Published online September 13, 2006

Stream Buffer Effectiveness in an Agriculturally Influenced Area, Southwestern Georgia: Responses of Water Quality, Macroinvertebrates, and Amphibians

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ABSTRACT

To determine useful metrics for assessing stream water quality in the Southeastern Coastal Plain, we examined differences among two buffered and three unbuffered streams in an agricultural landscape in southwestern Georgia. Potential indicators included amphibian diversity and abundance, aquatic macroinvertebrate populations, riparian vegetative structure, water quality, and stream physical parameters. Variability among sites and treatments (buffered vs. unbuffered) existed, with sites in the same treatment as most similar, and disturbances from a nearby eroding gully strongly affecting one unbuffered site. Of the invertebrate metrics examined, percentages of clingers, Ephemeroptera-Plecoptera-Trichoptera (EPT), Elmidae (Coleoptera), Crustacea (Decapoda and Amphipoda), and dipterans were found to be possible indicators of stream health for perennial streams within this region. Overall, buffered sites showed higher percentages of sensitive invertebrate groups and showed lower and more stable concentrations of nitrate N, suspended solids, and fecal coliforms (FCs). Percent canopy cover was similar among sites; however, riparian vegetative coverage and percent leaf litter were greatest at buffered sites. No differences in amphibian abundance, presence, and absence within the riparian area were apparent between sites; however, instream larval salamanders were more abundant at buffered streams. In this study, stream buffers appeared to decrease nutrient and sediment loads to adjacent streams, enhancing overall water quality. Selected benthic macroinvertebrate metrics and amphibian abundance also appeared sensitive to agricultural influences. Amphibians show potential as indicator candidates, however further information is needed on their responses and tolerances to disturbances from the microhabitat to landscape levels.

The paradigm of a healthy stream ecosystem (Meyer, 1997) includes the concepts of 'sustainability' and 'resilience,' incorporating both ecological integrity (maintaining structure and function) and societal values. Aquatic ecosystems worldwide have been subject to many anthropogenic disturbances, severely degrading the health of streams and rivers. Over 60% of reported waterway problems within the Southeastern USA have been associated with land altering practices such as agriculture, which continue to be a major contributor of nonpoint source (NPS) pollution (USEPA, 2000). Hydraulic alterations of waterways, alteration of flow rates, and the disruption of wildlife habitats through changes in chemical concentrations and increases in sedimentation, are additional consequences of intensive agriculture (Schultz et al., 1995).

Conservation buffers are areas of land permanently maintained in vegetation designed to intercept NPS pollutants such as sediment, nutrients, and pesticides within and from agricultural fields (Lowrance, 1992; Lowrance et al., 1997; Verchot et al., 1997; Vellidis et al., 2002; Vellidis et al., 2003). Although extensive information is available, limited data exist on buffer zone effectiveness in the Southeastern U.S. Coastal Plain. In particular, previous studies predominately consist of small plot-size treatments rather than entire functioning farms. Additionally, conservation efforts have focused on the protection and restoration of public lands even though >75% of the land base in the Southeast is in private ownership (NRCS, 1996). A consideration of private as well as public land in conservation research is extremely important to develop an understanding of ecological processes in human modified landscapes.

Water quality monitoring is one tool used to assess the health of aquatic systems (Rosenberg and Resh, 1993). Biological monitoring in particular has been recognized as one of the most useful tools in assessing water quality (Chandler, 1970; Karr and Chu, 1999; Linke et al., 1999). Ideally, a useful biotic indicator would have the combined attributes of responding quickly to problems, diagnostic of differing stressors, readily sampled, and present in sufficient numbers for comparison with reference conditions (Rapport, 1992). Benthic macroinvertebrate assemblages have been often used in biological monitoring, especially to assess agricultural impacts to streams (e.g., Lenat, 1984; Berkman et al., 1986; Rosenberg and Resh, 1993; Gregory, 1996; Davis et al., 2003). Macroinvertebrates offer a spectrum of responses to disturbance and inhabit a wide range of environments while integrating the effects of short-term environmental variation (e.g., Lenat et al., 1980; Abel, 1989; Barbour et al., 1999). In the Gulf Coastal Plain, studies by Gregory (1996) and Davis et al. (2003) have shown that macroinvertebrate communities are valuable tools to evaluate the effectiveness of best management practices (BMPs) on streams affected by human land use.

Amphibians as Bioindicators

The Southeastern USA supports a diverse assemblage of amphibians, with over 144 species occupying a variety of freshwater habitats (Duellman and Sweet, 1999; Dodd et al., 2004). Many amphibians are highly sensitive to disturbance, have complex life histories which often require

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Published in J. Environ. Qual. 35:1924–1938 (2006).
Technical Reports: Surface Water Quality
doi:10.2134/jeq2005.0456

Abbreviations: EPT, Ephemeroptera-Plecoptera-Trichoptera; PCA, principal components analysis; FC, fecal coliform; FS, fecal streptococci; IBI, index of biotic integrity; AFDM, ash free dry mass; DBH, diameter at breast height.

both terrestrial and aquatic environments, and often require specific microhabitats (Wake, 1990; Dunson et al., 1992; Blaustein et al., 1994; Stebbins and Cohen, 1995). Amphibians, especially salamanders, are excellent indicators of local conditions because they have fairly limited home ranges (Blaustein and Wake, 1995), they are relatively long-lived, and may have extended larval periods (Petranka, 1998). Furthermore, many species have lotic dwelling larval stages which are highly specialized, using stream microhabitats for both cover and foraging (Welsh and Ollivier, 1998). In small headwater streams, salamanders can occur in large numbers and provide another potential tool to assess stream health where other species assemblages, such as fish, may be less diverse or absent (Corn et al., 2003; Rocco et al., 2004).

Throughout North America some amphibian populations have been reduced due to habitat loss and alterations from wetland drainage, timber harvesting, agriculture, urbanization, stream pollution and siltation, and the introduction of exotic species (e.g., Blaustein and Wake, 1995; Petranka, 1998; Welsh and Lind, 2002; Dodd and Smith, 2003). These taxa play a vital role in aquatic ecosystem dynamics, and their decline may indicate a degradation of ecosystem health (Vitt et al., 1990; Semlitsch, 2003).

Although amphibians as a group show promise as ecological indicators, few studies in the Southeastern USA have specifically examined amphibian responses to disturbance within aquatic ecosystems and this faunal group is rarely considered in monitoring programs (but see Chazal and Niewiarowski, 1998; Bowers et al., 2000; Homyack and Giuliano, 2002; Ryan et al., 2002; Willson and Dorcas, 2002). However, there are examples in the Western USA where amphibians were incorporated into assessments of impacts of forest management practices in riparian zones (e.g., Gomez and Anthony, 1996; Wilkins and Peterson, 2000; Welsh and Lind, 2002). Also, the Ohio Environmental Protection Agency has established an amphibian IBI (index of biotic integrity) for isolated wetlands and is developing similar indices for headwater streams (Micacchion, 2002). Given the high diversity of amphibians in the Southeast, their inclusion in biological monitoring programs seems warranted.

We examined a suite of stream health indicators including macroinvertebrate and amphibian community composition and abundance, riparian composition, physical habitat composition, and overall water quality to detect differences between buffered (fenced from cattle access) and unbuffered (unfenced streams; cattle have access to streams) sites in southwestern Georgia. It was important to also determine potentially useful metrics for stream water quality and habitat in the Coastal Plain region, and to address the role amphibian species may play in bioassessment.

MATERIALS AND METHODS

Study Area Description

The study took place on a private farm in Early County, Georgia which is located within the Eastern Gulf Coastal Plain Fall Line Hills (FLH) physiographic district. The FLH is an area characterized by flat-topped ridges, and is dissected by meandering streams creating valleys that lie 15 to 75m below the adjacent ridge tops. Underlain by easily eroded sands, clays, and gravels, this region contains numerous sinkholes and springs, with narrow stream terraces; these streams also receive considerable amounts of groundwater discharge (Brantly, 1916; Clark and Zisa, 1976; Couch et al., 1996). The erodability of the strata, coupled with the elevation of the plain above the streams, abundant rainfall, cultivation of the land, and timber removal have caused the area to experience extensive erosion, forming steep gullies or washes [Southwest Georgia Regional Development Center (SWG RDC), 2005]. Geomorphically, streams in this region are sinuous, typically lacking riffles and shoals.

The climate of the region is warm and humid, with long hot summers. Winter temperatures fall below freezing for only brief periods of time [Southeast Regional Climate Center (SERCC), 2004]. Average monthly temperatures range from 2.7 to 15.3°C in January to 20.8 to 33.3°C in July (SERCC). Average annual precipitation is 141.8 cm, with the average minimum rainfall occurring in October (6.6 cm) and the maximum in January (15.9 cm) (SERCC).

Southwest Georgia is one of the USA's most productive agricultural regions (SWG RDC, 2005). Early County is a mix of cropland and pasture, with over 50% of the county in farmland. Cotton, corn, soybeans, sorghum, small grain, peanuts, and pecans are some of the main crops grown, and due to the large number of frost free days, it is not uncommon for the land to be double cropped. One of many farmland management concerns in the county has been soil erosion and low soil fertility. Due to these problems, agriculture is dependent on nutrient additions and irrigation (SWG RDC, 2005).

The study was conducted on the Brownlee Farm, located within the 68 km^2 sub-watershed of Factory Creek (Fig. 1). The farm is a diversified row crop and beef cattle operation that has been in production for >30 yr. Factory Creek is a second order tributary of the Lower Chattahoochee River, which has been designated as impaired under the Clean Water Act Section 303(d) due to concerns of altered lead, dissolved oxygen, and FC bacteria concentrations (USEPA, 2002).

Sampling Regime

Five sampling sites in the Factory Creek basin were selected for biological, chemical, and physical assessment. Three sites were unbuffered (UB-1, UB-2, and UB-3), and two were fenced or reference buffered sites (B-1 and B-2). Study site descriptions are as follows:

UB-1 was located near a heavily eroding gully and was surrounded by cotton fields, common rye grass [*Secale cereale* (L.)], and bahiagrass [*Paspalum notatum* (Flueggé)] pastures. Before the study, the upland area surrounding the gully was fenced to exclude cattle and seeded with grass to restore the area and prevent sediment from washing into the stream. However, due to drought conditions, little vegetation was established, thus allowing sediment to continue to wash into the stream following heavy rainfall. During the study, cattle had access to the stream and traveled down steep slopes (8%) for water, further contributing to the erosion.

UB-2 and UB-3 were surrounded by bahiagrass pastures. Cattle had access to the streams before and during the study. Compacted soil and severe bank erosion were evident. Understory vegetation was sparse, tree roots were exposed, and many trees had fallen both within the stream (especially at site UB-3) and in the riparian zone.

B-1 and B-2 were located within the same sub-basin. Cattle were excluded from both streams with fences >25 yr before this study. Fences were located downslope from pastures,

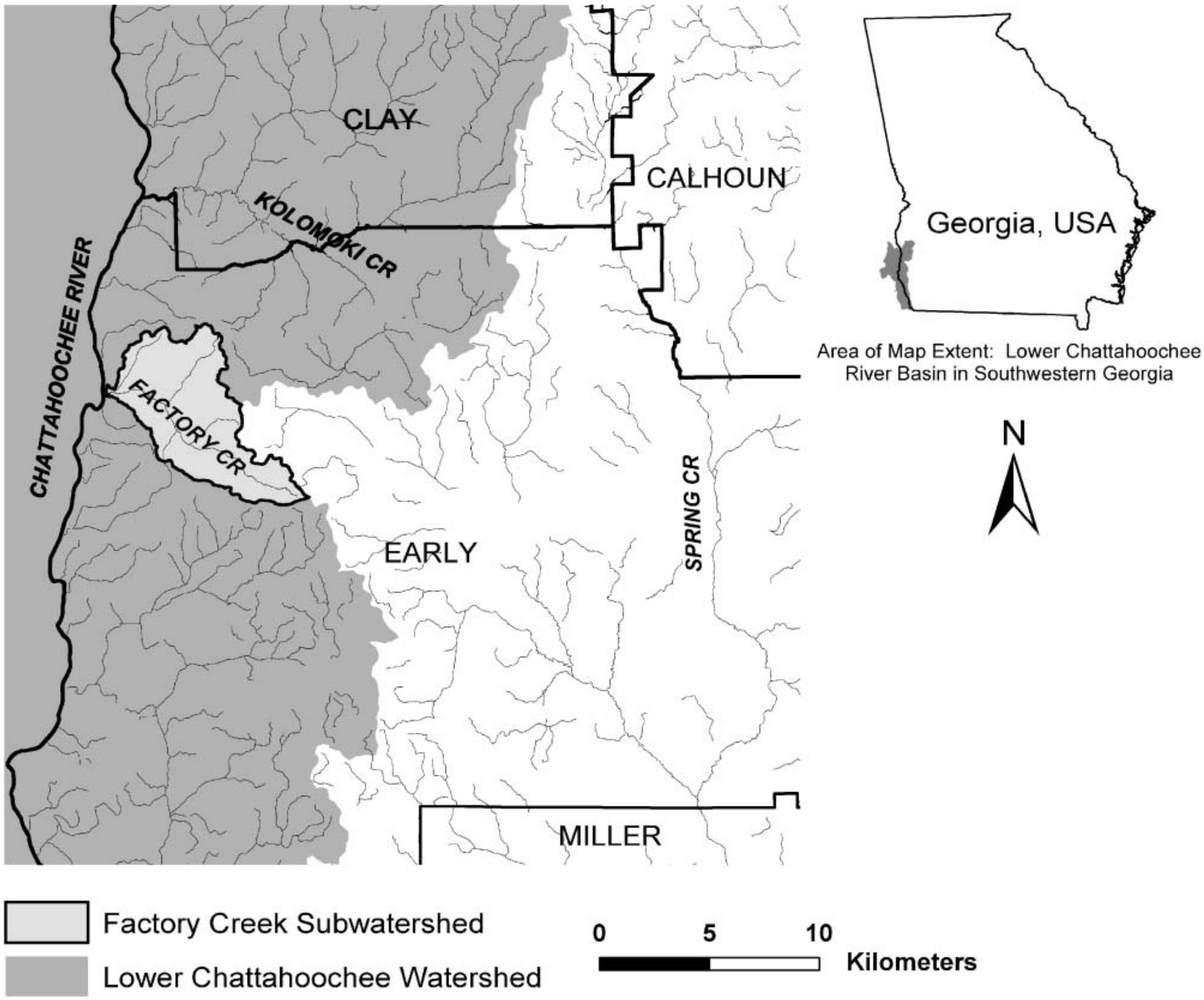

Fig. 1. Map of study area location in Early County, Southwest Georgia and watershed boundaries.

cotton fields, and pine plantations (M. Brownlee, personal communication, 2003).

Physical Measurements

Physical characterizations of each site included a description of general land use, stream origin and type, and measurements of stream bankfull width and depth. Stream flow velocity (FLOW-MATE, Marsh-McBirney, Fredrick, MD) and depth were measured at 20-cm intervals across each cross-stream transect. In addition, temperature and dissolved oxygen concentrations were measured biweekly with a dissolved oxygen meter (YSI Model 50B, Yellow Springs, OH). To obtain a representation of conditions in the benthic habitat, readings were taken at the sediment water interface. Water depth was measured with stationary staff gauges, although at some sites gauges periodically were dislodged during spates. Temperature loggers (HOBO Temperature logger, Pocasset, MA) were placed at each site to record air temperature (°C) hourly, from February 2002 to March 2003.

Chemical and Biological Water Quality Measurements

Grab samples (5000 mL) were collected biweekly from each stream. All samples were kept on ice until processing. Nitrate (NO_3–N), ammonium (NH_4–N), ortho-phosphate (PO_4–P), and chloride (Cl^-) concentrations were determined using a Brann and Luebbe TRAACS 800 autoanalyzer following EPA-approved colormetric techniques (Greenberg et al., 1992). Total kjeldahl nitrogen (TKN) was quantified using an EPA-approved digestion and titration technique (Tecator Autosampler 1035/1038) (Greenberg et al., 1992). Total suspended solids concentrations were determined using conventional techniques, and were dried at 103 to 105°C (Method 2540 D) (Greenberg et al., 1992). Apparent color measurements were made within 24 h of sample collection, and unfiltered room temperature samples were analyzed for apparent color with a platinum-cobalt standard scale (HACH, DR/2000 Spectrophotometer, Method #120). At each grab sample date, one 100-mL water sample from each site was collected in a single-use sterile sampling bag, stored on ice, and analyzed for FC and fecal streptococci (FS) colonies within 6 h of collection following conventional membrane filtration techniques (Greenberg et al., 1992). Due to laboratory analytical availability, only three stream sites were sampled for bacterial concentrations biweekly from February through June 2002.

Aquatic Macroinvertebrates

Benthic macroinvertebrates were sampled bimonthly using a modified 500-μm mesh Hess sampler (Wildco Wildlife Supply

Company, Saginaw, MI). Within a selected 100-m reach at each site, three crossstream transects were randomly chosen. At each transect, two Hess collections, sampling representative habitat types within the stream channel, were composited. Stream bed composition (sand, gravel, roots, etc.) was estimated visually by the line intersect method (Davis et al., 2003). Samples were rinsed into plastic bags, preserved in the field with 70% ethanol, and stained with rose Bengal dye. In the laboratory, processing of entire samples involved washing invertebrates from organic matter through a 1-mm and 500-μm sieve to be later sorted. Identifications were made to the lowest practical taxonomic level, most often genus, under a low power dissecting microscope and compound light microscope (Berner and Pescador, 1988; Stewart and Stark, 1988; Pescador et al., 1995; Epler, 1996, 2001; Wiggins, 1996; Needham et al., 2000; Thorp and Covich, 2001). Larval Chironomidae (Diptera) samples from two dates in February and August 2002, were mounted on microscope slides in carboxylmethyl cellulose (CMC) and identified to genus (Epler, 2001). Samples of >500 individuals were subsampled and adjusted to a final volume of 100 mL with 70% ethanol, placed on a magnetic stirrer to produce a homogenous solution, and then three 5-mL subsamples were removed with a wide-bore pipette (Hax and Golladay, 1993). Coarse organic material was removed from invertebrate samples, oven-dried (70°C, 24h), ground in a Wiley Mill, subsampled, weighed, ashed, and reweighed to obtain ash free dry mass (AFDM) as an estimate of benthic organic matter standing stock.

Amphibians

Amphibian diversity and abundance was determined using: (1) searches under natural cover objects (Jaeger, 1994); (2) searches under artificial cover boards (Fellers and Drost, 1994); (3) tree pipe surveys (Boughton, 1997), and (4) larvae collected in Hess samples. Amphibians observed opportunistically were also noted. Artificial and natural cover searches were conducted bimonthly (March 2002 to March 2003), and tree pipes were surveyed monthly (June 2002 to March 2003). The sampling area was a 100 by 4 m belt transect along one side of each stream. At each amphibian sampling date we measured air temperature above the ground level (>3ft) in a shaded location and soil temperature <5cm below the soil surface. Cloud cover was estimated (i.e., clear sky (<10%), partly cloudy (10 to 80%), overcast (>80%)), and weather conditions noted.

Artificial cover arrays consisted of five stations spaced 20 m apart in each belt transect. Each station included three pairs of boards (30 × 30 × 2.54 cm), with one pair located 4 m, 2 m, and 0 m from the stream edge, for a total of 6 boards per station, and 30 boards per site. Surveys began 6 wk after boards were placed at the study site. Natural cover objects within the belt transects were surveyed by turning (and replacing) all logs, sticks, and rocks within the belt transect. Number of cover objects turned was also recorded.

White polyvinyl chloride (PVC) pipes which provide artificial refugia for tree frogs were placed at each cover board station (Moulton et al., 1996; Boughton, 1997). Pipe dimensions followed the methods of Borg et al. (2004). Two pipes were hung vertically on trees at each station, for a total of ten pipes per site; pipes were placed on 12 different species of trees predominantly consisting of tulip poplar [*Liriodendron tulipfera* (L.)] and swamp tupelo [*Nyssa biflora* (Walter) Sarg.]. Pipes were put in place 4 wk before the first survey period and were checked monthly between 0900h and 1600h EST from June 2002 to March 2003. Salamander larvae recovered from preserved invertebrate Hess samples were identified (Petranka, 1998) and enumerated.

Vegetation Surveys

Vegetation transects consisted of four consecutive plots (5 × 25 m) positioned adjacent to the stream on the side with the wider floodplain terrace. All trees >2.5 cm in diameter at breast height (DBH) were identified and DBH was recorded. For each transect, mean density, basal area, and importance values were determined by species. Within each plot, shrubs (DBH < 2.5 cm, but > 30 cm in height) were identified to species and percent cover of each species was estimated using the following seven coverage classes: <1%; 1 to 5%; 6 to 15%; 16 to 25%; 26 to 50%; 51 to 75%, and 76 to 100%. Coarse woody debris (CWD) > 5 cm in diameter was quantified by size class. Standing dead trees (snags) were noted and DBH was recorded. A robel pole was used to estimate vertical obstruction of the understory (Robel et al., 1969). Two pole readings per plot were taken, one upstream and one downstream from a fixed height of 1 m. Ground layer cover type was estimated using a line transect; across each plot, a vertical pole was dropped at 1-m increments, and cover was categorized as sand, live vegetation, wood, leaf litter, or other organic matter.

Five 1 × 1 m quadrats (sub-plots) were established in each 5 × 25 m plot. Within each sub-plot, ground cover (<30 cm in height) species (Wunderlin, 1998) and graminoids, forbs, woody debris (all sizes; coarse and fine), and bare ground were recorded using the same cover classes as for the shrubs. Percent canopy cover was determined within each sub-plot with a spherical densiometer (Model-C, R.E. Lemmon Forest Densiometers, Bartlesville, OK). Two densiometer readings were also taken at each invertebrate transect collection, one upstream and one downstream from the middle of the stream channel. Soil compression was measured in each sub-plot with a pocket penetrometer (kg cm^{-2}). On both sides of the stream channel, within the selected transect area, floodplain width was measured and slope was estimated with a percent scale clinometer.

Data Analysis

Multiple parameters and metrics shown in previous studies to be valuable indicators of water quality were tested in this study, and new indicators were developed that might prove effective in assessment of streams disturbed by agriculture. Invertebrate metrics were chosen based on those found valuable for bioassessment (and for this region) and those able to be included due to taxonomic resolution (see Rosenberg and Resh, 1993; Barbour et al., 1996; Davis, 2000) (Table 1). A multimetric approach using an IBI for benthic macroinvertebrates was not utilized in this study because either many of the organisms were in early instars or the expertise to identify the individuals to

Table 1. Expected responses of invertebrate metrics to increased stress and degradation based on a study by Davis et al. (2003). Percent Elmidae was designed for this study.

Metric	Increasing stress
EPT abundance	decrease
No. families	decrease
Percent Amphipoda	decrease
Percent burrowers	increase
Percent Chironomidae	increase
Percent clingers	decrease
Percent Crustacea	variable
Percent Diptera	increase
Percent dominant family	increase
Percent Ephemeroptera	decrease
Percent EPT/Chironomidae	decrease
Percent non Diptera	decrease
Percent Plecoptera	decrease
Percent Trichoptera	decrease
Percent Elmidae	decrease

species was not available. Oligochaeta, Hydracarina, and Copepoda were not adequately sampled during Hess invertebrate collections, and were excluded from the analyses.

Principal components analysis (PCA) was used to compare physical, vegetative, water quality parameters, and macroinvertebrate metrics among sites. The PCA was also used as a form of exploratory data analysis for the selection of metrics to be further tested with inferential statistics. Some parameters were measured with more than one technique (i.e., soil coverage methods), and if a strong correlation was evident between parameters ($r^2 > 0.70$), then the more rapid technique was used in the PCA. Principal components analysis options were Euclidean distance and cutoff r^2 was set at 0.25 (McCune and Mefford, 1995). In each PCA, three axes were selected for interpretation based on broken-stick eigenvalues. An outlier analysis was conducted to identify sites that were located more than two standard deviations from all other sites.

Based on the results of the PCA, variation in metrics and parameters that appeared to be important was compared between stream sites for all dates combined. Comparisons were preformed using a Kruskal–Wallis Test ($p < 0.05$) (SAS Institute, 2002). Due to different collection dates for bacterial samples, statistical analyses were not used for these data. Coverage percentages used in analyses for shrub and ground cover classes were generated by using the median value of the cover class. Overstory plant species importance values (IV) were calculated by averaging the relative dominance (as expressed by basal area), relative density, and relative frequency for each site. Jaccard's index of similarity (JI) was used to compare vegetation by site (Ludwig and Reynolds, 1988).

Abundance data were reported for all hylid frogs, salamanders, and for individual species in which statistically significant differences were found between sites. Comparisons were preformed using a Kruskal–Wallis Test ($p < 0.05$) (SAS Institute, 2002).

RESULTS

Physical Parameters

The PCA for physical parameters determined that the first two axes explained 49.4% of the variation, and cumulatively 65.4% with the inclusion of the third axis (Fig. 2; Axis 3 not shown). Percent open canopy (over the stream) was negatively correlated with Axis 1 ($r^2 = 0.48$), whereas percent leaves, stream depth, and AFDM were positively correlated with Axis 1 ($r^2 = 0.52$, 0.83, and 0.53, respectively). Dissolved oxygen concentrations were positively correlated with Axis 2 ($r^2 = 0.52$), and one variable (percent pebble or rock) was positively correlated with Axis 3 ($r^2 = 0.57$). Sites B-1, B-2, and UB-2 tended to have deeper water, lower percent of open canopy, higher amounts of benthic organic matter (AFDM), and higher percentages of leaf debris and wood/roots. UB-1 was characterized by higher percentages of open canopy and exposed streambed, and was similar to UB-3 with high percentages of sand/silt in the streambed.

Unlike the buffered sites, the unbuffered sites differed in the measured physical parameters. UB-1 had the highest percent sand/silt, percent exposed streambed, percent open canopy (over the stream), and width/depth ratios, and lowest percent wood/roots, percent leaves, and AFDM ($p < 0.05$) (Table 2). UB-2, overall the widest and deepest stream, had the highest percent leaves and AFDM, and lowest width/depth ratios ($p < 0.01$) (Table 2). UB-1 also had the lowest percent pebble/rock. UB-3 was the narrowest stream, and maintained the fastest velocities with higher percentages of pebbles within the streambed ($p < 0.01$) (Table 2). Sites B-1 and B-2 overall had slower stream velocities, the lowest percentages of sand/silt, and highest percent wood/roots (Table 2).

Water Quality

The first two axes of the water quality PCA explained 56.3% of the variation, with an additional 13.6% explained by the third axis (Fig. 3). NH_4–N, suspended solids, and apparent color were positively correlated with Axis 1 ($r^2 = 0.48$, 0.53, and 0.61, respectively). Total nitrogen and NO_3–N were strongly positively correlated with Axis 2 ($r^2 = 0.94$ and 0.77, respectively). UB-3 was characterized by higher concentrations of total N and NO_3–N, whereas UB-2 had lower concentrations. The unbuffered sites were similar in that they had relatively higher concentrations of Cl^-, TKN, suspended solids, apparent color, NH_4–N, and PO_4–P with UB-1 having the highest concentrations of all sites. Sites B-1 and B-2 were characterized by low concentrations of nutrients and sediments.

Apparent color and suspended solid concentrations were lowest at the buffered sites ($p < 0.0001$) and those sites were more stable over time, whereas unbuffered sites showed pulses of suspended sediments after storms. The lowest mean concentrations for apparent color, suspended solids, TKN, PO_4–P, and NH_4–N were measured at buffered sites ($p < 0.0001$) (Table 3). Mean FC and FS concentrations varied over time, with overall highest FC levels peaking in the late summer/fall season (Aug. to Oct. 2002) and highest FS levels occurring in late winter and early spring (Feb. to Apr. 2002) and again in the winter (Nov. to Dec. 2002). Highest concentrations of both fecal forms were consistently found at UB-3 and UB-2, and lowest concentrations were found at B-1, B-2, and UB-1 (Table 3). Overall, mean bacterial concentrations fluctuated, especially at unbuffered sites. Mean FC/FS ratios were mostly in the range of 0.1 to 0.7, indicating bacterial contaminations originated from livestock (FC/FS $>$ 4, human contamination; $0.1 <$ FC/FS < 0.7, livestock; FC/FS < 0.1, wildlife; Geldreich, 1967).

Benthic Macroinvertebrates

A total of 23 840 individual organisms were collected at the five sites (Muenz, 2004). Forty-two taxa were identified, with greatest numbers of taxa in the orders Trichoptera (10 taxa) and Diptera (9 taxa), followed by Coleoptera (5 taxa) and Odonata (4 taxa). Throughout the study, the dominant groups of macroinvertebrates were: Dipterans (87%), of which 88% were of the family Chironomidae, and Coleoptera (8%) of which 73% were of the family Elmidae.

From the invertebrate metric PCA, the first two axes explained 47.5% of the variation, and 56.8% with the inclusion of the third axis (Fig. 4; Axis 3 not shown).

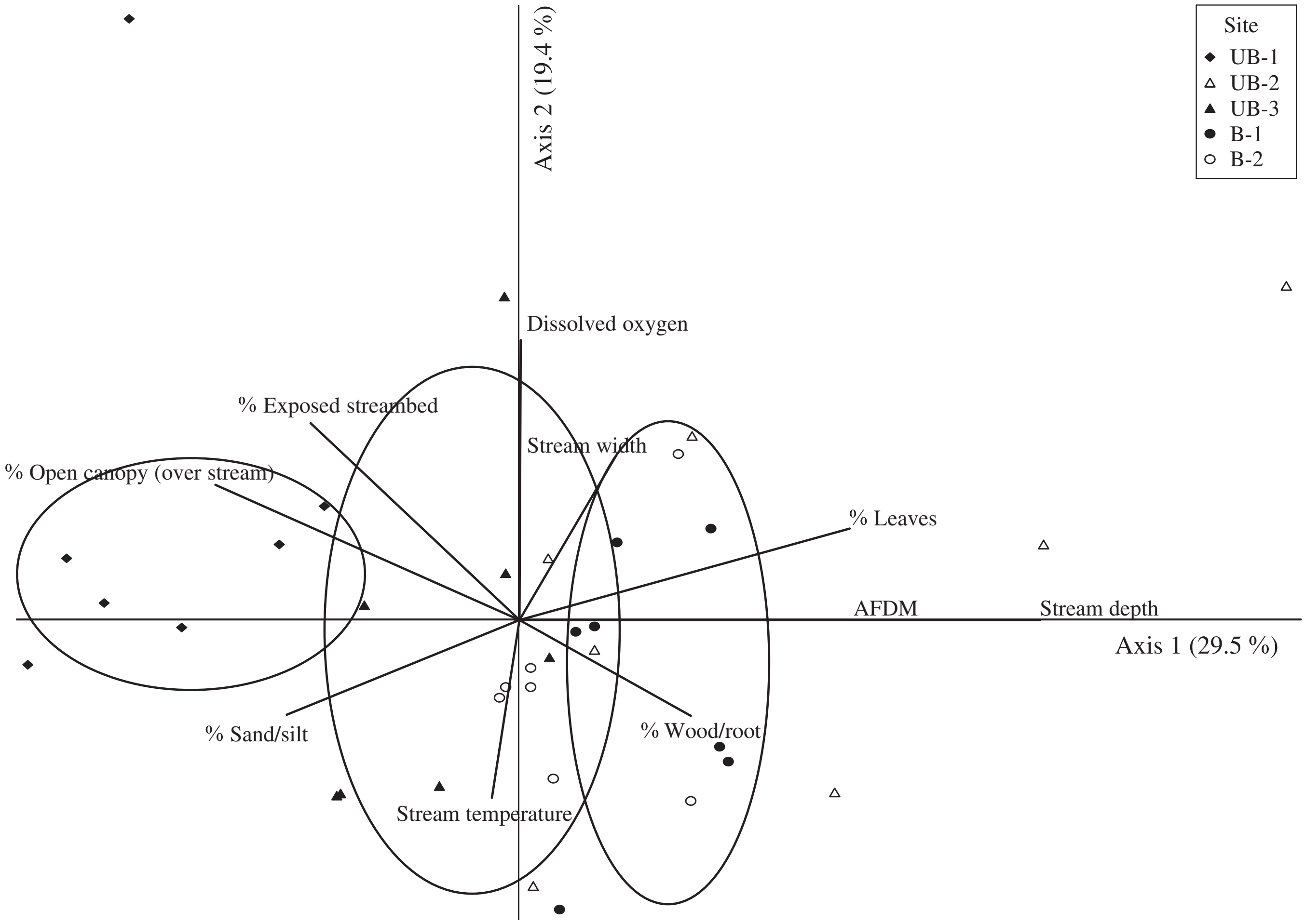

Fig. 2. First and second axes of the principal components analysis (PCA) for physical parameters at five stream sites in Early County, GA. Points represent individual site/dates and vectors indicate physical stream variables. Percent variation is indicated parenthetically on each axis. Ellipses were subjectively drawn around the central locations of points from each site to aid in discerning patterns.

Axis 1 represents a shift from a group consisting of EPT and Elmidae to dipterans. Buffered sites were positively associated with Axis 1 and the metrics percent EPT/Chironomidae, percent Elmidae, percent clingers, and percent EPT (r^2 = 0.69, 0.70, 0.68, and 0.67, respectively). A cluster of points from the unbuffered sites were characterized by high percent Diptera and percent dominant family, and were negatively correlated with Axis 1 (r^2 = 0.86 and 0.61, respectively). In addition, percent Chironomidae (r^2 = 0.49) was negatively correlated with Axis 2. Percentages of Ceratopogonidae (r^2 = 0.72) and burrowers (r^2 = 0.91) were both positively correlated with Axis 2 and associated with a group of points from UB-1.

Table 2. Mean, minimum, and maximum physical stream measurements for all study sites, January 2002 through March 2003. For each site and parameter, values with different letters are significantly different (Kruskal–Wallis test with respective *p* value).

Parameter	UB-1†	UB-2	UB-3	B-1	B-2	*p* value
Width, m	2.1cb (0.7–5.7)	2.9a (1.7–4.5)	1.7c (1.2–2.4)	2.2b (1.3–3.4)	1.9cb (1.5–2.5)	<0.0001
Depth, cm	1.8d (0.3–4.0)	11.8a (7.1–24.4)	4.4c (2.6–8.2)	7.7b (4.6–6.6)	6.4b (1.8–11.9)	<0.0001
Width/Depth, m	61a	7c	10cb	12b	11cb	<0.0001
Stream temperature, °C	19a (14–23)	18a (12–23)	19a (14–23)	18a (13–22)	19a (15–22)	0.8984
Dissolved oxygen, mg L^{-1}	7.0a (4.4–8.9)	6.6a (4.5–8.1)	7.1a (5.3–9.2)	6.6a (5.1–8.5)	7.0a (5.3–8.7)	0.5407
Velocity, m s^{-1}	0.09ab (0.01–0.27)	0.12ab (0–0.34)	0.15a (0.05–0.35)	0.07b (0.002–0.28)	0.07b (0.01–0.35)	0.0011
Sand/silt, %	68a (0–100)	57ab (0–93)	57ab (0–100)	43b (0–82)	53ab (29–84)	0.0130
Leaves, %	10b (0–36)	28a (0–100)	15ab (0–56)	15ab (0–48)	17ab (3–38)	0.0436
Wood/roots, %	<1d (0–6)	11bc (0–40)	5cd (0–24)	26a (0–70)	18ab (0–51)	<0.0001
Algae, %	0.5a (0–10.3)	0a (0)	0a (0)	0.1a (0–2.8)	0.4a (0–8.1)	0.7356
Pebbles, %	6c (0–50)	6c (0–30)	22a (0–83)	16ab (0–54)	11abc (0–28)	0.0004
Exposed streambed, %	15a (0–63)	<1b (0–4)	3b (0–18)	1b (0–12)	<1b (0–6)	<0.0001
Canopy opening (over stream), %	26a (5–52)	6b (2–23)	8b (3–20)	6b (1–17)	8b (0–27)	<0.0001
AFDM, kg m^{-2}	0.07c (0–0.20)	0.47a (0.05–1.33)	0.15bc (0.01–0.74)	0.20bc (0.03–0.90)	0.24ab (0.01–1.05)	<0.0001

† Numbers outside of parentheses are means; numbers inside parentheses are min. and max. values.

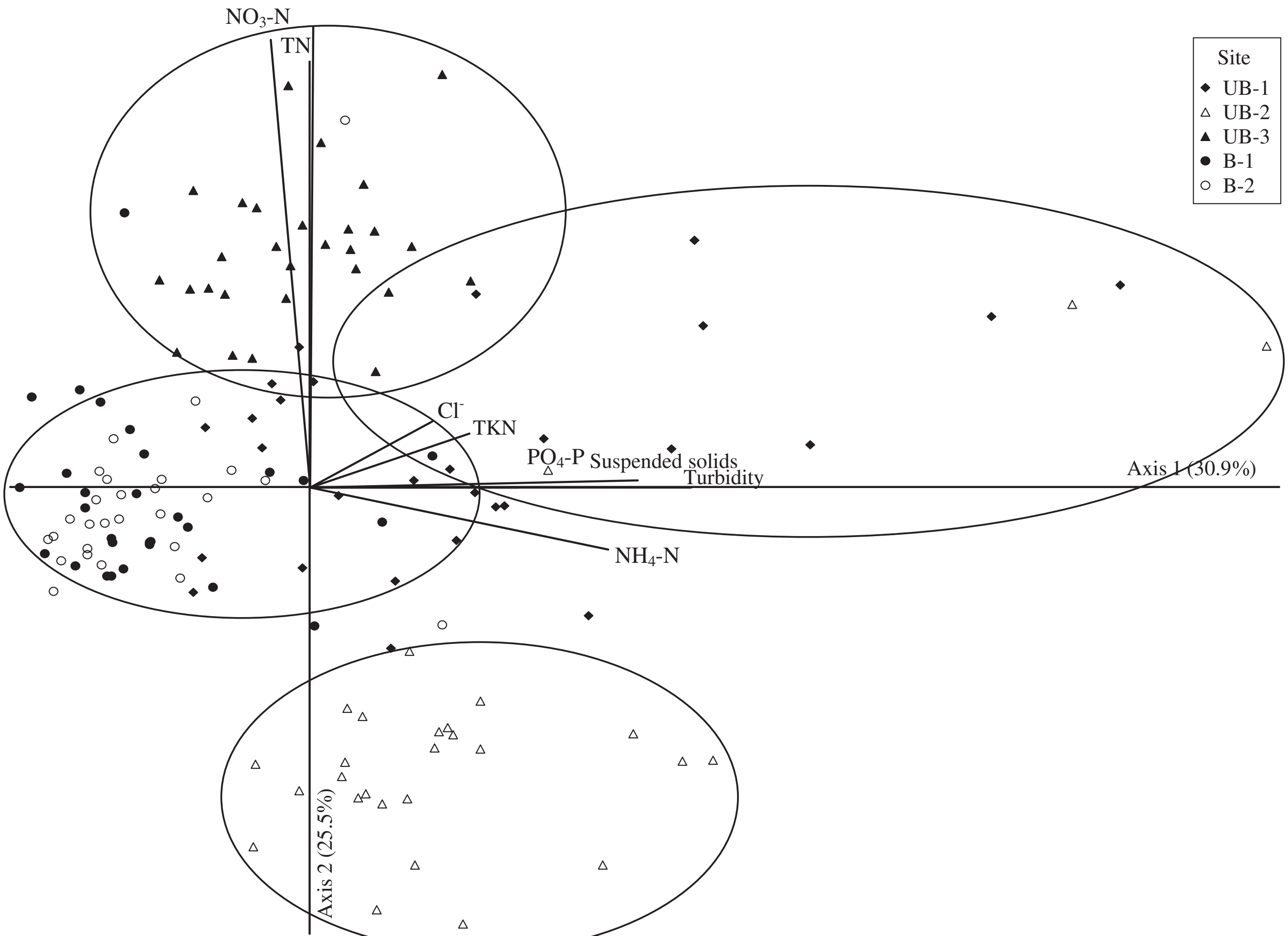

Fig. 3. First and second axes of the principal components analysis (PCA) for stream water quality parameters at five stream sites in Early County, GA. Points represent individual site/dates and vectors indicate water quality variables. Percent variation is indicated parenthetically on each axis. Ellipses were subjectively drawn around the central locations of points from each site to aid in discerning patterns.

Certain invertebrate taxa were found only at particular stream sites. Although many were low in abundance, they are worthy of mention. The buffered sites contained more unique taxa, many of which are sensitive to environmental disturbances (Lenat, 1993). The unique taxa at B-1 were: *Dixa* and *Dixella* (Dixidae), *Perlesta* (Perlidae), *Lepidostoma* (Lepidostomatidae), and *Molanna* (Molannidae). B-2 also had sensitive taxa such as *Lype* (Psychomyiidae) and *Hydroptila* (Hydroptilidae), as well as less sensitive invertebrates such as Curculionidae, *Trepobaks* (Gerridae), and *Habrophlebia* (Leptophlebiidae). The unbuffered sites had fewer numbers of unique taxa, mostly Coleoptera and Diptera, with none listed as sensitive. UB-1 had three taxa: *Hydrobiomorpha* (Hydrophilidae), *Prionocyphon* (Scirtidae), and one Hemipteran family, the Hebridae. *Ancyronyx* (Elmidae) and

Table 3. Mean, minimum, and maximum water quality measurements for all study sites, February 2002 to March 2003. (Kruskal–Wallis test with respective p-value).

Parameter	UB-1‡	UB-2	UB-3	B-1	B-2	p value
Apparent color, PtCo	79ab (21–393)	89a (17–369)	43b (3–101)	29c (0–235)	18c (0–102)	<0.0001
Suspended solids, mg L^{-1}	5.90a (1.42–27.08)	3.45a (0.43–11.51)	2.98a (0.20–7.14)	0.81b (0–5.21)	0.77b (0–3.67)	<0.0001
NO_3–N, mg L^{-1}	0.57b (0.46–0.70)	0.24c (0.17–0.42)	0.80a (0.71–0.97)	0.57b (0.50–0.80)	0.56b (0.29–0.67)	<0.0001
PO_4– P, mg L^{-1}	0.03a (0–0.08)	0.02ab (0–0.06)	0.01bc (0–0.07)	0.01c (0–0.05)	0.01c (0–0.05)	<0.0001
NH_4–N, mg L^{-1}	0.05a (0.02–0.11)	0.06a (0.03–0.18)	0.03b (0.01–0.009)	0.02c (0.01–0.007)	0.02c (0–0.06)	<0.0001
TKN, mg L^{-1}	0.23a (0.004–0.70)	0.21ab (0–0.56)	0.15abc (0–0.36)	0.14bc (0–0.30)	0.13c (0–0.75)	0.0005
Total N, mg L^{-1}	0.81b (0.52–1.21)	0.45d (0.19–0.92)	0.95a (0.79–1.13)	0.71c (0.52–1.07)	0.69c (0.52–1.32)	<0.0001
Cl^-, mg L^{-1}	3.51c (3.09–4.51)	3.95b (3.48–5.21)	4.27a (3.83–5.33)	3.49c (3.10–4.30)	3.59c (3.24–4.40)	<0.0001
F.coliform, col 100 mL^{-1}	281 (10–1100)	418 (60–1500)	532 (120–1800)	156 (28–760)	237 (23–3200)	N/A
F.streptococci, col 100 mL^{-1}	671 (16–2300)	1452 (100–12 000)	1593 (17–10 000)	1222 (40–13 000)	631 (13–5500)	N/A
pH†	5.06	5.32	5.12	5.03	5.05	N/A
Alkalinity†	8.60	9.11	5.45	3.81	4.75	N/A

† Denotes measurements taken once during the entire study.
‡ Numbers outside parentheses are means; numbers inside parentheses are min. and max. values.

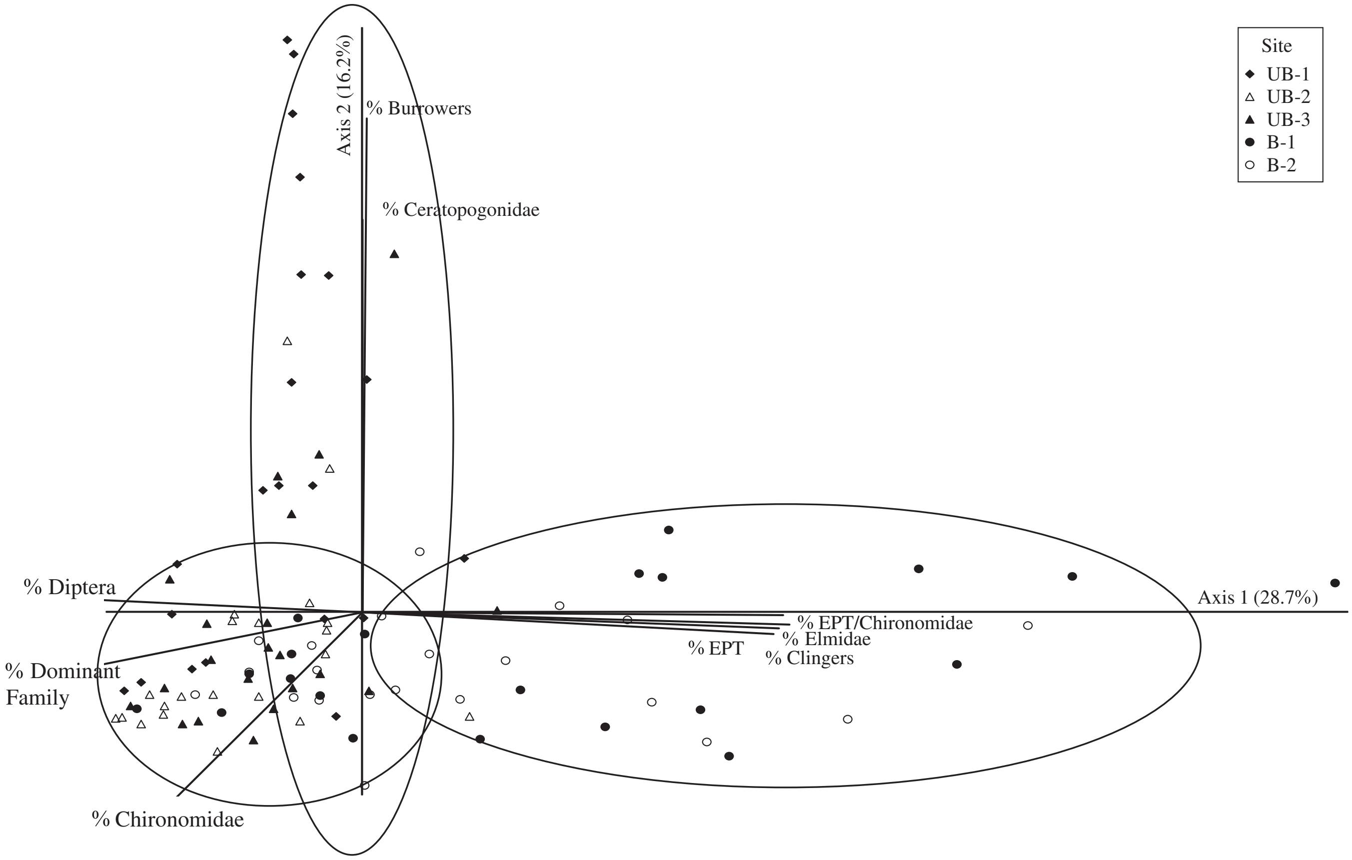

Fig. 4. First and second axes of the principal components analysis (PCA) for benthic macroinvertebrate metrics at five stream sites in Early County, GA. Points represent individual site/dates and vectors indicate metrics. Percent variation is indicated parenthetically on each axis. Ellipses were subjectively drawn around the central locations of points from each site to aid in discerning patterns.

Nemotelus (Stratiomyidae) were only found at UB-2, which also harbored noticeably greater numbers of Mollusca than the other sites (Muenz, 2004). UB-3 had no unique taxa.

Crustacean metrics including percent Crustacea, percent Amphipoda and percent Decapoda were significantly higher at the buffered sites ($p = 0.001$) (Fig. 5). No crustaceans were found at UB-1 and UB-3. Of all invertebrate samples, only one amphipod genus was identified, *Crangonyx*, and two decapod genera, *Cambarus* and *Procambarus*, all considered tolerant taxa. (Lenat, 1993) (Muenz, 2004). The metrics percent clingers, percent EPT, and percent Elmidae were also greatest at buffered sites ($p < 0.0001$) (Fig. 5). One metric was found lowest at buffered sites, percent Diptera ($p < 0.0001$) (Fig. 5). UB-1, the gully site, was characterized by the lowest percentages of many metrics including EPT, Chironomidae, and Elmidae ($p < 0.001$) (Fig. 5). Collectively, the unbuffered sites showed higher percentages of dipterans, with lower percentages of Elmidae and Crustacea (Amphipoda and Decapoda) ($p < 0.001$).

Eleven genera of chironomids were identified from the August 2002 sample as within three subfamilies: the Chironominae, Orthocladiinae, and Tanypodinae. Lowest diversity occurred at UB-1 (2 genera) and greatest diversity at B-2 (9 genera). Only one genus was found at all sites, *Polypedilum*, and UB-2 had a noticeably higher abundance of *Saetheria*, whereas highest numbers of *Tanytarsini* were found at B-2 (Muenz, 2004).

Amphibians

Species Richness

Twelve amphibian species were observed during the study. Three species of hylid frogs utilized tree pipes: Gray Treefrog [*Hyla chrysoscelis* (Cope)], Green Treefrog [*H. cinerea* (Schneider)], and the Squirrel Treefrog [*H. squirella* (Bosc)]. One Barking Treefrog [*H. gratiosa* (LeConte)] was found within a treepipe at UB-3, after the conclusion of this study. Four salamander species were observed during cover searches: Apalachicola Dusky Salamander [*Desmognathus apalachicolae* (Means and Karlin)], Southern Two-lined Salamander [*Eurycea cirrigera* (Green)], Southeastern Slimy Salamander [*Plethodon grobmani* (Allen and Neill)], and Red Salamander [*Pseudotriton ruber* (Latreille)]. American Bullfrog [*Rana catesbeiana* (Shaw)], Southern Leopard Frog [*R. sphenocephala* (Cope)] and Bronze Frogs [*R. c. clamitans* (Latrielle)] were observed at the sites during the course of the sampling. The ANOVA indicated differences in species richness ($p = 0.04$) although the means comparison (list test) was unable to resolve those differences. How-

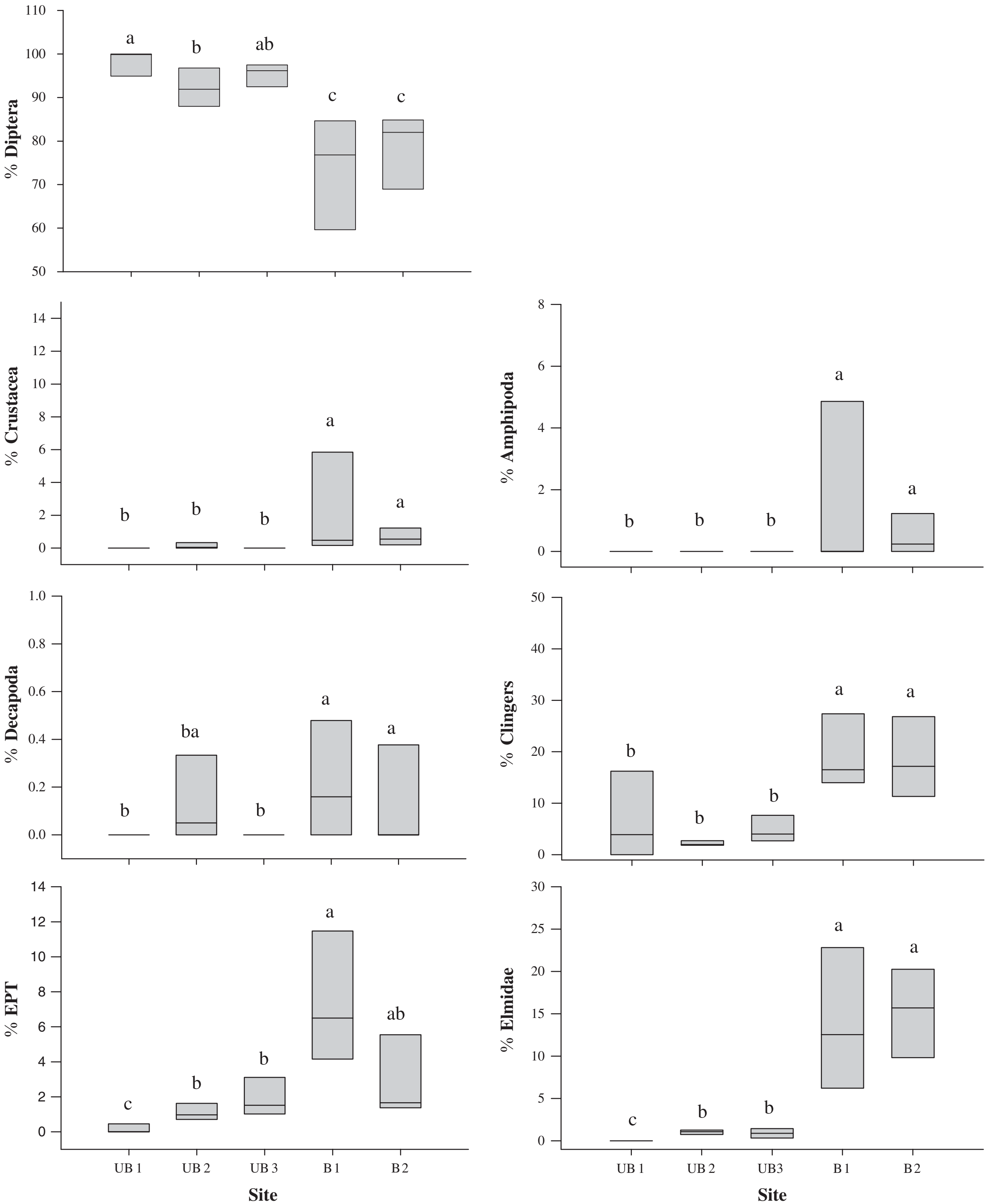

Fig. 5. Box plots of invertebrate metrics that showed significant differences among sites (median, 25th and 75th percentiles, maximum value, minimum value, and outliers). For each stream site and metric, boxes with different letters are significantly different (Kruskal–Wallis test, $p < 0.001$).

ever, species richness tended to be higher at the buffered sites.

Cover Board and Natural Cover Searches

Salamanders observed under cover boards and natural cover objects included: *Eurycea cirrigera* (67% of total observations), *P. ruber* (17%), *D. apalachicolae (*12%), and *P. grobmani* (4%). Observation rates were insufficient to compare cover boards to natural cover occurrences for salamanders. However, salamander abundance levels were similar between methods (39 individuals under natural cover objects and 44 individuals under artificial cover). Mean abundance of salamanders did not differ between sites for natural cover objects ($p = 0.19$) or for both cover types combined ($p = 0.41$); however, cover boards yielded slightly more salamanders at B-1 and B-2 than at other sites ($p = 0.005$). No differences in abundance of individual species were found when combining survey methods, or from individual methods, yet *P. ruber* was found in greatest abundance under natural cover at UB-1, and this difference approached statistical significance ($p = 0.06$). *Plethodon grobmani* and *D. apalachicolae* were found only at unbuffered sites. Overall, most salamanders (56% of total captures) were found at boards located closest to the stream (0m) ($p = 0.03$).

Larval Salamander Sampling

Forty-one Southern Two-lined Salamander larvae were captured in Hess samples. Correlations between larval abundance and stream measurements made at the time of the invertebrate sampling showed no strong relationships. Nor was a relationship detected between invertebrate and larval salamander abundance. However, the amount of organic matter present in the environment (AFDM), was correlated with the abundance of larval *E. cirrigera* ($r = 0.40$, $p = 0.05$).

Tree Pipe Surveys

Four hundred and eight individual treefrogs were observed. *Hyla squirella* was the most abundant species, comprising 87% of captures ($n = 355$), followed by *H. cinerea* (11%, $n = 45$) and *H. chrysoscelis* (2%, $n = 8$). All species were present at each site, except for UB-3, at which no *H. chrysoscelis* were found. There was no difference in treefrog abundance between buffered and unbuffered sites. Site B-2 had the highest mean number of treefrogs, whereas the lowest abundance occurred at B-1 ($p < 0.0001$).

Vegetation

Eighty-five plant species were identified in the floodplain forest [(40 trees/shrubs, 18 forbs, 16 vines, 10 graminoids, and 1 unidentified moss species)] (Wunderlin, 1998). Of all species, 4 were nonnative, and 18% considered ruderal (Wunderlin, 1998). Mean species richness was significantly higher at B-1 (42 species total), B-2 (43 species), and UB-1(34 species) than the other sites ($p < 0.0001$). Low vegetative similarity occurred between sites; however, the greatest affinity existed between sites that were similar in treatment, with B-1 and B-2 having the greatest similarity (46.6%) followed by UB-2 and UB-3 (26.3%).

Sweetbay magnolia [*Magnolia virginiana* (L.)] was the dominant canopy species at both of the buffered sites and at UB-3, while hazel alder [*Alnus serrulata* (Ait.)Willd] was the dominant species at UB-1 and the tulip tree, *L. tulipfera* at UB-2. Mean density of canopy trees was greatest at the buffered sites and UB-1; however, UB-1 was dominated by *A. serrulata,* a species noted for its shrub-like, and multiple branching structures, which perhaps also contributed to the low basal area of 19.65 m^2 ha^{-1} measured at this site (Table 4). The highest basal area was 86.28 m^2 ha^{-1}, measured at UB-2.

Floodplain width ranged from 10 m (UB-1) to over 30 m (UB-2) (Table 4). Most sites had steep, narrow stream terraces on at least one side of the stream, contributing to large variation in percent slope. Percent canopy cover within the riparian zone was similar to that measured over the stream channel at all sites, except UB-1 which consistently had the lowest canopy cover ($p < 0.0001$) (Table 4). Soil compaction in the riparian area was highest at sites UB-2 and UB-3, and lowest at sites B-1, B-2, and UB-1 ($p < 0.0001$). Two variables that de-

Table 4. Summary of riparian structural characteristics: Mean, minimum, and maximum measurements at each site. For each site and parameter, values with different letters are significantly different (Kruskal–Wallis test with respective *p* value).

Parameter	UB-1†	UB-2	UB-3	B-1	B-2	*p* value
Basal area, m^2 ha^{-1}	20	86	42	54	67	N/A
Floodplain width, m	15 (10–23)	30 (23–36)	20 (15–23)	25 (18–30)	25 (24–27)	N/A
Slope, %	22a (7–29)	18a (5–48)	12a (9–18)	15a (6–27)	18a (5–38)	0.9252
Canopy opening (riparian), %	21a (4–54)	7b (0–29)	8b (1–24)	5b (1–16)	7b (1–23)	<0.0001
Sand cover, %	79a (61–100)	74a (50–92)	49ab (15–77)	15b (0–31)	12b (0–31)	0.0004
Leaf litter cover, %	13b (0–23)	25b (8–46)	42ab (23–77)	83a (69–100)	75a (46–100)	0.0003
Vegetative cover, %	4a (0–8)	0a	2a (0–8)	2a (0–8)	10a (0–15)	0.1429
Robel pole coverage, %	50a (0–100)	4b (0–18)	2b (0–14)	14a (3–31)	23a (2–68)	0.0001
Woody, %	4a (0–8)	0a	2a (0–8)	0a	4a (0–8)	0.2930
CWD frequency, no. per 100 m^2	20a (7–32)	7a (3–8)	19a (4–34)	13a (11–18)	10a (6–14)	0.1257
CWD volume, m^3	1.39a	0.53a	1.66a	0.58a	0.54a	0.0711
Tree frequency, no. per 100 m^2	17a (8–40)	8b (6–10)	7b (3–13)	15a (12–18)	12ab (10–15)	0.1373
Snag frequency, no. per 100 m^2	1a (0–2)	<1a (0–2)	<1a (0–2)	2a (1–3)	<1a (0–2)	0.0599
Soil temperature, °C	17.3a (11.3–25)	16.6a (9.4–26)	17.4a (11.3–25.9)	17.7a (12.1–24.9)	18.5a (12.4–25.3)	0.7029
Soil compression, kg cm^{-2}	0.5c (0–4.5)	2a (0–4.5)	1b (0–4.5)	0.5c (0–1.75)	0.4c (0–3.3)	<0.0001

† Numbers outside parentheses are means; numbers inside parentheses are min. and max. values.

scribed the ground layer, percent woody and percent vegetative cover, did not significantly differ at any of the sites. Percent leaf matter cover was highest at the buffered sites, whereas sand cover was highest at unbuffered sites UB-1 and UB-2 ($p < 0.001$). Vertical structure of the understory also was highest at both buffered sites, and at UB-1 ($p = 0.0001$).

DISCUSSION

Site Differences

Numerous studies in North America have documented the positive effects of buffers on water quality and aquatic biota (e.g., Lowrance et al., 1983, 1984; Peterjohn and Correll, 1984; Jacobs and Gilliam, 1985; Owens et al., 1996; Line et al., 2000; Thomas, 2002; Vellidis et al., 2003). This study showed significant positive effects of buffers on physical, chemical, and vegetative measures, as well as benthic macroinvertebrate communities. Amphibian abundance and diversity did not show differences between treatments, yet greater abundance of larval salamanders suggests instream habitat quality may be better at the buffered sites (i.e., greater habitat heterogeneity).

We found that variability among streams in the parameters measured could be attributed to the geographic location as well as surrounding land uses (past and present) (see Weigel et al., 2000; Sarr, 2002). Even the buffered sites had a long history of disturbance from logging and cattle activity and may still be recovering. One of the unbuffered sites (UB-1) in particular had a long history of disturbance (i.e., cattle intrusion), but major impacts seemed to originate from an upland eroding gully. However, this site appeared similar to the buffered sites (i.e., vegetative measures). Lastly, sites UB-2 and UB-3 were very similar in most measurements, reflecting similar patterns of riparian disturbance and upstream land use.

Buffer Effects on Streams and Riparian Zones

Contrary to other studies (Kauffman and Krueger, 1984; Belsky et al., 1999; Maloney et al., 1999; Strand and Merritt, 1999), we did not observe increases in stream temperature and channel width, and decreases in dissolved oxygen at cattle-impacted sites. Water quality at buffered streams appeared more stable than unbuffered streams, with lower peak flows in response to rainfall and lower and less variable bacterial, nutrient, and sediment concentrations. Similar results were noted by Thomas (2002), who found that restricting cattle access to streams in the Georgia Piedmont resulted in decreases in nutrients (17 to 72%) and FCs (95%). Elevated nutrients, apparent color, sedimentation, and greater percent sand/silt within the channel bed were also found in Coastal Plain stream sites influenced by agriculture (Davis et al., 2003). Nutrient and bacteria concentrations in this study were generally lower compared to other studies examining intensively grazed areas (Barker and Sewell, 1973; Doran et al., 1981; Thomas, 2002). Stream buffers also seemed to promote more diverse stream habitat, with more wood/roots and organic matter and less sand/silt within the stream channel.

A protected riparian zone in an agriculturally influenced area is widely viewed as a critical component in the protection of surface water quality, instream habitat, and the biotic integrity of aquatic ecosystems (Roth et al., 1996; Stevens and Cummins, 1999; Paine and Ribic, 2002). Detritus, a critical habitat and principal component of the food base for many streams, is produced by the terrestrial and emergent plant communities of the surrounding riparian areas (Felley, 1992). Most riparian areas with cattle access have decreased leaf litter accumulation, higher levels of soil compaction, and a greater percentage of bare ground (Belskey et al., 1999). High levels of soil compaction, a common result of cattle trampling, greatly reduces plant productivity and plant cover (Kauffman and Krueger, 1984; Belskey et al., 1999). Our study showed this pattern; however, even among similar treatments vegetation structure and soil stability were shaped and affected by factors other than cattle access/exclusion. For example, soil compaction was not the dominating factor shaping plant structure at UB-1 the gully site, where moisture-laden, loose sand resulted from soil erosion. It was apparent that effects from the gully on riparian soil structure were localized, and that stable substrate at this site was available either upstream or outside the study area affected by the gully. Perhaps this pattern of increased stability downstream from the gully had an effect on the riparian plant community, and thus similar percentages of vegetative cover were observed at UB-1 and buffered sites.

Biotic Responses to Buffers

Macroinvertebrates

A number of variables determine the composition of aquatic invertebrate communities; one of which, stream habitat, is strongly influenced by streambed substrate and riparian vegetation (Strand and Merritt, 1999). Invertebrate communities are also influenced by regional land characteristics (i.e., geology) and by the presence of disturbances such as those resulting from grazing activities within the riparian area (Strand and Merritt, 1999). Within the Gulf Coastal Plain, streams are typically sandy-bottomed and low to medium gradient, where one would expect Chironomidae to be the dominant taxa (e.g., Felley, 1992; Gregory, 1996; Davis et al., 2003). The streams within this study were somewhat unusual in that cobble and coarse particles were present, typically a characteristic of more northern streams. Even so, over 70% of taxa identified within this study were Chironomidae. Chironomids have been found in association with streams having higher levels of pollution (i.e., sedimentation), and tend to increase in abundance with disturbances such as agriculture (Kerans et al., 1992; Clements, 1994; Strand and Merritt, 1999; Davis et al., 2003). In this study, abundances of chironomids were not found to be different among treatments, nor did they characterize unbuffered sites as has been reported (Davis et al., 2003; Reed, 2003). However, diversity within the Chironomidae differed among sites, with highest diversity and number of sensitive chironomid taxa at the buffered sites.

The impacts of agriculture (i.e., increased sediment, nutrients, and oxygen demand) can also result in greater macroinvertebrate diversity by eliminating less tolerant organisms, providing for a shift to more environmentally tolerant generalists (Richards et al., 1993; Delong and Brusven, 1998). We found differences in richness estimates between sites; however, highest richness of EPT and all taxa combined occurred at sites with the least disturbance (buffered sites) and greatest habitat heterogeneity. Numbers of 'unique' (and sensitive) taxa were also greater at buffered sites which has been predicted (i.e., Lenat, 1984). Richness estimates were comparable to recent studies within the Gulf Coastal Plain (Gregory, 1996; Davis et al., 2003); however, estimates were influenced by the lack of taxonomic resolution of some groups, predominantly the Ephemeroptera and Plecoptera.

Higher percentages of more sensitive groups such as EPT, Crustacea (Decapoda and Amphipoda), and clingers were found to characterize the buffered sites, whereas percentages of more tolerant taxa, the dipterans and dominant family, were highest at most of the unbuffered sites (Lenat, 1984; Davis et al., 2003). Higher percentages of Ceratopogonidae and burrowers appeared to characterize the gully site (UB-1), which in the case of burrowers are expected to increase with greater abundance of silt and sand (Barbour et al., 1999). One metric in particular, percent Elmidae, showed strong differences among treatments, with higher percentages at the buffered sites. The Elmidae (Riffle beetles), are inhabitants of the swifter portions of streams such as riffles, which have coarse sediments and high oxygen concentration (Merritt and Cummins, 1996). Coleopterans display a wide range of tolerances to disturbances (Gilbert, 1989) and in a study by Lenat (1984) Elmidae were not found to be affected by agricultural runoff. However, many sensitive taxa occur within the Elmidae, such as *Microcylloepus* and *Stenelmis* which were most common in our study.

Of the metrics examined, percentages of burrowers, clingers, crustaceans (amphipods and decapods), EPT, EPT/Chironomidae, dipterans, and Elmidae were most useful for perennial streams within the Fall Line Hills District of the Coastal Plain. Although taxonomic resolution of taxa groups within the EPT was low, richness estimates were different among sites, and abundances of these taxa (35 ind m^{-2}) were much higher than other Coastal Plain studies (0 to 5 ind m^{-2}) (Gregory, 1996; Davis et al., 2003). Percent Elmidae, a metric suggested by this study, appeared to be appropriate and captured differences in the study streams. Elmids were reliably collected and occurred in large enough quantities for comparison among sites (Rosenberg and Resh, 1993; Barbour et al., 1999).

Amphibians

Salamander larvae were captured in significantly higher numbers at the buffered sites, but no difference was observed in the number of adults in the riparian areas of buffered and unbuffered sites. No pattern in abundance of treefrogs was observed between treatments, although numbers of observations were highly variable among sites. In fact, buffered sites displayed both the highest and lowest abundances. A number of variables such as tree type (hardwood or pine), DBH, distance to breeding sites, and midstory vegetation appear to influence the capture of treefrogs within PVC pipes (Boughton et al., 2000). Although structural differences in DBH and tree species were not apparent in this study, canopy cover was highest at the unbuffered gully site. Little is known about the life history of wetland-breeding amphibians away from breeding sites, but it is possible that there were enough structural differences among sites that artificial refugia offered better habitat than their natural environment. Proximity to and availability of breeding areas is another factor that may explain variation in treefrog abundance at buffered sites. Location of breeding sites was not considered in this study, but perhaps distance or simply the presence/absence of sites coupled with a fragmented landscape were factors influencing treefrog abundance.

Amphibian abundance and richness have been shown to be lower in disturbed habitat than in undisturbed habitat (Chazal and Niewiarowski, 1998). However, in one study (Homyack and Giuliano, 2002) there was no difference in abundance and species richness of herpetofauna at fenced (1 to 2 yr) versus unfenced streams, even though fenced streams had greater amounts of litter cover, an important habitat component for amphibians (Petranka, 1998). The authors attributed the lack of differences in herpetofauna to similarities in water quality and the need for a longer recovery time for vegetative diversity and structure (>2–4 yr).

In this study, streams had been fenced for >20 yr and showed substantial differences in vegetative, chemical, and physical stream parameters, and macroinvertebrate communities. Yet differences in salamander abundance within the riparian zone were not apparent. Important components of salamander habitat are suitable woody debris and leaf litter (Pough et al., 1987; Petranka et al., 1993). Leaf litter was highest at the buffered sites in this study, but coarse woody debris did not differ between treatments. Canopy cover and soil/air temperature also did not differ between treatments, suggesting that twenty years is sufficient for recovery of vegetative and chemical parameters, yet may be insufficient for the recovery of salamander populations. Petranka et al. (1993) and Pough et al. (1987) noted that recovery required about 60 yr in areas disturbed by clearcutting, and Hyde and Simons (2001) also suggested that the effects of habitat disturbance (logging or agriculture) on salamander diversity and abundance may persist for more than 60 yr.

Southern Two-lined Salamander larvae were the only amphibians that exhibited higher abundances at buffered sites in this study. Lower sediment levels and greater heterogeneity within the streambed may help explain these differences. Smith and Grossman (2003) examined microhabitat use by larval *E. cirrigera* in Georgia Piedmont streams, and found a close association between larval abundance and substrata heterogeneity. The presence of pools at the buffered sites, which were lacking at the unbuffered sites, may also have provided suitable habitat for larvae (Petranka, 1984).

The Use of Macroinvertebrates and Amphibians to Detect Buffer Effects

As aquatic ecosystems continue to be degraded from land use disturbances such as agriculture, the need to assess the condition of such systems increases (Fore et al., 1996). Amphibian monitoring is important because threats to their populations are two-fold within an agricultural landscape, including alterations of upland habitat crucial to hibernation and migration to and from breeding sites, while also facing the threat of pollution within the stream from agrochemicals (Beebee, 1996). Understanding the changes in aquatic biota (such as amphibians and macroinvertebrates) in relation to disturbance forms the basis of many biomonitoring methodologies used in aquatic ecosystems (Cao et al., 1997). Resident aquatic biota such as macroinvertebrates and fish have been widely adopted to evaluate water quality because they are considered good indicators of localized disturbances, revealing and expressing the consequences of human activities (Lenat et al., 1980; Abel, 1989; Barbour et al., 1999).

Amphibians exhibit traits of a good indicator taxa, and their inclusion in monitoring and conservation programs has been suggested in many studies (i.e., Dunson et al., 1992; Welsh and Ollivier, 1998; Rocco et al., 2004; Boughton et al., 2000; Micacchion, 2002; Semlitsch, 2003). However, efforts to establish such programs have been hindered by uncertainties associated with their varied and complex life histories, detectibility, natural population fluctuations, species-specific habitat requirements, and a poor understanding of sampling technique accuracy (Hyde and Simons, 2001; Dodd, 2003). In this study overall observations of amphibians were low, but perhaps if a larger sample area had been used differences in abundance would have been detected.

Ultimately, a metric or tool used in biological monitoring programs must reflect specific and predictable responses to changes in the environment. *Eurycea cirrigera* (larvae) and *Hyla squirella* show promise as 'indicator species,' for they were found consistently at both buffered and unbuffered sites and in high enough numbers for comparison. The Hess sampler shows promise as a technique for quantifying larval amphibian densities because sampling was rapid and quantifiable.

It is suggested that the most effective approach to monitoring aquatic health is one that is synthetic, adopting a group of relevant metrics, while recognizing the need for approaches that consider the 'many attributes of biological condition simultaneously' (Karr and Chu, 1999). The inclusion of both macroinvertebrates and amphibians in biological assessment would concurrently provide for a more complete and strengthened assessment of a site's condition. It would also offer a broader spectrum of responses and signals about the effects of human activities, as both groups are highly sensitive to disturbances, and are linked to aquatic and terrestrial environments for their life history stages. If we acquire the missing information needed to incorporate amphibians into monitoring programs, another biological 'tool' would be available, especially in the case where other species, such as fish, may be rare or absent.

ACKNOWLEDGMENTS

This research was supported by the following organizations and agencies: the Joseph W. Jones Ecological Research Center, the University of Georgia, and the U.S. Environmental Protection Agency. We are particularly grateful to the energy and efforts of Frankie Hudson, Jan Battle, Raynie Bambarger, Lee Anne Bledsoe, Eamonn Leonard, Anna Liner, Brian Yahn, Slaton Wheeler, and Shannon Hoss for assisting with sampling activities conducted during the project. We thank the two anonymous reviewers and the editorial staff of the *Journal of Environmental Quality* who provided valuable comments on earlier versions of this manuscript. We also extend our gratitude to the Brownlee family for permitting access to the study area and sharing their knowledge of the land.

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