IMPROVING INFERENCE FROM BIOLOGICAL ASSESSMENTS:

LINKAGES BETWEEN ECOLOGICAL DRIVERS & AQUATIC COMMUNITIES IN PIEDMONT STREAMS

by

GWENDOLYN DENISE CARROLL

(Under the Direction of C. Rhett Jackson)

ABSTRACT

Since the establishment of the US Clean Water Act in 1972, biological assessments using fish and macroinvertebrates have been employed to detect stream perturbations and are required to assess water quality in locations across the globe. Despite the widespread use of such techniques, many questions remain unanswered concerning relationships between biological factors and physical habitat. As Georgia continues to grow, it is important to identify how that growth will impact aquatic communities and water quality. A better understanding of linkages between ecological drivers and aquatic communities is useful for predictive modeling of watershed conditions and the development of management strategies, providing suggestions to help maintain or improve the health of a watershed. This study investigates relationships between aquatic communities and 1) urbanization and 2) turbidity, SSC, and percent fines, as well as examining interrelationships between fish and macroinvertebrate assemblages. Our results indicate biotic community metrics were highly sensitive to land cover, specifically the fractions of urban land, imperviousness, and forest. Nearly all fair, good, and excellent fish IBI scores occurred in basins with more than 50 percent forest and less than 15 percent urban area and less than 4 percent impervious surface, indicating that Georgia Piedmont streams may be more sensitive to urbanization effects than streams in other parts of the country. Overall, sites with less urbanization had greater litter standing crops during December; however, higher rates of retention occurred in more urbanized areas. We infer that urban streams balance litter export with additional horizontal inputs from storm drains that act to increase the litter source area. Macroinvertebrate shredder taxa richness was negatively affected by watershed landuse, but shredder abundance and percent composition were not. Shredder abundance and composition were not correlated to litter availability. Neither baseflow suspended sediment concentrations, baseflow turbidity, nor bed particle size distributions were significantly related to watershed imperviousness or other land use metrics. However, these metrics added significantly to explanatory models of important biological metrics. Macroinvertebrate and fish biotic indices do not provide the same information with regard to water quality, although fish index of well being scores are more highly correlated than IBI scores.

INDEX WORDS: leaf litter inputs, leaf litter budget, litter availability, standing crop, shredder abundance, landuse, urbanization, biotic indices, sediment, percent fines, turbidity, SSC.

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CHAPTER 1

Introduction: Improving inference from biological assessments

Biological assessments using fish and macroinvertebrates have been employed to detect stream perturbations since the establishment of the US Clean Water Act in 1972 (Plafkin et al. 1989, Angermeier and Karr 1994, Naiman et al. 1995). Current legislation requires their usage to assess water quality in locations across the globe (Gordon 2004, Hawkins 2006, Uriarte and Borja 2009). Many questions remain unanswered concerning relationships between biological factors and physical habitat in the Piedmont ecoregion of Georgia, despite the widespread use of such techniques. Interpretation of biological assessments is clouded by geomorphological differences between streams and among sites within the same stream (Karr 1981, Plafkin et al. 1989, Merritt and Cummins 1996), local versus watershed scale effects of land-use characteristics (Stribling et al. 1998), legacy effects of historical land use, information provided by the organisms used in the assessment (Fitzpatrick et al. 2004), and the particular index used to calculate stream health (Iliopoulou-Georgudaki et al. 2003).

Background: Use of Biotic Indices

According to Carigan and Steedman (2000), two of the most important questions concerning sustainability of healthy waters are related to land use issues: (1) How has human activity threatened the ability of watersheds to produce clean water? and (2) How will new activity exacerbate or counteract existing conditions? Indices of biotic integrity (IBIs) are attractive water quality indicators to help answer these questions because biological community health integrates temporal variability of water quality conditions.

Indices of biotic integrity (IBI) have been used extensively to measure the ecological health of small to medium-sized freshwater streams and wetlands (Barbour et al. 1999, Stevenson and Hauer 2002). They have been used, more recently, for assessing large rivers, lakes, reservoirs, oxbows, estuaries, and coral reef communities (Dauer et al. 2000, Seegert 2000, Hughes et al. 2002). Although they were originally developed to detect organic contamination from treated wastewater discharges, they have since been used: 1) to assess aquatic communities in response to mitigation/restoration efforts and best management practices, 2) for detection of contaminants in the water column and sediments (e.g. metals, hazardous wastes, acidification, wastewater), 3) to identify thermal alterations, 4) to measure effects of increased sedimentation and turbidity, 5) for detection of salinization from both salt water intrusion and road deicing salts, 6) to determine the influence of various land cover conditions, and 7) as a tool to evaluate enrichment, eutrophication, and organic loading (Barbour et al. 1999).

Biotic assessments are usually conducted as before and after surveys, such as a pollutant release or treatment application, or as part of a regular sampling regime to test water quality in accordance with government mandates (Rosenberg and Resh 1993). Each IBI consists of metrics which fit into five categories: 1) richness, 2) taxa composition, 3) relative abundance, 4) actual population abundance, and 5) reproduction (Radar et al. 2001). Calculation of these metrics requires collection of quantitative and qualitative data that can be used to identify the distribution of tolerant and intolerant species, individual health (i.e., lesions or parasite load), and the presence or absence of specific indicator species, such as mayflies, stoneflies, and caddisflies for aquatic insects and darters for fish communities (Rosenberg and Resh 1993).

The Environmental Protection Agency reported, in 2002, that 440,000 stream and river miles had been assessed nationwide using at least one form of biocriteria (i.e. fish, macroinvertebrates, algae, or a combination of organisms). Biotic surveys in Georgia are most often conducted to determine if streams and rivers meet their aquatic life designated use standards in conjunction with total maximum daily load (TMDL) requirements. Georgia recently established a standard protocol for sampling and calculating fish IBIs, but is still in the process of finalizing standard procedures for computing macroinvertebrate IBIs.

Observed relationships between urbanization and riparian cover, shredder abundance, and stream leaf litter standing crops

Organic inputs from the surrounding landscape are a significant contributor to total stream web function, providing food and habitat for aquatic organisms. However, allochthonous contributions in aquatic ecosystems are complex (Rowe et al. 1996, Benfield 1997, Hutchens and Wallace 2002) because they are controlled by biological, physical, and chemical factors that can be highly variable and dependent upon both local and catchment conditions (Webster and Benfield 1986, Allan 1995, Benfield 1997). Understanding how human activities within the watershed alter ecological health of streams can aid conservation efforts by directing management decisions. In chapter two, data from 13 streams were analyzed to determine the effects of urbanization on the quantity of leaf litter inputs, the quantity of leaf litter standing crop, and leaf litter retention.

Use of simple suspended and bed sediment metrics to explain biotic index variability and relationships between fish and macroinvertebrate assemblage metrics in Georgia Piedmont Streams

Water quality is strongly influenced by the soil conditions, vegetation coverage, and climate within the watershed (Carignan and Steedman 2000, Tong and Chen 2002). The USDA cited soil erosion and its associated impacts as the greatest threat to the sustainability of our water resources (Huang et al. 2006). Although sedimentation occurs naturally, it is exacerbated by anthropogenic disturbances, such as urban development, agriculture, forestry, and mining (Waters 1995). Biotic impairment related to sedimentation results from a variety of stressors, such as reduced primary production, alteration of suitable substrate habitats, gill damage that inhibits breathing, death of eggs or larvae, and increasing tolerant species (Barrett et al. 1992, Waters 1995, Vuori and Joensuu 1996, Whol and Carline 1996, Boyle 1997, Wood and Armitage 1997, Wulff et al. 1997, Angradi 1999). Simple sedimentation metrics such as turbidity, suspended sediment concentration, and particle size analysis may serve as predictive tools for evaluating biotic condition.

Fish, macroinvertebrate, periphyton, and bird communities have all been used to assess the ecological status of streams and rivers. Fish and macroinvertebrates are the most commonly used organisms to assess biotic impairment, in Georgia. Streams that do not meet their designated use guidelines based on either community index are placed on the 303d list, established by the Clean Water Act to identify impaired waters, and referred to the Georgia Environmental Protection Division for the development of a TMDL prescription. When follow-up assessments are conducted either one of the indices may be used in the assessment, but both are not required. As such, it is important to understand how the two indices are related and if they provide similar information regarding stream degradation.

Chapter three evaluates the relationship between 1) baseflow turbidity and biotic conditions, and 2) correlations between the percentage of fines in streambed sediment and biotic condition. It also examines the use of biotic indices, both fish and macroinvertebrates, as a predictor of water quality and attempts to determine which metrics within the indices are most highly correlated. The analysis is confined to the Piedmont physiographic region where past and present activities have eroded channels and sediment is considered a major driver of biotic conditions.

Discussion - Further Consideration for "Urbanization Influences on Aquatic Communities in Northeastern Illinois Streams"

Chapter four addresses the question of which biotic index is the most appropriate for water quality assessments. Chapter four uses published data by Fitzpatrick et al. (2004) to directly compare fish and macroinvertebrate indices as predictors of the influence of land use on water quality. The work by Fitzpatrick *et al.* (2004) addresses the influence of urbanization on aquatic communities, but fails to compare results of the fish and macroinvertebrate biotic indices directly. They conclude that the Illinois fish alternative index of biotic integrity (AIBI) and macroinvertebrate index (MBI) scores respond similarly to land use changes, decreasing as agricultural land undergoes surbanization. We compared the relationship between their AIBI and MBI using a simple linear regression and found that the MBI overestimates water quality; using the two indices interchangeably requires calibration. We further discuss the difficulties involved in relating the two indices and the necessity to further investigate these interconnections in order to develop effective biomonitoring surveys.

Overall Conclusions

Our results indicate biotic community metrics were highly sensitive to land cover, specifically the fractions of urban land, imperviousness, and forest. Nearly all fair, good, and excellent fish IBI scores occurred in basins with more than 50 percent forest and less than 15 percent urban area and less than 4 percent impervious surface, indicating that Georgia Piedmont streams may be more sensitive to urbanization effects than streams in other parts of the country. Overall, sites with less urbanization had greater litter standing crops during December; however, higher rates of retention occurred in more urbanized areas. We infer that urban streams balance litter export with additional horizontal inputs from storm drains that act to increase the litter source area. Macroinvertebrate shredder taxa richness was negatively affected by watershed land use, but shredder abundance and percent composition were not. Shredder abundance and composition were not correlated to litter availability. Neither baseflow suspended sediment concentrations, baseflow turbidity, nor bed particle size distributions were significantly related to watershed imperviousness or other land use metrics. However, these metrics added significantly to explanatory models of important biological metrics. Macroinvertebrate and fish biotic indices do not provide the same information with regard to water quality; While MBI stream condition categories are more closely aligned with the IWB categories than with IBI scores, calibration is required to use one in place of the other. Baseflow turbidity, specific conductivity, and bed particle size distribution measurements should be collected with all biotic assessments to help explain variability in the indices.

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CHAPTER 2

Observed relationships between urbanization and riparian cover, shredder abundance, and stream leaf litter standing crops.

Abstract:

Leaf litter inputs, an essential driver of river ecosystem structure, are heavily influenced by local and basin wide changes. Urbanization can affect leaf litter inputs at both scales, but the consistency of ecosystem response is unknown. In this study, we investigated relationships between basin land use coverage, reach-scale canopy conditions, and macroinvertebrate shredder metrics in 42 streams across a gradient of land use in the Piedmont of Georgia, USA. We then evaluated how urbanization affects vertical leaf litter inputs and also autumn and spring leaf litter availability in a subset of 13 streams. Reach-scale riparian conditions controlled canopy cover and vertical litter inputs, but leaf litter standing crop dynamics were complex and seemed to be controlled at the catchment level by factors beyond land use coverage. Overall, sites with less urbanization had greater litter standing crops during December; however, higher rates of retention occurred in more urbanized areas. We infer that urban streams balance litter export with additional horizontal inputs from storm drains that act to increase the litter source area. Macroinvertebrate shredder taxa richness was negatively affected by watershed land use, but shredder abundance and percent composition were not. Shredder abundance and composition were not correlated to litter availability. The apparent

importance of horizontal leaf litter inputs to urban stream detritus budgets indicates that further studies are warranted to characterize and quantify litter inputs from storm drains and to gain a better understanding of how these inputs effect shredder populations.

Introduction:

Stream condition can be studied at multiple scales ranging from small patch analysis to assessments of basin-wide impacts (Allan 1995). Historically, lotic system research has focused on local conditions, either within the stream channel or reach-interactions with the immediate riparian area; technological advances and a greater understanding of whole system ecology over the past 25 years have led to catchment-scale analyses of aquatic ecosystems (Gregory et al. 1991, Allan and Johnson 1997). Leaf litter processing integrates all hierarchical scales because it is linked to both local conditions (e.g., leaf species, shredder abundance, and quantity of riparian vegetation) and watershed features (e.g., hydrologic condition, sedimentation, and percent impervious surface) (Hauer and Lamberti 1996, Johnson and Covich 1997, Royer and Minshall 2003, Paul et al. 2006).

Inputs of leaf litter are an essential component of river ecosystem foodwebs (Hall et al. 2001). Leaves and woody debris are substantial contributors to total stream system function because they serve as food and habitat for heterotrophic organisms from bacteria to fish, both directly and indirectly (Vannote et al. 1980, Cummins et al. 1989, Tait et al. 1994, Hall et al. 2001, Wipfli 2005). However, leaf litter input and standing crop dynamics are complex in aquatic ecosystems (Rowe et al. 1996, Benfield 1997, Hutchens and Wallace 2002) because they are controlled by biological, physical, and chemical factors that can be highly variable (Webster and Benfield 1986, Allan 1995, Benfield 1997). For example, leaf breakdown generally occurs at higher temperatures or increased nutrient concentrations due to faster microbial activity, but several studies have shown that temperature is not always the key factor controlling breakdown. Short et al. (1980) found rapid breakdown rates for water temperatures at or near 0^{o}C, while other research indicates no significant relationship with temperature (Short and Ward 1980, Grubbs and Cummins 1994, Rowe et al. 1996, Sponseller and Benfield 2001).

The variability of such factors and mixed results from previous studies makes it difficult to draw conclusions about the relationship between leaf litter inputs or standing crops and land use management (Meyer and Johnson 1983, Webster and Benfield 1986, Suberkropp 1995, Petts and Calow 1996, Paul 1999, Graca et al. 2001, Herbert 2003). Earlier research has suggested that urbanization decreases leaf litter inputs and accelerates leaf litter breakdown and flushing, because it removes riparian vegetation reducing the quantity of allochthonous inputs and increasing stream water temperatures, shifting detrital contributions to non-native and planted species (Paul and Meyer 2001, Walsh et al. 2005c), or from heterotrophic to autotrophic energy sources (Minshall 1978, Fletcher et al. 2000, Broadmeadow and Nisbet 2004). In contrast, Miller and Boulton (2005) found that urbanization increased leaf litter inputs by extending the source area to gutters and ditches of the storm drainage system. Moreover, urban streams impacted by heavy sedimentation bury leaves slowing processing rates, because of the reduced abrasion and fewer numbers of macroinvertebrates (Webster and Benfield 1986, Naiman and Décamps 1997).

In this study, we examined the effects of urbanization on the quantity of leaf litter inputs, the quantity of leaf litter standing crop, and leaf litter retention across 13 streams. We hypothesized that urbanization would decrease canopy cover, resulting in reduced leaf litter inputs and lower autumn leaf litter standing crops in more urbanized streams. We also expected that physical loss of leaf material due to higher storm flows in urban streams would result in overall lower retention of leaf litter in spring. We expected that the resulting reduction of leaf litter standing crops in more urban streams would result in reduced abundances of shredding macroinvertebrates.

Background:

Aquatic systems that receive limited light penetration due to dense canopy cover, turbidity, or depth are primarily supported by allochthonous inputs from the surrounding landscape (Fisher and Likens 1973, Webster and Benfield 1986, Allan 1995, Jones 1997, Pozo et al. 1997). In fact, some high-order river systems are supported by inputs from small headwater streams and tributaries that connect to them (Vannote et al. 1980, Wallace et al. 1995). For example, in a southeastern Alaskan stream, Wipfli (2005) found that fishless headwater streams provide enough resources to support 100-2000 young-of-year salmon per kilometer of downstream reach.

Processing of allochthonous leaf litter involves three steps; 1) leaching, 2) microbial colonization, and 3) fragmentation (Petersen and Cummins 1974, Webster and Benfield 1986). Soluble organics initially released from litter supply microorganisms with nutrients. Microorganisms quickly colonize leaves, which are then fragmented by 1) macroinvertebrates that process the conditioned litter or 2) by abrasion and mechanical breakdown as leaves tumble along the stream (Vannote et al. 1980, Cummins et al. 1989, Perry et al. 1996). Leaf breakdown rates occur along a continuum based on species, but are also influenced by other physical factors. Woody vegetation decomposes slower than non-woody plants due to the percentage of support tissue and essential nutrients within the leaf (Webster and Benfield 1986).

While temperature has been found to increase processing rates, as previously stated, several studies have shown it is not always the key factor controlling breakdown. Short et al. (1980) found rapid breakdown rates for water temperatures at or near $0^{o}C$, while other research indicates no significant relationship with temperature (Short and Ward 1980, Grubbs and Cummins 1994, Rowe et al. 1996, Sponseller and Benfield 2001). Goncalves et al. (2006) suggest that effect of temperature on breakdown rates can be overridden by nutrient content and presence of invertebrate shredders.

Indirect changes within the stream basin influence hydrology of the receiving stream and therefore channel geometry. Velocity is an important function because it affects erosion and sedimentation rates, nutrient and contaminant spiraling, biogeochemical exchanges between the water column and benthos, and flushing of invertebrates and fish larvae (Van Nieuwenhuyse 2005). Within a given physiographic region, assuming comparable basin areas, urban streams experience more frequent and larger peak flows and usually feature reduced channel complexity (Morisawa and Clayton 1985, Booth and Jackson 1997, Pizzuto et al. 2000, Paul and Meyer 2001). Herbert (2004) found that the higher breakdown rates were associated with increased velocity, and that both factors were amplified at sites with greater impervious surface, rather than larger drainage basin areas. Straight channels and those with less in-stream wood reduce retention time, slowing the biological breakdown process and decreasing the overall amount of food available for organisms (Oelbermann and Gordon 2001). Alternatively, the higher velocities can accelerate the physical breakdown.

Methods:

The study was conducted in the Piedmont physiographic region of Georgia, USA, which consists of rolling hills of weathered schist, gneiss, and granite mantled by Ultisols (Griffith et al. 2001). Before 1800, this region was covered by hardwood and pine forests that were converted to row crop agriculture, principally cotton, from approximately 1810 to 1930 (Trimble 1974, Richter and Markewitz 2001). Intense rainfall, erodible soils, and rolling topography made this a poor place for row crop agriculture, and extensive rill and gully erosion resulted (Trimble 1974, Richter and Markewitz 2001). Sediment from this period still stored on floodplains of streams and rivers in the region (Jackson et al. 2005). After 1930, much of the region returned to mixed oak and pine forests (Griffith et al. 2001). Today the area is heavily influenced by the urban sprawl of metropolitan Atlanta.

Single stream reaches from forty-two Piedmont streams were used to investigate relationships between canopy coverage and basin land use and between shredder taxa and basin land use. A subset of 13 streams was used to investigate vertical leaf litter inputs and fall and spring leaf litter standing crops. The larger group of 42 sites was selected from the Georgia Department of Natural Resources (DNR) – Wildlife Resources Division data base of wadeable streams (Fig. 1). Site selection was stratified to ensure sites covered a wide range of water quality conditions, including streams in each drainage basin within the study physiographic province (Savannah, Oconee, Ocmulgee, Chattahoochee, Coosa, and Tallapoosa). Water quality was quantified with Index of Biotic Integrity (IBI) and Index of Well Being (IWB) scores, calculated by Georgia DNR from fish assemblage data. An equal number of streams were chosen from each possible IBI category (excellent, good, fair, poor, and very poor). Selected watersheds ranged in size from 3 to 171 km^2 and encompassed a mixture of land use types (Appendix 1). Stream reaches encompassed 35 times the stream width and usually began three to four channel widths upstream of a bridge crossing and proceeded upstream from that point.

Catchment land use, reach-scale canopy cover, macroinvertebrate shredder abundance, and stream velocity were determined for all 42 study sites. Vertical litter inputs and leaf litter standing crops for winter and spring were originally measured on a subset of 18 streams selected from the larger set of 42 streams. To select streams we sorted canopy cover measurements in ascending order and grouped streams into four classes with ten sites in each group; classes were based on percent coverage (0-59, 60-79, 80-89, and 90-100). Four streams were selected from each class, with two sites being selected at random. Five sites were subsequently removed from the analysis because either beaver activity or construction projects altered stream and riparian conditions during the study period. Thus, the subset consisted of 13 streams with varying amounts of canopy cover.

Geographical information system software (ArcGIS 9.0) was used to quantify land cover and percent impervious surface within each watershed. Watershed boundary information for each site was obtained from Wildlife Resources Division, Georgia Department of Natural Resources. Land use data was derived from United States Geologic Survey 2001 National Land Cover Database (30-m pixels). Land cover was classified as urban (low, medium, and high intensity and developed open space), forest (deciduous, evergreen, and mixed), agriculture (pasture hay and cultivated crops), and other (e.g. open water, grassland, and wetlands). Sites were then determined to be forested, urbanized, or mixed-use based on the total amount of forested land or urbanization within each watershed. Streams with less than 20 percent urbanization and greater than 59 percent forest within their watersheds were judged to be forested sites. Streams with greater than 20 percent urbanization and less than 59 percent forest were deemed urbanized sites. All other sites were selected as mixed-use sites. None of the study sites contained less than five percent urbanization.

Canopy cover was determined during full leaf-out using a spherical densiometer (Lemmon 1956). Canopy cover was measured at three points within the study reach; at each point four measurements were taken (up, down, left, and right). Data were taken in summer 2003 on all but three sites (Long Cane, Lazar, and Raccoon Creek), which were unwadeable; therefore these data were collected in summer 2004. The three measurements were averaged for each site.

Mid-channel velocity was estimated using the tracer dye method over 30-m of the thalweg distance during baseflow conditions (Laenen and Dunnette 1997, Kondolf and Piégay 2003, Gordon 2004). Fluorescent yellow/green dye (Bright Dyes, Kingscote Chemicals) was released at the upstream end of the subreach and transit time was determined at the front edge, mid-point, and trailing edge of the dye plume. Average velocity was determined by dividing the total distance by average transit time for the front edge and trailing edge of the dye plume. The sampled subreach included a minimum of three habitat types (e.g. pool, riffle, glide, run, bend) that were characteristic of overall reach-scale morphology. Characteristic reach morphology was based on prevalence of habitat types within the reach.

Leaf litter traps were placed at three stations along the riparian zone to collect vertical needle and leaf fall. Traps were staked as close to the active channel as possible. Each trap was made from 1-m^3 screen netting mounted to a plastic frame. Litter collections were emptied monthly, beginning in October 2003 until the end of March 2004. Collected litter was placed in paper bags and returned to the laboratory. In the laboratory samples were rinsed over a sieve. Dry mass (DM) was determined by drying the washed litter at 60 ^{o}C for at least 48 hours and to a constant mass prior to weighing.

Leaf litter standing crop was determined by collecting all organic material within the bankfull channel in five 0.5-m transects across the channel during peak leaf fall in December 2003 and again in late winter during March 2004 (Cummins et al. 1989, Herbert 2003). All transects were within the 100-m macroinvertebrate sampling reach. Litter material, collected from the entire width of the stream, was placed in polypropylene sand bags and returned to the lab. Each sample was washed to remove inorganic debris, woody branches were also removed, and then samples were dried at 60 ^{o}C for at least 48 hours to a constant mass prior to weighing to determine DM. All data were combined to determine total leaf litter availability and standing crop for each site. Differences in weight and percent change in standing crop were calculated based on area (i.e. weight of organic matter divided by area of stream sampled within each transect) and per stream length (i.e. weight divided by length of stream sampled). Percent change in standing crop was based on the difference in December inputs and material retained in March, where exports are assumed to be all material not retained within the transect.

Following Georgia Standard Operating Procedures (SOP), established by the Environmental Protection Division, we collected macroinvertebrates from October 2002 to February 2003 and October 2003 to February 2004 within a 100 m sub-sampling reach at each site. Sampling between mid-September and February allows for collection of late instars, which are representative of the balanced indigenous communities inhabiting all accessible habitats (Plafkin et al. 1989, Barbour et al. 1999). During peak emergence (e.g., spring and fall) macroinvertebrates are too small to collect and sufficiently characterize the community (Cummins and Klug 1979).

All available productive habitat types were sampled, with preference given to the most productive habitats (Plafkin et al. 1989, Georgia Environmental Protection Division 2004a). In high gradient streams riffles (6 jabs), leaf packs (3 handfuls), sand (3 jabs), woody debris (5 jabs), and undercut banks (3 jabs) were sampled. In low gradient systems the riffle samples were reallocated to the remaining habitat types (leaf packs – 3 handfuls, sand – 3 jabs, woody debris – 8 jabs, undercut banks – 6 jabs). A total of 20 jabs were collected from each reach, with a single jab consisting of an upward/forward thrust of a D-frame dip net for a distance of 1-m. All samples were compiled prior to subsampling. Samples were fixed in 100% ethanol, which was diluted to a 90% ethanol solution for preservation (Georgia Environmental Protection Division 2004a).

In the laboratory, samples were washed through a 250 micron (# 60) sieve to remove sand, silt, and clay. Any large organic material (whole leaves, twigs, algal or macrophyte mats) not removed in the field were rinsed, visually inspected for macroinvertebrates, and discarded. Washed samples were uniformly distributed in a Caton Macroinvertebrate Subsampler (Caton 1991). Squares were selected from the grid using a random number table; all material (organisms and debris) was removed from the square and placed in a sorting tray. Organisms were separated from the organic matter and stored in a 70% ethanol solution. Squares were removed in this manner until a 200 (± 20%) organism subsample was obtained. Each grid selected for sorting was sorted in its entirety, regardless of number of organisms obtained (Georgia Environmental Protection Division 2004).

We identified organisms to genus level when possible, using taxonomic keys in the following sources: Merritt and Cummins (1996), Brigham et al. (1982), Wiggins (1996), Thorp and Covich (2001) and Epler (1996). Tolerance values and functional feeding groups were assigned according to designations by the Georgia Environmental Protection Division (2004). Shredder abundance and proportion of shredders were determined from total counts.

The Pearson Product Moment correlation coefficient determined the degree of association between variables. The effect of land use on canopy cover and shredder variables were measured using a Kruskal-Wallis one-way ANOVA followed by Dunn's *post hoc* test to isolate differences between urban, forested and mixed land use classes. The leaf litter data sub-set were then analyzed using regression analysis. Proportional data sets (e.g. % canopy cover, % shredders) were arcsin transformed and tests for assumption of normality were conducted prior to analysis. All data analyzed using regression analysis were normally distributed. The two seasons of macroinvertebrate data were averaged for each site because there were no detectable differences among years. SigmaStat® and SigmaPlot® software (Systat Software Inc., San Jose California) were used to conduct and plot separate simple linear regressions to determine correlative relationships.

Results:

Mean canopy cover varied between 0 and 95 percent for the entire data set and leaf litter subset. Regardless of adjacent land use, a minimal riparian buffer remained intact for most sites. Two streams had no canopy cover, all but four streams had canopy cover exceeding 50 percent and twenty-six of 42 sites had canopy cover > 70 percent (Table 2.1). Good riparian conditions were common even in heavily urbanized streams, and poor riparian conditions could be found along some streams in forested watersheds due to recreational activities (i.e., food plots, all-terrain vehicle trails, railroad) (Fig 2.2). After classifying the streams into forested, mixed, and urban groups, ANOVA analysis indicated mean percent canopy cover for urbanized sites was significantly less ($\alpha = 0.05$), decreasing by 21 percent, than that of forested sites and ten percent less than sites with mixed land use (Table 2.2). As urbanization increased the range in canopy cover also increased, suggesting that urbanization increases the probability of canopy clearing and the potential for impacts (Fig 2.2). Overall, the data indicate that canopy cover was more closely tied to local landowner decisions rather than watershed-scale land use coverage.

Baseflow mid-channel velocity measurements ranged between 0.17 m s^{-1} and 0.72 m s^{-1} (Table 2.1). Baseflow velocities were not significantly correlated to December leaf litter standing crop ($R^2 = 0.192$, $p = 0.652$). A weak relationship existed between velocity and March standing crop ($R^2 = 0.391$, $p = 0.022$), but did not explain the percent change leaf litter standing crop ($R^2 = 0.0841$, $p = 0.977$). The correlation between urbanization and velocity was weak and not significant (Table 2.3).

Average vertical leaf litter inputs, ranging between 0.094 g DM $m^{2\ -1}$ and 62.65 g DM m^{-2}, were highly and positively correlated to canopy cover ($R^2 = 0.835$, p = < 0.001) (Fig. 2.3), but this relationship was strongly influenced by two extreme points with zero canopy cover. When these two extreme points were removed from the data set, the relationship was still significant but not as highly correlated ($R^2 = 0.403$, $p = 0.036$). Vertical leaf litter inputs varied as expected with local riparian conditions. Correlations between vertical leaf litter inputs and the abundance, relative composition, and taxa numbers of shredders were weak (Table 2.3).

Mean leaf litter standing crop was highly variable for December and March (Fig 2.4). Standing crop in December, unlike March, was significantly correlated to basin forest cover ($R^2 = 0.491$, $p = 0.008$, $R^2 = 0.280$, $p = 0.063$, respectively), but neither December nor March standing crops were related to local canopy cover or leaf litter input. Streams with less urbanization had greater inputs in December. December and March standing crops were negatively correlated with percent urbanization and tended to be low in urban streams (Table 2.3). However, the forested sites had higher net exports of leaf litter over the winter than urbanized sites, which actually retained or gained leaf litter (Fig 2.5). Less leaf litter was available in March for all sites, with the exception of Britten's Creek, with less than 34 percent urbanization. In the more forested streams, winter flushing of fall litter inputs reduced March leaf litter standing crops as expected; yet these sites still had more leaf litter, overall, than highly urbanized streams that retained or gained leaf litter, which was contrary to expectations.

Macroinvertebrate shredders were among the least numerous functional feeding group present in all streams (Table 2.1), with collector-gatherers of the family Chironomidae being the most abundant. Shredders comprised only five percent of the total number of macroinvertebrates. Shredder composition and abundance did not vary with land use, but the number of shredder taxa was greater in forested sites than in urban or mixed sites (Table 2.2). For most sites the number of shredder taxa was low with a median of 6 per site, overall. The reduced number of shredder taxa in urban streams relative to forested streams was expected, but the apparent uniformity of shredder composition and abundance was not.

Discussion:

Our first objective was to determine if canopy cover was lower in urbanized streams than in rural systems, resulting in reduced leaf litter inputs. Reduced canopy cover results in predictable decreases in vertical litter fall. However, local reach-scale deforestation did not directly impact leaf litter standing crop availability or shredding macroinvertebrates as we hypothesized. Rather, leaf litter standing crops were driven by complicated basin-scale factors.

As expected, autumn leaf litter standing crops generally increased with forest cover and decreased with urban cover in the basins. We found low autumn leaf litter standing crops in all but one highly urbanized stream. Our results indicated that autumn leaf litter standing crops are most closely tied to basinwide forest cover which we assume correlate with basin-wide riparian forest condition.

During winter the leaves are processed and/or exported downstream. In forested systems litter inputs occur when leaves fall in autumn, and once processed leaves are exported the streams do not receive further inputs of organic matter. Previous studies of small streams, large rivers, and wetlands have shown that direct vertical inputs dominate leaf litter standing crop with negligible lateral inputs that usually do not exceed ten percent of the total litter input (Naiman and Decamps 1997, Pozo et al. 1997, Paul 1999, Batzer and Palik 2007). Brittens Creek, the one forested stream with higher standing crop in March, received direct inputs from a pond at the upstream section of our study reach. Urban streams, on the other hand, receive less litter than forested systems, but can receive continual year-round inputs from lawns, garden maintenance, and storm drainage systems (Fig 2.6). Unlike the forested streams, the highly urbanized streams retained or gained leaf standing crops over the winter, indicating that processed or exported leaves were being replaced by new sources. These findings are similar to work by Australian researchers who recorded significantly higher amounts of coarse particulate organic matter in urban streams than in rural reference systems. They attributed their results to increased inputs from planted trees, often exotic species, lining the streets that were connected to storm drainage pipes (Miller and Boulton 2005).

Although not recorded, we also observed variability in riparian vegetation species patterns between urban and forested sites. Urban sites were dominated by kudzu, *Pueraria sp,* and privet, *Ligustrum sp,* with a few oak species, while forested sites were comprised mainly of mixed-oak and pine species. Current literature does not report leaf breakdown rates for kudzu, therefore we do not know if this difference is a key determinant affecting leaf litter loss. We found that kudzu leaves and vines remained attached to the streambank for most of the dormant season. Also, unlike oak-hickory forests, Chinese privet (*Ligustrum sinense*), a common urban species, can provide litter year round.

Other factors that play a major role in the determination of leaf breakdown rates in disturbed systems are velocity and substratum (Reice 1980, Allan 1995, Herbert 2003, Paul et al. 2006). We found that March standing crop was related to velocity. Increased flows associated with urban streams have the potential to flush organic matter downstream reducing retention rates. However, continuous year-round litter inputs may counter the flushing effects of increased flows associated with urbanization. Greater retention in our urban streams may also be attributed to sediment deposition slowing processing rates and export. Most of the leaf litter at urban sites was either partially or mostly covered by sand. Because sand can slow processing rates and reduce fragmentation, more litter may be available until swifter velocities uncover litter and carry it downstream (Reice 1974, Webster and Benfield 1986, Wallace et al. 1995).

Our final objective involved determining if shredding macroinvertebrates differed in urban streams due to reduced leaf litter availability. The insignificant differences in abundance and proportion of shredding macroinvertebrate feeding groups between sites are possibly related to sample collection methodology. In the majority of macroinvertebrate leaf-litter studies, macroinvertebrates are sampled from leaf bags or baskets that are monitored for breakdown. Our samples were collected from all available habitats and, although leaf packs were collected, they were not the primary focus of sampling.

The link between macroinvertebrate communities and available resources is complicated by: 1) human activities that increase sedimentation, decreasing stable macroinvertebrate habitat (Webster and Benfield 1986, Sponseller and Benfield 2001); 2) planting of non-native trees, which may be less palatable to macroinvertebrates; 3) changes in water chemistry and quality of allochthonous inputs that may reduce the distribution of shredder species (Paul and Meyer 1996, Sponseller and Benfield 2001); and 4) stronger relationships between predator-prey interactions than availability of food resources (Yamamuro and Lamberti 2007). The apparent importance of horizontal leaf litter inputs to urban stream detritus budgets suggested by our data and that of Miller and Boulton (2005) indicates that further studies to characterize and quantify litter inputs from storm drains are warranted.

Conclusion:

Canopy cover and leaf litter inputs/retention are linked to land use activities, but at different scales. Canopy cover is influenced by riparian management at the reach scale, whereas leaf litter standing crop appears to be controlled by catchment conditions and factors indirectly tied to land use, such as timing of inputs and alterations of leaf litter source areas. Overall, we found that sites with less urbanization have greater litter inputs and thus a larger standing crop during December, but also have greater loss of standing crop by March. In fact, highly urbanized streams retained or gained leaf litter over the winter. These urban systems are apparently receiving continuous inputs from lawns and gardens contributing litter to the storm drain system, and these continued inputs replace leaf litter processed and flushed over the winter. Continuous inputs coupled with losses that are deposited in the wetted perimeter during higher flows, which can become re-entrained in the aquatic food web, may serve as a steady source of litter material in more urbanized watersheds. The number of macroinvertebrate taxa and intolerant species were negatively affected by watershed land use, although the proportion and abundance of shredders were not. In fact, shredder proportion and abundance was low in all streams, possibly reflecting the ubiquitousness of unstable channels due to the effects of past land uses. Lack of high shredder proportions may not be a significant factor in rates of litter processing or amount of litter standing crop loss in the Georgia Piedmont. Clearly, landcover patterns and human activities affect leaf litter resources in Georgia Piedmont streams, but the controls are more complicated than previously assumed.

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Table 2.1. Physical characteristics of 42 study streams within the Piedmont ecoregion of Georgia from 2001 USGS National Land Cover Database. Sites 1 - 13 are the subset of streams used for the leaf litter study.

Site No.	Site	Characterization of Land use (%) Urban	Forest	Agriculture	Other	Average Canopy cover (%)	Average Shredder Abundance (Ct)	Average Shredders (%)	Average Shredder Taxa (Ct)	Velocity (m s^{-1})
1	Copeland Creek	3.4	77.7	10.6	8.3	95	11.3	4.4	3.0	0.58
2	Indian Creek	5.1	65.1	13.5	16.3	92	3.2	1.3	1.7	0.30
3	Whooping Creek	5.7	62.5	22.4	9.4	87	10.0	4.7	5.5	0.20
4	Snake Creek	11.4	60.8	19.1	8.7	88	16	5.2	4.5	0.22
5	Brittens Creek	3.3	51.9	22.3	22.5	77	4.5	2.0	3.5	0.19
6	Cabin Creek	37.7	46.2	10.2	5.9	91	2.5	1.2	1.0	0.33
7	Tobesofkee Creek	34.4	43.6	14.8	7.2	76	17.5	8.1	2.0	0.17
8	Yellow Water Creek	12.1	39.6	35.8	12.6	87	6.5	3.0	4.5	0.31
9	Kendall Creek	4.0	73.9	9.2	12.9	67	3.5	1.5	2.0	0.39
10	Lightwood Log	7.5	38.4	41.2	12.8	94	15.0	6.6	1.5	0.49
11	Noonday Creek	61.4	25.9	5.2	7.5	0	3.5	1.7	2.5	0.40
12	Settingdown Creek	19.1	46.8	28.0	6.1	58	12	9.1	5.0	0.49
13	Noonday Creek	70.9	24.1	2.0	3.0	0	1.0	2.0	1.5	0.72
14	Zoie Brown Creek	5.2	73.8	4.7	16.2	95	44.5	22.1	2.5	0.08
15	Sandy Run	3.9	76.6	0.0	18.9	94	7.8	3.2	2.2	0.23
16	Crooked Creek	2.9	32.6	0.6	18.1	92	11.5	6.1	1.5	0.21
17	Little Buck Creek	6.4	50.9	31.0	11.7	93	26.0	12.8	4.0	0.03
18	Buck Creek	7.7	53.6	25.8	12.9	88	37.0	17.9	7.0	0.11
19	Wahoo Creek	20.5	54.1	15.2	10.2	69	26.0	9.6	2.0	0.20
20	Little Chehaw	6.0	68.9	14.1	10.9	93	2.5	1.1	1.5	0.20
21	Walnut Creek	7.2	57.2	16.8	18.8	92	23.0	15.9	3.5	0.51
22	Heads Creek	18.8	45.4	22.9	12.8	86	48.0	23.1	3.0	0.17
23	Gum Creek	4.1	61.3	29.3	5.3	94	13.5	5.8	3.5	0.22
24	Flat Creek	4.1	62.1	16.8	17.0	94	14.5	6.8	4.0	0.47
25	Long Cane Creek	6.1	61.2	15.2	17.5	80	15.5	6.6	5.0	0.17
26	Hillabahatchee Creek	2.2	74.1	8.6	15.2	68	14.5	6.1	2.0	0.41
27	Auchumpkee Creek	2.3	71.5	11.4	14.8	65	17.0	6.4	3.5	0.15
28	Lazar Creek	5.4	69.7	11.2	13.7	65	7.5	3.1	4.0	0.52
29	Potato Creek	7.7	51.2	28.0	13.1	88	19.0	8.3	5.5	0.42

Table 2.1 Continued

30	Rooty Creek	0.0	72.0	9.0	19.0	94	10.5	4.8	2.0	0.35
31	Mountain Creek	23.7	58.9	9.5	7.8	61	15	8.4	4.0	0.15
32	Red Oak Creek	4.2	68.7	12.1	15.0	92	11.5	4.5	3.5	0.14
33	Biger Creek	14.0	47.2	28.3	10.5	85	3.3	1.5	0.8	0.45
34	Bull Creek	19.3	59.9	9.0	11.6	53	1.0	0.5	0.5	0.06
35	Bear Creek	5.5	42.6	36.4	15.5	85	4.0	1.4	1.5	0.49
36	Butler Creek	59.8	33.8	2.7	3.8	91	0.5	1.1	1.0	0.24
37	Allatoona Creek	34.6	53.9	6.9	4.6	75	2.5	1.6	2.0	0.21
38	Etowah River	2.7	94.4	1.6	1.3	82	7.5	3.7	3.5	0.75
39	Shoal Creek	5.9	75.3	9.1	9.7	53	32.5	10.6	6.0	0.27
40	Beach Creek	16.7	58.2	21.7	3.4	86	12.0	4.7	4.0	0.27
41	Raccoon Creek	2.7	83.1	3.0	11.1	69	9.5	4.1	5.0	0.38
42	Little Tallapoosa River	16.5	48.3	21.2	14.0	89	2.5	1.1	1.0	0.44

Table 2.2. Canopy cover, shredder abundance, proportion of shredders and shredder taxa by land use class.

Land use	Canopy Cover (%)[1]	Shredder Abundance (No. per sample)	Proportion of Shredders (%)	Shredder Taxa (No. per sample)
Forest	86.16 (±1.59)a	14.65 (±1.47)a	6.97 (±0.77)a	3.54 (±3.54)a
Mixed	75.62 (±2.71)b	11.45 (±1.19)a	5.02 (±0.46)a	2.99 (±2.98)b
Urban	65.49 (±7.09)b	12.00 (±2.96)a	5.64 (±1.34)a	2.06 (±0.21)b

[1]Values are means (±SE) of means for each stream within a land-use class.
[2]Within a column, means values with same letter are not significantly different (Kruskal-Wallis with Dunn's post *hoc test*; p > 0.05).

Table 2.3. Pearson product moment correlation among available litter, standing crop, watershed land use and velocity for 13 streams within the Piedmont ecoregion of Georgia. Relationships between variables with p-values less than 0.050 are significant and are in bold; pairs with positive coefficients tend to increase together and for those with negative coefficients, one variable tends to increase while the other decreases.

	Leaf Litter (gDMm^{-1})	December Standing Crop (g m^{-2})	March Standing Crop (g m^{-2})	Difference in Standing Crop (g m^{-2})	Percent Change (g m^{-2})	December Standing Crop (m^{-1})	March Standing Crop (m^{-1})	Difference in Standing Crop (m^{-1})	Percent change (m^{-1})	Urban (%)	Forest (%)	Agriculture (%)	Canopy Cover (%)	Shredder Abundance	Shredder (%)	Shredder Taxa	Velocity (m s^{-1})
Leaf Litter (g DM m^{-1})	1																
December Standing Crop (g m^{-2})	0.423	1															
March Standing Crop (g m^{-2})	0.199	0.616	1														
Difference in Standing Crop (g m^{-2})	-0.388	**-0.809**	-0.036	1													
Percent Change (g m^{-2})	**-0.696**	-0.496	-0.171	0.500	1												
December Standing Crop (m^{-1})	0.420	**0.729**	0.307	**-0.696**	-0.52	1											
March Standing Crop (m^{-1})	0.160	0.526	**0.786**	-0.081	-0.129	**0.632**	1										
Difference in Standing Crop (m^{-1})	-0.435	**-0.605**	0.087	**0.832**	**0.584**	**-0.883**	-0.195	1									
Percent change (m^{-1})	-0.548	-0.476	0.033	0.629	**0.808**	**-0.636**	-0.047	**0.777**	1								
Urban (%)	**-0.795**	-0.530	-0.272	0.468	**0.719**	**-0.574**	-0.237	0.583	0.594	1							
Forest (%)	0.495	0.701	0.248	**-0.704**	**-0.627**	**0.721**	0.273	**-0.747**	-0.552	**-0.781**	1						
Agriculture (%)	**0.676**	-0.031	0.045	0.073	-0.428	0.049	0.009	-0.057	-0.345	-0.553	-0.051	1					
Canopy Cover (%)	**0.904**	0.535	0.378	-0.396	**-0.690**	0.522	0.338	-0.456	-0.407	**-0.838**	**0.650**	0.525	1				
Shredder Abundance	0.532	0.494	0.460	-0.283	-0.422	0.417	0.519	-0.213	-0.385	-0.339	0.161	0.496	0.434	1			
Shredder (%)	0.340	0.213	0.252	-0.082	-0.385	0.167	0.285	-0.038	-0.330	-0.171	-0.027	0.501	0.210	**0.880**	1		
Shredder Taxa	0.099	0.083	0.015	-0.094	-0.248	0.330	0.254	-0.264	-0.287	-0.388	0.212	0.428	0.171	0.345	0.388	1	
Velocity (m s^{-1})	-0.289	-0.138	**-0.626**	-0.291	-0.009	-0.248	-0.690	-0.103	-0.179	0.390	-0.242	-0.21	-0.472	**-0.247**	-0.039	-0.307	1

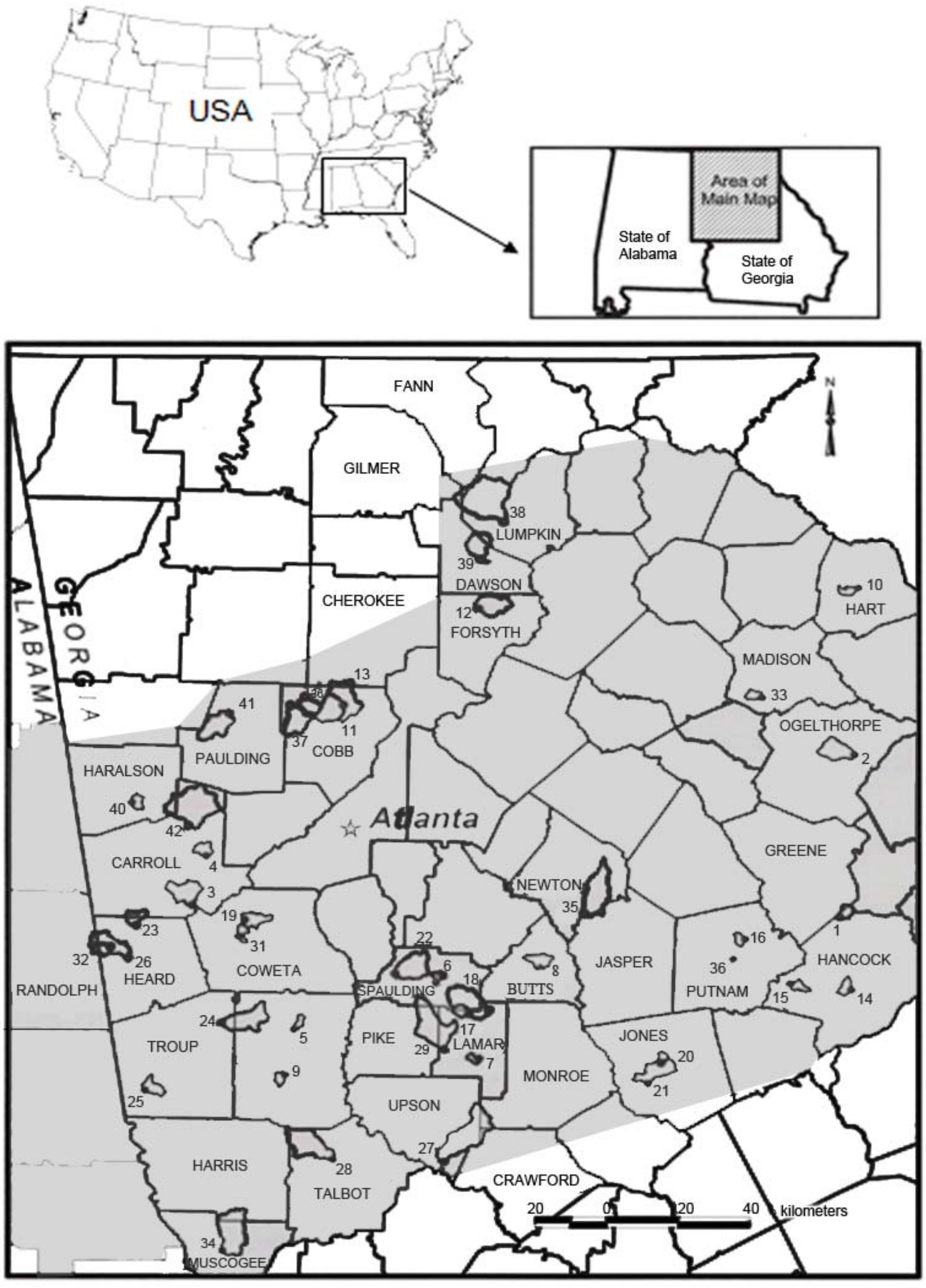
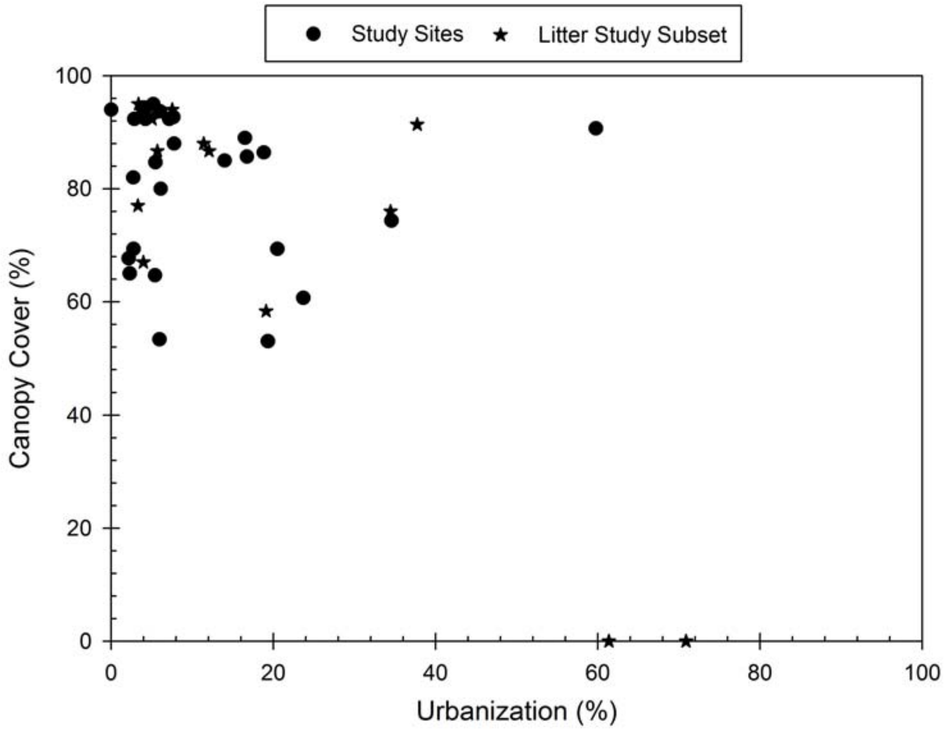


Figure 2.1. Locations of watersheds (outlined in black) and study sites (represented by black dots) within the Piedmont ecoregion of Georgia. The Piedmont ecoregion is indicated by the shaded area.

(A) Urbanization



(B) Forested Land use

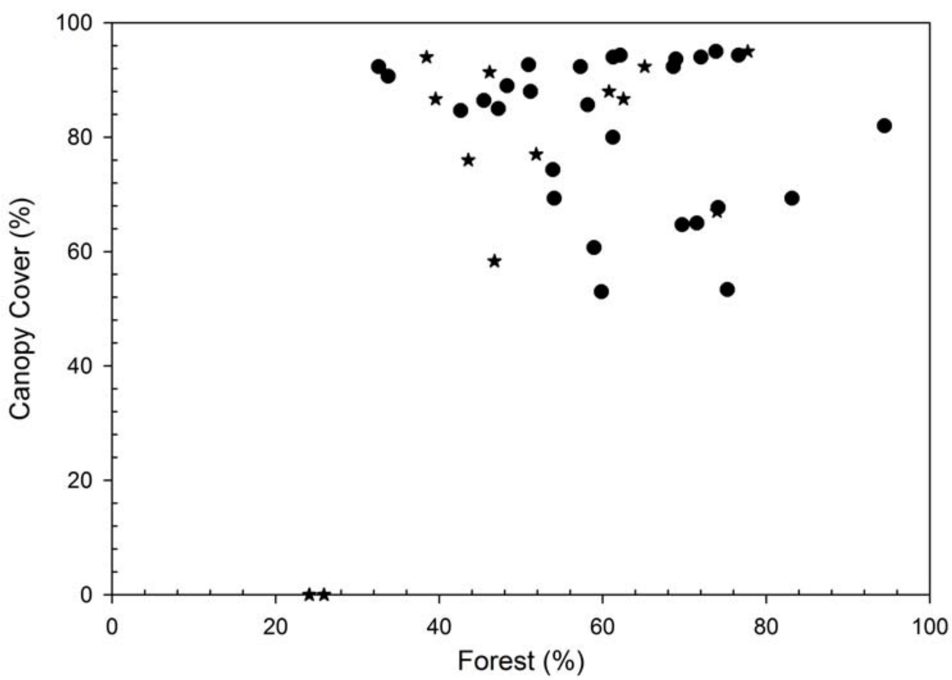


Figure 2.2. Relationship between mean percent local canopy cover and percent a) urbanization and b) forested land use for 42 study sites within the Piedmont ecoregion of Georgia.

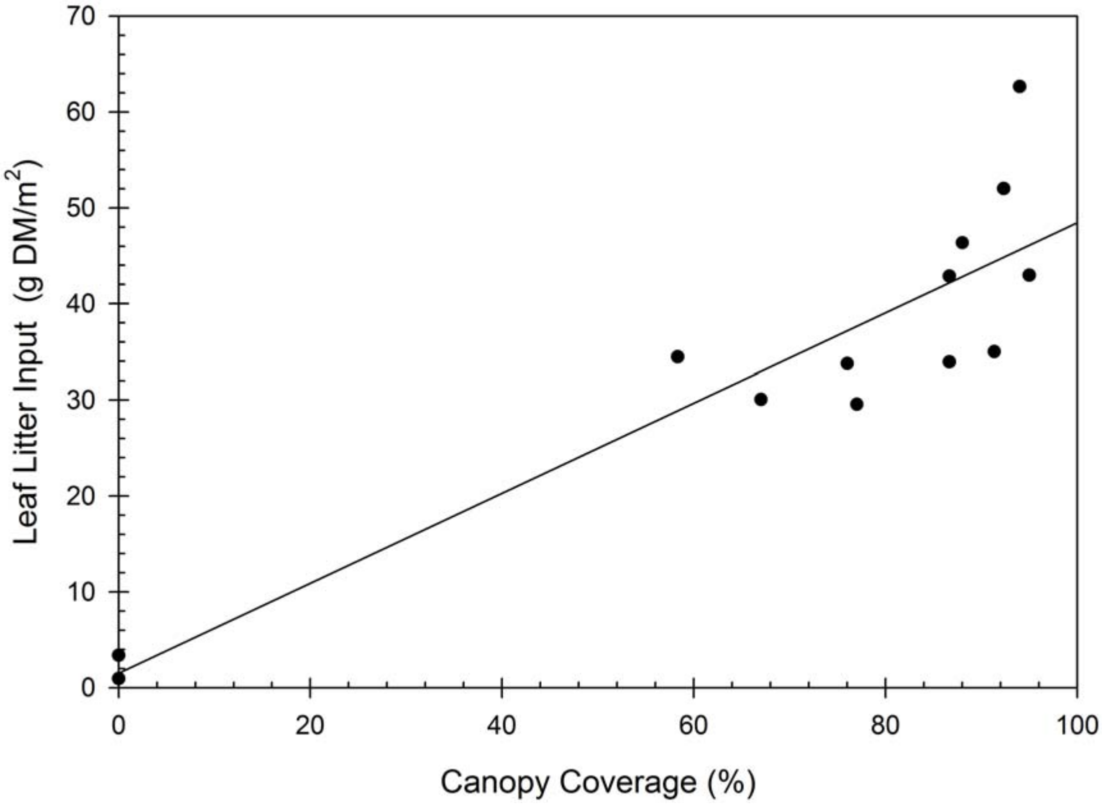


Figure 2.3. Relationship between mean percent canopy cover and urbanization and b) mean leaf litter quantity collected in litter traps for 13 study sites within the Piedmont ecoregion of Georgia. Line indicates best fit for regression.

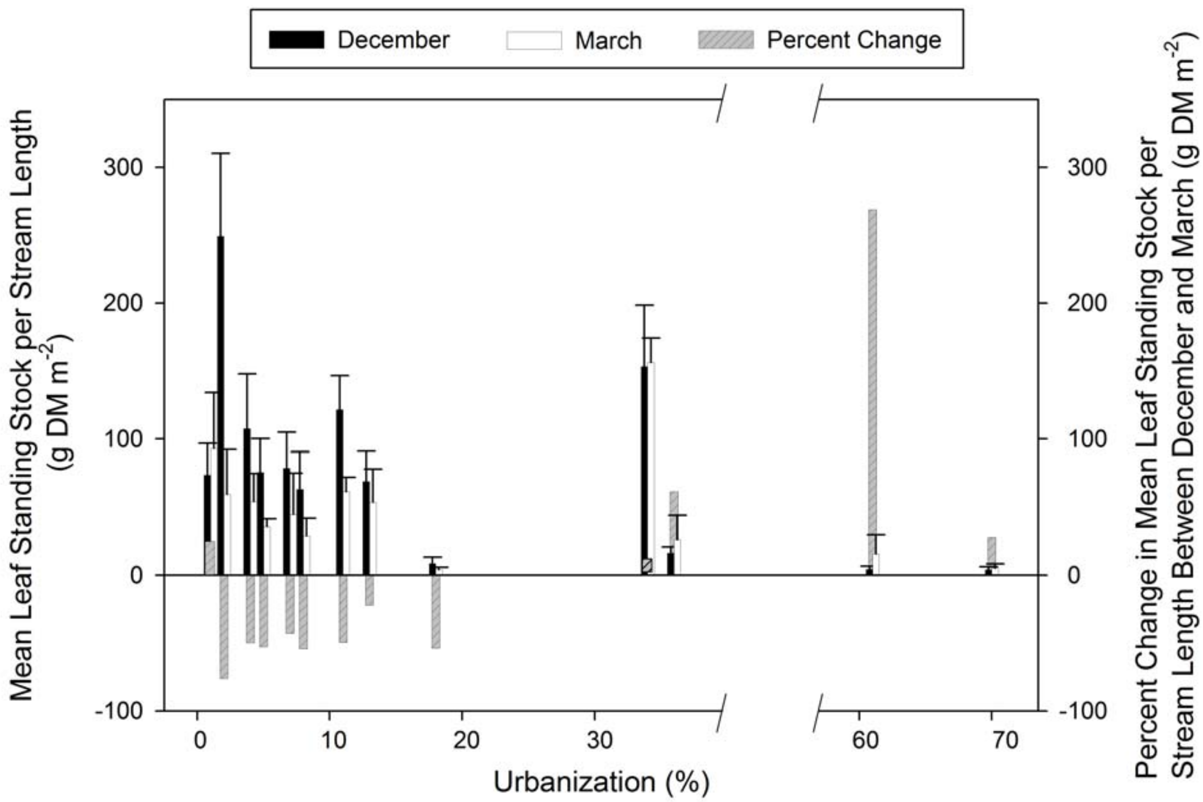
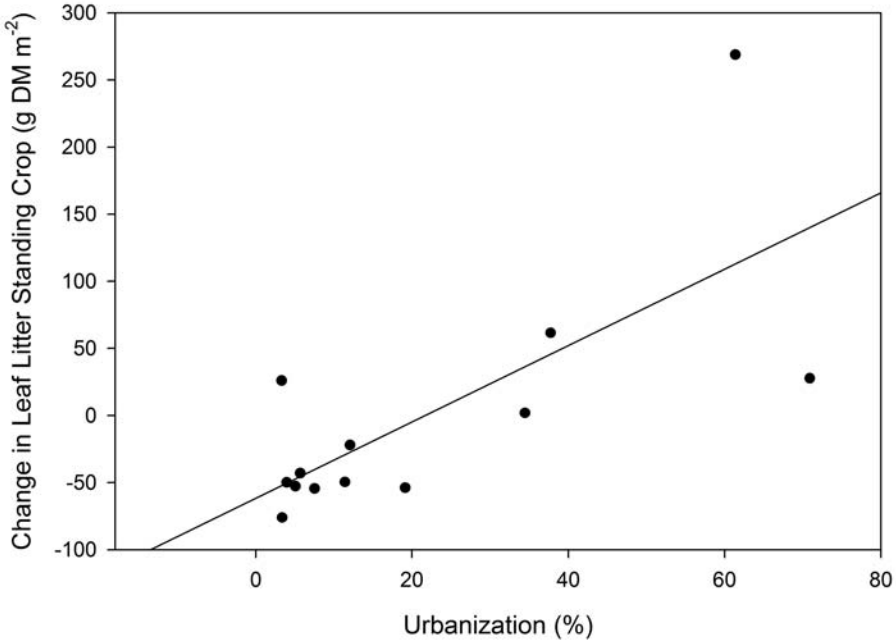


Figure 2.4. Mean leaf standing crop g DM m^{-2} in December 2005, March 2006, and the difference between the two months (percent change). Percent change was calculated as December standing crop minus March standing crop divided by December standing crop, where standing crop was calculated as dry mass per m^2.

(A) Urbanization



(B) Forested Land use

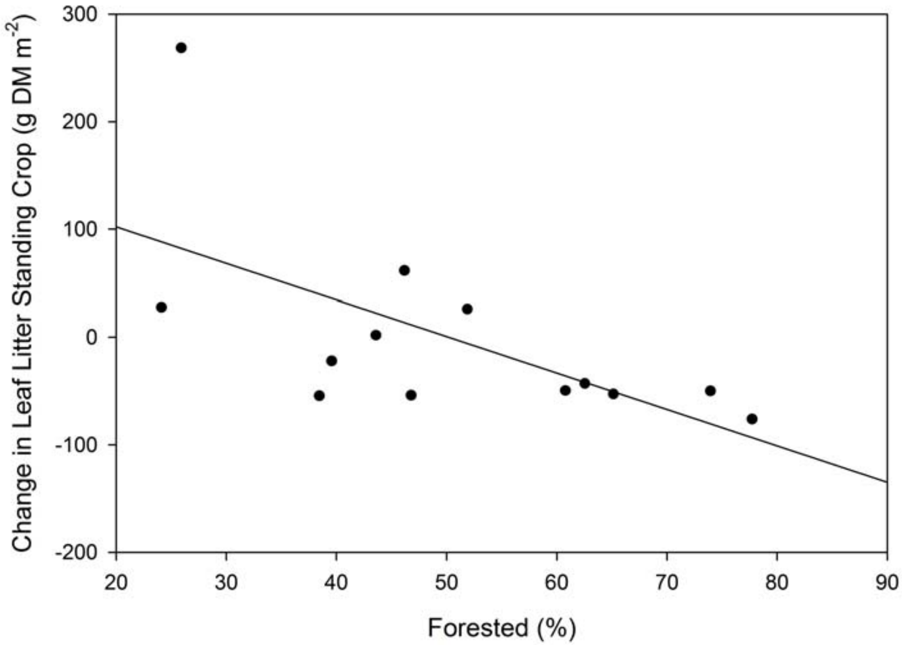


Figure 2.5. a) Percent change in mean leaf standing crop and a) percent urbanization and b) percent forested land use. Percent change was calculated as December standing crop minus March standing crop divided by December standing crop, where standing crop was calculated as dry mass per m^2

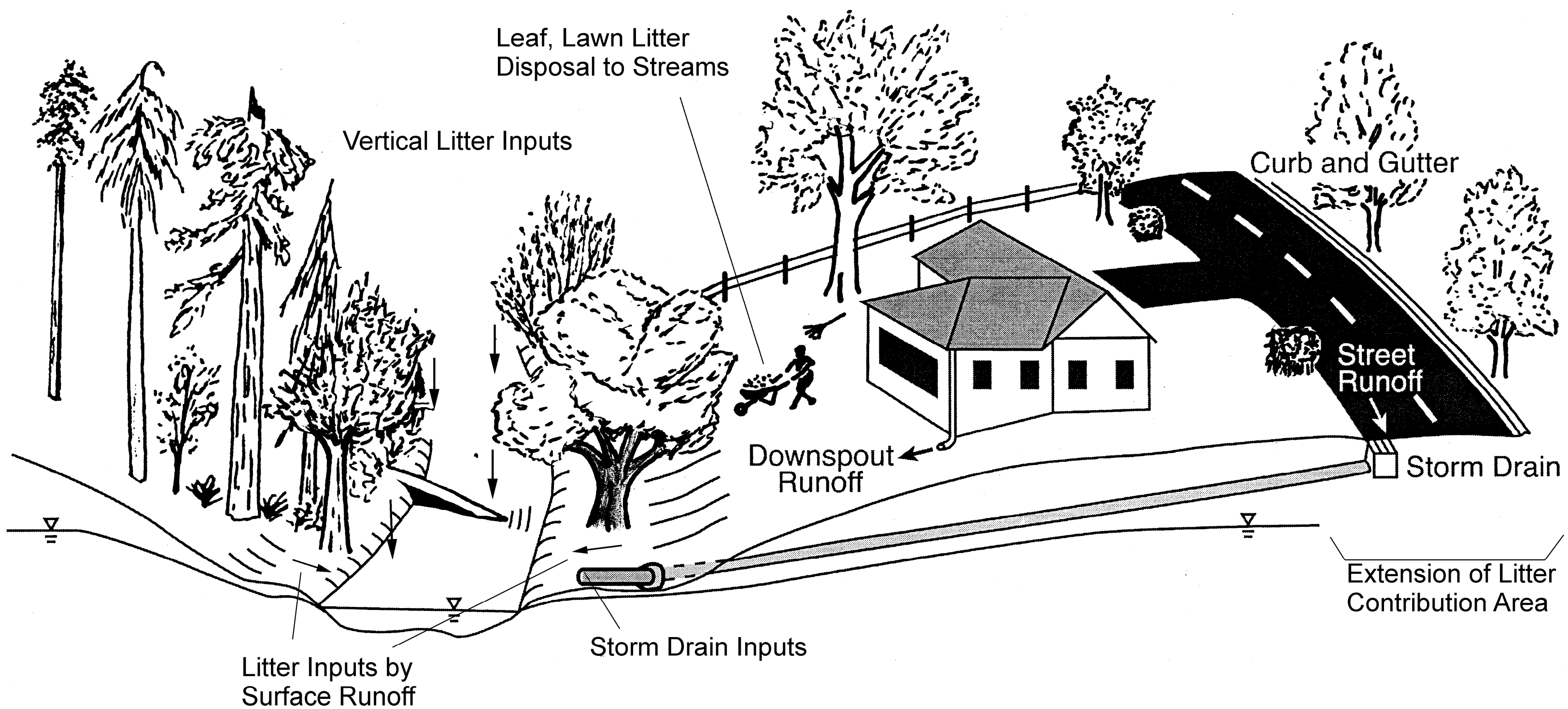


Figure 2.6. Schematic illustrating extended area of litter contribution; input of leaf and lawn litter is dependent upon occurrence of lawncare.

CHAPTER 3

Use of simple suspended and bed sediment metrics to explain biotic index variability and relationships between fish and macroinvertebrate assemblage metrics in Georgia Piedmont Streams

Abstract:

Stream geomorphology changes have accompanied shifting land use patterns of Georgia Piedmont watersheds during the past century. The resulting stream sedimentation and erosion, remnants of earlier agricultural land use and current urbanization, can have deleterious effects on resident biota. Acquiring inexpensive methods of measuring these impacts and having a better understanding of sedimentation-biota interactions can provide more accurate estimates of stream health, allowing for better engineering and management decisions. We sampled 42 streams to determine the relationship between 1) baseflow turbidity and biotic conditions, 2) correlations between the percentage of fine bed substrate and biotic condition, and 3) linkages between fish and macroinvertebrate biotic indices that were developed for the state of Georgia. Our results indicate that easily measured sediment metrics did not correlate strongly to watershed land use but did help explain biological conditions but should be included in descriptive models; conductivity and unit stream power are also important predictors of biotic health. There are enough dissimilarities in the fish and macroinvertebrate responsiveness to landscape factors to suggest that both fish and macroinvertebrate

populations should be sampled. Specific conductivity was an important parameter for both biotic assemblages, but fish indices were also highly correlated to and stream power, whereas EPT taxa and number of taxa were most highly correlated to water clarity and suspended sediment. In the absence of fish data, calculating EPT taxa is important as it can offer insight into the condition of native fish species, benthic invertivore species, and fish catch per unit effort.

Introduction:

Biological assessments of aquatic systems are used to provide indications of sedimentation, changing land use patterns, incursion of contaminants, and hydrologic alterations. These assessments use resident biota, usually fish and macroinvertebrates, to detect ecological degradation in surface waters throughout the United States (U.S.) (Georgia Environmental Protection Division 2005, Hawkins 2006, Georgia Environmental Protection Division 2007). Biological indicators are particularly valuable for predicting degradation related to multiple stressors and cumulative impacts, which is most commonly the case (Beever 2006, Van Sickle et al. 2006). Studies have traditionally examined either macroinvertebrates or fish in relationship to a stressor (i.e, landuse, sedimentation), some have investigated both, but only a few have compared how biotic valuation of water quality differs based on the assemblage used in the assessment (Flinders et al. 2008).

Two or more community assemblages are frequently used to assess the health of a water resource. Although the relationship between biotic integrity (i.e., fish and macroinvertebrate) and stressors varies with respect to the assemblage being surveyed, the U.S. Environmental Protection Agency recommends sampling multiple assemblages to "provide a more complete assessment of biological condition" (Barbour et al. 1999). The harmful effects of anthropogenic sediments on trout were first identified in waters of the United States during the late 1800s by ichthyologist, David Starr Jordan, who recognized that habitat loss was related to sedimentation from mining operations (Waters 1995). Sediment deposition in streams can also clog macroinvertebrate gills, kill aquatic plants and reduce primary production by elevating turbidity levels, blocking sunlight, reducing visibility. Together, sedimentation and turbidity can cause shifts in community structure by decreasing abundance, reducing sensitive species, and escalating numbers of tolerant species (Barrett et al. 1992, Waters 1995, Vuori and Joensuu 1996, Whol and Carline 1996, Boyle 1997, Wood and Armitage 1997, Wulff et al. 1997, Angradi 1999).

The USDA cites soil erosion and its associated impacts as the greatest threat to the sustainability of our water resources (Huang et al. 2006). The southeastern United States does not differ from the nation as a whole in regard to water degradation related to impacts of sedimentation. Sixteen percent of streams in Georgia are listed for sediment or biota problems, and sediment is typically the assumed cause of biotic impairment (Georgia Environmental Protection Division 2004b, 2006).

Relating biotic stress to sediment conditions, however, is hampered by the difficulty of accurately measuring either loads or average or median concentrations of sediment. Sediment moves and manifests itself in two biologically significant ways: as suspended load moving in the water column and as bedload which slides, rolls, and bounces along the streambed. In the Piedmont, suspended load is primarily colloidal clays with some silt and fine sands, whereas bedload is predominantly sand. Either form is a mechanistically plausible source of biotic stress (Doeg and Milledge 1991, Davies and Nelson 1993, Schofield et al. 2004). Both suspended and bedload sediment concentrations increase with increasing flows, and vary horizontally and vertically across a channel cross section. Therefore, accurate determinations of sediment loads or the distribution of concentrations requires long-term spatially and temporally intensive sampling; typically well beyond the scope of water quality assessments.

The difficulty of accurately measuring pollutant concentrations and loads has been a primary motivator for biotic monitoring, as stream organisms integrate water quality over time (Kerans and Karr 1994, Barbour et al. 1999, Cereghino et al. 2003). The problem interpreting biological data without accompanying detailed physical and chemical data is that many water quality factors can alter biological communities in similar ways (Paul and Meyer 2001, Roy et al. 2003, Roy et al. 2005). With biological data alone, it is difficult to infer the source of impairment.

In contrast to estimating sediment loads or central tendencies of sediment concentrations, average baseflow concentrations and bed particle size distributions are easy to measure. Developing relationships between biotic conditions, baseflow turbidity, and percent fines in the bed may provide easy and inexpensive methods upon which to base water quality assessments and sediment Total Maximum Daily Loads (TMDLs). This project explores whether these easily measured sediment metrics explain significant variability in fish and macroinvertebrate metrics in the Georgia Piedmont. Specifically, this work evaluates the relationship between 1) baseflow turbidity and biotic conditions, and 2) correlations between the percentage of fines in bed substrate and biotic condition. Our analysis also investigates linkages between fish and macroinvertebrate biotic indices that were developed for the state of Georgia. The analysis is confined to the Piedmont physiographic region where channels are highly disturbed from past and present activities and sediment is considered a major driver of biotic conditions. Stream sampling and data analysis were designed to address the following hypotheses:

1. Turbidity, baseflow suspended sediment concentration (SSC) and percent fine sediments will increase with increasing urbanization.
2. Fish IBIs will strongly and negatively correlate with basin urbanization.
3. Fish and invertebrate indices (i.e. fish Index of Biotic Integrity (IBI) scores and index of well being (IWB) scores, and number of invertebrate taxa, and number of EPT taxa) will decrease with increasing baseflow SSC.
4. Fish and invertebrate indices will decrease with increasing percent fine sediment (i.e. fish Index of Biotic Integrity (IBI) scores and index of well being (IWB) scores, and number of invertebrate taxa, and number of EPT taxa).
5. Baseflow SSC and percent fine sediments in any reach can be used to create a simple model explaining significant variability in biotic conditions. Residuals from this model could possibly be used to examine water quality conditions not explained by sediment.
6. Fish IBIs correlate with benthic macroinvertebrate indices, providing redundant information. Macroinvertebrate data will accurately predict fish assemblage conditions in the absence of fish data.
7. Turbidity is an accurate surrogate for SSC in Piedmont streams.

Background:

In humid forested landscapes, hillslope erosion and sediment contributions to channels are typically small (except during episodic disturbance by wildfires or tropical storms) and can be easily assimilated by stream processes (Wolman 1967a, Julien 1995, Waters 1995). Human activities, specifically row-crop cultivation and cattle grazing, forestry operations, mining, and urbanization (i.e. increased impervious surface, construction runoff) can all greatly accelerate erosion processes and increase sediment inputs to streams (Waters 1995, Wood and Armitage 1997, Roy et al. 2005). Human activities alter physical features of streams by altering the volume and timing of runoff, increasing sediment delivery and rates of erosion up to 100 times the natural rate (Julien 1995). Rates are dependent upon many factors, such as climate, catchment geology, soil properties, topography, and land cover type (Brooks 1997, Wood and Armitage 1997).

Erosion and sedimentation can decrease water clarity, an important component of water quality and productivity. Turbidity, a measure of the cloudiness of fluid caused by suspended and colloidal organic and inorganic particles that scatter or absorb light, is a common measure of water clarity (Wetzel and Likens 2000, American Public Health Association. et al. 2005). Turbidity can be measured directly or indirectly; it is reported in nephelometric turbidity units (NTU), Formazin turbidity units (FTU), or Formazin attenuation units (FAU) depending on method of determination (Wetzel and Likens 2000, American Public Health Association. et al. 2005). Suspended Sediment Concentration (SSC), is an alternative measure of suspended particles, and may be more precise. It is indicative of all erosional activity within the watershed and provides an exact dry weight of all suspended materials within a whole sample. This differs from Total Suspended

Sediment (TSS) concentration, which measures an aliquot of a sample. While both measures can be substantially dissimilar from turbidity, they can be calibrated to accurately reflect water clarity.

Methods:

Study Region Background

Sediment issues in the Piedmont ecoregion are complicated by historic deposition related to poor agricultural practices (Trimble 1974, Richter and Markewitz 2001, Jackson et al. 2005), placer mining of gold (Leigh 1994, 1997), and extirpation of beaver (Naiman et al. 1988, Naiman et al. 1999). Bennett reported that 126,000 acres of productive Piedmont farm land was destroyed by erosion in 1933 (Bennett 1934). Poor management practices caused severe gullying on cultivated hills, washing sand and mud onto once fertile land. Overall, the Georgia Piedmont lost approximately 18-20 cm of topsoil to erosion during this era (Trimble 1974).

After 1930, with the abrupt end of cotton farming, much of the region returned to mixed oak and pine forests (Griffith et al. 2001). Because of such past abuse, today, even forested Piedmont streams tend to have sandy bottoms and high turbidities. The majority of soil that eroded during the agricultural period remains stored within local valley bottoms and has not washed downstream (Trimble 1975, 1999). For example, in the Savannah River watershed only four percent of eroded soil from the Piedmont uplands has been transported past the Fall Line (Trimble 1975, Jackson et al. 2005). Superimposed upon this agricultural legacy, the restored forests of the Piedmont are now being urbanized extensively in radial directions from major cities.

Study Sites

Sites were selected within the Southeast Piedmont physiographic province of Georgia and Alabama (Fig 3.1), an ecoregion underlain by metamorphic and igneous rock forming rolling landscapes with some steep-sided hills and plains areas. Elevations range from 152 m in the Lower Piedmont to 426 m in the Upper Piedmont. Mean annual temperature for the region is 23.3 ^{o}C, with hot summer temperatures averaging 31.7 – 33.4 ^{o}C and cool winter temperatures averaging 0.3 – 13.9 ^{o}C. Mean annual rainfall for this province is 1,250 mm. This study evaluated 42 streams draining watersheds ranging in size from 3 to 171 km^2 that had varying degrees of urbanization (Table 3.1).

Sites were selected from the Georgia Department of Natural Resources Wildlife Resources Division database of fish population surveys. Selected sites included reaches from each possible Index of Biotic Integrity category (e.g. excellent, good, fair, poor, and very poor) and each drainage basin within the study ecoregion (Savannah, Oconee, Ocmulgee, Chattahoochee, Coosa, and Tallapoosa).

Land Use

Land cover within each watershed was quantified with Geographical Information System software (ArcGIS 9.0). Watershed boundary information for each site was obtained from Wildlife Resources Division, Georgia Department of Natural Resources. Land use data was derived from United States Geologic Survey 2001 National Land Cover Database (30-m pixels), and included 29 land use categories. Overall basin land use was determined, as well as land use within 7.62-m buffer – the minimal buffer required by state law.

After determining percentages for each land use type within the watershed, sites were then grouped into three broad categories: forested, urbanized, or mixed-use. Streams with less than 20 percent urbanization and greater than 59 percent forest within their watersheds were judged to be forested sites. Streams with greater than 20 percent urbanization and less than 59 percent forest were deemed urbanized sites. All other sites were selected as mixed-use sites (Table 3.1).

Land use changes were evaluated for 26 streams between 1998 and 2001. The 1998 Land use data was determined using the Georgia Land Use Trends (GLUT) Land Cover layer for the state of Georgia, which was produced by the Natural Resources Spatial Analysis Laboratory (NARSAL).

Bed Sediment Composition

Bed texture was evaluated using a modified pebble count and by dry sieving riffle samples (Bevenger and King 1995, Roy 2000, Kondolf and Piégay 2003). The percent fine sediment was based on the percentage of particles less than 2 mm (silt and clay) from pebble count samples that consisted of 200 pebbles; pebble counts were conducted over the entire length of the study reach. In addition, three replicate 1-L samples were collected from riffle areas. Samples were dried, sieved, and weighed to determine the proportion of fine sediment (particles less than 2 mm). Assessment of riffle habitat was selected because of 1) the sensitivity of riffles to increased sediment supply (Parker and Klingeman 1982, Dietrich et al. 1989) and 2) the correlation between percent fines in riffles and IBI score within Georgia's Etowah river system (Walters et al. 2001, Walters et al. 2003c, Walters et al. 2009). For analysis, particulate classes were converted to the *phi* scale and mean sediment size (Dg), median or 50th percentile of bed sediment size (D50), and the 84th percentile bed sediment size (D84) (Maidment 1993, Leopold 1994).

Turbidity, SSC, Conductivity, and pH

Turbidity, SSC, specific conductance, and pH measurements were recorded monthly between October 2002 and June 2005 during baseflow conditions; baseflow was considered to be conditions not attributable to stormflows, or less than a 30% change in flow over a three day period. A total of twelve baseflow samples were collected for each site. Turbidity (NTU) was evaluated from grab samples using a portable turbidimeter (Turbidimeter Model 2100P, Hach Company, Loveland, CO) (Greenberg et al. 1992). Recorded values were based on the mean of three samples measured as NTU. Suspended sediment concentration samples, 450 mL each, were collected using a US-DH48 depth integrated sampler (Hauer and Lamberti 1996). Samples were returned to the lab, filtered on Whatman glass fiber filter paper (93-AH), dried for 24 hours at 103 - 105 ^{o}C, and weighed to determine concentration of suspended sediments (Greenberg et al. 1992, Hauer and Lamberti 1996). Specific conductivity and pH were measured *in situ* using a portable water analyzer (Quanta by Hydrolab Corporation, Austin Texas), which was calibrated with standards prior to each sampling event. All samples and water quality data were collected mid-stream within a glide or run unit.

Physical Habitat Data

Total large woody debris (LWD) was counted and placed into a size class by diameter: 10-20 cm, 20-40 cm, 40-60 cm. In log jams, where number and size of counts were impractical, a size estimation was determined based on two classes 68-80 cm and >80 cm. All structures >10 cm in diameter and >1 m in length, which were in contact with the stream channel, were considered LWD (Hairston-Strang and Adams 1998). LWD density was determined by dividing the number of large wood structures present by study reach length. The Shannon-Weiner diversity index was calculated for each reach based on number of LWD within each category and volume of LWD within each category.

Canopy cover was determined during full leaf-out using a spherical densiometer (Lemmon 1956). Canopy cover was measured at three points within the study reach; at each point four measurements were taken (up, down, left, and right). Data were taken in summer 2003 on all but three sites (Long Cane, Lazar, and Raccoon Creek), which were unwadeable; therefore these data were collected in summer 2004. The three measurements were averaged for each site.

Unit stream power was calculated as $\omega_w = VS$, where V is the mean velocity (m/s) in a stream cross section and S is the energy slope (Gordon 2004). Energy slope or surface water gradient was calculated using the formula $S = \Delta E/ L$, where S represents average slope; ΔE is the water surface elevation at the upstream reach minus the water surface elevation at the downstream reach, and L is the reach length (Hauer and Lamberti 1996).

Habitat assessments were conducted within each study reach. The physical characteristics associated with the stream and stream bank were tabulated on site, including a general description of adjacent land use (Table 3.2). Individual habitat metrics related to substratum type and stability, channel morphology, and stream bank stability were rated and used to calculate a riffle/run habitat assessment score (Table 3.2) (Georgia DNR-WRD 2005). Instream measurements included average width and depth of each stream unit type (i.e., glide, pool, run, riffle, bedrock cascade). Stream channel cross sectional area was measured at two transects within each study reach. At each transect, we measured bankfull area and wetted perimeter area (active channel). Depth measurements were taken across the channel at each elevation change (Leopold 1994, Peterson and Rabeni 1995, Murphy et al. 1996).

Fish data

Fish population surveys were conducted between 1998 and 2002 by Georgia DNR, using backpack electrofishing and seining techniques (Georgia Environmental Protection Division 2005). These data were used to assess the water quality of streams based on Index of Biotic Integrity (IBI) and Index of Well Being (IWB) scores. The IBI is based on 13 metrics related to richness, composition, trophic dynamics, abundance and condition (Table 3.3). Possible IBI scores range between 8 and 60 with higher scores indicating excellent conditions and lower scores indicating very poor conditions (Georgia Environmental Protection Division 2005). Sites are assigned to one of five integrity classes based on their total IBI score (Table 3.4). The IWB is a composite index that combines two parameters of diversity and two parameters of abundance (Table 3.5).

Invertebrate collection and processing

Macroinvertebrates were collected between October to February 2002 and October to February 2003 within a 100 m sampling reach (Georgia Environmental Protection Division - Water Protection Branch 2004). One set of samples were collected at each site per sampling season; a total of two samples were collected for 37 sites, at five sites, seasonal samples were collected, for a total of four samples per site. All available productive habitat types were sampled at each location using 500-µm mesh dip net

(WaterMark®). In high gradient streams riffles, leaf packs, sand, woody debris, and undercut banks were sampled. In low gradient systems the riffle jabs are reallocated to the remaining habitat types (Table 3.6). A total of 20 jabs were collected from each reach. A single jab consists of an upward/forward thrust for a distance of 1-m. All 20 jab samples were compiled prior to subsampling. Samples were fixed in 100% ethanol, which was diluted to a 90% ethanol solution for preservation.

Samples were washed through a # 60 sieve, in the lab, to remove sand, silt, and clay. Any large organic material (whole leaves, twigs, algae or macrophyte mats) not removed in the field was rinsed, visually inspected for macroinverte-brates, and discarded. Washed samples were uniformly distributed in a Caton Macroinvertebrate Subsampler, with a standardized screen (595-µm screen, 30 squares, each 6 cm^2). Squares were selected to be removed from the grid via use of a random number table generated in an Excel spreadsheet (Microsoft Office 2003). All material (organisms and debris) were removed from the square and placed in a sorting tray. Organisms were separated from the organic matter and stored in a 70% ethanol solution. Squares were removed in this manner until a 200 (± 20%) organism subsample was obtained. Each grid selected for sorting must be sorted in its entirety, regardless of number of organisms obtained.

We identified organisms to genus level when possible, with the exception of Acari, Planaria, Lepidoptera, Hirudinea, Ceratopogonidae, Oligochaeta, and Chironomidae. Organisms were identified using the following taxonomic keys: Brigham et al. (1982), Epler (1996), Merritt and Cummins (1996), Wiggins (1996), and Thorp and Covich (2001).

Tolerance and functional feeding group values were assigned according to designations by the Georgia Environmental Protection Division (2007). Ecological integrity was characterized using a subset of metrics selected by the Georgia Environmental Protection Division for sub ecoregion 45b (Table 3.7). Final counts, total number collected, were used to tabulate metrics associated with species richness, composition, tolerance, and functional feeding groups (Barbour et al. 1999, Georgia Environmental Protection Division 2004a, 2007) (Table 3.8).

Analysis

Prior to analysis, all parameters were evaluated for normality and constant variance. Proportional data sets (e.g. % landuse, % fines) were arcsin transformed when necessary. All data points were used for determining the relationship between turbidity and SSC; all other analyses used only baseflow measurements. Regression analysis was used to examine relationships between each variable and turbidity/SSC or percent fine sediments. Sites with anomalous electrical conductivity (e.g. active construction or effluent inputs) were removed from the data set.

An Analysis of Variance (ANOVA) was used to compare relationships between land use and biotic communities. The Holm-Sidak test was used to separate means of the macroinvertebrate data (i.e., total taxa, EPT taxa, and Macroinvertebrate Site Index), which were normally distributed. Because fish IBI and IWB data were not normally distributed, a Kruskal-Wallis test was preformed and Dunn's Method was used to separate the means. Forward stepwise regression was used to determine which metrics should be used to create a simple model explaining significant variability in biotic conditions. Variables that were highly correlated, such as turbidity and SSC, were analyzed separately. Akaike's Information Criteria (AIC) with the small sample bias adjustment (AIC_c) was used to determine the two "best" models, or models with lowest AIC_c, from all possible models; the fit of each candidate model was then evaluated based on Akaike model weight (Akaike 1973, Hurvich and Tsai 1989, Burnham and Anderson 2004). Akaike model weights were also used to analyze differences in models with correlated variables.

The Pearson Product Moment correlation coefficient determined the degree of association between landuse variables and biotic index scores, as well as the degree of association between metrics used in the fish indices and macroinvertebrate indices. These analyses evaluate whether fish and macroinvertebrate indices are consistent. Fish IBI and IWB scores were regressed against MBI scores to determine if the indices were providing the same categorical information about water quality. Because IWB categories are based on drainage basin area (DBA), scores within each category were averaged to create the comparison of fish IWB and MBI.

Results:

Land use data were combined into four major cover types (i.e., urban, agriculture, forest, and wetlands) because initial analysis at finer resolution indicated there was not a significant difference in levels of urbanization (i.e., low, medium, or high) or forest types (i.e., deciduous or coniferous). Agricultural areas included pasture/hay and cultivated crop land uses. Forest areas were a combination of all forest types; hardwood, oak, pine, and mixed forests. Wetlands were comprised of woody, palustrine forested, and palustrine scrub/shrub wetlands. All other land use types (e.g., grasslands, open water, and shrub/scrub forest) were combined in an “other” category. Categorizing land use into three broad groups resulted in 17 urban sites, 16 mixed-use sites, and 8 urban sites. One urban site and one mixed-use site was excluded due to beaver activity. None of the study sites contained > 71 % urbanization. Overall, forested land use for 2001 ranged between 26 and 83 percent, decreasing an average of 3.6 percent from 1998. Urbanization significantly increased by an average of 8.6 percent between 1998 and 2001.

Overall basin land use was not significantly correlated ($\alpha = 0.05$) to baseflow SSC, baseflow turbidity, or percent fine sediments (Table 3.9). During baseflow conditions, turbidity and SSC values were all below 30 (NTU and mg/L respectively), and highly variable (Table 3.10). Bed sediments ranged in size from silts/clays to bedrock, with percent fine sediment ranging between 0.44 and 99.00 percent (Table 3.10). Mean percent fines were numerically greater at urban sites than forested or mixed-use sites, but this relationship was not significant based on either method of data collection (i.e., pebble count or particle size analysis) (Table 3.10). Turbidity, SSC, and percent fines were significantly correlated to local riparian decreasing with increasing percent forest (Table 3.9). There were no differences in the relationship between turbidity, SSC, or percent fines and 1998 versus 2001 land use.

Biotic communities were analyzed based on overall health and by examining the composition of each assemblage. Fish IBI scores, which ranged between 16 and 54, were strongly correlated with basin urbanization (Fig 3.2a). In basins with low forest cover and high urbanization, IBI scores were in either the poor or very poor category. Basins with greater than four percent impervious surface had equivalent results as basins with high urbanization- IBI scores in the poor and very poor category (Fig 3.2b). Fish IBI and

IWB scores and macroinvertebrate metrics differed significantly among land use categories. Urban sites always had lower IBI and IWB scores and fewer macroinvertebrate taxa than forested sites, whereas results varied for the mixed sites (Table 3.11). Urbanization was most closely associated with decreases in catch per unit effort, IBI and IWB scores, and increases in fish deformities (DELTs). When comparing these relationships to percent land use within a 7.62-m (25-ft) buffer, percent forested land use adjacent to the channel explained more of the variability among biotic metrics than watershed-basin land use (Table 3.12). Differences in metrics assessed by IBI and IWB were apparent in that the IBI was more highly correlated to basin-wide land use and percent forested buffer than the IWB, which was not related to percent forested buffer.

Generally, aquatic community indices decreased with increasing turbidity, suspended sediment concentration, and percent fines but considerable variability was unexplained (Figs 3.3 – 3.5). Turbidity was a better predictor of fish IBI scores than SSC, but SSC was a better predictor of fish IWB scores than turbidity (Fig 3.3). We further evaluated IBI metrics attempting to determine which, if any, were affected by the sediment parameters. A forward regression indicated that native fish species, benthic invertivores, sensitive fish species, and fish catch per unit effort were the variables that best predicted IBI scores. Akaike analysis indicated that impervious surface, urbanization, and sediment metrics (i.e, D_{84}, percent fines) are the most influential factors affecting fish metrics and overall fish IBIs and IWBs; 85 percent of the models contained a land use variable and 71 percent contained a sediment variable (Table 3.13).

Not unlike the fish biotic indices, differences in the invertebrate EPT taxa and total taxa were only weakly explained by turbidity and SSC concentrations. Both the total number of taxa and number of EPT taxa decreased with increasing baseflow sediment metrics (Fig 3.4). The benthic biotic index (MBI) was correlated to turbidity and SSC, but not percent fines (Fig 3.5). Turbidity, SSC, percent fines (PSA <2) and land use were significant components in the models to explain variability in the benthic macroinvertebrate communities (Table 3.13). Drainage basin area was a significant factor in both fish and macroinvertebrate models despite selecting basins less than 259 sq kilometers in area to reduce natural variation due to stream size and calculating fish indices based on basin size.

Fish and macroinvertebrate indices (IBI, IWB, and MBI) were significantly related, but the relationships exhibit considerable scatter, with r-square values of 0.396 and 0.312 (Figure 3.6). Both the IBI and IWB have a stronger relationship with number of EPT taxa than with the MBI score (Table 3.14). When indices are placed into their water quality categories, discrepancies in ranking are frequent (Figure 3.7 and 3.8). The categorical ratings of macroinvertebrate and fish indices agree in only 45.2 percent (with IWB) and 23.8 percent (with IBI) of sites and 29 to 31 percent (IWB and IBI, respectively) of sites differ by two categories.

Fish indices and metrics are correlated to thirteen environmental variables, whereas the macroinvertebrate indices are correlated to only seven environmental variables. Fish indices are most highly correlated to specific conductivity and stream power (Table 3.15), while EPT taxa and number of taxa are most highly correlated to water clarity, suspended sediment, and specific conductivity (Table 3.16). The macroinvertebrate index is not strongly correlated to any individual environmental variables we measured (Table 3.16). EPT taxa are providing similar information to native fish species (r = 0.601), CPUE (r = 0.601), and benthic invertevores (r = 0.702) and may be useful in interpreting fish assemblages when fish data are absent.

All three biotic index scores fell within a narrow range, with few points in either the very poor (i.e, lowest) or excellent (i.e., highest) categories. IWB has a slightly stronger relationship with the MBI (Fig 3.6) and categorically corresponds to the MBI (Fig 3.7). The fish IBI consistently predicted poorer water quality conditions than the fish IWB (Fig 3.9). Principal component analysis indicates that the first four factors explain 66.13 percent of the variance in the dataset (Figure 3.10 and Table 3.17). None of the component one factors explains a large amount of variability in scores. The factors in component two that explain most of the variability are related to canopy cover, wetted perimeter active channel (WPAC), and wetted perimeter bankfull channel (WPBF).

Turbidity was not an accurate surrogate for SSC in Piedmont streams without detailed rating curves data. Baseflow conditions are relatively similar among these streams, and therefore turbidity and SSC were not strongly correlated without additional stormflow data (Fig 3.11). In addition, replacing turbidity with SSC parameters decreased overall biotic index model significance between 1.1 and 75.6 percent (mean = 14 percent), further suggesting that turbidity and SSC are not redundant measures for these streams (Table 3.18).

Discussion:

Although urbanization disturbs channel equilibrium, usually leading to channel widening and waves of bank instability, bed aggradation, and bed degradation (Wolman 1967b, Paul and Meyer 2001, Booth et al. 2002), our study did not find a relationship between overall watershed land use and either SSC, turbidity, or percent fines. We infer that effects of urbanization on sediment metrics were masked by legacy sedimentation and channel disturbance imposed by the agricultural history of the Georgia Piedmont. Wolman (1967) found that, in the Maryland Piedmont, channel erosion and sedimentation continued years after initial construction of urban areas was complete and that concrete bank stabilization did not completely eliminate deposition. Wolman proposed, but did not show, that sediment inputs and exports should decrease over time. Local land use changes can , such as urbanization, can moves channels out of equilibrium and initiate a dynamic progression to a new channel state (Thorne et al. 1996). The influence of local riparian disturbance on water quality was not overshadowed by the residual sediments, as evidenced by the significant relationships of percent forested buffer with turbidity, SSC, and percent fines.

Our data reflect the unpredictability of natural disturbance at baseflow, as well as the energy required to move particles that is only available during stormflow. Price and Leigh (2006) observed baseflow SSC ranges between 1 mg/L and 10 mg/L in lightly-impacted streams and 13mg/L and 37 mg/L in moderately-impacted streams. In Piedmont Etowah River tributaries, Leigh et al. (2002) reported TSS values ranging between 2 and 50 mg/L for basins with 27 to 87 percent forested land. The maximum SSC value monitored for any land use was 13.8 mg/L. Sediment transport occurs when particles overcome frictional resistance and become entrained in the water column; particles remain suspended until the stream lacks energy to carry the load (Gordon 2004). Channels are relatively stable during low flows producing minute amounts of sediment from biological and physical processes (Estrany et al. 2009). It is high flows and flooding during storm events that move sediment and shape channel morphology (Wondzell and Swanson 1999, Estrany et al. 2009).

The impact of land use change on biota has been studied extensively since the late 1970s. Results from this study are comparable to existing works which suggest urbanization decreases diversity, abundance, and homogenization of aquatic populations by altering hydrology, chemistry, and/or geomorphology (Walters et al. 2001, Brasher 2003, Nilsson et al. 2003, Walters et al. 2003a, Walters et al. 2003b, Marchetti et al. 2004, Morgan and Cushman 2005, Roy et al. 2005, Walsh et al. 2005a, Chadwick et al. 2006). However, it is debated as to whether local or watershed scale changes are most influential to water quality and aquatic community health (Table 3.10), as well as the degree of urbanization that alters community structure (Walsh et al. 2005a). Previous studies have indicated a general threshold of 10 percent impervious surface or greater cause degradation (Booth and Jackson 1997). Walters et al. (2001) proposed a baseflow turbidity threshold of 10 NTU for biotic impact; the study found that values greater than 10NTU produced changes in fish assemblages. When considered in the context of other studies of urbanization effects, the results indicate that aquatic ecosystem responses to urbanization have high consistency, but they are not uniform. Our percent impervious surface was less than the suggested threshold, but our turbidity values were above the upper limit for all land use categories. Ecoregion characteristics and effects of historic land use activities both affect stream response to urbanization.

One reason there is not a definitive answer may be because of the multiple mechanisms that control responses of aquatic communities to urbanization (Paul and Meyer 2001, Roy et al. 2003, Roy et al. 2005). Georgia’s fish and macroinvertebrate indices assemblages exhibit considerable scatter in their direct relationships and the indices reveal important differences in their responsiveness to land use and landscape variables. For example, independently percent fines can adversely affect aquatic communities, but percent fines can have greater impacts when coupled with urbanization because the particles carry contaminants that degrade stream health. Percent fines were negatively correlated to forest cover within the riparian zone, most likely because vegetation stabilizes streambanks reducing erosion (Kreycik 2001), but were not correlated to catchment land use patterns. Nonetheless, fish IBI scores were more strongly associated with watershed urbanization than percent fines or riparian forest. Strayer et al. (2003) reported similar findings, indicating that scale resolution is dependent upon the specific questions being asked. It is also important to note that biotic index scores are dependent upon the organism being sampled. Fitzpatrick (2004), found that streambed sedimentation had affected macroinvertebrates more than fish

While some of the important fish indices are highly correlated to EPT taxa, it is clear that the MBI cannot substitute for fish assemblage data. There are enough dissimilarity in the fish and macroinvertebrate responsiveness to landscape factors to suggest that both fish and macroinvertebrate indices should be sampled, as they are related to environmental conditions. Walters et al. (2009) observed that in newly urbanizing watersheds, macroinvertebrates were more sensitive to percent urbanization, conductivity, and percent fines in riffles than fish. Fish responses were more closely tied to turbidity and forest cover. Several other studies have shown the importance of cross taxon surveys (Triest et al. 2001, Passy et al. 2004, Feio et al. 2007, Carlisle et al. 2009). In the absence of fish data, calculating EPT taxa is important as it can offer insight into the condition of native fish species, benthic invertivore species, and catch per unit effort. Cross taxonomic assessment provide a complete picture of how biota are reacting to changes at multiple scales and stressors.

Stream ecologists in Georgia are interested in using biotic indices to maintain or improve stream health with respect to human influence and landscape condition. Each year billions of dollars are spent on restoration for impaired stream systems (Moerke et al. 2004). Many restoration methodologies exist, but there is still much concern regarding identification, sustainability, and effectiveness of projects (Gore and Shields 1995, Moerke et al. 2004, Walsh et al. 2005b, Hey 2006, Ekness and Randhir 2007). One tool to aid identification of restoration sites may be to determine which sites are most likely to be sustainable based on the watershed land use for similar streams. From our data there are two fair sites, two poor sites, and two very poor sites that may be suitable candidates for successful restoration, all located in the lower right-hand quadrant of Fig 3.2a.

Contrary to expectations, baseflow suspended sediment concentrations and turbidity levels in Georgia Piedmont streams did not correlate well, despite the significant relationship when stormflow is included. These data do not suggest that one can be used in place of the other in this ecoregion. Previous studies examining the relationship between turbidity and SSC have produced results varying from an $R^2 = 0.68$ to an almost one-to-one relationship (Gray et al. 2000, Pavanelli and Bigi 2005, Stubblefield et al. 2007). Studies with higher correlation come from streams with similar surrounding soils and geologic formations, or studies in which dilutions were not necessary to obtain an accurate turbidity reading. Several studies have also examined the validity of using a combination of turbidity and acoustic Doppler velocity measurements as a surrogate for SSC, but concluded that calibration curves are necessary for each body of water sampled because of variability related to water quality and sediment characteristics (Chanson et al. 2008). While the samples for this study were collected within the same ecoregion and similar drainage basin areas, streams varied greatly in basin morphology and historic land use effects which were not accounted for.

Conclusion:

In the Georgia Piedmont, neither baseflow suspended sediment concentrations, baseflow turbidity, nor bed particle size distributions were significantly related to overall watershed imperviousness. Together these simple sediment metrics accounted for a significant amount of variation biotic condition, which elucidate biological conditions of streams. Biotic community metrics were highly sensitive to land cover, specifically the fractions of urban land, imperviousness, and forest. Nearly all fair, good, and excellent fish IBI scores occurred in basins with more than 50 percent forest and less than 15 percent urban area and less than four percent impervious surface, indicating that Georgia Piedmont streams may be more sensitive to urbanization effects than streams in other parts of the country. The fact that easily measured sediment metrics, such as turbidity and suspended sediment concentration, did not correlate strongly to land use but did help explain biological condition argues for their inclusion in water quality assessments. We recommend measuring baseflow turbidity, baseflow SSC, and unit stream power in water quality assessments, albeit a superior explanatory metric could not be determined with this dataset.

Macroinvertebrate stream health categories (MBI) are more closely aligned with fish IWB categories than with IBI classes, but their disparate classification of biotic integrity indicates calibration is required to use one in place of the other. If surveying both populations is cost prohibitive, it is imperative to define your purpose as one assemblage may be able to answer your question more effectively than the other. Results from multiple taxonomic groups can better direct management decisions that protect biodiversity from numerous stressors.

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Table 3.1. Physical characteristics of 42 study streams within the Piedmont ecoregion of Georgia from 2001 Georgia GAP data.

Site No.	Site	County	Drainage Basin	DBA (km^2)	Urban	Forest	Other	Land use Designation
1	Copeland Creek	Hancock	Oconee	12.3	3.4	77.7	8.3	Forested
2	Indian Creek	Oglethorpe	Savannah	51.3	5.1	65.1	16.3	Forested
3	Whooping Creek	Carroll	Chattahoochee	68.3	5.7	62.5	9.4	Forested
4	Snake Creek	Carroll	Chattahoochee	19.1	11.4	60.8	8.7	Mixed-Use
5	Brittens Creek	Meriwether	Flint	9.7	3.3	51.9	22.5	Forested
6	Cabin Creek	Spalding	Ocmulgee	10.8	37.7	46.2	5.9	Urban
7	Tobesofkee Creek	Lamar	Ocmulgee	6.3	34.4	43.6	7.2	Urban
8	Yellow Water Creek	Butts	Ocmulgee	53.6	12.1	39.6	12.6	Urban
9	Kendall Creek	Meriwether	Flint	11.4	4.0	73.9	12.9	Forested
10	Lightwood Log	Hart	Savannah	15.4	7.5	38.4	12.8	Mixed-Use
11	Noonday Creek	Cobb	Coosa	45.8	61.4	25.9	7.5	Urban
12	Settingdown Creek	Forsyth	Coosa	48.7	19.1	46.8	6.1	Mixed-Use
13	Noonday Creek	Cherokee	Coosa	107	70.9	24.1	3.0	Urban
14	Zoie Brown Creek	Hancock	Oconee	15.8	5.2	73.8	16.2	Forested
15	Sandy Run	Putnam	Oconee	13.6	3.9	76.6	18.9	Forested
16	Crooked Creek	Jasper	Oconee	10.1	2.9	32.6	18.1	Mixed-Use
17	Little Buck Creek	Lamar	Ocmulgee	8.0	6.4	50.9	11.7	Forested
18	Buck Creek	Lamar	Ocmulgee	83.8	7.7	53.6	12.9	Forested
19	Wahoo Creek	Coweta	Chattahoochee	53.3	20.5	54.1	10.2	Urban
20	Little Chehaw	Jones	Ocmulgee	7.9	6.0	68.9	10.9	Forested
21	Walnut Creek	Jones	Ocmulgee	80.3	7.2	57.2	18.8	Forested
22	Heads Creek	Spalding	Flint	55.9	18.8	45.4	12.8	Urban
23	Gum Creek	Heard	Chattahoochee	20.0	4.1	61.3	5.3	Forested
24	Flat Creek	Troup	Chattahoochee	70.2	4.1	62.1	17.0	Forested
25	Long Cane Creek	Troup	Chattahoochee	171	6.1	61.2	17.5	Forested
26	Hillabahatchee Creek	Heard	Chattahoochee	58.5	2.2	74.1	15.2	Mixed-Use

Table 3.1 (continued). Physical characteristics of 42 study streams within the Piedmont ecoregion of Georgia from 2001 Georgia GAP data.

27	Auchumpkee Creek	Upson	Flint	112	2.3	71.5	14.8	Forested
28	Lazar Creek	Talbot	Flint	83.6	5.4	69.7	13.7	Forested
29	Potato Creek	Lamar	Flint	161	7.7	51.2	13.1	Forested
30	Rooty Creek	Putnam	Oconee	22.6	0.0	72.0	19.0	Forested
31	Mountain Creek	Coweta	Chattahoochee	19.9	23.7	58.9	7.8	Mixed-Use
32	Red Oak Creek	Heard	Chattahoochee	14.8	4.2	68.7	15.0	Mixed-Use
33	Biger Creek	Madison	Savannah	12.4	14.0	47.2	10.5	Mixed-Use
34	Bull Creek	Muscogee	Chattahoochee	87.4	19.3	59.9	11.6	Mixed-Use
35	Bear Creek	Newton	Ocmulgee	87.8	5.5	42.6	15.5	Mixed-Use
36	Butler Creek	Cobb	Coosa	24.1	59.8	33.8	3.8	Urban
37	Allatoona Creek	Cobb	Coosa	48.4	34.6	53.9	4.6	Mixed-Use
38	Etowah River	Lumpkin	Coosa	143	2.7	94.4	1.3	Forested
39	Shoal Creek	Dawson	Coosa	53.9	5.9	75.3	9.7	Mixed-Use
40	Beach Creek	Haralson	Tallapoosa	14.2	16.7	58.2	3.4	Mixed-Use
41	Raccoon Creek	Paulding	Coosa	19.6	2.7	83.1	11.1	Mixed-Use
42	Little Tallapoosa River	Carroll	Tallapoosa	119	16.5	48.3	14.0	Urban

Table 3.2. Description of habitat parameters (Georgia EPD-WRD 2005).

Habitat Parameter	Description
Epifaunal Substrate/Instream Cover	The amount of substrates that are available as cover for aquatic organisms
Embeddedness	Degree to which cobble, boulders, and other rock substrates are surrounded by fine sediment and silt.
Velocity/Depth Combinations	A stream's characteristic velocity/depth regime.
Channel Alteration	Measure of large-scale alteration in stream morphology that affects flow, instream habitat, and/or sedimentation rates.
Sediment Deposition	Relates to the amount of sediment that has accumulated and the changes that have occurred to the stream bottom as a result of deposition.
Riffle Frequency	Estimates the frequency of occurrence of riffles and thus the heterogeneity occurring in a stream
Channel Flow Status	Degree to which the channel is filled with water when the stream reach is sampled.
Bank Vegetative Protection	Amount of stream bank that is covered by vegetation.
Bank Stability	Evidence of active and potential to erosion.
Riparian Vegetation Zone Width	Measures the width of natural vegetation from the edge of the upper stream bank out through the floodplain.

Table 3.3. Subset of metrics used in characterizing ecological integrity of macroinvertebrate communities and their response to impairment (Barbour et al. 1999, Georgia WRD/EPD 2004, 2007).

Metric	**Description**	**Response to impairment**
Native Species	Total number of native fish species	Decrease
Benthic Invertevore Species	Total number of darters, madtoms, and sculpins	Decrease
Sensitive Species	Number of species designated as intolerant	Decrease
Catch per unit effort (CPUE)	Number of fish collected per 200 meters of stream	Decrease
IBI Score	Multimetric index comprised of 13 metrics related to richness, composition, trophic dynamics, abundance and condition	Decrease
IWB Score	Composite index that combines two parameters of diversity and two parameters of abundance	Decrease

Table 3.4. Total IBI scores, integrity classes and the attributes of those classes – modified from Karr 1981 and Schleiger 2000 (Georgia Environmental Protection Division 2005).

Total IBI Score	**Integrity Class**	**Attributes**
60-52	Excellent	Comparable to the best ecoregional reference conditions; all regionally expected species for the habitat and stream size, including the most intolerant species are present with a full array of size classes; significant proportion of the sample composed of benthic fluvial specialist and insectivorous cyprinid species; number of individuals abundant, representing a balanced trophic structure.
50-44	Good	Species richness somewhat below expectation, especially due to the loss of the most intolerant forms; good number of individuals, with several species of suckers, minnows, and benthic invertivores present; trophic structure shows some signs of stress.
42-34	Fair	Species richness declines as some expected species are absent; few, if any, intolerant or headwater intolerant species present; trophic structure skewed toward generalist, herbivorous, and *Lepomis* species as the abundance of insectivorous cyprinid and benthic fluvial specialist species decreases.
32-26	Poor	Sample dominated by generalist, herbivorous, and *Lepomis* species; proportion of non-native species and hybrids increases; intolerant and headwater intolerant species absent; benthic fluvial specialist and insectivorous cyprinid species in low abundance or absent; growth rates and condition factors commonly depressed and diseased fish are often present; number of individuals in low abundance.
24-8	Very Poor	Few fish present, mostly generalist and *Lepomis* species; condition factors poor as unhealthy and juvenile individuals dominate the sample; fish with disease, eroded fins, lesions, and tumors common.
No Fish		No fish collected in the sample

Table 3.5. Index of well-being scoring criteria and integrity classes for wadeable streams in the Piedmont ecoregion of Georgia (Georgia Environmental Protection Division 2005).

Total IWB Score	**DBA (Sq ki)**	**Integrity Class**	**Attributes**
> 8.1 > 9.6	< 38.84 > 38.84	Excellent	Comparable to the best regional reference conditions; all regionally expected species for the habitat and stream size, including the most intolerant species, are present with a full array of size classes; healthy species diversity within the fish community, indicated by elevated evenness scores; number of individuals abundant; total biomass is high, with each level of the food web represented, indicating a balanced trophic structure.
8.1 - ≥ 7.3 9.6 - ≥ 8.6	< 38.84 > 38.84	Good	Species richness somewhat below expectation; evenness scores decrease as species diversity falls, especially due to the loss of the most intolerant forms; good number of individuals in the sample, with several species of benthic fluvial specialists and insectivorous cyprinids present; some decreases in total biomass as trophic structure shows some signs of stress.
7.3 - ≥ 5.7 8.6 - ≥ 6.6	< 38.84 > 38.84	Fair	Species richness and diversity decline as some expected species are absent; abundance of individuals declines; total biomass continues to decline as some levels of the food web in low abundance or missing; trophic structure skewed toward generalist feeders and/or *Lepomis* species as the abundance of insectivorous cyprinid and benthic fluvial specialist species decreases.
5.7 - ≥ 4.9 6.6 - ≥ 5.6	< 38.84 > 38.84	Poor	Number of individuals is low; species richness and diversity are very low, with benthic fluvial specialist and insectivorous cyprinid species in low abundance or absent; sample dominated by generalist feeders, herbivores, and *Lepomis* species; increase in the proportions of non-native species and hybrids; growth rates depressed as sample is heavily skewed to the smaller size classes; total biomass low.
< 4.9 < 5.6	< 38.84 > 38.84	Very Poor	Sample represented by few individuals, mainly generalist feeders and *Lepomis* species; some sites dominated by non-native species; total biomass very low.

Table 3.6. Prioritized list of habitat types for sampling and sample reallocation for the Georgia EPD Macroinvertebrate 20-jab method (Georgia Environmental Protection Division - Water Protection Branch 2004).

		Number of Stream Samples	
Priority	**Habitat Type**	**High Gradient**	**Low Gradient**
1	Riffles	6	0
2	Woody Debris/Snags	5	8
3	Undercut Banks/Rootwads	3	6
4	Leaf Packs	3	3
5	Soft Sediment/Sand	3	3

Table 3.7. Macroinvertebrate multi-metric index for the Southern Outer Piedmont ecoregion (45b).

Metrics	**Standardized Metric Scores/Index Score/Site Ranking**	**Response to Impairment**
EPT Taxa	Number of taxa in Ephemeroptera, Plecoptera, and Trichoptera insect orders	Decrease
Coleoptera Taxa	Number of taxa in the order Coleoptera	Decrease
% Oligochaeta	Percent Oligochaeta Taxa	Increase
% Plecoptera	Percent Plecoptera Taxa	Decrease
Shredder Taxa	Number of shredder taxa	Decrease
Scraper Taxa	Number of scraper taxa	Decrease
Swimmer Taxa	Number of swimmer taxa	Decrease
Site Index Score	Total score for individual site	Decrease
Numeric Ranking	Ranking of 1-5, rating health of stream	Increase
Narrative Description	Narrative description of numeric ranking and stream health rating	-
Stream Health Rating	Rating of A-C describing health of stream	-

Table 3.8. Macroinvertebrate multi-metric index a) scoring criteria and integrity classes and b) management decisions for the Southern Outer Piedmont ecoregion (45b).

a)

Numeric Ranking	Narrative Description	Stream Health Rating	Index Score
1	Very Good	A	> = 84
2	Good	A	56 - 83
3	Fair	B	32 - 55
4	Poor	C	17 - 31
5	Very Poor	C	< = 16

b)

Numeric Ranking	Management Decision
1	Continue periodic monitoring to detect change baseline reference condition
2	Continue periodic monitoring to detect change baseline reference condition
3	Frequent monitoring critical to detect change in ecological status, lower range especially
4	Frequent monitoring necessary to determine remediation needs and if remediation has been successful
5	Frequent monitoring necessary to determine remediation needs and if remediation has been successful

Table 3.9. Relationship between turbidity, SSC, percent fines and land use (percent urban and percent forest).

Relationship		Watershed Land use[1] R-square	Watershed Land use[1] Slope[3]	Watershed Land use[1] p-value[4]	7.62 m Buffer[2] R-square	7.62 m Buffer[2] Slope[3]	7.62 m Buffer[2] p-value[4]
Turbidity vs. SSC[5]		**0.775**	**(+)**	**<0 .001**			
Turbidity vs. SSC[6]		**0.531**	**(+)**	**<0 .001**			
Turbidity vs.							
	Urban	0.021		0.363	0.023		0.344
	Impervious	0.019		0.395	0.015		0.452
	Forest	0.014		0.467	**0.270**	**(-)**	**<0.001**
SSC vs.							
	Urban	0.003		0.734	<0.001		0.994
	Impervious	0.002		0.763	0.007		0.609
	Forest	0.040		0.210	**0.153**	**(-)**	**0.011**
Particle size vs Pebble Ct		**0.745**	**(+)**	**<0.001**			
Percent fines (particle size analysis) vs.							
	Urban	<0.001		0.921	0.011		0.520
	Impervious	<0.001		0.994	<0.001		0.858
	Forest	0.021		0.365	**0.215**	**(-)**	**0.002**
Percent fines (pebble count) vs.							
	Urban	0.025		0.324	0.033		0.258
	Impervious	0.028		0.294	0.016		0.438
	Forest	0.074		0.086	**0.185**	**(-)**	**0.005**

[1]Land use determined for entire watershed;
[2]Land use determined within 7.62 m buffer (25 ft buffer required by state law)
[3]Slope direction; indicates positive or negative relationship between variables. Included for significant relationships only.
[4]Power of performed test with alpha = 0.050 *(Note: Bold values are significant)*
[5]Linear regression between turbidity and SSC conducted on stormflow and baseflow data
[6]Linear regression between turbidity and SSC conducted on baseflow data only, all other comparisons were determined by baseflow data only.

Table 3.10. Descriptive statistics for physiochemical and biological parameters by land use categories (urban, forested, and mixed).

	Forested Sites				Mixed Sites				Urban Sites			
Parameter	**Min**	**Max**	**Median**	**Mean**	**Min**	**Max**	**Median**	**Mean**	**Min**	**Max**	**Median**	**Mean**
Drainage Basin Area	7.9	171.0	37.0	52.8	8.0	161.0	51.2	54.2	6.3	107.0	35.0	39.5
Water Quality												
SpC (µs/cm)	15.6	154.0	51.8	59.7	32.8	155.5	54.5	62.3	62.3	1480	1449	313.8
pH	6.2	7.0	6.6	6.6	6.1	6.9	6.6	6.6	6.2	7.6	7.0	6.9
Turbidity (NTU)	3.9	16.9	8.2	9.1	5.3	17.5	10.8	10.5	5.8	13.8	7.9	8.8
SSC (mg/L)	1.3	13.8	5.6	5.6	2.2	13.4	5.7	6.4	2.4	7.8	6.0	5.7
Percent Fines												
Particle size distribution (%)	3.6	98.8	68.9	55.7	1.1	98.0	83.0	62.5	0.4	97.2	64.2	62.4
Pebble count distribution (%)	10.5	81.0	37.5	43.3	2.0	85.5	59.5	50.5	8.0	92.5	61.0	58.9
Fish Metrics												
Native species (Ct)	4.0	35.0	19.0	17.5	5.0	25.0	14.5	15.2	7.0	17.0	9.5	11.3
Benthic invertivores (Ct)	0	5.0	2.5	2.4	0	3.0	2.0	1.9	0	5.0	0	1.0
Sensitive species (Ct)	0	6.0	2.0	2.1	0	5.0	2.0	1.9	0	4.3	1.0	1.3
Catch per unit effort (CPUE)	78.9	1176	618.3	633.6	86.5	785.3	417.2	443.1	25.2	491.9	171.5	202.6
IBI score	16.0	54.0	45.0	39.8	14.0	48.0	36.0	32.8	14.0	32.0	22.0	22.8
IWB score	3.3	10.2	8.1	7.6	2.2	8.8	7.8	7.2	3.8	7.1	5.4	5.4
Macroinvertebrate Metrics												
EPT taxa richness	2.0	21.0	11.5	10.9	2.0	14	10.5	9.0	2.0	8	5	4.6
Taxa richness	11.0	35.0	26.8	24.9	12.0	32.0	22.0	22.4	11.0	23.0	19	17.8
Coleoptera taxa	1.0	6.0	3.7	3.7	1.0	6.0	3.5	3.6	2.0	5.0	2.0	3.2
Percent Oligochaeta	0	3.3	0.8	1.0	0.2	5.7	0.9	1.6	0.47	3.5	1.7	1.8
Percent Plecoptera	0	15.4	5.3	5.4	0	15.0	2.4	4.7	0	5.2	0	0.96
Shredder taxa	2.0	6.0	3.5	3.4	0.5	7.0	3.5	3.2	1.0	4.0	2.0	2.0
Scraper taxa	2.0	13.0	8.6	8.4	0	13.0	7.5	7.4	3.0	9.0	6.5	6.3
Swimmer taxa	1.0	5.0	3.0	2.8	1.0	4.5	3.0	2.8	0.5	3.5	1.5	1.6
MBI (Site index score)	25.5	73.0	55.5	53.3	22.0	67.5	52.5	49.0	25.5	46.5	39.8	37.9

Table 3.11. Means and standard errors of biotic community metrics for basins differing in dominant land cover.

Land use	**IBI**[1]	**IWB**[2]	**Total Taxa**[3] **(Ct)**	**EPT Taxa (Ct)**	**Macro Site Index**
Forest	40.75 (±2.95)a	7.81 (±0.35)a	25.76 (±1.57)a	11.41 (±1.42)a	54.91 (±2.37)a
Mixed	32.87 (±2.63)ab	7.23 (±0.43)a	22.35 (±1.48)a	9.04 (±1.08)a	49.00 (±3.27)b
Urban	22.75 (±2.47)b	5.39 (±0.38)a	17.75 (±1.70)a	4.65 (±0.92)b	37.93 (±2.87)b

[1]Values are means (±SE) of means for each stream within a land-use class.
[2]Within a column means values with same letter are not significantly different (Kruskal-Wallis with Dunn's post *hoc test*; $p < 0.05$).
[3]Macroinvertebrate data were normally distributed and analyzed using an ANOVA; Holm-Sidak was used to separate means; $p<0.05$).

Table 3.12. Relationship between biotic parameters and land use (percent urban and percent forest).

	Watershed Land use[1]			7.62 m Buffer[2]		
Relationship	**R-square**	**Slope[3]**	**p-value[4]**	**R-square**	**Slope[3]**	**p-value[4]**
Urban vs.						
IBI	**0.245**	**(-)**	**<0.001**	**0.125**	**(-)**	**0.023**
IWB	**0.130**	**(-)**	**0.019**	0.083		0.067
EPT	**0.172**	**(-)**	**0.006**	**0.163**	**(-)**	**0.009**
Total Taxa	**0.174**	**(-)**	**0.006**	**0.165**	**(-)**	**0.008**
Invert Index	**0.218**	**(-)**	**0.002**	**0.220**	**(-)**	**0.002**
Impervious vs.						
IBI	**0.198**	**(-)**	**0.003**	**0.145**	**(-)**	**0.014**
IWB	**0.101**	**(-)**	**0.040**	0.063		0.114
EPT	**0.160**	**(-)**	**0.009**	**0.142**	**(-)**	**0.015**
Total Taxa	**0.142**	**(-)**	**0.014**	**0.111**	**(-)**	**0.034**
Invert Index	**0.176**	**(-)**	**0.006**	**0.137**	**(-)**	**0.017**
Forest vs.						
IBI	**0.192**	**(+)**	**0.004**	**0.209**	**(+)**	**0.003**
IWB	**0.115**	**(+)**	**0.028**	**0.187**	**(+)**	**0.005**
EPT	**0.156**	**(+)**	**0.010**	**0.423**	**(+)**	**<0.001**
Total Taxa	**0.172**	**(+)**	**0.006**	**0.310**	**(+)**	**<0.001**
Invert Index	**0.136**	**(+)**	**0.016**	**0.205**	**(+)**	**0.003**

[1]Land use determined for entire watershed;
[2]Land use determined within 7.62 m buffer (25 ft buffer required by state law)
[3]Slope direction; indicates positive or negative relationship between variables. Included for significant relationships only.
[4]Power of performed test with alpha = 0.050 *(Note: Bold values are significant)*
[5]Linear regression between turbidity and SSC conducted on stormflow and baseflow data
[6]Linear regression between turbidity and SSC conducted on baseflow data only, all other comparisons were determined by baseflow data only.

Table 3.13. Models explaining biotic variability based on R-square values, Mallows' Cp, and Akaike Information Criterion.

Parameter	Variables	R-Square	Cp	S	AIC	AIC_c	Δ_i	w_i
Fish Metrics								
Native species (Ct)	DBA + D84	0.369	-1.99	5.43	145.08	146.16	0	0.94
	DBA + Mean PS	0.281	2.75	5.80	150.58	151.66	5.5	0.06
Benthic invertivores (Ct)	Impervious (%) + PC (<2) + DBA	0.494	5.81	1.09	11.34	13.01	0	0.88
	Impervious (%) + PC (<2)	0.409	10.48	1.17	15.85	16.93	3.92	0.12
Sensitive species (Ct)	D-84 + % Forest	0.355	1.91	1.38	30.60	31.68	0	0.54
	PSA (<2) + % Forest	0.350	2.21	1.39	30.91	31.99	0.31	0.46
Catch per unit effort (CPUE)	Urban (%)	0.267	-5.02	266.22	471.03	471.66	0	0.64
	Forest (%) + Ag (%)	0.289	-4.03	263.51	471.74	472.82	1.16	0.36
DELTs	Urban (%)	0.261	3.76	0.50	-56.05	-55.41	0	0.30
	Urban (%) + pH	0.331	-1.81	0.48	-58.24	-57.15	-1.74	0.70
IBI score	Urban (%) + PSA (<2)	0.375	-2.44	9.98	196.17	197.25	0	0.70
	Urban (%) + D84	0.316	-1.07	10.18	197.85	198.93	1.68	0.30
IWB score	PSA (<2) + Urban (%) + DBA	0.423	0.049	1.39	31.38	33.04	0	0.87
	PSA (<2) + Urban (%)	0.327	3.69	1.48	35.83	36.92	3.86	0.13
Macroinvertebrate Metrics								
EPT taxa richness	D84 + Turbidity + Urban (%) + DBA	0.595	10.48	3.61	112.54	114.94	0	0.56
	Urban (%) + Turbidity + PSA (<2)	0.562	11.92	3.70	113.79	115.46	0.52	0.44
Taxa richness	SSC + Urban (%) + PSA (<2) + DBA	0.581	14.49	4.56	132.23	134.63	0	0.94
	Turbidity + Urban (%) + PSA (<2)	0.480	22.65	4.97	138.53	140.20	5.57	0.06
Coleoptera taxa	PSA (<2) + DBA	0.212	8.17	1.22	19.89	20.52	0	0.73
	SSC	0.134	10.54	1.26	21.85	22.48	1.96	0.27

Table 3.13 (continued). Models explaining biotic variability based on R-square values, Mallows' Cp, and Akaike Information Criterion.

Percent Oligochaeta	Turbidity	0.449	2.60	1.42	31.54	32.17	0	0.84
	SSC	0.402	6.01	1.47	34.92	35.55	3.38	0.16
Percent Plecoptera	Urban (%) + DBA	0.205	-0.23	4.10	121.52	122.60	0	0.82
	pH	0.094	4.32	2.78	125.03	125.67	3.06	0.18
Shredder taxa	Urban (%) + DBA	0.179	2.29	1.48	35.65	36.73	0	0.66
	Impervious (%)	0.101	3.89	1.52	37.43	38.06	1.33	0.34
Scraper taxa	Urban + Turbidity	0.208	8.24	2.47	78.02	79.10	0	0.80
	Forest (%)	0.103	10.95	2.57	81.29	81.92	2.82	0.20
Swimmer taxa	Impervious (%)	0.205	4.78	0.92	-4.69	-4.06	0	0.30
	Urban (%)	0.255	3.08	0.91	-6.41	-5.78	-1.72	0.70
MBI (Site Index score)	Urban (%) + SSC + DBA	0.409	19.56	9.96	196.91	198.58	0	0.83
	Urban (%) + SSC	0.323	25.35	10.52	200.61	201.69	3.11	0.17

Table 3.14. Pearson product moment correlation among fish indices, macroinvertebrate indices and the metrics that comprise each index for streams within the Piedmont ecoregion of Georgia. Relationships between variables with p-values less than 0.050 are significant and are in bold; pairs with positive coefficients tend to increase together and for those with negative coefficients, one variable tends to increase while the other decreases. Note: The only relationships tested for significance were between fish indices and macroinvertebrate indices; relationships between indices and the individual metrics used to create them were not tested.

	IBI Score	IWB Score	MBI	EPT Taxa	Native Sp	Benthic Invertivore	Sensitive Species	Insect Cyprinids	Sucker Species	CPUE	DELTs	No of Taxa	Plecoptera (%)	Oligochaeta (%)	Coleoptera Taxa	Shredder Taxa	Swimmer Taxa	Scrapper Taxa
IBI Score	1																	
IWB Score	0.867	1																
MBI	**0.543**	**0.630**	1															
EPT Taxa	**0.625**	**0.711**	0.853	1														
Native Species	0.735	0.830	**0.482**	**0.601**	1													
Benthic Invertivore	0.742	0.746	**0.544**	**0.702**	0.713	1												
Sensitive Sp	0.743	0.692	**0.470**	**0.577**	0.726	0.708	1											
Insect Cyprinids	0.783	0.821	**0.509**	**0.524**	0.845	0.598	0.714	1										
Sucker Species	0.540	0.656	**0.351**	**0.537**	0.810	0.566	0.583	0.531	1									
CPUE	0.688	0.624	**0.535**	**0.601**	0.521	0.450	0.506	0.622	0.294	1								
DELTs	-0.432	-0.311	-0.291	-0.231	-0.222	-0.348	-0.355	-0.272	-0.046	-0.310	1							
No of Taxa	**0.585**	**0.698**	0.876	0.907	0.559	0.651	0.530	0.529	0.443	0.574	-0.281	1						
Plecoptera (%)	**0.422**	**0.503**	0.705	0.570	0.419	0.317	0.358	0.531	0.294	0.433	-0.053	0.519	1					
Oligochaeta (%)	-0.192	-0.205	-0.266	-0.389	-0.243	-0.155	-0.115	-0.207	-0.217	-0.116	0.029	-0.340	-0.060	1				
Coleoptera Taxa	**0.290**	**0.443**	0.703	0.483	0.334	0.322	0.291	0.275	0.232	0.256	-0.349	0.621	0.228	-0.165	1			
Shredder Taxa	**0.377**	**0.432**	0.765	0.627	0.354	0.365	0.210	0.369	0.183	0.322	-0.160	0.690	0.684	-0.194	0.450	1		
Swimmer Taxa	**0.553**	**0.528**	0.530	0.452	0.318	0.411	0.319	0.413	0.179	0.355	-0.330	0.484	0.216	-0.194	0.383	0.400	1	
Scrapper Taxa	**0.560**	**0.621**	0.775	0.670	0.487	0.444	0.439	0.556	0.303	0.544	-0.265	0.831	0.491	-0.320	0.644	0.631	0.403	1

Table 3.15. Pearson product moment correlation among fish indices and the metrics that comprise each index for streams within the Piedmont ecoregion of Georgia. Relationships between variables with p-values less than 0.050 are significant and are in bold; pairs with positive coefficients tend to increase together and for those with negative coefficients, one variable tends to increase while the other decreases. Note: The only relationships tested for significance were highly correlative and were between fish indices and macroinvertebrate indices; relationships between indices and the individual metrics used to create them were not tested.

	IBI Score	IWB Score	Native Sp	Benthic Invertivore	Sensitive Species	Insect Cyprinids	Sucker Species	CPUE	DELTs (Ct)	Urban (%)	Forest (%)	Impervious (%)	Agriculture (%)	Turbidity (NTU)	SSC (mgL^{-1})
IBI Score	1														
IWB Score	0.867	1													
Native Species	0.735	0.830	1												
Benthic Invertivore	0.742	0.747	0.713	1											
Sensitive Sp	0.743	0.692	0.726	0.708	1										
Insect Cyprinids	0.783	0.821	0.845	0.598	0.714	1									
Sucker Species	0.540	0.656	0.810	0.566	0.583	0.531	1								
CPUE	0.688	0.624	0.521	0.450	0.506	0.622	0.294	1							
DELTs	-0.432	-0.311	-0.222	-0.348	-0.355	-0.272	-0.046	-0.310	1						
Urban (%)	**-0.533**	-0.335	-0.319	-0.393	-0.392	-0.445	-0.193	-0.489	-0.189	1					
Forest (%)	0.430	0.335	0.250	0.460	0.430	0.347	0.167	0.426	-0.277	0.648	1				
Impervious (%)	-0.441	-0.223	-0.192	-0.332	-0.323	-0.352	-0.018	-0.439	0.344	-0.964	-0.713	1			
Agriculture (%)	0.124	0.126	0.028	-0.053	-0.119	0.079	-0.028	0.131	-0.320	0.023	-0.439	-0.024	1		
Turbidity	-0.248	-0.392	-0.328	-0.447	-0.394	-0.109	-0.420	-0.123	-0.058	-0.097	-0.201	-0.040	0.291	1	
SSC	-0.250	-0.436	-0.354	**-0.520**	-0.377	-0.147	-0.421	-0.155	0.086	-0.014	-0.252	-0.026	0.302	0.830	1
SpC	**0.624**	**0.703**	**0.525**	**0.629**	**0.552**	**0.513**	0.395	0.421	-0.464	0.416	0.484	-0.379	0.146	-0.426	-0.392
pH	-0.327	-0.244	-0.060	-0.123	-0.070	-0.187	-0.044	-0.185	0.480	-0.238	-0.232	0.311	-0.346	-0.264	-0.133
Fines - PSA (%)	-0.344	-0.424	-0.337	-0.497	-0.448	-0.121	-0.447	-0.170	0.185	-0.104	-0.135	0.046	0.066	0.457	0.549
Fines - PC (%)	-0.384	-0.376	-0.359	**-0.560**	-0.452	-0.161	-0.402	-0.201	0.107	-0.296	-0.274	0.262	0.166	0.501	0.546
D_{50}	-0.384	-0.379	-0.338	**-0.577**	-0.485	-0.135	-0.359	-0.197	0.261	-0.236	-0.309	0.259	0.137	0.421	0.515
D_{84}	-0.264	-0.285	-0.333	-0.415	-0.279	-0.190	-0.276	-0.120	0.007	-0.072	-0.085	0.052	0.065	0.115	0.214
D_g	-0.368	-0.364	-0.337	**-0.548**	-0.469	-0.130	-0.363	-0.186	0.181	-0.249	-0.269	0.248	0.136	0.451	0.506
DBA	-0.061	-0.354	**-0.522**	-0.316	-0.162	-0.459	-0.414	-0.150	-0.169	-0.039	-0.022	-0.071	0.073	0.143	0.138
Canopy Cover (%)	0.224	0.070	-0.002	0.109	0.003	0.011	-0.037	0.151	**-0.507**	0.320	0.317	-0.479	0.360	0.165	0.086
SWCt	-0.318	-0.245	-0.296	-0.420	-0.411	-0.134	-0.392	-0.192	0.152	-0.335	-0.309	0.321	0.113	0.519	0.426
SWVol	-0.234	-0.132	-0.243	-0.321	-0.296	-0.046	-0.372	-0.078	0.102	-0.207	-0.212	0.166	0.175	0.391	0.350
Woody Debris (Ct)	0.017	0.200	0.317	0.144	0.092	0.245	0.313	0.118	-0.125	-0.371	0.013	0.283	0.004	0.133	-0.049
Woody Debris (m^{-1})	-0.070	0.029	0.073	-0.069	-0.048	0.114	0.043	0.062	-0.190	-0.443	-0.041	0.254	0.082	0.364	0.140
Bankfull Area	0.086	0.347	0.475	0.329	0.287	0.277	**0.555**	0.096	0.063	-0.109	-0.056	0.271	-0.141	-0.484	-0.392
Wetted Perimeter (AC)	0.113	0.373	**0.525**	0.358	0.289	0.375	0.585	0.262	0.091	-0.006	0.055	0.157	-0.173	-0.387	-0.393
Gradient	0.412	0.281	0.222	0.369	0.330	0.090	0.253	0.137	-0.253	0.231	0.155	-0.264	0.031	-0.224	-0.221
Power Index	**0.526**	**0.584**	**0.594**	**0.647**	0.467	0.419	**0.641**	0.223	-0.128	0.227	0.184	-0.155	-0.004	-0.408	-0.359
Width-Depth Ratio	0.161	0.289	0.347	0.241	0.305	0.354	0.353	0.273	0.069	0.061	-0.007	0.041	0.042	-0.193	-0.336
Habitat	0.464	0.405	0.411	**0.570**	**0.535**	0.253	0.467	0.291	-0.261	0.376	0.340	-0.357	-0.027	-0.388	-0.512

Table 3.15 (continued). Pearson product moment correlation among fish indices and the metrics that comprise each index for streams within the Piedmont ecoregion of Georgia. Relationships between variables with p-values less than 0.050 are significant and are in bold; pairs with positive coefficients tend to increase together and for those with negative coefficients, one variable tends to increase while the other decreases.

	SpC	pH	Fines - PSA (%)	Fines – PC (%)	D_{50}	D_{84}	D_g	DBA	Canopy Cover (%)	SWCt	SWVol	Woody Debris (Ct)	Woody Debris (Ct m^{-1})	Bankfull Area (m^2)	Wetted Perimeter (AC)	Gradient	Power Index	Width-Depth Ratio	Habitat
SpC	1																		
pH	-0.560	1																	
Fines - PSA (%)	-0.444	-0.043	1																
Fines - PC (%)	-0.422	-0.121	0.833	1															
D_{50}	-0.442	-0.012	0.836	0.908	1														
D_{84}	-0.162	-0.186	0.451	0.637	0.588	1													
D_g	-0.424	-0.090	0.858	0.959	0.977	0.654	1												
DBA	-0.029	-0.172	-0.063	-0.076	-0.057	0.000	-0.082	1											
Canopy Cover (%)	0.328	-0.406	-0.038	-0.099	-0.142	0.059	-0.114	0.300	1										
SWCt	-0.392	0.041	0.556	0.624	0.638	0.219	0.647	-0.012	-0.172	1									
SWVol	-0.254	-0.024	0.532	0.509	0.528	0.261	0.536	-0.052	-0.092	0.809	1								
Woody Debris (Ct)	0.101	0.051	-0.118	0.152	0.074	0.048	0.111	-0.417	-0.002	0.064	-0.142	1							
Woody Debris (m^{-1})	0.017	-0.040	-0.053	0.228	0.138	0.108	0.172	-0.113	0.126	0.142	-0.093	0.910	1						
Bankfull Area (m^2)	0.293	0.174	-0.339	-0.233	-0.134	-0.005	-0.165	-0.534	-0.316	-0.162	-0.103	0.350	0.100	1					
Wetted Perimeter (AC)	0.195	0.188	-0.207	-0.126	-0.085	-0.041	-0.088	-0.741	-0.369	-0.143	-0.104	0.520	0.195	0.781	1				
Gradient	0.385	-0.201	-0.574	-0.693	-0.786	-0.481	-0.780	0.287	0.233	-0.620	-0.467	-0.292	-0.264	-0.138	-0.218	1			
Power Index	0.465	-0.161	-0.521	-0.570	-0.638	-0.401	-0.616	-0.343	0.000	-0.563	-0.404	0.002	-0.192	0.299	0.286	0.687	1		
Width-Depth Ratio	0.094	0.214	-0.163	-0.180	-0.141	-0.237	-0.153	-0.350	-0.272	-0.106	-0.057	0.320	0.144	0.204	0.606	-0.001	0.108	1	
Habitat	0.446	-0.146	-0.661	-0.676	-0.689	-0.319	-0.686	-0.032	0.176	-0.686	-0.678	0.064	-0.031	0.087	0.248	0.516	0.465	0.388	1.000

Table 3.16. Pearson product moment correlation among macroinvertebrate indices and the metrics that comprise each index for streams within the Piedmont ecoregion of Georgia. Relationships between variables with p-values less than 0.050 are significant and are in bold; pairs with positive coefficients tend to increase together and for those with negative coefficients, one variable tends to increase while the other decreases.

	MBI (Site index score)	EPT Taxa	No of Taxa	Plecoptera (%)	Oligochaeta (%)	Coleoptera Taxa	Shredder Taxa	Swimmer Taxa	Scrapper Taxa	Urban (%)	Forest (%)	Impervious (%)	Agriculture (%)	Turbidity (NTU)	SSC (mgL^{-1})
MBI (Site index score)	1.000														
EPT Taxa	0.853	1.000													
No of Taxa	0.876	0.907	1.000												
Plecoptera (%)	0.705	0.570	0.519	1.000											
Oligochaeta (%)	-0.266	-0.389	-0.340	-0.060	1.000										
ColeopteraTaxa	0.703	0.483	0.621	0.228	-0.165	1.000									
ShredderTaxa	0.765	0.627	0.690	0.684	-0.194	0.450	1.000								
SwimmerTaxa	0.530	0.452	0.484	0.216	-0.194	0.383	0.400	1.000							
ScrapperTaxa	0.775	0.670	0.831	0.491	-0.320	0.644	0.631	0.403	1.000						
Urban (%)	0.447	0.488	0.402	0.386	-0.244	0.037	0.237	0.375	0.265	1.000					
Forest (%)	0.373	0.401	0.416	0.299	-0.162	0.101	0.262	0.299	0.315	0.648	1.000				
Impervious (%)	-0.378	-0.367	-0.303	-0.305	0.216	-0.027	-0.208	-0.337	-0.177	-0.964	-0.713	1.000			
Agriculture (%)	0.212	0.116	0.112	0.092	-0.063	0.110	0.155	0.375	0.110	0.023	-0.439	-0.024	1.000		
Turbidity (NTU)	-0.335	**-0.594**	**-0.504**	-0.122	0.423	-0.285	-0.163	-0.111	-0.336	-0.097	-0.201	-0.040	0.291	1.000	
SSC (mgL^{-1})	-0.413	**-0.586**	**-0.578**	-0.175	0.314	-0.437	-0.271	-0.176	-0.373	-0.014	-0.252	-0.026	0.302	0.830	1.000
SpC ($\mu s\ cm^{-1}$)	0.492	**0.619**	**0.543**	0.421	-0.326	0.171	0.292	0.492	0.423	0.416	0.484	-0.379	0.146	-0.426	-0.392
pH	-0.239	-0.142	-0.111	-0.328	0.035	0.023	-0.243	-0.435	-0.021	-0.238	-0.232	0.311	-0.346	-0.264	-0.133
Fines - PSA (%)	-0.234	-0.429	-0.393	0.099	0.186	-0.310	0.080	-0.115	-0.192	-0.104	-0.135	0.046	0.066	0.457	0.549
Fines - PC (%)	-0.172	-0.448	-0.323	0.074	0.346	-0.130	0.060	-0.073	-0.119	-0.296	-0.274	0.262	0.166	0.501	0.546
D_{50}	-0.252	-0.482	-0.408	0.058	0.268	-0.208	-0.080	-0.074	-0.199	-0.236	-0.309	0.259	0.137	0.421	0.515
D_{84}	-0.122	-0.243	-0.212	0.049	0.074	-0.076	-0.021	0.000	-0.149	-0.072	-0.085	0.052	0.065	0.115	0.214
D_g	-0.212	-0.456	-0.367	0.081	0.285	-0.168	-0.008	-0.055	-0.157	-0.249	-0.269	0.248	0.136	0.451	0.506
DBA	-0.339	-0.331	-0.361	-0.333	0.110	-0.274	-0.274	0.070	-0.262	-0.039	-0.022	-0.071	0.073	0.143	0.138
SWCt	0.057	-0.013	-0.012	-0.032	-0.147	0.058	0.032	0.243	0.009	0.320	0.317	-0.479	0.360	0.165	0.086
SWVol	-0.152	-0.403	-0.362	0.085	0.431	0.004	0.064	-0.071	-0.121	-0.335	-0.309	0.321	0.113	0.519	0.426
Canopy Cover (%)	-0.064	-0.285	-0.288	0.107	0.372	0.029	0.099	0.050	-0.153	-0.207	-0.212	0.166	0.175	0.391	0.350
Woody Debris (Ct)	0.100	0.041	0.188	0.177	0.075	0.181	0.117	-0.108	0.228	-0.371	0.013	0.283	0.004	0.133	-0.049
Woody Debris (m^{-1})	-0.075	-0.181	-0.004	0.062	0.166	0.033	0.011	-0.148	0.105	-0.443	-0.041	0.254	0.082	0.364	0.140
Bankfull Area	0.204	0.361	0.240	0.267	-0.083	0.155	0.058	-0.111	0.107	-0.109	-0.056	0.271	-0.141	-0.484	-0.392
Wetted Perimeter (AC)	0.359	0.471	0.421	0.343	-0.064	0.281	0.151	-0.067	0.309	-0.006	0.055	0.157	-0.173	-0.387	-0.393
Gradient	0.027	0.253	0.155	-0.142	-0.328	-0.071	-0.059	0.119	-0.001	0.231	0.155	-0.264	0.031	-0.224	-0.221
Power Index	0.345	**0.573**	0.463	0.165	-0.440	0.137	0.225	0.120	0.239	0.227	0.184	-0.155	-0.004	-0.408	-0.359
Width-Depth Ratio	0.276	0.339	0.408	0.212	-0.052	0.220	0.055	0.133	0.404	0.061	-0.007	0.041	0.042	-0.193	-0.336
Habitat	0.377	**0.562**	**0.530**	0.108	-0.232	0.256	0.059	0.244	0.292	0.376	0.340	-0.357	-0.027	-0.388	-0.512

Table 3.16 (continued). Pearson product moment correlation among macroinvertebrate indices and the metrics that comprise each index for streams within the Piedmont ecoregion of Georgia. Relationships between variables with p-values less than 0.050 are significant and are in bold; pairs with positive coefficients tend to increase together and for those with negative coefficients, one variable tends to increase while the other decreases.

	SpC	pH	Fines - PSA (%)	Fines – PC (%)	D_{50}	D_{84}	D_g	DBA	Canopy Cover (%)	SWCt	SWVol	Woody Debris (Ct)	Woody Debris (Ct m^{-1})	Bankfull Area (m^2)	Wetted Perimeter (AC)	Gradient	Power Index	Width-Depth Ratio	Habitat
SpC	1.000																		
pH	-0.560	1.000																	
Fines - PSA (%)	-0.444	-0.043	1.000																
Fines - PC (%)	-0.422	-0.121	0.833	1.000															
D_{50}	-0.442	-0.012	0.836	0.908	1.000														
D_{84}	-0.162	-0.186	0.451	0.637	0.588	1.000													
D_g	-0.424	-0.090	0.858	0.959	0.977	0.654	1.000												
DBA	-0.029	-0.172	-0.063	-0.076	-0.057	0.000	-0.082	1.000											
SWCt	0.328	-0.406	-0.038	-0.099	-0.142	0.059	-0.114	0.300	1.000										
SWVol	-0.392	0.041	0.556	0.624	0.638	0.219	0.647	-0.012	-0.172	1.000									
Canopy Cover (%)	-0.254	-0.024	0.532	0.509	0.528	0.261	0.536	-0.052	-0.092	0.809	1.000								
Woody Debris (Ct)	0.101	0.051	-0.118	0.152	0.074	0.048	0.111	-0.417	-0.002	0.064	-0.142	1.000							
Woody Debris (m^{-1})	0.017	-0.040	-0.053	0.228	0.138	0.108	0.172	-0.113	0.126	0.142	-0.093	0.910	1.000						
Bankfull Area (m^2)	0.293	0.174	-0.339	-0.233	-0.134	-0.005	-0.165	-0.534	-0.316	-0.162	-0.103	0.350	0.100	1.000					
Wetted Perimeter (AC)	0.195	0.188	-0.207	-0.126	-0.085	-0.041	-0.088	-0.741	-0.369	-0.143	-0.104	0.520	0.195	0.781	1.000				
Gradient	0.385	-0.201	-0.574	-0.693	-0.786	-0.481	-0.780	0.287	0.233	-0.620	-0.467	-0.292	-0.264	-0.138	-0.218	1.000			
Power Index	0.465	-0.161	-0.521	-0.570	-0.638	-0.401	-0.616	-0.343	0.000	-0.563	-0.404	0.002	-0.192	0.299	0.286	0.687	1.000		
Width-Depth Ratio	0.094	0.214	-0.163	-0.180	-0.141	-0.237	-0.153	-0.350	-0.272	-0.106	-0.057	0.320	0.144	0.204	0.606	-0.001	0.108	1.000	
Habitat	0.446	-0.146	-0.661	-0.676	-0.689	-0.319	-0.686	-0.032	0.176	-0.686	-0.678	0.064	-0.031	0.087	0.248	0.516	0.465	0.388	1.000

Table 3.17. Principal components on correlations for all variables in dataset.

Number	**Eigenvalue**	**Percent**	**Cumulative Percent**
1	8.9286	30.79	30.79
2	4.4871	15.47	46.26
3	3.7556	12.95	59.21
4	2.0586	7.10	66.31
5	1.6113	5.56	71.87
6	1.3089	4.51	76.38
7	1.0796	3.72	80.10
8	0.9362	3.23	83.33
9	0.9189	3.17	86.50
10	0.7473	2.58	89.08
11	0.6286	2.17	91.24
12	0.6172	2.13	93.37
13	0.3623	1.25	94.62
14	0.3551	1.22	95.85
15	0.2624	0.90	96.75
16	0.2225	0.77	97.52
17	0.1493	0.51	98.03
18	0.1370	0.47	98.51
19	0.1043	0.36	98.87
20	0.0854	0.29	99.16
21	0.0724	0.25	99.41
22	0.0575	0.20	99.61
23	0.0379	0.13	99.74
24	0.0229	0.08	99.82
25	0.0213	0.07	99.89
26	0.0167	0.06	99.95
27	0.0071	0.02	99.97
28	0.0063	0.02	99.99
29	0.0012	0.01	100.00

Table 3.18. Summary of eigenvectors for PCA analysis on all factors within the dataset.

Factor	Factor Acronym	Eigenvalues Component 1	Component 2	Component 3	Component 4
Urban (%)	Urban	0.1299	0.3312	-0.2218	0.0033
Forest (%)	Forest	-0.1535	-0.1583	0.1224	-0.1371
Impervious	Imp	0.1263	0.3375	-0.2042	-0.0026
Agriculture (%)	Ag	0.01961	-0.1896	0.1178	0.1537
Turbidity (NTU)	Turb	0.1677	-0.2099	0.0747	0.4099
SSC mg/L	SSC	0.1765	-0.1992	0.0412	0.3528
SpC µs/cm	SpC	0.1049	0.1328	-0.1957	0.0060
pH	pH	0.0322	0.2978	-0.1855	-0.0774
Particle Size (%)	PSA	0.2632	-0.0700	0.1415	-0.1501
Pebble Ct	PC	0.2845	-0.0238	0.2006	-0.0639
D_{50}	D_{50}	0.2897	0.0177	0.1880	-0.1174
D_{84}	D_{84}	0.1745	-0.0452	0.1265	-0.2175
Mean Particle Size	D_g	0.2880	-0.0053	0.2141	-0.1197
Drainage Basin Area	DBA	-0.0292	0.2084	0.3106	0.0667
Woody Debris Ct	SWCt	0.2428	0.0095	0.1513	0.0214
Woody Debris Vol	SWVol	0.1945	-0.0343	0.1687	-0.1211
Canopy Cover (%)	CanCvr	-0.0492	-0.3297	0.0157	0.1093
Total Woody Debris	WoodyTot	0.0051	0.2421	0.2052	0.4292
Wood per meter	Woodpermet	0.0716	0.1517	0.1277	0.4798
Bankfull Area	WPBF	-0.1198	0.3271	0.1944	0.0261
Active Channel Area	WPAC	-0.1174	0.3060	0.2730	0.0159
Gradient (%)	Gradient	-0.2109	-0.1240	-0.2389	0.1243
Power Index	PowerIndex	-0.2403	0.0453	0.0091	0.1432
Width-Depth Ratio	WDRatio	-0.1121	0.2140	0.1631	0.0102
Habitat Score	Habitat	-0.2838	-0.0276	-0.0302	0.0171
Fish IBI Score	IBI	-0.2137	-0.1266	0.1845	0.0327
Fish IWB Score	IWB	-0.2239	0.0077	0.2576	0.0223
Benthic Index Score	MBI	-0.1826	-0.0388	0.3059	-0.1919
EPT Taxa	EPT	-0.2575	0.0164	0.2130	-0.1881

Table 3.19. Comparison of models replacing PC with PSA and turbidity with SSC.

Parameter	Variables	K	AIC	AIC_c	Δ_i	w_i	Δ_j
Fish Metrics							
Benthic invertivores (Ct)	Impervious (%) + PC (<2) + DBA	5	11.34	13.01	0.00	0.63	1.67
	Impervious (%) + PSA (<2) + DBA	5	12.37	14.04	1.03	0.37	
Sensitive species (Ct)	PSA (<2) + % Forest	4	30.91	31.99	0.00	0.79	3.80
	PC (<2) + % Forest	4	33.58	34.66	2.67	0.21	
IBI score	Urban (%) + PSA (<2)	4	196.17	197.25	0.00	0.70	2.36
	Urban (%) + PC (<2)	4	197.89	198.97	1.72	0.30	
IWB score	PSA (<2) + Urban (%) + DBA	5	31.38	33.05	0.00	0.95	19.79
	PC (<2) + Urban (%) + DBA	5	37.35	39.02	5.97	0.05	
	PSA (<2) + Urban (%)	4	35.83	36.91	0.00	0.93	13.26
	PC (<2) + Urban (%)	4	41.00	42.08	5.17	0.07	
Macroinvertebrate Metrics							
EPT taxa richness	D84 + Turbidity + Urban (%) + DBA	6	112.54	114.94	0.00	0.91	10.43
	D84 + SSC + Urban (%) + DBA	6	117.23	119.63	4.69	0.09	
	Urban (%) + Turbidity + PSA (<2)	5	113.79	115.46	0.00	0.99	75.57
	Urban (%) + SSC + PC (<2)	5	122.44	124.11	8.65	0.01	
Taxa richness	SSC + Urban (%) + DBA + PSA (<2)	6	132.23	134.63	0.00	0.97	37.15
	Turbidity + Urban (%) + DBA + PC (<2)	6	139.46	141.86	7.23	0.03	
	Turbidity + Urban (%) + PSA (<2)	5	138.53	140.20	0.00	0.95	17.99
	SSC + Urban (%) + PC (<2)	5	144.31	145.98	5.78	0.05	
Coleoptera taxa	SSC	3	21.85	22.48	0.00	0.79	3.88
	Turbidity	3	24.56	25.19	2.71	0.21	
	PSA (<2) + DBA	3	19.89	20.52	0.00	0.92	12.24

Table 3.19 (continued). Comparison of models replacing PC with PSA and turbidity with SSC.

Percent Oligochaeta	Turbidity	3	31.54	32.17	0.00	0.84	5.42
	TSS	3	34.92	35.55	3.38	0.16	
Scraper taxa	Urban + Turbidity	4	78.02	79.10	0.00	0.53	1.14
	Urban + SSC	4	78.29	79.37	0.27	0.47	
MBI (Site Index score)	Urban (%) + SSC + DBA	5	196.91	198.58	0.00	0.51	1.06
	Urban (%) + Turbidity + DBA	5	197.03	198.70	0.12	0.49	
	Urban (%) + SSC	4	200.61	201.69	0.00	0.53	1.14
	Urban (%) + Turbidity	4	200.87	201.95	0.26	0.47	

Table 3.20. Literature review of land use effects on aquatic communities 1981-2006.

Author	Year	Location	Number of Sites	Variable	Local or Regional Effect	Findings
Omernik et al.	1981	USA	904	Nutrients	Catchment	Proximity of forest and agriculture to main streams do not bear a significant relationship to stream nutrient concentrations
Frissell et al.	1986	USA	-	Streams	Both	Hierarchical framework for viewing streams from microhabitats to watershed geomorphic features.
Steedman, R.	1988	Southern Ontario	209	Fish	Local	The 10-100 km2 of drainage basin immediately above a station was most important in predicting stream quality
Richards et al.	1996	Saginaw Bay (Michigan)	45	Invert	Catchment	Results suggest that catchment-wide geology and land-use characteristics may be more important than stream buffers for maintaining or restoring stream ecosystems.
Roth et al.	1996	River Raisin (Michigan)	23	Fish	Catchment	Habitat index and IBI correlated strongly with regional land use throughout the catchment upstream of the site.
Allan et al.	1997	River Raisin (Michigan)	23	Fish	Local	Flow stability and percent forested land within 100-m buffer explained 44% of the variation in IBI scores.
Richards et al.	1997	Saginaw Bay (Michigan)	58	Invert	Local	Macroinvertebrate traits best related to reach-scale physical features. Effects of land use were masked by geology among the catchments.
Townsend et al.	1997	Taieri plain, South island of New Zealand	8	Invert	No scale effect	Community patterns were similar for watershed, subcatchment, and reach. Elevation, riffle length, large substrate, and phosphorus content were all related to land use.
Wang et al.	1997	Wisconsin	134	Fish	Catchment	Habitat scores and IBI were significantly correlated with amount of agriculture land in the entire watershed and in a 100-m buffer, but correlations were stronger for the entire watershed.
Wang and Yin	1997	Great Miami River (Ohio)	6	Conductivity	-	Conductivity increases in relationship to percent urbanized land, not agriculture, despite 5 of 6 catchments having ~80% agricultural land use.

Table 3.20 (continued). Literature review of land use effects on aquatic communities 1981-2009.

Vinson and Hawkins	1998	Review of Literature	-	Inverts	Local	Most consistent patterns of richness exist with substrate size, disturbance regime, predation, annual temperature range, flow intermittency, and biome type. Invertebrate richness is jointly structured by historical events and by the physical and chemical conditions unique to each location.
Barbour et al.	1999	USA	-	Fish and Inverts	Both	Fish species richness associated with land cover within the entire watershed; Macroinvertebrate richness associated with riparian corridor land cover.
Lammert and Allen	1999	Southeastern Michigan	6	Fish and Inverts	Local	Land use within 100-m of the stream was significantly related to biotic integrity.
Stauffer et al.	2000	Minnesota River Basin	20	Fish	Local and runoff potential	Streams with wooded riparian zones and low potential for runoff had higher IBI scores.
Stewart et al.	2000	Northwest Indiana	3	Inverts	Local	Macroinvertebrate communities are driven by in-stream habitat and local scale land use factors.
Van Sickle et al.	2000	Western Oregon	137	Fish	Catchment	Ecoregions and large catchments have utility for classifying stream vertebrate assemblages.
Leigh et al.	2001	Etowah River, Georgia	32	Fish and Inverts	Local	Stream-reach geomorphic and physical habitat measures from were more important than basin-wide land cover measures.
Wang et al.	2001	Southeastern Wisconsin	47	Fish	Local	Impervious surface within 100-m buffer is a better predictor than comparable levels further away. Above 12% watershed imperviousness non-urban influences are negated.
Stewart et al.	2001	Eastern Wisconsin	38	Fish Inverts	Both	Stream health was related to environmental factors at a variety of scales; land cover resolution needs to vary depending on the scale of analysis and the specific questions at hand.
Papanicolaou et al.	2003	Clearwater River basin (Idaho)	61	Fish	Both	Watershed and instream parameters affect the aquatic life. The most controlling factors for fish are water temperature, physiographic characteristics, watershed gradient, and river density.

Table 3.20(continued). Literature review of land use effects on aquatic communities 1981-2009.

Roy et al.	2003	Etowah River Watershed, Georgia	30	Inverts	Catchment	Catchment land use was related to taxon richness and other biotic indices. Urban land cover explained 29-38% of the variation. Reduced water quality was detectable at >15% urban land cover.
Snyder et al.	2003	Opequon Ck watershed, West Virginia	20	Fish	Catchment	Catchment patterns were more strongly correlated to biological integrity than riparian patterns, suggesting that forested buffers were of little value in mitigating deleterious effects of urbanization.
Strayer et al.	2003	Chesapeake Bay (eastern USA)	944 (fish) 269 (inverts)	Fish and Inverts	Both	Fish species richness associated with land cover within the entire watershed; Macroinvertebrate richness associated with riparian corridor land cover.
Weigel et al.	2003	Northern Lakes (MN, WI, MI)	94	Inverts	Both	Catchment and reach variables were equally influential in defining assemblage attributes. Reach scale was more important in determining relative abundance and presence/absence.
Urban et al.	2006	Rural-urban gradient	18	Inverts	Both	Local riparian vegetation and watershed landscape structure were best predictors of community diversity and abundance.
Walters et al.	2009	Etowah River Watershed, Georgia	31	Fish and Inverts	Both	Macroinvertebrate descriptors were better predicted by land cover, whereas fish descriptors were better predicted by geomorphology.

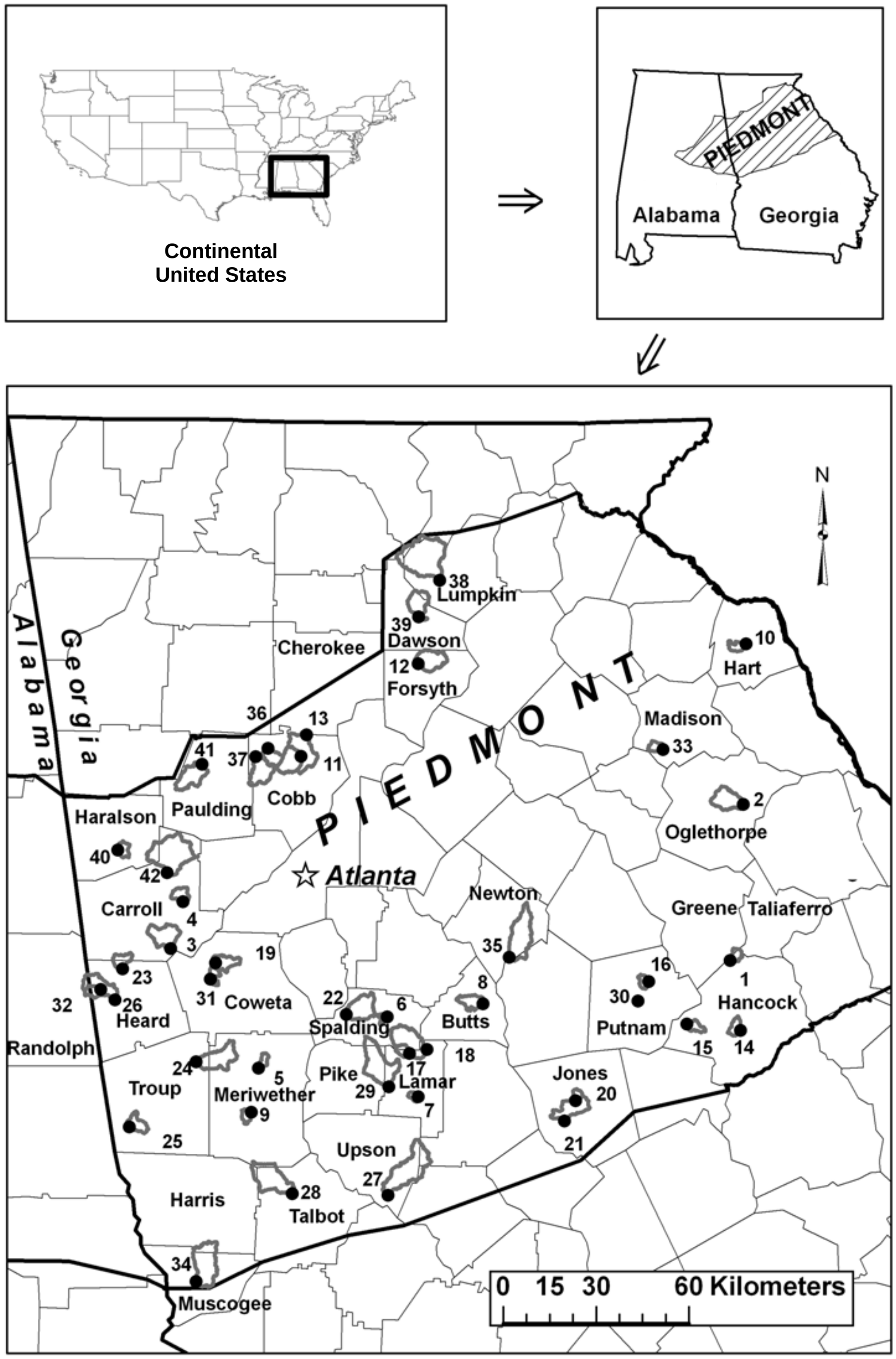
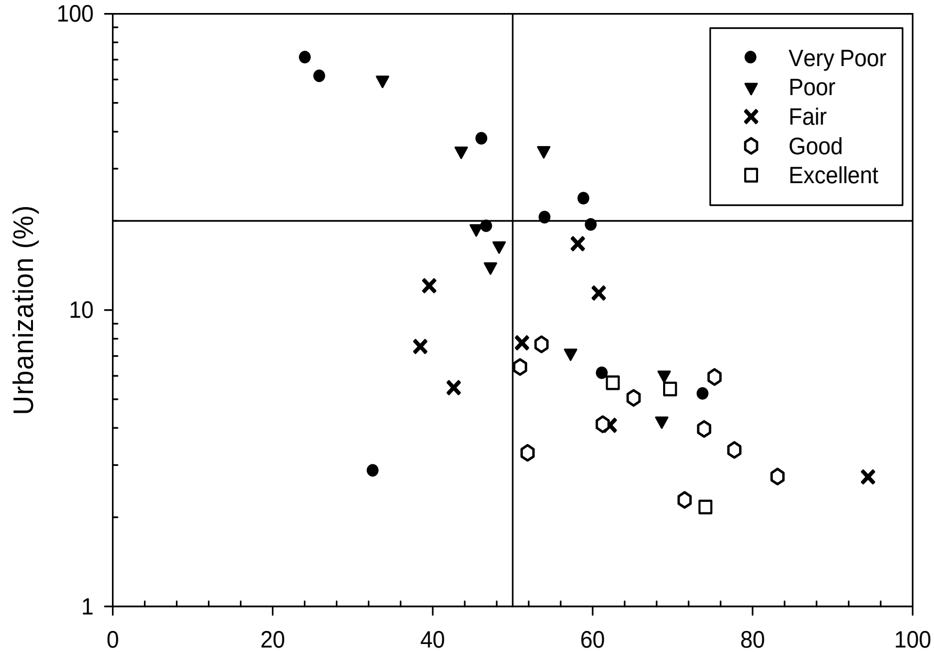


Figure 3.1. Locations of watersheds (outlined in black) and study sites (represented by black dots) within the Piedmont ecoregion of Georgia.

(A)



(B)

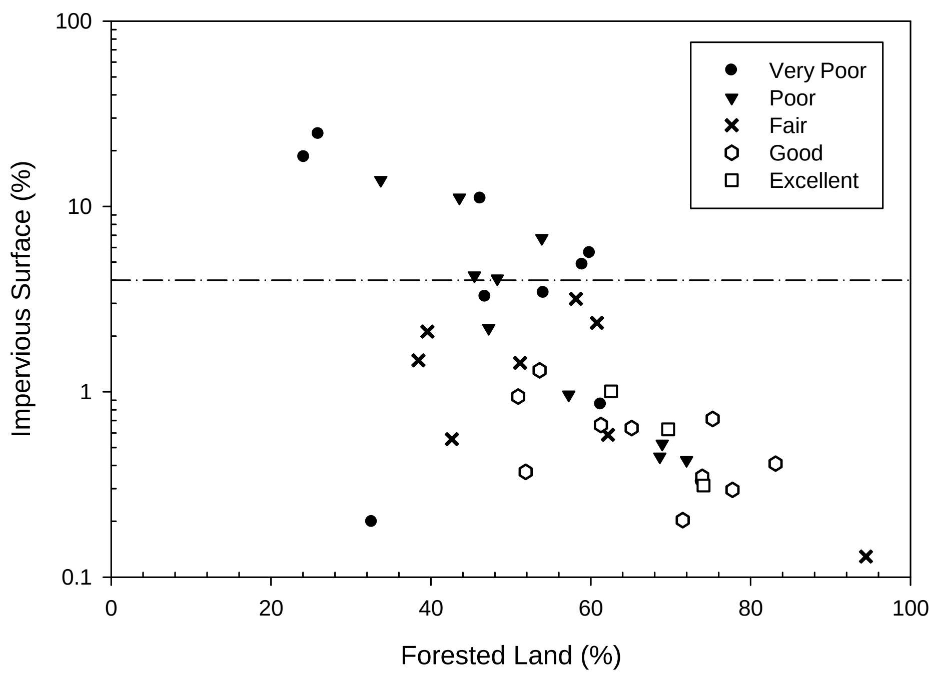


Figure 3.2. Relationship between fish IBI scores, forested land use, a) urbanization and b) impervious surface.

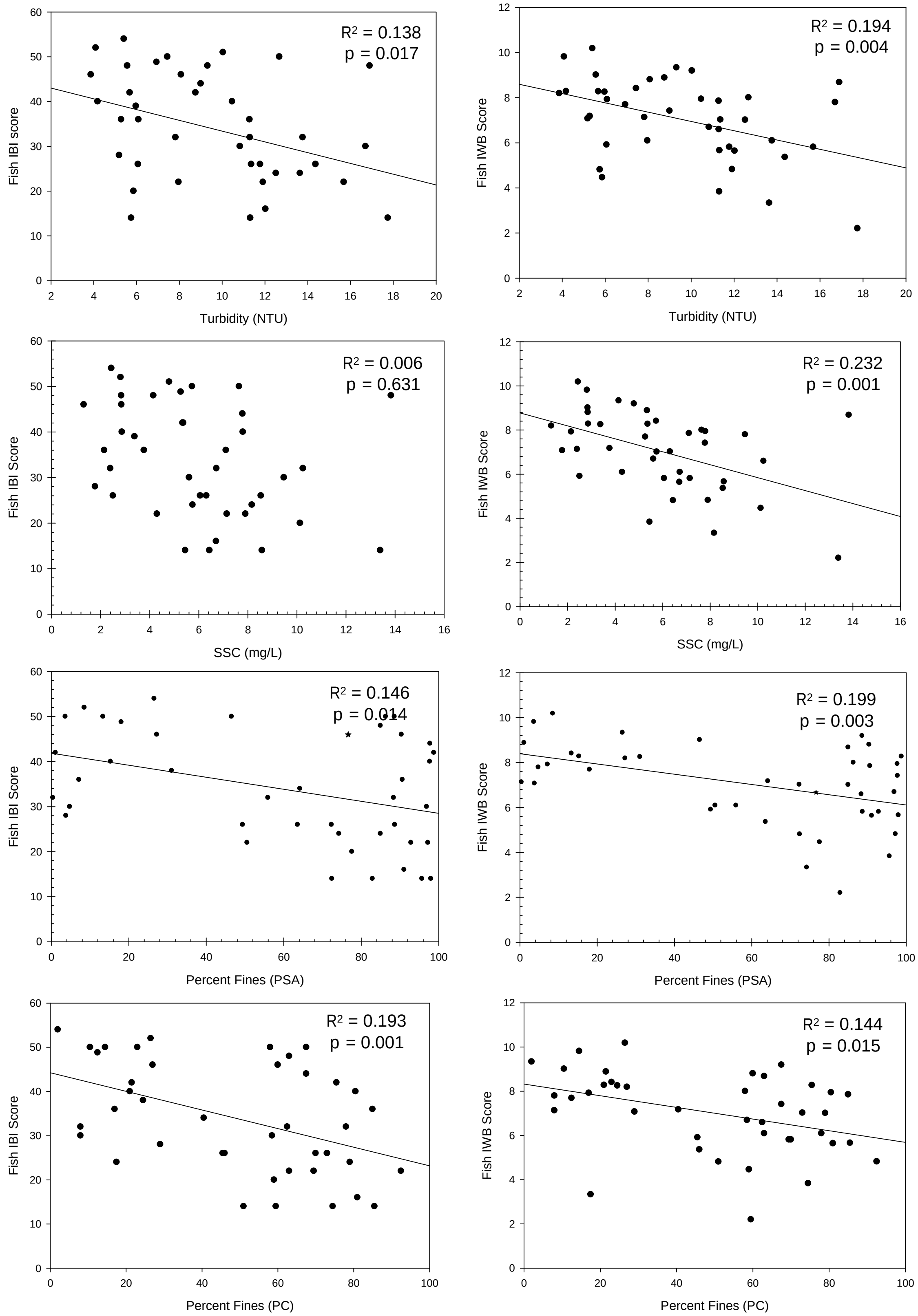


Figure 3.3. Comparison of fish IBI and IWB scores to water clarity (turbidity and suspended sediment concentration) and bed sediment metrics (particle size analysis and pebble count).

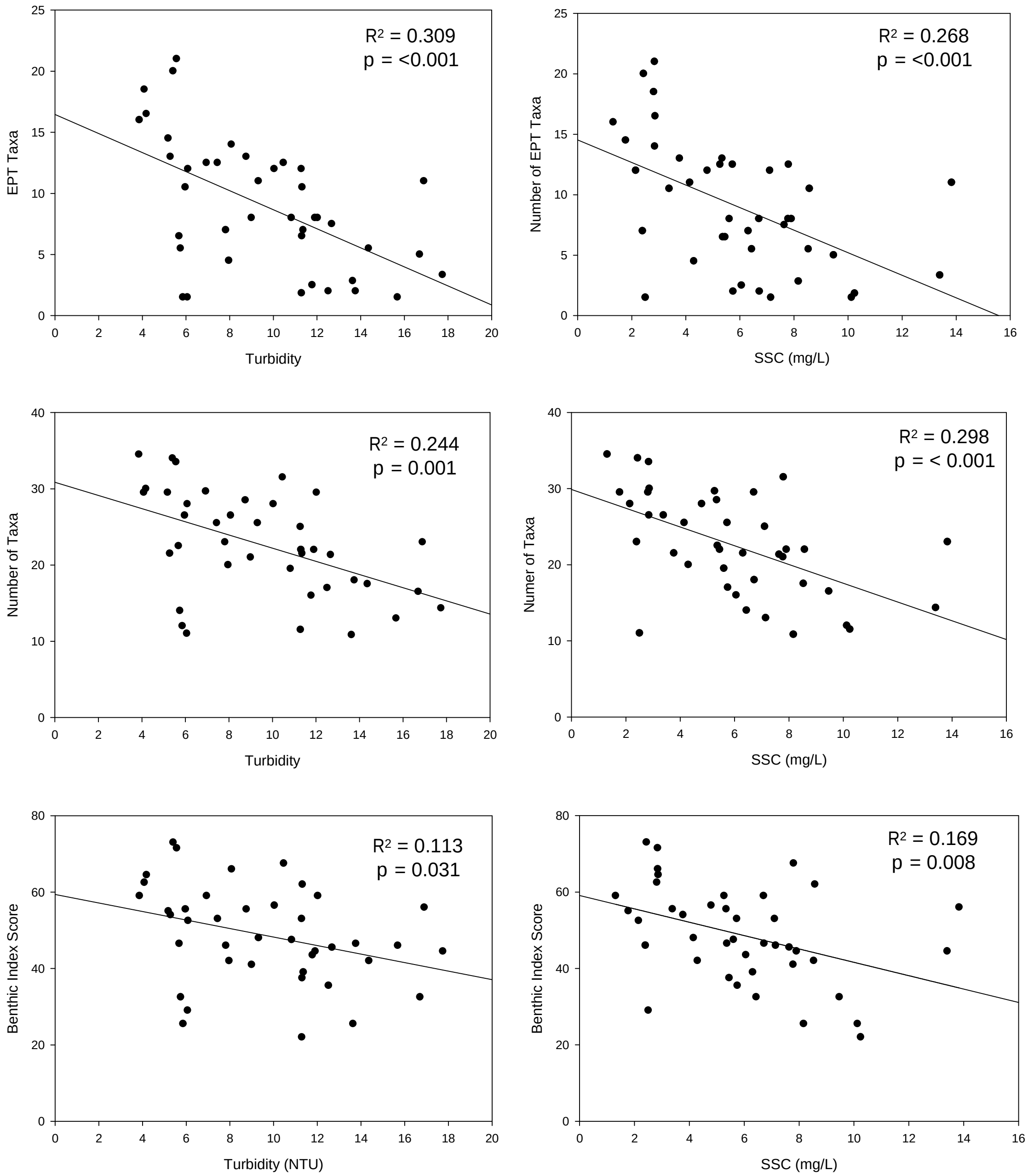


Figure 3.4. Comparison of EPT taxa, total number of taxa, and benthic index scores with turbidity and suspended sediment concentration.

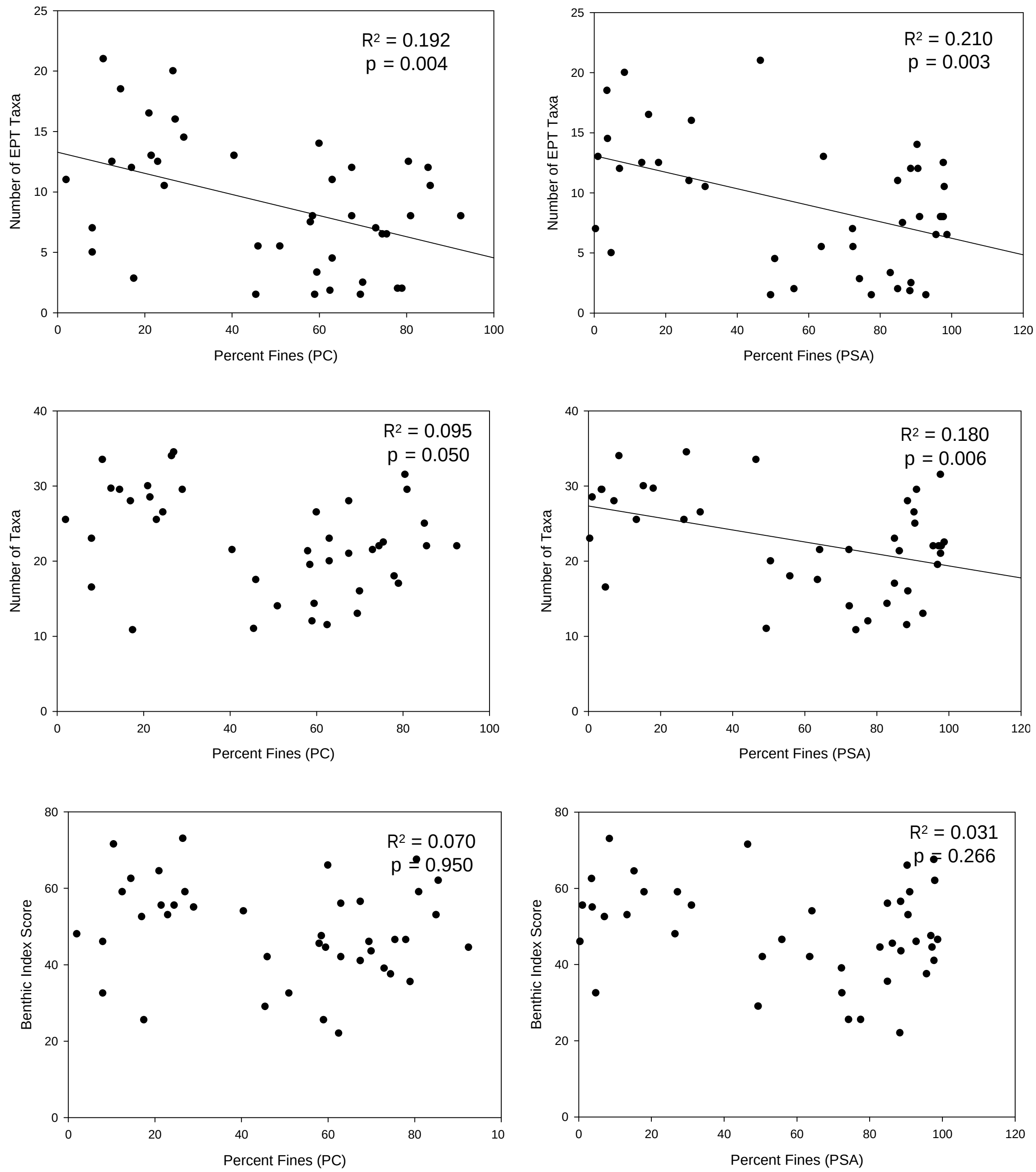


Figure 3.5. Comparison of EPT taxa, total number of taxa, and benthic index scores with percent fines using both the pebble count and particle size analysis method.

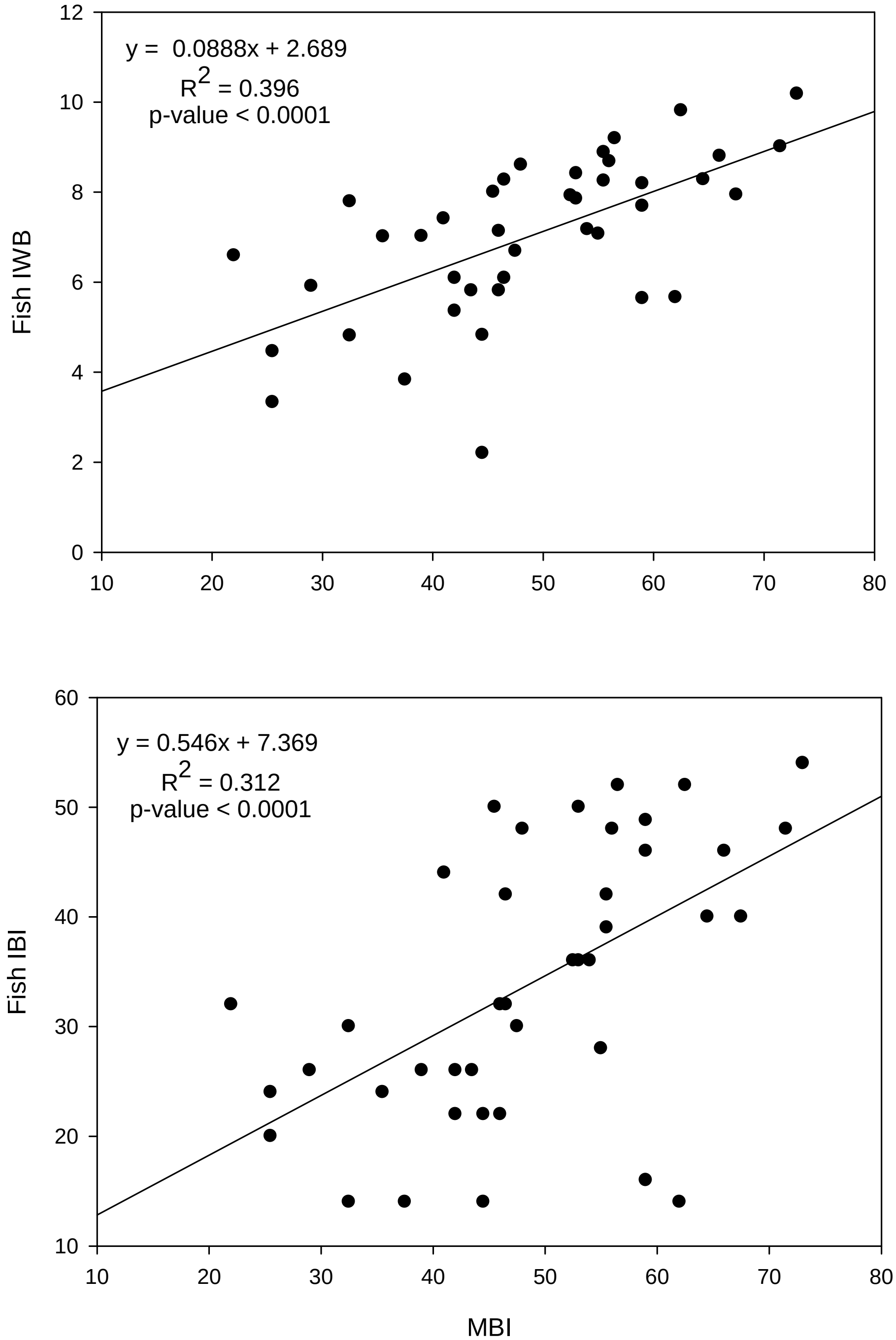


Figure 3.6. Regression line representing the relationship between MBI and a) Fish IWB and b) Fish IBI Scores for streams in the Georgia Piedmont.

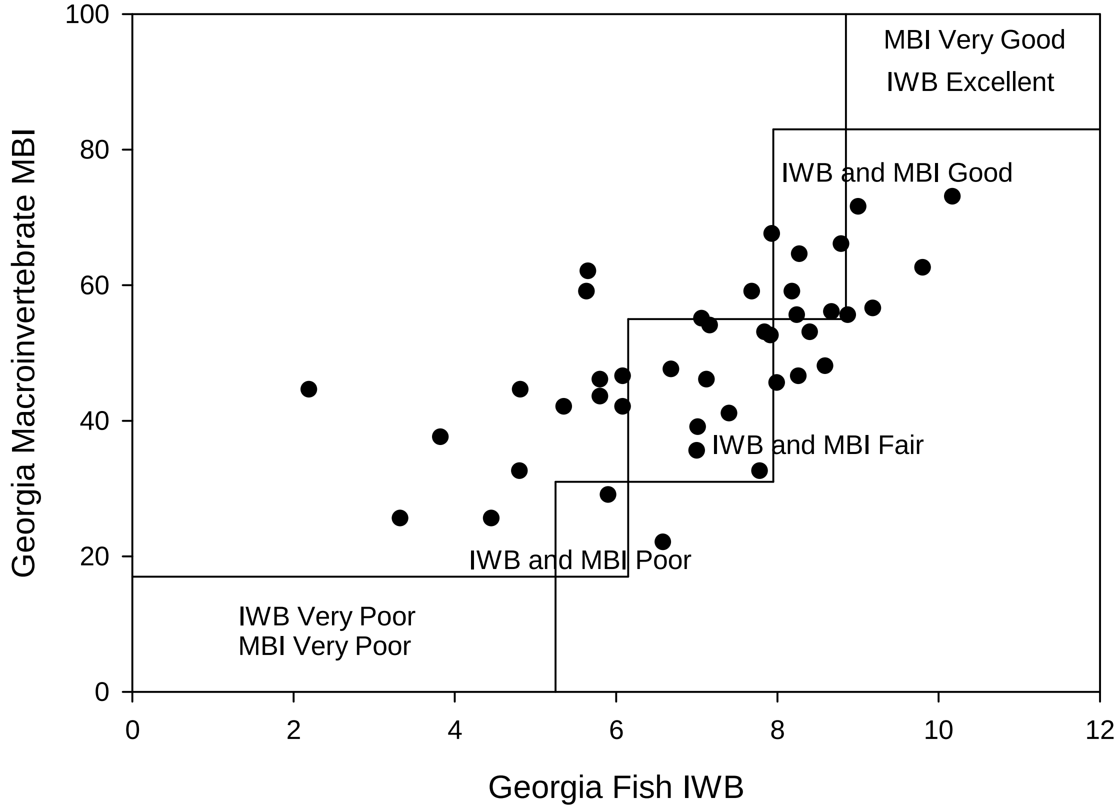


Figure 3.7. General Description of the Relationship of IWB and MBI for Georgia Piedmont streams.

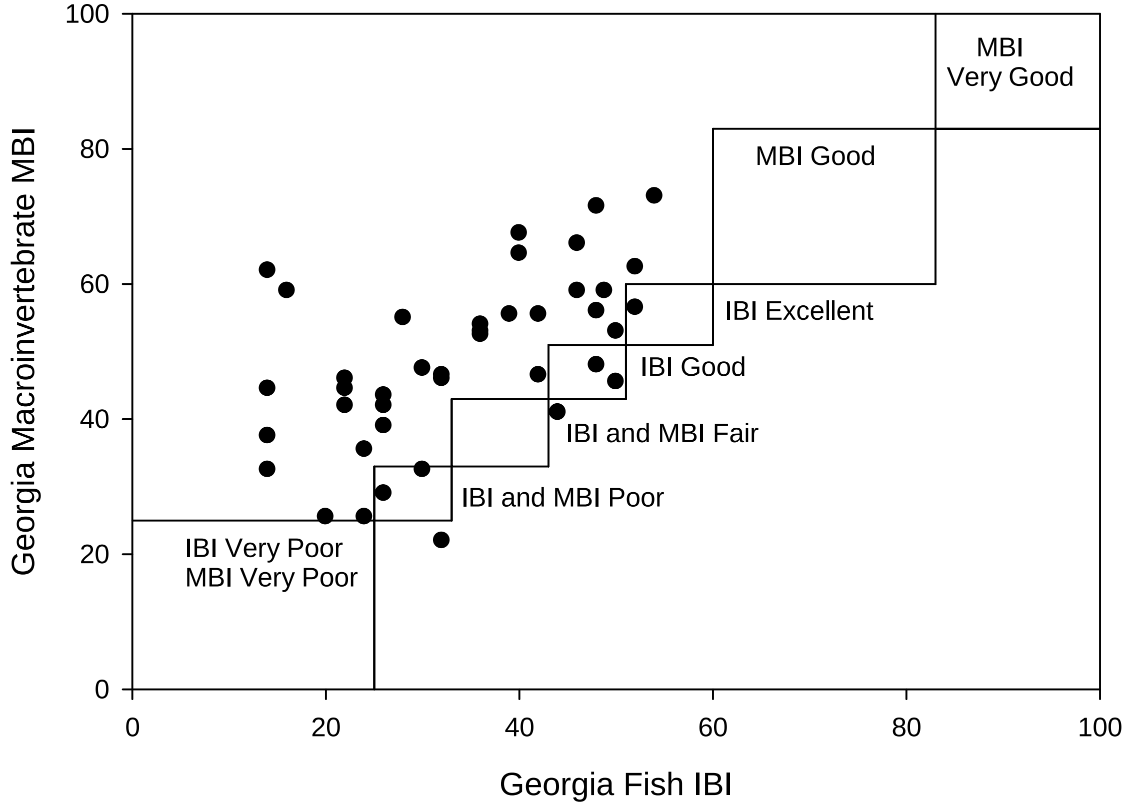


Figure 3.8. General Description of the Relationship of IBI and MBI for Georgia Piedmont streams.

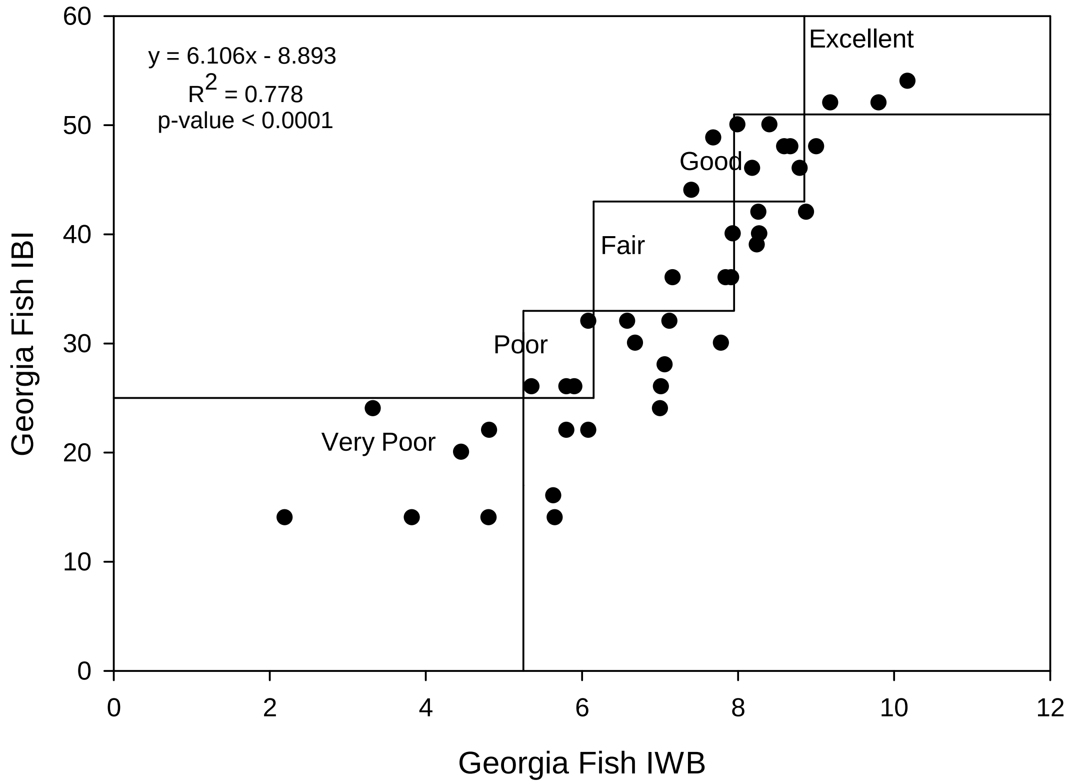


Figure 3.9. Regression line representing the relationship between IWB and IBI and general description of categorical stream health ratings.

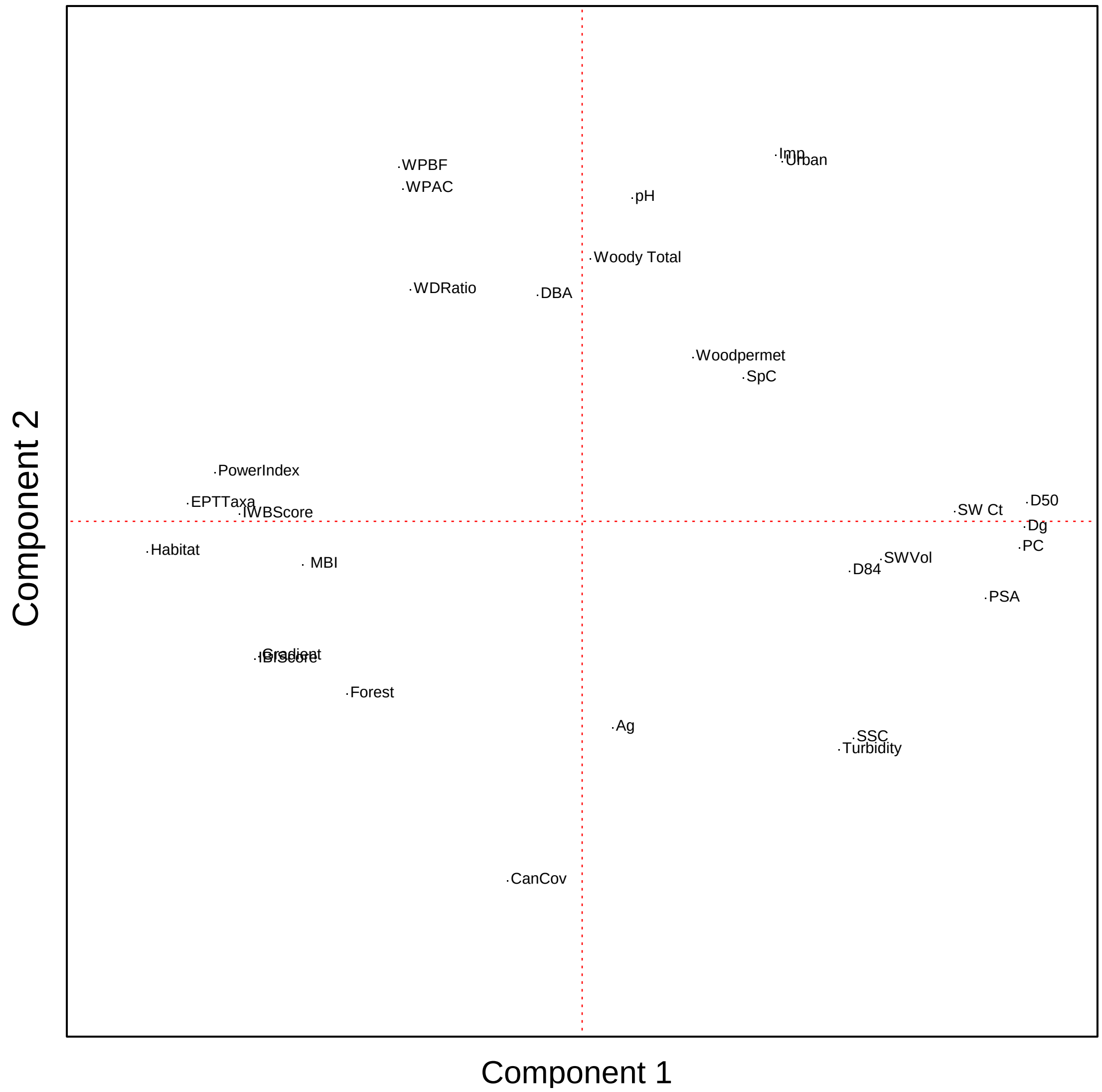


Figure 3.10. Principal component analysis of all variables in the dataset. Component one explains 30.79 percent of the variability in the data, the addition of component two increases adds 15.47 percent. The factors in component two that explain most of the variability are related to stream size (Drainage basin area (DBA), wetted perimeter active channel (WPAC), and wetted perimeter bankfull channel (WPBF).

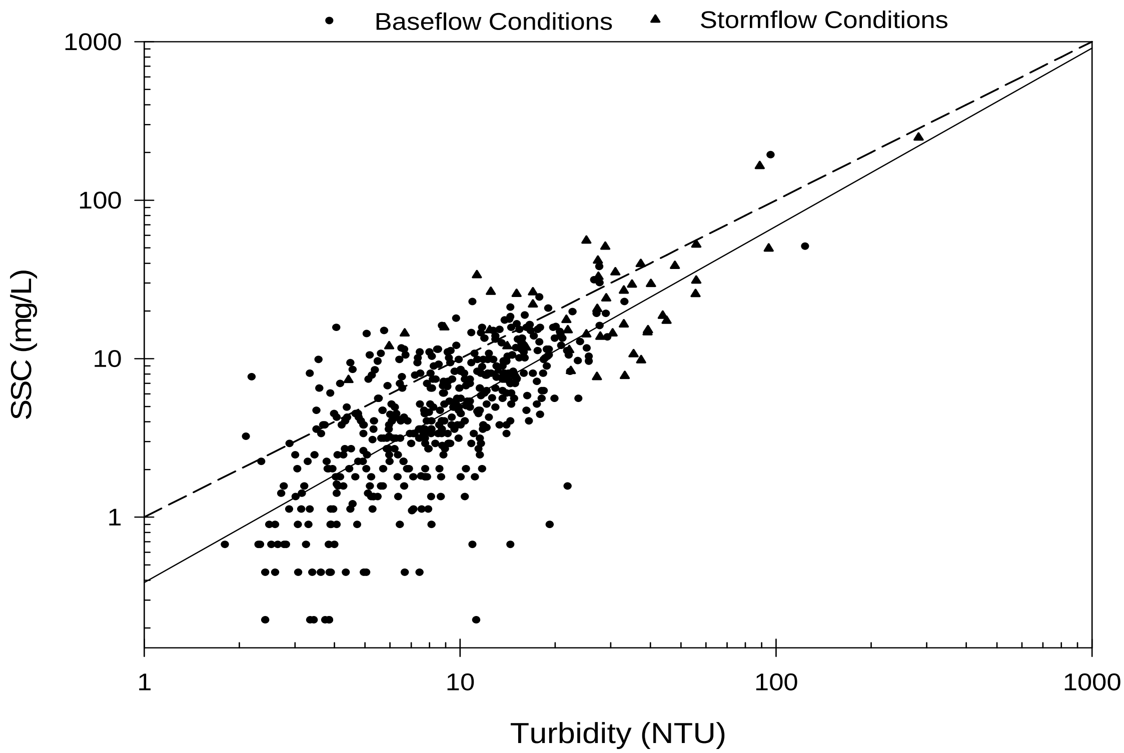


Fig 3.11. Relationship between turbidity (NTU) and suspended sediment concentration (mg/L) (solid line) during stormflow and baseflow conditions (R-square 0.531, $p<0.001$). The one-to-one line is indicated by the dashed line.

CHAPTER 4

Further Considerations for "Urbanization Influences on Aquatic Communities in Northeastern Illinois Streams"

By Faith Fitzpatrick, Michael Harris, Terri Arnold, and Kevin D. Richards

Most water quality assessments based on biotic health address either macroinvertebrate or fish community conditions, but rarely are the two communities used conjunctively. Even fewer studies exist that compare indices based on the two different aquatic communities. The work by Fitzpatrick et al, (2004) addresses the influence of urbanization on aquatic communities using both macroinvertebrate and fish biotic indices, but they do not use their data to compare fish and macroinvertebrate biotic indices directly. They conclude that the Illinois fish alternative index of biotic integrity (AIBI) and macroinvertebrate index (MBI) scores respond similarly to land use changes, decreasing as agricultural land undergoes urbanization. The authors point out that macroinvertebrate assessments are useful in limited situations where fisheries data are unavailable or in streams with limited restricted aquatic resource. The protocol in Illinois is to use the MBI only when fish data are not available. This begs two questions: how well do fish and macroinvertebrate indices of biotic integrity (IBIs) correlate and do fish and macroinvertebrate IBIs respond similarly to stressors?

Using their data, we compared the relationship between the AIBI and MBI with a simple linear regression (Fig 1). The relationship is significant ($p < 0.0001$) but weak

(R2 = 0.3579). The large unexplained variability may be due to the fact that macroinvertebrates and fishes respond differently to stressors. For example, macroinvertebrate communities experience species loss following urbanization, while fish communities often see replacement of native species by non-natives (Wang and Lyons 2003). This fact manifests itself in individual biotic indices; fish indices include a nonnative species metric, a metric absent in benthic indices. Also, as the authors pointed out, benthic organisms may react differently than more mobile fish species to substrate trapped contaminants, such as copper (Fitzpatrick et al 2004).

Another interesting factor of the Fitzpatrick et al (2004) dataset is the narrow range of scores produced by the biomonitoring surveys (Fig 5.2). Of the 43 streams, only three have excellent fish scores and only one has a very poor fish score. The remaining streams, 39 of 43, fall into only three of five categories for the fish AIBI (excellent, good, fair, poor, very poor). The breadth of MBIs is even smaller with scores ranging from 4.4 to 7.9. Illinois created only three categories for the MBI score (good, fair, poor). All of the streams fall into two MBI categories (good and fair). The category definitions suggest use of the Illinois MBI scores will tend to overestimate water quality if fish conditions are the reference.

To use the MBI and AIBI interchangeably, it is essential that the two indices are calibrated to show the same state of degradation. Previous studies by Seegert (2000), Houston et al. (2002), and Iliopoulou-Georgudaki et al. (2003) have shown the need for IBI calibration due to variation among IBI assessments. Seegert (2000) found that using the same dataset from the Pigeon River in North Carolina, three different fish IBIs produced scores differing by as much as 18 IBI units. These differences placed river condition in different categories (i.e., poor, fair, good) depending on which IBI was used. In Peloponnisos Greece, Iliopoulou-Georgudaki *et al.* (2003) used nine biotic indices and five different species to evaluate water quality. They found that macroinvertebrates were the most suitable bioindicator. However, stream health was dependent upon which biotic index was used (e.g., the Agios Dimitrios 3 site scored good, moderate, and poor using five different indices). Fitzpatrick et al.'s (2004) dataset also suggests that stream size, as quantified by basin area, also affects IBI scores (Figs 3 and 4 in Fitzpatrick et al 2004). Larger basins tend to have better biotic conditions as determined by the Illinois MBI and AIBI.

The work by Fitzpatrick et al. (2004) addresses the important but not well understood relationship of macroinvertebrate and fish biotic indices. Similar studies which further investigate these interconnections are essential to developing effective biomonitoring surveys.

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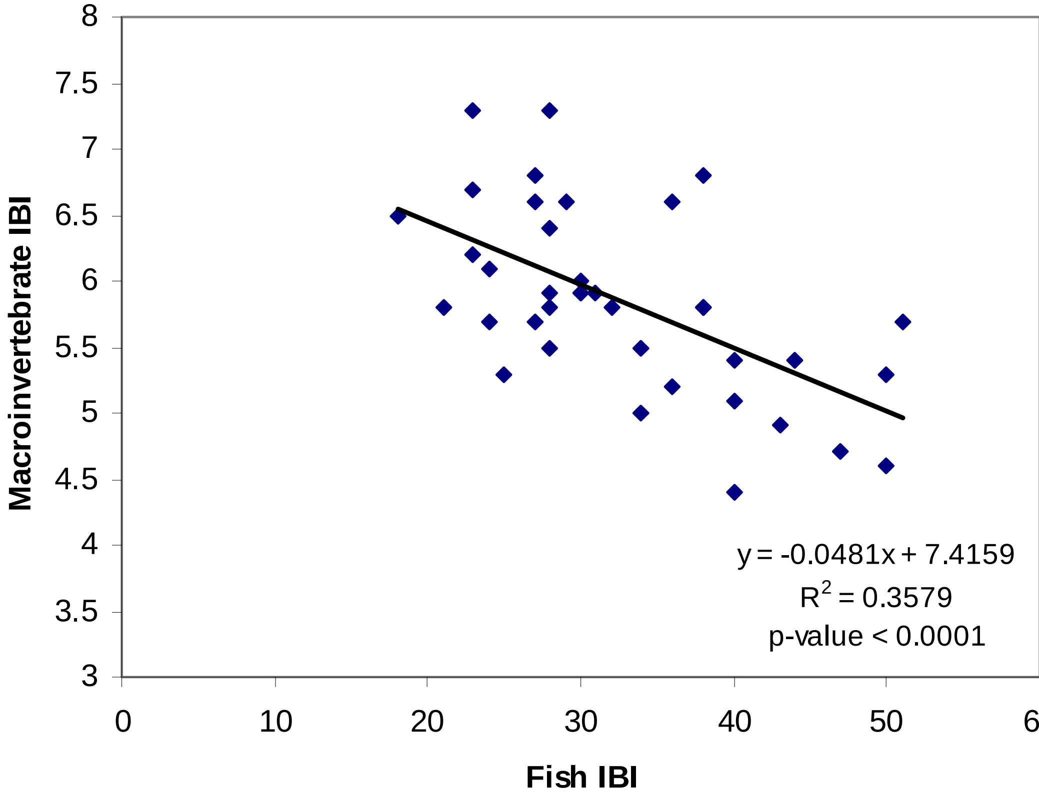

Figure 4.1. Regression line representing relationship between MBI and AIBI scores for Northeastern Illinois streams.

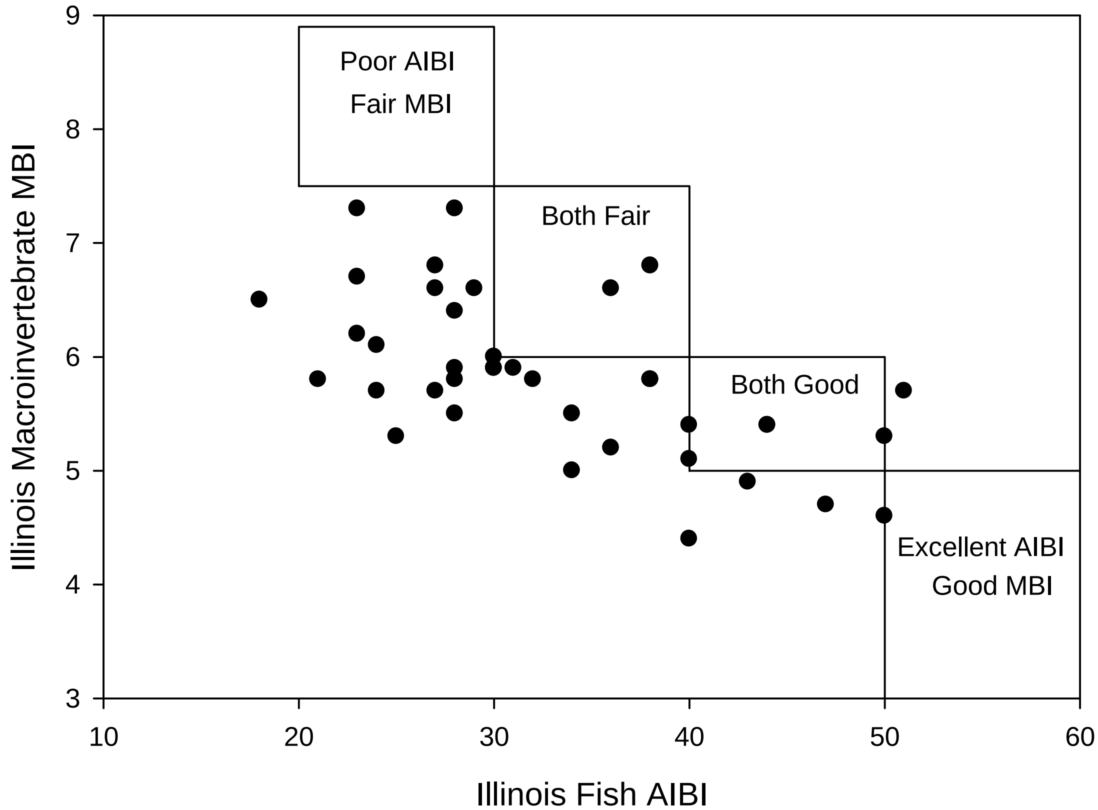

Figure 4.2. General description of the relationship of AIBI and MBI.

CHAPTER 5

Conclusions

The deleterious effects of changing land use on water quality, particularly urbanization, have been and continue to be a major concern (Nilsson et al. 2003b, Walsh et al. 2005c). How we assess our waters to inform management decisions that will protect the resource is of the utmost importance, as drought and legislation threaten its availability in the Georgia Piedmont ecoregion. Our results indicate biotic community metrics were highly sensitive to land cover, specifically the fractions of urban land, imperviousness, and forest. This is not surprising given that "urbanization is second only to agriculture as an agent of stream degradation" (Morgan and Cushman 2005). Nearly all fair, good, and excellent fish IBI scores occurred in basins with more than fifty percent forest and less than 15 percent urban area and less than four percent impervious surface, indicating that Georgia Piedmont streams may be more sensitive to urbanization effects than streams in other parts of the country. The legacy effects of cotton farming coupled with the multiple physical changes attributed to urbanization (i.e., altered flow, silt deposition) may account for the increased sensitivity. Despite the numerous causes, it is clear fish community composition responds to land use changes more so than to annual variation (Schweizer and Matlack 2005).

Macroinvertebrate assemblages responded similarly to urbanization, with reduced EPT taxa and total taxa. Macroinvertebrate shredder taxa richness was negatively affected by watershed land use, but shredder abundance and percent composition were not. Shredder abundance and composition were not correlated to litter availability. Overall, sites with less urbanization had greater litter standing crops during December; however, higher rates of retention occurred in more urbanized areas. We infer that urban streams balance litter export with additional horizontal inputs from storm drains that act to increase the litter source area.

The changes in fish and macroinvertebrate assemblages are related to different factors. Fish indices are most highly correlated to specific conductivity and stream power, while EPT taxa and number of taxa are most highly correlated to water clarity, suspended sediment, and specific conductivity. Neither baseflow suspended sediment concentrations, baseflow turbidity, nor bed particle size distributions were significantly related to watershed imperviousness or other land use metrics. However, these metrics should be collected with all biotic assessments to help explain the variability in the indices.

The fish and macroinvertebrate indices can be useful tools to answer questions regarding preservation and restoration of aquatic resources. The two indices together provide a complete picture of how biota are reacting to changes in the watershed. Calibration is required to use one index in the place of another to explain stream health.

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