

**Influence of Agricultural Land Use on Allochthonous Input  
and Leaf Breakdown in Southern Appalachian Streams**

by

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## **Abstract**

Streams and terrestrial ecosystems are linked through allochthonous organic matter inputs from streamside vegetation. This allochthonous material makes up the energy base for forested aquatic food webs. Therefore, removal of riparian vegetation associated with agricultural land use affects stream ecosystem structure and function. The objectives of this study were to measure and compare allochthonous input and leaf breakdown rates along a gradient of agricultural land use in southern Appalachian streams. Study streams were placed into the following land use categories: forest and light, moderate, and heavy agriculture. Several physical, chemical, and biological parameters also were measured including discharge, temperature, nutrient concentrations, macroinvertebrate abundance and density, periphyton biomass, and chlorophyll *a* concentration. In forested, light agricultural, and moderate agricultural streams, the quantity and quality of allochthonous input were not significantly different. However, the timing and composition of allochthonous materials were related to land use. Chlorophyll *a* and periphyton biomass did not vary among land use types. Leaf breakdown rates were significantly faster in light and moderate agricultural streams in comparison to forested and heavy agricultural streams. Slow breakdown rates in forested streams resulted from low nutrient concentration and cool stream temperature. The scarcity of shredding macroinvertebrates and sedimentation probably limited leaf breakdown in heavy agricultural streams. Though limited riparian vegetation along agricultural streams resulted in an energy supply equivalent to forested streams, agricultural land use may still have long term impacts on stream structure including nutrient concentrations, temperature, macroinvertebrate community, and sedimentation thus affecting stream ecosystem function.

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## CHAPTER I

### **Influence of agricultural land use on southern Appalachian streams**

The southern Appalachian Mountains are located in West Virginia, southwestern Virginia, eastern Kentucky and Tennessee, western North Carolina, western South Carolina, northern Georgia, and northeastern Alabama (Yarnell 1998). This region is characterized primarily by forested land use at high elevations with agricultural land use concentrated in river valleys. The region experienced two major peaks in agricultural development, between 1400-1500 and between 1800-1860 (Gragson and Bolstad 2004). These peaks were followed by widespread abandonment of agricultural lands, between 1550-1660 and between 1945-1970. The percentage of agricultural land use in the southern Appalachians has decreased from 34% in 1969 to 25% in 1987 (SAMAB 1996). In 1992, 2.6% of the southern Appalachian population depended on agriculture for employment (SAMAB 1996) and that value is less than 2% today (Gragson and Bolstad 2004). At present, economic development, rapid population growth including a high number of recent retirees, highway networks connecting metropolitan areas, and growth in the retail and service sector has hastened the decline of agriculture (SAMAB 1996). Nevertheless, in 1992, over 1.1 billion ha (95%) of privately owned land in the southern Appalachian region was designated primarily as agriculture (SAMAB 1996). This agricultural land use has left a long lasting footprint on the landscape. In fact, research suggests that past agricultural land use predicts current aquatic macroinvertebrate and fish diversity better than present land use conditions (Harding et al.1998).

Conversion of land use from forest to agricultural land use often involves disturbance of the riparian zone, the interface between the stream and the surrounding land (Winterbourn 1986). In the southern Appalachians, agricultural land use is most often used for livestock grazing, which results in a reduced vegetation in the riparian zone, altering macroinvertebrate communities, temperature, nutrient concentrations, sedimentation, and channel morphology. These factors can result in changes in stream structure and function. For example, cattle grazing in riparian zones alter riparian vegetation, aquatic macroinvertebrate biodiversity, and nutrient cycling (Fleischner 1994). Cattle grazing in the riparian zone also can have substantial influence on stream ecosystems via direct degradation of stream and bank habitat, which indirectly affect aquatic insects (Strand and Merritt 1999). Benfield et al. (1977) attributed the absence of shredders in a pasture stream to the reduction of riparian vegetation quantity and diversity. Low



shredder abundance may contribute to slow leaf breakdown rates (Benfield and Webster 1985). While some studies show higher overall macroinvertebrate density in pastoral streams, the proportion of macroinvertebrates in the orders Ephemeroptera, Plecoptera, and Trichoptera tend to be low as are the number of shredding macroinvertebrates (Harding and Winterbourn 1995, Quinn et al. 1997, Townsend et al. 1997). Harding and Winterbourn (1995) found lower macroinvertebrate community richness and biomass in streams disturbed by pastoral land use in comparison to forested streams.

Conversion of forest to agriculture often results in a reduction in riparian vegetation, which causes reduced shading, less terrestrial organic matter entering the stream, and decreased bank stability. Reduced shading and subsequent increased light input indirectly result in higher primary production. Primary production in Appalachian streams is largely limited by light input; therefore, the shift from heterotrophy to autotrophy in streams mainly depends on the degree of shading (Minshall 1978, Vannote et al. 1980). This is particularly evident in agricultural development associated with small streams because the direct effect of shade decreases as channel width increases (Quinn 2000). The effects of riparian shading substantially decrease in streams with widths greater than 10 m due to natural gaps in the forest canopy over the stream channel (Davies-Colley and Quinn 1998).

Elevated light infiltration promotes primary productivity increasing food supplies to invertebrate grazers and filter-feeders (Lamberti et al. 1989). Additionally, macrophyte growth increases habitat availability for aquatic invertebrates (Biggs and Malthus 1982). However, removal of riparian vegetation drastically reduces the amount of allochthonous input to stream ecosystems limiting food availability to many aquatic organisms. The removal of riparian vegetation can be particularly disturbing to shredding macroinvertebrates that rely heavily on leaves and leaf fragments as their primary food source (Cummins et al. 1989). Dance and Hynes (1980) noted a shift from an allochthonous based systems in which detritivores and shredders dominated to an autochthonous system predominated by algae production and fewer shredders in streams disturbed by agriculture. The paucity of shredders was attributed to a lack of food availability. Shifting the primary energy source in streams from allochthonous to autochthonous production can have great effects on stream invertebrate assemblages, causing a decline in the number of shredders and an increase in scraper/grazer abundance (Gurtz and Wallace 1984).

Leaf litter is not only an important food source for macroinvertebrates but provides stream habitat and stability as well. Additions of leaves, as a form of habitat, to artificial stream channels along a pasture stream resulted in 70% higher invertebrate species richness and 140% higher invertebrate density than in streams with no leaves added (Quinn et al. 1999). In addition to providing in stream habitat, studies suggest that adult aquatic macroinvertebrates (e.g., Trichoptera and Plecoptera) inhabit riparian zones prior to laying eggs (Quinn 2000). Many species of aerial adult aquatic macroinvertebrates depend on forested riparian zones for shaded resting areas (Strand and Merritt 1999). Thus removal of riparian vegetation associated with agricultural land use may affect the fecundity and survival of adult aquatic macroinvertebrates.

A reduction in shading and subsequent insolation causes warmer temperatures in agricultural streams. Elevated temperatures often result in higher algal and macroinvertebrate production (Li et al. 1994). However, warm temperatures can also be detrimental to aquatic macroinvertebrates. Sweeney et al. (1986) found stream temperatures over 15°C to cause faster growth and development but increased mortality.

Riparian vegetation contributes to the interception and evapotranspiration of precipitation. Thus removal of woody riparian vegetation results in higher flows, flow variability, and surface runoff (Hewlett and Hibbert 1961). The reduction of large woody material within stream systems further alters stream morphology, structure, and retention (Bilby and Likens 1980, Davies-Colley 1997). Alterations to flow regime, channel morphology, and water quality have been shown to negatively impact macroinvertebrates (Winterbourn 1986). Furthermore, debris dams are an important site for organic matter accumulation thus providing a food source and habitat for shredding macroinvertebrates (Smock et al. 1989).

Nutrient concentrations, especially nitrogen and phosphorous, tend to be high in agricultural streams due to nitrogen fixing riparian vegetation, fertilizer application, and livestock wastes. Alder, a nitrogen fixer, often dominates riparian vegetation along agricultural streams (Torrey 1978). Crops of the legume family including peas and soybeans also have the ability to fix nitrogen. The presence of these plants in the riparian zone may contribute to high nitrogen fluxes to agricultural streams. The application of fertilizer to agricultural fields may contribute to phosphorous and nitrogen loading (Cooper 1993). For example, increased nitrogen flux to the Mississippi River, which is largely attributed to agricultural fertilizer, is causing eutrophication and seasonal hypoxia in the Gulf of Mexico (Alexander et al. 2000). Urine and

manure also can elevate stream nitrate and phosphorous concentrations (Lemly 1982). Livestock grazing in the riparian zone can contribute potentially toxic levels of ammonia to streams via urination directly into the stream and as runoff during precipitation (Quinn 2000). However, Del Rosario et al. (2002) found manure enrichment to be an important food source for macroinvertebrate gatherers, in particular chironomids. Despite manure providing a food source for chironomids, streams draining agricultural land use tend to have degraded food and water quality for most aquatic macroinvertebrates due to fertilizer and pesticide application and the contribution of livestock waste (Cooper 1993).

In addition to contributing to nutrient loading in agricultural streams, livestock grazing has been shown to negatively reduce riparian vegetation, bank stability, and stream habitat (Kauffman and Krueger 1984). Trampling of stream banks by livestock decreases bank stability and increases bank erosion leading to higher rates of sedimentation in agricultural streams. Riparian vegetation provides bank stability; therefore, by eliminating riparian vegetation, agricultural stream banks are more easily eroded resulting in higher rates of sedimentation. Sedimentation due to agricultural land use is the most damaging type of pollution in North American streams (Waters 1995). Sediment loads can be detrimental to benthic macroinvertebrate diversity and abundance. Not only will sediment impair macroinvertebrate respiration by coating gill structures (Strand and Merritt 1999), but the accumulation of fine sediment on substrate surfaces also reduces substrate heterogeneity (Lake et al. 2000). Elevated sediment loads can fill interstitial spaces, possibly burying aquatic organisms and reducing hyporheic habitat (Lenat et al. 1981). However, Kownacki (1983) did find elevated abundances of chironomids in livestock grazed streams suggesting that midges may be tolerant of siltation. Sediment loads also can cause high turbidity, potentially reducing light infiltration necessary for primary production. Sedimentation caused by cattle grazing can limit algal production reducing the quality of food for grazing macroinvertebrates (Davies-Colley et al. 1992).

The effects of agricultural land use on stream ecosystems are complex, affecting both stream structure and function (Fig. 1). To measure the influence of agricultural land use on stream structure and function in the southern Appalachians (North Carolina and Georgia), I studied streams ranging from forested (no agriculture present) to streams heavily impacted by agriculture (little to no riparian vegetation and grazing by cattle). Streams were placed into categories of agricultural land use (forested, light agriculture, moderate agriculture, and heavy

agriculture) based on tree density, tree basal area, canopy cover, light infiltration, and cattle grazing in the riparian zone (within 10-m of each stream bank). The objectives of this research were to assess the influence of agricultural land use on organic matter dynamics in southern Appalachian streams. Specifically, I was interested in how allochthonous input and leaf breakdown rates are altered by agricultural land use. In addition to quantifying annual allochthonous input and leaf breakdown rates, I measured several chemical, physical, and biological parameters including nutrient concentrations, temperature, discharge, macroinvertebrate community structure, chlorophyll *a*, and periphyton biomass.

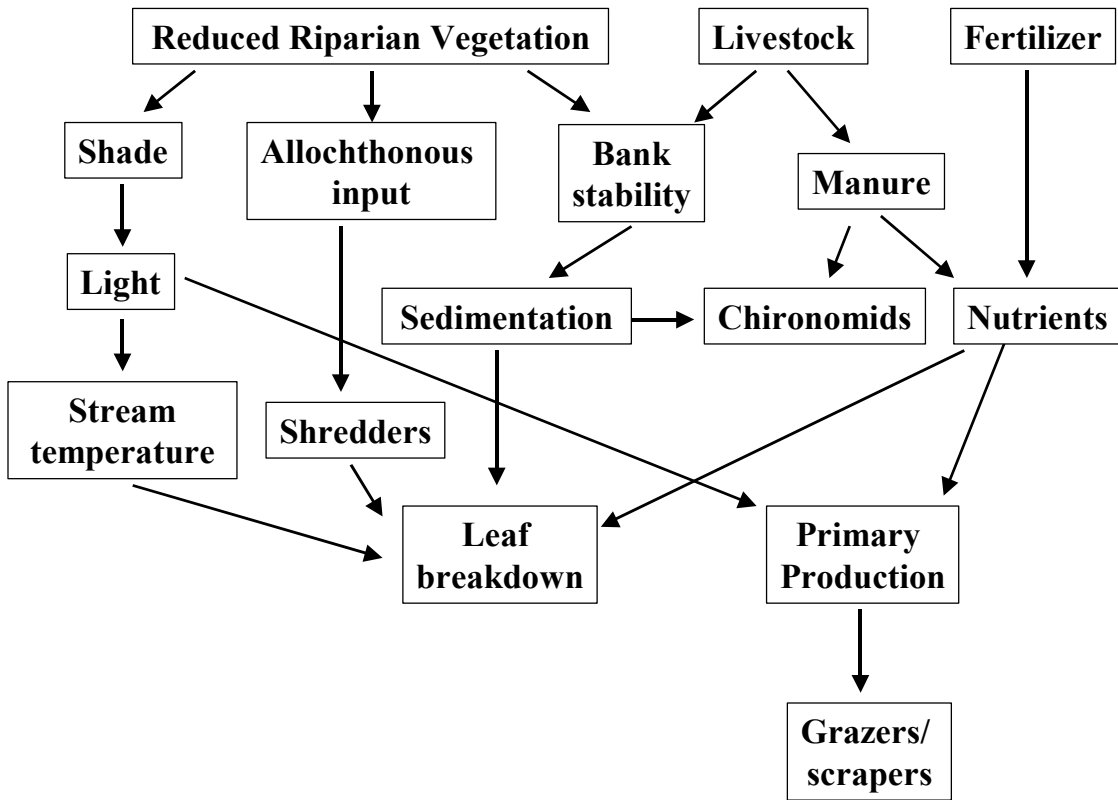


Fig. 1. Common effects of agricultural land use on stream ecosystems. Boxes represent structural and functional components of stream ecosystems that may be altered by agricultural land use.

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## CHAPTER II

### **Effects of agricultural land use on allochthonous input to southern Appalachian streams**

#### **Abstract**

The influence of agricultural land use on the timing, quantity, composition, and quality of allochthonous inputs to southern Appalachian streams was examined. Annual allochthonous input was measured in streams with different land use: forest, light agriculture, moderate agriculture, and heavy agriculture (September 2002-2003). Additionally, the density, size, and distribution of large, woody riparian vegetation were measured 3-m and 10-m from each stream bank to examine linkages between the riparian community and composition of the leaf litter entering the stream ecosystem. Allochthonous input was highest in November in forested streams, but peaked in September and November in agricultural streams. Annual allochthonous input ranged from 6 to 530 g AFDM/m<sup>2</sup>/y and was significantly lower in heavy agricultural streams than in other streams. While tree species composition varied along the land use gradient, the quality of leaf input was not related to land use type. Stream substrate chlorophyll *a* and periphyton biomass were not related to land use type. Along the land use gradient from forested to heavy agricultural streams, a shift from an allochthonous to an autochthonous based production was not evident.

## Introduction

Streams and terrestrial ecosystems are linked by allochthonous organic matter inputs from streamside vegetation, which provide the main energy source to forested stream communities (e.g., Cummins 1974, Vannote et al. 1980, Wallace et al. 1997). Heavy shading often limits primary production in forested streams; therefore, forested stream food webs tend to be highly dependant on allochthonous material to form their trophic base (Webster and Meyer 1997). Fisher and Likens (1973) found allochthonous inputs to make up more than 99% of the annual energy supply available to a small headwater stream in New Hampshire. Conners and Naiman (1984) found allochthonous material to comprise 81 to 95% of carbon supply to undisturbed Canadian streams. Allochthonous material entering streams from riparian vegetation includes leaves, leaf fragments, floral parts, wood, cones, nuts, and fruit (Benfield 1997). Leaf litter generally makes up the largest component of allochthonous organic material entering streams (69 to 80%) (Wallace et al. 1995). Autumn litter fall can make up to 79% of annual allochthonous input (Abelho and Graca 1998). In fact, the energy base in forested southern Appalachian streams is primarily composed of leaves that enter the stream during autumn leaf fall (Benfield 1997, Wallace et al. 1997). Once allochthonous material enters a stream, it may be broken down, retained, or transported downstream (Webster et al. 1999). The rates in which these processes occur depend on a number of variables including type of allochthonous input and stream characteristics.

Climatic variables, such as air temperature and precipitation, and physical attributes of the stream, including stream order, latitude, and slope, will influence allochthonous input (Benfield 1997, Abelho 2001). Additionally, the type of riparian vegetation largely controls the timing, quantity, and quality of allochthonous input received by streams (Webster and Meyer 1997). For example, Scarsbrook et al. (2001) found lower leaf litter inputs to pasture streams in comparison to streams draining forest or pine catchments. Agricultural land use often results in the reduction of large woody riparian vegetation, which may potentially lower allochthonous input to agricultural streams. The reduction of allochthonous input may strongly influence stream ecosystem structure and function. For example, the exclusion of allochthonous input to a small, forested, headwater stream in the southern Appalachians resulted in strong bottom up effects that influenced a wide variety of aquatic organisms (Wallace et al. 1997, Wallace et al. 1999).

Not only does agricultural land use reduce the supply of organic matter to aquatic organisms by reducing allochthonous input to streams (Campbell et al. 1992), but less stream shading and subsequent light infiltration may also cause higher rates of primary productivity, mainly by algae. Autotrophy becomes an important energy pathway in streams not heavily shaded by riparian vegetation (Mulholland et al. 2001). Furthermore, the shift from an allochthonous based system to an autochthonous based system can cause changes to macroinvertebrate community structure (Townsend et al. 1997, Delong and Brusven 1998). Macroinvertebrate communities tend to respond by having an increased dependence on autotrophic production (e.g., periphyton) (Feminella et al. 1989, Winterbourn 1990). Thus, the conversion from forested to agricultural land use results in a shift from streams dominated by shredders and detritivores to streams composed primarily by scrapers and grazers (Benfield et al. 1977, Dance and Hynes 1980).

Many of the effects of agricultural land use on stream ecosystem structure are well understood, such as altered stream temperature, nutrient concentration, sedimentation, macroinvertebrate community, and channel morphology (e.g., Strand and Merritt 1999, Lake et al. 2000, Quinn 2000). Yet, there have only been a few studies assessing the influences of agricultural land use on stream ecosystem function. McTammany (2004) measured whole-stream metabolism in streams recovering from agricultural land use in the southern Appalachians. A limited number of studies have assessed leaf breakdown rates in agricultural streams (Benfield et al. 1977, Bird and Kaushik 1992, Tuchman and King 1993, Huryn et al. 2002). However, very few studies have assessed the effects of agricultural land use on allochthonous inputs to streams or how a gradient of agricultural land use influences stream organic matter dynamics. The objectives of this study were to measure the timing, quantity, composition, and quality of allochthonous inputs to streams along a land use gradient from forest to agricultural land use. In addition, chlorophyll a and periphyton biomass were measured to assess the trophic base of stream food webs along the land use gradient.

## Methods

### *Site description*

This study was conducted in the southern Appalachians in Macon County, North Carolina and Rabun County, Georgia. The southern Appalachian region is characterized by forested land use within the mountains and agricultural land use within river valleys. Agricultural development in the southern Appalachians increased in importance through the 1960's. However, since the 1970's agricultural activity has declined by 31% due to human population growth, economic growth, urban development, a shift to the retail and service sector, and an expanding highway network connecting metropolitan areas, resulting in widespread abandonment of agricultural lands (SAMAB 1996, Sponseller et al. 2001). While less than 2% of the southern Appalachian population currently depends on agriculture for employment (Gragson and Bolstad 2004); agriculture is the primary designation of 95% of privately owned land (over 1.1 billion ha) (SAMAB 1996). In North Carolina alone, there are 41,000 small-scale farms (Lewis 2004). Due to steep hill slopes and high soil erosion rates, the land is not well suited for row crops. Therefore, the primary use of agricultural lands is pasture for livestock grazing.

Twelve experimental stream sites were located within the Little Tennessee River drainage system along a gradient of agricultural land use. Land use categories included: forest, light agriculture, moderate agriculture, and heavy agriculture. Forested streams were located in the Nantahala National Forest. Light agricultural streams were located in the Nantahala National Forest or on private land, and all moderate and heavy agricultural streams were located on private land. Streams were placed into land use categories based on riparian basal area and tree density at 3-m and 10-m from each stream bank, percentage canopy cover in the riparian zone, percentage stream canopy cover, percentage grass groundcover, and agricultural influence (e.g., livestock grazing in riparian zone).

Mean monthly air temperature was 13.8°C and annual precipitation was 176 cm in 2002 (Coweeta LTER 2004). Rainfall fluctuated between 121 to 232 cm/y from 1982 to 2002 (Coweeta LTER 2004). Frequent rainstorms occurred during the study, while base flows occurred in summer and early autumn (Webster et al. 1999). This study was conducted during a wet year in comparison to previous years (annual mean stream flow in the Little Tennessee River, Needmore, North Carolina, was 15.9 m<sup>3</sup>/s in 2000 and 15.3 m<sup>3</sup>/s in 2001) (Ragland et al.

2002, Ragland et al. 2003). Mean stream flow throughout the study's duration, September 2002-2003 was 35.0 m<sup>3</sup>/s (Ragland et al. 2004).

Stream orders ranged from 1<sup>st</sup> to 4<sup>th</sup> order, and stream discharge during base flow conditions ranged from 0.02 to 0.76 m<sup>3</sup>/s with no pattern among land use types. Nutrient concentrations generally increased along the gradient from forested to agricultural land use (Chapter III). Mean NO<sub>3</sub>-N ranged from 30.4 ppb in forested streams to 130.9 ppb in heavy agricultural streams. NH<sub>4</sub>-N and PO<sub>4</sub>-P were highest in moderate agricultural streams (10.3 and 3.3 ppb, respectively). Dissolved oxygen significantly decreased, and specific conductance and stream temperature significantly increased along the land use gradient from forest to agricultural streams.

#### ***Land use category determination***

At each site the riparian zone was defined as the 3-m and 10-m strip of land on both sides of the stream reach. An adapted point-center quarter method (Mitchell 2001) was used to measure the density, size, and distribution of large woody riparian vegetation (> 4-cm diameter at breast height (DBH), approximately 1.4 m). Four 100-m transects were established, two transects on each side of the stream bank. The two transects ran parallel to the stream 3-m and 10-m from each stream bank. Ten random transects were established perpendicular to the 3-m and 10-m transects. Everywhere two transects intersected, a point was established (Fig. 1). The 40 points at each stream were divided into quarters and the DBH and distance to the closest tree in each quarter were measured. Additionally, trees were identified to species. From these data, tree basal area and tree density were calculated for both the 3-m and 10-m riparian zone along each stream reach. Due to the low number of trees at the agricultural sites, Warde and Petranka's (1981) correction factor was used when calculating tree density in sample quarters with no trees. Also, tree species importance values were determined, which were the sum of relative frequency, relative density, and relative basal area. At each of the 40 points used to measured riparian vegetation, riparian canopy cover was measured using a concave spherical densiometer, Model C (Robert E. Lemmon, Forest Densiometers, Bartlesville, OK, USA). Also, grass groundcover was estimated within the 10-m riparian zone along each 100-m stream reach.

Stream canopy cover was measured along each 100-m stream reach. This was determined by replicate light measurements collected at each site between 1100 and 1500 hr on sunny days, summer 2003. One Onset HOBO light intensity data logger (ONSET Computer

Corp., Pocasset, MA, USA) recorded light intensity, every 10 seconds, as I walked along the 100-m stream reach, while a second data logger, set in a clearing, recorded light intensity every 10 seconds to serve as a baseline. Additionally, the light meters were placed together to correct for differences in light intensity measurements. The light measurements collected along the stream reach were subtracted from the measurements collected in the clearing to determine the amount of light blocked by the stream canopy cover. Values were converted to the percentage stream canopy cover and averaged for each stream reach.

Riparian basal area ranged from 0.0 m<sup>2</sup>/ha along Hoglot Branch to 47.2 m<sup>2</sup>/ha for Ball Creek along the 3-m riparian zone of each stream reach (Table 1). Basal area within the 3-m riparian zone also was low along North Fork Skeenah (0.6 m<sup>2</sup>/ha) and Payne Creek (0.8 m<sup>2</sup>/ha). Basal area along the 10-m riparian zone of each stream reach ranged from 0.0 m<sup>2</sup>/ha in Hoglot Branch to 39.2 m<sup>2</sup>/ha in Hugh White Creek. Riparian basal area was similar in the 3-m and 10-m riparian zone of Caler Fork, Hugh White Creek, Jones Creek, North Fork Skeenah Creek, Payne Creek, and Sutton Branch. Yet, riparian basal area was substantially less in the 10-m riparian zone of Dryman Branch, North Shope Fork, and Tessentee Creek than in the 3-m riparian zone. Tree density ranged from 0.0 trees/ha along Hoglot Branch to 1299.0 trees/ha along Hugh White Creek in the 3-m riparian zone (Table 1). Hoglot Branch and Hugh White Creek had the lowest and highest tree densities along the 10-m riparian zone, respectively. High tree density (> 1000 trees/ha) was measured in Ball Creek, Hugh White Creek, and Jones Creek. North Prong Little Ellijay, North Shope Fork, Sutton Branch, and Tessentee Creek had intermediate tree density along both the 3-m and 10-m riparian zone. Caler Fork, Hoglot Branch, North Fork Skeenah Creek, and Payne Creek had low tree density (< 100 trees/ha) along the 3-m and 10-m riparian zones.

Percentage stream canopy cover was highest along Ball creek (39%), Jones Creek (37%), and Sutton Branch (32%) (Table 2). Caler Fork, Hoglot Branch, Payne Creek, and North Fork Skeenah Creek had very low percentage stream canopy cover (1 to 8%) in comparison to other stream reaches. Percentage riparian canopy cover ranged from 0 to 86% among study streams with substantially less riparian canopy cover in Hoglot branch, North Fork Skeenah Creek, and Payne Creek (Table 2). Percentage grass groundcover was highest along Hoglot Branch and Payne Creek (Table 2). Grass groundcover was not present along the riparian zone in Ball Creek, Hugh White Creek, and Jones Creek. Grass groundcover also was low along North Prong

Little Ellijay Creek (6%). Agriculture was not present within catchments draining three stream reaches, Ball Creek, Hugh White Creek, and Jones Creek, which were located in the Nantahala National Forest (Table 2). Differing degrees of agricultural land use were present in the remaining stream watersheds.

Based on results of vegetation analysis, including riparian basal area and tree density, stream and riparian canopy cover, grass groundcover, and the influences of agriculture; streams were placed into land use categories (Table 3). Figure 3 shows the location of each stream reach and its respective land use category.



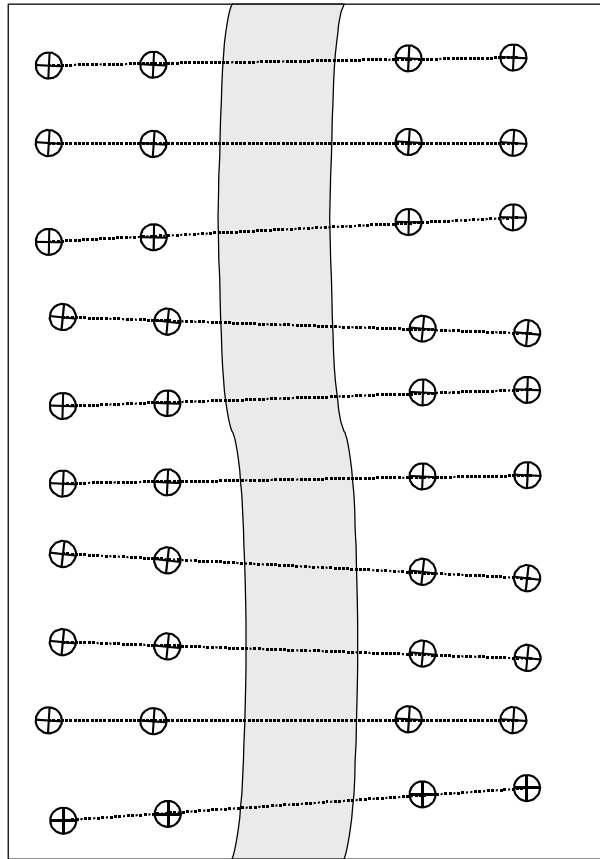


Fig.1. Point-center quarter method. Diagram represents a 100-m stream and approximate location of 40 points ( $\oplus$ ) used to sample riparian vegetation. Points were located 3-m and 10-m from each stream bank.

Table 1. Riparian basal area (m<sup>2</sup>/ha) and tree density (trees/ha) along study streams.  
Basal area and tree density were based on transects 3- and 10-m from each stream bank.

Stream	Stream abbreviation	3-m basal area (m <sup>2</sup> /ha)	10-m basal area (m <sup>2</sup> /ha)	3-m tree density (trees/ha)	10-m tree density (trees/ha)
Ball Creek	BAL	47.2	34.8	1251.9	1178.7
Caler Fork	CAL	4.6	0.3	31.8	62.5
Dryman Branch	DRY	32.1	1.9	459.7	70.4
Hoglot Branch	HOG	0.0	0.0	0.0	0.0
Hugh White Creek	HWC	31.8	39.2	1299.0	1620.2
Jones Creek	JON	33.6	36.6	1161.1	1554.1
North Fork Skeenah Creek	SKE	0.6	0.3	92.1	49.4
North Prong Little Ellijay	ELL	21.6	35.5	586.5	568.4
North Shope Fork	SHO	30.6	19.7	869.9	448.1
Payne Creek	PAY	0.8	0.4	13.7	6.8
Sutton Branch	SUT	10.7	6.1	692.6	77.2
Tessentee Creek	TES	34.9	15.4	771.9	367.3

Table 2. Study site characteristics of each stream reach. Stream abbreviations are given in Table 1.

Stream name	Stream canopy cover (%)	Riparian canopy cover (%)	Grass groundcover (%)	Agricultural influence
BAL	39	83	0	Agriculture was not present in watershed.
CAL	8	13	67	Pasture was directly adjacent to stream. Little to no vegetation was in the riparian zone. Livestock formerly had access to stream.
DRY	25	61	42	Pasture was separated from stream by vegetated riparian zone. Cattle did not have access to stream.
ELL	30	84	6	Agriculture was present in watershed, but not adjacent to stream.
HOG	1	0	95	Pasture was directly adjacent to stream. No vegetation was present in riparian zone. Livestock had direct access to stream.
HWC	33	86	0	Agriculture was not present in watershed.
JON	37	83	0	Agriculture was not present in watershed.
PAY	2	6	90	Pasture was directly adjacent to stream. Little to no vegetation was in the riparian zone. Cattle had direct access to stream.
SHO	26	80	31	Pasture was adjacent to stream, separated by patchy vegetated riparian zone. Livestock did not have access to the stream.
SKE	7	1	70	Pasture was directly adjacent to stream. Some vegetation was in riparian zone. Livestock were fenced from stream.
SUT	32	37	29	Pasture was adjacent to stream, separated by patchy vegetated riparian zone. Livestock did not have access to stream.
TES	27	61	17	Agriculture was separated from stream by vegetated riparian zone. Livestock were not present.

Table. 3. Land use category of each study stream.

Land use	Stream
Forest	Ball Creek
Forest	Hugh White Creek
Forest	Jones Creek
Light agriculture	Dryman Fork
Light agriculture	North Prong Little Ellijay Creek
Light agriculture	Tessentee Creek
Moderate agriculture	North Shope Fork
Moderate agriculture	Sutton Branch
Moderate agriculture	Caler Fork
Heavy agriculture	Hoglot Branch
Heavy agriculture	Payne Creek
Heavy agriculture	North Fork Skeenah Creek

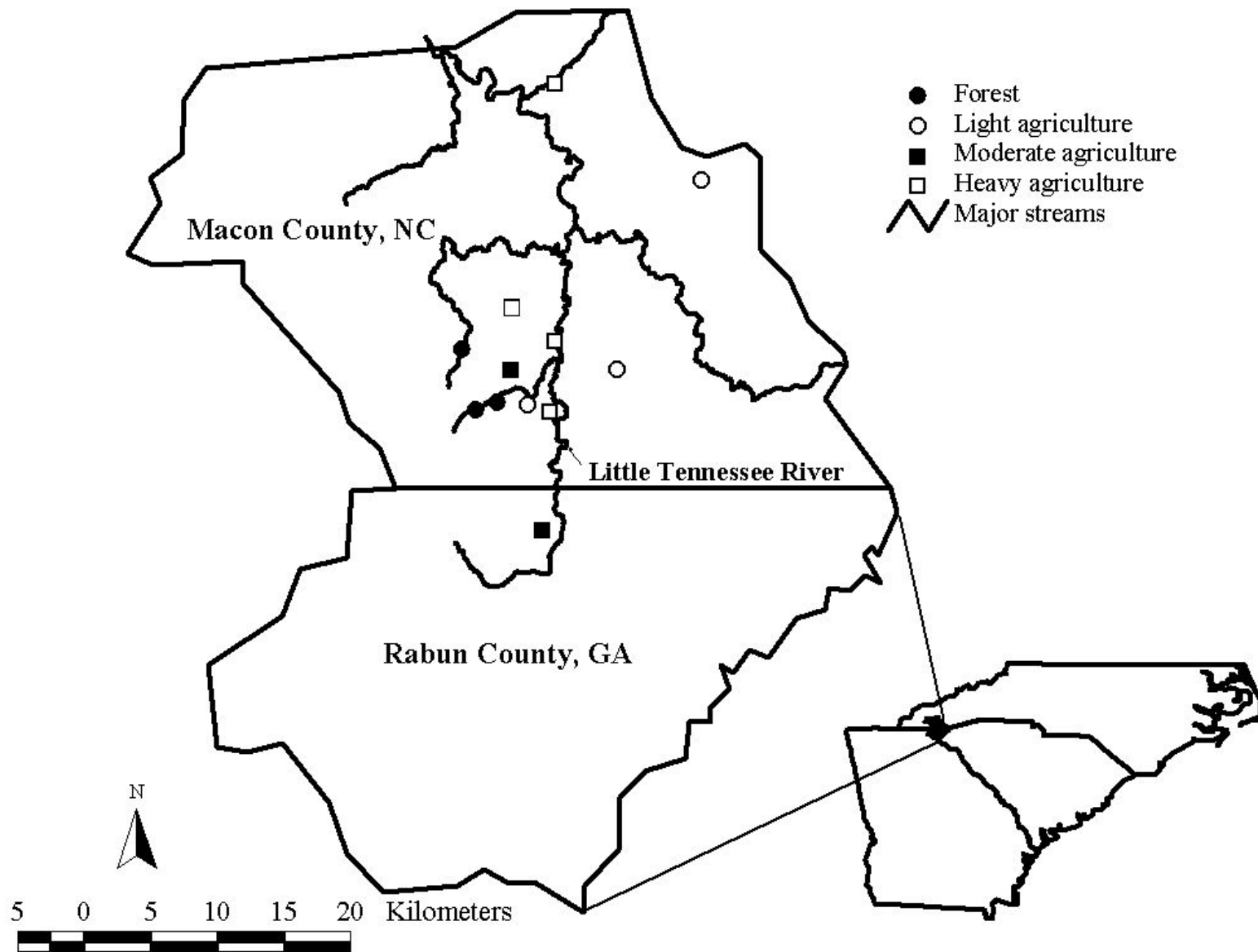


Fig. 2. Map of 12 study sites and associated land use within the Little Tennessee River basin.

### ***Allochthonous input***

Allochthonous input was measured September 2002 through September 2003 using litter traps. Litter traps consisted of 5-gal plastic buckets (surface area = 0.066 m<sup>2</sup>) anchored approximately 0.5 m above the stream to 1.5 m metal fence posts. Several holes were drilled into the bottom of the buckets to allow for water drainage and buckets were lined with aluminum screen with 2-mm openings. Ten litter traps were evenly dispersed along each 100-m stream reach. A total of 120 litter traps were used (12 sites, 10 litter traps per site). Allochthonous input was collected twice a month during heavy leaf fall (October and November) and monthly during the remainder of the study. While allochthonous material could be deposited laterally via wind, only direct litter fall was measured in this study. Lateral input was expected to range from 15-30% of total allochthonous input (Webster et al. 1990).

Litter was dried to a constant weight (50°C) and sorted into the following categories: leaves, wood, fruit and flowering parts, and grasses. Leaves were further divided into dominant tree species at each site. Samples were weighed, ground, ashed (550°C, 45 min), and reweighed to determine ash free dry mass (AFDM) for each category. Leaf quality was measured on recently senesced autumn leaves collected from litter traps. Leaf percentage carbon and percentage nitrogen were measured using a Vario MAX CNS Macro Elemental Analyzer (Elementar Americas, Inc., Mt. Laurel, NJ, USA). Carbon to nitrogen ratio was calculated as percentage carbon divided by percentage nitrogen. To compare leaf quality along the land use gradient, annual leaf input was multiplied by percentage nitrogen per tree species and summed for each stream. Percentage nitrogen rather than carbon to nitrogen ratio was used to assess leaf quality because leaf carbon percentage was approximately 50% regardless of tree species. Dissolved organic carbon (DOC) was measured seasonally by a wet persulfate digestion using an Oceanography International Corporation Model 700 Total Organic Carbon Analyzer. On each sampling date (December 2002 and May, June, and September 2003), 3 60-mL water samples were collected from each stream, filtered in the field (Whatman GF/F), stored in acid washed polyethylene bottles, and frozen until analysis.

### ***Chlorophyll *a* and periphyton biomass***

Benthic periphyton biomass and chlorophyll *a* were measured monthly, July through September 2003, using the AFDM method (Steinman and Lamberti 1996) and hot ethanol extraction (Sartory and Grobbelaar 1984), respectively. Three to 5 samples, consisting of 3 to 5

rocks each, were collected from each stream reach and stored on ice until return to the laboratory. Samples were then stored at 4°C until processed. Periphyton biomass, which included algal biomass, was removed from the rock surface with a wire brush, subsampled, and filtered onto two, ashed, 0.45 µm glass fiber filters (Whatman GF/F). One filter was dried at 50°C and ashed to determine periphyton AFDM. The second filter was frozen until chlorophyll *a* analysis could be completed. Additionally, the surface area of each rock was estimated using the “aluminum foil method” to express periphyton biomass and chlorophyll *a* measurements per unit area (Steinman and Lamberti 1996).

### ***Statistical analysis***

Differences in allochthonous input among land use types were assessed using a 1-way analysis of variance (ANOVA) followed by Fisher’s Least Significant Differences (LSD) multiple comparison test. Linear regression analysis was used to examine how riparian vegetation (riparian canopy cover, ground cover, tree basal area, tree density, light infiltrating the stream) and climatic factors (air temperature, precipitation) influenced leaf input dynamics. Differences in DOC among land use types were assessed using a repeated measure 2-way ANOVA with land use as a fixed variable and season as a repeated variable. Chlorophyll *a* and periphyton biomass were analyzed using a completely randomized split plot design with land use as a treatment, stream as the subject, and month as a repeated variable to test for differences among land use types. Linear regression analysis also was used to examine relationships between allochthonous input and algal growth. Results were considered significant if  $p < 0.05$ .

## **Results**

### ***Riparian vegetation***

Mean riparian tree basal area decreased significantly along the land use gradient from forested to heavy agriculture along both 3-m and 10-m riparian vegetation ( $p < 0.05$ ) (Figs. 3A, 3C). Mean forested riparian zone basal area was 34 m<sup>2</sup>/ha in the 3-m riparian zone and 36 m<sup>2</sup>/ha in the 10-m riparian zone. Mean basal area along light agricultural streams was similar in both 3-m (30 m<sup>2</sup>/ha) and 10-m (23 m<sup>2</sup>/ha) riparian zones. Mean basal area was higher in the 3-m riparian zone (21 m<sup>2</sup>/ha) than the 10-m riparian zone (15 m<sup>2</sup>/ha) of moderate agricultural streams (Figs. 3A, 3C). Yet, basal area varied substantially between the two moderate agricultural streams; North Shope Fork basal area was consistently higher than Sutton Branch along both the 3-m and 10-m riparian zones (Table 1). Riparian trees identified along the heavy agricultural

streams were primarily along the 3-m riparian zone (Table 1). Mean heavy agricultural riparian basal area was 2 m<sup>2</sup>/ha along the 3-m riparian zone and 1 m<sup>2</sup>/ha along the 10-m riparian zone.

Mean riparian tree density was substantially higher in forested riparian zones than in agricultural riparian zones ( $p < 0.01$ ) (Figs. 3B, 3D). Mean tree density was higher along the 10-m riparian zone (1300 tree/ha) than the 3-m riparian zone (1200 tree/ha) in the forested sites. Along light agricultural streams, mean tree density ranged from 300-800 tree/ha and was higher in the 3-m riparian zone. Mean 3-m tree density was 2-fold higher than 10-m tree density along moderate agricultural streams, coinciding with pasture land use generally being separated from the stream only by a narrow strip of riparian vegetation. Mean tree density was significantly lower in heavy agricultural riparian vegetation than other land use types at both 3-m and 10-m ( $p < 0.001$ ).

Riparian vegetation in the forested sites consisted predominantly of deciduous trees dominated by sweet birch (*Betula lenta* L.) and yellow-poplar (*Liriodendron tulipifera* L.). Rhododendron (*Rhododendron maximum* L.) and eastern hemlock (*Tsuga canadensis* (L.) Carr.) were also important riparian species along forested streams. Light agricultural riparian vegetation was primarily yellow-poplar (*L. tulipifera*) and yellow buckeye (*Aesculus octandra*). Red maple (*Acer rubrum* L.) and alder (*Alnus serrulata* (Ait. Willd.) were important species to moderate agricultural sites. Alder (*A. serrulata*), black walnut (*Juglans nigra* L.), sycamore (*Plantanus occidentalis* L.), and black cherry (*Prunus serotina* Ehrh.) had high importance values in heavy agricultural sites; however, importance values were inflated due to the limited number of trees and species at these sites. A complete species list and associated importance values for trees in the riparian zone of each study stream is in Appendix A. Additionally, the number of tree species within the riparian zones of forest and light agricultural streams were significantly higher than heavy agricultural streams based riparian vegetation analysis ( $p < 0.01$ ). Tree species richness did not differ in moderate and heavy agricultural streams.



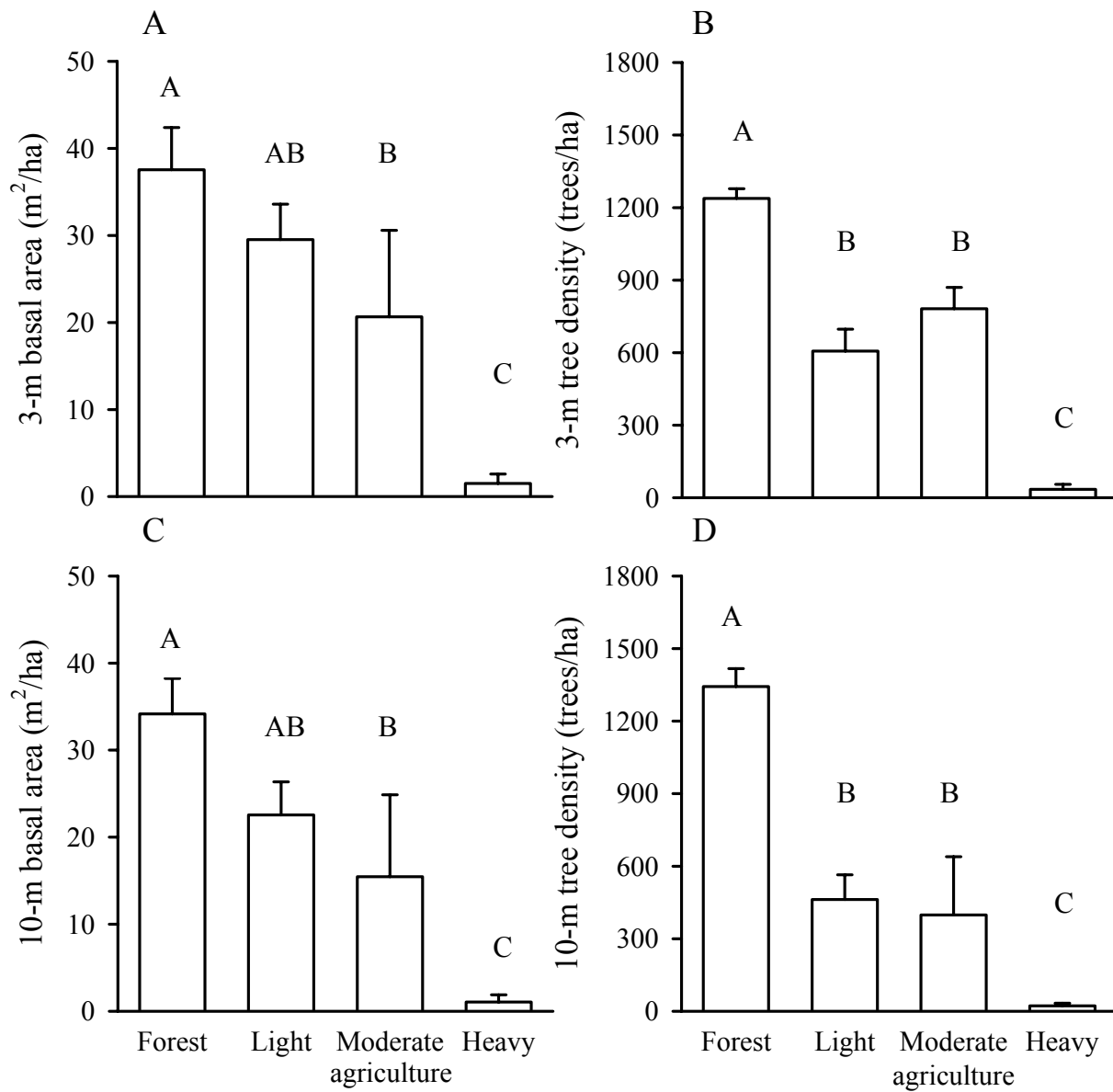


Fig. 3. Mean ( $\pm$  SE) riparian basal area (m<sup>2</sup>/ha) and tree basal area (number of trees/ha) in each land use type. Basal area and tree density were calculated for 3-m and 10-m riparian zones in each land use type. Values with the same letters were not significantly different. ANOVA followed by LSD,  $p < 0.05$ ).

### ***Allochthonous input***

The temporal distribution for allochthonous input to the streams throughout the year was typical for temperate deciduous forests in the southern Appalachians. The majority of input occurred in autumn, September through November (Figs. 4A, 4B), and this pattern was strongly driven by leaf litter input. One major peak in allochthonous input occurred in the forested streams in November consisting of 35% of annual leaf litter input. Eighty percent of annual leaf input occurred September through November 2002 in the forested streams. Two peaks in allochthonous input were observed in the agricultural streams (Figs. 4A, 4B). Light and moderate agricultural stream received 75% and 73%, respectively, of their annual leaf input September through November. Heavy agricultural streams received 52% of annual leaf input during autumn leaf fall (September through November 2002). A slight increase in allochthonous input also was observed in the forested and light agricultural streams in May 2003 (Fig. 4A). No trends were present among individual species of trees and time of abscission in agricultural streams. A peak in discharge in the Little Tennessee River was observed on 27 September 2003, which coincided with the first peak in allochthonous input to agricultural streams (Fig. 4C).

Annual allochthonous input ranged from 5.5 to 529.7 g AFDM/m<sup>2</sup>/y in Hoglot Branch and Dryman Fork, respectively (Fig. 5). Annual allochthonous input and leaf input were significantly higher in forested, light agricultural, and moderate agricultural streams in comparison to heavy agricultural streams ( $p < 0.05$ ) (Fig. 5, Table 4). Leaves comprised 51 to 87% of total input, with no pattern among land use types. Overall, leaf inputs were highest in light and moderate agricultural streams, 315 and 310 g AFDM/m<sup>2</sup>/y, respectively (Table 4). Wood input was highest to forested streams, while fruit and flowering parts contributed 5 to 18% of total allochthonous input with no trend among land use types. Grass input in moderate and heavy agricultural streams was 0.1 and 4.8 g AFDM/m<sup>2</sup>/y, respectively (Table 4). Grass input was significantly greater in heavy agricultural streams in comparison to other land uses. Sweet birch, yellow-poplar, and rhododendron made up the majority of leaf input to forested streams, 70, 39, and 28 g AFDM/m<sup>2</sup>/y, respectively (Table 5). Red maple (80 g AFDM/m<sup>2</sup>/y) and yellow-poplar (59 g AFDM/m<sup>2</sup>/y) made up the largest contribution to light agricultural streams. Red maple was the highest contributor of leaf input to moderate agricultural streams and all sites combined (95 g AFDM/m<sup>2</sup>/y). Also, alder input was important to moderate

agricultural streams (80 g AFDM/m<sup>2</sup>). Alder and sycamore leaves made up the majority of leaf fall to heavy agricultural streams, 30 and 28 g AFDM/m<sup>2</sup>/y, respectively.

Percentage carbon was approximately 50% regardless of tree species (Table 6). However, grass percentage carbon was substantially lower (25%). Percentage nitrogen ranged from 0.40 to 2.46% (Table 6). Leaf quality, measured as carbon to nitrogen ratio, varied from 20.4 to 127.7 in black willow and rhododendron leaves, respectively. Mean leaf litter input weighted by percentage nitrogen was not significantly different among land use types (Fig. 6). However, the variability among moderate and heavy agricultural streams was higher in comparison to variability among light agricultural and forested streams.

DOC ranged from 0.21 ppm in Hugh White Creek (Forest) in the spring to 3.28 ppm in Hoglot Branch (Heavy agriculture) in the summer. DOC concentration was not related to land use type or season ( $p < 0.05$ ) (Fig. 7). However, DOC concentration was generally higher in agricultural streams than in forested streams. High standard error to heavy agricultural streams during summer sampling is largely due to elevated DOC concentration in Hoglot Branch (mean = 2.63 ppm) in comparison to other heavy agricultural streams (0.49-0.90 ppm). Due to elevated discharge and high turbidity, only one light agricultural stream and one heavy agricultural stream were sampled in the spring resulting in no error bars (Fig. 7).

The positive relationship between annual leaf input per stream and percentage riparian canopy cover was significant ( $r^2 = 0.76$ ,  $p < 0.001$ ) (Fig. 8A). Annual leaf input was negatively related to percentage grass groundcover ( $r^2 = 0.76$ ,  $p < 0.001$ ) (Fig. 8B). Percentage stream canopy was positively related to annual leaf inputs ( $r^2 = 0.80$ ,  $p < 0.001$ ) (Fig. 8C). Annual allochthonous input and leaf input per stream versus 3-m basal area showed a stronger positive relationship than 10-m basal area, yet all relationships were significant (allochthonous input vs. 3-m basal area:  $r^2 = 0.65$ ,  $p < 0.01$  and leaf input vs. 3-m basal area:  $r^2 = 0.60$ ,  $p < 0.01$ ) (Fig. 9A). Tree density at 3-m significantly predicted annual leaf input ( $r^2=0.56$ ,  $p < 0.01$ ), but leaf input was not related to tree density 10-m (Fig. 9B). Minimum, mean, and maximum monthly air temperature, and precipitation were not significantly related to autumn leaf input.

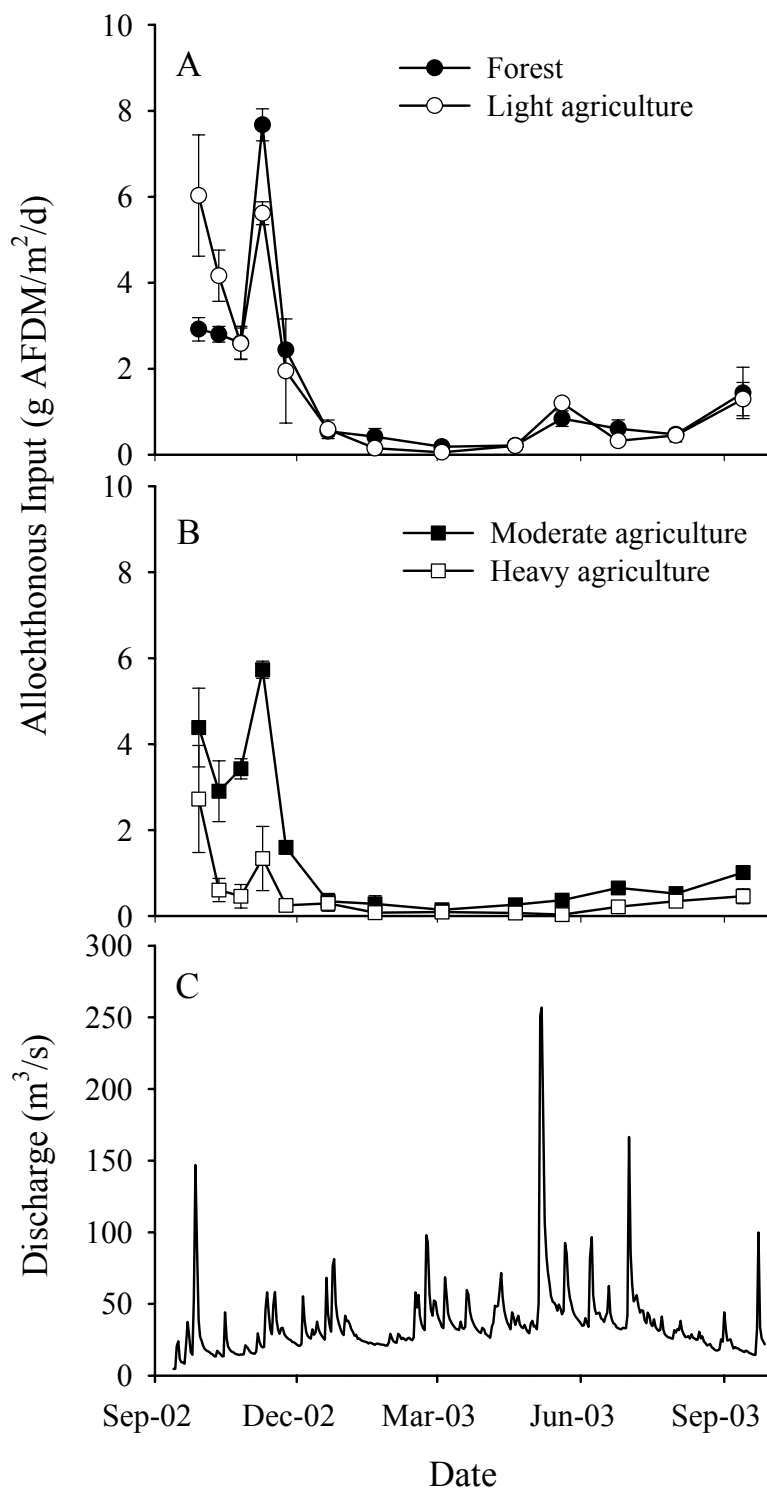


Fig. 4. Mean ( $\pm$  SE) allochthonous input (g AFDM/m<sup>2</sup>/d) to (A) forest and light agricultural streams and (B) moderate and heavy agricultural streams in the southern Appalachians (North Carolina and Georgia). (C) Mean daily discharge (m<sup>3</sup>/s) was measured in the Little Tennessee River, Needmore, North Carolina (Ragland et al 2003, Ragland et al. 2004).

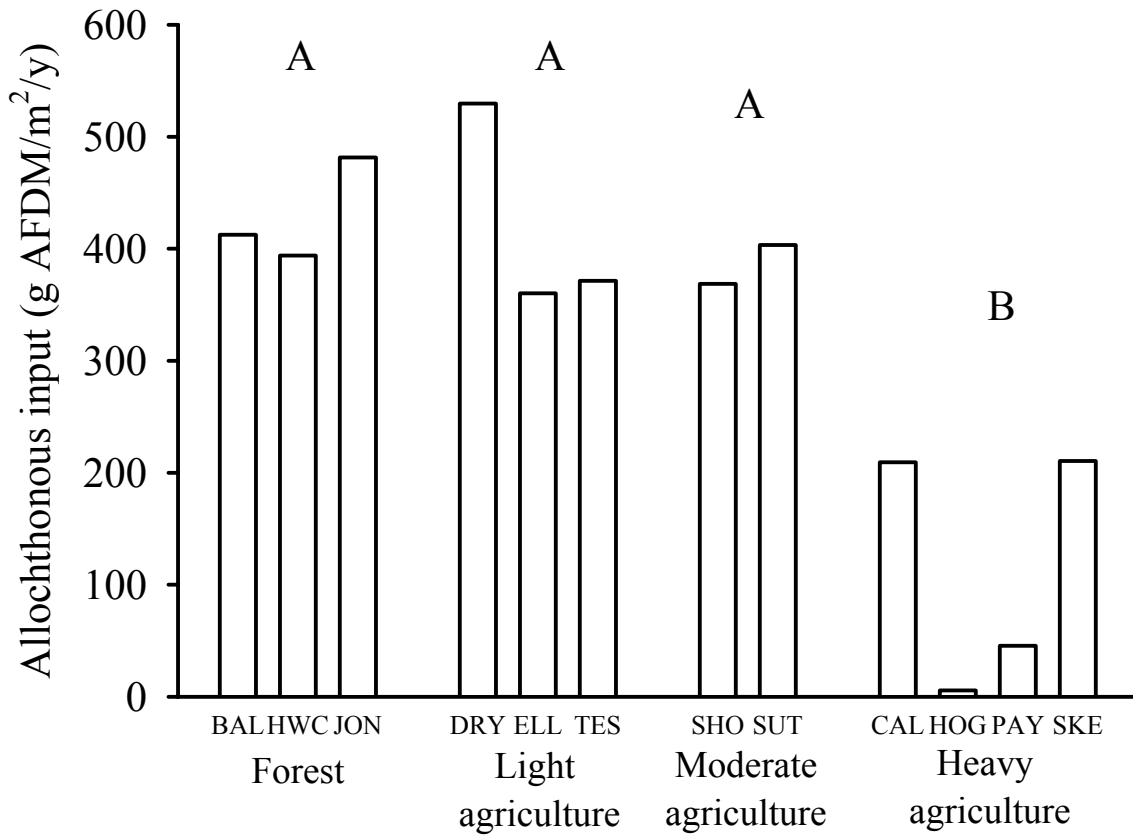


Fig. 5. Annual allochthonous input (g AFDM/m<sup>2</sup>/y) to 12 study streams. Values with the same letters are not significantly different among land use types (ANOVA followed by LSD,  $p < 0.05$ ). Stream abbreviations are given in Table 1.

Table 4. Annual mean ( $\pm$  SE) allochthonous input (g AFDM/m<sup>2</sup>/y) per land use type. Data were collected September 2002-2003. Values with the same letters were not significantly different among land use types (ANOVA followed by LSD,  $p < 0.05$ ).

Input	Forest	Light agriculture	Moderate agriculture	Heavy agriculture
Leaves	298 (6) A	315 (16) A	310 (6) A	95 (45) B
Wood	87 (32) A	55 (19) AB	34 (4) AB	10 (7) B
Fruit and flowering parts	44 (3) A	50 (19) A	41 (15) A	8 (4) B
Grass	0 (0) A	0 (0) A	0 (0) A	5 (2) B
Total allochthonous input	429 (27) A	420 (55) A	386 (17) A	118 (54) B

Table 5. Species composition of leaf input (g AFDM/m<sup>2</sup>/y) to each land use type collected September 2002-2003.  
 Values are means ( $\pm$  SE) for species totaling at least 2% of total leaf input to at least 1 study stream.

Tree species		Forest	Light agriculture	Moderate agriculture	Heavy agriculture
American basswood	<i>Tilia americana</i>	3.5 (0.5)	16.1 (15.8)		
American beech	<i>Fagus grandifolia</i>	8.0 (7.4)	7.5 (5.8)		
Bigleaf magnolia	<i>Magnolia macrophylla</i>	4.9 (2.3)	3.1 (3.1)		
Black cherry	<i>Prunus serotina</i>		6.9 (5.6)	1.8 (1.7)	5.9 (2.2)
Black walnut	<i>Juglans nigra</i>		17.7 (7.6)		17.6 (6.0)
Black willow	<i>Salix nigra</i>			9.0 (9.0)	8.0 (8.0)
Eastern hemlock	<i>Tsuga canadensis</i>	23.1 (15.5)			
Flowering dogwood	<i>Cornus florida</i>	0.4 (0.4)	8.8 (8.5)	0.5 (0.5)	
Alder	<i>Alnus serrulata</i>		6.8 (6.6)	80.0 (54.7)	29.5 (27.4)
Northern red oak	<i>Quercus rubra</i>	16.2 (7.6)	24.7 (23.9)	0.5 (0.5)	0.1 (0.1)
Pignut hickory	<i>Carya glabra</i>	0.6 (0.6)			
Red maple	<i>Acer rubrum</i>	15.7 (6.3)	80.6 (55.1)	94.8 (35.1)	
Rhododendron	<i>Rhododendron maximum</i>	28.4 (8.0)		3.8 (3.8)	
Sugar maple	<i>Acer saccharum</i>	9.4 (9.4)			
Sweet birch	<i>Betula lenta</i>	70.1 (7.9)	8.9 (6.3)	5.9 (5.9)	
Sycamore	<i>Platanus occidentalis</i>	12.4 (10.3)	12.3 (2.6)	15.9 (1.8)	27.9 (27.9)
White oak	<i>Quercus alba</i>	2.6 (1.7)	0.1 (0.1)		0.1 (0.1)
White pine	<i>Pinus strobus</i>	0.1 (0.1)		15.5 (15.5)	
Yellow buckeye	<i>Aesculus octandra</i>	5.2 (3.6)	16.4 (6.6)		
Yellow-poplar	<i>Liriodendron tulipifera</i>	39.4 (5.2)	58.8 (22.2)	15.9 (12.6)	
Miscellaneous leaves		53.5 (7.9)	41.8 (8.5)	58.7 (24.8)	5.3 (2.1)
Total Input		297.8 (97.2)	315.2 (182.9)	310.3 (173.8)	95.2 (75.5)

Table 6. Percentage carbon and nitrogen, and carbon to nitrogen ratios of recently senesced leaves collected fall 2002.

Tree species	Percentage carbon	Percentage nitrogen	C:N ratio
Black willow	50.31	2.46	20.43
Black walnut	49.22	2.20	22.35
Hazel alder	52.40	2.16	24.27
American basswood	48.94	1.82	26.86
Grass	25.43	0.88	28.84
Yellow-poplar	51.61	1.23	41.94
Sweet birch	52.67	1.20	43.85
Sycamore	50.77	1.15	44.02
Bigleaf magnolia	49.86	1.13	44.16
Flowering dogwood	50.68	1.04	48.71
Eastern hemlock	52.10	0.90	57.98
American beech	49.19	0.84	58.33
Northern red oak	51.67	0.85	60.47
Chestnut oak	51.30	0.81	63.40
Red maple	51.58	0.76	68.12
Sugar maple	47.18	0.60	78.68
White pine	53.31	0.58	91.83
Rhododendron	50.86	0.40	127.65



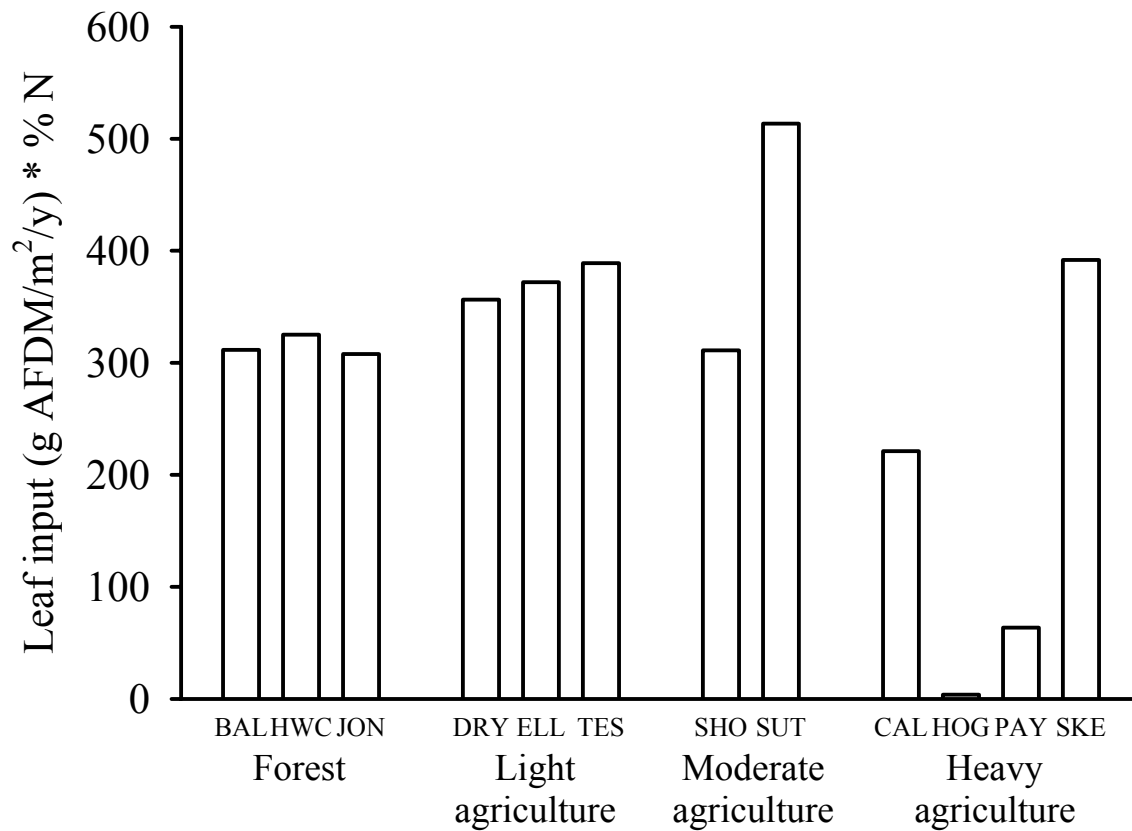


Fig. 6. Mean ( $\pm$  SE) leaf input weighted by percentage nitrogen to 12 study streams. Leaf input was multiplied by percentage nitrogen content per tree species and summed for each stream. If percentage nitrogen was not measured for a species of tree, then leaf input was multiplied by 1.30, the average percentage nitrogen content of deciduous leaves measured (Table 5). Leaf input weighted by percentage nitrogen was not significantly different among land use types. Stream abbreviations are given in Table 1.

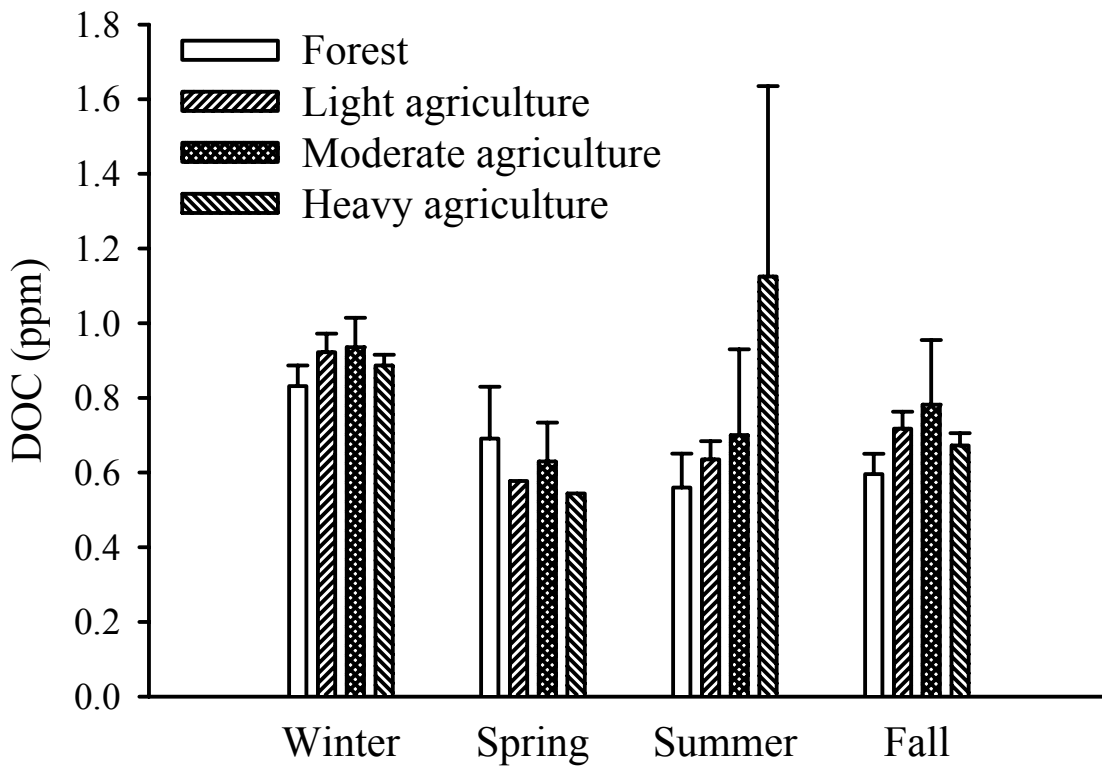


Fig. 7. Mean ( $\pm$  SE) dissolved organic carbon concentrations per season and land use.

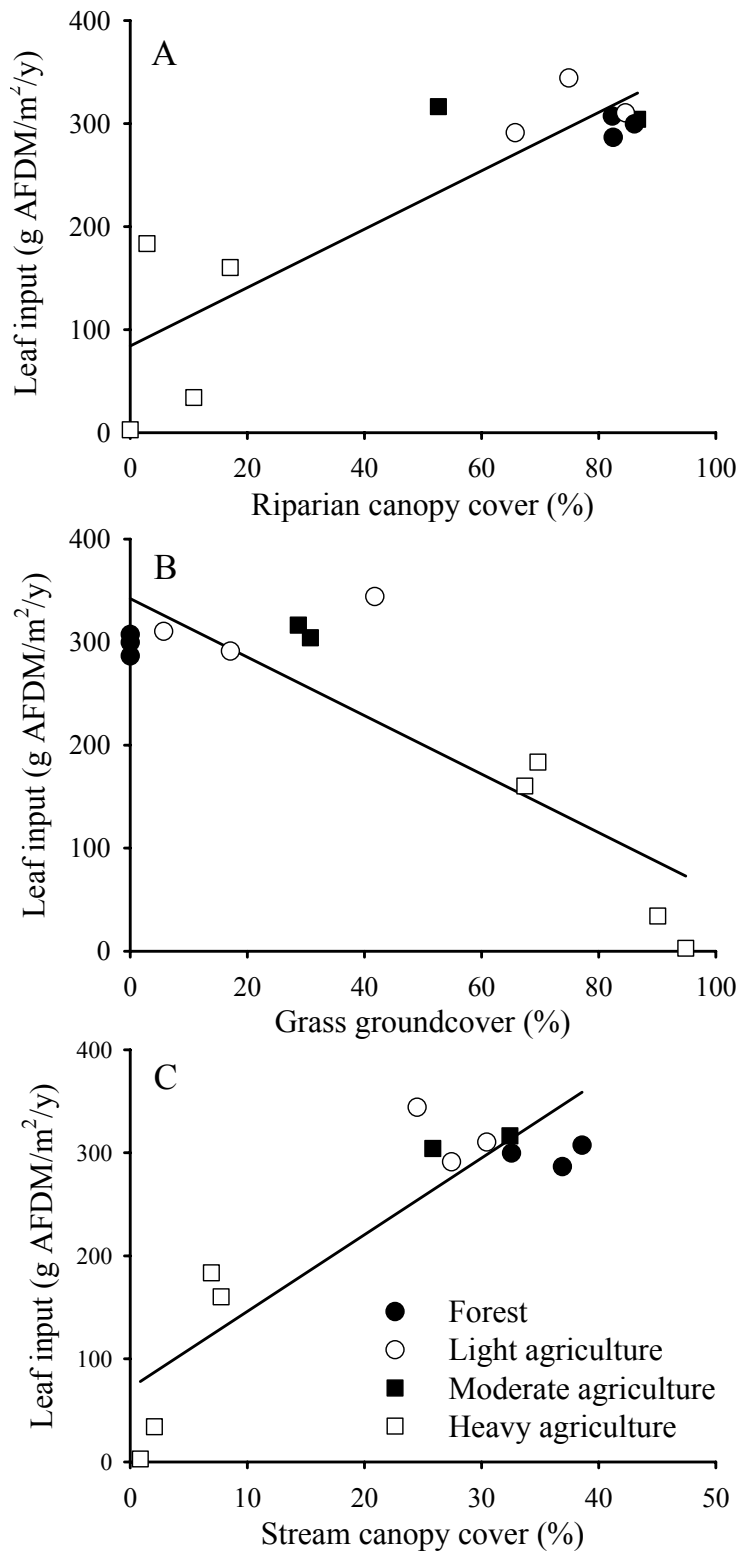


Fig. 8. Relationships between annual leaf inputs (g AFDM/m<sup>2</sup>/y) and riparian variables, including (A) percentage riparian canopy cover along the 3-m riparian strip and (B) percentage grass groundcover and (C) percentage stream canopy cover along the 10-m riparian strip.

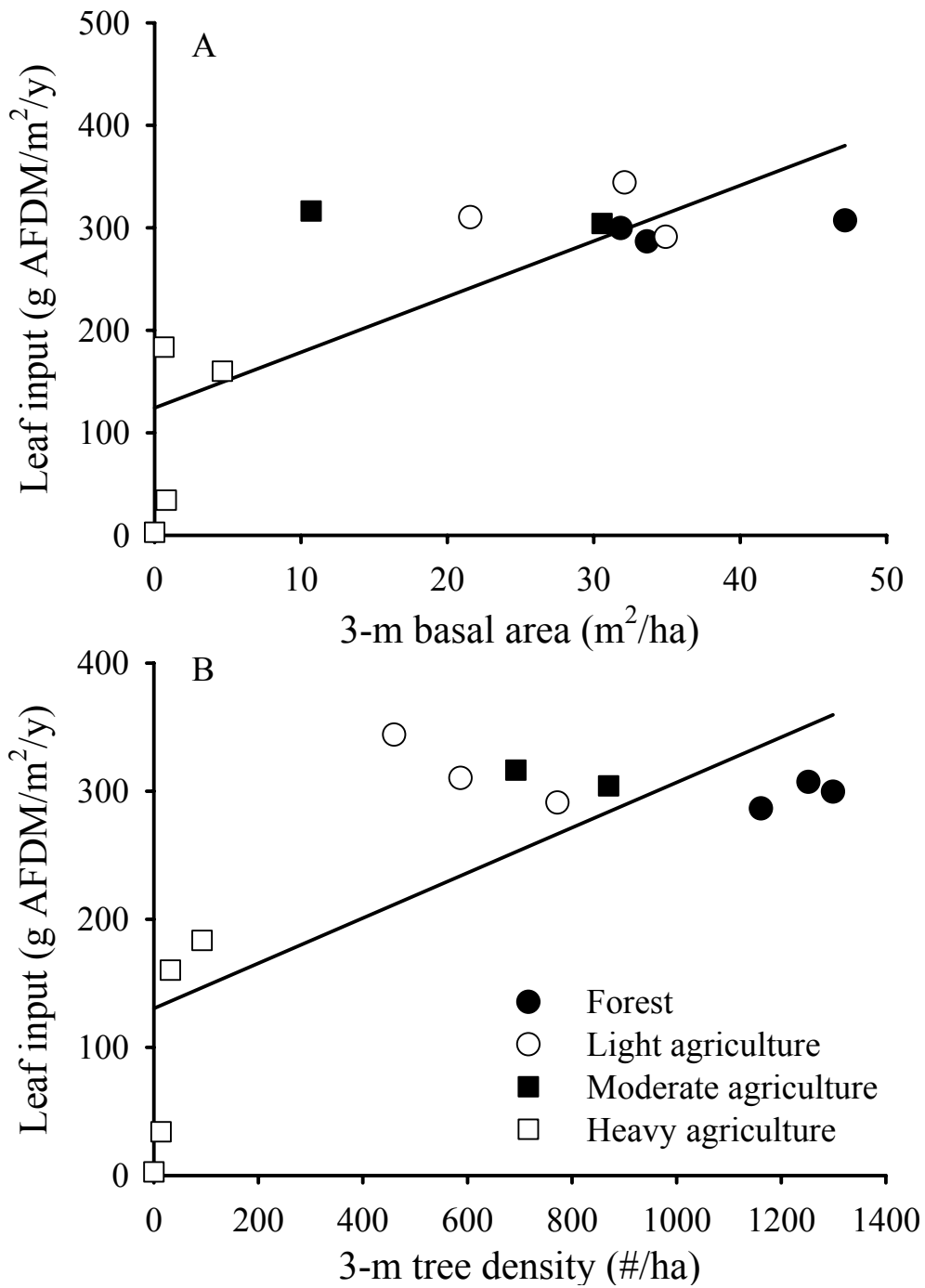


Fig. 9. Relationships between leaf input (g AFDM/m<sup>2</sup>/y) and (A) 3-m basal area and (B) 3-m tree density.

### ***Chlorophyll *a* and periphyton biomass***

Chlorophyll *a* values ranged from 0.2 mg/m<sup>2</sup> in Sutton Branch (Moderate agriculture) in September to 37.4 mg/m<sup>2</sup> Hugh White Creek (Forested) in July. Overall, chlorophyll *a* concentrations did not vary significantly among land use or sampling date ( $p > 0.05$ ) (Fig. 10A). However, within land use types, July and September chlorophyll *a* concentrations were 2.7 to 3.7 times higher than August measurements in forested streams. July and September chlorophyll  $\alpha$  concentrations were 0.9 to 1.3 times higher than August concentrations in light agricultural streams. August chlorophyll *a* concentrations were consistently lower than July and September concentration in moderate and heavy agricultural streams. Periphyton biomass ranged from 0.5 g AFDM/m<sup>2</sup> in Payne Creek in August to 25.7 g AFDM/m<sup>2</sup> Sutton Branch in September. Periphyton biomass did not vary significantly by land use or sampling date ( $p > 0.05$ ). Overall periphyton biomass was elevated in forested and light agricultural streams during the July sampling date and in moderate agricultural streams in September (Fig. 10B). Both total allochthonous input and leaf input were not significantly related to chlorophyll *a* concentration or periphyton biomass.

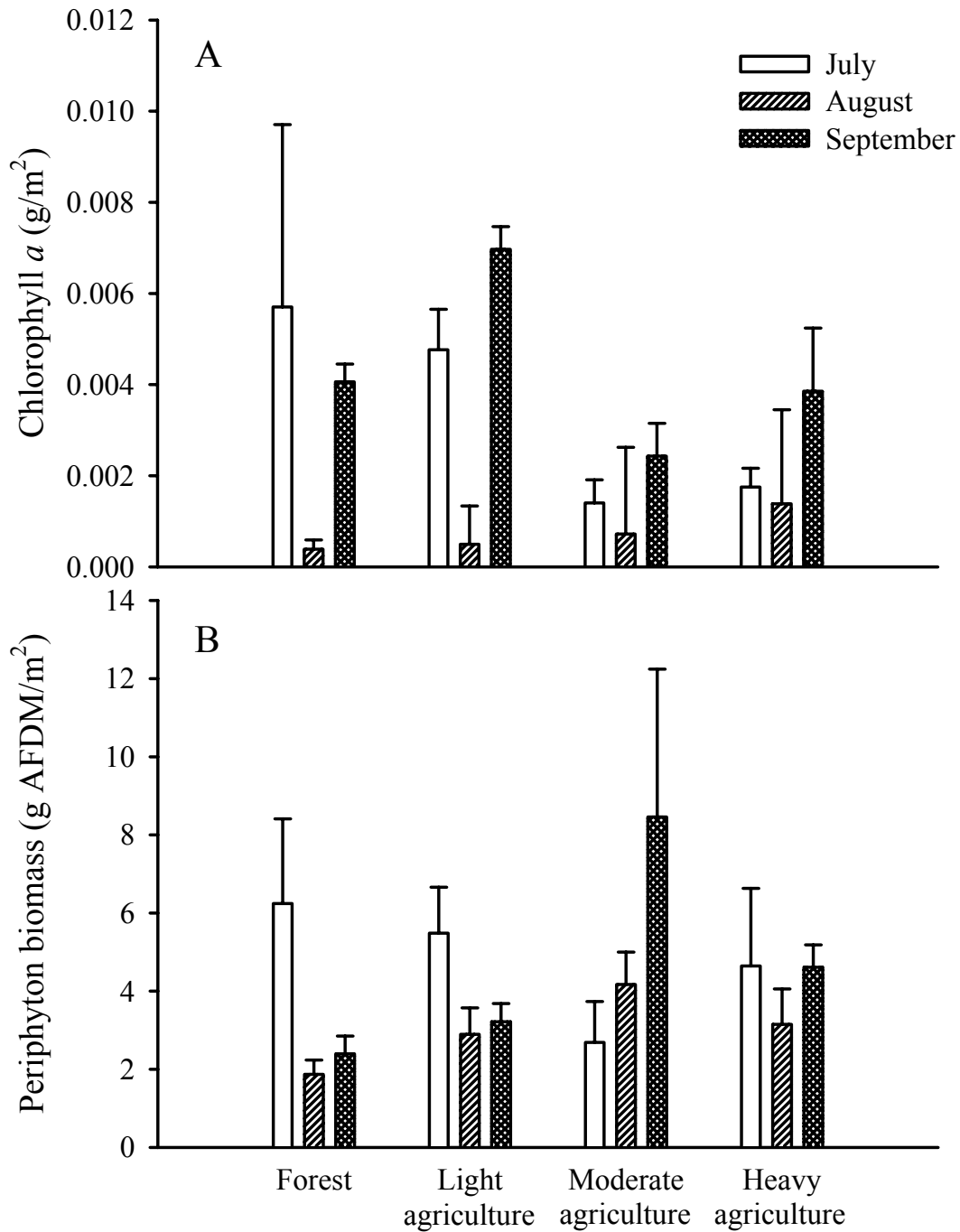


Fig. 10. Mean ( $\pm$  SE) (A) chlorophyll *a* concentration (g/m<sup>2</sup>) and (B) periphyton biomass (g AFDM/m<sup>2</sup>) collected during summer 2003 base flow conditions. The comparison of land use and month showed no significant difference in chlorophyll *a* concentration or periphyton biomass ( $p = 0.46$  and  $0.07$ , respectively).

## Discussion

Allochthonous input was expected to decline along the land use gradient from forested to heavy agricultural streams. Vegetation type within the riparian zone is a strong determinant of allochthonous input into streams (Benfield 1997). Scarsbrook et al. (2001) found leaf litter input dynamics to vary significantly with land use (i.e., native forest, pine, or pasture) but more specifically with amount and type of riparian vegetation. Yet, annual allochthonous and leaf input in light and moderate agricultural streams was equivalent to forested streams despite lower riparian basal area and tree density along adjacent riparian habitat. Annual allochthonous input to forested, light agricultural, and moderate agricultural streams were similar to previous studies in eastern deciduous forests. Wallace et al. (1999) measured 338 to 387 g AFDM/m<sup>2</sup> of annual leaf input to 2 streams at Coweeta Hydrological Laboratory, North Carolina. Additional litter fall studies conducted at Coweeta Hydrologic Laboratory measured litter fall between 295 to 482 g/m<sup>2</sup>/y in streams draining mature forests (Webster et al. 1995). Allochthonous inputs to heavy agricultural streams (5 to 211 g AFDM/m<sup>2</sup>/y) were similar to values found in arid regions, 17 g/m<sup>2</sup>/y (Schade and Fisher 1997), high latitude streams, 0 to 62 g/m<sup>2</sup>/y (Benfield 1997), and pasture New Zealand streams, 112 g DM/m<sup>2</sup>/y (DM = dry mass) (Scarsbrook et al. 2001).

Most likely, light and moderate agricultural streams received an equivalent supply of allochthonous material because individual trees contributed larger quantities of leaf input in comparison to individual trees along the forested streams. Isolated trees tend to grow larger and produce more leaves due to lack of competition for resources (light, nutrients, water, space) with surrounding trees. While individual trees may contribute less to annual allochthonous input in forested streams, the greater overall tree basal area and density compensated for lower leaf inputs per individual tree. Despite similar quantities of leaf litter inputs to forest, light agricultural, and moderate agricultural streams, species composition differed along the land use gradient. Thus suggesting that leaf quality may be related to land use type. Leaf quality, measured as C:N ratio, was high in black willow, black walnut, and alder (C:N ratios were 20.4, 22.4, and 24.3, respectively), species that were only present in riparian zones along agricultural streams. Yet, leaf input weighted by leaf quality was not significantly related to land use suggesting that agricultural streams received a range of leaf quality similar to forested streams.

Percentage N and C:N values on senesced leaves were somewhat higher than previous studies for certain species. For example, Royer and Minshall (2001), measured 0.92% N on

birch leaves and 0.65% N on dogwood leaves. This may be because leaves used in C:N analysis were not collected directly from trees. Rather, leaves were collected from litter traps and so leaching may have occurred due to rain. Any leaching of soluble compounds, particularly nitrogen, would result in a higher C:N ratio and lower leaf quality measurements, but is probably a better indicator of leaf quality available to stream macroinvertebrates.

Leaf quality or C:N ratio influences how fast leaves breakdown in streams thus becoming an available energy source to stream organisms. Leaves with high nitrogen contents break down more rapidly in stream ecosystems (Kaushik and Hynes 1968) due to rates of microbial colonization and macroinvertebrate fragmentation. Shredding macroinvertebrates depend on conditioned leaves as a food source, which will change throughout the year as leaves with different nitrogen contents become colonized by microbial organisms (Cummins and Klug 1979, Cummins et al. 1989). However, agriculture often results in riparian vegetation composed of species that have high leaf quality and faster breakdown rates (e.g., alder, black walnut, black willow), thereby limiting the availability of leaves to stream organisms throughout the year disrupting the leaf processing continuum (Petersen and Cummins 1974). Yet, in the current study, agricultural streams and forested streams received similar leaf inputs when weighted by leaf quality, even though tree density, basal area, and number of species were significantly higher in forested streams than in heavy agricultural streams.

The timing of allochthonous inputs were similar to previous litter fall studies, which tend to show an increase in leaf input September through November coinciding with autumn leaf fall (e.g., Wallace et al. 1995). Gosz et al. (1972) and McDowell and Fisher (1976) measured peak leaf input in mid October. However, the bimodal peak in allochthonous input in agricultural streams has not been documented in previous studies. The timing of autumn leaf fall has been associated with air temperature and rainfall (Scarsbrook et al. 2001). The first date during autumn leaf fall that ambient air temperature was below freezing at Coweeta Hydrological Laboratory was 18 October 2002 (Coweeta LTER 2004). Yet, the first freezing air temperature was not associated with either peak in litter fall. Additionally, monthly air temperature and precipitation were not significantly related to autumn leaf fall. The first peak in agricultural streams in September, however, coincided with elevated discharge resulting from torrential rains caused by Hurricane Isidore as it moved through the southeastern United States (Fig. 4) (Avilia 2003). Most likely the first peak in allochthonous input was only observed in agricultural



streams because riparian vegetation along agricultural streams were unprotected from heavy winds associated with Hurricane Isidore, which caused trees to lose leaves prematurely. Forest streams were part of an entirely forested catchment; therefore, high winds did not affect riparian vegetation along the forested streams to the extent that occurred in agricultural streams. Aside from the pulse in input in September and November, allochthonous input was generally low throughout the remainder of the study regardless of land use. A small pulse in allochthonous input in forested and light agricultural streams occurred in May, which consisted primarily of fruit and flowering parts. Peterson and Rolfe (1982) also measured an increase in spring litterfall to forested ecosystems due to reproductive and flowering parts.

Leaves comprised the largest proportion of allochthonous input in all streams (51 to 87% of total input) with no pattern among land use types. These findings are similar to Wallace et al. (1995) who found leaf litter to make up 69 to 80% of total input to Appalachian Mountain streams. Annual leaf input was only substantially less to heavy agricultural streams. The overall amount of wood input tended to decrease along the gradient of land use from forested to agricultural streams, but the percentage of wood input in comparison to total allochthonous input was not significantly different among land use types. Grass inputs only occurred in moderate and heavy agricultural streams. Grass input comprised 32% of annual allochthonous input to Hoglot Branch, which had no large woody vegetation in its riparian zone. Grass inputs may become an important source of organic material in agricultural pasture because other sources of allochthonous input are limited. However, grass is not very palatable to aquatic macroinvertebrates (Niyogi et al. 2003). Grass was a low quality food source due to its low nitrogen content (0.9% N).

While not significant, across season, DOC concentration tended to be higher in agricultural streams than in forested streams. However, very little is known about the bioavailability and fate of DOC in agricultural streams (Royer 2004). Meyer and Tate (1983) measured high DOC in a forested stream in comparison to a recently clear-cut stream, which they attributed to undisturbed riparian vegetation. Hill et al. (2002) measured higher DOC concentration in forested streams in comparison to open canopy streams.

Overall, chlorophyll *a* concentrations were highest in forested and light agricultural streams in July and September 2003. Sponseller et al. (2001) measured 7.0-121.1 mg/m<sup>2</sup> chlorophyll *a* concentrations along a range of land use in the southern Appalachians. While in

the current study, overall chlorophyll *a* concentrations were lower than reported by Sponseller et al. (2001), forest measurements were similar to previous studies. Summer chlorophyll *a* concentrations were approximately 4.5 mg/m<sup>2</sup> in Hubbard Brook Experimental Forest, New Hampshire (Bernhardt and Likens 2004). Despite high levels of stream shading, Scarsbrook and Halliday (1999) measured high (1.1-18.0 mg/m<sup>2</sup>) levels of chlorophyll *a* in 2 New Zealand forested streams. In addition, infrequent algal blooms have occurred in the Hubbard Brook Experimental Forest regardless of heavy shading (Bernhardt and Likens 2004).

Periphyton biomass measurements were highest in forested and light agricultural streams in July and in moderate agricultural streams in September. Biomass measurements from the forested streams were similar to previous studies. Findlay et al. (2002) measured epilithon biomass of 1.3 g AFDM/m<sup>2</sup> in Ball Creek in October 1996. Periphyton biomass in Walker Branch, a forested headwater stream, was 3.8 g AFDM/m<sup>2</sup> (Mulholland et al. 2000). In the current study, periphyton biomass values were closer to values reported by Mulholland et al. (2000). Agricultural streams were expected to have similar biomass estimates to grassland and desert streams due to similar amounts of light infiltration; however, biomass estimates from agricultural streams more similar to forested streams. Findlay et al. (2002) estimated epilithon biomass of 76.0 g AFDM/m<sup>2</sup> in a tall grass prairie stream and 18.3 g AFDM/m<sup>2</sup> in a desert stream.

In agricultural streams that are not heavily shaded by riparian vegetation, autotrophy is expected to become an important energy pathway (Wiley et al. 1990). Scarsbrook and Halliday (1999) measured higher epilithon biomass in a pasture stream than in forested streams. However, this trend did not occur in the current study. Chlorophyll *a* concentrations and periphyton biomass measurements did not vary significantly among land use types. This may be the result of nitrogen limitation across study streams. Yet, NO<sub>3</sub>-N concentrations were 4 times higher in heavy agricultural streams in comparison to forested stream (Chapter III). Stream temperature significantly increased and stream shading significantly decreased along the land use gradient (Chapter III). Therefore, despite elevated nutrient concentrations, warmer temperatures, and less stream shading; agricultural stream algal biomass was similar to forested streams most likely due to other factors related to agricultural land use, such as turbidity, scouring of the streambed during high discharge, and presence of macroinvertebrate grazers (Townsend et al. 1997, Townsend and Riley 1999). Young and Huryn (1996) measured depressed periphyton

biomass and chlorophyll *a* concentrations in agricultural streams, despite higher nutrient concentrations and warmer temperatures, which they attributed to elevated sedimentation and turbidity. High turbidity caused by sedimentation would potentially block light infiltration thus limiting primary productivity in agricultural streams. Findlay et al. (1993) and Rosemond (1993) suggested that periphyton growth might be more limited by light or grazers than nutrients. Scrapers and grazers have been shown to limit periphyton growth (Feminella et al. 1989). In the current study, scraper abundance and density were not related to chlorophyll *a* or periphyton biomass; however, scraper abundance and density were generally higher in agricultural streams (Chapter III). Moreover, several heavy rainstorms occurred summer 2003 (Fig. 4C); therefore, high flows may have scoured the streambeds, limiting algal growth.

Additionally, the technique used to measure chlorophyll *a* and periphyton biomass may not be sufficient to measure algal biomass in agricultural streams. The technique was well suited for measuring periphyton biomass on cobble size substrate, typical in forested streams. However, the majority of substrate in heavy agricultural streams was silt and sand. Therefore, while higher algal biomass may have been present in agricultural streams, it was unable to be quantified. Also, the technique used to measure periphyton biomass does not distinguish among algal, bacterial, or fungal biomass growing on rock surfaces. Bacteria and fungi biomass may comprise a larger component of periphyton biomass in forested streams, while algae was expected to represent a larger proportion of periphyton biomass in agricultural streams. Mulholland et al. (2001) found that algae made up 3 to 8% of total autotrophic biomass in 2 forested streams, while algae made up 98 and 100% of autotrophic biomass in a tall grass prairie and desert stream, respectively.

### ***Conclusions***

Our results showed that the quantity and quality of allochthonous inputs did not vary in forest, light agricultural, and moderate agricultural streams, despite decreasing riparian basal area and tree density along the land use gradient. This study suggests that as long as some large woody vegetation is present within the riparian zone, the quantity of allochthonous inputs will be similar to forested streams. Riparian variables at 3-m were the best predictor of leaf litter input, which was largely due to close proximity to the stream reaches. While tree species composition changed along the land use gradient, the quality of leaf input was not significantly different along the land use gradient. Also, an increase in algal biomass in agricultural streams was not

observed. Rather, chlorophyll *a* was generally higher in forested and light agricultural streams despite high amounts of shading over the stream. Along the land use gradient, a major shift in energy resources from forested stream food webs depending primarily on allochthonous organic material and agricultural stream food webs depending mainly on autochthonous carbon sources was not evident. Rather, light and moderate agricultural streams received allochthonous input equivalent to forested streams. Despite high light infiltration and nutrient concentrations; algal growth was probably limited in agricultural streams because of high sedimentation, scraping macroinvertebrates, and scouring of the streambed due to elevated discharge. Yet, we were only able to access algal biomass, primary productivity may have been higher in agricultural streams than forested streams.

Studies have shown that improved management of riparian zones is one the most effective ways to protect stream ecosystems from the effects of pastoral land use (Quinn et al. 1993, Collier et al. 1995). This study suggests that in order to restore natural food web dynamics in heavy agricultural streams, at least a partially intact riparian zone is necessary. Although the quality and quantity of allochthonous input may be equivalent to undisturbed forested streams, other negative effects of agriculture (warmer temperatures, high nutrient concentrations, high sedimentation, and altered macroinvertebrate communities) may still cause long term impairments to stream structure and function.

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## CHAPTER III

### **Influence of agricultural land use on leaf breakdown rates in southern Appalachian streams**

#### **Abstract**

The effects of agriculture on red maple (*Acer rubrum* L.) leaf breakdown rates were measured in southern Appalachian streams with differing land use: forest, light agriculture, moderate agriculture, and heavy agriculture. Leaf breakdown rates were significantly higher in streams with light ( $0.0138\text{ d}^{-1}$ ) and moderate agricultural ( $0.0139\text{ d}^{-1}$ ) land use in comparison to forested ( $0.0082\text{ d}^{-1}$ ) and heavy agricultural ( $0.0086\text{ d}^{-1}$ ) streams. Macroinvertebrates colonizing litterbags increased throughout the study with highest abundance, richness, and density in light agricultural streams. Macroinvertebrate richness was an important factor in explaining leaf breakdown rates. The number of shredding macroinvertebrates was highest in forested streams and lowest in heavy agricultural streams. Temperature, nutrient concentration, and specific conductance increased along the land use gradient from forest to heavy agriculture, while dissolved oxygen significantly decreased along that gradient. Results suggested that fast breakdown rates in light and moderate agricultural sites were due to the combined effects of high nutrients, temperature, and macroinvertebrate shredders. Slow breakdown rates in forested streams probably result from low temperatures and limited nutrients, while breakdown may be limited by sedimentation and lack of shredders in heavy agricultural streams. Due to the confounding effects that agricultural land use can have on breakdown rates, leaf breakdown was not an accurate indicator of stream integrity.

## Introduction

Allochthonous input from surrounding riparian vegetation is an important energy source to forested streams (e.g., Wallace et al. 1997). Over 99% of forested streams' energy may come from allochthonous input (Fisher and Likens 1973). Even streams draining catchments without a significant number of trees, such as tall grass prairie streams, receive organic matter input from the surrounding riparian zone, mainly in the form of grasses (Stagliano and Whiles 2002). Likewise, agricultural streams with few to no trees still receive organic matter from the riparian zone.

Once leaves enter the stream, they generally break down via a 3-step process: chemical leaching of soluble compounds, aerobic degradation by microbial organisms, and physical fragmentation by macroinvertebrate shredders and abrasion (Webster and Benfield 1986). Leaching consists of the rapid loss of soluble organic and inorganic compounds and typically occurs within the first 24 hours of immersion. During this phase leaves can lose up to 25% of their initial dry mass (Cummins 1974). The following 2 steps are much slower than the leaching phase. Aquatic microorganisms, primarily fungi and bacteria, colonize leaves within a few days after immersion in freshwater. This conditioning stage by microbes is necessary to promote further leaf breakdown by enhancing leaf litter palatability and nutritional value for macroinvertebrate shredders (Cummins 1974, Gessner et al. 1999). During the fragmentation stage, leaves are converted to smaller particles through invertebrate ingestion and production of feces (Cummins et al. 1989).

Leaf breakdown in streams is an important ecosystem processes that has been studied for over 40 years. Leaf breakdown is influenced by a number of factors including temperature, dissolved oxygen, and concentration of dissolved nutrients. Elevated nitrogen and phosphorous concentration in particular have been shown to increase breakdown rates (Robinson and Gessner 2000). Many studies have shown sedimentation, water velocity, leaf species, microbial activity, invertebrate composition, and changes in riparian vegetation and surrounding land use to alter leaf breakdown rates as well (e.g., Webster and Benfield 1986, Gessner et al. 1999).

While the factors influencing leaf breakdown rates in forested streams are well known, there have only been a few studies assessing leaf breakdown in agriculturally impacted streams (e.g., Benfield et al. 1977, Tuchman and King 1993, Huryn et al. 2002). Many of the factors that influence leaf breakdown are altered by the conversion of forest to agriculture. These

include temperature and light regimes, dissolved oxygen and nutrient concentrations, sedimentation, riparian vegetation, and aquatic macroinvertebrates. Agricultural land use often results in a reduction of riparian vegetation thus limiting shading, increasing stream insolation, and elevating stream temperatures. Dissolved oxygen levels decrease with warmer temperatures. Nutrient concentrations increase with agricultural land in response to runoff of fertilizers and grazing animals eliminating adjacent to and directly in the stream system (Townsend and Riley 1999). Agricultural land use often results in increased sedimentation, soil erosion, and runoff in streams (Allan et al. 1997). While the movement of sediments, especially during storms, can cause abrasion and physical breakdown of leaf material, sediment can bury leaves resulting in anaerobic conditions, which in turn prevent fungal or macroinvertebrate colonization (Cummins et al. 1980, Webster and Waide 1982). Additionally, high rates of sedimentation can be detrimental to aquatic biota further slowing biological fractionation of leaf litter (Zweig and Rabeni 2001).

Allochthonous material, primarily in the form of leaves, is an important energy source for macroinvertebrates in stream food webs (e.g., Cummins 1974, Minshall et al. 1985). Several studies have shown macroinvertebrate abundance to positively correlate with leaf breakdown (Peterson and Cummins 1974, Iversen 1975, Wallace et al. 1982, Benfield 1996, Dangles et al. 2001). These shredding macroinvertebrates play an important role in leaf breakdown and particulate organic matter production in streams (Wallace et al. 1982, Kirby et al. 1983, Benfield and Webster 1985, Sponseller and Benfield 2001). However, agricultural land use often reduces the quantity of allochthonous inputs to stream ecosystems. A reduction in leaf litter input may reduce shredder biomass and production due to limited food availability, which in turn potentially affects leaf breakdown (Dance and Hynes 1980, Wallace et al. 1997, Sponseller and Benfield 2001). Harding and Winterbourn (1995) found lower macroinvertebrate richness and biomass in streams with pastoral land use in comparison to forested streams. In addition to providing a food source, allochthonous material also provides stream stability and habitat. Large woody debris, an important source of retention, also tends to be limited in agricultural streams. Reduction of retention materials can limit macroinvertebrate habitat and subsequent establishment causing lower macroinvertebrate density and diversity.

Because leaf breakdown rates are influenced by a multitude of factors, breakdown rates can be used to assess the effects of anthropogenic disturbances on stream ecosystem integrity

(Webster and Benfield 1986). While the definition of integrity is subject to debate (Karr 1991), for the purpose of this study integrity will refer to a systems wholeness, which includes the presence of appropriate components and occurrence of processes at appropriate rates (Angermeier and Karr 1994). Stream integrity consists of two components: structural integrity and functional integrity (Minshall 1996). Structure includes biotic assemblages, while function applies to ecosystem processes. Streams with minimal deviation from a natural reference conditions will be defined as having high integrity (Bunn and Davies 2000). Macroinvertebrate communities are commonly used to assess stream integrity (Lenat and Crawford 1994, Karr 1999) because water quality has an overt impact on macroinvertebrate community structure. However, it is often unclear how changes in stream structure impact ecosystem function (Bunn et al. 1999). Measures of ecosystem function (e.g., nutrient cycling, leaf breakdown, and primary production) have been used as indicators of stream integrity (Meyer 1997, Bunn et al. 1999, Gessner and Chauvet 2002). Even though, ecosystem processes may not change in response to shifts in structure (e.g., species diversity) (Angermeier and Karr 1994). Several studies suggest using leaf breakdown as a measure of stream integrity (Gessner and Chauvet 2002, Niyogi et al. 2003) because leaf breakdown is an ecosystem level process that is controlled by a variety of structural components including allochthonous input, microbial and invertebrate activity, and chemical and physical conditions (Benfield 1996, Gessner and Chauvet 2002, Pascoal et al. 2003).

The objective of this study was to compare leaf breakdown rates in streams along a gradient of agricultural land use in the southern Appalachians (North Carolina and Georgia). In order to better understand the factors controlling leaf breakdown in agricultural streams biological, chemical, and physical parameters of each stream reach were characterized and related to different types of agricultural land use. Study stream conditions varied along a gradient of agricultural land use ranging from reference (forested) to heavy agricultural land use. Forested streams had high integrity because these streams represented the natural reference condition. Conversion to agricultural land use resulted in a number of alterations to the stream ecosystem level processes that may have affected stream integrity including reduced allochthonous input, increased light infiltration, warmer temperatures, increased nutrient concentrations, and sedimentation (Bunn et al. 1999). These factors can further impact microbial and macroinvertebrate communities. Therefore, stream integrity of agricultural streams was

expected to decrease as streams deviated from natural forested conditions. The use of breakdown rates as measures stream integrity was assessed along the land use gradient from forest to agriculture.

## Methods

### *Site description*

Study sites were located in the southern Appalachians (Macon County, North Carolina and Rabun County, Georgia) in 12 catchments along a gradient from forest to agricultural land use in the Little Tennessee drainage system. Land use categories were forest, light agriculture, moderate agriculture, and heavy agriculture. Due to steep terrain and rocky soil in the southern Appalachians, land was not well suited for row crops. Therefore, the primary use of agricultural land was pasture for livestock grazing. Streams were placed into land use categories based on riparian tree density and basal area, percentage riparian canopy cover within 10-m from each stream bank, percentage stream canopy cover, percentage grass groundcover 0-10 m from each stream bank, and influence of agriculture (e.g., presence of livestock) (Table 1). Forested streams had an intact, forested riparian zone and no agricultural land use present in its watershed. Aside from logging around the early 1900's, the forested sites had very little anthropogenic impact. The forested streams were located in the Nantahala National Forest; Ball Creek and Hugh White Creek were also part of the Coweeta Hydrologic Laboratory. Light agricultural streams had intact, forested riparian zones with areas of pastoral land use close to the stream. Large patches of undisturbed forest were present in light agricultural stream catchments. Light agricultural streams were located in the Nantahala National Forest (North Prong Little Ellijay) or on private land (Dryman Fork and Tessentee Creek). Moderate and heavy agricultural streams were located on private land. Streams with moderate agriculture had limited sections of intact riparian vegetation with pasture directly adjacent to the stream bank. Livestock were fenced from moderate agricultural streams. Heavy agricultural streams had little to no riparian vegetation and cattle had direct access to the stream or had direct access in recent years.

Riparian vegetation along forested streams consisted of a mixed deciduous forest composed primarily of birches (*Betula* sp.), maples (*Acer* sp.), oaks (*Quercus* sp.), eastern hemlock (*Tsuga canadensis* (L.) Carr.), and yellow-poplar (*Liriodendron tulipifera* L.), with a dense understory of rhododendron (*Rhododendron maximum* L.). Light agricultural riparian vegetation was primarily yellow-poplar, yellow buckeye (*Aesculus octandra* Marsh.), and red

maple (*A. rubrum* L.). Red maple and alder (*Alnus serrulata* (Ait.) Willd.) were dominant species in moderate agricultural riparian zones. The few trees present in heavy agricultural riparian zones included alder and sycamore (*Platanus occidentalis* L.).

### ***Physicochemical variables***

Water samples were collected approximately monthly, December 2002 through September 2003, from each stream reach. On each sampling date, 3 60-mL water samples were filtered in the field (Whatman GF/F), stored in acid washed polyethylene bottles, and frozen until analysis. Nitrate-nitrogen (NO<sub>3</sub>-N) and ammonium-nitrogen (NH<sub>4</sub>-N) were measured 7 times (December 2002 and January, March, April, May, June, and September 2003) using a Technicon Autoanalyzer II (Saskatoon, SK, Canada) or Dionex DX500 Chromatography System (Ion Chromatography/High Pressure Liquid Chromatography) (Sunnyvale, CA, USA). The Technicon Autoanalyzer II was always used to measure NH<sub>4</sub>-N concentration. Phosphate-phosphorous (PO<sub>4</sub>-P) was measured 5 times (January, March, April, May, and September 2003) using the Technicon Autoanalyzer II. One half detection limits were used for nutrient concentration below instrument resolution.

Dissolved oxygen (DO) (mg/L) and specific conductance (μS/cm) were measured monthly, October 2002-September 2003, at each stream site, using YSI dissolved oxygen and conductivity probes (Model 55 DO probe and Model 30 conductivity probe, YSI inc., Yellow Springs, OH, USA). Stream temperatures were recorded every four hours, September 2002-2003, using temperature data loggers (HOBO, Pocasset, MA, USA). Average annual temperature was calculated for each stream reach. Additionally, mean temperature and number of cumulative degree days (dd) above 0°C throughout the study's duration were calculated.

Discharge was measured at 3 locations in each stream during summer base flow conditions. To calculate discharge each stream's wetted width was multiplied by average depth and average velocity. Velocity was measured with a Marsh-McBirney Flo-Mate Model 2000 flow meter (Marsh-McBirney, Inc., Frederick, MD, USA) and depth was recorded at 3 to 10 evenly spaced points across each stream's wetted width. The 3 discharge measurements were averaged for each stream reach.

### ***Leaf breakdown***

Red maple leaf breakdown was measured over a 5-mo period beginning November 2002 using the litterbag method (Benfield 1996). Red maple leaves were used because it was a



dominant riparian species at most of the stream sites and was a major contributor to leaf input. Senescent leaves from a single tree were collected shortly after abscission in autumn 2002 and air-dried to a constant weight. Twelve litterbags (4-mm mesh size) were filled with 6.0-g dried leaves and placed in each of the 12 streams during natural litter fall. Leaf packs were anchored by attaching each leaf pack to a wire cable that was secured to a fence post in the streambed. Upon initial placement of litterbags, 5 additional leaf packs were brought back to the lab to account for handling loss.

Three leaf packs were retrieved at random from each site 4 times over the 5-mo study and stored on ice until processed. Within 48 h of collection, litterbags were rinsed over a 250- $\mu$ m mesh sieve to remove sediment and collect aquatic invertebrates. Leaves were dried (50°C) to a constant mass, weighed, ground, subsampled, and ashed (550°C, 45 min) to determine the ash-free dry mass (AFDM) remaining.

A short-term (39-h) leaching study was conducted to account for the initial and rapid loss of soluble compounds from the leaves upon entering the stream. Nine leaf packs, each with 6.0-g red maples leaves, were placed in Ball Creek on 21 December 2002. Three leaf packs were recovered after 12, 24, and 39 h. To account for handling loss, 4 additional leaf packs were taken to the field but never placed in the stream. AFDM was determined on leaves used in the leaching experiment.

### ***Macroinvertebrates***

Macroinvertebrates inhabiting the leaf packs were preserved in 80% ethanol, enumerated, and identified to genus when possible except for Oligochaeta and Chironomidae, which were identified to the class and family, respectively. Invertebrates were placed into functional feeding groups according to Merritt and Cummins (1996) and Voshell (2003). Mean number of macroinvertebrates per leaf pack, macroinvertebrate richness per stream, and percentage EPT ((Ephemeroptera + Plecoptera + Trichoptera) / total abundance \* 100%) per leaf pack were calculated. Additionally, macroinvertebrate diversity and evenness were estimated using the Shannon-Wiener Diversity Index (e.g., Hauer and Resh 1996, Morin 1999). Total macroinvertebrate, shredder, and scraper density per leaf pack were determined by dividing the number of invertebrates by g AFDM leaf litter remaining.

### ***Statistical analyses***

Water chemistry variables (NO<sub>3</sub>-N, NH<sub>4</sub>-N, PO<sub>4</sub>-P, DO, and specific conductance) were response variables and differences among land use were analyzed using a 2-way repeated measure (RM) analysis of variance (ANOVA) with land use as a fixed variable and time as a repeated variable to test for differences among land use category. Differences in annual stream temperature among land use types were analyzed using a 1-way ANOVA followed by Fishers Least Significant Differences (LSD) multiple comparison test. Temperature was a response variable and land use was a fixed variable.

To compare leaf breakdown rates among land use, a breakdown rate was calculated using a negative exponential decay model (Petersen and Cummins 1974). Breakdown rates were calculated as the slope of the regression line of the natural log of g AFDM remaining verses time. A general linear model with dummy variables for land use type and time as a covariate was used to compare breakdown rates among land use types. Each land use category was compared individually to identify differences in breakdown rates among land use.

Macroinvertebrate assemblages (abundance of total macroinvertebrates, shredders, scrapers, collector-filters, collector-gathers, and predators) were response variables. Differences among macroinvertebrate assemblages and land use were analyzed using a 2-way RM ANOVA with land use as a fixed variable and time as a repeated variable. Fisher's LSD multiple comparison test was used to measure differences among land use types. Additionally, total macroinvertebrate, shredder, and scraper density were response variables and differences among land use were assessed using a 2-way RM ANOVA with land use as a fixed variable and time as a repeated variable. Macroinvertebrate richness, percentage EPT, diversity, and evenness were response variables analyzed using a 1-way ANOVA followed by Fisher's LSD multiple comparison test to test for differences among land use types.

We performed regression analysis to examine the effects of biological, chemical, and physical factors on breakdown rates. We used forward stepwise multiple regression to select the best model for predicting leaf breakdown rates among land use types. NO<sub>3</sub>-N and PO<sub>4</sub>-P concentration, annual stream temperature, and shredder density were used as independent variables. Results were considered significant if  $p < 0.05$ .

Table 1. Characteristics of land use types. Different letters represent statistical differences among means (ANOVA, followed by LSD,  $p < 0.05$ )

Land Use	Tree density (#/ha)	Basal area (m <sup>2</sup> /ha)	Riparian canopy cover (%)	Stream canopy cover (%)	Grass groundcover (%)	Agricultural influence
Forest	1207-1464 A	26-40 A	83-86 A	33-39 A	0 B	Agriculture was not present in watershed.
Light agriculture	256-580 B	15-58 AB	61-84 A	25-30 B	6-42 B	Agriculture was present in watershed but not adjacent to stream.
Moderate agriculture	157-639 B	6-25 B	37-80 A	26-32 B	29-31 B	Agriculture was separated from stream by a thin, patchy vegetated riparian zone. Livestock did not have access to stream.
Heavy agriculture	0-49 C	0-3 C	0-13 B	1-8 C	67-95 A	Agriculture was directly adjacent to stream. Little to non-vegetated riparian zone was present. Livestock had direct access to stream

## Results

### *Physicochemical variables*

Stream nutrient concentrations generally increased along the land use gradient from forested to heavy agricultural streams. Mean NO<sub>3</sub>-N increased 4-fold from forested to heavy agricultural streams and NH<sub>4</sub>-N more than doubled along the land use gradient, yet concentrations were not significantly related to land use (Table 2). Mean yearly NO<sub>3</sub>-N values were high for forested streams due to high NO<sub>3</sub>-N in Jones Creek (69 ppb) in comparison to 11 ppb NO<sub>3</sub>-N recorded in Ball Creek and Hugh White Creek. NO<sub>3</sub>-N ranged from 26-77 ppb in light agricultural streams and ranged from 25-192 ppb in moderate agricultural streams. Heavy agricultural streams consistently had high NO<sub>3</sub>-N concentrations (63-211 ppb). Mean NH<sub>4</sub>-N per land use ranged from 3-10 ppb and was highest in moderate and heavy agricultural streams. Mean PO<sub>4</sub>-P was almost 2 times higher in agricultural streams than in forested streams.

Chloride, specific conductance, and temperature were significantly related to land use, showing significantly higher values in agriculturally impacted streams (Table 2). Annual stream temperature (September 2002-2003) ranged from 10.7°C in Ball Creek (forested) to 14.4°C in Payne Creek (heavy agriculture). Throughout the study, November 2002-April 2004, average stream temperature was coldest in North Prong Ellijay (6.4°C) and warmest in Payne Creek (8.9°C). The number of degree-days (of the 148 day study) ranged from 939 in North Prong Ellijay to 1316 in Payne Creek (Table 3). The number of degree-days increased along the forested to heavy agricultural stream gradient. Dissolved oxygen decreased along the land use gradient from forested to agricultural land use ( $p < 0.001$ ) (Table 2). Width, depth, velocity, and discharge were not significantly different among land use. Average discharge was 0.30 m<sup>3</sup>/s (range: 0.02 to 0.76 m<sup>3</sup>/s) with no pattern among land use (Table 2).

Table 2. Mean ( $\pm$  SE) chemical and physical characteristics for each land use type. Water was sampled October 2002-September 2003. *P* values less than 0.05 were considered significantly different among land use types.

Parameter	Forest	Light agriculture	Moderate agriculture	Heavy agriculture	<i>p</i> value
NO <sub>3</sub> -N (ppb)	30.4 (3.7)	50.6 (3.4)	108.2 (19.8)	130.9 (8.2)	0.28
NH <sub>4</sub> -N (ppb)	3.2 (0.4)	2.7 (0.3)	10.3 (1.5)	8.2 (0.8)	0.06
PO <sub>4</sub> -P (ppb)	1.7 (0.2)	3.2 (0.3)	3.3 (0.4)	3.1 (0.3)	0.16
Dissolved oxygen (mg/L)	8.38 (0.39)	8.42 (0.34)	7.98 (0.39)	7.77 (0.30)	<0.001
Specific conductance ( $\mu$ S/cm)	14.2 (1.4)	21.6 (2.3)	25.2 (0.7)	29.5 (1.2)	0.03
Temperature ( $^{\circ}$ C)	10.9 (0.1)	11.7 (0.0)	12.3 (0.0)	13.7 (0.2)	0.01
Width (m)	3.7 (1.1)	5.6 (0.2)	2.9 (1.6)	2.0 (0.6)	0.06
Depth (m)	0.14 (0.04)	0.20 (0.03)	0.16 (0.04)	0.16 (0)	0.50
Velocity (m/s)	0.50 (0.15)	0.47 (0.01)	0.51 (0.24)	0.60 (0.07)	0.85
Discharge (m <sup>3</sup> /s)	0.31 (0.14)	0.54 (0.11)	0.17 (0.06)	0.17 (0.04)	0.08

Table 3. Red maple leaf breakdown rates in 12 streams with different land use (dd = degree days).

Land use	Stream name	Stream abbreviation	Breakdown rate (d <sup>-1</sup> )	r <sup>2</sup>	Breakdown rate (dd <sup>-1</sup> )	r <sup>2</sup>	Cumulative degree days
Forest	Ball Creek	BAL	0.0059	0.98	0.0009	0.93	963
Forest	Hugh White Creek	HWC	0.0075	0.95	0.0011	0.99	1088
Forest	Jones Creek	JON	0.0113	0.96	0.0018	1.00	974
Light agriculture	Dryman Fork	DRY	0.0121	0.90	0.0018	0.96	1071
Light agriculture	North Prong Ellijay Creek	ELL	0.0180	0.90	0.0030	0.98	939
Light agriculture	Tessentee Creek	TES	0.0114	0.97	0.0017	0.98	1018
Moderate agriculture	North Shope Fork	SHO	0.0138	0.94	0.0018	0.97	1199
Moderate agriculture	Sutton Branch	SUT	0.0140	0.90	0.0020	0.97	1095
Heavy agriculture	Caler Fork	CAL	0.0083	0.99	0.0011	0.94	1092
Heavy agriculture	Hoglot Branch	HOG	0.0053	0.94	0.0007	0.99	1079
Heavy agriculture	Payne Creek	PAY	0.0093	0.96	0.0010	0.92	1316
Heavy agriculture	North Fork Skeenah Creek	SKE	0.0115	0.88	0.0015	0.96	1232

### ***Leaf breakdown***

Red maple leaves lost 20% of initial dry weight within the first 12 hours of the leaching study (Fig. 1). To account for loss of leaf mass due to handling (11%) and leaching (20%), the initial leaf pack weight used in determining breakdown rates was 4.2 g AFDM. Red maple leaf breakdown rates ranged from 0.0053 to 0.0180 d<sup>-1</sup> and 0.0007 to 0.0030 dd<sup>-1</sup> across all study streams (Table 3). Breakdown rates were significantly faster in light (0.0138 d<sup>-1</sup>) and moderate (0.0139 d<sup>-1</sup>) agricultural streams in comparison to forested (0.0082 d<sup>-1</sup>) and heavy (0.0086 d<sup>-1</sup>) agricultural streams ( $p < 0.05$ ) (Fig. 2A). When measured using cumulative degree-days, breakdown rates also were significantly slower in forested and heavy agricultural streams and substantially faster rates in light and moderate agricultural streams ( $p < 0.05$ ) (Fig. 2B). Leaf breakdown rates in forested streams ranged from 0.0059-0.0113 d<sup>-1</sup> with the fastest rate measured in Jones Creek (forested) (Table 3). Breakdown rates in light and moderate agricultural streams were similar with fastest rates in North Prong Ellijay (Table 3). Leaf breakdown rates in heavy agricultural streams ranged from 0.0053 to 0.0115 d<sup>-1</sup> (Table 3).

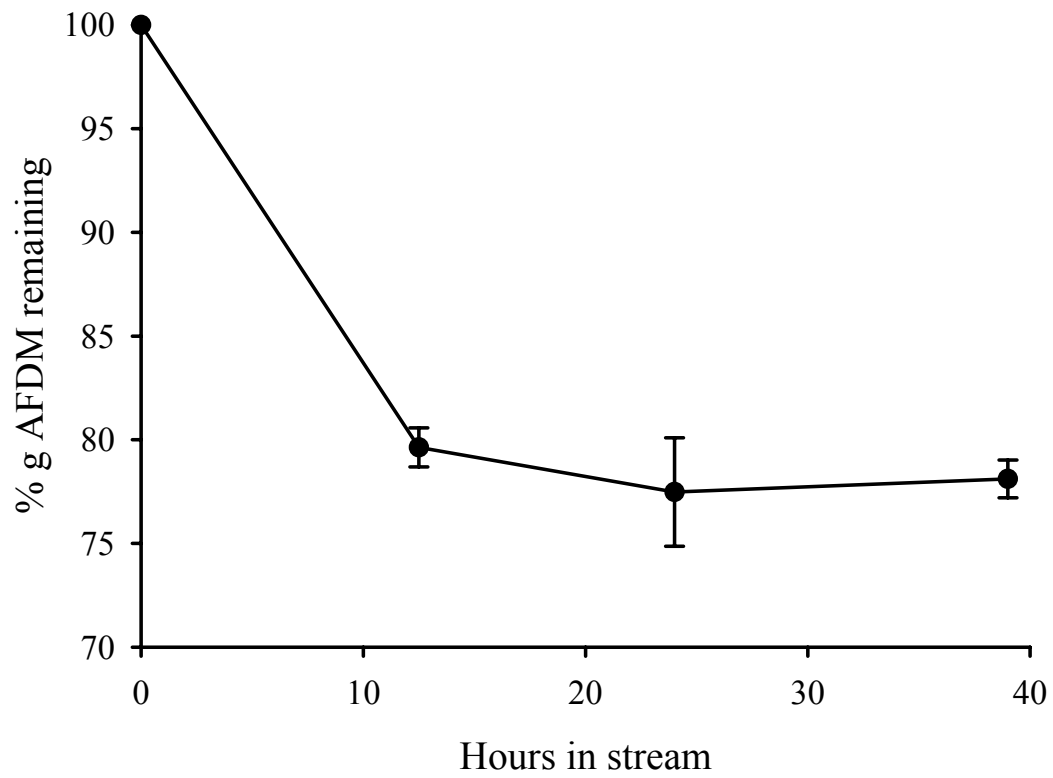


Fig. 1. Mean ( $\pm$  SE) percentage ash free dry mass remaining after 12.5, 24, and 39 hours in Ball Creek. After correcting for transportation loss, red maple leaves lost 20% of initial mass within the first 12 hours.



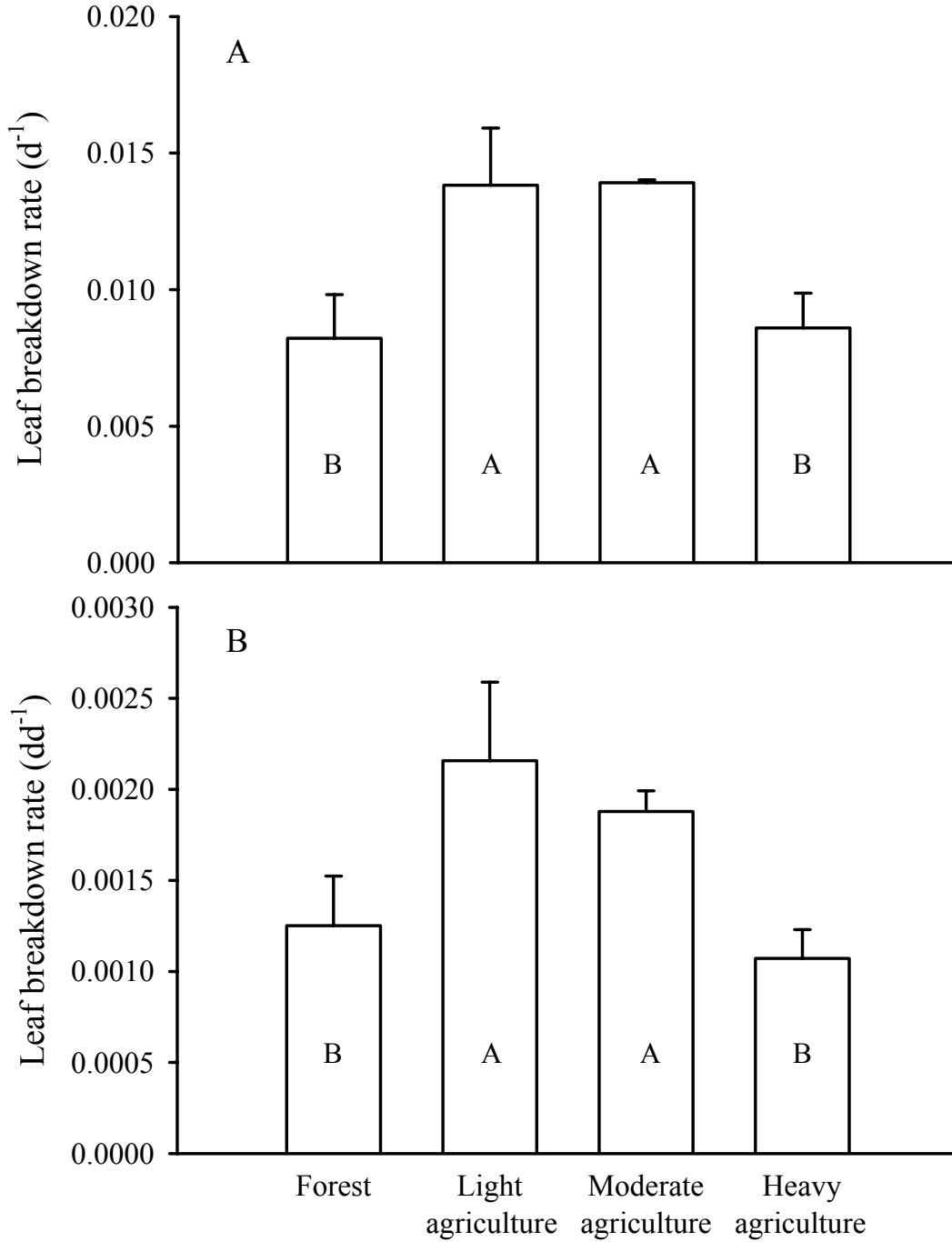


Fig. 2. Mean ( $\pm$  SE) red maple leaf breakdown rates calculated for each land use type. Breakdown rates (A) per day and (B) per degree day were significantly different among land use types ( $p < 0.05$ ) (dd = degree days).

### ***Macroinvertebrates***

The mean number of total macroinvertebrates ranged from 35 to 84 individuals per leaf pack (Table 4). The number of macroinvertebrates colonizing leaf packs in light agricultural streams was 1.5 to 1.7 times higher than the other land use types ( $p = 0.03$ ). The number of macroinvertebrates colonizing litterbags increased throughout the study with significantly higher colonization after 100 days in the stream ( $p < 0.001$ ). The number of macroinvertebrates inhabiting leaf packs was highest in light agricultural streams (Fig. 2). Heavy agriculture and forested streams consistently had the fewest number of macroinvertebrates. Macroinvertebrate richness increased in each land use category from day 0 to day 148; light agricultural streams consistently had the highest number of genera, while heavy agricultural streams had the lowest.

The mean number of shredding macroinvertebrates ranged from 2 to 19 individuals per leaf pack and was related to land use ( $p = 0.03$ ) (Table 4). Seven times more shredders were observed in forested streams than in heavy agricultural streams (Table 4). Over the course of the study, the highest numbers of shredders were found in forested streams and the lowest numbers in heavy agricultural streams, except for day 27 when more shredding invertebrates were found in light agricultural streams (Fig. 3). Shredder abundance (per leaf pack) increased throughout the course of the study ( $p = 0.03$ ). *Tallaperla* sp. were observed in high abundance in Ball Creek, while *Tipula* sp. were important shredders in light and moderate agricultural streams. *Lepidostoma*, *Pychnopsyche*, and *Lectura* were also important shredding macroinvertebrates in forest, light agriculture, and moderate agricultural streams (Appendix B).

Among land use types, low numbers of scrapers colonized leaf packs. Heavy agricultural streams spanned the range of scraper abundances (range: 1-11 individuals per leaf pack) (Table 4). Aside from heavy agricultural streams, scraper abundance did not increase with time in leaf pack. Abundant scrapers in agriculturally impacted streams include the mayfly, *Stenonema*, and the beetle, Elmidae (Appendix B).

More collector-filtering invertebrates were identified in the heavy agricultural streams (Table 4), which was largely due to a high number of *Chematopsyche* caddisflies collected in Caler Fork. Most invertebrates inhabiting litterbags belonged to the collector-gather functional feeding group (21 to 67% of total invertebrates collected). This was mainly the result of Chironomidae and *Ephemerella* occurring in high abundances in each land use type (Appendix B). The mayfly, *Ephemerella*, was especially predominant in light agricultural streams ( $p =$

0.03). The number of collector-gathers colonizing leaf packs in light agricultural streams was 2.3 times higher than in forested streams, 1.6 times higher than in moderate streams, and 1.5 times higher than in heavy agricultural streams ( $p = 0.01$ ) (Table 4). The number of collector-gathers increased 5-fold from day 27 to day 100 ( $p < 0.001$ ). Similar numbers of predators were collected in each land use type (Table 4). However, abundance significantly increased over the course of the study ( $p < 0.001$ ).

Mean macroinvertebrate density in leaf packs ranged from 14 to 93 individuals per g AFDM leaf litter remaining in Hoglot Branch and Dryman Fork, respectively (Table 5). Mean macroinvertebrate density in light agricultural streams were approximately 3 times higher than forested and heavy agricultural streams and 1.4 times higher than moderate agricultural streams ( $p < 0.05$ ) (Table 5). In each land use type, macroinvertebrate density increased with days in stream and was highest in light agricultural streams and lowest in moderate and heavy agricultural streams ( $p < 0.001$ ).

Among land uses, shredder density was significantly lower in heavy agricultural streams ( $p < 0.01$ ) (Table 5). Mean shredder density also increased throughout the study ( $p < 0.05$ ). Mean scraper density and land use type were not significantly related (Table 5); however, scraper density increased with time ( $p < 0.001$ ). Scraper density was low in each land use type from day 0 to 100 but increased in moderate (29 individuals/g AFDM remaining) and light agricultural (16 individuals/g AFDM remaining) streams by day 148. However, mean scraper density in the moderate agricultural streams was quite variable with 3 individuals/g AFDM remaining in Sutton Branch and 10 individuals/g AFDM remaining in North Shope Fork. Collector-filter and predator density increased significantly across land use and time ( $p < 0.05$ ), yet there was no relationship among collector-filter density with land use or time.

Percentage EPT was not related to land use, however, macroinvertebrate diversity and evenness significantly decreased along the forested to agricultural gradient ( $p < 0.05$ ). Shannon Wiener diversity was approximately 1.3 times higher in forest and light agricultural streams in comparison to heavy agricultural streams (Table 4). Evenness was 1.2 times lower in the heavy agricultural streams in comparison to forested streams (Table 4). Leaf breakdown was not significantly related to diversity or evenness.

The relationship between physicochemical and macroinvertebrate variables, and breakdown rates were examined to determine what factors were controlling leaf breakdown

rates. Breakdown rates were positively related to macroinvertebrate richness ( $r^2 = 0.36$ ,  $p = 0.04$ ) (Fig. 4), but no other variables explained a significant portion of breakdown. Shredding density was the best predictor of breakdown rates in all land use types and agricultural land use types using forward stepwise regression analysis.

Table 4. Mean number of macroinvertebrates per leaf pack, community richness per stream, mean number of shredders (SH), scrapers (SC), collector-filters (C-F), collector-gathers (C-G), and predators (PR) per leaf pack, percentage EPT (invertebrates in the orders Ephemeroptera, Plecoptera, and Trichoptera) per stream, Shannon-Wiener Diversity Index (H') per stream, and Shannon Evenness (E) per stream. Macroinvertebrates were collected 4 times, December 2002 - April 2003. *P* values are shown if there was a significant difference among land use types. Different letters indicate significant differences among land use types (1 way ANOVA followed by Fisher's LSD, *p* < 0.05 for macroinvertebrate richness, % EPT, H' and E; 2-way RM ANOVA with time as a repeated measure for # SH, # SC, # C-F, # C-G, and # PR followed by Fisher's LSD, *p* < 0.05). Stream abbreviations are given in Table 3.

Land use	Stream	Total number of macroinvertebrates		SH		C-G			PR	EPT	H'		E			
		( <i>p</i> = 0.03)	Richness	( <i>p</i> = 0.03)	Richness	SC	C-F	( <i>p</i> = 0.01)			( <i>p</i> = 0.04)	( <i>p</i> = 0.01)				
Forest	BAL	63.9	26	19.2	2.8	1.9	13.5	26.5	75.5	2.39	0.73					
Forest	HWC	34.7	B	28	9.8	A	1.8	3.8	11.7	B	7.5	62.5	2.51	A	0.75	A
Forest	JON	45.8		37	9.8		2.3	0.9	23.2		9.3	54.4	2.62		0.72	
Light agriculture	DRY	83.8		37	8.2		5.9	2.8	37.9		28.6	60.7	2.46		0.68	
Light agriculture	ELL	74.3	A	39	9.8	A	3.5	4.1	32.4	A	23.8	83.1	2.50	A	0.68	AB
Light agriculture	TES	85.2		40	16.2		5.7	2.2	43.7		17.0	61.9	2.51		0.68	
Moderate agriculture	SHO	62.7	B	32	11.3	AB	3.8	1.9	25.2	B	20.3	75.4	2.41	AB	0.70	BC
Moderate agriculture	SUT	43.2		27	2.4		7.8	2.8	21.3		8.5	19.7	1.98		0.60	
Heavy agriculture	CAL	70.2		34	4.3		10.8	16.8	26.4		12.0	67.6	2.37		0.67	
Heavy agriculture	HOG	39.7	B	16	0.3	B	1.1	1.1	23.9	B	13.1	8.4	1.59	B	0.57	C
Heavy agriculture	PAY	40.8		20	0.9		7.5	1.3	27.3		3.8	18.4	1.85		0.63	
Heavy agriculture	SKE	38.8		26	1.1		2.1	3.1	24.1		8.5	43.3	1.49		0.60	

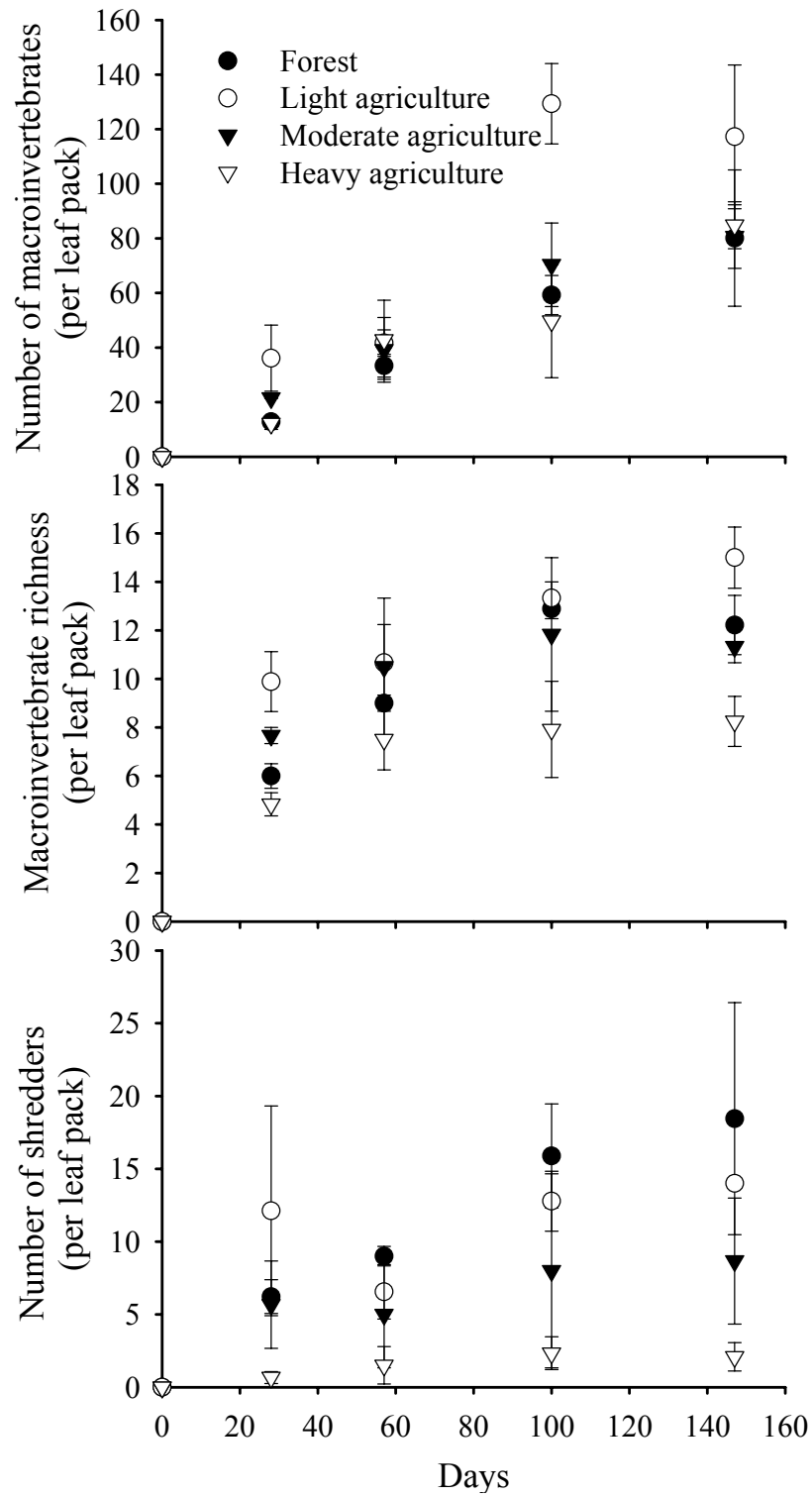


Fig. 3. Mean ( $\pm$  SE) number of macroinvertebrates, macroinvertebrate richness, and number of shredders per leaf pack in each land use type. Samples were collected December 2002 through April 2003.

Table 5. Mean ( $\pm$  SE) macroinvertebrate density, shredder density, and scraper density (# individuals per g AFDM remaining). Total macroinvertebrate and shredding macroinvertebrate density were significantly different among land use types (2-way RM ANOVA, followed by LSD,  $p < 0.05$ ). Different letters indicate statistical difference among land use types. Stream abbreviations are given in Table 3.

Land use	Stream	Mean macroinvertebrate density ( $p < 0.001$ )		Mean shredder density ( $p = 0.01$ )		Mean scraper density
Forest	BAL	25 (14)		8 (5)		1 (1)
Forest	HWC	19 (10)	C	5 (3)	A	1 (1)
Forest	JON	32 (18)		6 (3)		2 (10)
Light agriculture	DRY	93 (74)		6 (4)		5 (3)
Light agriculture	ELL	87 (59)	A	14 (11)	A	4 (3)
Light agriculture	TES	78 (47)		12 (6)		4 (3)
Moderate agriculture	SHO	66 (44)		10 (6)		2 (1)
Moderate agriculture	SUT	58 (45)	B	3 (3)	A	13 (11)
Heavy agriculture	CAL	33 (13)		2 (1)		5 (2)
Heavy agriculture	HOG	14 (8)		0 (0)		0 (0)
Heavy agriculture	PAY	29 (19)	C	1 (0)	B	5 (4)
Heavy agriculture	SKE	46 (41)		2 (2)		2 (1)

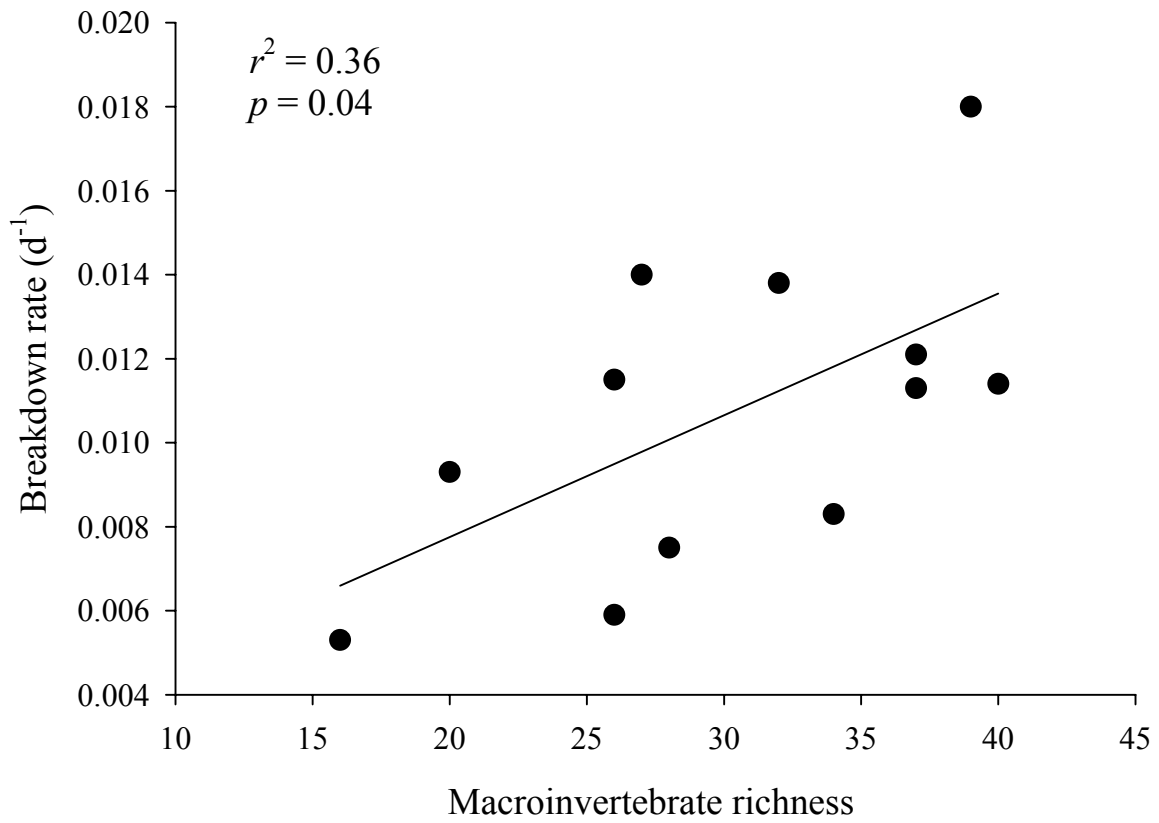


Fig. 4. Relationship between leaf breakdown rate (d<sup>-1</sup>) and macroinvertebrate richness.



## Discussion

### *Leaf breakdown*

Red maple breakdown rates of 0.0053-0.0180 d<sup>-1</sup> were similar to previous studies. Petersen and Cummings (1974) reported red maple breakdown rates of 0.0062 and 0.0093 d<sup>-1</sup> and Hutchens and Wallace (2002) reported breakdown rates of 0.0175-0.0318 d<sup>-1</sup> for red maple leaves in forested streams. Huryn et al. (2002) measured red maple breakdown rates of approximately 0.0027 and 0.0024 d<sup>-1</sup> in forest and agricultural streams, respectively. In the current study, red maple breakdown rates were fast in all streams except for 2 forested and 3 heavy agricultural streams, in these streams leaves broke down at a medium rate (Petersen and Cummins 1974). Breakdown rates calculated using degree-days were slow to medium in forested and heavy agricultural streams and fast in light and moderate agricultural streams (Cummins et al. 1989). By using degree days to make comparisons among land use, the effects of temperature on breakdown rates were removed (Young et al. 1994). Leaf breakdown rates were significantly faster in light and moderate agricultural streams when measured using degree-days suggesting additional factors are influencing breakdown rates.

Agricultural land use generally was associated with higher nutrient concentrations (NO<sub>3</sub>-N, NH<sub>4</sub>-N, and PO<sub>4</sub>-P). In this study NO<sub>3</sub>-N concentrations increased along the gradient from forested to agricultural land use and NH<sub>4</sub>-N concentrations were higher in moderate and heavy agricultural streams in comparison to forested and light agricultural streams. Faster breakdown rates in Jones Creek compared to the other forested streams were likely due to high NO<sub>3</sub>-N levels (69 ppb). Overall, breakdown rates were fastest in North Prong Ellijay coinciding with higher NO<sub>3</sub>-N concentration in comparison to the other light agricultural streams studied. Elevated NO<sub>3</sub>-N and NH<sub>4</sub>-N values in the moderate and heavy agricultural streams were probably due to a large percentage of nitrogen fixing alder trees in the riparian zone (Chapter II). Riparian zones dominated by alder, a nitrogen fixer, have been shown to increase stream nitrate concentrations (Gregory et al. 1991). Elevated nutrients in the heavy agricultural streams can also be attributed to cattle grazing in the riparian zone. Cattle grazing in riparian zones can increase nutrient loading to streams and increase erosion by physically altering near stream habitat (Del Rosario et al. 2002).

Breakdown rates were similar in both moderate agricultural streams despite higher NO<sub>3</sub>-N and NH<sub>4</sub>-N concentrations in Sutton Branch. Breakdown was probably limited by low

shredder density and high sedimentation in Sutton Branch in comparison to North Shope Fork. The range in breakdown rates in heavy agricultural streams was greater than other land use types assessed (0.0053-0.0115 d<sup>-1</sup>). North Fork Skeenah Creek breakdown rates were 2 times higher than in Hoglot Branch. However, NO<sub>3</sub>-N and NH<sub>4</sub>-N concentrations were substantially higher in Hoglot Branch. Temperature regime and shredder density were similar; therefore, variation in breakdown rates is most likely due to low sedimentation in North Fork Skeenah Creek. Overall, mean rates of leaf breakdown were faster in light and moderate agricultural streams; however, breakdown rates were quite variable within each land use type. These findings are similar to those of Huryn et al. (2002) who did not find a significant difference in red maple breakdown rates between forested and agricultural streams. They attributed this to the influence of shredding macroinvertebrates and nutrients (nitrate and soluble reactive phosphorous) negating one another. In the current study, fast breakdown rates in light and moderate agricultural sites were due to the additive effects of high nutrients, temperature, and macroinvertebrate richness and shredder density. Slow breakdown rates in forested streams were probably due to low temperatures and limited nutrients, and breakdown rates were probably limited by sedimentation in heavy agricultural streams. High nutrients and warm temperatures were expected to increase breakdown rates; however, sedimentation was expected to slow breakdown rates (Townsend and Riley 1999).

### ***Macroinvertebrates***

Shredding macroinvertebrates were the strongest predictor of leaf breakdown rates. While elevated nutrient concentrations were consistent with fast breakdown rates in light and moderate agricultural streams, Pascoal et al. (2003) found the presence or absence of macroinvertebrates to explain a higher degree of breakdown rates than nutrient concentrations in a river polluted by a wastewater treatment plant. In the current study, shredder abundance and density dramatically decreased from forested to heavy agricultural streams. Jonsson et al. (2001) found shredder species richness and shredder abundance to strongly correlate with leaf breakdown rates across a range of streams sizes in Sweden. In the current study, breakdown rates were related to macroinvertebrate richness but not with shredder density. These results agree with findings by Huryn et al. (2002).

Townsend et al. (1997) observed maximum macroinvertebrate richness in streams with intermediate levels of disturbance (measured as bed movement in response to flooding). In this

study, though not significant, higher macroinvertebrate richness coincided with intermediate levels of agricultural land use (light and moderate) thus supporting Connell's (1978) intermediate disturbance hypothesis. Additionally, macroinvertebrate density adheres to the intermediate disturbance hypothesis with highest density in light and moderate agricultural streams.

The presence of shredding macroinvertebrates was significantly less in heavy agricultural streams suggesting that other factors are dominating the leaf breakdown process in these streams, such as microbial degradation and fractionation by physical breakage. Leaf softening by microbial degradation followed by physical fragmentation is the assumed mechanism of leaf breakdown in agricultural streams (Benfield et al. 1977, Bird and Kaushik 1992, Tuchman and King 1993, Young et al. 1994). Thus mechanism of leaf breakdown rates in agricultural streams may be more influenced by physical rather than biological mechanisms. However, Tank and Winterbourn (1996) found increased sedimentation to slow microbial degradation of leaf material.

Lower shredder abundance and density in heavy agricultural streams may also be in response to reduction in the quantity and diversity of riparian vegetation (Benfield et al. 1977). Annual leaf inputs to study streams ranged from 3 to 344 g AFDM/m<sup>2</sup>/y but were only significantly reduced in heavy agricultural streams (Chapter II). In comparison to the riparian vegetation in forested, light agricultural, and moderate agricultural streams, which was composed of a greater variety of tree species, riparian vegetation along heavy agricultural streams was limited to a few trees and tree species. Therefore, the reduction of food quantity may explain the lower number of shredders in the heavy agricultural streams.

Scraper abundance and density were not significantly different among land use types. More scraping macroinvertebrates were expected in agriculturally impacted streams due to higher light inputs, which increase food availability for scraping invertebrates (Feminella et al. 1989, Reed et al. 1994). However, chlorophyll *a* and microbial biomass measurements were not significantly related to land use (Chapter II) possibly explaining why scraper abundance and density did not increase with increases in agricultural land use. Scrapers recorded in forested stream were primarily habrophlebiid and heptageniid mayflies and elmids (Appendix B). Strand and Merritt (1999) also noted an increase in heptageniid mayflies in a forested versus pasture reach.

While overall macroinvertebrate and shredder abundance, density, and richness were lowest in heavy agricultural streams, these values may be artificially elevated due to the method used to assess the macroinvertebrate community. Leaf packs placed in agricultural streams may attract colonizing macroinvertebrates by providing a more palatable food source and more suitable habitat than what was currently present. And while macroinvertebrate abundance, density, and richness estimates may be accurate for the leaf packs, they may be an overestimation of the macroinvertebrate community at the reach scale (Webster and Waide 1982, Tuchman and King 1993). Also, leaf packs probably underestimated scraper abundance because they provide optimal food and habitat resource for shredding macroinvertebrates.

### ***Breakdown as a measure of stream integrity***

The basis for using leaf breakdown to assess stream integrity focuses on the link between stream structure and function (e.g., biological structure affects ecosystem processes and ecosystem processes influence biological structure). This study shows that several aspects of stream structure were disturbed by agricultural land use including temperature, dissolved oxygen, nutrient concentrations, and the macroinvertebrate community. Furthermore, changes in stream structure influenced leaf breakdown rates. But note that Angermeier and Karr (1994) found that process (leaf breakdown) rates did not change in response to shifts in biotic elements (species diversity). In order for leaf breakdown to serve as a successful indicator of stream integrity it must respond uniformly to a given stress (Gessner and Chauvet 2002). However, Agricultural land use can have confounding effects on leaf breakdown. In the current study, higher nutrients and temperatures increased breakdown rates in light and moderate agricultural streams, while lack of shredders in addition to higher sedimentation probably slowed breakdown in heavy agricultural streams despite higher nutrients and temperatures. Huryn et al. (2002) found that rates leaf breakdown did not differ among land use types attributing this to the multitude of confounding factors that influence breakdown rates. In spite of this, several studies suggest using leaf breakdown as an indicator of stream integrity.

Gessner and Chauvet (2002) provided a method for assessing stream integrity using breakdown rates, where breakdown rates above or below the rates measured in pristine sites receive a lower score. This method has been applied to studies that showed faster breakdown rates in agriculturally impacted streams and accurately depicted those streams as having “poor” stream integrity (Niyogi et al. 2003). In the current study heavy agricultural streams had similar

breakdown rates as the forested (pristine) streams. Therefore, using Gessner and Chauvet's method to calculate integrity scores, light and moderate agricultural sites received lower scores than heavy agricultural streams suggesting that stream function in heavy agricultural streams were the least impacted of land use types assessed. Niyogi et al. (2003) also reported similar integrity scores for an agricultural stream and attributed it to sedimentation. While overall the positive effects of nutrients tended to outweigh the negative effects of sedimentation on tussock grass breakdown along a gradient of agricultural development, Niyogi et al. (2003) stressed that caution was necessary when using breakdown rates to assess stream integrity. In the current study, high nutrient concentration and temperature were not negated by sedimentation in the light and moderate agricultural stream. Also, the shredding macroinvertebrate community was not as impaired in light and moderate agricultural streams as it was in the heavy agricultural streams. Thus, light and moderate agricultural leaf breakdown rates were faster than in forested streams suggesting lower stream integrity. The effects of higher nutrient concentration and temperature in heavy agricultural streams were negated by sedimentation and lack of shredding macroinvertebrates. Therefore, heavy agricultural land use seemed to have no effect on leaf breakdown or stream integrity. Thus, I suggest that leaf breakdown is not an accurate indicator of stream integrity along a gradient of agricultural land use.

Furthermore, while several studies have shown higher leaf breakdown rates in response to agricultural land use (Young et al. 1994), these studies are limited in that they compare reference stream to one type of agricultural land use rather than assessing the gradient of land use from forest to agriculture (Benfield et al. 1977, Dance and Hynes 1980, Tuchman and King 1993). By examining the gradient from forest to heavy agricultural land use, the complex influence of agriculture on leaf breakdown emerges. Agricultural land use does not simply result in elevated levels of leaf breakdown. Rather this study suggests that the effects of agricultural land use (e.g., altered nutrient concentration, temperature, sedimentation, and macroinvertebrate assemblage) have both positive and negative effects on leaf breakdown resulting in streams somewhat effected by agricultural land use (light and moderate agriculture) exhibiting faster breakdown rates than forested or heavy agricultural streams.

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**Appendix A.** Riparian tree importance values for trees > 4 cm DBH. Importance values were calculated by summing relative density, relative cover, and relative frequency for a possible score of 300. Species are ranked by importance values. Stream abbreviations are given in Table 1.

Tree species	Forest			Light agriculture			Moderate agriculture		Heavy agriculture			
	BAL	HWC	JON	DRY	ELL	TES	SHO	SUT	CAL	HOG	PAY	SKE
Alder <i>Alnus serrulata</i>				29.3		11.5		104.5	40.3			244.2
Rhododendron <i>Rhododendron maximum</i>	76.9	165.4	29.2		6.8		34.0					
Black walnut <i>Juglans nigra</i>				33.1	9.5	10.3		15.0	64.8		136.2	
Sycamore <i>Platanus occidentalis</i>			36.0	42.1	10.5	30.3	17.4	12.9	108.6			
Black cherry <i>Prunus serotina</i>				14.8		46.1			86.2		93.2	
Red maple <i>Acer rubrum</i>	5.2		13.9	28.3	9.0	26.3	83.0	53.6				
Sweet birch <i>Betula lenta</i>	36.8	53.3	80.3		21.3							
Eastern redcedar <i>Juniperus virginiana</i>				9.9							70.6	30.2
Eastern hemlock <i>Tsuga canadensis</i>	64.4	30.1	33.4									
Yellow-poplar <i>Liriodendron tulipifera</i>	48.0	20.8	12.8	31.8	57.8	64.1	20.5	25.7				
Eastern white pine <i>Pinus strobus</i>					2.3		55.5					
Flowering dogwood <i>Cornus florida</i>			18.7	47.5	4.6	13.0	6.5					25.6
Black willow <i>Salix nigra</i>								43.2				
Yellow buckeye <i>Aesculus octandra</i>	3.4		15.3	40.7	30.3	38.1						
Russian-olive <i>Elaeagnus angustifolia</i>								37.6				
American hornbeam <i>Carpinus caroliniana</i>	4.3						33.1					
Black tupelo <i>Nyssa sylvatica</i>							32.4					
Northern red oak <i>Quercus rubra</i>	3.5	7.3	3.6		32.1							
Butternut <i>Juglans cinerea</i>					27.8	19.3		7.5				
Fringetree <i>Chionanthus virginicus</i>		1.9		22.5		6.1						

Tree species		Forest			Light agriculture			Moderate agriculture		Heavy agriculture			
		BAL	HWC	JON	DRY	ELL	TES	SHO	SUT	CAL	HOG	PAY	SKE
American beech	<i>Fagus grandifolia</i>			10.8		18.3							
Sourwood	<i>Oxydendrum arboreum</i>	14.7				5.8		2.9					
American basswood	<i>Tilia americana</i>	10.3		14.3		13.7	6.4						
Witch-hazel	<i>Hamamelis virginiana</i>	7.4	2.2	9.6		13.5							
Mountain-laurel	<i>Kalmia latifolia</i>	9.8				2.2		5.4					
Yellow birch	<i>Betula alleghaniensis</i>					9.7							
Common persimmon	<i>Diospyros virginiana</i>					4.4	9.2	9.4					
Sugar maple	<i>Acer saccharum</i>			8.9									
Pignut hickory	<i>Carya glabra</i>	1.7	9.6			7.7	3.2						
Bigleaf magnolia	<i>Magnolia macrophylla</i>		8.5										
Carolina ash	<i>Fraxinus caroliniana</i>	6.5											
Plum sp.	<i>Prunus sp.</i>						6.0						
Pumpkin ash	<i>Fraxinus profunda</i>			1.7		5.8							
Black locust	<i>Robinia pseudoacacia</i>	4.8	6.5			4.5	4.1						
Sassafras	<i>Sassafras albidum</i>						3.0						
White oak	<i>Quercus alba</i>	2.2		1.7			3.0						
Cockspur hawthorn	<i>Crataegus crus-galli</i>					2.3							
Mockernut hickory	<i>Carya tomentosa</i>			2.3									
Striped maple	<i>Acer pensylvanicum</i>			1.7									

**Appendix B.** Composition of macroinvertebrate fauna in each study stream. Values are mean number of macroinvertebrates from each taxa (FFG functional feeding group, PR = predators, CG = collector-gathers, SC = scrapers, CF = collector-filters, and SH = shredders). Stream abbreviations are given in Table 3.

Taxa	Forest			Light agriculture			Moderate agriculture		Heavy agriculture					
	FFG	BAL	HWC	JON	DRY	ELL	TES	SHO	SUT	CAL	HOG	PAY	SKE	
TURBELLARIA														
Planariidae					0.1	0.1			0.2			0.1		
OLIGOCHAETA	PR	1.0	0.8	1.3	4.2	1.1	2.8	1.1	6.6	0.3	11.9	1.4	0.8	
HIRUDINEA														
DECOPODA		0.1	0.1					0.1						
EPHEMEROPTERA														
Siphonuridae	<i>Isonychia</i>	CG					0.1		0.1					
Siphonuridae	<i>Siphonurus</i>	CG	0.1											
Baetidae	<i>Baetis</i>	CG				0.3								
Baetidae	<i>Cloeon</i>	CG			0.1		0.1							
Oligoneuriidae	<i>Homoeoneuria</i>	CF				0.4								
Heptageniidae		SC				0.2					0.1			
Heptageniidae	<i>Epeorus</i>	CG	0.2		0.1						0.1			
Heptageniidae	<i>Stenacron</i>	CG								2.0				
Heptageniidae	<i>Stenonema</i>	SC	0.2		0.3	1.2	0.6	0.6	1.0	1.5	5.8	0.5	1.2	1.0
Ephemerallidae	<i>Ephemerella</i>	CG	1.2	1.1	5.6	13.0	24.9	18.3	12.7	1.0	12.3	0.1	2.1	5.4
Tricorythidae	<i>Leptohyphes</i>	CG						0.1						
Baetiscidae	<i>Baetisca</i>	CG				0.2			0.1				0.1	
Leptophlebiidae	<i>Habrophlebia</i>	CG		0.3				0.1						
Leptophlebiidae	<i>Habrophlebiodes</i>	SC	1.5	1.3	1.6	1.8	1.3	2.8	2.3	0.8	0.1		0.1	
Leptophlebiidae	<i>Leptophlebia</i>	CG						0.2						
Ephemeridae	<i>Ephemera</i>	CG				0.1		0.5	0.3					

Taxa	Forest				Light agriculture			Moderate agriculture		Heavy agriculture			
	FFG	BAL	HWC	JON	DRY	ELL	TES	SHO	SUT	CAL	HOG	PAY	SKE
ODONATA													
Cordulegastridae	<i>Cordulegaster</i>	PR		0.2	0.2			0.2		0.1			
Gomphidae	<i>Erpetogomphus</i>	PR											0.2
Gomphidae	<i>Hagenius</i>	PR	0.2		0.1	0.2		0.3					
Gomphidae	<i>Lanthus</i>	PR		0.1					0.3				
Aeshnidae	<i>Basiaeschna</i>	PR							0.1			0.2	0.2
Libellulidae	<i>Libellula</i>	PR					0.1						
Agriidae	<i>Agrion</i>	PR							0.1	0.1	0.1		
PLECOPTERA													
Pteronarcidae	<i>Pteronarcys</i>	SH			0.3	0.3	0.6	0.5	0.1	0.1			
Peltoperlidae	<i>Peltoperla</i>	SH		0.1	0.1						0.3		0.2
Peltoperlidae	<i>Talloperla</i>	SH	9.5	4.3	2.5	1.8	2.0	0.4	1.1		0.8		0.3
Peltoperlidae	<i>Viehoperla</i>	SH		0.1	1.3	0.4	3.3				0.4		0.3
Taeniopterygidae	<i>Taeniopteryx</i>	SH	0.1		0.3		0.3	0.3			0.3		0.1
Taeniopterygidae	<i>Strophopteryx</i>	SC	0.5			0.3		0.8			0.5		
Nemouridae	<i>Amphinemura</i>	SH	1.8	0.8	0.1	0.3		0.3	0.2	0.3	0.1		
Nemouridae	<i>Paranemoura</i>	SH					0.2	0.7					0.2
Nemouridae	<i>Soyedina</i>	SH			0.6	0.1							
Leuctridae	<i>Leuctra</i>	SH	3.6	0.3	0.8	0.2	0.6	3.6	1.1	0.6	0.1		
Capniidae	<i>Allocapnia</i>	SH	0.1	0.1	0.2								
Capniidae	<i>Paracapnia</i>	SH			0.3								
Perlidae		PR					0.1						
Perlidae	<i>Beloneuria</i>	PR							0.1		0.1		
Perlidae	<i>Anacroneuria</i>	PR					1.8						
Perlidae	<i>Perlinella</i>	PR	1.2	0.1									

Taxa	Forest			Light agriculture			Moderate agriculture		Heavy agriculture					
	FFG	BAL	HWC	JON	DRY	ELL	TES	SHO	SUT	CAL	HOG	PAY	SKE	
PLECOPTERA														
Perlodidae													0.1	
Perlodidae	<i>Isogenoides</i>						3.1						0.3	
Perlodidae	<i>Remenus</i>											0.3	0.3	
Perlodidae	<i>Isoperla</i>	PR	12.3	1.3	3.4	1.0	7.3	2.8	2.3	0.2	7.2	0.6	5.7	
Perlodidae	<i>Oconoperla</i>					9.4		7.5	3.6	0.3	3.7	1.1		
Chloroperlidae	<i>Alloperla</i>	PR	10.8	4.2	3.3	11.6	8.4	2.8	12.5	0.6	0.3	1.1	0.2	0.3
Chloroperlidae	<i>Haploperla</i>	PR	0.1											
Chloroperlidae	<i>Sweltsa</i>	PR					0.4							
MEGALOPTERA														
Corydalidae	<i>Corydalus</i>	PR									0.1			
Corydalidae	<i>Chauliodes</i>	PR							0.1	0.1			0.3	
TRICHOPTERA														
Philopotamidae	<i>Chimarra</i>	CF					0.2							
Psychomyiidae	<i>Psychomyia</i>	CG					0.1							
Polycentropodidae	<i>Neureclipsis</i>	CF									0.2			
Polycentropodidae	<i>Polycentropus</i>	CF	1.0	0.9	0.2		0.3	0.4	0.3		0.5	0.1	0.1	
Hydropsychidae	<i>Cheumatopsyche</i>	CF	2.1	2.9	0.7	1.9	3.0	1.4	1.5	2.2	14.2	1.1	1.3	2.8
Hydropsychidae	<i>Hydropsyche</i>	CF								0.4	1.3			
Rhyacophilidae	<i>Rhyacophila</i>	PR			0.2	2.0			0.3					
Glossosomatidae	<i>Glossosoma</i>	SC					0.2	0.2			0.3			
Phryganeidae	<i>Oligostomis</i>	PR						0.2						
Brachycentridae	<i>Brachycentrus</i>	CF				0.7	0.2		0.2		0.6			
Brachycentridae	<i>Micrasema</i>	SH				0.4		1.5	0.1					
Lepidostomatidae	<i>Lepidostoma</i>	SH	1.2	2.6	2.5	2.9	2.0	3.5	6.2	0.3	0.7	0.1	0.2	0.3

Taxa	Forest				Light agriculture			Moderate agriculture		Heavy agriculture				
	FFG	BAL	HWC	JON	DRY	ELL	TES	SHO	SUT	CAL	HOG	PAY	SKE	
TRICHOPTERA														
Lepidostomatidae	<i>Theliopsyche</i>	SH					0.1							
Limnephilidae	<i>Pychnopsyche</i>	SH	1.2	0.8	0.6	0.8	0.2	3.1	1.4	0.2		0.3	0.3	
Goeridae	<i>Goerita</i>	SC		0.1	0.1									
Beraeidae	<i>Beraea</i>	CG		0.4		0.2								
COLEOPTERA														
Haliplidae	<i>Haliphus</i>				0.1							0.1		
Hydrophilidae	<i>Laccobius</i>	PR					0.1					0.1		
Psephenidae	<i>Psephenus</i>	SC					0.1	0.1						
Elmidae	<i>sp. A</i>	SC				0.5		0.4				0.3		
Elmidae	<i>sp. B</i>	SC	0.6	0.3	0.3	1.9	1.4	0.8	0.5	5.5	3.8	0.5	6.3	1.1
Curculionidae	<i>Phytobius</i>	SH			0.1									
DIPTERA														
Tipulidae	<i>Tipula</i>	SH		0.3	0.3	0.9	0.8	2.1	1.0	1.0	1.6		0.3	0.1
Tipulidae	<i>Antocha</i>	SH	0.4					0.5						
Tipulidae	<i>Eriocera</i>	SH	0.5	0.4		0.3	0.2	0.2	0.2					
Simuliidae	<i>Simulium</i>	CF			0.1	0.3	0.1	0.3		0.2	0.1			
Chironomidae		CG	12.1	9.8	12.2	23.3	6.7	24.3	11.5	19.6	13.7	15.8	16.6	17.7
Stratiomyidae	<i>Stratiomys</i>	CG		0.2	0.1	0.3	0.5	0.1			0.3			0.1
Tabanidae	<i>Chrysops</i>	PR			0.1									
Athericidae	<i>Atherix</i>	PR	0.9	0.8	0.8	0.3	1.4	0.5	0.1	0.6	0.2			0.5
Sciomyzidae		PR									0.2			5



Taxa	Forest				Light agriculture			Moderate agriculture		Heavy agriculture			
	FFG	BAL	HWC	JON	DRY	ELL	TES	SHO	SUT	CAL	HOG	PAY	SKE
<b>GASTROPODA</b>													
Physidae	<i>Physa</i>	CG									1.9		0.4
Viviparidae		CG		5.3	0.9			0.7	0.7		6.1	8.5	0.5
Sphaeriidae	<i>Sphaerium</i>	CF											0.3

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**of**  
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**EDUCATION**

**Master of Science in Biology, May 2004**

Virginia Polytechnic Institute and State University, Blacksburg, Virginia

Thesis Title: Influence of agricultural land use on allochthonous input and leaf breakdown in southern Appalachian streams.

**Bachelor of Arts, graduated Magna Cum Laude, May 2001**

Wittenberg University, Springfield, Ohio

Major: Biology, with Honors

Minors: Chemistry and Environmental Studies

**ACADEMIC HONORS**

Best Student Presentation Award, Virginia Water Research Annual Meeting. 2003

Phi Beta Kappa – National Honor Society. 2001

Wittenberg University Scholar Award (4 time recipient). 1997-2001

Wittenberg University Dean's List (7 semesters). 1997-2001

Beta Beta Beta – Biology Honor Society. 1999-2001

Beta Beta Beta – Outstanding Achievement Award. 2001

Biology Faculty Award (two time recipient), Wittenberg University. 1999 and 2001

Emmet Bodenberg Award – For outstanding work in the environmental field. 2000

Omicron Delta Kappa – National Honor Society. 2000

**RESEARCH EXPERIENCE**

*Graduate Research*, Studied the influence of agricultural land use on organic matter dynamics in southern Appalachian streams. Master's research, Virginia Polytechnic Institute and State University. Dr. J. R. Webster, 2001-2004.

**Graduate Research Assistant**, Virginia Polytechnic Institute and State University.  
*Leaf Litter Input* - Measured leaf litter input to streams affected by the hemlock woolly adelgid, an insect pest species. Dr. J. R. Webster, 2003-2004.

*Lotic Intersite Nitrogen eXperiment (LINX II)* - Studied nitrate uptake and retention in streams impacted by human disturbance. Drs. J. R. Webster and H. M. Valett, 2003-2004.

*Litter Exclusion* - Studied the effects of leaf litter exclusion and wood removal on phosphorus and nitrogen retention in a first-order forest stream. Dr. J. R. Webster, 2001-2004.

**Undergraduate Research**, Investigated the effects of habitat and microclimate conditions on bat species presence, relative abundance, and activity in the Huron National Forest, MI. using the Anabat II bat detector. Huron National Forest, Dr. T. Lewis and P. Huber. 2000.

**Undergraduate Research**, Studied the effects of dams on macroinvertebrate assemblages along Buck Creek River, Clark County, OH. Wittenberg University. Dr. H. H. Hobbs III. 2000.

**Undergraduate Research**, Completed a yearlong study of the biological and physicochemical parameters of Freeland's Cave, Adams County, OH. Wittenberg University, Dr. H. H. Hobbs III. 1998-2000.

**Undergraduate Research Assistant**, Baseline study of physicochemical characteristics of five inland blue holes on North Andros Island, Bahamas. Wittenberg University, Dr. H. H. Hobbs III. 1998.

## RELEVANT PROFESSIONAL EXPERIENCE

**Graduate Teaching Assistant**, Virginia Polytechnic Institute and State University, Blacksburg, VA.

*Ecosystem Dynamics (5034)* Fall 2003

*Field and Lab Ecology (3114)* Spring 2003 and 2004

*Principles Biology Laboratory (1115 and 1116)* Spring 2002 and Fall 2002

*General Biology Laboratory (1015 and 1016)* Fall 2001, Spring 2002, and Fall 2002

**Naturalist**, Glen Helen Nature Preserve, Yellow Springs, OH. 2001.  
Worked with children age 5-14 teaching environmental education.

**Undergraduate Employee**, Wittenberg University Biology Department, Springfield, OH. 1999-2001. Conducted projects related to caves and karst in Ohio and Kentucky, prepared labs for Limnology Laboratory, sorted and identified fish and macroinvertebrate collections, and maintained aquatic ecology research database.

***National Forest Service Internship***, Huron National Forest, Mio, MI. 2000.

Focused efforts on the Kirtland warbler, an endangered species. Provided daily bird tours, participated in yearly census, and Recovery Team. Gained experience in electroshocking, radio telemetry, and bat mist-netting techniques. Conducted skipper surveys, plant surveys, and bear-bait indexes, in addition to other wildlife fieldwork.

***Fair Housing Assistant***, City of Springfield Fair Housing Department, Springfield, OH.

1997-1999. Revised the *Affordable Housing Handbook*, mediated neighbor disputes, developed and implemented surveys, wrote articles for the *Doing the Right Thing* column in the *Springfield-News Sun* newspaper, and organized *Everybody Counts Day*, which encouraged awareness of persons with disabilities to elementary aged children.

***Chemistry Internship***, Ohio State University Chemistry Department, Columbus, OH.

1996-1997. Lab technician in organic chemistry lab. Research focused on synthetic and mechanistic organic chemistry.

#### **ABSTRACTS, PRESENTATIONS, POSTERS**

Hagen, Elizabeth M. and Jackson R. Webster. Allochthonous input in southern Appalachian streams impacted by agricultural land use. Virginia Lakes and Watersheds Association: Virginia Water Conference, Virginia Beach, VA. March 2004.

Hagen, Elizabeth M. and J. R. Webster. Effects of agricultural disturbance on autumn allochthonous input to southern Appalachian streams. Virginia Water Research Symposium, Blacksburg, VA. October 2003.

Hagen, Elizabeth M. and J. R. Webster. Poster: Effects of agricultural disturbance on organic matter dynamics in southern Appalachian streams. 2003 LTER All Scientist Meeting, Seattle, WA. September 2003.

Hagen, Elizabeth M. and J. R. Webster. Poster: Effects of agricultural disturbance on allochthonous input in southern Appalachian streams. Coweeta LTER Summer 2003 Science Meeting, Franklin, NC. June 2003.

Hagen, Elizabeth M. and J. R. Webster. Poster: Effects of agricultural disturbance on allochthonous input in southern Appalachian streams. Annual Meeting of the North American Benthological Society, Athens, GA. May 2003.

Hagen, Elizabeth M. and Phillip W. Huber. Species diversity and habitat use of bats in the Huron National Forests, MI. National Conference on Undergraduate Research, Lexington, KY. March 2001.

Hagen, Elizabeth M. Poster: A biological and physicochemical assessment of Freeland's Cave, Adams County, Ohio (1998-1999). Ohio Academy of Sciences Annual Meeting, Ohio Northern University, Ada, OH. April 2000.

## **PUBLICATIONS**

Hagen, Elizabeth M. and Kathryn A. Gogolin. 2001. A biological and physicochemical assessment of Freeland's Cave, Adams County, Ohio (1998-1999). *Pholeos* 19:2-15.

Athy, Erin, Kate Greico, and Beth Hagen. 2001. A comparison of salamander populations in Cobble Crawl and Coon-in-the-Crack I, Carter Co., KY. *Pholeos* 19:16-21.

Hagen, Elizabeth M. and Jackson. R. Webster. 2004. Effects of agricultural disturbance on autumn allochthonous input in southern Appalachian streams. *Virginia Water Research Symposium 2003: Proceedings Water Resource Management for the Commonwealth* 191-195.

## **GRANTS**

Travel Fund Project, Graduate Student Assembly, Virginia Polytechnic Institute and State University, \$411 awarded for travel to North American Benthological Society Annual Meeting in Vancouver, B.C. 2004.

Leo Bourassa Scholarship, Virginia Lakes and Watersheds Association, \$2700 awarded in support and acknowledgement of academic and personal accomplishments in the field of water resources. 2003.

Graduate Research Development Project, Graduate Student Assembly, Virginia Polytechnic Institute and State University, \$300 awarded for proposal entitled "Effects of agricultural disturbance on leaf litter input and primary production in southern Appalachian streams." 2003.

Faculty Research Fund Board, Wittenberg University, \$2500 awarded for proposal entitled "Species diversity and habitat use of bats in the Huron National Forests, MI." 2000.

Faculty Research Fund Board, Wittenberg University, \$2500 awarded for proposal entitled "A biological and physicochemical assessment of Freeland's Cave, Adams County, Ohio (1998-1999)." 1999.

Faculty Research Fund Board, Wittenberg University, \$250 awarded for proposal entitled "A biological and physicochemical assessment of Freeland's Cave, Adams County, Ohio (1998-1999)." 1999.

## **PROFESSIONAL MEMBERSHIPS**

North American Benthological Society (Graduate Resources Committee 2003), member since 2002.

Biology Graduate Student Assembly, Virginia Polytechnic Institute and State University (Secretary 2003, Seminar Committee 2002-2003), member since 2001.

Ohio Academy of Sciences, member since 1999.

National Speleological Society, member since 1992.

Wittenberg University Speleological Society (President 1999-2001, Vice President 1998-1999), member since 1997.

Virginia Lakes and Watersheds Association, member since 2003.

National Ground Water Association, member since 2004.

Virginia Museum of Natural History Save Our Streams Program, certified water monitor since 2002.