) and D had highest mean percent Elmidae, followed by C, B, and A.

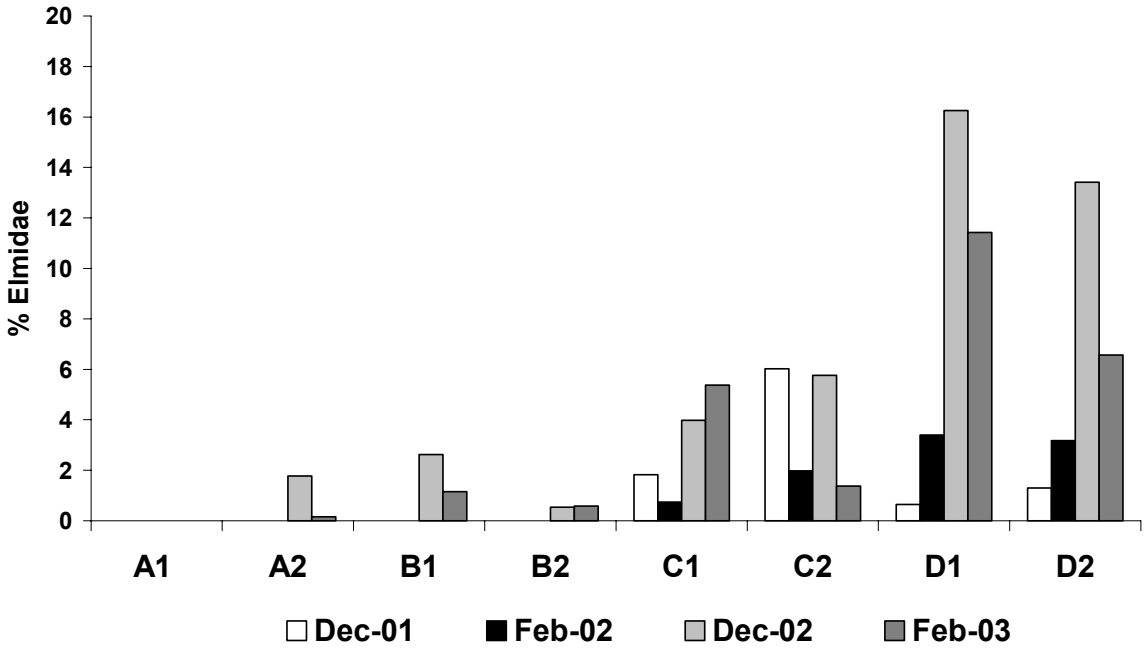

Figure 4-28. Percent of the total assemblage represented by Elmidae for upstream and downstream sites of individual streams (A-D) over the entire study period.

Table 4-6. Repeated measures analyses for the effects of time (Dec 01, Feb 02, Dec 02, Feb 03) and position (upstream vs. downstream) on macroinvertebrate metrics. Significant results are highlighted.

Clinger Taxa				
Between subjects				
Position	1	23.205	5.224	0.06
error	6	4.442		
Within subjects				
Time	3	1.841	0.945	0.44
Position*Time	3	11.654	5.98	0.005
error (time)	18	1.949		
% Filtering Collectors				
Between subjects				
Position	1	0.006	2.71	0.15
error	6	0.002		
Within subjects				
Time	3	0.023	1.99	0.15
Position*Time	3	0.013	1.147	0.35
error (time)	18	0.012		
% Elmidae				
Between subjects				
Position	1	0.000	0.017	0.89
error	6	0.004		
Within subjects				
Time	1.218	0.009	3.854	0.08
Position*Time	1.218	0.001	0.496	0.53
error (time)	7.307	0.002		
FL SCI				
Between subjects				
Position	1	204.905	0.405	0.54
error	6	505.827		
Within subjects				
Time	3	75.279	1.031	0.40
Position*Time	3	208.151	2.852	0.06
error (time)	18	72.992		
GA AAS				
Between subjects				
Position	1	0.633	0.014	0.91
error	6	45.82		
Within subjects				
Time	3	276.865	19.850	<0.001
Position*Time	3	5.531	0.397	0.75
error (time)	18	13.948		
EPD				
Between subjects				
Position	1	8.000	0.044	0.84
error	6	183.875		
Within subjects				
Time	3	327.167	3.220	0.04
Position*Time	3	89.500	0.881	0.47
error (time)	18	101.611		
EPD (200 individual subsample)				
Between subjects				
Position	1	5.281	0.025	0.87
error	6	209.99		
Within subjects				
Time	3	323.208	3.357	0.04
Position*Time	3	53.375	0.554	0.65
error (time)	18	96.292		

Table 4-7. Repeated measures analyses for the effects of time (Dec 01, Feb 02, Dec 02, Feb 03) and stream (A, B, C, D) on macroinvertebrate metrics. Significant results are highlighted.

Clinger Taxa				
Between subjects				
Stream	3	8.601	1.43	0.35
error	4	6.014		
Within subjects				
Time	3	1.841	0.367	0.77
Time*Stream	9	1.098	0.219	0.98
error (time)	12	5.013		
% Filtering Collectors				
Between subjects				
Stream	3	0.004	1.744	0.29
error	4	0.002		
Within subjects				
Time	3	0.023	1.97	0.17
Time*Stream	9	0.012	1.026	0.47
error (time)	12	0.012		
% Elmidae				
Between subjects				
Stream	3	0.002	34.075	0.003
error	4	0.000		
Within subjects				
Time	3	0.003	13.88	<0.001
Time*Stream	9	0.002	6.464	0.002
error (time)	12	0.000		
FL SCI				
Between subjects				
Stream	3	215.518	5.275	0.07
error	4	40.853		
Within subjects				
Time	1.158	194.962	1.137	0.35
Time*Stream	3.475	329.229	1.921	0.25
error (time)	4.633	171.406		
GA AAS				
Between subjects				
Stream	3	87.258	25.327	0.005
error	4	3.445		
Within subjects				
Time	3	276.865	27.149	<0.001
Time*Stream	9	16.142	1.583	0.22
error (time)	12	10.198		
EPD				
Between subjects				
Stream	3	329.188	10.646	0.02
error	4	30.922		
Within subjects				
Time	3	327.167	3.501	0.50
Time*Stream	9	108.472	1.161	0.390
error (time)	12	93.437		
EPD (200 individual subsample)				
Between subjects				
Stream	3	328.802	4.717	0.08
error	4	69.703		
Within subjects				
Time	3	323.208	4.389	0.02
Time*Site	9	112.181	1.523	0.24
error (time)	12	73.646		

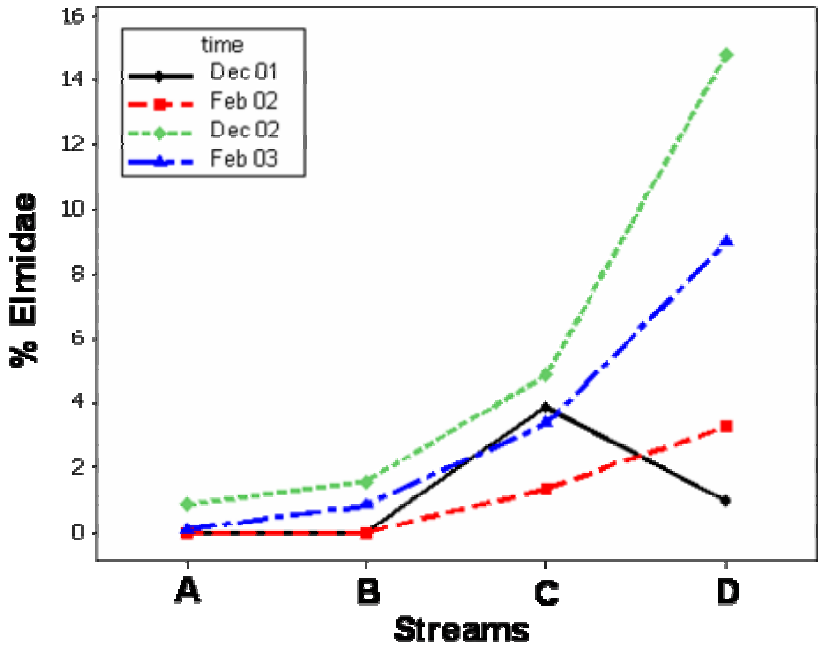

Figure 4-29. Interaction plot (data means) for percent Elmidae. Site 1, 2, 3, 4, represents A, B, C, D, respectively and Time 1,2,3,4 represents Dec 01, Feb 02, Dec 02, and Feb 03, respectively.

Feeding Type and Habitat Type

The relative contribution of functional feeding groups to the total assemblage of each stream varied. The assemblages of B, C, and D were co-dominated by shredders, predators, and collectors, while collectors were dominant in A (Figure 4-30). Within the collector functional feeding group, collector-gatherers were dominant over filter feeders.

Each site for all sampling periods had <15% filter feeders (Figure 4-31), which are thought to be sensitive in low gradient streams with high percent filter feeders indicating healthy coastal plain streams (Barbour et al. 1996). Repeated measures ANOVA detected no differences due to time, position, or site (Table 4-6 and Table 4-7).

The number of clinger taxa generally increased at each site from December 2001 through February 2003 (Figure 4-32); however, there was no significant effect due to

time of sampling, sampling position, or interaction between position and time (Table 4-6 and Table 4-7).

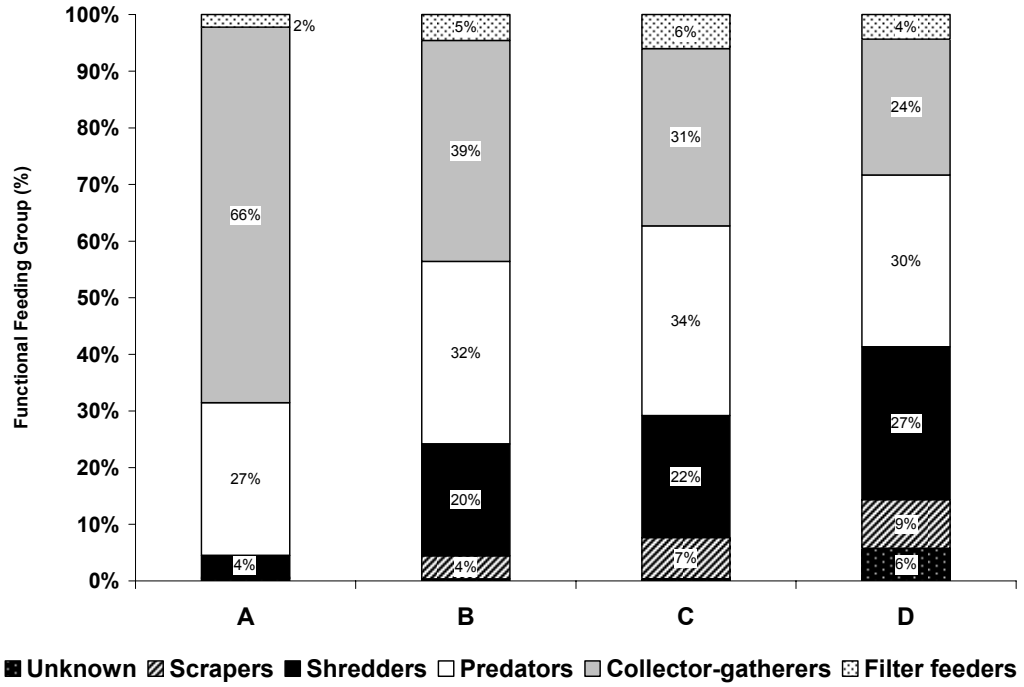

Figure 4-30. Percentage of the total assemblage contributed by individual functional feeding groups for individual streams (A-D) over the entire study period.

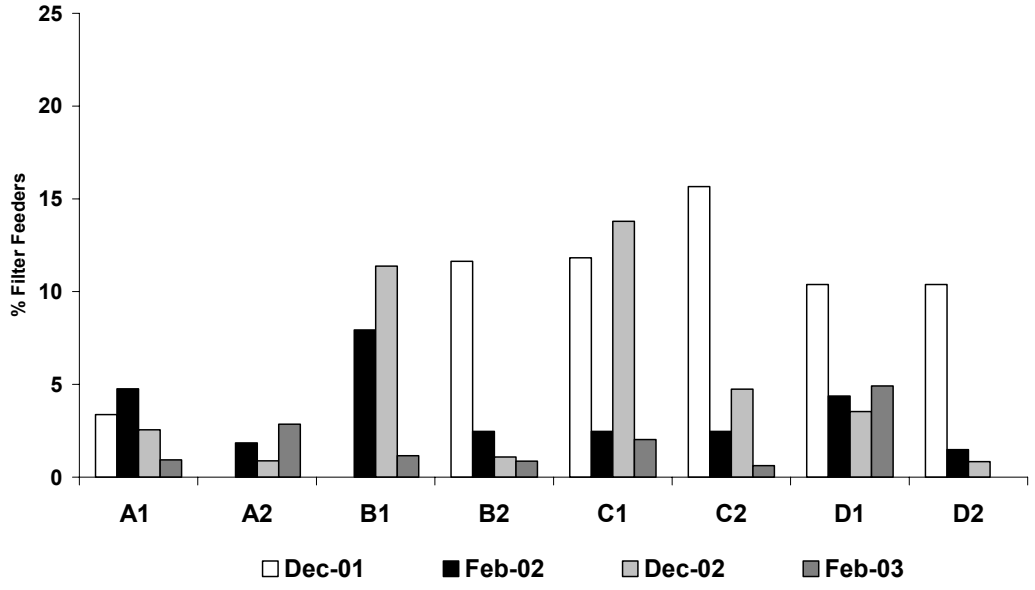

Figure 4-31. Percentage of the total assemblage represented by filter feeders for upstream and downstream sites of individual streams (A-D) over the entire study period.

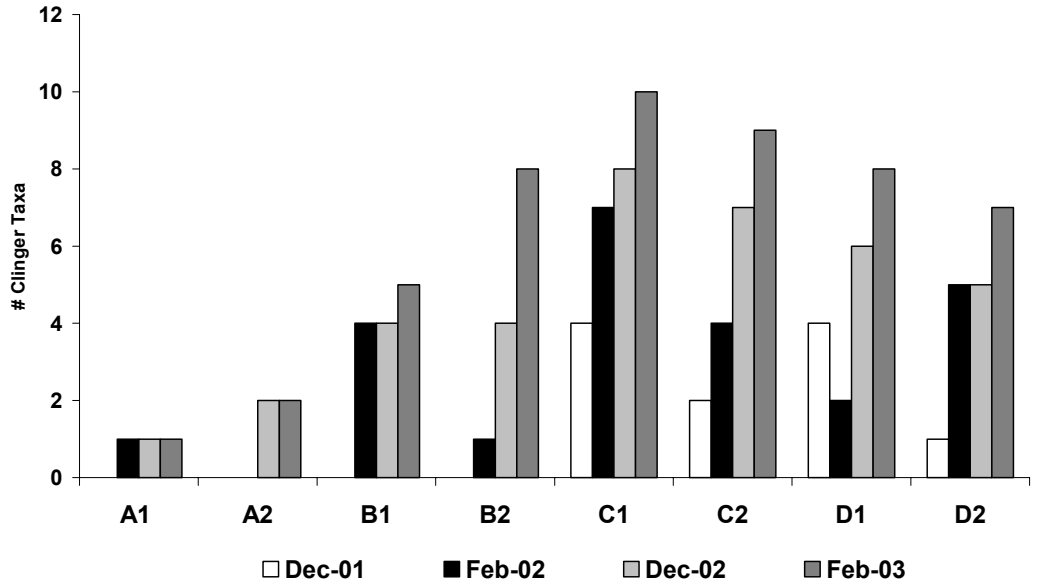

Figure 4-32. Clinger taxa for upstream and downstream sites of individual streams (A-D) over the entire study period.

Biotic Indices

The Florida Stream Condition Index (SCI) scored A1 as Very Poor for every sampling period (Figure 4-33). December 2001, February 2002, and February 2003 samples for A2 scored Poor, while the December 2002 sample fell within the Very Poor category. The earliest sample for B1, December 2001, scored Very Poor, with the remaining samples in the Poor category. December 2001, December 2002, and February 2003 samples for B2 scored Poor, while February 2002 was Fair. C1 had the greatest number of samples (December 2001 and December 2002) in the Fair category, with the remaining samples, February 2002 and February 2003, listed as Poor. All C2 samples scored Poor. December 2001, February 2002, and February 2003 samples for D1 scored Poor, while the December 2002 sample fell into the Fair category. All D2 samples scored Poor.

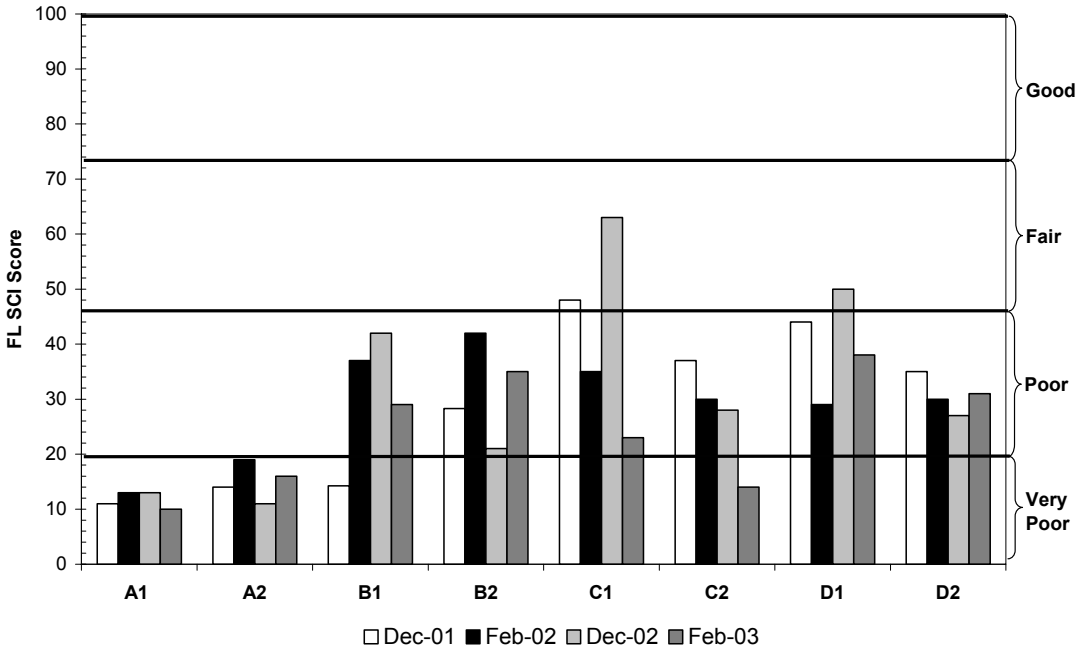

Figure 4-33. Florida Stream Condition Index (SCI) scores for upstream and downstream sites of each stream (A-D) over the entire study period.

Repeated measures ANOVA did not detect significant differences due to time, position, or site (Table 4-6 and Table 4-7). Overall, the Florida SCI scored 9.4% of the samples for the eight sites as Fair, 59.3% Poor, and 31.3% Very Poor . No sites were classified as Good. C1 and A1 had the highest and lowest average score, respectively, for every sampling period.

Results from the Georgia Environmental Protection Division (EPD) Biological Assessment were more favorable. 12.5% of scores for the four samples each at the eight sites were Very Good, 53.1% Good, 21.9% Fair, 6.3% Poor, and 3.1% Very Poor when compared to the Georgia DNR reference stream from the Southeastern Plains Ecoregion, Tifton Upland sub-ecoregion. A1 and A2 December 2001 samples scored Poor, but the February 2002, December 2002, and February 2003 samples of both scored Fair (Figure 4-34). The December 2001 sample for B1was the only one in the study to score Very Poor. The remainder of B1 samples scored Good. December 2001 and February 2002

samples for B2 scored Good, while the December 2002 sample fell into the Good condition, and the February 2003 was in the Very Good category. C1 had the greatest number of samples (December 2002, February 2002, and February 2003) in the Very Good condition, with one sample (December 2001) in the Good category. All samples for C2 scored in the Good category. December 2001 and February 2003 samples for D1 scored Good, while the February 2002 sample scored Fair, and the December 2002 sample scored Very Good. All samples for D2 scored Good (Figure 4-34).

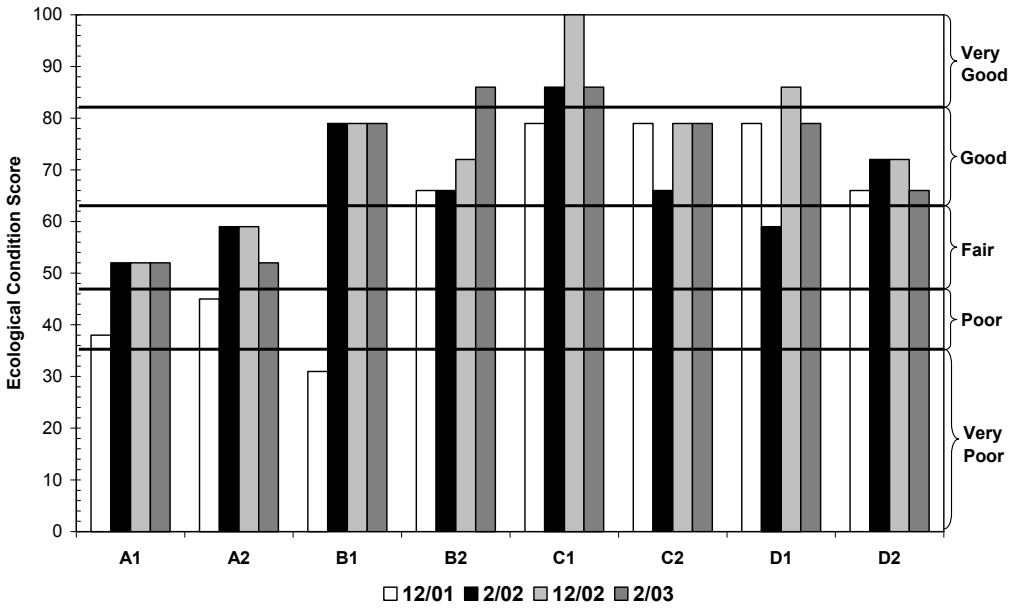

Figure 4-34. Georgia EPD Biological Assessment scores for upstream and downstream sites of each stream (A-D) over the entire study period.

For the Georgia EPD Biological Assessment applied to a 200 individual subsample, as called for in Georgia EPD standard operating procedures, site condition scores changed slightly from results presented above, with 11 samples increasing by 10%, 11 samples decreasing by 10%, and 10 sites remaining unchanged. Overall scores improved slightly with 15.6% listed as Very Good, 56.2% Good, 18.8% Fair, 6.2% Poor, and 3.1% Very Poor (Figure 4-35). For both the complete and 200 individual subsample, the

Georgia EPD Biological Assessment was similar to the Florida SCI in scoring C1 the highest and A1 the lowest across all sampling periods.

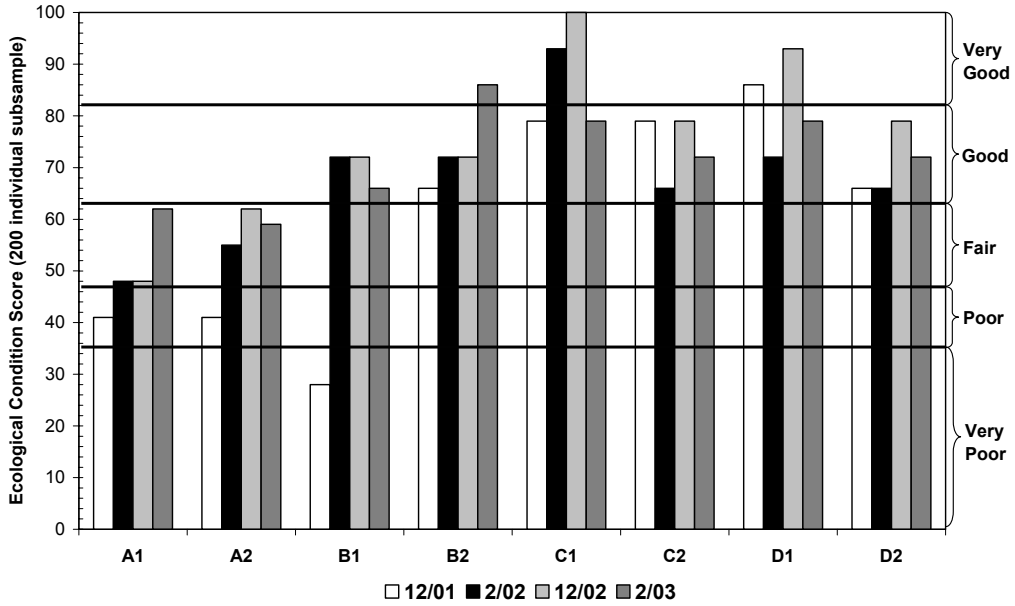

Figure 4-35. Georgia EPD Biological Assessment scores (based on a 200 individual subsample) for upstream and downstream sites of each stream (A-D) over the entire study period.

Repeated measures ANOVA detected weak but significant differences due to time (Table 4-6) for the Georgia EPD Biological Assessment scores calculated from both the complete sample (P=0.04) and 200 individual subsample (P=0.04), with varying significance between specific time periods. For the complete sample (P=0.08) (Figure 4-36) and 200 individual subsample (P=0.04) (Figure 4-37), December 2002 had significantly higher scores than February 2002, although results from the complete sample were marginally significant. No differences were detected due to position (Table 4.6). A significant difference (P=0.02) was found between sites for the Georgia EPD Biological Assessment scores calculated from the complete sample, but differences were not detected for the 200 individual subsample (Table 4.7). For the complete sample, stream A had significantly lower scores than stream C (P=0.01) (Figure 4-38).

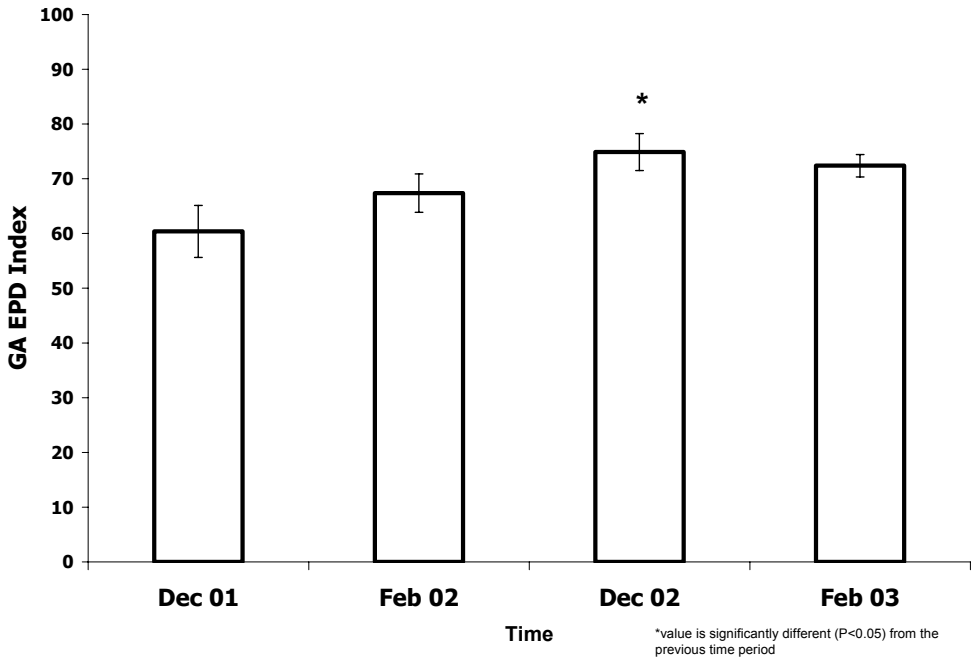

Figure 4-36. Means for GA EPD Index for all streams combined during each time period with standard error and repeated contrast results (alpha = 0.05) across time.

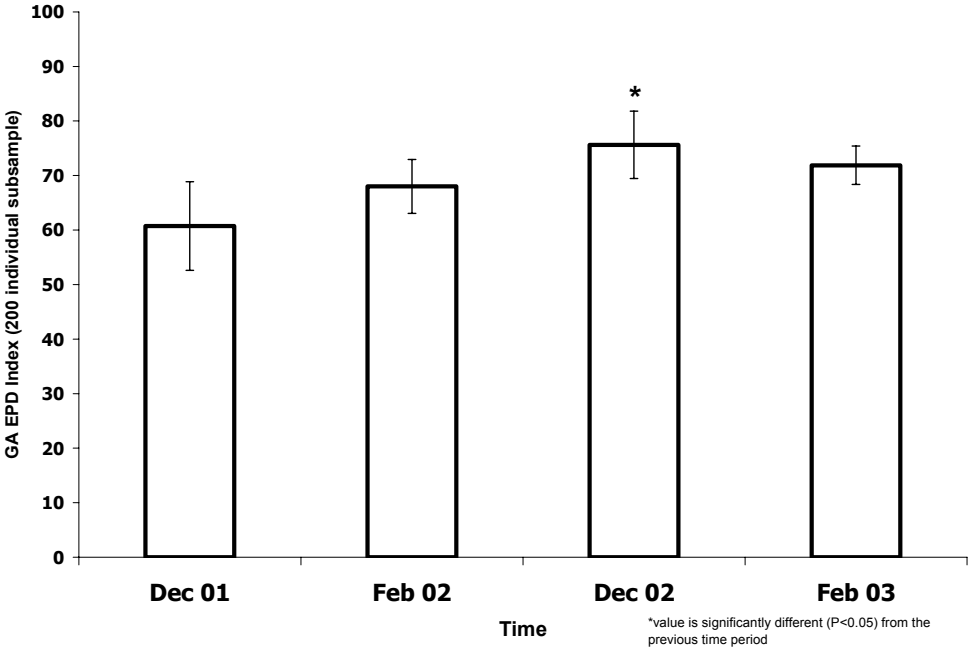

Figure 4-37. Means for GA EPD Index for all sites combined (200 individual subsample) within each time period with standard error and repeated contrast results (alpha = 0.05) across time.

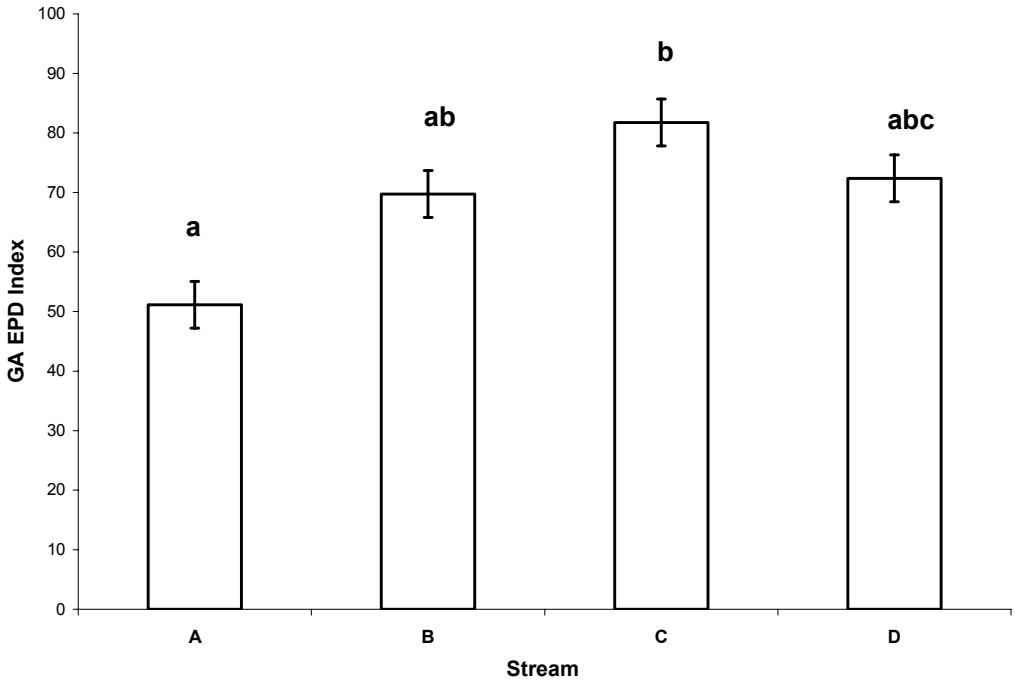

Figure 4-38. Means for GA EPD Index (individual streams for the entire study) and pairwise multiple comparison test results (Tukey's honestly significant difference (HSD) test, alpha = 0.05). Letters above bars indicate statistical groupings.

The Georgia Adopt-A-Stream water quality rating was excellent for 50% of samples, good for 12.5%, fair for 12.5%, and poor for 25%. Five of eight sites collected in December 2001 were rated as having poor water quality (Figure 4-39). Repeated measures ANOVA found significant differences due to time ($P<0.001$) and site ($P=0.005$), but not for position (Table 4-6 and Table 4-7). The December 2001 samples had significantly ($P=0.008$) lower scores than February 2002. February 2003 samples had significantly higher scores when compared to December 2002 ($P=0.03$) (Figure 4-40). Scores from stream A were significantly lower than scores for stream B ($P=0.02$), C ($P=0.006$), and D ($P=0.006$) (Figure 4-41).

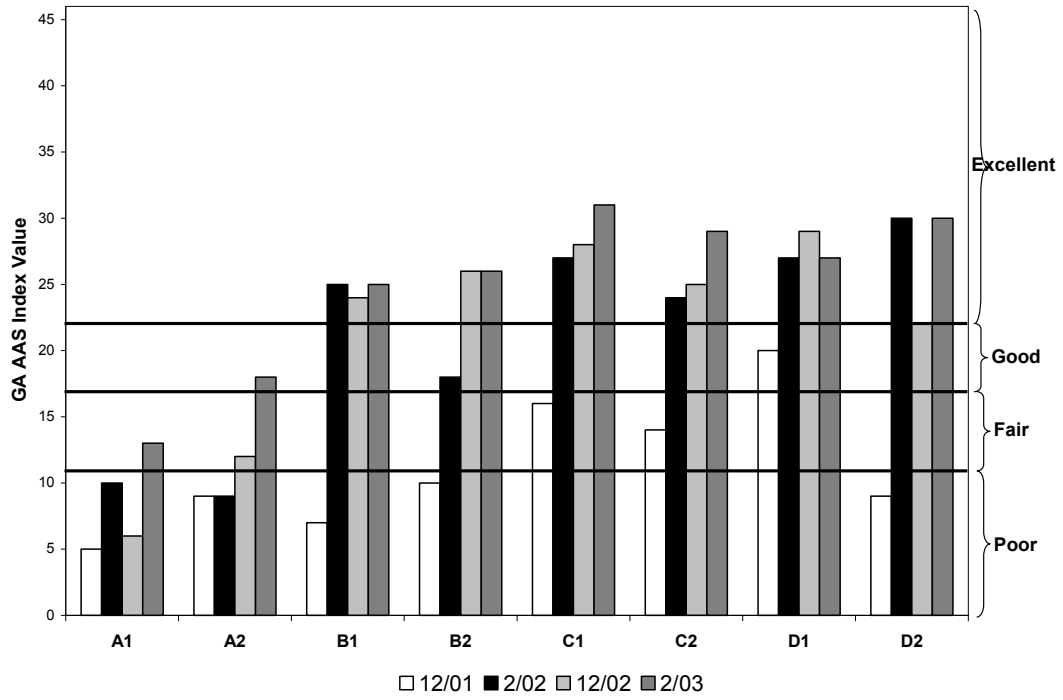

Figure 4-39. Georgia Adopt-A-Stream Index scores for upstream and downstream sites of each stream (A-D) over the entire study period.

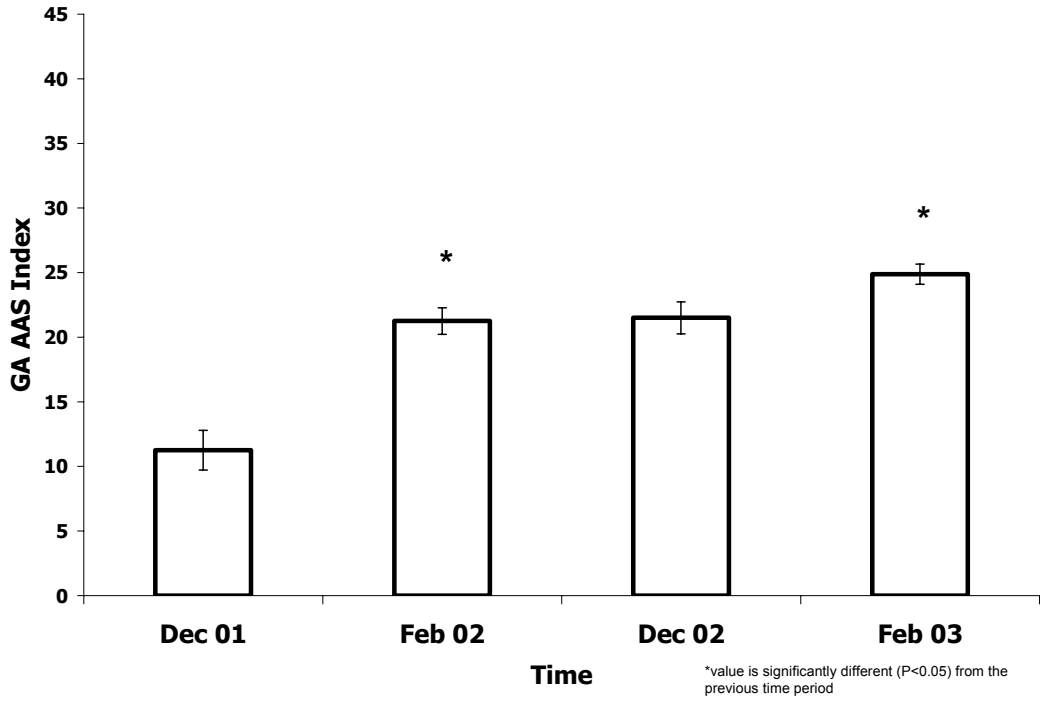

Figure 4-40. Means for GA AAS Index for all sites combined within each time period with standard error and repeated contrast results (alpha = 0.05) across time.

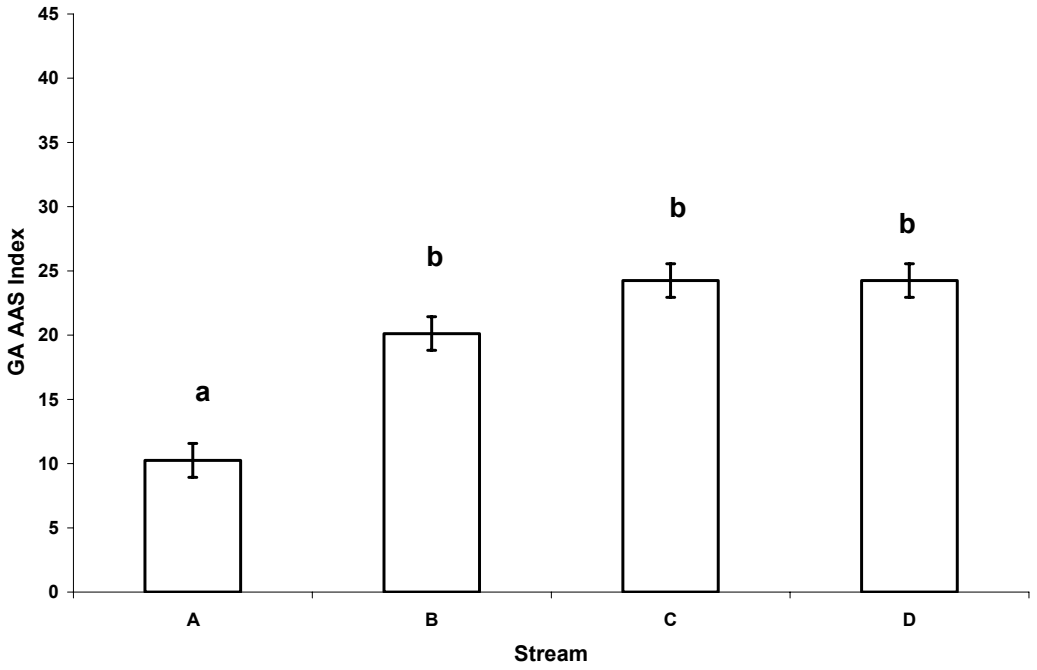

Figure 4-41. Means for GA AAS Index (individual streams for the entire study) and pairwise multiple comparison test results (Tukey's honestly significant difference (HSD) test, alpha = 0.05). Letters above bars indicate statistical groupings.

The Georgia Ecological Condition Worksheet was used to compare among sites for year to year, downstream vs. upstream, and stream to stream. Sites with lower percent comparability scores indicate greater similarity between sites, and higher scores indicate greater dissimilarity between sites. A score of 0 indicates 100% comparable assemblages, as shown in the comparison of B1 (study site) and C1 (reference) for the February 2002 samples (Table 4-8). Year to year site comparisons indicate generally higher percent comparability in year two (February 2002 to February 2003) of the study compared to year one (December 2002 to December 2003) (Table 4-9). Downstream vs. upstream percent comparability varied more in watersheds B and C across time than watersheds A and D (Table 4-10).

Table 4-8. Sample comparison of sites (B1 vs. C1 for February 2002)

ECOLOGICAL CONDITION WORKSHEET FOR B1 and C1 (2-02)

METRIC:	B1 Data:	C1 Data:	SOUTHEASTERN PLAINS (65) SCORE RANGES 5	3	1	0	B1 Score:	C1 Score:
Taxa Richness (Total # Taxa)	35	37	>30	30-16	<16		5	5
EPT Index	5	14	>6	6-4	<4		3	5
# Chironomidae Taxa	16	11	>8	8-5	<5		5	5
% Contribution of Dominant Taxon	28	23	<23	23-61	>61		3	3
% Diptera	91	68	-----	<51	>50		1	1
Florida Index	13	16	>15	15-8	<8		5	5
% Filterers	8	2	>11	11-6	<6		3	1
Total Habitat Score:			>89%	89-75%	74-60%	<59%		
Total Points Earned							25	25
% of Difference							**100**	
100 - % of Difference							0	

Table 4-9. Percent comparability scores for year to year comparison of sites.

	Year to Year							
	A1	**A2**	**B1**	**B2**	**C1**	**C2**	**D1**	**D2**
12-01 vs 12-02	15	31	156	11	26	9	8	11
2-02 vs 2-03	0	12	0	32	0	21	10	10

Table 4-10. Percent comparability scores for downstream vs. upstream comparison of sites.

	Downstream vs Upstream			
	12-01	**2-02**	**12-02**	**2-03**
A1 vs A2	15	12	12	0
B1 vs B2	53	21	10	8
C1 vs C2	0	32	26	9
D1 vs D2	21	19	19	21

Table 4-11 displays three watershed to watershed comparisons in grey or white boxes for each watershed for one sampling date. The lowest percent comparability score for each sampling date is highlighted. Across sampling dates, there is no consistently lower percent comparability score for one site over another. However, when the data are viewed for year one (12-01 to 2-02) and year two (12-02 to 2-03), the format shows no consistent results. Year two scores showed lowest percent comparability between A1 and D1, B1 and D1, C1 and D1 and D1 and B1.

Table 4-11. Percent comparability scores for stream to stream comparison of sites.

	Stream to Stream			
	12-01	2-02	12-02	2-03
A1 vs B1	**22**	35	35	35
A1 vs C1	52	40	48	40
A1 vs D1	52	**12**	**29**	**35**
B1 vs A1	**31**	67	53	53
B1 vs C1	65	**0**	21	8
B1 vs D1	65	47	**10**	**0**
C1 vs A1	17	67	93	67
C1 vs B1	188	**0**	26	9
C1 vs D1	8	47	**7**	**9**
D1 vs A1	17	**13**	67	53
D1 vs B1	188	32	**9**	**0**
D1 vs C1	**9**	32	14	8

Multivariate Analysis

The first two PCA axes explained 57% of variability in macroinvertebrate data (Figure 4-42). Points in the plot represent individual sites for four time periods (e.g., B11= site B1, at December 2001). The Georgia EPD index was highly correlated with Axis 1 for both the complete sample (r^2=0.91) and 200 individual subsample (r^2=0.89). EPT taxa, Georgia AAS index, and total taxa were also strongly correlated with Axis 1 (r^2=0.71, 0.71, and 0.62, respectively). Percent filter feeders was correlated (r^2=0.58) with Axis 2. Abundance and percent dominant taxa were negatively correlated (r^2=0.54 and 0.41, respectively) with the same axis. Samples from watershed A tended to separate along Axis 1.

Regression Analysis

Five models were run with: 1) all samples pooled (n=31), 2) December 2001 samples only (n=7), 3) February 2002 samples only (n=8), 4) December 2002 samples

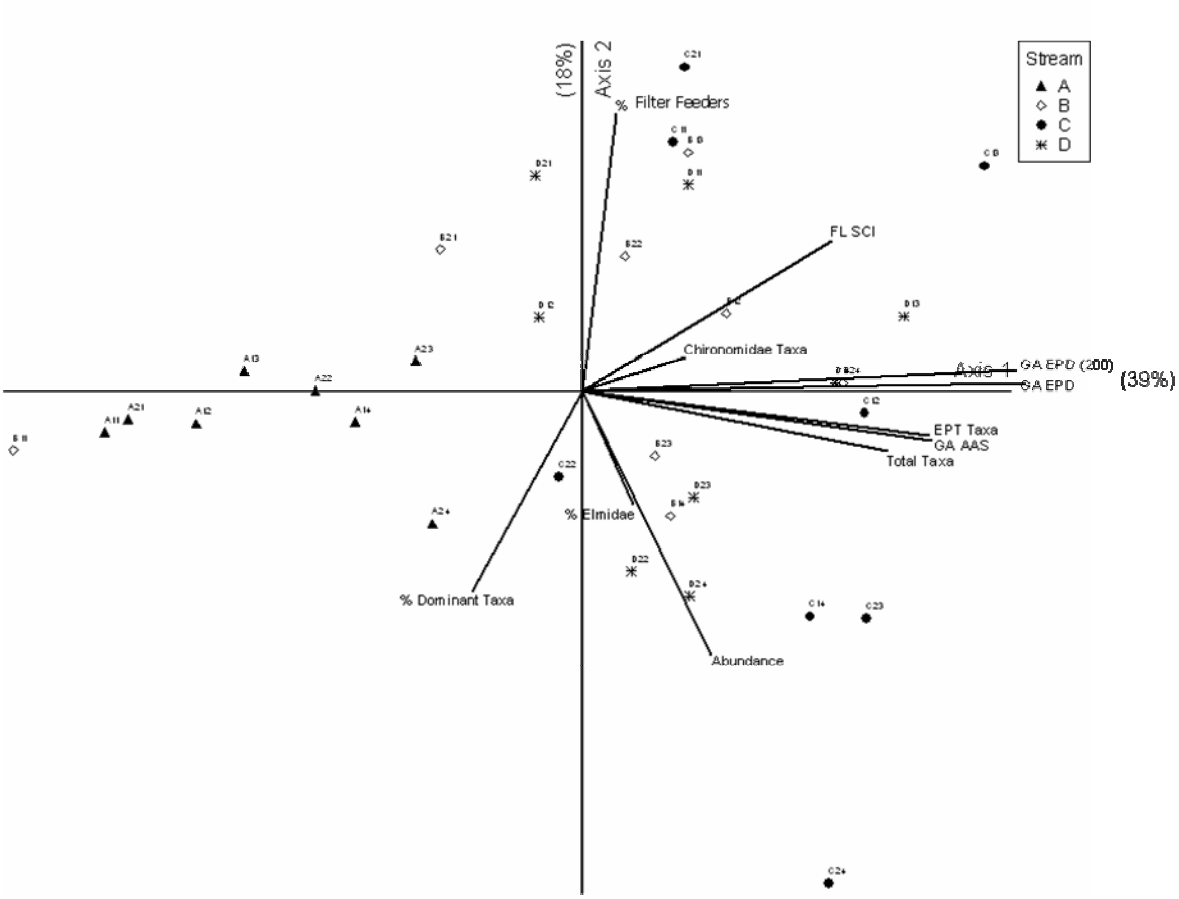

Figure 4-42. Principal components ordination for macroinvertebrate metrics and indices.

only (n=8), and 5) February 2003 samples only (n=8). These five models were run with water chemistry and hydrology data from one, three, and six months prior to macroinvertebrate sampling (Table 4-12). The models run with predictor variable data from three months prior to macroinvertebrate sampling cumulatively explained 51.8% of variation, whereas one and six month prior data explained 50.5% and 50.9%, respectively (Table 4-12). As a result, the remaining discussion will be restricted to the set of models run with predictor data from three months prior to macroinvertebrate sampling.

The first predictive model using all samples pooled (n=31) explained 82.7% of variation (Georgia AAS) to 15.8% (percent filter feeders). The remaining models with each time period as a separate model (n=8) generally had higher adjusted R^2 than the

Table 4-12. Adjusted R^2 values (%) for five *models: 1) all samples pooled (n=31), 2) Dec 01 samples only (n=7), 3) Feb 02 samples only (n=8), 4) Dec 02 samples only (n=8), 5) Feb 03 samples only (n=8). (--) = no variables entered model at alpha=0.05.

A. Predictor variables in models summarized for one month prior to macroinvertebrate sampling.

	*Model (%)					
	1	2	3	4	5	
Response Variable						Average
GA EPD	45.85	89.24	80.66	98.1	79.86	78.74
GA AAS	79.2	67.8	85.61	68.29	80.16	76.21
EPT taxa	67.69	--	90.81	79.35	93.02	66.17
Abundance	55.66	--	--	55.31	81.92	38.58
Total taxa	55.47	85.69	88.19	47.87	--	55.444
% Elmidae	66.29	51.28	72.96	94.53	86.73	74.358
% Dominant taxon	12.11	49.18	--	--	--	12.26
% Filter feeders	9.76	--	--	--	--	1.95
Average	49.0	42.9	52.3	55.4	52.7	**50.46**

B. Predictor variables in models summarized for three months prior to macroinvertebrate sampling.

	*Model (%)					
	1	2	3	4	5	
Response Variable						Average
GA EPD	54.46	83.16	80.66	60.63	90.34	73.85
GA AAS	82.65	70.31	85.61	87.56	80.16	81.26
EPT taxa	72.83	63.92	90.81	97.19	93.97	83.74
Abundance	74.53	--	--	53.26	71.26	39.81
Total taxa	61.39	93.89	88.19	--	--	48.69
% Elmidae	43.78	54.19	72.96	91.18	81.42	68.71
% Dominant taxon	16.77	58.83	--	--	--	15.12
% Filter feeders	15.79	--	--	--	--	3.16
	52.78	53.04	52.28	48.73	52.14	**51.79**

C. Predictor variables in models summarized for six months prior to macroinvertebrate sampling.

	*Model (%)					
	1	2	3	4	5	
Response Variable						Average
GA EPD	52.34	86.08	61.75	82.19	94.98	75.47
GA AAS	63.46	68.94	86.23	75.46	63.24	71.47
EPT taxa	59.14	70.92	51.42	75.53	91.18	69.64
Abundance	68.17	48.72	--	74.36	68.4	51.93
Total taxa	51.47	88.55	76.9	48.94	--	53.17
% Elmidae	49.85	50.98	85.04	92.09	84.94	72.58
% Dominant taxon	--	51.68	--	--	--	10.34
% Filter feeders	12.89	--	--	--	--	2.58
	44.67	58.23	45.17	56.07	50.34	**50.90**

*Models: 1) all samples pooled (n=31), 2) Dec 01 samples only (n=7),
3) Feb 02 samples only (n=8), 4) Dec 02 samples only (n=8), 5) Feb 03 samples only (n=8).
(--) = no variables entered model at alpha 0.05

pooled models, especially for regressions with Georgia EPD, Georgia AAS, and EPT taxa. Percent Elmidae, dominant taxa and filter feeders had the lowest number of

significant regressions (Table 4-12-B). EPT taxa had the highest cumulative adjusted R^2 of the response variables (83.7%) with 72.8% (Model 1-Pooled), 63.9% (Model 2-December 2001), 90.8% (Model 3-February 2002), 97.2% (Model 4-December 2002), and 94.0% (Model 5-February 2003) (Table 4-12-B).

Predictor variables selected for Models 1 (samples pooled), 2 (December 2001), 3 (February 2002), 4 (December 2002), and 5 (February 2003) varied. This is likely due to ANOVA results that indicated differences between time periods for abundance, EPT taxa, total taxa, Georgia EPD index, Georgia AAS index, and percent Elmidae. As a result, one time period, February 2003, was selected for discussion of predictor variable results because abundance, EPT taxa, total taxa, and Georgia AAS index had the highest average values for this time period. Also, annual rainfall data from 1967-2003 for the study site indicated that 2003 was closest to normal climatic conditions than 2001 or 2002 (Figure 4-43).

Variation in abundance was explained by average daily flow (P=0.005) for the February 2003 model (adjusted R^2=71.3%). The model for EPT taxa resulted in average daily flow (P<0.001) and inorganic phosphorus (P=0.03) explaining 94% (adjusted R^2) of variation (Table 4-13). Variation in Georgia EPD index was explained (adjusted R^2=90%) by specific conductance (P=0.002) and minimum daily flow (P=0.04), while specific conductance (P=0.002) alone explained variation (adjusted R^2=80%) in Georgia AAS index values (Table 4-14). Inorganic phosphorus (P=0.006) and dissolved oxygen (P=0.04) were significant predictors (adjusted R^2=81%) for percent Elmidae. Total taxa, percent dominant taxon, and percent filtering collectors did not result in significant regressions for February 2003 (Table 4-15).

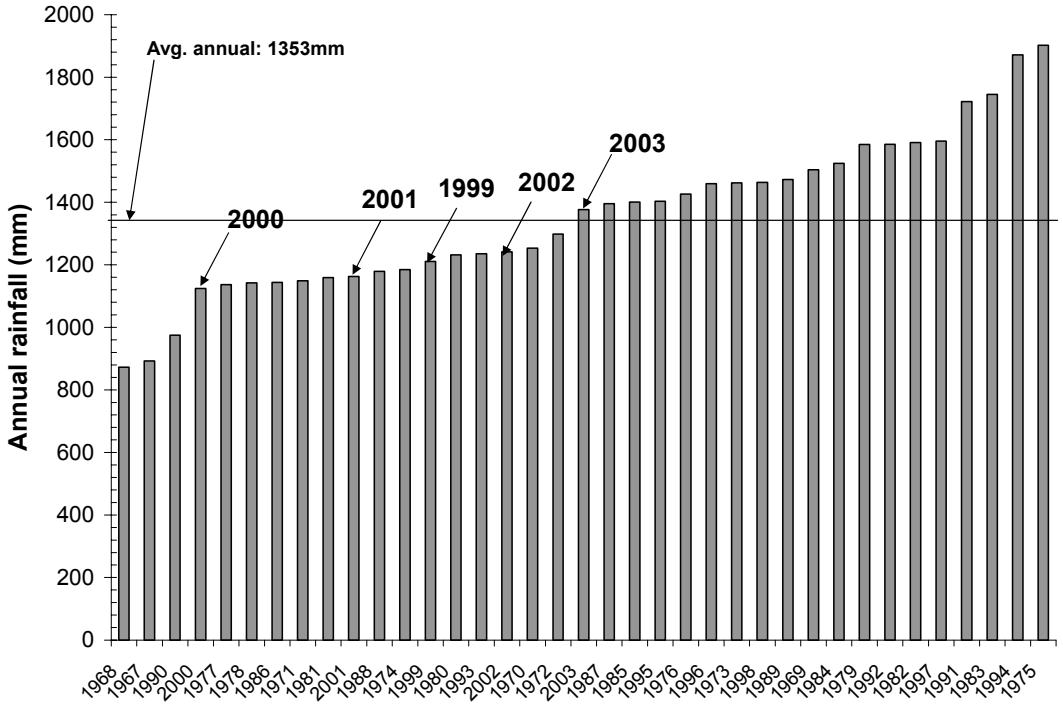

Figure 4-43. Total annual rainfall from 1967-2003 at the Bainbridge, GA station (90586) at International Paper (SRCC 2004) arranged from lowest to highest annual values.

As a result of hydrology being a significant predictor for abundance, EPT taxa, and Georgia EPD index, for the February 2003 time period, the relationship between hydrology for the entire study period was examined. Plots of abundance (Figure 4-44), EPT taxa (Figure 4-45), and Georgia EPD index (Figure 4-46) with average daily flow for all sites and time periods (indicated with different symbols), showed an increasing positive relationship, particularly for abundance.

For abundance (Figure 4-47), when a linear regression fit was applied to each time period, the December 2001 and February 2002 time period had no relationship with average daily flow, but the December 2002 time period was marginally significant (P=0.09) with an r^2 of 40% and finally the February 2003 time period resulted in a highly significant relationship (P=0.005) with an r^2 of 75%.

Table 4-13. Stepwise regression models of the relationship between EPT taxa and abundance (response) with water chemistry and hydrology parameters (predictors). Alpha to Enter/Remove: 0.05.

Response: EPT Taxa

Step	Predictor(s)	P value	R^2 (%)	R^2 (adj) (%)	PRESS
Pooled					
1	DO	<0.001	59	57	248
2	DO	<0.001	74	72	163
	Turbidity	<0.001			
Dec 01					
1	Temperature	0.01	69	63	109
Feb 02					
1	Turbidity	0.01	63	57	91
2	Turbidity	<0.001	93	90	36
	NH4	0.005			
Dec 02					
1	DO	0.002	81	78	32
2	DO	0.002	93	90	17
	Turbidity	0.03			
3	DO	0.001	98	97	9
	Turbidity	0.005			
	Inorg P	0.02			
Feb 03					
1	Avg df	0.001	87	85	20
2	Avg df	<0.001	95	93	14
	Inorg P	0.030			

Response: Abundance

Step	Predictor(s)	P value	R^2 (%)	R^2 (adj) (%)	PRESS
Pooled					
1	Maxdf	<0.001	50	49	3.1
2	Maxdf	<0.001	69	67	2.0
	Temp	<0.001			
3	Maxdf	<0.001	77	74	1.7
	Temp	<0.001			
	Min df	0.007			
Dec 01					
1	No variables entered or removed				
Feb 02					
1	No variables entered or removed				
Dec 02					
1	Inorg N	0.02	59	53	0.3
Feb 03					
1	Avg df	0.005	75	71	0.1

EPT taxa (Figure 4-48) was slightly different in that the December 2001 and time period had no relationship with average daily flow, but February 2002 and December 2002 had marginally significant regressions (P=0.17 and r^2= 28%, P=0.16 and r^2= 30%, respectively). Finally, the February 2003 time period resulted in a highly significant regression (P=0.001) for EPT and average daily flow with an r^2 of 87%.

For Georgia EPD index (Figure 4-49), when a linear regression fit was applied to each time period, the December 2001 time period had no relationship with average daily flow. February 2002 and December 2002 displayed similar trends with EPT index in that both time periods had marginally significant regressions (P=0.15 and r^2= 30%, P=0.09 and r^2= 40%, respectively). Finally, the February 2003 time period resulted in a highly

significant regression (P=0.005) for Georgia EPD index and average daily flow with an r^2 of 75%.

For each metric, overall the increasing positive relationship improves with each time period with a marked difference in relationship between the December 2001 time period and February 2003.

Table 4-14. Stepwise regression models of the relationship between Georgia EPD index and Georgia AAS index (response) with water chemistry and hydrology parameters (predictors). Alpha to Enter/Remove: 0.05.

Response: GA EPD Index

Step	Predictor(s)	P value	R^2 (%)	R^2 (adj) (%)	PRESS
Pooled					
1	DO	<0.001	35	33	5407
2	DO	<0.001	57	54	4101
	SC	0.001			
Dec 01					
1	Inorg N	0.03	64	56	1460
2	Inorg N	0.006	88	83	2250
	Inorg P	0.04			
Feb 02					
1	SC	0.02	59	52	622
2	SC	0.003	86	80	347
	NH4	0.02			
Dec 02					
1	SC	0.01	66	60	938
Feb 03					
1	SC	0.002	82	79	319
2	SC	0.002	93	90	254
	Mindf	0.04			

Response: GA AAS Index

Step	Predictor(s)	P value	R^2 (%)	R^2 (adj) (%)	PRESS
Pooled					
1	DO	<0.001	60	58	978
2	DO	<0.001	81	80	500
	Turbidity	<0.001			
3	DO	<0.001	84	82	463
	Turbidity	<0.001			
	InorgN	0.03			
Dec 01					
1	Inorg N	0.01	75	70	46
Feb 02					
1	pH	0.001	87	85	83
Dec 02					
1	Turbidity	0.003	78	75	193
2	Turbidity	0.001	91	87	118
	Inorg P	0.04			
Feb 03					
1	SC	0.002	83	80	95

Table 4-15. Stepwise regression models of the relationship between percent dominant taxa, percent filter feeders, total taxa, and percent Elmidae (response) and water chemistry and hydrology parameters (predictors). Alpha to Enter/Remove: 0.05.

Response: % Dominant taxon

Step	Predictor(s)	P value	R^2 (%)	R^2 (adj) (%)	PRESS
Pooled					
1	SC	0.02	16	13	0.5
Dec 01					
1	pH	0.04	58	50	0.2
Feb 02					
1	No variables entered or removed				
Dec 02					
1	No variables entered or removed				
Feb 03					
1	No variables entered or removed				

Response: % Filter feeders

Step	Predictor(s)	P value	R^2 (%)	R^2 (adj) (%)	PRESS
Pooled					
1	SC	0.01	18	15	0.3
Dec 01					
1	No variables entered or removed				
Feb 02					
1	No variables entered or removed				
Dec 02					
1	No variables entered or removed				
Feb 03					
1	No variables entered or removed				

Response: Total taxa

Step	Predictor(s)	P value	R^2 (%)	R^2 (adj) (%)	PRESS
Pooled					
1	DO	<0.001	51	49	1253
2	DO	<0.001	63	61	989
	Maxdf	0.004			
Dec 01					
1	Inorg N	0.02	67	61	138
2	Inorg N	0.001	95	93	25
	Turbidity	0.006			
Feb 02					
1	SC	0.01	68	63	179
2	SC	0.001	91	88	93
	NH4	0.01			
Dec 02					
1	No variables entered or removed				
Feb 03					
1	No variables entered or removed				

Response: % Elmidae

Step	Predictor(s)	P value	R^2 (%)	R^2 (adj) (%)	PRESS
Pooled					
1	Inorg P	0.002	28	26	0.06
2	Inorg P	<0.001	47	43	0.04
	Max df	0.004			
Dec 01					
1	Inorg N	0.03	61	54	0.00
Feb 02					
1	Inorg P	0.004	76	72	0.00
Dec 02					
1	Inorg P	0.001	85.00	82	0.01
2	Inorg P	0.001	93	91	0.01
	DO	0.04			
Feb 03					
1	Inorg P	0.01	68	62	0.01
2	Inorg P	0.006	86	81	0.01
	DO	0.04			

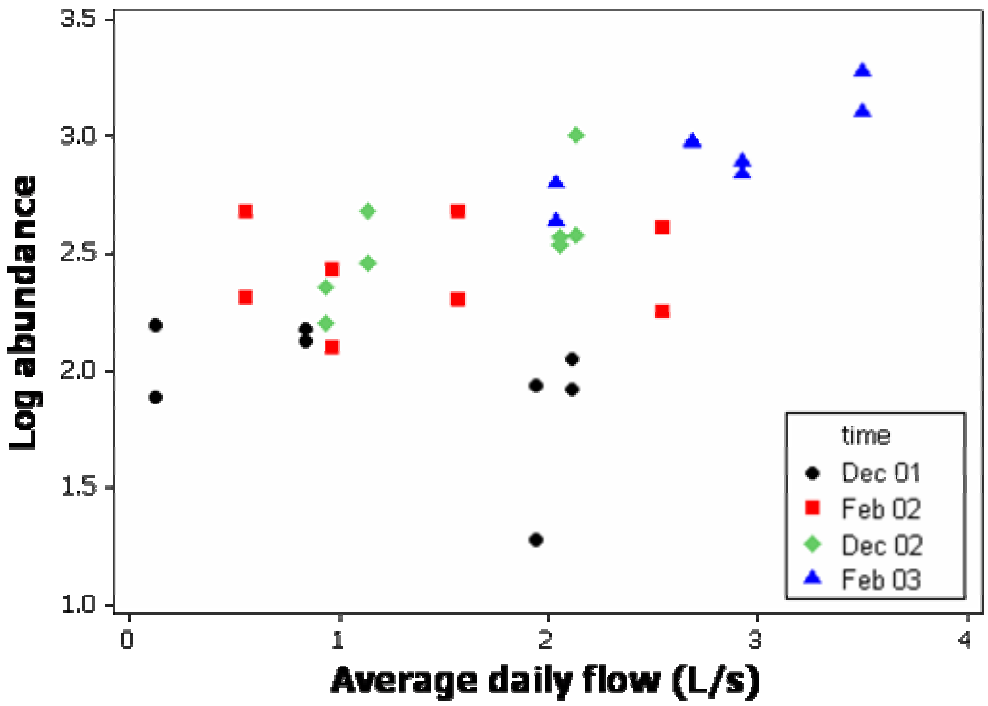

Figure 4-44. Abundance vs. average daily flow for all sites and time periods (1-December 2001, 2-February 2002, 3-December 2002, 4-February 2003).

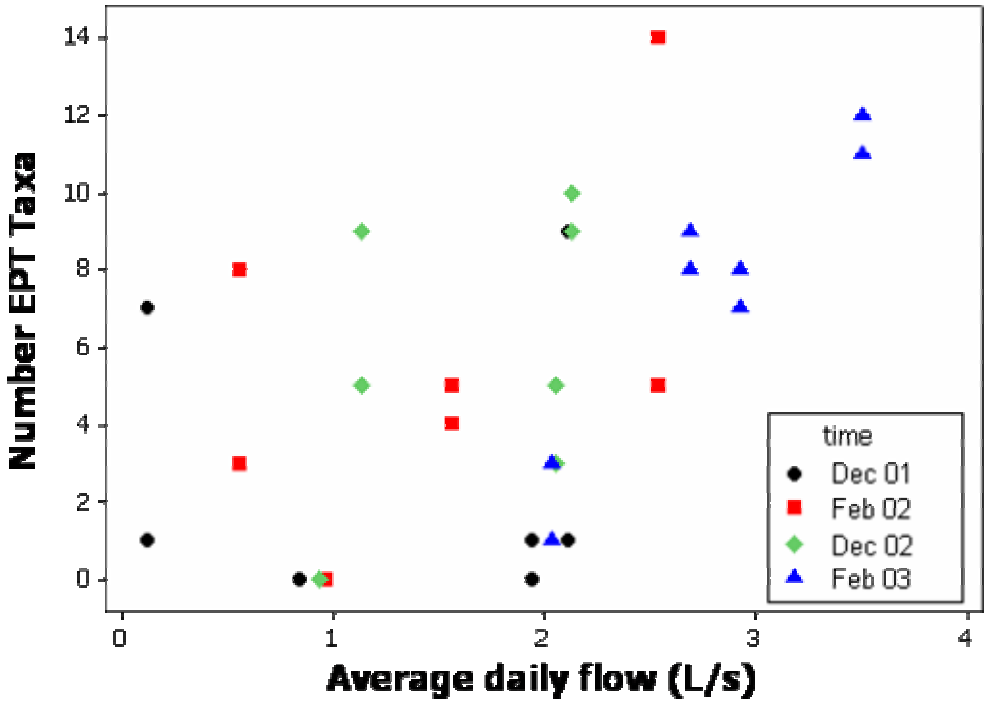

Figure 4-45. EPT taxa vs. average daily flow for all sites and time periods (1-December 2001, 2-February 2002, 3-December 2002, 4-February 2003).

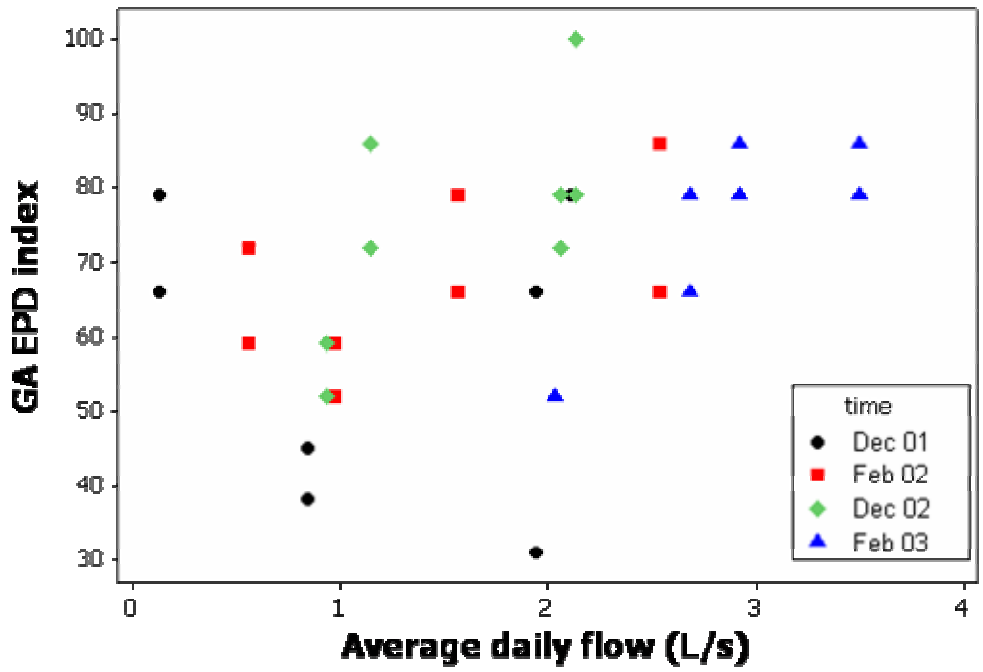

Figure 4-46. GA EPD Index vs. average daily flow for all sites and time periods (1-December 2001, 2-February 2002, 3-December 2002, 4-February 2003).

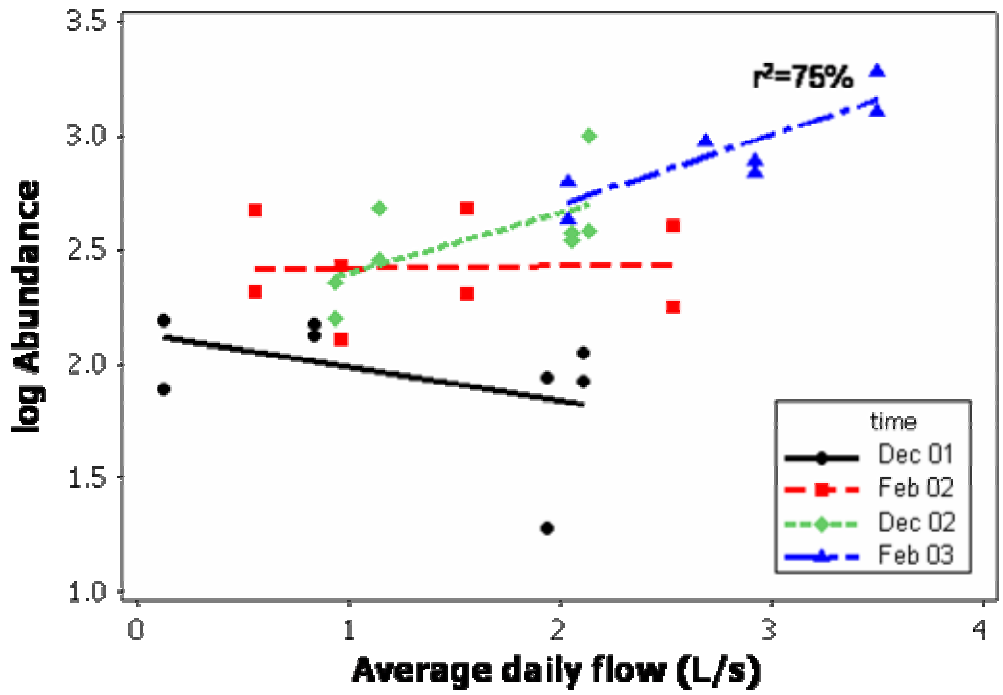

Figure 4-47. Abundance vs. average daily flow for all sites and time periods with linear regression fit for each time period (1-December 2001, 2-February 2002, 3-December 2002, 4-February 2003).

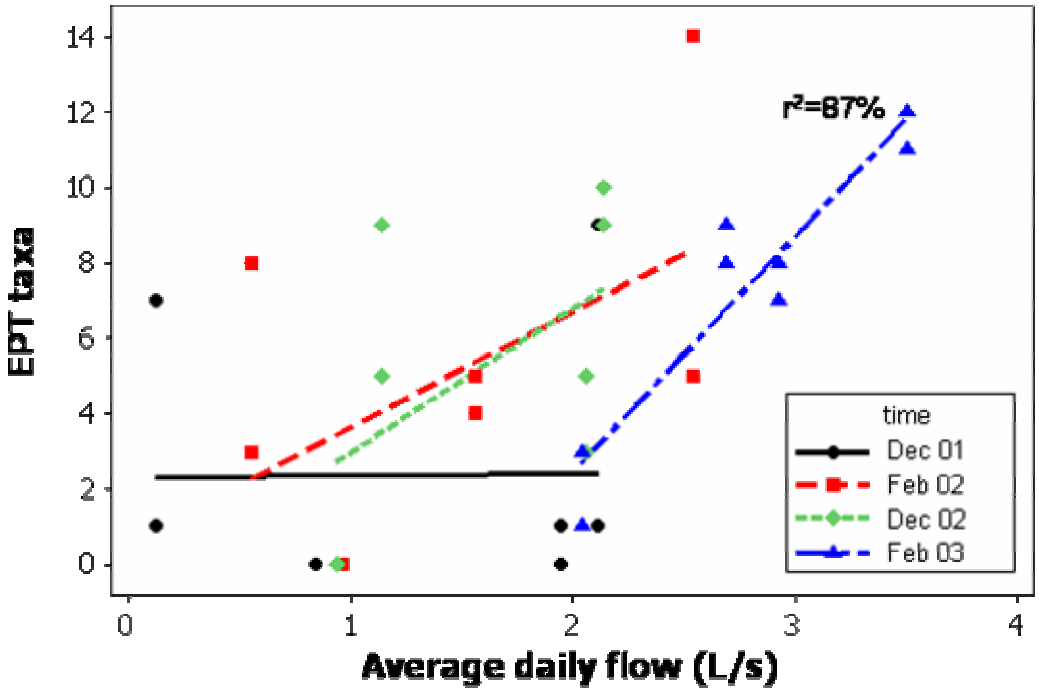

Figure 4-48. EPT taxa vs. average daily flow for all sites and time periods with linear regression fit for each time period (1-December 2001, 2-February 2002, 3-December 2002, 4-February 2003).

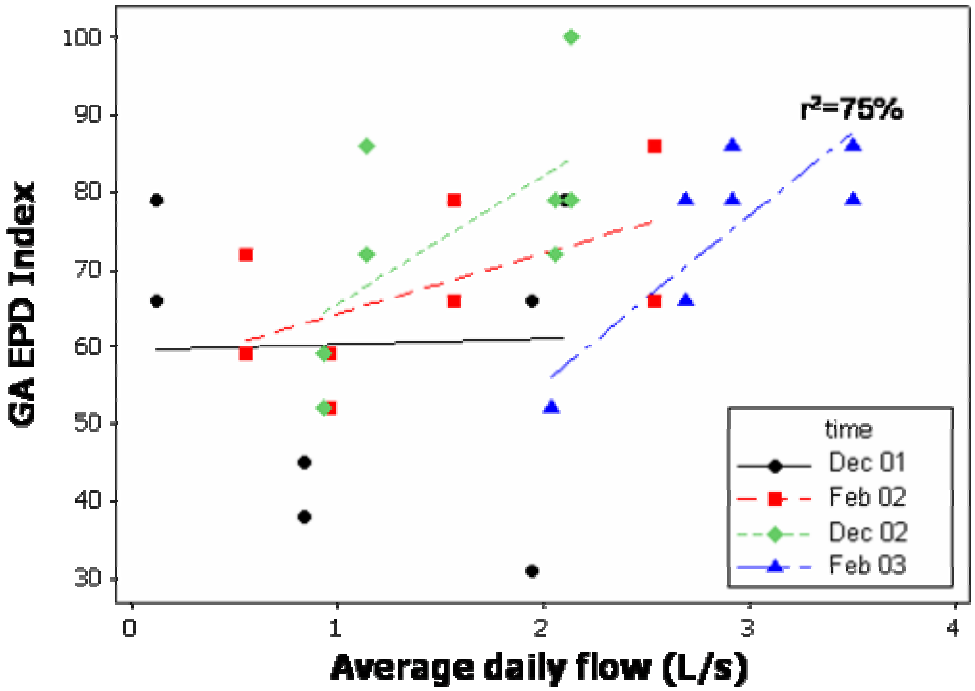

Figure 4-49. GA EPD Index vs. average daily flow for all sites and time periods with linear regression fit for each time period (1-December 2001, 2-February 2002, 3-December 2002, 4-February 2003).

CHAPTER 5
DISCUSSION AND CONCLUSIONS

Discussion

Macroinvertebrate Assemblages – All Streams

Although Diptera comprised >60% of the total macroinvertebrate assemblages in the sand dominated study streams, they were still quite diverse with EPT taxa and other taxa sensitive to disturbance being well represented. Sand is generally considered a poor substrate for macroinvertebrates because of its instability and lack of interstitial oxygen, but some macroinvertebrates are specialists of this habitat. For example, the ephemeropteran, *Hexagenia limbata*, found in this study creates a U-shaped burrow in fine sediments, then beats its gills to create a current through its burrow (Allan 1995). A study of three steephead streams adjacent to the Dry Creek watershed in the International Paper Southlands Forest found that macroinvertebrate assemblages in all streams had high diversity with some taxa typically found in the southern Appalachians (Entrekin et al. 1999). The total number of EPT taxa (genus level) from all streams of the current study (26) was greater than reported for other streams in southwestern Georgia, 20 (Muenz 2004), 11 (Gregory 1996), and 15 (Davis 2000). A study of low gradient, higher order streams in Georgia had 31 taxa (Benke et al. 1984), a low gradient, low order stream in southeastern Virginia had 34 (Wright and Smock 2001), and in high gradient, low order streams of North Carolina, 29 EPT taxa were present (Stone and Wallace 1998).

Repeated measures ANOVA results for abundance, total taxa, EPT taxa, Georgia AAS index, Georgia EPD index, and percent Elmidae indicated that there were differences in the macroinvertebrate assemblages due to sampling period. More specifically, ANOVA results for Georgia EPD index and percent Elmidae were similar with the December 2002 sampling period having significantly greater average values than December 2001 and February 2002. The reason for the similarity in higher average values for December 2002 is not apparent and may be coincidental because the EPD index is composed of seven metrics (taxa richness, EPT index, number of Chironomidae taxa, percent contribution of dominant taxon, percent Diptera, Florida Index, and percent filterers), and Elmidae would only influence one of these metrics, taxa richness.

Abundance, total taxa, EPT taxa, and GA AAS index consistently indicated that the December 2001 sampling period had significantly lower values than February 2003 and that February 2002 did not differ from December 2002. Macroinvertebrate assemblages respond to temporal variability, whether seasonal (Gibbins et al. 2001, Hutchens et al. 1998) or interannual (Hutchens et al. 1998). Seasonal variation is expected in macroinvertebrate communities due to life history patterns, especially for taxa that complete their life cycle within one year. The aforementioned ANOVA results for abundance, total taxa, EPT taxa, and GA AAS index indicate that seasonal variation was not the controlling factor for differences because the two seasons collected in 2002 did not differ, while the same two seasons from different years, 2001 and 2003, were significantly different.

Interannual variation in macroinvertebrate communities can be influenced by drought (Feminella 1996), although Hutchens et al. (1998) reported that no consistent

drought-induced pattern in macroinvertebrate assemblages was apparent. This apparent contradiction of results may be due to the substrate sampled. Hutchens et al. (1998) suggested that drought-induced effects were possibly not detected because mixed substrates, in this case red maple litter bags, were less sensitive to disturbance than other habitats, such as bedrock outcrops. In the study of macroinvertebrate assemblages in small streams of Alabama along a gradient of flow permanence, riffles were the substrate sampled and total taxa was found to significantly differ between years; 1994, a wet year preceded by a dry year and 1995 a normal year preceded by a wet year (Feminella 1996). EPT taxa were not significantly different between years, but showed a strong positive relationship with stream permanence (Feminella 1996).

In southwestern Georgia, drought conditions occurred during 1998-2002 and resulted in an accumulated rainfall deficit of 711-1270 mm in some areas (Pam Knox, Assistant Georgia State Climatologist, oral communication as cited in Warner and Norton, 2003). The depressed rainfall totals resulted in low average daily flows (all streams combined) that were most severe in December 2001 (1.25 L/s), then steadily increased through February 2002 (1.40 L/s) and December 2002 (1.57 L/s), to February 2003 (2.78 L/s). The biota was affected by these conditions as evidenced by the generally increasing positive relationship through time between average daily flow and abundance, EPT taxa, and Georgia EPD index.

Poff and Ward (1989) analyzed long-term discharge records for 78 streams across the US to develop quantitative characterizations of streamflow variability and predictability. Acknowledging that there is generally a lack of empirical data on stream organisms relative to long-term flow data, this characterization was used to suggest

relationships between hydrology and ecological processes. For streams that have some degree of intermittency, they hypothesized that the biological community would be trophically simple with high predictability of successional patterns upon resumption of flow.

The results of this study support Poff and Ward's hypothesis because samples collected during December 2001 (higher degree of intermittency) had lower abundance, total taxa, and EPT taxa than subsequent periods (lower degree of intermittency). Also, as streams recovered from drought conditions and flow increased, the predictive power of the relationship between flow and metrics improved.

Macroinvertebrate Assemblages – Within Streams

ANOVA was also used to detect effects due to sampling position (i.e. upstream vs. downstream), but none were detected for the metrics and indices tested. The River Continuum Concept (Vannote et al. 1980) suggests that faunal changes will occur along the length of a river due to changes in the amount and type of production along that gradient. In this study, upstream and downstream sampling positions ranged from approximately 50-250 meters apart, which is not a large enough distance to incur substantial differences in allochthonous inputs or differences in primary production that would result in faunal changes. Stream order did not change between sites sampled within each stream which also lowered the potential for instream variation.

Macroinvertebrate Assemblages – Among Streams

Abundance, total taxa, EPT taxa, Georgia EPD index, Georgia AAS index, and percent Elmidae displayed significant differences among streams; however the power of these analyses was very low due to a sample number of two, which could have underestimated the among-stream variation. Comparisons between streams for EPT taxa,

Georgia EPD index, Georgia AAS index, and percent Elmidae demonstrated that stream A had significantly lower values than stream C. In general, water chemistry was different in stream A versus C, with mean specific conductance, pH, dissolved oxygen, turbidity, and inorganic nitrogen being lower in A.

Inorganic phosphorus and dissolved oxygen were significant predictors for percent Elmidae. Percent Elmidae ANOVA results indicated that A had significantly lower percent Elmidae than C, but that C had significantly lower percent Elmidae than D. Average inorganic phosphorus concentrations (for the entire study period, upstream and downstream positions) in D (0.044 ug/L) were an order of magnitude greater than concentrations in A (0.0045 ug/L), B (0.0035 ug/L), and C (0.0053ug/L). The source of the elevated phosphorus in D is unknown, but possibly reflects dissolution of limestone, because D is in a relatively undisturbed condition with no fertilization or other treatments. Average dissolved oxygen (for the entire study period, upstream and downstream positions) was highest in C (6.45 mg/L), followed by D (5.99 mg/L), B (4.90 mg/L), A (4.05 mg/L). Inorganic phosphorus is often the limiting nutrient for growth of algae and macrophytes (Allan 1995). Since 95% of individuals collected were Elmidae of the genus *Stenelmis*, which is a scraper (Merritt and Cummins 1996) that feeds on decayed plant materials and algae (Epler 1996), the significantly higher percent Elmidae in D versus other streams is plausible. Mean periphyton chlorophyll *a* concentrations collected during the summer of 2003 were not significantly different among sites, but these samples only quantified the periphyton community of the sediment, not of stable substrates such as exposed rock. Since *Stenelmis* is also a clinger (Merritt and Cummins 1996), they would more likely be associated with more stable substrate.

EPT taxa, Georgia EPD index, Georgia AAS index had significantly lower values in A than C. Significant predictors in regressions for EPT taxa were average daily flow and inorganic phosphorus, specific conductance and minimum daily flow for Georgia EPD index, and specific conductance for the Georgia AAS index. Average daily flow for three months prior to the February 2003 sampling period was 2.04 L/s in A and 3.50 L/s in C. For February 2003, Pearson correlations indicated that specific conductance was highly correlated (0.846, P=0.008) with average daily flow. C receives flow primarily from groundwater seepage, while A receives groundwater seepage and discharge from a small headwater pond. Specific conductance can be used as an indicator of baseflow (i.e. groundwater) input to a stream (Kuwabara 1992, Pilgrim et al. 1979, as cited in Black 1996) because groundwater has elevated conductivity relative to the water column. Streams with very low ionic concentrations generally have a flora and fauna characterized by low abundance and richness (Allan, 1995). However, this relationship has not been adequately established for aquatic insects. In a study that identified the abiotic factors best predicting species richness and abundance of macroinvertebrates from Patagonian streams and rivers, current speed, conductivity, substrate type, and abundance of aquatic plants were the main factors (Miserendino 2001). These predictors are similar to the four primary factors controlling river fauna (Hynes 1966): 1) dissolved salts, 2) current, 3) temperature, and 4) dissolved oxygen. Hynes also stressed the importance of plants that influence biota by providing shelter, food, and surface area for growth of attached algae, which are later grazed by invertebrates.

Significant predictors for EPT taxa, Georgia EPD index, and Georgia AAS index were either related to flow or specific conductance, which were highly correlated to flow.

Flow controls many structural attributes of streams, such as current velocity, substratum stability, channel geomorphology, and habitat volume (Poff and Ward 1989), which in turn influence factors of critical importance to macroinvertebrates such as routing and retention of organic matter (Gomi et al. 2002). Water velocity, coupled with the abiotic and biotic factors that it influences, may be the most important environmental factor influencing stream ecology (Allan 1995, Poff and Ward 1989, Hynes 1970). Hydrologic disturbances such as flood and drought can affect biota because frequency, duration, and intensity of such disturbances influence the response and recovery time of communities (Gomi et al. 2002).

Differences in streamflow magnitude and origin between streams A and C may explain why these streams would be different in terms of EPT taxa, Georgia EPD index, Georgia AAS index, and percent Elmidae, but D also had low flow compared to C. Over a 638 day period, D had a higher number of zero flow days (206) compared to A (161). What is not expressed in this metric of intermittent conditions is the amount of water still present in the channel but not detected at the flume. Zero flow conditions at A versus D were very different. The downstream reaches of A were sometimes completely dry, whereas those of D always had water and could have provided a hyporheic refuge. Griffin and Perry (1993) noted that the hyporheic zone may provide a refuge from a longer-term disturbance, such as drought. This may partially explain why the lower flow conditions in D did not result in a depressed fauna as detected in A.

Another difference between A and D lies in dissolved oxygen concentrations. For the entire study period, average dissolved oxygen in A was 4.05 mg/L (42% saturation of dissolved oxygen), while D was 6.00 mg/L (64% saturation of dissolved oxygen).

Georgia Department of Natural Resources (2004) water quality regulations specify a daily average of 5.0 mg/L and no less than 4.0 mg/L for waters supporting warm water species of fish, which indicates that dissolved oxygen conditions in A are considered low for warm water fish species. Crisman et al. (1998) investigated the relationship of oxygen concentrations in shallow Florida lakes to humic color, trophic state, and lake size. Percent oxygen saturation was negatively correlated with color. Although quantitative measurements of humic color were not done in the current study, qualitative observations indicate that A is highly colored relative to D. The source of color for A is likely export of organic matter from a small, headwater pond that contributes flow to A during wet periods. Decomposition of dissolved organic matter consumes oxygen. Allan (1995) indicated that the biota of flowing waters is highly dependent on availability of oxygen, but low dissolved oxygen is usually not limiting. However, under certain conditions, such as drought, it can be important. The low dissolved oxygen potentially influenced by color in A coupled with low flow conditions may have restricted the fauna.

Another indication of the influence of the headwater pond in watershed A is the representation of functional feeding groups for stream A relative to B, C, and D. The assemblages of streams B, C, and D had functional feeding groups similar to headwater streams described in the River Continuum Concept (RCC) (Vannote et al. 1980), with shredders and collectors being co-dominant. However, predators were equally dominant in these streams. Collectors dominated stream A, which according to the RCC is an assemblage typical of large rivers with smaller detrital particle size distribution. The small headwater pond is likely the primary source of detritus for stream A because leaf litter inputs to each stream were not significantly different, and the pond would provide

the required residence time for large organic matter to be broken down into smaller particle sizes.

Metrics Assessment

Many metrics were calculated both to describe individual macroinvertebrate assemblages and to compare assemblages between upstream and downstream, stream to stream, and year to year. The effectiveness of each metric in describing the study streams varied. Percent dominant taxa, number of Chironomidae taxa, percent Diptera, percent Chironomidae, percent filter feeders, and number of clinger taxa were not different between sampling periods, position, or streams. Percent dominant taxa, number of Chironomidae taxa, percent Diptera, and percent Chironomidae generally described the importance of Chironomidae in the assemblage. The percent dominant taxa was greatly influenced by Chironomidae because 21 of the 32 total samples were dominated by this group. Percent Diptera was also heavily influenced by the Chironomidae because 44 of 63 total dipteran taxa were Chironomidae. Metrics should have a specific range of variability in order to discriminate between sites (Barbour et al. 1999), but Chironomidae were simply ubiquitous at all sites; therefore, the methods of comparison employed in this study did not detect differences in metrics that described this group.

Percent filter feeders and number of clinger taxa also were not different between sampling periods, position, or streams. When present, filter feeders and clinger taxa were not abundant. Barbour et al. (1999) suggested that metrics with too many zero values should be eliminated from a pool of potential metrics, suggesting that these metrics may not be useful for evaluating the describing the macroinvertebrate assemblage of study streams. Furthermore, Karr (1999) indicated that metrics based on functional feeding groups can respond differently to disturbance in different streams.

Abundance, EPT taxa, and total taxa showed significant differences due to time and stream and consistently indicated that December 2001 had significantly lower values than February 2003. February 2002 did not differ from December 2002. Abundance can vary greatly naturally (Karr 1999). In a study of macroinvertebrate assemblages in small streams of Alabama, total taxa was significantly different between years with varying degrees of flow permanence, and EPT taxa showed the strongest relationship with stream permanence (Feminella 1996). Of abundance, total taxa, and EPT taxa, the latter had the strongest relationship with flow in the current study as there was significantly lower EPT taxa in stream A versus C.

The number of EPT taxa is often lower in headwater reaches relative to larger downstream reaches and this may limit the usefulness of this metric in biological monitoring unless it is used in comparisons of streams with similar size (Wallace et al. 1996). As mentioned previously, streams in this study have numbers of EPT taxa comparable with studies of low and higher order streams; however, it is still advisable to compare streams of similar size and characteristics in biological monitoring studies. EPT taxa, a single metric, can detect changes in macroinvertebrate assemblages as can biotic indices that integrate several metrics together to determine condition (Karr and Chu 1999). Wallace et al. (1996) investigated the ability of the North Carolina Biotic Index (NCBI) (Lenat 1993) and EPT taxa to track manipulation of macroinvertebrate communities in headwater streams due to insecticide treatments. NCBI and EPT taxa reflected changes due to treatment and effectively indicated improved conditions during recovery, with EPT taxa being the easier to use (time for sample processing and ease of application). EPT taxa were sensitive to disturbance but relatively insensitive to natural

disturbances such as extreme discharges. However, Stone and Wallace (1998) found that EPT taxa increased and NCBI decreased in headwater streams that had undergone forest harvesting, suggesting that more than one metric or index should be used in biological monitoring studies.

Percent Elmidae was a metric that was utilized in southwestern Georgia to differentiate between fenced and unfenced agricultural streams (Muenz 2004). It was effective in detecting interannual variation and was sensitive to influences of inorganic phosphorus. In biomonitoring, a metric is selected because it reflects some aspect of the system biological condition (Karr and Chu 1999). Percent Elmidae reflects the range of conditions that affect one family of aquatic beetles. While these conditions may have similarities with other groups of insects, the metric may be too specific for use as an indicator of the assemblage as a whole. However, this metric may have utility when combined with other metrics.

Three indices were calculated to describe the macroinvertebrate assemblage and to compare assemblages between upstream and downstream, stream to stream, and year to year. The effectiveness of each index in describing the study streams varied. The Georgia EPD index and Georgia AAS index detected differences in the macroinvertebrate assemblages due to time and stream, but the Florida SCI did not. Also, the Georgia EPD index and Georgia AAS index scored most streams as Good or Excellent, respectively, whereas Fl SCI scored the majority of samples as Poor. There are similarities and inherent differences among these indices. Georgia EPD index and Florida SCI both combine results from selected metrics that are weighted based on regional differences, then summed to provide a final score. For the Georgia EPD, the score is

compared to a regional reference stream score, and this score is converted into a final assessment of ecological condition (GA DNR 2002). The Florida SCI converts the score for the stream in question into a final assessment of ecological condition by determining where it falls within certain ranges of values that were established by statistical analysis of FDEP reference stream data from northwestern Florida (FDEP 2004). The Georgia EPD index scored most streams Good in their undisturbed condition, which was somewhat surprising given that the average area of the watersheds in the study was 0.40 km^2 compared to 38.4 km^2 for reference watershed used for comparisons. The number of species is generally thought to increase from headwaters to mid-order streams, then to decrease again in larger rivers (Allan 1995, Vannote et al. 1980). Overall, Georgia EPD index scores might improve if compared to a reference stream closer to the size of the study streams.

Florida SCI scores were low for the study streams. The Florida SCI scores were calculated based on a 100 individual subsample required as a standard operating procedure (FDEP 2004). In general, when a 100 individual subsample was randomly generated from the entire list of species present in a sample, dominant taxa tended to displace more sensitive taxa such as EPT taxa that had a detrimental effect on the overall SCI score.

Doberstein et al. (2000) analyzed 500 random 100-individual subsamples from a stream in the Puget Sound area of Washington that was minimally disturbed. They found that the ability to discern biological condition was reduced to a point where the potential for an ill-informed water resource decision was high when a 100-individual subsample was used. Additional subsamples at different intervals (100, 200, 500-individual

subsamples) and subsequent calculations of the SCI may be warranted to determine if subsampling was the cause of the depressed SCI scores in the current study. Subsampling did not seem to have as drastic an effect on Georgia EPD index scores, as overall condition of streams actually increased slightly as a result of calculating the index with a 200-individual subsample versus the entire sample.

The Georgia AAS index is calculated as presence/absence of invertebrates at the order level (GA DNR 2000) versus genus and species levels, which are used in calculation of Georgia EPD index and Fl SCI. Genus or species-level taxonomy can yield the greatest benefits for biological monitoring studies especially when results could influence management decisions (Lenat and Resh 2001). However, studies utilizing higher taxonomic levels (e.g., family, order) discriminate among ecoregions (Feminella 2000) and describe community patterns (Bowman and Bailey 1997) equally as well as lower taxonomic levels (e.g., genus, species). One argument against using lower taxonomic levels is that genus and/or species information increases the cost and ecological noise of bioassessment (Bailey et al. 2001). Feminella (2000) also proposed that coarser taxonomic levels may provide adequate resolution for relatively unimpaired streams that may differ due to natural within-catchment variation. Analysis of Georgia AAS index and Georgia EPD values indicated that there were significant differences due to time and stream. However, comparisons between time for AAS and EPD varied in that December 2001 had significantly lower AAS index values than February 2003. February 2002 and December 2002 were not significantly different. December 2002 EPD index values were significantly higher than the remaining sampling periods, which were not significantly different. Georgia AAS results for differences between time

periods agreed with the results for abundance, total taxa, and EPT taxa, whereas EPD index results only agreed with percent Elmidae. For differences due to stream, Georgia AAS index and Georgia EPD index had similar results as EPT taxa and percent Elmidae in that A had significantly lower values than C. These results do not suggest that the Georgia AAS index is better at describing differences in this study than the Georgia EPD, but they do suggest that a great deal of information is not lost due to the coarser taxonomic resolution employed in Georgia AAS index.

Conclusions

The biota of small, sandy substrate headwater streams in the Gulf coastal plain are dominated by Diptera, but they are not low in taxa richness and have taxa from the orders, Ephemeroptera, Plecoptera, and Trichoptera, considered sensitive to human disturbance.

Since all sites in this study were in a relatively undisturbed condition and were similar in location and physical characteristics, the macroinvertebrate fauna was expected to be comparable between sites. However, there was variation in abundance, EPT taxa, total taxa, Georgia AAS index, and Georgia EPD index between the four undisturbed adjacent headwater streams and from year to year. Natural variability, year-to-year and stream to stream variance can be significant, even within sub-watersheds of a small catchment. Hydrology may be the controlling variable for the natural variability displayed in this study. However, due to small sample size, additional data are needed to strengthen this relationship. Hydrology should be monitored and considered in manipulative studies where macroinvertebrates are used as a response variable, because even small interannual differences in hydrology can have a significant effect on organisms. Establishing this natural variability is important to consider in manipulative

studies; however, a robust baseline dataset can lessen the potential of natural variation clouding identification of a treatment effect.

Of all metrics examined in this study, abundance, EPT taxa, total taxa, GA AAS index, and GA EPD index detected differences in macroinvertebrates due to time and stream, and therefore best described differences in the macroinvertebrate assemblage.

Differences in the macroinvertebrate assemblages between A and C, but not between A and B or C and D, support the overall Dry Creek study design and suggest that A and D would be appropriate reference streams for B and C, respectively. No differences were detected between upstream and downstream sampling locations within each stream suggesting that upstream reaches within the same stream will be appropriate references for manipulations occurring on downstream reaches.

APPENDIX
SPECIES LIST AND TOTAL ABUNDANCE FOR EACH SITE

Order	Family	Subfamily	Genus	Species	Site							
					A1	A2	B1	B2	C1	C2	D1	D2
Diptera												
	Chironomidae											
		Chironominae	*Chironomus*	*sp.*	15	1	2	0	0	9	0	2
		Chironominae	*Cryptochironomus*	*sp.*	0	1	18	4	5	54	5	0
		Chironominae	*Microtendipes*	*pedellus gp.*	0	15	7	6	3	16	4	0
		Chironominae	*Paracladopelma*	*sp.*	0	0	10	1	0	0	0	0
		Chironominae	*Paralauterborniella*	*nigrohalterales*	0	0	2	0	0	0	3	2
		Chironominae	*Paratendipes*	*sp.*	1	29	3	7	7	7	10	1
		Chironominae	*Polypedilum*	*fallax*	0	0	34	9	9	16	5	1
		Chironominae	*Polypedilum*	*flavum*	0	33	80	58	168	928	63	109
		Chironominae	*Polypedilum*	*halterale*	0	0	3	10	7	0	0	1
		Chironominae	*Polypedilum*	*illinoense*	32	8	5	22	9	12	1	2
		Chironominae	*Rheotanytarsus*	*sp.*	0	0	22	3	43	25	45	0
		Chironominae	*Stenochironomus*	*sp.*	0	1	2	0	0	15	1	2
		Chironominae	*Tanytarsus*	*sp.*	10	4	53	12	22	40	38	19
		Chironominae	*Tribelos*	*fuscicorne*	7	0	0	0	0	0	0	0
		Chironominae	*Tribelos*	*jucundum*	122	38	31	84	33	38	0	0
		Chironominae	*Tribelos*	*sp.*	1	6	1	5	9	14	8	2
		Chironominae	*Zavrelia*	*sp.*	0	0	0	0	0	0	4	1
		Orthocladiinae	*Brillia*	*flavifrons*	0	0	0	0	2	0	0	0
		Orthocladiinae	*Corynoneura*	*sp.*	2	6	19	8	0	7	6	9
		Orthocladiinae	*Eukiefferiella*	*claripennis gp.*	0	0	0	0	0	0	3	0
		Orthocladiinae	*Heterotrissocladius*	*marcidus*	8	35	0	16	0	0	0	0
		Orthocladiinae	*Limnophyes*	*sp.*	0	1	0	0	0	1	0	0
		Orthocladiinae	*Nanocladius*	*distinctus*	0	7	0	111	0	1701	2	296
		Orthocladiinae	*Orthocladiinae*		1	14	0	4	0	0	0	0
		Orthocladiinae	*Orthocladius*	*lignicola*	0	0	0	15	7	18	3	0
		Orthocladiinae	*Parachaetocladius*	*sp.*	11	6	3	33	0	72	4	29
		Orthocladiinae	*Parametriocnemus*	*sp.*	10	14	401	87	581	715	183	75
		Orthocladiinae	*Pseudorthocladius*	*sp.*	3	0	0	8	0	30	0	0
		Orthocladiinae	*Pseudosmitta*	*sp.*	1	4	0	0	0	0	0	0
		Orthocladiinae	*Rheocricotopus*	*tuberculatus*	0	0	0	0	0	0	3	0
		Orthocladiinae	*Smittia*	*sp.*	6	0	0	0	0	0	0	0
		Orthocladiinae	*Thienemanniella*	*xena*	0	0	4	3	0	0	1	1
		Orthocladiinae	*Xylotopus*	*par*	0	0	0	0	0	48	1	15
		Prodiamesinae	*Odontomesa*	*fulva*	0	0	3	0	0	0	0	0
		Tanypodinae	*Ablabesmyia*	*mallochi*	0	0	7	5	9	52	0	0
		Tanypodinae	*Alotanypus*	*sp.*	26	80	10	4	0	72	1	1
		Tanypodinae	*Cantopelopia*	*gesta*	1	4	0	11	0	48	0	11
		Tanypodinae	*Clinotanypus*	*pinguis*	0	1	0	0	0	1	0	1
		Tanypodinae	*Conchapelopia*	*sp.*	13	10	279	65	186	213	108	271
		Tanypodinae	*Labrundinia*	*pilosella*	0	255	0	38	2	24	0	5
		Tanypodinae	*Larsia*	*sp.*	0	0	0	27	2	36	0	0
		Tanypodinae	*Procladius*	*sp.*	0	2	5	10	2	15	5	1
		Tanypodinae	*Tanypodinae*		0	28	0	46	0	120	0	142
		Tanypodinae	*Zavrelimyia*	*sp.*	25	22	35	119	35	255	75	50
	Tipulidae				0	1	1	3	3	0	1	1
			Tipula/Nippotipula		0	0	16	12	12	23	20	11
			Pseudolimnophila		1	3	24	13	95	174	32	109
			Pilaria		3	3	5	4	4	3	1	8
			Dicranota		0	0	0	0	0	0	0	0
			Hexatoma		0	2	25	1	1	32	3	3
			Erioptera		1	5	2	0	0	1	0	0
			Austrolimnophila	*sp.*	0	0	0	0	0	0	1	0
			Limnophila		0	1	0	0	0	0	0	0
	Ceratopogonidae		*Bezzia/Palpomyia*		100	104	169	125	98	266	89	117
			Alluaudomyia		6	0	0	0	0	0	0	0
			Dasyhelea		0	0	0	0	0	0	1	0

Order	Family	Subfamily	Genus	Species	Site							
					A1	A2	B1	B2	C1	C2	D1	D2
	Ptychopteridae		*Bittacomorpha*		4	0	7	18	0	7	0	2
	Simulidae				83	67	34	73	102	189	131	23
	Tabanidae				4	0	1	0	13	29	17	5
	Culicidae				9	5	0	3	0	7	1	0
	Dixidae				0	0	0	0	2	0	0	11
	Psychodidae		*Pericoma*		0	0	0	0	1	1	0	0
			Psychoda		2	0	0	0	0	0	0	0
	Dolichopodidae				2	0	0	0	0	6	1	0
Isopoda												
	Asellidae		*Caecidotea*		1	220	1	0	0	0	1	0
Amphipoda												
	Crangonyctidae		*Cragonyx*		310	194	21	37	4	32	55	0
Odonata												
	Gomphidae				0	0	1	0	0	0	1	0
			Ophiogomphus		0	0	0	0	2	0	0	0
			Gomphus		0	0	0	0	1	3	0	0
			Progomphus		0	0	0	0	9	0	0	1
	Cordulegastridae				0	1	0	0	0	0	0	0
			Cordulegaster		0	0	2	2	2	2	8	0
				sayi	0	0	0	0	0	3	1	1
	Aeshnidae		*Boyeria*	*vinosa*	0	0	0	0	1	0	0	0
	Libellulidae				0	1	2	4	0	0	0	2
		Libellulinae			0	0	0	0	0	2	0	0
		Corduliinae			0	0	2	1	0	0	0	0
	Calopterygidae		*Calopteryx*		0	0	0	1	0	4	2	0
Plecoptera												
	Nemouridae		*Amphinemura*		0	1	117	65	229	55	228	126
	Perlodidae		*Ciloperla*	*cilo*	0	0	4	3	11	3	7	5
	Perlidae		*Perlesta*		0	0	0	2	8	1	0	0
	Capniidae		*Allocapnia*		0	0	0	0	0	0	70	189
Ephemeroptera												
	Baetidae		*Acerpenna*	*pygmaea*	0	0	2	1	26	198	0	0
			Baetis	*intercalaris*	0	0	0	0	1	0	0	0
			Diphetor	*hageni*	0	0	0	0	2	0	3	0
			Pseudocloeon	*sp.*	0	0	0	1	0	1	1	0
	Ephemerelliae		*Ephemerella*	*sp.*	0	0	2	1	4	0	0	0
			Eurylophella	*doris*	0	0	0	0	0	1	0	0
	Ephemeridae		*Hexagenia*	*sp.*	0	0	0	0	7	7	0	0
			Hexagenia	*limbata*	0	0	0	0	0	5	0	0
	Heptageniidae		*Stenonema*	*smithae*	0	0	0	0	13	0	1	0
	Leptophlebiidae		*Habrophlebiodes*	*sp.*	0	0	16	25	170	19	34	35
			Leptophlebia	*sp.*	0	0	2	1	21	19	36	10
Trichoptera												
	Calamoceratidae		*Anisocentropus*	*pyraloides*	0	0	1	0	27	5	9	10
	Odontoceridae		*Psilotreta*	*frontalis*	0	0	0	0	8	0	4	8
	Hydropsychidae		*Diplectrona*	*modesta*	0	0	4	1	1	9	0	0
			Cheumatopsyche		0	0	0	0	7	12	0	0
	Philopotamidae		*Chimarra*		0	0	0	0	24	22	3	0
	Limnephilidae		*Pycnopsyche*		2	5	12	16	18	23	24	14
	Molannidae		*Molanna*	*blenda*	0	0	0	0	0	0	0	1
	Leptoceridae		*Triaenodes*		0	0	0	0	1	2	0	0
	Sericostomatidae		*Agarodes*		0	0	0	0	1	0	0	0
	Dipseudopsidae		*Phylocentropus*		0	1	0	0	0	2	0	0
	Rhyacophilidae		*Rhyacophila*	*carolina*	0	0	1	4	0	0	0	0
	Polycentropodidae		*Polycentropus*		0	0	0	1	0	0	0	0

Order	Family	Subfamily	Genus	Species	Site							
					A1	A2	B1	B2	C1	C2	D1	D2
Coleoptera												
	Dryopidae		*Helichus*		0	0	59	29	2	9	10	30
			Pelonomus		0	0	0	0	0	0	0	1
	Dytiscidae				2	0	1	21	0	0	3	0
			Thermonectus	*basilaris*	1	0	0	0	0	0	0	0
			Agabus	*sp.*	2	0	0	0	0	0	0	0
			Agabus	*astrictovittatus*	1	0	0	0	0	0	0	0
			Copelatus	*glyphicus*	0	0	0	0	0	0	1	0
			Rhantus	*sp.*	0	0	0	0	0	0	0	0
			Rhantus	*calidus*	1	0	1	0	0	1	0	0
			Coptotomus	*loticus*	0	0	0	0	0	1	0	0
			Hydaticus	*bimarginatus*	0	0	0	0	0	1	0	0
			Neoporus	*sp.*	0	1	1	32	4	1	16	34
			Neoporus	*striatopunctatus*	0	0	9	7	0	1	10	7
			Neoporus	*blanchardi*	0	4	0	0	1	0	0	0
			Neoporus	*undulatus*	0	0	0	5	0	0	0	0
	Elmidae				0	0	0	0	0	0	0	0
			Stenelmis		0	5	18	6	79	162	156	134
			Dubiraphia		0	0	0	0	8	4	5	8
			Macronychus		0	0	0	0	2	0	0	0
	Scirtidae				3	1	0	3	0	6	1	10
	Hydrophilidae				0	0	0	0	0	0	0	2
			Helocombus		3	0	1	0	0	2	0	0
	Lampyridae				1	1	0	1	0	0	0	0
	Ptilodactylidae		*Anchytarsus*		0	0	0	0	1	4	0	1
Hemiptera												
	Notonectidae		*Notonecta*		1	0	0	0	0	0	0	0
Megaloptera												
	Chauliodinae		*Nigronia*		0	0	0	0	2	0	3	2
			Chauliodes		1	0	0	1	0	1	0	0

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BIOGRAPHICAL SKETCH

I graduated from Clemson University with a B.S. in biological sciences in 1997. After graduation I took an internship position with International Paper to work on a forestry best management practices effectiveness-monitoring project. It was during this project that I first began working with benthic macroinvertebrates. In the past 6 years with International Paper I have had various responsibilities for the company, such as providing technical support to company foresters and other employees on matters related to water quality and water resource management; leading or assisting in study installation, data collection, analysis, reporting, and pursuit of funding for internal and external research projects; leading or assisting in the development of and presentation of training on company environmental management policies; and representing International Paper at technical, policy meetings/activities. I have also participated in state and local programs such as Georgia Adopt-A-Stream, Georgia Rivers Alive, and Keep America Beautiful. While in graduate school I remained employed by International Paper as a Watershed Specialist, then later as the Coordinator of Partnerships.