Land use influences on benthic invertebrate assemblages in southern Appalachian agricultural streams.

Barbara Loraine Bennett

Thesis submitted to the Faculty of the
Virginia Polytechnic Institute and State University
in partial fulfillment of the requirements for the degree of

Master of Science
in
Biology

E.F. Benfield, Chair
J.R. Webster
J.R. Voshell
J.S. Harding

14 May 1998
Blacksburg, Virginia

Keywords: Stream, Macroinvertebrate, Agriculture, Riparian, Historical Land Use, Diversity, GIS, Community Structure

Land use influences on benthic invertebrate assemblages in southern Appalachian agricultural streams.

Barbara Loraine Bennett

(ABSTRACT)

I investigated the role of land use in structuring benthic invertebrate assemblages in agricultural streams in the French Broad River drainage in western North Carolina. I sampled six agricultural streams (3 with cleared headwaters and 3 with forested headwaters) at three points along a gradient (headwaters, a midpoint, and a downstream site). At each site, I measured a variety of physico-chemical parameters, including temperature, chlorophyll *a*, discharge, nutrients, and suspended solids. Invertebrates were sampled at all sites in October 1996 and April 1997. Riparian vegetation was assessed for each site at mutiple spatial scales using GIS data from the 1950s, 1970s, and 1990s. Forested agricultural (FA) streams had more riparian vegetation than cleared agricultural (CA) streams in both the 1950s and the 1970s. Cleared agricultural streams had less organic matter, more primary production, higher nitrates, and warmer temperatures than FA streams. Total and EPT taxa richness was greater in FA streams. Pollution-sensitive Plecoptera were relatively more abundant in FA streams, while tolerant Diptera were more abundant in CA streams. High diversity and Plecoptera abundance was related to high habitat quality, more riparian vegetation, low nitrates, and low summer temperatures. Higher invertebrate diversity was related to the land use 25-50 years as well as the current land use (forested, moderate agriculture, or heavy cattle impact). These results indicate a long-term legacy of agricultural influences on stream invertebrate assemblages.

This study was supported by two Sigma Xi Grants In Aid of Research grants and one Virginia Academy of Science Small Project Research Grant awarded to the author. Additional support was provided by Virginia Tech Biology Department, NSF-DEB9632854, and Coweeta Hydrologic Laboratory.

ACKNOWLEDGEMENTS

First, I owe many thanks to Dr. Fred Benfield for being a far more than “tolerable” advisor. Fred, thank you for your support and for always finding time to listen and give feedback. I appreciate your editorial comments on the many drafts that traveled across your desk. I am also grateful to my committee members, Dr. Jack Webster, Dr. Reese Voshell, and Dr. Jon Harding, for providing not only guidance but also enthusiasm for my project. Dr. Bob Jones also provided statistical guidance. The Virginia Tech “Stream Team,” in all its various combinations of graduate students over the years, has been a great source of discussions, idea generation, and advice. It was nice to have a lab to call home and to have company while I stared into the microscope all those hours.

My thanks also go to the many landowners who graciously allowed access to the streams in this study; I appreciate their hospitality. Many people helped with the field work on sampling trips. I thank Matt McTammany, Alicia Schultheis, Jon Harding, and Matt Neatrour for the many long hours they spent driving around western North Carolina and bending over equipment in streams. I also appreciate Wendy Schwieger McTammany for taking part of her summer vacation to help with the field work, and Bob and Helen McTammany for providing a comfortable place to come home to at the end of the long field days. Many undergraduates provided help sorting the endless samples in the lab, especially Mollie Whitcraft who was there to help from the first equipment preparation to the very last sample to be sorted. I am also indebted to my brother, Brad Bennett, who graciously spent part of his vacation wrapping rocks with aluminum foil.

Finally, I could never have come so far without the love and support of my friends and family. I am delighted to have made friends with both Alicia Schultheis and Matt McTammany. They have been with me through tears and laughter, and were unending sources of strength. Jen Schaeffer was a great friend as well as upstairs neighbor, and I appreciate her for providing balance and sanity in my life when things were hectic. Kristin Meirose and Bethany Looney were terrific long-distance supporters, and I am thankful to both of them for being friends that help me to be fascinated with the great outdoors. Dr. Joe Holomuzki got me started in stream ecology while I was at Transylvania University, and he remains an important advisor even now. And last, but certainly not least, I am very grateful to my parents, Jim and Linda Bennett, for giving me my start in this field by letting me play in the creek to my heart's content as I was growing up. They have been my number-one fans through all the ups and downs, and without their love and encouragement all that I have accomplished would have been impossible. I have been blessed with a wonderful, supportive family.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

Land use influences on benthic invertebrate assemblages in southern Appalachian agricultural streams.

INTRODUCTION

Benthic community structure varies considerably from stream to stream and is influenced by a wide range of physico-chemical characteristics, including substrate size and complexity, temperature and flow regimes, nutrient levels, pH, hardness, and riparian vegetation (Hynes 1970; Hynes 1975; Cummins *et al.* 1984; Townsend *et al.* 1983; Allan 1995). These stream characteristics are in turn largely dependent upon the properties of the watershed the stream drains, including geomorphology, hydrology, and land use (Hynes 1975; Allan 1995). The River Continuum Concept (RCC) suggests that stream community structure and function shifts both in response to the shift in physico-chemical properties of the drainage network along the gradient from headwaters to mouth and in response to upstream conditions (Vannote *et al.* 1980; Minshall *et al.* 1985). Although developed for the entire river system gradient from small headwater streams to large rivers, the tenets of the RCC may also apply at smaller spatial scales, such as the passage of small, first- to third-order streams from forested headwater regions into cleared, agricultural areas.

Converting forested land to agriculture results in a complex interaction of factors which may severely disturb stream communities. Central to the differences between forested and agricultural land is the removal of riparian vegetation. Allochthonous inputs (*e.g.*, leaves and wood) are the energy base in forested streams (Minshall 1967; Hynes 1975), tightly linking streams with the land. This linkage is severed in open streams, and the energy base is shifted to autochthonous primary production due to both decreased riparian inputs and increased light exposure (Vannote *et al.* 1980; Minshall *et al.* 1983; Hauer and Hill 1996; Corkum 1996). Based on this shift in energy sources, it seems intuitive that a corresponding shift in functional feeding groups (Cummins 1973) would occur, *i.e.*, shredders (convert leaves into smaller fragments) would decrease and scrapers (ingest periphyton from rocks) would increase. However, investigators have encountered varying responses by functional feeding groups to differences in land cover. The intuitive shift from shredders to scrapers in open streams has been documented in some studies (Minshall *et al.* 1983; Tuchman and King 1993; Reed *et al.* 1994), but others have found either no apparent trend (Dudgeon 1989) or nearly opposite trends (Giller and Twomey 1993; Hawkins *et al.* 1982).

The removal of riparian vegetation also eliminates wood inputs to streams. Large woody debris creates retention structures that trap organic matter and sediments and alter the direction and energy of stream flow (Bilby 1981; Ehrman and Lamberti 1991). Debris dams provide greater substrate stability to streams which reduces the impact of flooding on invertebrate fauna (Cobb *et al.* 1992; Palmer *et al.* 1996). The impact of floods on agricultural streams lacking debris dams is compounded by the increased flow variability ("flashiness") that occurs as a result of increased surface run-off and reduced evapotranspiration from riparian vegetation.

Agricultural streams experience higher temperature maxima due to decreased shading compared to forested streams and are likely to have greater temperature variability (Hynes 1970; Quinn *et al.* 1992). Small, low-order streams are particularly vulnerable to large temperature variations in open, agricultural systems (Smith and Lavis 1975; Wiley *et al.* 1990). Temperature interacts with food quality and availability to influence benthic macroinvertebrate assemblages. Each species has an optimum temperature range which balances optimum temperatures for each developmental stage (Sweeney *et al.* 1986; Söderström 1988). Thus assemblages found in agricultural streams may be structured, in part, by limiting the types of invertebrates that are able to survive the highly variable, summer-warm temperature conditions.

Perhaps the most serious impact of agriculture on streams is the addition of sediment through a variety of processes, including erosion of top-soil and destabilization of stream banks due to channelization and stock access (Lenat 1984; Quinn *et al.* 1992; Waters 1996). Sediment in streams can occur either as suspended solids contributing to stream turbidity or as deposited sediments. Waters (1996) found that suspended sediments can impact stream fauna directly (by abrasion, interference with filter-feeding mechanisms, or reducing visual feeding efficiency) or indirectly (by reducing light levels to the point of triggering drift behavior). Additionally, Graham (1990; as cited by Ryan 1991) demonstrated that clay-sized suspended inorganic sediments stick to periphyton surfaces, reducing the quality of the periphyton as a food source. Fine sediments impact stream fauna by settling into coarse substrates and filling interstitial spaces, thereby reducing substrate complexity (Minshall 1984; Richards *et al.* 1993). Lenat *et al.* (1981) found that added sediment reduced available rock habitat, resulting in reduced benthic invertebrate density although community composition remained unchanged. However, periods of low flow may allow deposited sediments to be colonized with an assemblage suited to that habitat.

Finally, run-off of pesticides, fertilizers, and organic pollution from agricultural lands also contributes to chemical differences between forested and agricultural streams. Chemical concentrations in general were higher in streams in basins which were 15-55% agricultural than in forested basin streams in North Carolina (Simmons and Heath 1979). Additionally, in agricultural basins total phosphorous concentrations were 2-13 times higher in storm runoff than at baseflow, and total nitrogen concentrations were 5 times higher (Simmons and Heath 1979). Townsend *et al.* (1983) found streams draining agricultural land tended to have higher pH than wooded streams. Inorganic nitrogen in New Zealand pasture streams was present at concentrations five times higher than those in native forests (Quinn *et al.* 1994). Richards *et al.* (1993) found that intolerant taxa were associated with sites with lower nutrient levels, although the relationship was stronger with sites with higher habitat scores. A negative correlation between levels of oxygen and nitrogen indicate the possibility of over-enrichment in some streams (Townsend *et al.* 1983). In streams impacted by non-point pollution such as agricultural inputs, periods of low flow may concentrate nutrients to the point that the richness of sensitive taxa decreases (Lenat 1990).

The physical and chemical impacts of agriculture on streams described above combine to alter stream biotic assemblages (Dance and Hynes 1980). Several studies have found that overall benthic invertebrate abundance tends to be higher and taxa diversity tends to be lower in open, agricultural streams than in forested streams. The abundance of sensitive EPT taxa also tends to be lower in open streams (Quinn *et al.* 1984; Lenat 1984; Harding and Winterbourn *et al.* 1995). However, not all studies have shown such clear trends (*e.g.*, Townsend *et al.* 1983; Giller and Twomey 1993; Reed *et al.* 1994).

The specific objectives of this study were: (1) to examine differences in taxonomic diversity and benthic assemblage composition along a forest to agriculture land-use gradient; (2) to determine whether there are differences in invertebrate taxa richness and density between cleared and forested agricultural streams; and (3) to identify the physico-chemical characteristics important in influencing differences in invertebrate assemblages between these stream types. To address these objectives, I compared benthic invertebrate assemblages and physico-chemical variables in three agricultural streams with forested headwaters to three similar streams with agricultural headwaters (Figure 1).

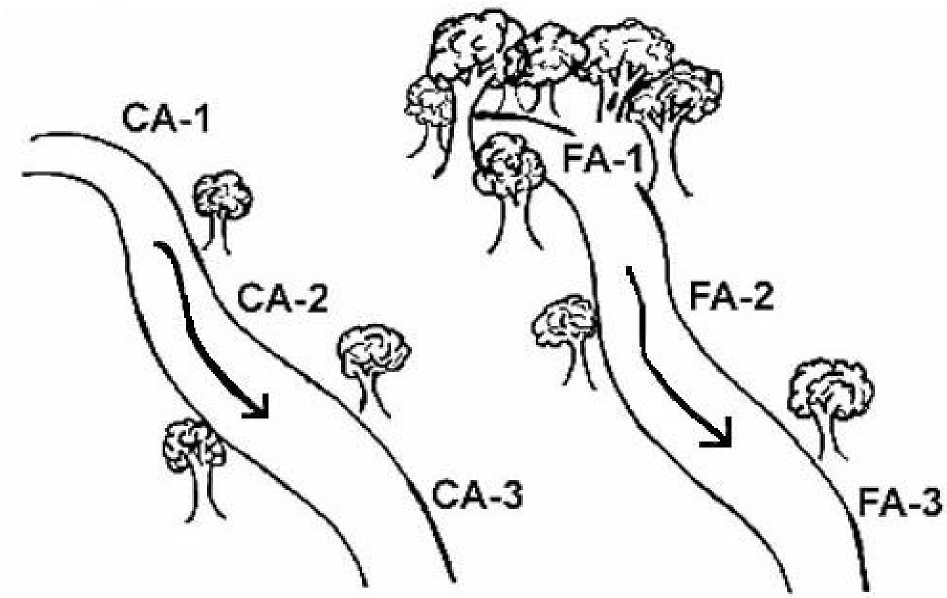


Figure 1. Schematic diagram of study design. There were two types of agricultural streams: cleared agricultural (CA) streams and forested agricultural (FA) streams. Three sites were sampled on each stream: a headwater site (1), an intermediate site (2), and a downstream site (3).

MATERIALS AND METHODS

Site description

The study sites were three second- to third-order agricultural streams with forested headwaters and three similar agricultural streams which were cleared to the headwaters (Figure 2). The six streams are located in the French Broad River Basin near Asheville, North Carolina, within Buncombe and Madison counties in the Blue Ridge Province of the southern Appalachian mountains. These counties together contain over 2500 kilometers of stream length with an average stream density of 8.97 m/hectare (Southern Appalachian Man and the Biosphere [SAMAB] 1996). This area is a primarily micaceous schist and gneiss geologic zone, which is generally characterized by slightly acidic water with low dissolved ions (Simmons and Heath 1979).

The three forested headwater agricultural (FA) streams originated in forests which gradually opened to cleared, agricultural areas: Heck Creek (a tributary of Walnut Creek), North Fork of Ivy Creek, and Upper Flat Creek. Heck Creek and Upper Flat Creek originated in privately owned forests, and North Fork of Ivy Creek originated in Pisgah National Forest. The three cleared headwater agricultural (CA) streams drained areas which have largely been cleared to the headwaters: North Turkey Creek, Pawpaw Creek, and Willow Creek (a tributary of Sandymush Creek). Agricultural land-use along all streams includes cattle and horse pasture, dairy operations, and row crops such as corn and tobacco. Three sites per stream were sampled representing the gradient

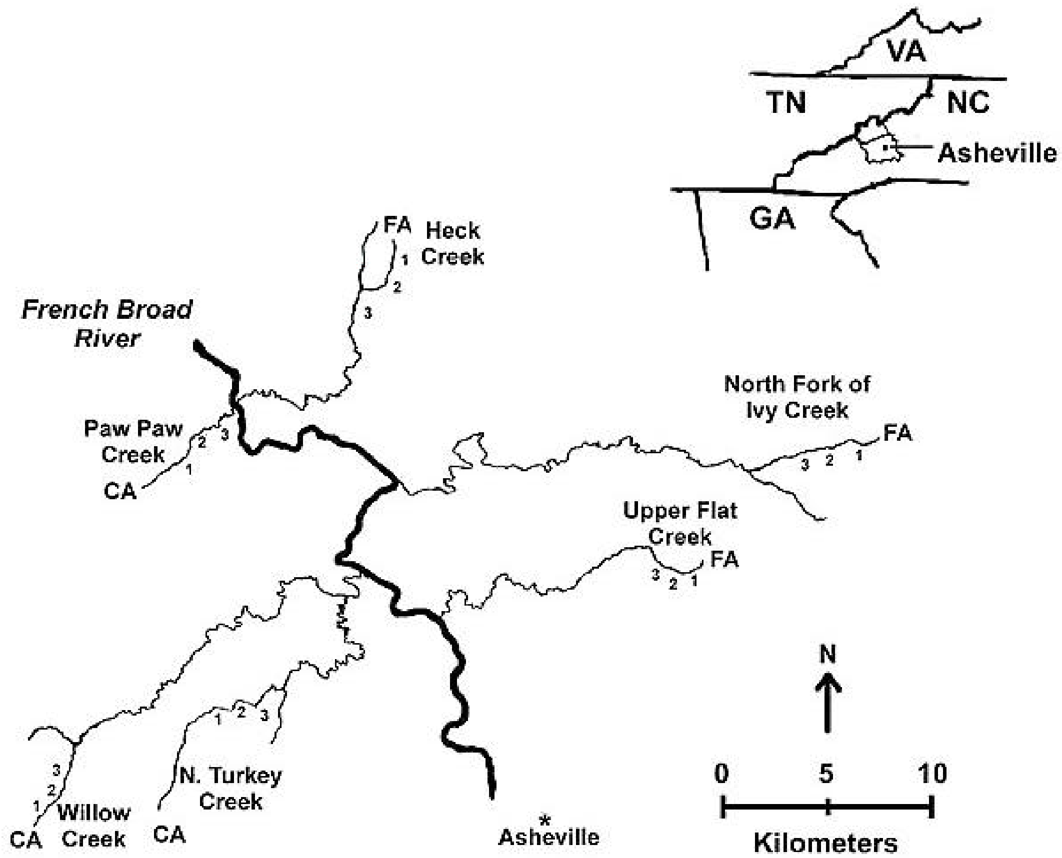


Figure 2. Study location in western North Carolina, French Broad River drainage. Upstream (1), intermediate (2), and downstream (3) sites are indicated on each stream. (FA) denotes agricultural streams with forested headwaters and (CA) denotes cleared agricultural streams.

from the headwaters (Site 1) to an intermediate site (Site 2) to a downstream site (Site 3; Figure 1). The 18 sites along these six streams are indicated in Figure 2.

Sampling regime

The six streams were sampled at three sites (headwaters, intermediate, and a downstream site). Invertebrates were sampled in October 1996 and April 1997. A range of physical and chemical variables were measured up to six times during the study. These variables included stream discharge, substrate size and heterogeneity, benthic organic matter, nutrient concentrations, and periphyton biomass. Temperature was recorded continuously by dataloggers at each intermediate site (Site 2) and by maximum-minimum thermometers at each upstream (Site 1) and downstream (Site 3) site.

Macroinvertebrate sampling and analysis

Benthic invertebrates and organic matter were sampled using a standard Surber sampler (0.13 m^2) fitted with a 500-μm mesh catch net. Five benthic samples were collected randomly in riffles at each site (Townsend *et al.* 1983; Giller and Twomey 1993) in October 1996 and April 1997. Qualitative samples were also taken from all areas of the riffle habitat, including large boulders, debris dams, and edges on each sampling occasion. Samples were preserved with 10% formalin and returned to the lab. Following separation from organic matter, invertebrates were stored in 80% ethanol. Invertebrates were identified to the lowest practical taxonomic level, usually genus or species, using several identification keys and local invertebrate references (Merritt and Cummins 1995; Brigham et al. 1982; Penrose et al. 1982; Wiggins 1996; Stewart and Stark 1993; Epler 1995; Pennak 1989). Invertebrates were also assigned to a functional feeding group according to Merritt and Cummins (1995) and Pennak (1989).

The invertebrate assemblage was assessed by calculating various invertebrate metrics, such as total density, total taxa richness, diversity indices, EPT index, percent abundance per functional feeding group, etc. Additionally, invertebrate density and presence/absence information for all taxa was analyzed using multivariate statistics.

Benthic organic matter sampling and analysis

Benthic organic matter (BOM) was sampled by collecting ten random 0.1296-m^2 samples of leaves and wood in each study reach on five dates during the year (November 1996; January, March, April, and July 1997). The samples were dried at 55°C and separated into leaves, wood, and seeds. Each component was then weighed.

Measurement of physico-chemical variables

Discharge was calculated from cross-sectional area and current speed at each site (Gore 1996), and temperature was recorded at each site. Alkalinity, hardness, and pH were measured in the field using a Hach field kit. Chlorophyll *a* concentration was measured by placing five randomly selected large pebbles

(32-64 mm; Wentworth Classification as cited by Minshall 1984) in individual plastic containers and chilled in the dark for transport to the lab. Chlorophyll was extracted in 90% acetone and analyzed colorimetrically (Clesceri *et al.* 1989). Surface area of stones was determined by the aluminum foil method described by Steinman and Lamberti (1996). Water samples taken at each site were analyzed for nitrate (NO_3-N) and phosphate (o-PO_4) at Coweeta Hydrologic Lab. Three 1-L water samples were collected and filtered to measure total suspended solids.

Habitat assessment

Stream habitat was assessed at each site by two methods. Substrate composition was measured by randomly selecting and recording the size category (modified Wentworth Classification) of approximately 100 rocks (based on Dunne and Leopold 1978). From this information, the median particle size (Minshall 1984) and the percent of substrate in each size category were calculated. Additionally, the general stream habitat was surveyed using the EPA Habitat Assessment (Barbour and Stribling 1994), which assigns values to traits such as summer flow conditions, channelization, canopy shading, and bank erosion.

Land-use information

Spatial data were assembled from a variety of sources and organized in a geographical information system (GIS) by Paul Bolstad and staff, University of Minnesota. Bolstad provided the following description of how the GIS data were obtained. Separate data layers were developed for 1950s, 1970s, and 1990s landcover, and for watershed area, elevation, and slope. Elevation data were obtained from U.S. Geological Survey (USGS) 7.5' digital elevation quadrangles, delivered at a 30-m horizontal resolution. Watershed boundaries were determined from elevation data to identify drainage divergence, and slope was derived from digital elevation maps using a third-order finite difference method (Skidmore 1989). Landcover for the 1950s were manually interpreted from 1:20,000 panchromatic aerial photographs, hand-digitized, and terrain corrected using single-photo resection (Wolf 1983). Control points in this and subsequent digitizing were obtained from road intersections and other photo-identifiable points which were also represented on USGS 1:24,000 scale USGS quadrangles. There was some variation in map production date. Although most were initially produced between 1950 and 1955 (1950s data), production dates for this set ranged from 1940 to 1962. The 1990s maps were based on updates, the majority of which were conducted between 1983 and 1991, although some were conducted as early as 1978. Landcover for the 1990s was derived from Landsat Thematic Mapper™ collected in leaf-on periods during 1990 and 1991, classified using a supervised maximum likelihood classification, with post-classification sorting used to improve classification accuracy. Randomly selected points were photo-interpreted from 1:58,000 and 1:40,000 U.S. Department of Agriculture photographs and were used to assure classification to forest/non-forest categories at or above 95%. Land use (forest/non-forest) was determined for the watershed area above each stream benthic invertebrate sampling point by intersecting watershed boundaries with landcover and sampling locations. Buffer layers were derived from the stream layers, identifying regions within specified buffer distances of a stream. Stream buffers were intersected with land use layers. Summary statistics for upstream buffer zones were then extracted, including land use proportions within the buffer zones.

Statistical methods

Benthic invertebrate metrics and physico-chemical data were compared using two-way analysis of variance (ANOVA). The two main factors tested in the two-way ANOVA were stream types (*i.e.*, FA *vs.* CA streams; n = 3 of each stream type) and longitudinal position (*i.e.* Sites 1 *vs.* 2 *vs.* 3; n = 6 at each gradient position). When significant differences were detected by two-way ANOVA (α = 0.05), the means were then compared using Tukey's HSD multiple comparison technique (SigmaStat 1995). Statistical tests were two-way ANOVA unless otherwise noted. Invertebrate assemblages were compared using multivariate statistics run by PC-ORD (McCune and Mefford 1997). Finally, multiple regression analyses were used to investigate relationships between the invertebrate assemblage and environmental as well as land-use variables (SAS 1996). All statistical tests were run at the 0.05 significance level (α).

RESULTS AND DISCUSSION

Physico-chemical characteristics

Physico-chemical characteristics (Table 1), such as discharge, hardness, alkalinity, conductivity, and pH, were generally similar among all sites on both cleared agricultural (CA) and forested headwater agricultural (FA) streams. Hardness (range: 12.0 - 53.1 mg/L $CaCO_3$), alkalinity (range: 7.4 - 29.1 mg/L $CaCO_3$), and conductivity (range 9.7 - 41.1 mS) were similar among most sites. The pH was circumneutral, ranging from 6.8 to 7.3. These physico-chemical variables were similar to other agricultural streams in this area of western North Carolina (Lenat 1984). The mean discharge from October 1996 to July 1997 ranged from 0.02 m^3/s at a small forested headwater (FA-1) site to 0.45 m^3/s at a larger FA-3 site, but most sites were more similar in discharge. Discharge generally did not change dramatically (<0.05 m^3/s) within each stream from the headwater site (1) to the downstream site (3). North Fork of Ivy Creek, did increase between Site 1 (0.08 m^3/s) and Site 3 (0.45 m^3/s). The sites on this stream were separated by a greater longitudinal distance than the other streams in the study.

Land use analysis

The land use data collected using GIS techniques provided information about the percent of the watershed forested in the 1950s, 1970s, and 1990s. Forest versus non-forest was also assessed at multiple spatial scales, both

Table 1. Mean values (±1 S.E.) of physico-chemical characteristics at each site. Means and standard errors were calculated from measurements made three to six times during the study.

Physico-chemical trait	**Cleared agricultural streams**			**Forested agricultural streams**		
	CA-1	**CA-2**	**CA-3**	**FA-1**	**FA-2**	**FA-3**
Discharge (m^3/s)	0.08 (±0.02)	0.09 (±0.02)	0.12 (±0.02)	0.04 (±0.02)	0.14 (±0.10)	0.19 (±0.13)
Hardness (mg/L $CaCO_3$)	33.7 (±1.5)	40.3 (±6.8)	27.2 (±4.4)	34.0 (±3.7)	17.1 (±3.6)	30.2 (±4.4)
Alkalinity (mg/L $CaCO_3$)	22.8 (±1.4)	25.3 (±2.5)	16.7 (±1.4)	21.5 (±2.5)	11.4 (±2.3)	16.4 (±1.3)
Conductivity (μS)	28.2 (±1.5)	32.8 (±4.6)	22.0 (±2.8)	27.8 (±3.0)	14.3 (±2.9)	23.3 (±2.9)
pH	7.2 (±0.1)	7.2 (±0.0)	7.2 (±0.1)	7.0 (±0.0)	6.9 (±0.1)	7.1 (±0.0)

longitudinally (entire upstream watershed, the upstream segment between sites, or between the top site and the headwaters) and laterally (entire watershed, 100 m riparian width, or 30 m riparian width). In all three periods (Figure 3a), cleared agricultural (CA) streams were less forested (in the upstream, 30 m riparian zone) than the forested headwater agricultural (FA) streams (1950, $P = 0.004$; 1970, $P = 0.001$; 1990 < 0.001). Headwater sites (Figure 3b) were more forested than the downstream sites in the 1970s ($P = 0.04$) and 1990s ($P = 0.02$), but there were no significant differences between headwaters and downstream in the 1950s. These trends were similar when considered at the various spatial scales.

The mountains in western North Carolina have been used for agriculture since at least the middle to late 1800s. Agricultural records for Buncombe and Madison counties indicate the development of mountain sides and tops for growing tobacco and corn, as well as for grazing cattle, since around 1880 (North Carolina State Board of Agriculture 1896). The area also provided about 40 percent of the United States timber in 1910, but timber production began to fall away when much of the land had been cleared by 1920 (SAMAB 1996). Following those times of widespread logging and agricultural development, the percentage of the watershed area in forest cover among all sites increased from the 1950s to the 1970s (paired t-test; $P < 0.001$). This likely resulted from cleared farm lands being allowed to revert to forest as land was taken out of agriculture (Allan *et al.* 1997). From the 1970s to the 1990s, there was no consistent trend in forest cover change (paired t-test; $P > 0.05$). However, forest

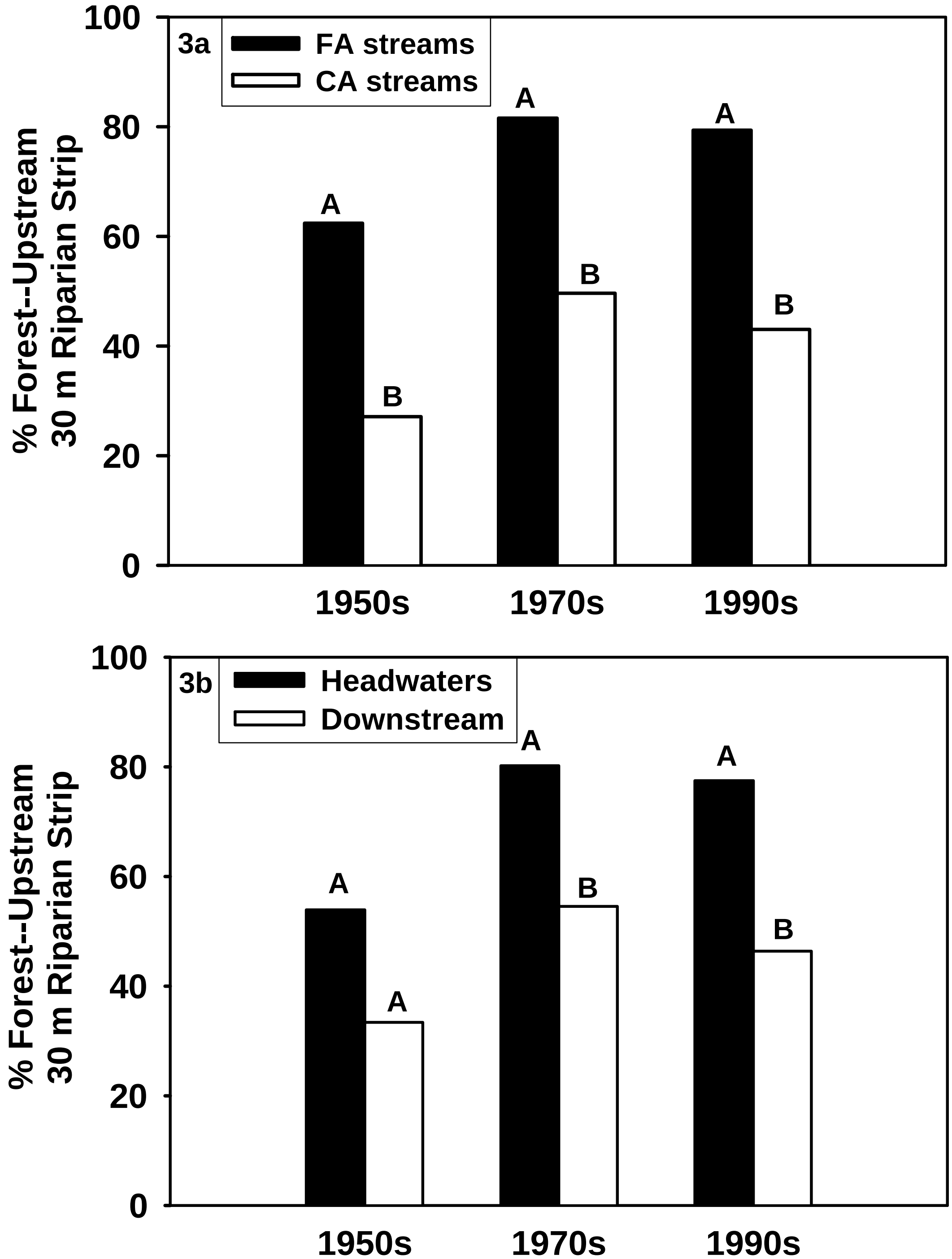


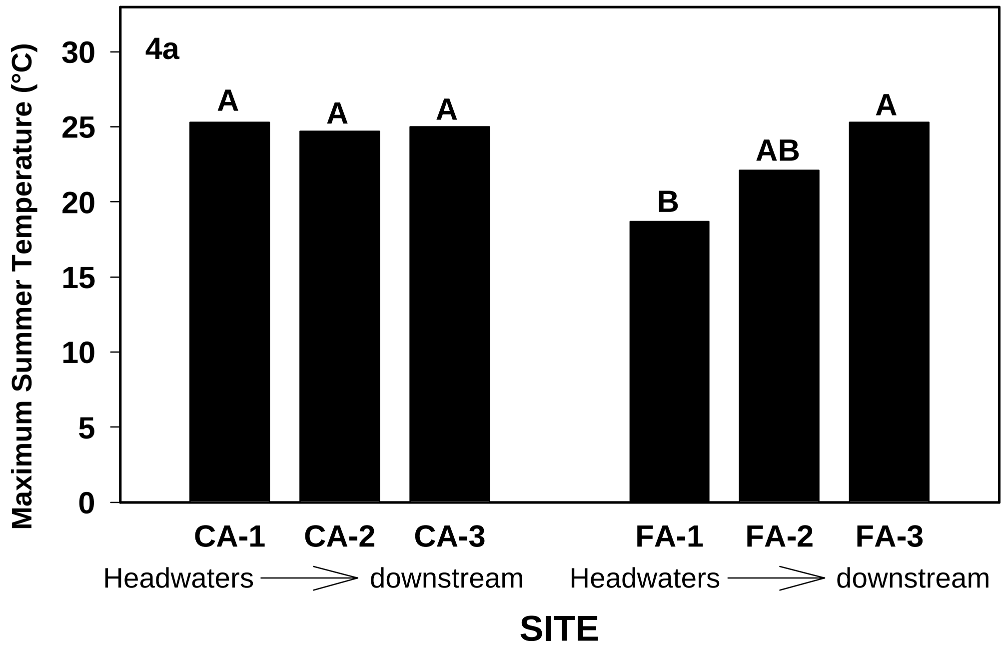
Figure 3. Mean percent of riparian area in forest cover in 1950s, 1970s and 1990s. Comparisons were made at each time period between stream types (3a) and gradient positions (3b). Bars marked with different letters are significantly different (2-way ANOVA, α = 0.05).

cover at individual sites increased or decreased up to 20 percent of the riparian area from the 1970s to 1990s. These historical land use patterns of increased forest cover from 1950s to 1970s, as well as individual site changes from 1970s and 1990s, may help explain to current stream invertebrate assemblages. Relationships between past land use and the invertebrates are discussed later, in the discussion section.

Temperature analysis

Temperatures were recorded for six intervals: autumn (mid-October through mid-November 1996), winter (mid-November 1996 through January 1997; mid-January through mid-March 1997), spring (mid-March through mid-April 1997), and summer (mid-April through mid-July 1997; mid-July through mid-September 1997). There were some difficulties in piecing together a complete temperature data set due to equipment losses during high flows at various times throughout the study.

The winter temperature range was generally similar among all sites (0 °C - 12 °C), while summer temperatures were more variable among sites. Summer minimum temperatures ranged from 3 °C to around 6 °C, and maximum temperatures ranged from 18 °C under full forest cover (FA-1 site) to 29 °C in an open FA-3 site. Maximum summer temperatures (Figure 4a) were not explained solely by stream type or gradient position but were different among the sites ($P = 0.03$). The maximum temperature under full forest cover (FA-1 sites; mean 18.7 °C) was significantly lower than the forested stream downstream sites (FA-3;



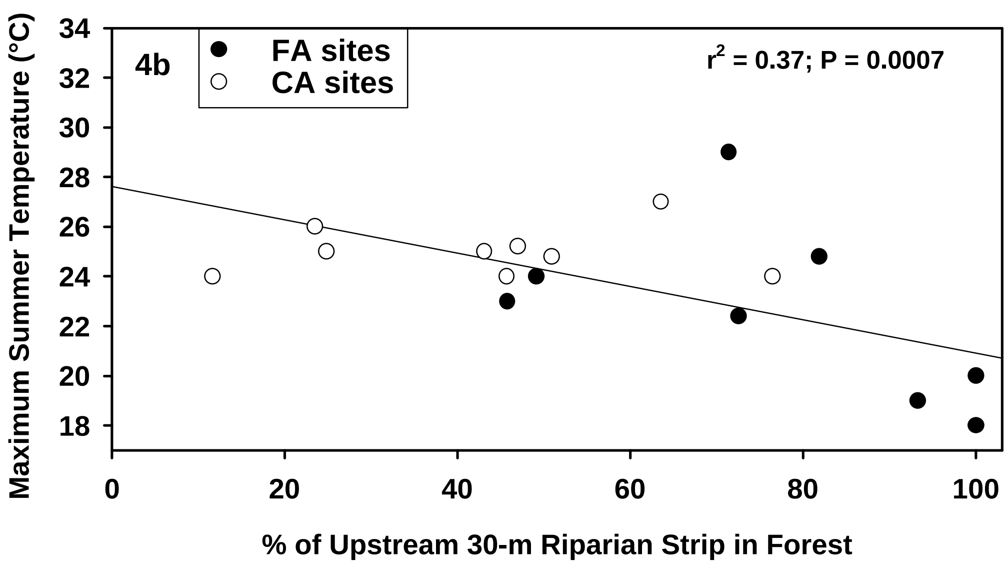


Figure 4. Mean maximum summer temperature (°C) compared among sites (4a). Bars marked with different letters are significantly different (P = 0.03; 2-way ANOVA). Relationship (4b) between maximum temperature and riparian forest cover (% forest in 30-m riparian zone in upstream segment).

mean 25.3 °C) as well as all sites in the cleared streams. The gradient of maximum summer temperatures among all sites was inversely related to forest cover in the riparian area (Figure 4b; r^2 = 0.37; P = 0.007). The degree of upstream riparian forest cover probably reflects stream shading influences on the stream temperature ranges (Allan 1995).

This summer temperature pattern, *i.e.,* higher temperature in open streams than in shaded streams, has been documented globally (Dance and Hynes 1980; Weiderholm 1984; Quinn *et al.* 1992; Quinn *et al.* 1994; Harding and Winterbourn 1995). The maximum temperature of the cleared streams (*ca.* 25 °C) in the present study could influence invertebrate assemblage structure by exceeding tolerance limits of some taxa, particularly some stoneflies and mayflies (Wiederholm 1984). The high temperatures could also increase invertebrate biomass while shortening development time, resulting in altered life history patterns and thereby shifting the assemblage structure found in cleared in streams compared to shaded streams (Wiederholm 1984; Allan 1995; Hawkins *et al.* 1997).

Nutrient analysis

Nitrate and phosphate were measured several times during the study in both the forested agricultural (FA) and cleared agricultural (CA) streams. Mean phosphate concentrations ranged from 0.0003 mg P/L to 0.023 mg P/L, and both the high and the low extremes occurred at FA-2 sites. However, there were no significant differences between stream types (Figure 5a) or gradient position.

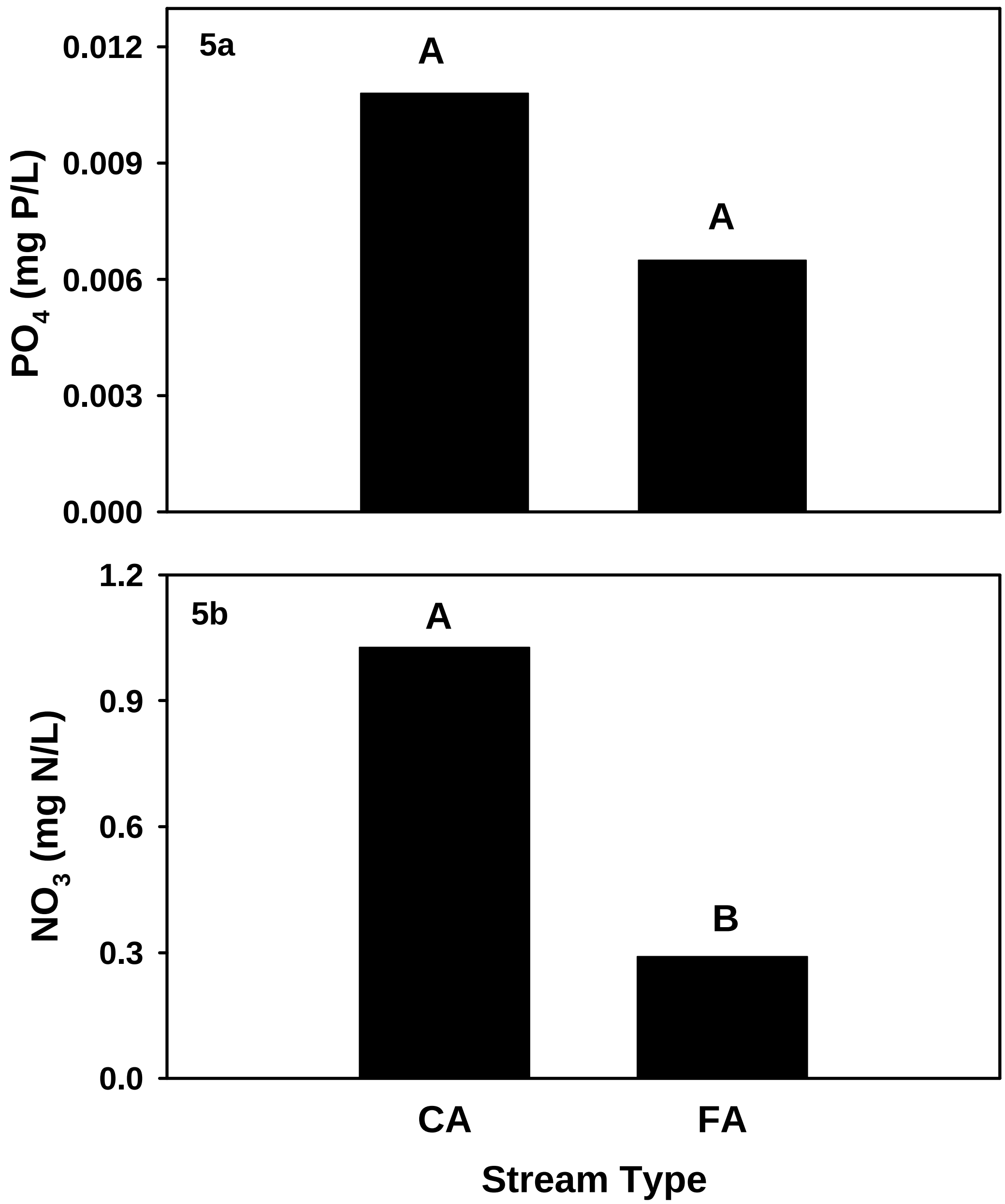


Figure 5. Mean phosphate (a) and nitrate (b) concentration compared between stream types. Bars marked with different letters are significantly different (2-way ANOVA, $\alpha = 0.05$).

Mean nitrate concentrations ranged from 0.124 mg N/L in a forested headwater (FA-1) to 1.589 mg N/L at a CA-2 site which is directly downstream of a dairy operation. The nitrate concentrations in the cleared agricultural streams were significantly higher than the forested agricultural streams ($P < 0.001$; Figure 5b), although there were not significant differences along the longitudinal gradient.

Mean nitrate concentrations for both CA streams (1.03 mg N/L) and FA streams (0.29 mg N/L) were greater than background concentrations of 0.002 - 0.02 mg N/L recorded for forested streams in western North Carolina (Simmons and Heath 1979; Swank and Waide 1988). Mean phosphate concentrations (CA = 0.011; FA = 0.006 mg P/L) were also higher than forested streams (0.001-0.002 mg P/L) in western North Carolina (Swank and Waide 1988). These elevated concentrations were probably due to the presence of a dairy operation on Willow Creek (a CA stream), grazing cattle on several of the streams, and row crop fertilization (Dance and Hynes 1980; Allan 1995). The large dairy operation on Willow Creek has been a source of elevated nutrients since at least the 1970s (Willow Creek Project Summary 1982).

Nitrate concentrations recorded in other agricultural areas around the world vary: 0.05 mg N/L in Australian pasture streams (Reed *et al.* 1994), 0.47 - 1.16 mg N/L in Illinois prairie streams (Wiley *et al.* 1990), 0.35 - 1.75 mg N/L in English streams (Townsend *et al.* 1983), and 0.33 - 1.09 mg N/L in agricultural basins in Michigan (Richards *et al.* 1993; Tuchman and King 1993). Small streams throughout the United States average up to 0.15 mg N/L in catchments with >75% agriculture, and up to 3.5 mg N/L in catchments with >90% agriculture

(Allan 1995). The nitrate concentrations found in the present study (0.124 - 1.589 mg N/L) indicate organic enrichment and were similar to other agricultural stream systems.

Total suspended solids

Total suspended solids at baseflow ranged from 5.0 mg/L at a forested headwater (FA-1) site to 44.5 mg/L at a cleared headwater (CA-1) site. Overall, CA streams had about twice the concentration of total suspended solids as FA streams ($P = 0.01$; Figure 6a), but there were no significant differences among longitudinal positions. Both inorganic and organic components of the suspended solids were also significantly higher in CA compared to FA streams ($P = 0.01$ for each component) but similar among longitudinal positions.

Forested streams throughout North Carolina have been found to average 6 mg/L TSS at baseflow (Simmons and Heath 1979), compared with 10 mg/L in FA streams and 21 mg/L in CA streams in this study. Erosion of top-soil, as well as destabilization of stream banks due to channelization and stock access (Lenat 1984; Quinn *et al.* 1992; Waters 1996), contribute to elevated TSS concentrations. In Buncombe County poorly managed agricultural areas lose up to 50.3 metric tons of soil per watershed hectare while forested watersheds lose less than 13.6 metric tons of soil per hectare (Lenat 1984). Other stream studies have shown similar TSS differences between cleared and forested streams (Tuchman and King 1993; Reed *et al.* 1994; Quinn *et al.* 1994; Allan *et al.* 1997). For example, in New Zealand TSS concentrations were roughly four times

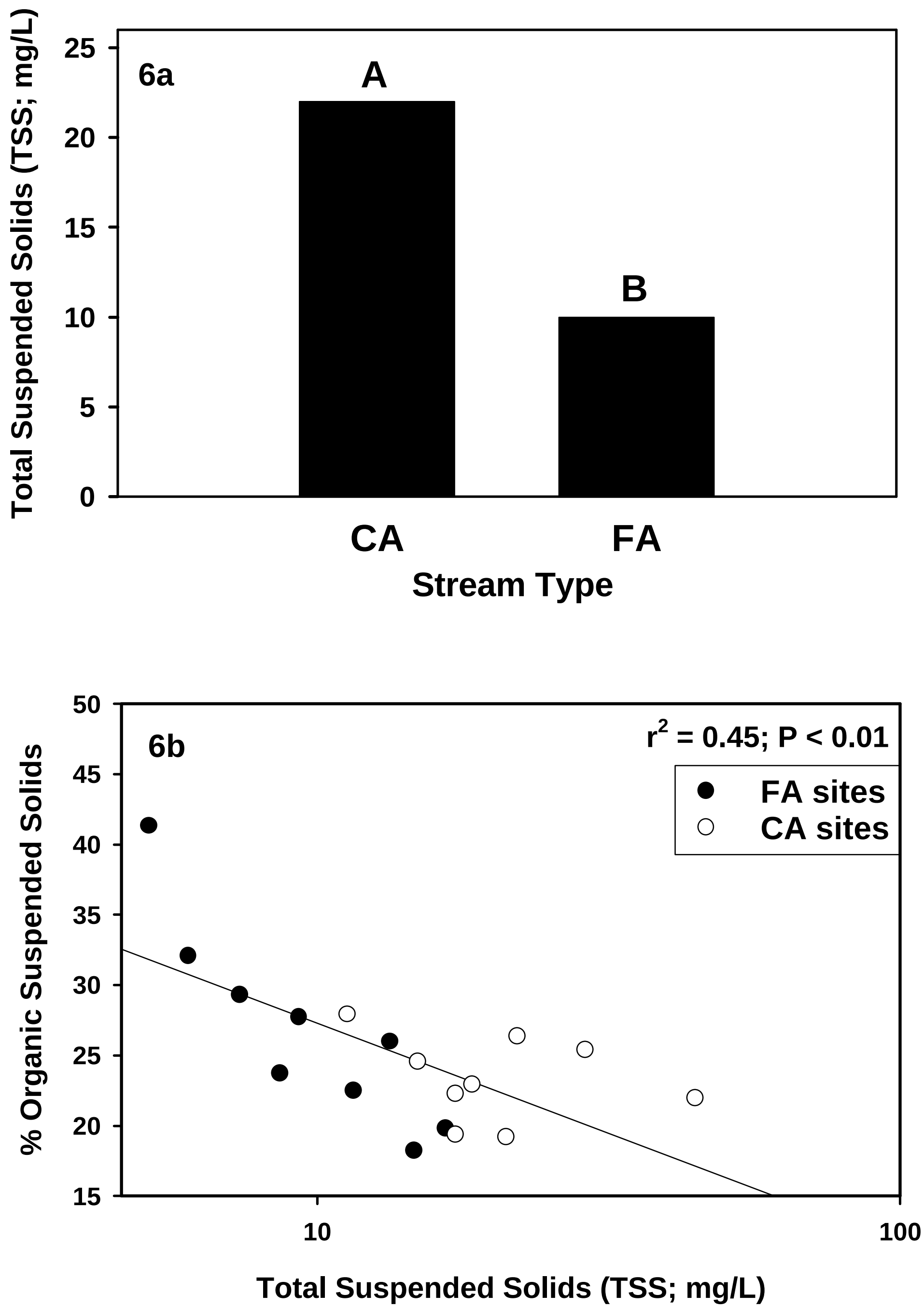


Figure 6. Mean total suspended solids (TSS) in cleared agricultural (CA) versus forested agricultural (FA) streams (6a). Bars marked with different letters are significantly different (2-way ANOVA, $\alpha = 0.05$). Relationship between TSS and organic fraction of TSS (6b).

greater in pasture streams than in streams flowing through native forest (Quinn *et al.* 1994).

Suspended solids ranged from 18.2% to 41.3% organic material, but the organic fraction was not significantly different between stream types. Generally, lower concentrations of solids contained a greater proportion of organic matter (Figure 6b; $r^2 = 0.45$; $P < 0.01$). This relationship has been documented in North Carolina streams by Simmons and Heath (1979) as well as in a Michigan stream (Tuchman and King 1993) and appears to be related to riparian vegetation. Therefore, while streams with high TSS provide more suspended organic material as food for filter-feeding invertebrates (Cummins 1973), the corresponding elevated suspended sediment concentrations may also scour invertebrates and lower periphyton food quality and habitat heterogeneity.

Benthic organic matter

Benthic organic matter (BOM) was separated into leaves, wood (< 5 cm diameter), and seeds (<5% of total). Mean total benthic organic matter ranged from 5.2 g/m^2 dry-weight at an intermediate cleared (CA-2) site to 138.2 g/m^2 within a forested headwater (FA-1) site. This high extreme was largely due to woody material washed into the stream following a strong storm. Forested agricultural streams had more BOM than cleared agricultural streams (Figure 7a; $P < 0.001$). In addition, headwater sites (all six Site 1's) had more BOM than the downstream sites (Figure 7b; $P = 0.03$; 2-way ANOVA on log transformed BOM data).

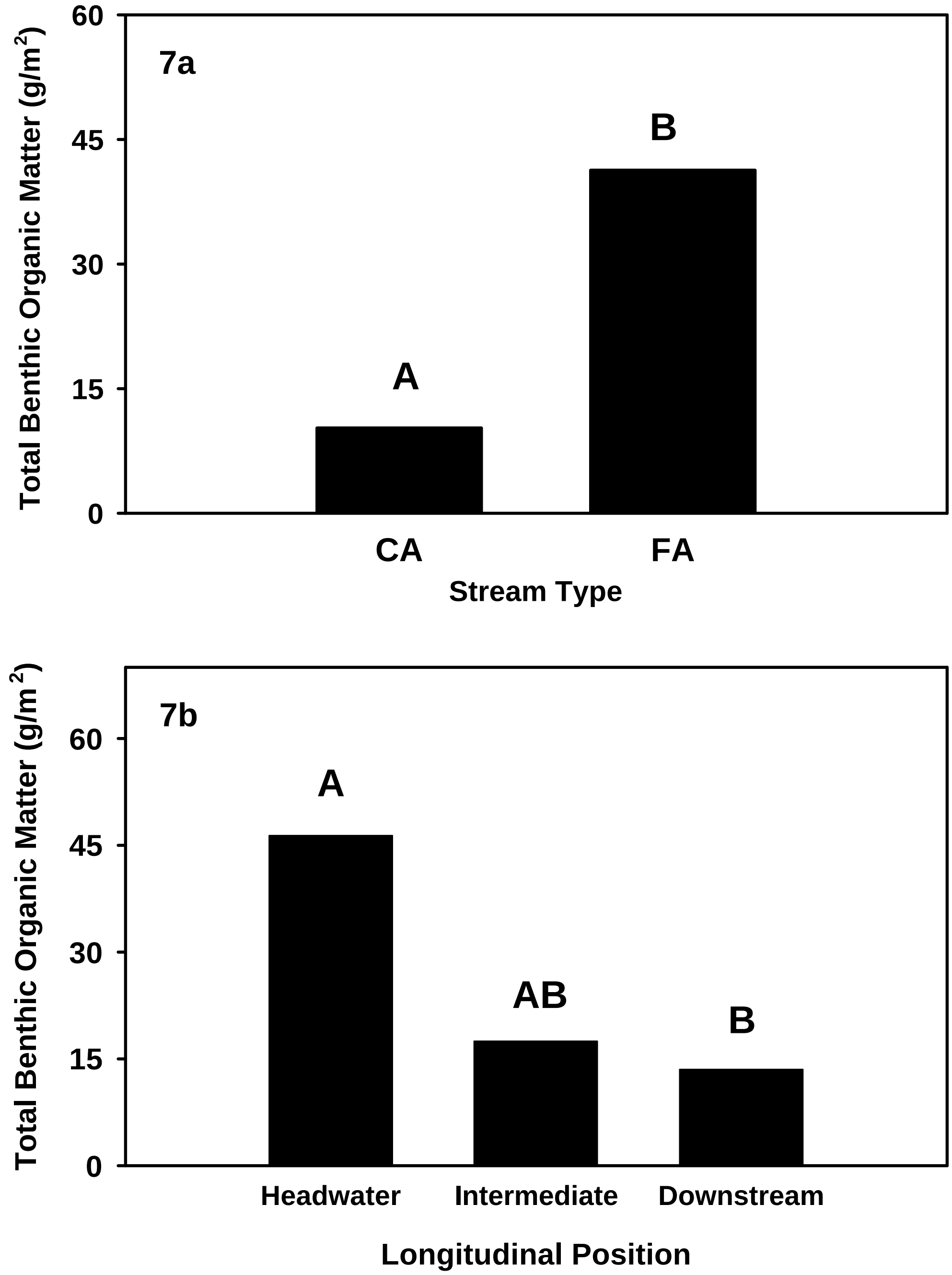


Figure 7. Mean total benthic organic matter compared by stream type (7a) and among longitudinal positions (7b). Bars marked with different letters are significantly different ($P < 0.05$; 2-way ANOVA on log-transformed data).

The leaf portion of the total organic matter ranged from 1.9 g/m^2 at a cleared intermediate site (CA-2) to 24.6 g/m^2 at a forested headwater (FA-1) site. Differences in leaf material were not accounted for by either stream type or gradient position, but by specific sites (*i.e.,* CA-2 versus FA-2; Figure 8a). The forested headwater (FA-1) sites had significantly greater amounts of leaf material than all other sites. The sites immediately downstream of the forested headwaters (FA-2) also had significantly more leaves than the downstream sites (FA-3) as well as all of the entirely agricultural sites ($P < 0.001$). As might be expected, the amount of leaf material was highly related to the percent of forest in the riparian area (Figure 8b; $r^2 = 0.71$, $P < 0.001$). Forested stream intermediate (FA-2) sites may also receive leaf material exported from the forested headwaters (FA-1), adding to the leaf material from the riparian vegetation (Vannote *et al.* 1980; Reed *et al.* 1994). However, it is not possible to separate the influences of riparian vegetation and transport based on this study.

Open, agricultural streams often have less BOM than forested streams (Reed *et al*. 1994; Quinn *et al.* 1994; Allan 1995; Harding and Winterbourn 1995). Woody debris in streams serves as the base of debris dams which form as leaves and small organic matter fragments pack against obstructions. These debris dams filter organic matter and sediments from the water, disperse energy of the flowing water, and provide stable and diverse habitat for invertebrates (Bilby 1981; Ehrman and Lamberti 1991; Cobb *et al*. 1992; Palmer *et al*. 1996). Standing crops of BOM in this study, including the forested headwater (FA-1) sites, were generally lower than standing crops reported for several forested

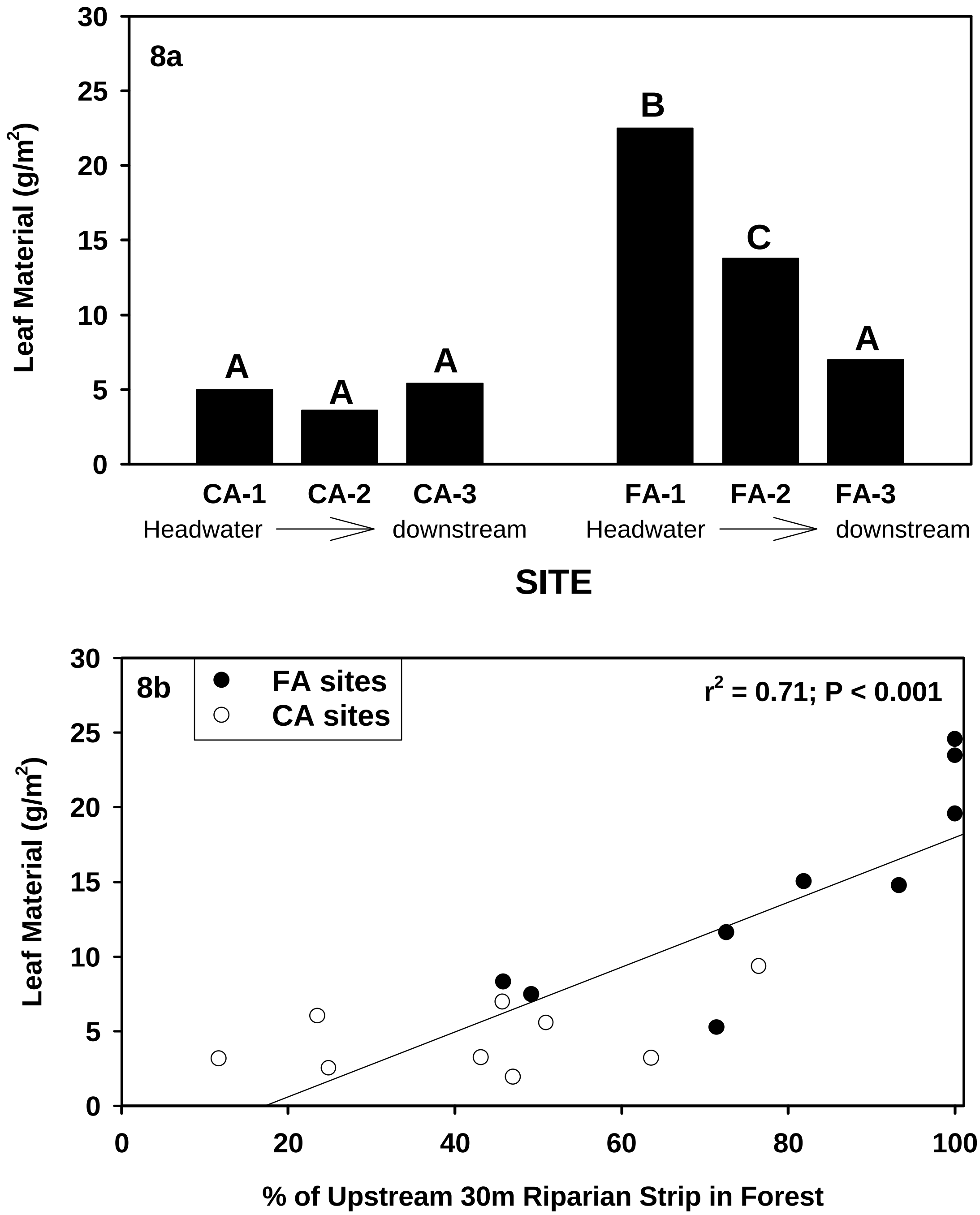


Figure 8. Mean standing biomass of leaf material compared among all sites (8a). Bars marked with different letters are significantly different ($P < 0.001$; 2-way ANOVA, $\alpha = 0.05$). Relationship between riparian vegetation and leaf material (8b).

streams in the United States and Europe (Table 2). However, standing crops in this study were similar to New Zealand and Australian pasture and forested streams. The differences in BOM between the forested reaches of this study and the United States forested streams may be related to the nature of the forest. The United States forested streams used for comparison are found within forests that were never completely cleared and have been relatively undisturbed for at least 30 years (Mulholland 1997; Wallace *et al.* 1997; Webster *et al.* 1997; Findlay *et al.* 1997). In comparison, the sites in this study had less forest cover in the 1950s than in the 1990s, indicating disturbance as recently as 30 years ago.

Chlorophyll a

Chlorophyll *a* exhibited trends opposite of total benthic organic matter. Chlorophyll *a* was significantly greater in cleared agricultural (CA) streams than in forested agricultural (FA) streams ($P < 0.001$; Figure 9a). Mean chlorophyll *a* levels were also significantly higher at all downstream sites compared to all headwater sites ($P = 0.02$; Figure 9b). Chlorophyll a was higher at sites with lower percent forest cover in the riparian area (Figure 10; $r^2 = 0.50$, $P = 0.001$) presumably due to less shading in areas of less riparian cover (Lester *et al.* 1996). Algal production, and therefore chlorophyll *a*, may also be enhanced by elevated temperatures (Figure 4) and nutrient concentrations (Figure 5b) in the agricultural reaches compared to more forested reaches (Corkum 1996). Higher chlorophyll *a* in open, agricultural streams, at levels similar to those measured in

Table 2. Comparison of standing crops of benthic organic matter in selected forested and agricultural streams.

	Standing crop of benthic organic matter (BOM) > 1 mm (g AFDM / m^2)		
Stream, description, location	**Non-woody BOM**	**Wood BOM**	**Total BOM**
Walker Branch, 1st order **forested**, Tennessee [1]	175	50	225
Satellite Branch, 1st order **forested**, North Carolina [2]	161	419	580
Breitenbach, 1st order **forested**, Germany [3]	---	---	621
Hugh White Creek, 2nd order **forested**, North Carolina [4]	213	5446	5659
Bear Brook, 2nd order **forested**, New Hampshire [5]	610	530	1140
Several streams (means), 2nd order **native forest**, New Zealand [6]	---	---	40
Several streams (means), 2nd order **pasture**, New Zealand [6]	---	---	10
Several streams (means), 2nd order **forest**, Australia [7, 8]	---	---	36
Several streams (means), 2nd order **pasture**, Australia [7,8]	---	---	26
This study, 1st-2nd order **forested headwaters** (FA-1) [8]	19-23 Mean = 21	9-113 Mean = 53	32-131 Mean = 74
This study, 1st-2nd order **all cleared stream sites** (CA-1, -2, -3) [8]	2-9 Mean = 4	1-15 Mean = 6	4-20 Mean = 10

[1] Mulholland 1997; [2] Wallace *et al.* 1997; [3] Marxsen *et al.* 1997; [4] Webster *et al.* 1997; [5] Findlay *et al.* 1997; [6] Quinn *et al.* 1994; [7] Reed *et al*. 1994; [8] Dry weight measurements were converted to approximate AFDM measurements by formula: (g DW / m^2) (0.95) = g AFDM / m^2 (Webster pers. comm.).

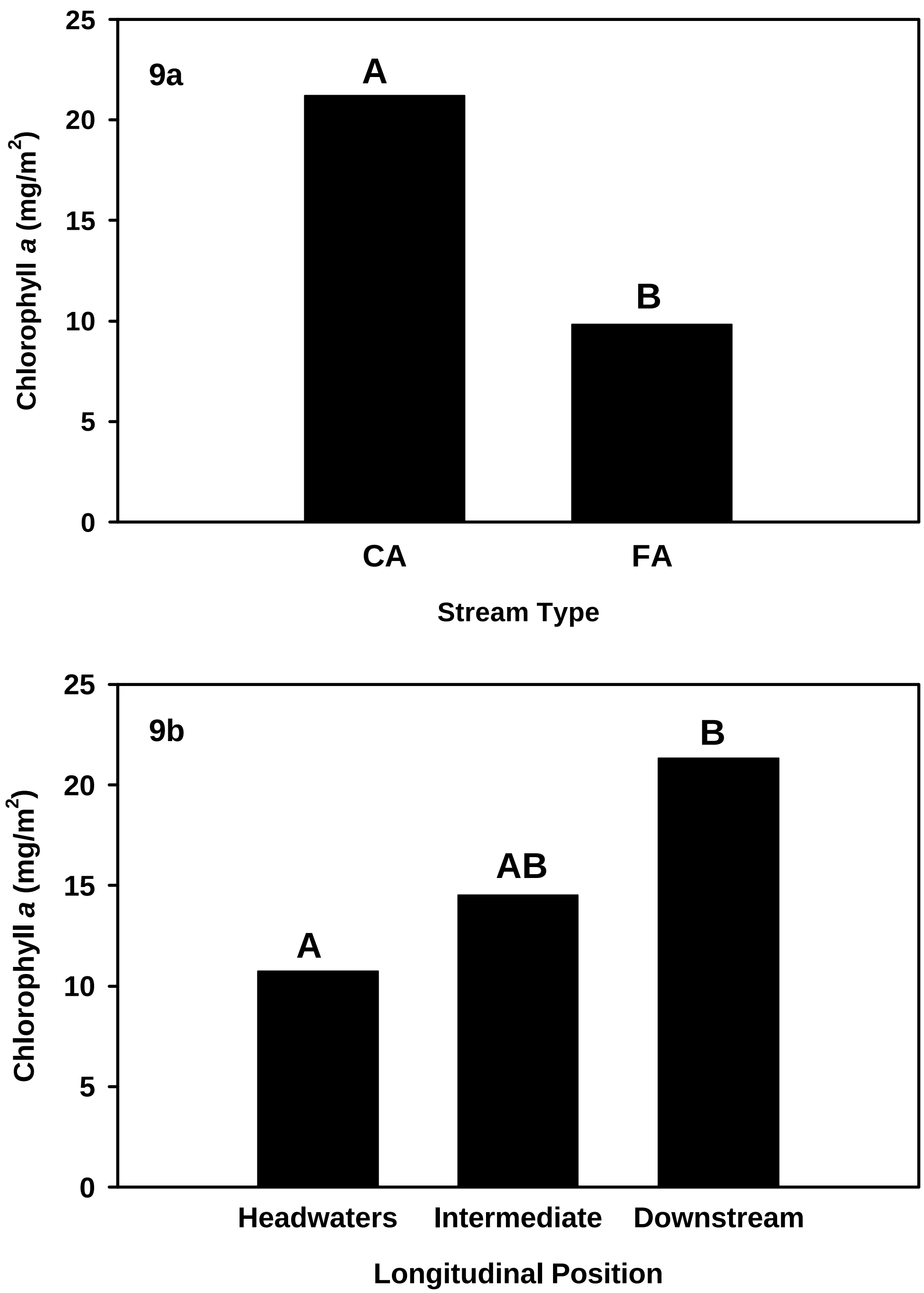


Figure 9. Mean chlorophyll a (mg/m^2) compared between stream types (9a) and among longitudinal positions (9b). Bars marked with different letters are significantly different (2-way ANOVA, α = 0.05).

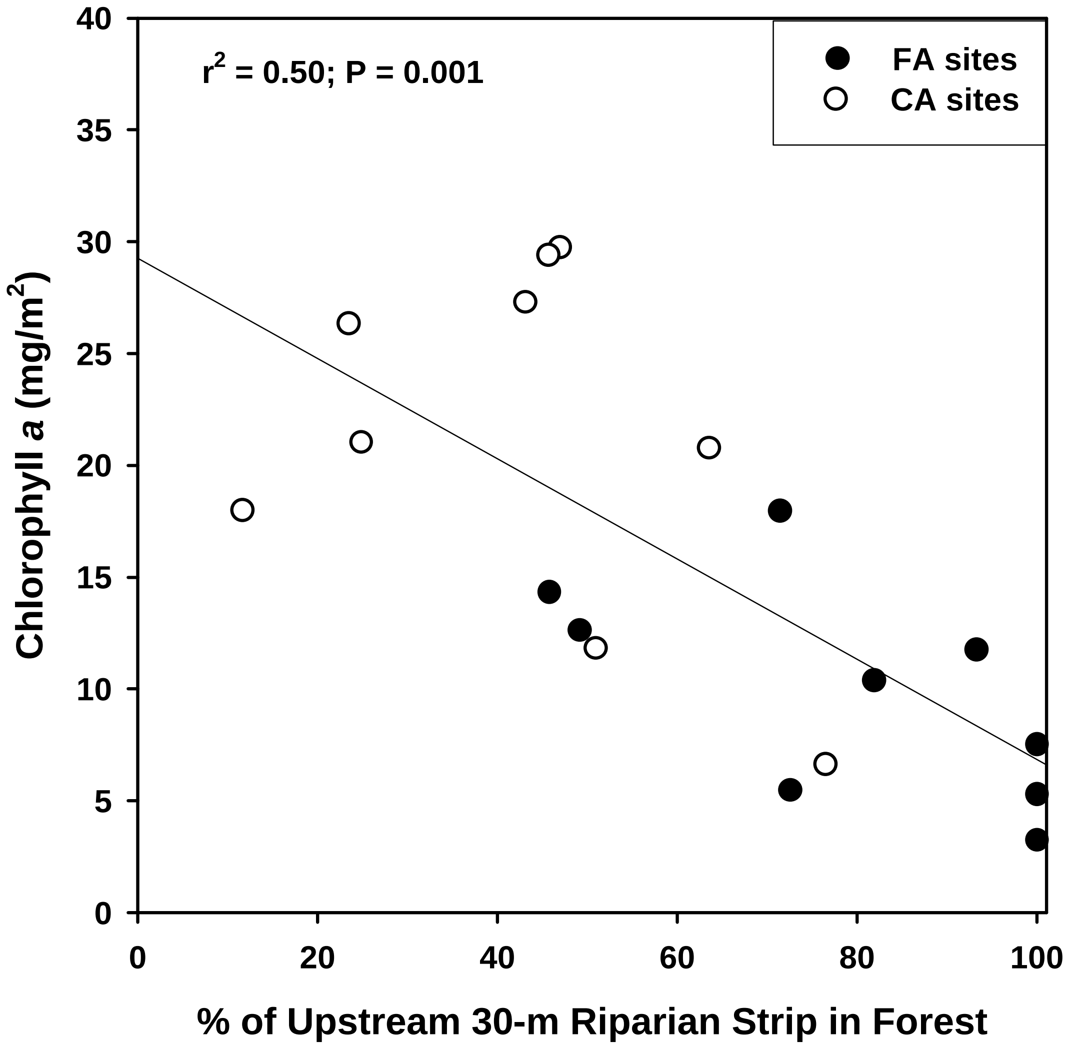


Figure 10. Relationship between chlorophyll *a* and percent of riparian area in forest (r^2 = 0.50; P = 0.001).

this study, has been well documented (Quinn *et al.* 1992; Quinn *et al.* 1994; Reed *et al.* 1994; Allan 1995; Corkum 1996).

Habitat conditions

Stream substrate composition, as assessed by particle size categories, was generally similar among all sites, with most sites dominated by cobble-sized stones (Wentworth classification; Table 3). The percent of substrate composed of boulders and bedrock was lower than all other categories, and one-third of the sites had no boulder/bedrock component. However, boulder/bedrock was most dominant in the FA streams (P = 0.01; Table 3), which was consistent with Lenat's (1984) substrate analysis of this region. Boulder/bedrock was also most dominant in the headwater sites compared to the intermediate and downstream sites (P = 0.01). Boulders provide substrate stability in streams by physically staying in place. Boulders also serve as foundations for debris dams, creating habitat stability. Turbulent areas created by boulders are likely to freeze over in cold winter temperatures, a phenomenon which could be important to invertebrates with multi-year life cycles (deMarch 1976).

Stream habitat was also quantified by assigning values to a list of variables described in the EPA Habitat Assessment worksheet. The results are presented in Table 4 as raw habitat scores (maximum score = 160) and the percent of maximum score. The amount of in-stream cover habitat, degree of large substrate embeddedness due to fine sediment, summer discharge, and canopy shading each composed up to 12.5% of the total possible score. The

Table 3. Mean (±1 S.E.) percent of substrate in each size class, compared between stream types. Values marked with different letters are significantly different (2-way ANOVA, $\alpha = 0.05$).

% Substrate Size Class	CA streams	FA streams	P-value
Boulder/Bedrock (> 25.6 cm)	2.8 A (±1.1)	8.2 B (±2.1)	0.01
Cobble (6.4 - 25.6 cm)	26.7 A (±2.5)	29.6 A (±2.9)	n.s.
Pebble (1.6 - 6.4 cm)	24.7 A (±18.7)	18.7 A (±3.0)	n.s.
Gravel (2-16 mm)	24.2 A (±1.8)	24.1 A (±3.9)	n.s.
Sand/Silt (< 2 mm)	22.4 A (±2.1)	21.1 A (±3.3)	n.s
Median Particle Size (cm)	2.4 A (±0.5)	3.8 A (±1.2)	n.s.

Table 4. Habitat assessment scores for each stream type (mean ±1 S.E.). Values marked with different letters are significantly different ($P = 0.02$; 2-way ANOVA).

Score	CA streams	FA streams
Mean Habitat Assessment	86 A (±3.1)	106 B (±6.6)
% of Maximum Possible	54 A (±1.9)	66 B (±4.1)

degree of channelization, scouring/deposition, pool and riffle ratios, and lower bank capacity parameters each received up to 9.4% of the total possible score. And both upper bank stability and bank vegetation levels accounted for up to 6.3% of the total possible score.

The FA streams with forested headwaters received higher habitat scores, including the highest of all the sites at 78.8% of the maximum score, compared to the CA streams ($P = 0.02$) which included the poorest habitat score at 44.4% of maximum. Bank stability and vegetation, although the least weighted parameters, encompassed the greatest discriminatory ability among sites with scores spread through 80% of the possible range. Habitat quality was generally higher at sites with more riparian forest cover (Figure 11; $r^2 = 0.21$; $P = 0.05$). Other studies have also shown this relationship between land use and habitat quality (Roth *et al.* 1996; Allan *et al.* 1997).

Invertebrate assemblage, taxonomic analysis

A total of 36 quantitative samples were collected from the 18 sites in October 1996 and again in April 1997. From the quantitative samples, 52,946 invertebrates were identified to the lowest practical taxonomic level (usually species) and counted. Additional taxa were identified from qualitative samples collected at each site and both dates. A list of all 194 taxa represented in this study can be found in Appendix A.

Of these 194 taxa, 117 taxa were common to both cleared and forested agricultural streams. There were 31 taxa (Appendix B) collected only in the

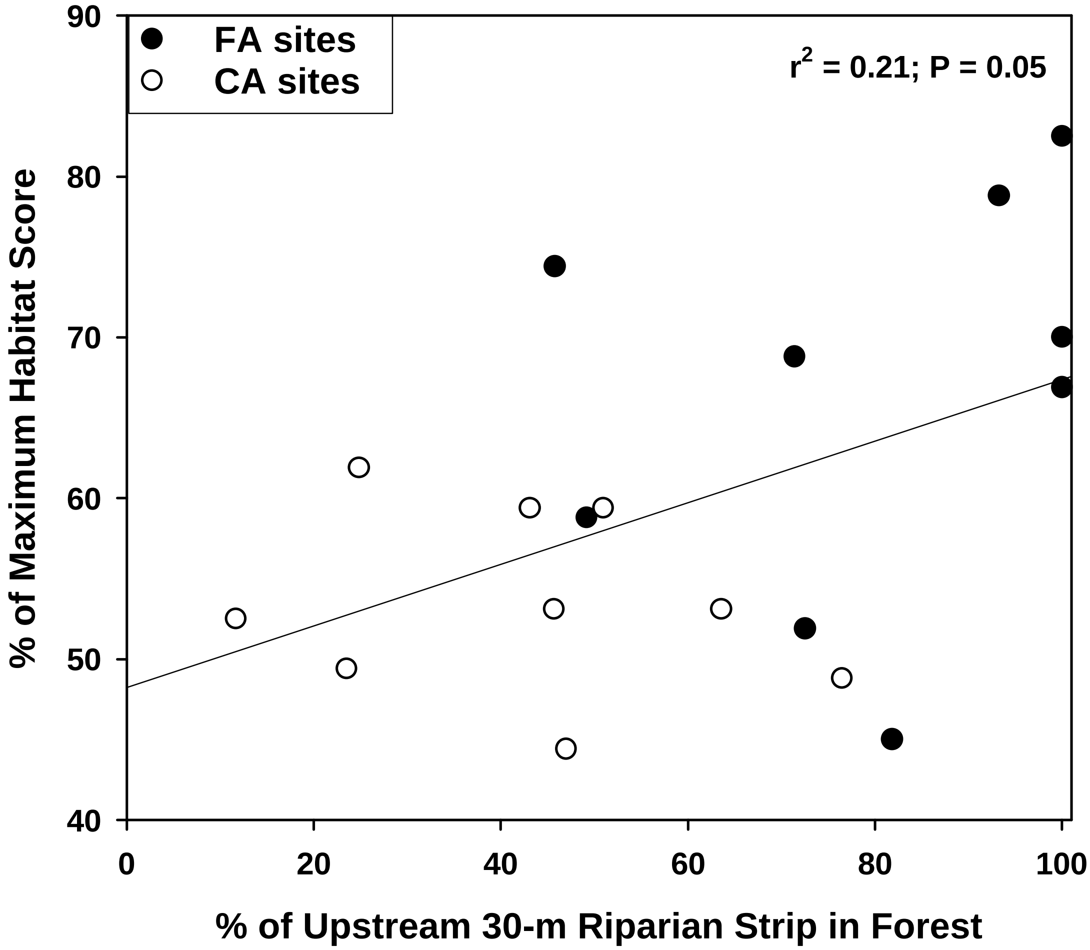


Figure 11. Relationship between habitat score and riparian forest cover. (r^2 = 0.21; P = 0.05)

cleared agricultural streams ("unique" taxa). In contrast, there were 46 taxa collected only in forested agricultural streams (Appendix C). Pooled presence-absence data from all sites of each stream type indicated a total taxa richness of 148 in CA streams (117 common taxa + 31 taxa unique to CA streams). In comparison, FA streams had a total taxa richness of 163 (117 common taxa + 46 unique taxa). While FA total taxa richness was not substantially greater than that of CA streams, the two stream types supported different assemblages of taxa. Diptera and Ephemeroptera were 35% and 25%, respectively, of the 31 taxa unique to the CA streams (Table 5). In FA streams, Diptera and Trichoptera represented 35% and 30%, respectively, of the unique taxa (Table 5). However, approximately 40% of the taxa unique to the FA streams (primarily Diptera and Trichoptera) were found only within the forested headwaters (FA-1) and not in the downstream agricultural (FA-2, FA-3) sites. Overall, this assemblage breakdown indicates that the two stream types supported different invertebrate assemblages, and the forested headwater (FA-1) sites had a different assemblage than the downstream agricultural (FA-2, FA-3) sites.

It is worth noting that one dipteran species unique to the cleared streams, *Chironomus riparius* (Chironomidae), is often found in sewage treatment plants (Epler 1995). However, this species was collected in Willow Creek at the sites downstream of a large dairy operation. The presence of this species and the elevated nitrate concentration in this stream may indicate organic pollution in this stream, perhaps from sewage lagoon overflow and subsequent run-off into the stream.

Table 5. Number of taxa unique to each stream type.

Number of unique taxa	CA streams	FA streams
Ephemeroptera	8	5
Plecoptera	5	6
Trichoptera	1	14
Coleoptera	4	3
Diptera	11	16
Other Misc.	2	2
Total	31	46

The distribution of taxa found at each site was also analyzed according to stream type and longitudinal position. Mean taxa richness (Table 6) was significantly higher ($P = 0.04$) in FA streams than in CA streams, but there were no differences among longitudinal positions. Most of the taxa were Ephemeroptera, Plecoptera, Trichoptera, Coleoptera, and Diptera, and the distribution among the orders was generally similar among all sites. However, the number of trichopteran taxa, as well as the sum of Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa, were significantly greater in FA streams (Table 6). Trichoptera diversity was highly related to the percent forest cover in the riparian zone (Figure 12; $P = 0.006$).

This general pattern of lower diversity, particularly in the sensitive EPT taxa, in agricultural streams has been documented in studies world-wide (Dance and Hynes 1980; Lenat 1988; Quinn and Hickey 1990; Lenat 1994; Harding and Winterbourn 1995), although other studies have not found such trends (Giller and Twomey 1993; Quinn et al. 1994; Reed et al. 1994). Although significantly higher in FA streams, taxa richness was not related to the percent of forest in the riparian area (upstream segment, 30 m riparian strip; Figure 13a). However, taxa richness was highly related to nitrate concentration (Figure 13b; $P < 0.001$), an indication that the invertebrate assemblages may be related to intensity of agricultural practices, particularly livestock grazing (Dance and Hynes 1980, Allan 1995), and not just the amount of cleared land.

The invertebrate assemblage was quantitatively sampled in both fall 1996 and spring 1997. In the fall (Table 7), invertebrate density was higher in CA

Table 6. Mean number of taxa (±1 S.E.) in cleared agricultural and forested agricultural streams. Values marked with different letters are significantly different (2-way ANOVA, $\alpha = 0.05$).

Number of taxa	CA streams	FA streams	P-value
Coleoptera	4.8 A (±0.4)	5.3 A (±0.4)	n.s.
Diptera	16.2 A (±1.1)	17.6 A (±0.6)	n.s.
Ephemeroptera	22.0 A (±1.5)	20.1 A (±0.7)	n.s.
Plecoptera	7.8 A (±1.0)	9.9 A (±0.9)	n.s.
Trichoptera	7.8 A (±.9)	14.9 B (±0.9)	< 0.001
Total (S)	66.2 A (±3.6)	76.7 B (±2.2)	0.04
EPT	36.3 A (±2.9)	44.9 B (±1.8)	0.04

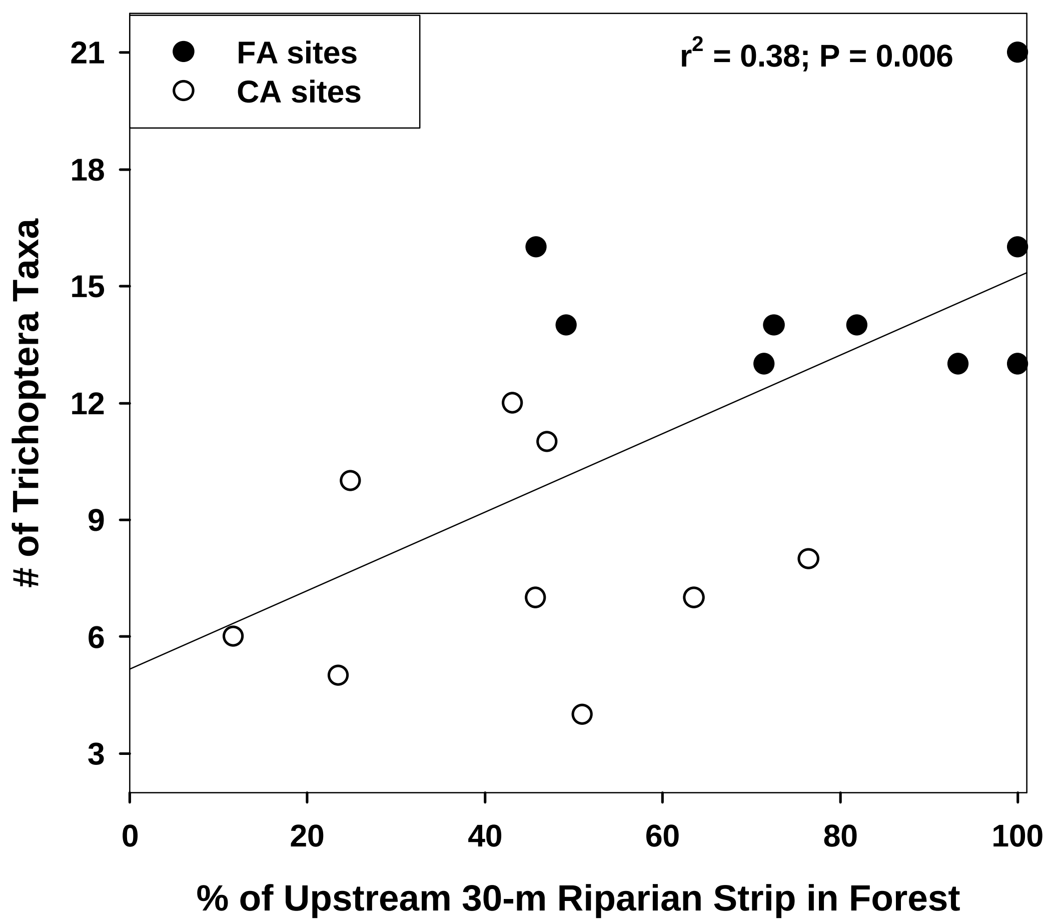


Figure 12. Relationship between Trichoptera diversity and riparian vegetation.

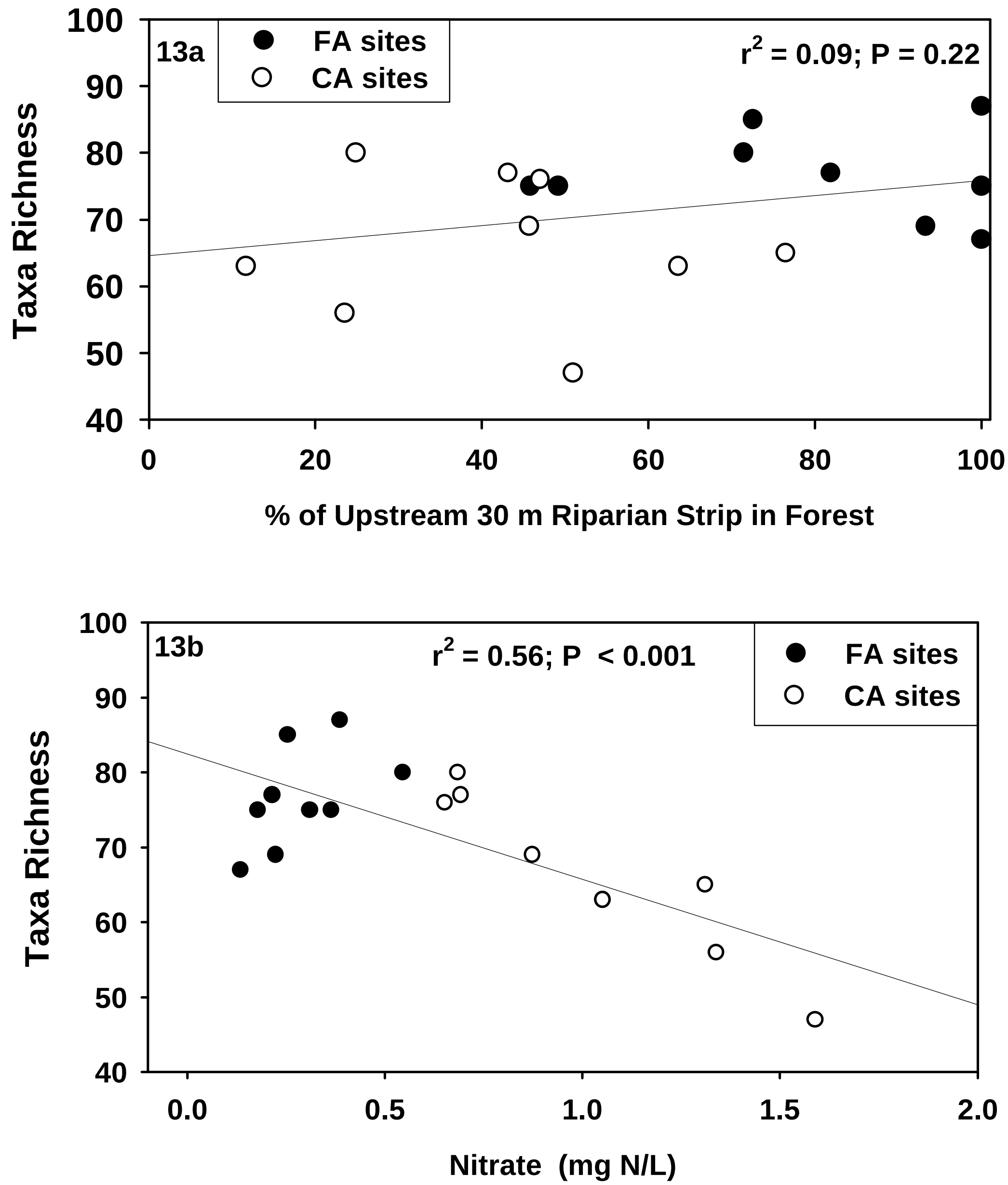


Figure 13. Relationship of taxa richness with riparian vegetation (13a) and NO_3-N concentration (13b).

Table 7. Mean metric values (±1 S.E.) describing the invertebrate assemblage in October 1996 compared between stream types. Values marked with different letters are significantly different (2-way ANOVA, $\alpha = 0.05$).

Invertebrate metrics	**CA streams**	**FA streams**	**P-Value**
Density (#/m^2)	5670 A (±1097)	2670 B (±425)	0.02
Taxa Richness	44.6 A (±4.3)	54.0 A (±2.0)	n.s.
Margalef's Index of Diversity	5.8 A (±0.6)	7.6 B (±0.3)	0.02
EPT	23.0 A (±3.3)	30.2 A (±1.7)	n.s.
MAIS [1]	13.8 A (±1.3)	14.7 A (±1.3)	n.s.

[1] MAIS = Macroinvertebrate Aggregated Index for Streams; Maximum score = 18.

streams (P = 0.02), but density was similar between CA and FA streams in the spring (Table 8; P > 0.05). While invertebrate density has been found in some other studies to be higher in agricultural streams than forested streams (Lenat 1984; Harding and Winterbourn 1996), this relationship has been somewhat variable at different sites or in different seasons (Lenat 1984; Quinn *et al.* 1994; Reed *et al.* 1994). Taxa richness was not significantly different between stream types in fall (P > 0.05), but in spring (Table 8) FA streams had a greater taxa richness than CA streams (P = 0.03). The EPT taxa were not significantly different between stream types in either season. Margalef's Index, a measure of diversity which reflects taxa richness but also incorporates the distribution among taxa (Magurran 1988), was higher in FA streams in both fall (P = 0.02) and spring (P = 0.03). A multi-metric assessment of the invertebrate assemblage at the family level (MAIS; Smith and Voshell 1997) showed no significant differences during the fall, but in the spring, FA streams received higher scores than CA streams (P = 0.008). All of these metrics considered together indicate a more diverse invertebrate assemblage in FA streams than in CA streams.

The distribution among major orders was similar between stream types and among longitudinal positions in the fall. In both seasons, mayflies composed the largest portion of total invertebrate density. In the spring, however, there were differences in the distribution of invertebrates among orders (Table 9). Coleoptera were a greater component of total density in FA streams than in CA streams (P = 0.02), while dipterans showed the opposite trend and were relatively more abundant in CA streams than in FA streams (P = 0.04).

Table 8. Mean metric values (±1 S.E.) describing the invertebrate assemblage in April 1997 compared between stream types. Values marked with different letters are significantly different (2-way ANOVA, $\alpha = 0.05$).

Invertebrate metrics	CA streams	FA streams	P-Value
Density ($\#/m^2$)	2140 A (±241)	2221 A (±245)	n.s.
Taxa Richness	40.9 A (±2.1)	48.9 B (±2.3)	0.03
Margalef's Index of Diversity	5.9 A (±0.3)	6.9 B (±0.3)	0.03
EPT	24.4 A (±1.7)	28.3 A (±1.2)	n.s.
MAIS [1]	14.0 A (±0.9)	17.3 B (±0.3)	0.008

[1] MAIS = Macroinvertebrate Aggregated Index for Streams; Maximum score = 18.

Table 9. Mean percent (±1 S.E.) of total invertebrate density (April 1997) in each major insect order compared between stream types. Values marked with different letters are significantly different (2-way ANOVA, $\alpha = 0.05$).

% of total invertebrate density in each order	CA streams	FA streams	P-value
Coleoptera	4.4 A (±1.4)	10.7 B (±2.3)	0.02
Diptera	28.8 A (±4.1)	17.3 B (±2.3)	0.04
Ephemeroptera	43.5 A (±5.6)	49.6 A (±3.5)	n.s.
Plecoptera	2.4 A (±0.6)	6.3 B (±1.6)	0.006
Trichoptera	6.9 A (±2.6)	10.8 A (±2.3)	n.s.

Riffle beetles (Elmidae) were the most abundant Coleoptera and accounted for the greater beetle abundance in FA streams. Elmidae as a group tend to be most abundant in clear, cool streams and may be an indicator of stream quality (White 1982). The common elmid taxa in these streams (*Optioservus ovalis* and *Oulimnius latiusculus*) have been assigned tolerance values for the North Carolina Biotic Index which indicate that they are relatively pollution sensitive (Lenat 1993), and Marsh and Water (1980) found that elmids were more abundant in forested stream segments. However, beetles, particularly the elmids, have also been found to be tolerant of sediment and agriculture (Lenat *et al.* 1981).

In the spring, stoneflies were relatively more abundant in FA streams than CA streams (Table 9; P = 0.006). Many studies have shown decreased Plecoptera density in agricultural streams, and stoneflies are commonly stated to be the most sensitive order (Dance and Hynes 1980; Lenat 1984; Quinn *et al.* 1994; Harding and Winterbourn 1996; Stewart and Harper 1996), therefore this result is not surprising. However, Plecoptera also comprised a greater portion of total density in all headwater (FA-1 and CA-1) sites than in intermediate (FA-2 and CA-2) or downstream (FA-3 and CA-3) sites (Table 10; P = 0.006). In this study, Plecoptera were relatively more abundant in sites with more riparian vegetation, higher habitat quality, and lower summer temperatures (Figure 14). These relationships reflect the more specific habitat requirements of stoneflies, which tend to be common in clear, cool streams (Stewart and Harper 1996).

Table 10. Mean percent (±1 S.E.) of total invertebrate density (April 1997) in each major insect order compared among longitudinal positions. Values marked with different letters are significantly different (2-way ANOVA, $\alpha = 0.05$).

% of total invertebrate density in each order	Head-waters	Inter-mediate	Down-stream	P-value
Coleoptera	10.0 A (±2.9)	8.0 A (±2.8)	4.6 A (±2.1)	n.s.
Diptera	20.4 A (±2.5)	26.5 A (±4.6)	22.3 A (±6.5)	n.s.
Ephemeroptera	46.1 A (±3.2)	46.7 A (±5.5)	46.9 A (±8.4)	n.s.
Plecoptera	7.7 A (±2.2)	2.8 B (±0.9)	2.6 B (±0.6)	0.006
Trichoptera	7.3 A (±1.3)	7.1 A (±2.7)	12.1 A (±4.4)	n.s.

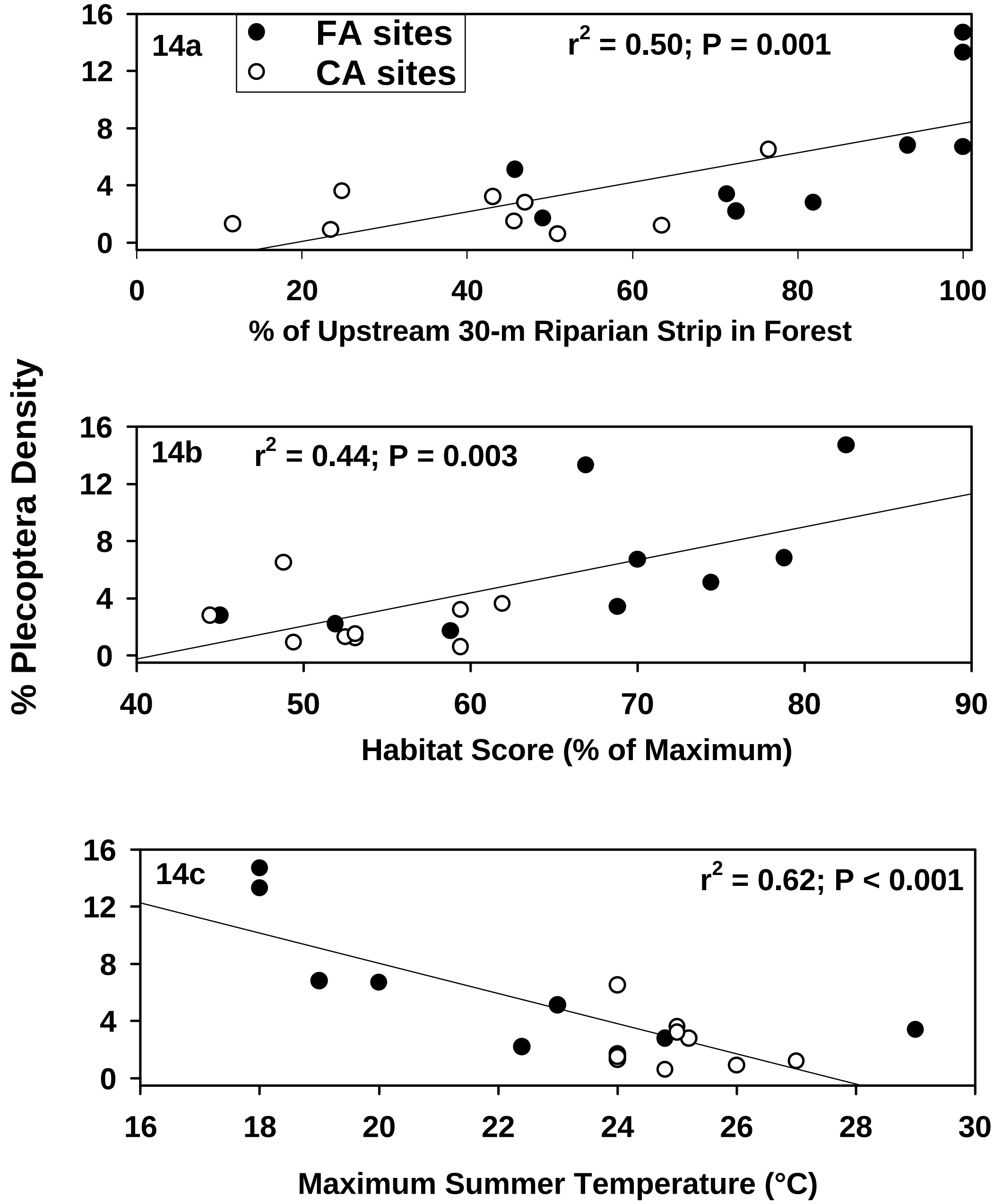


Figure 14. Relationship between relative Plecoptera density (% of total density) and three site variables: riparian forest cover, habitat quality, and maximum summer temperature.

Functional feeding group composition

Functional feeding group (FFG) composition was generally similar in the two stream types, forested agricultural (FA) and cleared agricultural (CA) streams. The bulk of the taxa were collector-gatherers (Table 11), ranging from 30% to 47% of the total number of taxa at each site. The FA streams had a higher percentage of collector-filterer taxa ($P = 0.002$) and lower percentage of collector-gatherer taxa ($P = 0.004$) than CA streams (Table 11). When analyzed according to longitudinal gradient position (Table 12), shredders made up a greater portion of taxa in the headwaters (FA-1 and CA-1) than in the downstream (FA-3 and CA-3) sites ($P = 0.04$).

The distribution of invertebrate density among functional feeding groups (FFG) was similar between stream types and along the longitudinal gradient in the fall. In the spring, shredders composed a greater portion of the assemblage in FA streams than in CA streams (Table 13; $P = 0.007$), and were also relatively more abundant in headwater sites than in intermediate or downstream sites (Table 14; $P = 0.007$). Shredder density (relative to total density) was significantly related to leaf material in the streams (Figure 15; $r^2 = 0.66$; $P < 0.001$), which corresponds to the findings of Egglishaw (1964) and predictions of the River Continuum Concept (Vannote *et al.* 1983). Scrapers were also more abundant in FA streams than in CA streams (Table 13; $P < 0.001$). Scrapers tended to be a greater component of invertebrate density as chlorophyll a decreased (Figure 16a; $r^2 = 0.24$; $P = 0.04$), which is opposite of what might be expected since scrapers consume algae on the rocks. However, elmids,

Table 11. Mean percent of total taxa (±1 S.E.) from all sites and all sampling dates in each of functional feeding groups, compared between stream type. Values marked with different letters are significantly different (2-way ANOVA, α = 0.05).

% of total taxa in functional feeding groups	**CA streams**	**FA streams**	**P-value**
Shredder	8.2 A (±0.7)	9.3 A (±0.9)	n.s.
Collector-Filterer	12.4 A (±0.9)	16.6 B (±0.7)	0.002
Collector-Gatherer	40.3 A (±1.4)	33.6 B (±1.1)	0.004
Predator	21.5 A (±1.6)	23.2 A (±1.0)	n.s.
Scraper	17.6 A (±0.6)	17.3 A (±0.8)	n.s.

Table 12. Mean percent of total taxa (±1 S.E.) from all sites and all sampling dates in each of functional feeding groups, compared among longitudinal positions. Values marked with different letters are significantly different (2-way ANOVA, $\alpha = 0.05$).

% of total taxa in functional feeding groups	**Head-waters**	**Inter-mediate**	**Down-stream**	**P-value**
Shredder	10.6 A (±1.1)	8.5 AB (±0.9)	7.2 B (±0.4)	0.04
Collector-Filterer	13.0 A (±1.4)	15.4 A (±0.9)	15.2 A (±1.36)	n.s.
Collector-Gatherer	37.1 A (±2.3)	38.2 A (±2.4)	35.4 A (±1.6)	n.s.
Predator	21.5 A (±1.6)	20.9 A (±2.0)	24.6 A (±1.0)	n.s.
Scraper	17.8 A (±1.2)	17.0 A (±0.7)	17.7 A (±0.7)	n.s.

Table 13. Mean percent (±1 S.E.) of total invertebrate density (April 1997) in each functional feeding group compared between stream types. Values marked with different letters are significantly different (2-way ANOVA, $\alpha = 0.05$).

% of total invertebrate density in functional feeding groups	CA streams	FA streams	P-value
Shredder	1.3 A (±0.7)	4.5 B (±1.6)	0.007
Collector-Filterer	6.0 A (±2.3)	9.3 A (±2.3)	n.s.
Collector-Gatherer	79.9 A (±4.4)	57.8 B (±3.2)	0.003
Predator	3.3 A (±0.4)	4.6 A (±0.8)	n.s.
Scraper	9.5 A (±1.9)	23.8 B (±3.1)	0.002

Table 14. Mean percent (±1 S.E.) of total invertebrate density (April 1997) in each functional feeding group compared among longitudinal positions. Values marked with different letters are significantly different (2-way ANOVA, $\alpha = 0.05$).

% of total invertebrate density in functional feeding groups	**Head-waters**	**Inter-mediate**	**Down-stream**	**P-value**
Shredder	6.7 A (±2.1)	1.3 B (±0.4)	0.7 B (±0.2)	0.001
Collector-Filterer	6.4 A (±1.4)	6.4 A (±2.5)	10.1 A (±3.6)	n.s.
Collector-Gatherer	66.9 A (±5.3)	69.7 A (±8.1)	70.0 A (±6.9)	n.s.
Predator	4.0 A (±0.6)	3.6 A (±1.2)	4.3 A (±0.8)	n.s.
Scraper	15.9 A (±3.6)	19.1 A (±6.1)	14.9 A (±3.2)	n.s.

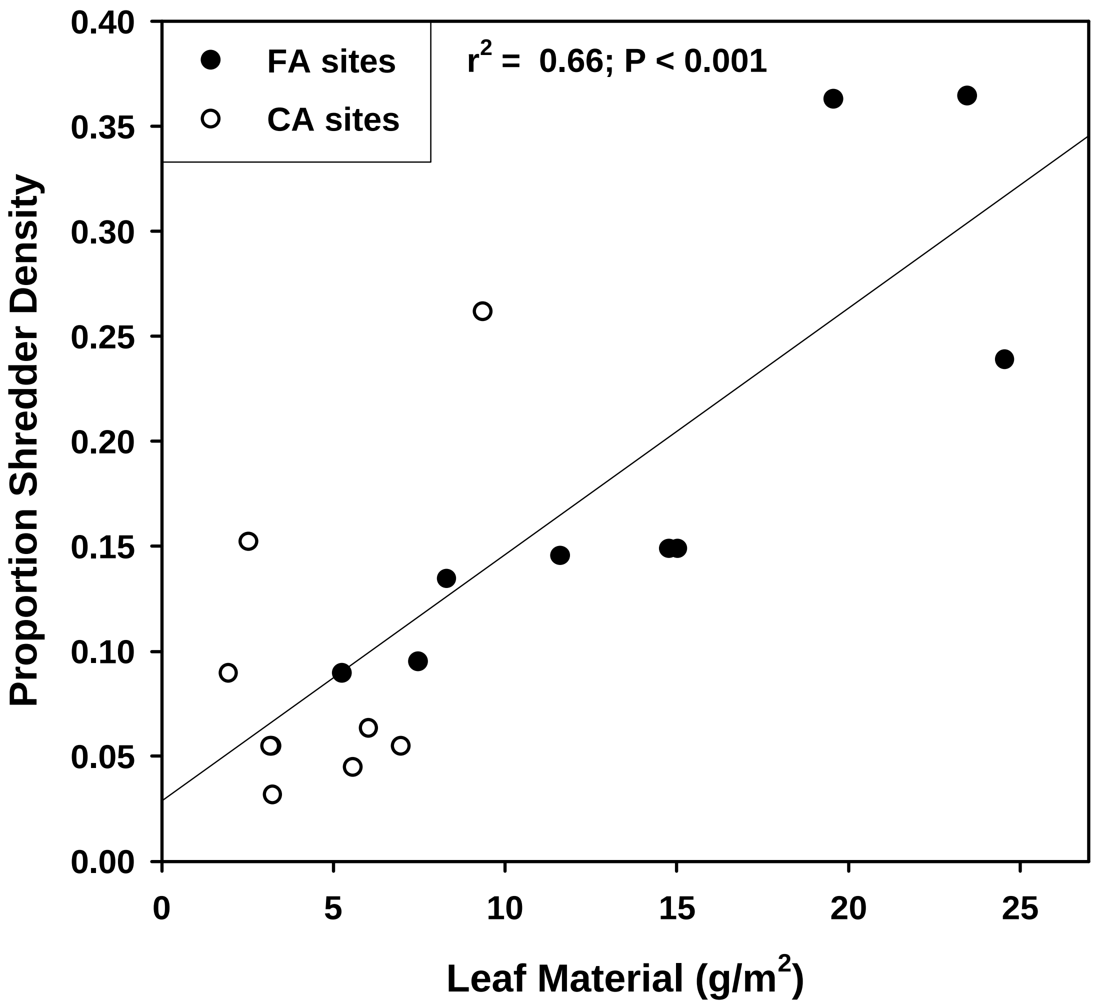


Figure 15. Relationship between shredder density (% of total density) and leaf material standing crop. Shredder proportions were arcsine square root transformed.

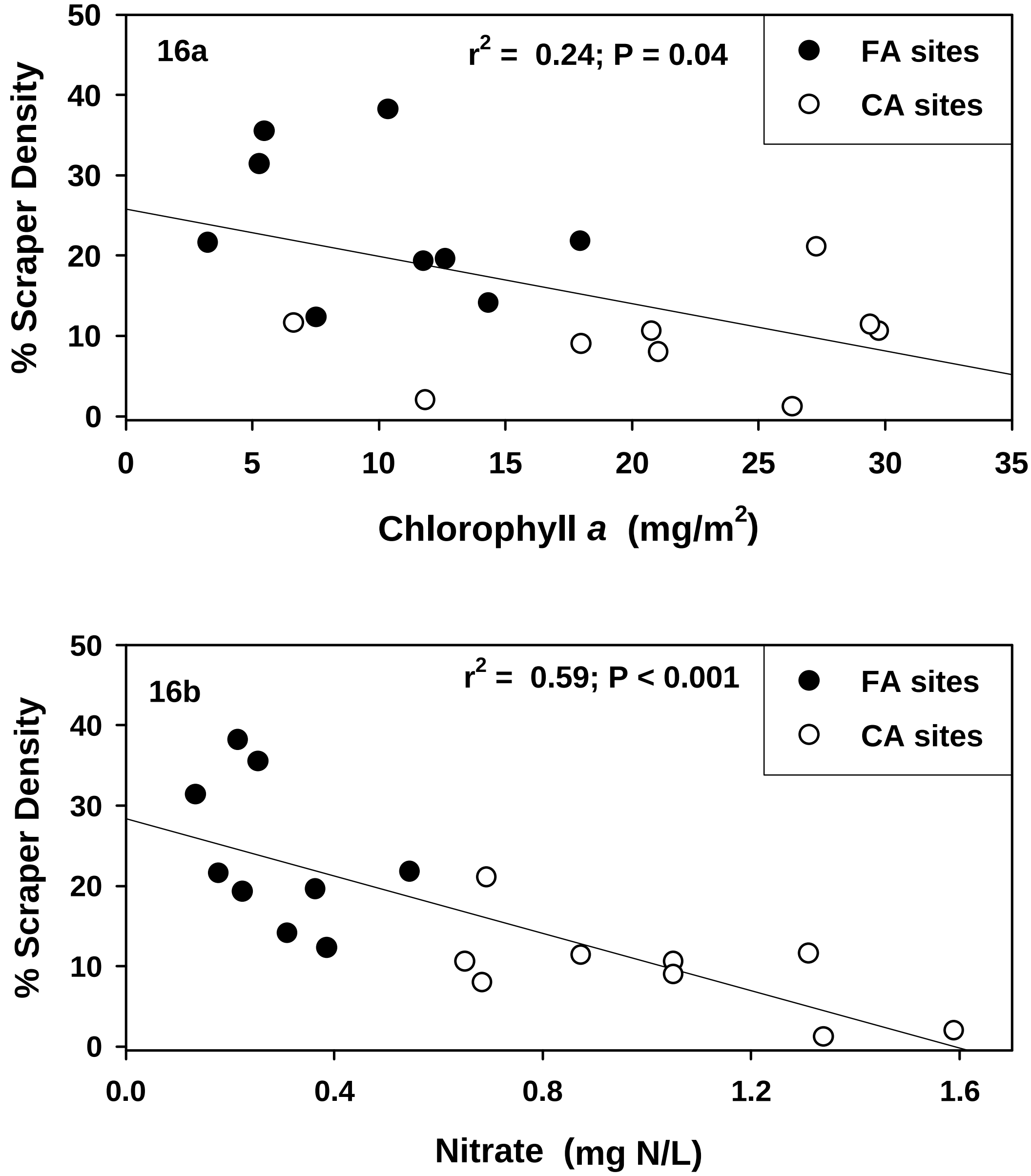


Figure 16. Relationship between scraper density (% of total density) and chlorophyll *a* (16a) and nitrate concentration (16b).

generally considered scrapers (White 1982; Merritt and Cummins 1996), were most of the scrapers in FA streams. This family was often as abundant in FA sites as all scrapers combined in CA sites. Elmids are secondarily considered collector-gatherers (Merritt and Cummins 1996), but scraper density was not related to leaf material in the streams ($r^2 = 0.06$). However, scraper density had a strong tendency to decrease as nitrate concentration increased (Figure 16b; $r^2 = 0.60$; $P < 0.001$). Elevated nutrient concentrations generally indicate agricultural impact, so this relationship between scrapers and nitrate concentration may reflect the tendency for elmids to be most abundant in cleaner streams (White 1982).

Multivariate analysis of invertebrate assemblage

Invertebrate density data from all sites and both sampling dates were analyzed using multivariate techniques to arrange sites according to similarity of the invertebrate assemblages. Cluster analysis is one method of defining groups of sites that have similar invertebrate assemblages (McCune and Mefford 1997). In the cluster analyses (Figures 17 and 18), sites were arranged using the relative Euclidean distance measurement and Ward's hierarchical linking method, parameters recommended as all-purpose for most data sets (McCune and Mefford 1997). The point in the cluster diagram where sites separate into groups to be discussed is indicated by a distance measurement. This measurement is specific to each analysis and is indicated on each cluster

```
                                    Distance (Objective Function)

                     .040          12.745   ↓     25.450           38.155          50.861
                     |-------+-------+-------+-------+-------+-------+-------+-------
                                    Information remaining (%)
                     100           75.0            50.0             25.0            0.00
                     |-------+-------+-------.+-------+-------+-------+-------+-------+
CA1 (NT)  Fall       |-|
CA2 (NT)  Fall       | ||
CA3 (NT)  Fall       |-||--------------|
FA3 (HK)  Fall       |  |              |--|
CA2 (NT)  Spring     ---|              |  |
CA2 (WL)  Spring     |-----------------|  |
CA3 (WL)  Spring     |                    |------------|
CA1 (PP)  Fall       |                    |            |
CA3 (PP)  Fall       |------|             |            |
FA2 (HK)  Fall       |      |------|      |            |---------------------------|
CA2 (PP)  Fall       -------|      |------|            |                           |
FA2 (UF)  Fall       -|------------|                   |                           |
FA3 (UF)  Fall       -|                                |                           |
                                                       |                           |
CA2 (WL)  Fall       |---------------------------------|                           |
CA3 (WL)  Fall       |                                                             |
                                                                                   |
CA1 (WL)  Fall       -----|----|                                                   |
CA1 (WL)  Spring     -----|    |-|                                                 |
FA2 (NFI) Fall       --|-------| |                                                 |
FA3 (NFI) Fall       --|         |------------|                                    |
FA1 (NFI) Fall       --------|--||            |                                    |
FA1 (NFI) Spring     ------|-|  ||            |                                    |
FA1 (UF)  Spring     |-----|    ||            |                                    |
FA1 (HK)  Spring     |          |             |                                    |
FA1 (UF)  Spring     -----|-----|             |                                    |
FA1 (HK)  Fall       -----|                   |------------------------------------|
                                              |
CA1 (NT)  Spring     ||                       |
CA1 (PP)  Spring     ||-------|               |
CA2 (PP)  Spring     ||       |               |
CA3 (NT)  Spring     -|       |------|        |
FA2 (NFI) Spring     |---|    |      |        |
FA3 (NFI) Spring     |   |----|      |--------|
FA3 (UF)  Spring     ----|           |
CA3 (PP)  Spring     |--|            |
FA3 (HK)  Spring     |  |------------|
FA2 (UF)  Spring     -|-|
FA2 (HK)  Spring     -|
```

Figure 17. There were four distinct clusters of sites based on log transformed invertebrate density data from both fall 1996 and spring 1997 (at a distance of 16.7, designated by ↓ on distance scale). Samples are indicated by site type (e.g., FA1 or CA1), a stream name code, and season. Cluster dendrogram was created using relative Euclidean distance and Ward's hierarchical linking method (McCune and Mefford 1997).

```
                            Distance (Objective Function)
             .040           ↓8.463         16.887          25.311           33.735
             |-------+-------+-------+-------+-------+-------+-------+-------+
                            Information remaining (%)
             100.000        75.000         50.000          25.000           .000
             |-------+-------+-------+-------+-------+-------+-------+-------+
CA1 (NT)     |
             ||
CA2 (NT)     ||
              |-------------|
CA3 (NT)     ||             |
             ||             |
FA3 (HK)     |              |
                            |
CA1 (PP)     |              |-------------|
             |              |             |
CA3 (PP)     |---|          |             |
             |   |          |             |
FA2 (HK)     |   |------|   |             |
                 |      |   |             |
CA2 (PP)     ----|      |---|             |--------------------------------|
                        |                 |                                |
FA2 (UF)     |          |                 |                                |
             |----------|                 |                                |
FA3 (UF)     |                            |                                |
                                          |                                |
                                          |                                |
                                          |                                |
CA2 (WL)     |                            |                                |
             |----------------------------|                                |
CA3 (WL)     |                                                             |
                                                                           |
                                                                           |
                                                                           |
CA1 (WL)     -----|                                                        |
                  |-|                                                      |
FA2 (NFI)    --|  | |                                                      |
               |--| |                                                      |
FA3 (NFI)    --|    |-|                                                    |
                    | |                                                    |
FA1 (UF)     ---|   | |                                                    |
                |---| |----------------------------------------------------|
FA1 (HK)     ---|     |
                      |
FA1 (NFI)    ---------|
```

Figure 18. There were three distinct clusters of sites based on log transformed invertebrate density data from October 1996 (at a distance of 8.4, designated by ↓ on distance scale). Each sample is indicated by site type (e.g., FA1 or CA1) and stream name code. Cluster dendrogram was created using relative Euclidean distance measure and Ward's hierarchical linking method (McCune and Mefford).

analysis figure. Invertebrate density data were log-transformed to equalize the high degree of variation among individual taxa and sites.

Seasonal differences appeared to be a major influence when comparing the invertebrate density from all sites and both sampling dates. There were four distinct clusters of sites (Figure 17; distance = 16.7). The top two clusters generally contain autumn samples, and the two sites that cluster together are sites with heavy cattle impact. The bottom two clusters generally contained spring samples. In comparison, the spring samples generally separated into two groups. One group included all the forested headwater (FA-1) sites from both seasons, and the other group contained the remaining spring sites with moderate agriculture. Reed *et al.* (1994) also noted strong seasonal influences when comparing invertebrate assemblages in forested and agricultural sites using cluster analyses. The clustering of FA-1 sites from both seasons, when most sites with agricultural influences generally separated according to season, suggests greater seasonal variability in agricultural sites and greater seasonal stability in forested sites. Lenat (1984) also noted such a pattern of seasonal variability in agriculturally impacted sites.

Invertebrate density data were also analyzed within each season using cluster analyses. The 18 sites in October 1996 clustered roughly into three groups (Figure 18; distance = 8.4). In the bottom cluster, all forested headwater (FA-1) sites clustered together with three sites influenced by light agricultural activity (two FA-2 sites and one CA-1 site). The middle cluster included the CA sites directly downstream of an active dairy farm located on a hillside adjacent to the stream. The remaining sites of moderate agriculture clustered together. Within each cluster, sites from each stream tended to be closely linked, a pattern also noted by similar analysis of invertebrate assemblages in forested and agricultural streams in Australia (Reed *et al.* 1994).

The quantitative invertebrate data analyzed using Reciprocal Averaging (RA; McCune and Mefford 1997) revealed patterns similar to the cluster analyses in each season, and I have presented the analysis of log transformed October 1996 data. Generally, CA stream sites separated from FA streams sites along RA Axis 1 (Figure 19). Correlation of invertebrate density data with axis values (McCune and Mefford 1997) indicated that the forested headwater sites to the right of RA Axis 1 tended to have high abundance of *Rhyacophila, Dolophilodes*, and *Epeorus*, all sensitive taxa (Lenat 1983). In contrast, the cleared sites with heaviest agricultural impact to the left end of RA Axis 1 tended to have high abundance of pollution tolerant baetid mayflies, oligochaetes, and chironomids (Lenat 1983). The horizontal (Axis 1) separation was most closely correlated with nitrate levels and the dominance of boulders in the substrate (best two-variable model, r^2 = 0.61; maximum r^2 multiple regression**;** SAS 1996). The forested headwater (FA-1) sites corresponded on this axis to low nitrate levels and a high percentage of boulders, and the agricultural sites below the dairy operation fell at the opposite end of the continuum. Richard *et al.* (1993) found similar results when using ordination techniques to analyze forested and agricultural streams in which the combination of nitrogen concentration and substrate explained the most variation among sites. The separation of sites

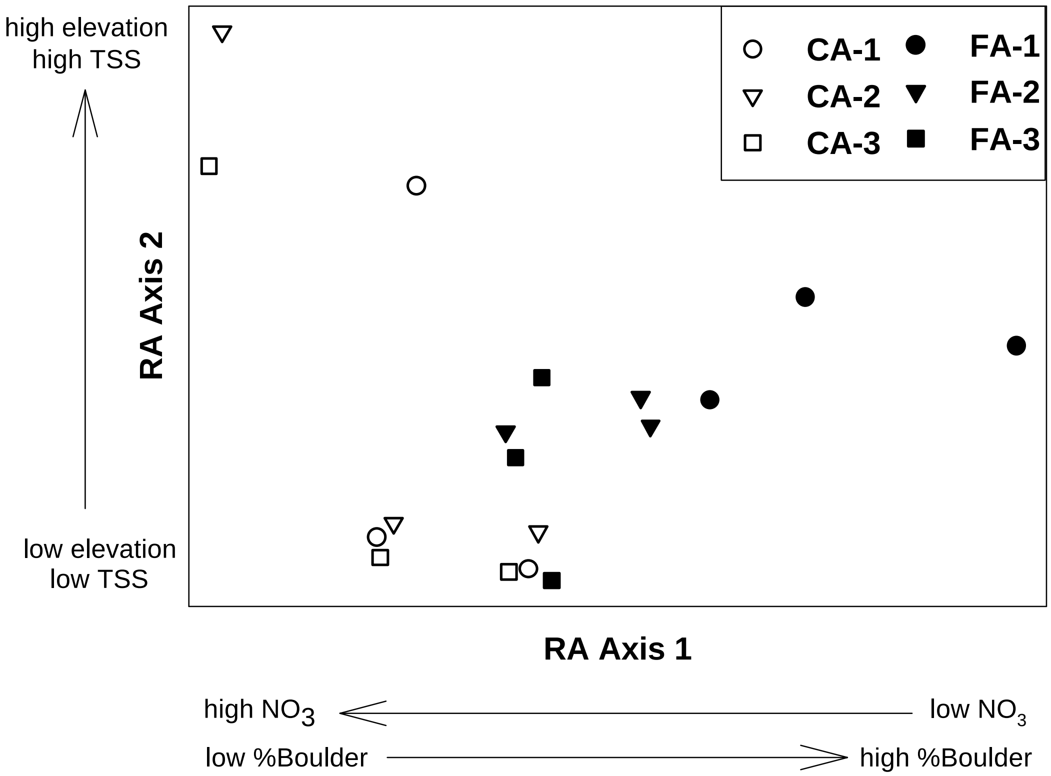


Figure 19. Reciprocal Averaging (RA) analysis of log transformed invertebrate density among all taxa collected October 1996. Correlated physico-chemical variables (best two-variable models; maximum r^2 multiple regression; SAS 1996) are shown with each RA axis.

along Axis 2, however, did not show as clear a trend. The CA stream sites around the dairy operation fell on the high end of Axis 2, the FA stream sites in the middle, and the remaining CA stream sites at the low end. The agricultural sites at the low end of Axis 2 tended to be dominated by *Ephemerella, Epeorus, Cheumatopsyche,* and *Stenonema* which are ubiquitous and generally tolerant taxa (Lenat 1983, 1984). Cleared agricultural stream sites near the dairy farm at the high end of Axis 2 contained an invertebrate not found in any other site as well as high abundance of baetid mayflies. The uncommon invertebrate taxon was the muscoid (Diptera: Muscidae) larvae which has often found on decaying organic matter mats on stream margins (Brigham *et al.* 1982), an environment which may have occurred if manure washed into the stream channel and onto the banks. Most of the forested stream sites fall in the middle of Axis 2, indicating an intermediate assemblage, *i.e.*, less dominated by the common taxa and generally more diverse, but with most taxa occurring in more than just one or two sites. The combination of elevation and total suspended solids (TSS) levels explained the most variation along RA axis 2 (best two-variable model, $r^2 = 0.60$). The cleared sites at the high end of Axis 2 had very high TSS concentrations and were higher in elevation than many of the other sites. The remaining sites displayed interactions between the two factors, and there were more differences in elevation than in TSS concentrations among these sites. The forested sites in the middle of the graph were generally higher in elevation but lower in TSS concentrations, while the agricultural sites at the low end were generally higher in TSS concentrations but lower in elevation.

Together, the multivariate analyses indicate that sites with extensive riparian cover (primarily FA-1 sites) and agricultural sites supported different invertebrate assemblages. Among agricultural sites, separation according to degree of agricultural impact. Similarly, Lenat (1984) found that agricultural stream invertebrate assemblages were different from those in forested streams and differences according to severity of agricultural impact could also be detected. This interpretation of how the invertebrate assemblages grouped together is supported by the correlation of RA axes with environmental variables that reflect degree of agriculture (*e.g.*, nitrate concentration, substrate size, and total suspended solids concentration).

Physico-chemical and land use relationships with invertebrates

Ten invertebrate diversity measures from each season's data and the combined qualitative data were compared with the environmental variables (maximum temperature, chlorophyll *a*, benthic organic matter, nutrient concentrations, total suspended solids, etc.) measured in this study. The environmental variables were evaluated using the maximum r^2 stepwise multiple regression method (SAS 1996) to identify the model that explains the most variation in each diversity measure. The best one-variable models were selected and results summarized in Table 15. Nine of the ten diversity measures were highly related to environmental variables ($P < 0.01$). Nitrate explained 45-63% of the variation among sites for 8 measures of diversity. Nitrate concentration reflected a gradient of agricultural impact ranging from full forest cover to heavy

Table 15. Best one-variable models relating diversity measures with various physico-chemical variables (maximum r^2 stepwise multiple regression; SAS 1996).

Invertebrate Diversity Measure	Season	Environmental Variable	r^2	P
Taxa Richness	Oct 1996	NO_3	0.58	< 0.001
	Apr 1997	NO_3	0.45	0.002
	Combined	NO_3	0.56	< 0.001
EPT Taxa	Oct 1996	NO_3	0.56	< 0.001
	Apr 1997	NO_3	0.41	0.004
	Combined	NO_3	0.47	0.002
Density	Oct 1996	PO_4	0.35	0.001
	Apr 1997	Discharge	0.12	n.s.
Margalef's Index	Oct 1996	NO_3	0.63	< 0.001
	Apr 1997	NO_3	0.45	0.002

influences of cattle operations on some cleared streams, and these impacted sites were generally characterized by low diversity. Thus, nitrate concentration may be an indicator for assessing agricultural impacts on stream invertebrate assemblages.

The various invertebrate diversity measures were also compared to assessments of present and historical land-use. The GIS data provided information about the percent of the watershed forested in 1950s, 1970s, and 1990s at several spatial scales. Spatial area was considered both longitudinally (entire upstream watershed or upstream segment, *i.e.,* the upstream segment between sites or between the top site and the headwaters) and laterally (entire watershed, 100m riparian width, or 30m riparian width). The land use variables were evaluated using the maximum r^2 stepwise multiple regression method (SAS 1996) to determine the model which explained the most variation in each diversity measure. Upper Flat Creek sites were not tested due to absence of 1950s GIS data. The best one-variable models were selected and results summarized in Table 16. Nine of the ten diversity measures were significantly correlated with a single land-use variable (Table 16). All diversity measures that were significantly correlated with a single variable were related to land use in either the 1950s or the 1970s. This indicates that past land use may have continuing influences on present invertebrate assemblages. The significantly correlated variables also provide information about the spatial scales that influence invertebrate diversity. Seven of these variables related to land-use not just adjacent to the stream but up-slope to the watershed boundary. Additionally,

Table 16. Best one-variable models relating diversity measures with the percentage of area in forest at multiple temporal and spatial scales (maximum r^2 stepwise multiple regression; SAS 1996).

Invertebrate Diversity Measure	**Season**	**Watershed Variable—spatial and temporal scale (Year—Upstream—Riparian)[1]**	**r^2**	**P**
Taxa Richness	Oct 1996	1970—Segment—WS	0.27	0.04
	Apr 1997	1950—Segment—WS	0.60	< 0.001
	Combined	1970—Segment—WS	0.26	0.05
EPT Taxa	Oct 1996	1970—Segment—WS	0.27	0.05
	Apr 1997	1950—Segment—WS	0.51	0.003
	Combined	1950—Segment—WS	0.30	0.04
Density	Oct 1996	1950—WS—100m	0.34	0.02
	Apr 1997	1970—Segment—100m	0.20	n.s.
Margalef's Index	Oct 1996	1970—Segment—WS	0.44	0.007
	Apr 1997	1950—Segment—100m	0.56	0.001

[1]Year—Upstream—Riparian:
Year = 1950, 1970, 1990 land use
Upstream = Entire watershed or segment immediately upstream
Riparian = Entire watershed, 30m, or 100m riparian width

most related variables assessed land-use in the segments directly upstream of the sampling site, not the entire watershed. Allan *et al.* (1997) discussed the relationship of land use variables at several spatial scales with stream biotic integrity in Michigan agricultural streams. They found land use within the entire watershed to be more important than stream-side land use, although contradicting studies were noted. The results of this study in North Carolina coincide somewhat with the Michigan study, indicating that while watershed conditions may be influential, it may be more useful to focus on land use in the area immediately upstream, particularly where point sources can be identified.

The relationships between the diversity measures and land use were not as strong (as indicated by percent of variation explained, r^2) as between diversity and nitrate concentrations (Table 15 and 16). This may indicate that while past land use is important in explaining current invertebrate assemblages, the specific type and degree of agriculture (i.e., intense cattle grazing and organic pollution problems) in the upstream area may have a stronger influence than the past land use legacy.

SUMMARY AND CONCLUSIONS

Physico-chemical variables indicated many differences between cleared and forested agricultural streams. Cleared agricultural (CA) streams tended to have lower benthic organic matter, more primary production, higher nitrate concentrations, warmer temperatures, and lower habitat heterogeneity and quality than forested agricultural (FA) streams. These findings followed general predictions based on published studies describing differences between forested and agricultural streams.

Forested and cleared agricultural streams also supported different invertebrate assemblages. The invertebrate assemblage in FA streams was generally more diverse, particularly in the sensitive EPT taxa. Trichoptera in particular were more diverse in sites with more riparian vegetation. However, total taxa richness was not related to riparian vegetation but instead to nitrate concentration. Low taxa richness at sites with high nitrate concentrations possibly indicates the impact of cattle operations. Additionally, Plecoptera taxa were relatively more abundant in FA streams. High plecopteran abundance was related to high habitat quality, more riparian vegetation, and lower summer temperatures.

Invertebrate diversity and assemblage structure was explained by both current agricultural practices and the history of land use at each site. The invertebrate assemblages were most similar among sites with similar land use: complete forest cover, light to moderate agriculture, and heavy cattle impact.

The differences between the riparian vegetation on FA and CA streams have been present for at least 50 years, and higher invertebrate diversity was related to the land use 25-50 years ago as well as the current land use. These results may indicate a long-term legacy of agriculture on current invertebrate assemblages, and therefore it may be necessary to know more about a watershed than just the current total percent forest to predict the type of invertebrate assemblage that may be present at a particular stream site.

LITERATURE CITED

Allan, J.D. 1995. Stream ecology: structure and function of running waters. Chapman & Hall. New York, NY. 388 p.

Allan, J.D., D.L. Erickson, and J. Fay. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. Freshwater Biology 37: 149-161.

Barbour, M.T. and J.B. Stribling. 1994. A technique for assessing stream habitat structure. Pg. 156-178 *in* Proceedings of the conference "Riparian ecosystems in the humid United States: function, values, and management." National Association of Conservation Districts, Washington, DC.

Bilby, R.E. 1981. Role of organic debris dams in regulating the export of dissolved and particulate matter from a forested watershed. Ecology 62: 1234-1243.

Brigham, A.R., W.U. Brigham, and A. Gnilka, eds. 1982. Aquatic insects and oligochaetes of North and South Carolina. Midwest Aquatic Enterprises, Mahomet, Illinois, USA. 837 p.

Clesceri, L.S., A.E. Greenburg, and R.R. Trussel, eds. 1989. Standard methods for the examination of water and wastewater. Port City Press, Baltimore, Maryland.

Cobb, D.G., T.D. Galloway, and J.F. Flannagan. 1992. Effects of discharge and substrate stability on density and species composition of stream insects. Canadian Journal of Fisheries and Aquatic Sciences 49: 1788-1795.

Corkum, L.D. 1996. Responses of chlorophyll-a, organic matter, and macroinvertebrates to nutrient additions in rivers flowing through agricultural and forested land. Archiv für Hydrobiologie 136: 391-411.

Cummins, K.W. 1983. Structure and function of stream ecosystems. BioScience 24: 631-641.

Cummins, K.W., G.W. Minshall, J.R. Sedell, C.E. Cushing, and R.C. Petersen. 1984. Stream ecosystem theory. Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie 22: 1818-1827.

Dance, K.W., and H.B.N. Hynes. 1980. Some effects of agricultural land use on stream insect communities. Environmental Pollution (Series A) 22: 19-28.

DeMarch, B.G.E. 1976. Spatial and temporal patterns in macrobenthic stream diversity. Journal of the Fisheries Research Board of Canada 33: 1261-1270.

Dudgeon, D. 1989. The influence of riparian vegetation on the functional organization of four Hong Kong stream communities. Hydrobiologia 179: 183-194.

Dunn, T. and L.B. Leopold. 1978. Water in environmental planning. W.H. Freeman and Company: San Francisco. 818 p.

Egglishaw, H.J. 1964. The distributional relationship between the bottom fauna and plant detritus in streams. Journal of Animal Ecology 33: 463-476.

Ehrman, T.P., and G.A. Lamberti. 1991. Hydraulic and particulate matter retention in a 3rd-order Indiana stream. Journal of the North American Benthological Society 11: 341-349.

Epler, J.H. 1995. Identification manual for the larval Chironomidae (Diptera) of Florida. Final report for Florida Division of Environmental Protection Contract Number WM579.

Findlay, S., G.E. Likens, L. Hedin, S.G. Fisher, and W.H. McDowell. 1997. Organic matter dynamics in Bear Brook, Hubbard Brook Experimental Forest, New Hampshire, USA. Journal of the North American Benthological Society 16: 43-46.

Giller, P.S., and H. Twomey. 1993. Benthic macroinvertebrate community organisms in two contrasting rivers—between-site differences and seasonal patterns. Biology and environment: proceedings of the Royal Irish Academy 93B: 114-126.

Gore, J.A. 1996. Discharge measurements and stream flow analysis. pp 53-74 *in* F.R. Hauer and G.A. Lamberti (eds.). Methods in stream ecology. Academic Press, New York. 674 p.

Graham, A.A. 1990. Siltation of stone-surface periphyton in rivers by clay-sized particles from low concentration in suspension. Hydrobiologia 199: 107-115.

Harding, J.S., and M.J. Winterbourn. 1995. Effects of contrasting land use on physico-chemical conditions and benthic asssemblages of streams in a Canterbury (South Island, New Zealand) river system. New Zealand Journal of Marine and Freshwater Research 29: 479-492.

Hauer, F.R., and W.R. Hill. 1995. Temperature, light, and oxygen. Pp. 93-106 *In* F.R. Hauer and G.A. Lamberti (eds.) Methods in Stream Ecology. Academic Press: New York. 674 p.

Hawkins, C.P., M.L. Murphy, and N.H. Anderson. 1982. Effects of canopy, substrate composition, and gradient on the structure of macroinvertebrate communities in Cascade Range streams of Oregon. Ecology 63: 1840-1856.

Hawkins, C.P., J.N. Hogue, L.M. Decker, and J.W. Feminella. 1997. Channel morphology, water temperature, and assemblage structure of stream insects. Journal of North American Benthological Society 16: 728-749.

Hynes, H.B.N. 1970. The ecology of running waters. Liverpool University Press, Liverpool. 555 p.

Hynes, H.B.N. 1975. The stream and its valley. Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie 19: 1-15.

Lenat, D.R. 1984. Agriculture and stream water quality: a biological evaluation of erosion control practices. Environmental Management 8: 333-344.

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

Lenat, D.R. 1990. Reducing variability in freshwater macroinvertebrate data. *In* Proceedings of the 1990 Midwest Pollution Control Biologists Meeting.

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

Lenat, D.R., D.L. Penrose, and K.W. Eagleson. 1981. Variable effects of sediment addition on stream benthos. Hydrobiologia 79: 187-194.

Lester, P.J., S.F. Mitchell, and D. Scott. 1996. Substrate and shade: mechanisms of willow tree influence on the macroinvertebrate community of Henney Creek, South Island, New Zealand. Archiv für Hydrobiologie 136: 145-158.

Magurran, A.E. 1988. Ecological diversity and its measurement. Princeton University Press, New Jersey, USA. 179 p.

Marsh, P.C., and T.F. Waters. 1980. Effects of agricultural drainage development on benthic invertebrates in undistrubed downstream reaches. Transactions of the American Fisheries Society 190: 213-223.

Marxsen, J., H. Schmidt, and D.M. Fiebig. 1997. Organic matter dynamics in the Breitenbach, Germany. Journal of the North American Benthological Society 16: 28-33.

McCune, B., and M.J. Mefford. 1997. Multivariate analysis of ecological data: Version 3.0. MjM Software, Gleneden Beach, Oregon, USA.

Merrit, R.W., and K.W. Cummins. 1996. An introduction to the aquatic insects of North America. Kendall-Hunt Publishing Company. Dubuque, Iowa. 862 p.

Minshall, G.W. 1967. Role of allochthonous detritus in the trophic structure of a woodland springbrook community. Ecology 48: 139-149.

Minshall, G.W. 1984. Aquatic insect-substratum relationships. *In* V.H. Resh and D.M. Rosenberg (eds.) The ecology of aquatic insects. Praeger Publishers, New York. 625 p.

Minshall, G.W., K.W. Cummins, R.C. Petersen, C.E. Cushing, D.A. Bruns, J.R. Sedell, and R.L. Vannote. 1985. Developments in stream ecosystem theory. Canadian Journal of Fisheries and Aquatic Sciences 42: 1045-1055.

Minshall, G.W., R.C. Petersen, K.W. Cummins, T.L. Bott, J.R. Sedell, C.E. Cushing, and R.L. Vannote. 1983. Interbiome comparison of ecosystem dynamics. Ecological Monographs 53: 1-25.

Mulholland, P.J. 1997. Organic matter dynamics in the West Fork of Walker Branch, Tennessee, USA. Journal of the North American Benthological Society. 16: 61-67.

North Carolina State Board of Agriculture. 1896. North Carolina and its resources. M.I. & J.C. Stewart, Winston, North Carolina. 413 p.

Palmer, M.A., P. Arensburger, A.P. Martin, and D.W. Denman. 1996. Disturbance and patch-specific responses: the interactive effects of woody debris and floods on lotic invertebrates. Oecologia 105: 247-257.

Pennak, R.W. 1989. Fresh-water invertebrates of the United States: Protozoa to Mollusca. John Wiley & Sons, Inc., New York.

Penrose, D.L., D.R. Lenat, and K.W. Eagleson. 1982. Aquatic macroinvertebrates of the Upper French Broad River Basin. Brimleyana 8: 27-50.

Quinn, J.M., A.B. Cooper, R.J. Davies-Colley, J.C. Rutherford, and R.B. Williamson. 1994. Land-use effects on New Zealand hill country streams and implications for riparian management. *In* Proceedings of the International Workshop on the Ecology and Management of Aquatic-Terrestrial Ecotones.

Quinn, J.M., Williamson, R.B., Smith, R.K., and Vickers, M.L. 1992. Effects of riparian grazing and channelization on streams in Southland New Zealand. 2. Benthic invertebrates. New Zealand Journal of Marine and Freshwater Research 26: 259-273.

Reed, J.L., I.C. Campbell, and P.C.E. Bailey. 1994. The relationship between invertebrate assemblages and available food at forest and pasture sites in three south-eastern Australian streams. Freshwater Biology 32: 641-650.

Richards, C., G.E. Host, and J.W. Arthur. 1993. Identification of predominant environmental factors structuring macroinvertebrate communities within a large agricultural catchment. Freshwater Biology 29: 285-294.

Roth, N.E., J.D. Allan, and D.L. Erickson. 1996. Landscape influences on stream biotic integrity assessed at multiple spatial scales. Landscape Ecology 11: 141-156.

Ryan, P.A. 1991. Environmental effects on New Zealand streams: a review. New Zealand Journal of Marine and Freshwater Research 25: 207-221.

SigmaStat for Windows, Version 2.0. 1995. Jandel Corporation.

Simmons, C.E., and R.C. Heath. 1979. Water-quality characteristics of streams in forested and rural areas of North Carolina. *In* Water Quality of North Carolina Streams. Geological Survey Water—Supply paper 2185 A-D.

Skidmore, A.K. 1989. A comparison of techniques for calculating gradient and aspect from a gridded elevation model. International Journal of Geographic Information Systems 3: 323-334.

Smith, E. P., and J. R. Voshell, Jr. 1997. Studies of benthic macroinvertebrates and fish in stream within EPA Region 3 for development of biological indicators of ecological conditoin. Report submitted to U.S. Environmental Protection Agency. 23 pp + appendices.

Smith, K., and M.E. Lavis. 1975. Environmental influences on the temperature of a small upland stream. Oikos 26: 228-246.

Söderström, O. 1988. Effects of temperature and food quality on life-history parameters in *Parameletus chelifer* and *P. minor* (Ephemeroptera): a laboratory study. Freshwater Biology 20: 295-303.

Southern Appalachian Man and the Biosphere (SAMAB). 1996. The Southern Appalachian Assessment Aquatics Technical Report. Report 2 of 5. Atlanta: U.S. Department of Agriculture, Forest Service, Southern Region.

Steinman, A.D., and G. A. Lamberti. 1996. Biomass and pigments of benthic algae. Pp. 295-313 *in* F.R. Hauer and G.A. Lamberti (eds.) Methods in Stream Ecology. Academic Press: New York.

Stewart, K.W. and B.P. Stark. 1993. Nymphs of North American stonefly genera (Plecoptera). University of North Texas Press, Denton, Texas. 460 p.

Stewart, K.W. and P.P. Harper. 1996. pp. 217-266. *In* Merrit, R.W. and K.W. Cummins. An introduction to the aquatic insects of North America. Kendall-Hunt Publishing Company. Dubuque, Iowa. 862 p.

Swank, W.T. and J.B. Waide. 1988. Characterization of baseline precipitation and stream chemistry and nutrient budgets for control watersheds. Pp. 57-79 *In* Forest hydrology and ecology at Coweeta. W.T. Swank and D.A. Crossley, Jr. (eds.). Springer-Verlag: New York.

Sweeney, B.W., R.L. Vannote, and P.J. Dodds. 1986. The relative importance of temperature and diet to larval development and adult size of the winter stonefly, *Soyedina carolinensis* (Plecoptera: Nemouridae). Freshwater Biology 16: 39-48.

Townsend, C.R., A.G. Hildrew, and J. Francis. 1983. Community structure in some southern English streams: the influence of physicochemical factors. Freshwater Biology 13: 521-544.

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

Wallace, J.B., T.F. Cuffney, S.L. Eggert, and M.R. Whiles. 1997. Organic matter dynamics in Satellite Branch at Coweeta Hydrological Laboratory, North Carolina, USA. 16: 67-74.

Waters, T.F. 1995. Sediment in streams: sources, biological effects and control. American Fisheries Society, Bethesda, Maryland. 251 pp.

Webster, J.R., J.L. Meyer, J.B. Walace, and E.F. Benfield. 1997. Organic matter dynamics in Hugh White Creek, Coweeta Hydrologic Laboratory, North Carolina, USA. Journal of the North American Benthological Society. 16: 74-82.

White, D.S. 1982. Elmidae, pp. 10.99-10.110 *In* A.R. Brigham, W.U. Brigham and A. Gnilka (eds.). Aquatic insects and oligochaetes of North and South Carolina. Midwest Aquatic Enterprises, Mahomet, Illinois.

Wiederholm, T. 1984. Responses of aquatic insects to environmental pollution. 1984. *In* V.H. Resh and D.M. Rosenberg (ed.) The ecology of aquatic insects. Praeger Publishers, New York. 625 p.

Wiggins, G.B. 1996. Larvae of the North American caddisfly genera (Trichoptera). University of Toronto Press, Toronto. 457 p.

Wiley, M.J., L.L. Osborne, and R.W. Larimore. 1990. Longitudinal structure of an agricultural prairie river system and its relationship to current stream ecosystem theory. Canadian Journal of Fisheries and Aquatic Sciences 47: 373-384.

Willow Creek Project Summary. 1982. North Carolina Division of Environmental Management, internal report.

Wolf, P.R. 1983. Elements of Photogrametry. McGraw-Hill, New York, 628 p.

Appendix A. List of all aquatic invertebrate taxa collected at all sites from both October 1996 and April 1997. Functional feeding groups assigned to each taxa are included (CG = Collector-Gatherer; CF = Collector Filterer; P = Predator; SC = Scraper; SH = Shredder).

Insect Taxa

Order Coleoptera

Family Curculionidae

Listronotus (SH)

Family Elmidae

Ancyronyx variegatus (CG)

Dubiraphia (CG)

Macronychus glabratus (CG)

Optioservus immunis (SC)

Optioservus ovalis (SC)

Oulimnius latisculus (SC)

Promoresia tardella (SC)

Stenelmis bicarinata (SC)

Stenelmis concinna (SC)

Unknown but distinct (SH)

Family Psephenidae

Ectopria (SC)

Psephenus herricki (SC)

Family Ptiodactylidae

Anchytarsus bicolor (SH)

Family Scirtidae (SC)

Family—Unknown but distinct (CG)

Order Collembola (CG)

Order Diptera

Family Athericidae

Atherix lantha (P)

Family Blephariceridae

Blepharicera (SC)

Family Ceratopogonidae

Atrichopogon (CG)

Bezzia/Palpomyia (P)

Culicoides (P)

Dasyhelea (CG)

Probezzia (P)

Stilobezzia (P)

Family Chironomidae

Brillia flavifrons (SH)

Brillia sera (SH)

Cardiocladius obscurus (P)

Chironomus riparius (CG)

Conchapelopia (P)

Cricotopus (CG)

Cryptochironomus (P)

Diamesa (CG)

Eukiefferiella brehmi gr. (CG)

Eukiefferiella brevicalcar gr. (CG)

Eukiefferiella claripennis gr. (CG)

Eukiefferiella gracei gr. (CG)

Eukiefferiella pseudomontana gr. (CG)

Microtendipes pedellus gr. (CF)

Nanocladius (CG)

Natarsia baltimorea (P)

Orthocladius annectens (CG)

Parakiefferiella (CG)

Paramerina (P)

Parametriocnemus (CG)

Polypedilum aviceps (CG)

Polypedilum convictum gr. (CG)

Polypedilum fallax (CG)

Polypedilum halterale gr. (CG)

Potthasia (CG)

Pseudorthocladius (CG)

Psilometriocnemus (CG)

Rheotanytarsus (CF)

Tanytarsus sp. T (CF)

Thienemanniella xena (CG)

Family Dixidae

Dixa (CG)

Family Empididiae

Clinocera (P)

Hemerodromia (P)

Family Muscidae (P)

Family Psychodidae

Pericoma (CG)

Psychoda (CG)

Family Simuliidae

Cnephia (CF)

Prosimulium (CF)

Simulium (CF)

Family Stratiomyidae

Stratiomys (P)

Family Tabanidae

Chlorotabanus (P)

Family Tanyderidae

Protoplasa fitchii (CG)

Family Tipulidae

Antocha (CG)

Cryptolabis (CG)

Dicranota (P)

Hexatoma (P)

Molophilus (SH)

Tipula (SH)

Order Epemeroptera

Family Ameletidae

Ameletus lineatus (SC)

Family Baetidae

Acentrella ampla (CG)

Baetis brunneicolor (CG)

Baetis intercalaris (CG)

Baetis pluto (CG)

Baetis tricaudatis (CG)

Pseudocleon (w/ tail band) (CG)

Pseudocleon (w/o tail bands) (CG)

Family Baetiscidae

Baetisca carolina (CG)

Baetisca gibbera (CG)

Family Ephemerellidae

Drunella conestee (CG)

Drunella lata (CG)

Ephemerella catawba (CG)

Ephemerella crenula (CG)

Ephemerella dorothea (CG)

Ephemerella excrucians (CG)

Ephemerella hispida (CG)

Ephemerella inconstans (CG)

Ephemerella invaria (CG)

Ephemerella rotunda (CG)

Eurylophella doris (CG)

Serratella (CG)

Family Ephemeridae

Ephemera blanda (CG)

Ephemera guttalata (CG)

Family Heptageniidae

Cinygmula subaequalis (SC)

Epeorus dispar (CG)

Epeorus pleuralis (CG)

Epeorus rubidis (CG)

Epeorus vitreus (CG)

Heptagenia julia (SC)

Rithrogena amica (CG)

Rithrogena uhari (CG)

Stenacron carolina (CG)

Stenacron pallidum (CG)

Stenonema carlsoni (SC)

Stenonema exiguum (SC)

Stenonema ithaca (SC)

Stenonema mediopunctatum (SC)

Stenonema modestum (SC)

Stenonema pudicum (SC)

Stenonema terminatum (SC)

Family Isonychiidae

Isonychia (CF)

Family Leptophlebiidae

Paraleptophlebia (CG)

Family Neoephemeridae

Neoephemera purpurea (CG)

Order Hemiptera (P)

Order Lepidoptera (SH)

Order Megaloptera

Family Corydalidae

Corydalus cornutus (P)

Nigronia serricornis (P)

Family Sialidae

Sialis aequalis (P)

Order Odonata

Family Aeshnidae

Boyeria grafiana (P)

Boyeria vinosa (P)

Family Gomphidae

Gomphus lividis (P)

Lanthus parvulus (P)

Ophiogomphus mainensis (P)

Order Plecoptera

Family Capniidae

Allocapnia (SH)

Nemocapnia carolina (SH)

Family Chloroperlidae

Alloperla (P)

Haploperla brevis (P)

Utaperla gaspesiana (P)

Family Leuctridae

Leuctra (SH)

Family Nemouridae

Amphinemura wui (SH)

Family Peltoperlidae

Peltoperla (SH)

Tallaperla (SH)

Family Perlidae

Acroneuria abnormis (P)

Beloneuria (P)

Hansonoperla appalachia (P)

Paragnetina immarginata (P)

Perlesta (P)

Phasganophora capitata (P)

Family Perlodidae

Cultus decisus (P)

Diploperla (P)

Isogenoides (P)

Isoperla bilineata (P)

Isoperla holochlora (P)

Isoperla lata (P)

Isoperla nana (P)

Yugus arinus (P)

Family Pteronarcyidae

Pteronarcys (SH)

Family Taeniopterygidae

Oemopteryx contorta (SH)

Order Trichoptera

Family Brachycentridae

Brachycentrus (SH)

Micrasema (SH)

Family Glossosomatidae

Glossosoma (SC)

Family Goeridae

Goera (SC)

Family Hydropsychidae

Arctopsyche irrorata (CF)

Cheumatopsyche (CF)

Diplectrona modesta (CF)

Hydropsyche betteni (CF)

Hydropsyche bronta (CF)

Hydropsyche macleodi (CF)

Hydropsyche slossonae (CF)

Hydropsyche sparna (CF)

Hydropsyche venularis (CF)

Parapsyche cardis (CF)

Family Hydroptilidae

Leucotrichia pictipes (SC)

Family Limnephilidae

Pycnopsyche (SC)

Family Odontoceridae (SH)

Family Philopotamidae

Chimarra (CF)

Dolophilodes (CF)

Wormaldia (CF)

Family Polycentropidae

Neureclipsis crepuscularis (CF)

Nyctiophylax (P)

Polycentropus (CF)

Family Psychomyiidae

Lype diversa (SC)

Family Rhyacophilidae

Rhyacophila amicis (P)

Rhyacophila atrata (P)

Rhyacophila carolina ? (P)

Rhyacophila fuscula (P)

Rhyacophila nigrita (P)

Family Uenoidae

Neophylax consimilis (SC)

Neophylax mitchelli (SC)

Neophylax oligius (SC)

Neophylax ornatus (SC)

Other, Non-Insect Taxa

Phylum Annelida

Class Oligochaeta (CG)

Phylum Arthropoda

Subphylum Chelicerata

Class Hydracarina (P)

Subphylum Crustacea

Class Malacostraca

Order Isopoda (CG)

Order Decapoda (CG)

Phylum Mollusca

Class Bivalvia (CF)

Class Gastropoda

Subclass Prosobranchia

Family Pleuroceridae (SC)

Subclass Pulmonata

Family Ancylidae (SC)

Phylum Nematoda (P)

Appendix B. List of 31 taxa collected only in cleared agricultural (CA) streams.

Ephemeroptera
Baetis pluto
Baetis tricaudatis
Ephemera guttalata
Ephemerella inconstans
Ephemerella rotunda
Rithrogena uhari
Stenacron pallidum
Stenonema exiguum

Odonata
Boyeria vinosa

Plecoptera
Alloperla
Isoperla bilineata
Nemocapnia carolina
Peltoperla
Perlesta

Trichoptera
Neophylax oligius

Coleoptera
Anchytarsus bicolor
Ancyronyx variegatus
Coleoptera—Unknown but distinct
Elmidae—Unknown but distinct

Diptera
Chironomus riparius
Clinocera
Cryptochironomus
Muscidae
Paramerina
Polypedilum fallax
Polypedilum haltere gr.
Probezzia
Protoplasa fitchii
Psychoda
Stratiomys

Misc.
Isopoda

Appendix C. List of 46 taxa collected only in forested agricultural (FA) streams. Taxa followed by an asterisk (*) were collected only in the forested headwater sites (FA-1 sites) and not in the downstream agricultural sites (FA-2 or FA-3 sites).

Ephemeroptera
Drunella lata
Ephemerella excrucians *
Heptagenia julia
Neoephemera purpurea
Stenonema carlsoni

Odonata
Boyeria grafiana

Plecoptera
Diploperla *
Hansonoperla appalachia
Haploperla brevis *
Isogenoides
Isoperla holochlora
Utaperla gaspesiana

Megaloptera
Sialis aequalis *

Trichoptera
Arctopsyche irrorata *
Brachycentrus *
Diplectrona modesta
Goera
Hydropsyche slossonae
Lype divera *
Neophylax ornatus *
Neureclipsis crepuscularis

Trichoptera (cont.)
Nyctiophylax
Odontoceridae *
Parapsyche cardis *
Rhyacophila atrata
Rhyacophila carolina
Rhyacophila nigrita *

Coleoptera
Listronotus
Scirtidae *
Stenelmis concinna

Diptera
Atrichopogon
Brillia flavifrons
Cryptolabis
Dasyhelea *
Eukiefferiella brevicalcar *
Eukiefferiella pseudomontana
Nanocladius
Natarsia baltimorea
Parakiefferiella
Pericoma *
Potthasia
Pseudorthocladius *
Psilometriocnemus *
Rheotanytarsus
Stillobezzia *
Tanytarsus sp. T

Barbara Loraine Bennett

CURRENT ADDRESS

Department of Biology
Virginia Polytechnic Institute and State University
Blacksburg, VA 24061-0406

PERMANENT ADDRESS

918 Love Lane
Taylorsville, KY 40071

PERSONAL DATA

Date of Birth: 25 November 1973
Place of Birth: Fredericksburg, Virginia

EDUCATION

Master of Science, Biology, 1998
Virginia Polytechnic Institute and State University (Virginia Tech), Blacksburg, VA
Thesis title: Land use influences on benthic invertebrate assemblages in southern Appalachian agricultural streams.
Advisor: Dr. Fred Benfield

Bachelor of Arts, Biology; Minor: Chemistry, May 1995
Graduated Magna Cum Laude, with Honors in Biology
Transylvania University, Lexington, KY
Advisor: Dr. Joseph Holomuzki

HONORS/AFFILIATIONS

Cunningham Fellowship (Virginia Tech research fellowship), 1995-1998
North American Benthological Society, 1995-1998, Graduate Student Auction Commitee—1998
Sigma Xi, The Scientific Research Society, 1995-1998
Virginia Academy of Science, 1996-1998
Virginia Tech Biology Graduate Student Association, 1995-1998
Maxine Troxel MacIntyre Biology Award, Transylvania University, 1995
Transylvania University Science Honorary, 1993-1995
William T. Young Scholar (Transylvania University full scholarship), 1991-1995
Kentucky Governor's Scholars Program, 1990

EXPERIENCE

Research

Graduate Research, Dept. of Biology, Virginia Tech, Blacksburg, VA, 1995-1998

- Quantitative and qualitative sampling of stream benthic macroinvertebrates
- Identification of freshwater invertebrates
- Supervised several undergraduates in processing invertebrate samples
- Measured routine physical and chemical variables in streams
- Measured nutrients and chlorophyll a concentrations with standard laboratory instruments
- Set up and monitored long-term temperature monitoring network in study streams
- Data analysis and presentation using Microsoft Excel, SigmaStat, Minitab, PC-ORD, Sigma Plot, and PowerPoint

Research Assistant, Dept. of Biology, Virginia Tech, Blacksburg, VA 1996-1997

- Benthic invertebrate identification
- Coordinated and executed microbial respiration field study, measuring respiration with a Gilson respirometer
- Organized field work related to study of nutrient uptake lengths, including preparing equipment, executing procedures in the field, and supervising undergraduate and graduate students
- Data analysis using Microsoft Excel, Sigma Stat, and Sigma Plot

National Science Foundation—Research Experience for Undergraduates, Department of Biology, Truman State University, Kirksville, MO, 1994

- Designed and conducted research testing if the presence of an endophytic fungus in the diet of an insect herbivore influence host preference by a parasitic wasp.

Research Assistant—NSF/EPSCoR grant, Department of Biology, Transylvania University, Lexington, KY, 1994

- Helped design and conduct research assessing fish (*Cottus*) impacts on behavior of a mayfly (*Stenonema*).
- Faculty Researcher: Dr. Joseph Holomuzki

Lab Assistant—Organic Chemistry II, Department of Chemistry, Transylvania University, Lexington, KY 1994

- Operated HP 5890A Gas Chromatograph/5970 Mass Spectrometer for all independent research projects being conducted as a part of Organic Chemistry II lab.

Teaching

Teaching Assistant, Dept. of Biology, Virginia Tech, Blacksburg, VA, 1995-1997

- Taught lab sections of Freshwater Ecology course for two semesters (1996-97); supervised students in field and lab; evaluated student research projects.
- Taught lab sections of General Biology (for non-majors) and Principles of Biology (majors) for three semesters (1995-1996, 1998).

Teaching Assistant, Dept. of Biology, Transylvania University, Lexington, KY 1993
- Assisted professor with general biology labs, including some pre-lab lectures.

PAPERS SUBMITTED

Webster, J.R., J.L. Tank, J.B. Wallace, J.L. Meyer, S.L. Eggert, T.P. Ehrman, B.R. Ward, B.L. Bennett, P.F. Wagner, and M.E. McTammany. Effects of litter exclusion and wood removal on phosphorus and nitrogen retention in a forest stream. Submitted Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie, August 1998.

ABSTRACTS AND PRESENTATIONS

Bennett, Barbara L., E.F. Benfield, and J.S. Harding. Land use influences on benthic invertebrate assemblages in southern Appalachian agricultural streams. Presented at North American Benthological Society meeting, Prince Edward Island, Canada, 1998.

Bennett, Barbara L. and E.F. Benfield. Does headwater land-use influence downstream invertebrate assemblages in agricultural streams? Presented at the North American Benthological Society meeting, San Marcos, TX, and the All-Coweeta LTER Scientists meeting, Athens, GA, 1997.

Webster, J.R., J.L. Tank, J.B. Wallace, J.L. Meyer, S.L. Eggert, T.P. Ehrman, B.R. Ward, B.L. Bennett, P.F. Wagner, M.E. McTammany. Effects of leaf litter exclusion and wood removal on phosphorus and nitrogen retention in a first-order forest stream. Presented at the North American Benthological Society meeting, San Marcos, TX, 1997.

Bennett, Barbara L. Effects of fungal endophytes on parasitoid host choice: a tri-trophic investigation. Presented at the National Conference on Undergraduate Research, NY, 1995.

GRANT PROPOSALS FUNDED

Virginia Academy of Science. April 1997. Funded for $772; matched $500 by Virginia Tech Biology Department.

Sigma Xi. February 1997. Funded for $700; matched $500 by Virginia Tech Biology Department.

Sigma Xi. May 1996. Funded for $400; matched $500 by Virginia Tech Biology Department.

PROFESSIONAL DEVELOPMENT

North American Benthological Society Teaching Workshop participant, San Marcos, TX, May 1997.

Model Inquiries into Nature in the Schoolyard (MINTS) Program, Educators Workshop participant, Virginia Tech Museum of Natural History, Blacksburg, VA, May 1997.

GTAs using Writing: Issues and Challenges Workshop participant, Virginia Tech, Blacksburg, VA, November 1996.

DEPARTMENTAL SERVICE

Selection Committee member for Principles of Biology/General Biology Lab Coordinator position, Department of Biology, Virginia Tech, Blacksburg, VA, May 1997.

Treasurer and Fundraising Chair, Biology Graduate Student Association, Virginia Tech, Blacksburg, VA 1997.