

**Livestock Exclusion Effects on the Structure and Function of
Headwater Streams**

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THESIS

Submitted to the Faculty of
Virginia Polytechnic Institute and State University
in partial fulfillment of the requirements
for the degree of
Master of Science
in
Biology

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July 18, 2003
Blacksburg, Virginia

Keywords: Stream, Livestock, CREP, structure, function

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By

Kenneth Reid Cook

Abstract

The structure and function of headwater streams was evaluated in response to livestock exclusion implemented through Virginia's Conservation Reserve Enhancement Program (CREP). We measured riparian plant characteristics along with stream organic matter dynamics, benthic macroinvertebrates, algal biomass, and leaf breakdown in non-fenced, fenced, and forested streams. Increased growth of non-woody vegetation and the pre-existence of woody plants had a significant influence on stream organic matter dynamics. Tree basal area in a 20 m wide riparian corridor was predictive of stream coarse benthic organic matter standing stocks. Higher benthic organic matter standing stocks and differences in algal biomass in fenced and forested sites indicate different food resources may be structuring macroinvertebrate communities in these systems. We found a significant relationship between coarse benthic organic matter and percent shredder density, and scraper density generally followed patterns of algal biomass among treatments. Leaf breakdown rates among treatments were not indicative of differences in shredder density with two of the three fenced sites having the fastest overall breakdown rates observed. We attributed faster breakdown rates in these streams to available food

resources and shredder community structure existing prior to the implementation of livestock exclusion.

Our results suggest that a certain amount of ecological recovery may be possible through livestock exclusion. Macroinvertebrate structure in our study streams was primarily influenced by the presence or absence of riparian trees. Maturation and successional changes in woody riparian vegetation after livestock exclusion may allow certain characteristics of pastoral streams to return to those found in forested reaches.

Acknowledgements

I would like to thank Dr. E.F. Benfield for all of his help and guidance throughout this project. I would also like to thank Dr. J.R. Webster and Dr. R.H. Jones for their help with various aspects of sampling and data analysis. I would also like to thank all the individual landowners that allowed me to work on their property and the National Park Service, Blue Ridge Parkway for allowing access to streams. This project could not have been completed without the entire Virginia Tech Stream Team. I would like to thank the countless undergraduate workers who helped with sample processing. Above all I would like to thank my wife Kerrie, she has been the most valuable asset in my life and this project.

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Introduction

Livestock activities can have significant impact on stream community structure and ecosystem function through decreased riparian vegetation, geo-morphological changes, nutrient loading, and macroinvertebrate habitat loss (Kauffman and Krueger, 1984; Fleischner, 1994; Quinn et al., 1997). Livestock exclusion has been shown to result in the re-growth of riparian vegetation and to improve degraded water quality associated with grazing (Platts et al., 1984; Barling and Moore, 1994; Strand and Merritt, 1999; Scrimgeour and Kendall, 2003). However, little attention has specifically focused on the effects of livestock exclusion on stream structure and function.

A terrestrial landscape perspective is important in understanding the structure and function of streams (Hynes, 1975). Riparian zones incorporate both terrestrial and stream properties to create three-dimensional ecotones of direct interaction between the two environments (Gregory et al., 1991; Swanson et al., 1992). Low order streams with intact riparian vegetation are especially linked to the terrestrial environment and can be energetically dependent upon allochthonous inputs (Fisher and Likens, 1973; Vannote et al., 1980). Livestock grazing has been shown reduce vegetation biomass in riparian zones (Schulz and Leininger 1990), leading to loss of terrestrial organic matter input (Campbell et al., 1992) and increased dependency on autotrophic production in streams (Feminella, 1989; Winterbourn, 1990). Changes in the energetic resources of streams, due to shifts away from allochthonous inputs, can have significant effect on macroinvertebrates, and community structure can be regulated by autotrophic production

in these systems (Wallace et al., 1999; Delong and Brusven, 1998; Townsend et al., 1997).

Ecosystem processes, such as organic matter breakdown in streams, have been closely linked to macroinvertebrates and alteration in shredder abundances can affect leaf litter breakdown rates (Benfield et al., 1977; Kirby et al., 1983). Understanding changes in both structure and function may be important in evaluating the overall condition of streams under altered conditions (Schultheis et al., 1997; Nelson 2000).

Agricultural landscapes are common historical features in rural areas of the southern Appalachian Mountains (SAMAB, 1996). Due to the rugged topography of the landscape and high soil erosion rates, row crop production is limited, but the land has been amenable to livestock production. Land management programs have favored riparian zone restoration to mitigate the adverse effects of livestock grazing. Virginia's Conservation Reserve Enhancement Program (CREP) has implemented livestock exclusion and establishment of forested riparian corridors by enrolling streams and land in long-term contracts (10-15 years). In exchange, landowners are given assistance in fencing, tree planting, offsite watering, and annual payments for land taken out of use (FSA, 2000).

The objectives of this study were to evaluate the effects of livestock exclusion and changes in riparian vegetation on the structure and function of headwater streams. The effects of exclusion were assessed using nine streams in three stream treatments: fenced (FEN 1-3), non-fenced (NF 1-3), and forested (FOR 1-3). We quantified both riparian and stream properties in order to examine the connection between riparian vegetation differences and stream organic matter, macroinvertebrate community structure, algal

biomass, and leaf breakdown. We predicted differences in riparian vegetation due to livestock exclusions would influence stream structure and function and that each category would have unique properties along an ecological gradient from non-fenced to fenced to forested.

Methods

Site description

Study streams were located in the southern Appalachian region of southwest Virginia, in Floyd, Carroll, and Patrick counties, U.S.A. (Figure 1). The area lies within the Blue Ridge physiographic province and is characterized by gneiss and schist formations with sandy to clay soils well suited to pasture development (SAMAB, 1996). Precipitation averages 114 cm per year in the area and average daily air temperature ranges from 1.5 °C to 23 °C (VASS, 1999). Agriculture and mixed deciduous forests generally typify land use in the region.

Riparian vegetation at forested sites was characterized by *Fraxinus* spp., *Betula* spp., *Rhododendrom maximum*, *Acer* spp., *Liriodendron tulipifera*, and *Hamamelis virginiana*. Fenced sites were characterized by a high biomass of non-woody vegetation and woody species including *Fraxinus* spp., *Acer* spp, *Alnus rugosa*, and *Lindera benzoin*, *Rosa multiflora*, and *Liriodendron tulipifera*. Vegetation in non-fenced sites was typical of pasture streams characterized by a low biomass of non-woody vegetation and woody species consisting of *Prunus serotina*, *Alnus rugosa*, and *Carpinus caroliniana*.

Cattle were present in low to moderate stocking rates at each of the three non-fenced sites and outside exclosures at fenced sites for at least half of the study period. Cattle had unabated access to streams within the non-fenced treatment. Fenced sites had complete exclusion of cattle, two years or less, from the stream except for a single crossing that allowed movement across exclosures.

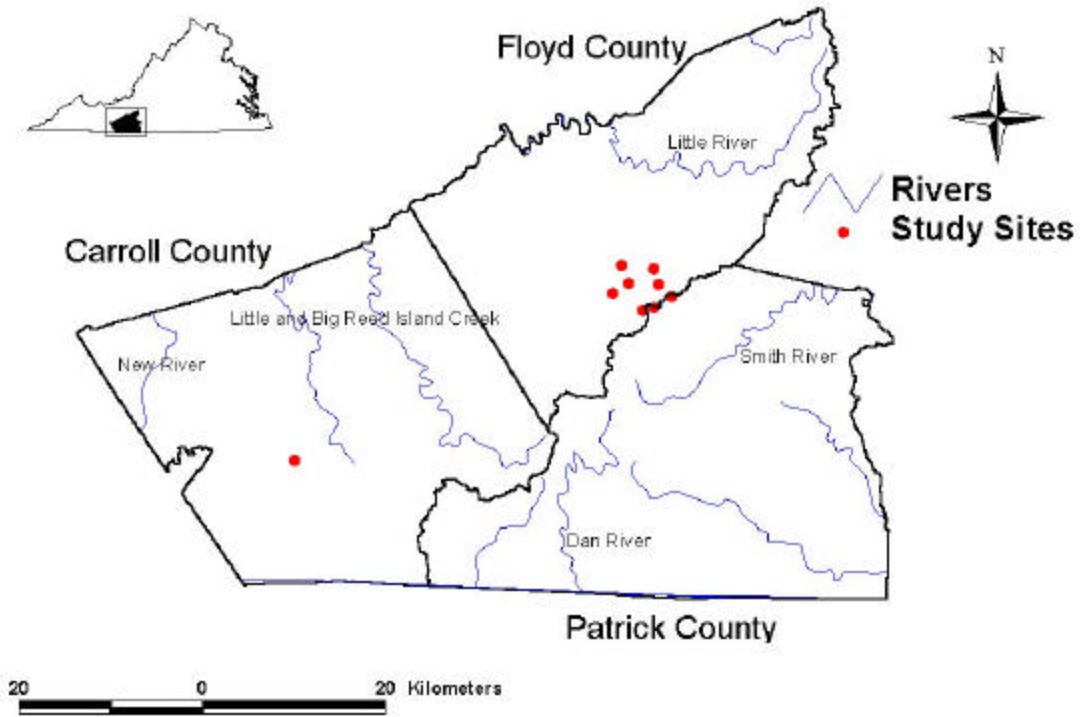


Figure 1: Location of study sites in southwest Virginia portion of the southern Appalachians.

Riparian Vegetation

Non-woody vegetation was sampled at fenced and non-fenced sites in August 2002, a period of peak plant biomass in the region. Twenty random 0.25 m² plots were sampled within two 20 x 50 m plots in the riparian area on each side of the stream. In each 0.25 m² plot, above ground biomass was removed and taken to the laboratory for further separation into grasses and forbs. Samples were dried at 50 °C and weighed to the nearest 0.1 g.

Woody vegetation was characterized at all sites over a 40 m reach at 10 m longitudinal intervals. At four lateral transects, tree diameter at breast height (DBH) was taken for all trees >1 cm DBH in 16 m² plots 5 and 10 m from the stream channel. Twenty total plots at each stream were sampled and trees in each were identified to the lowest possible taxonomic level. DBH values were used to calculate tree basal area in a 20 m riparian corridor along each side of the stream channel.

The amount of non-woody material that could potentially enter each fenced and non-fenced stream was measured from overhanging vegetation along the channel in fall 2002 by randomly placing an expandable grid over the wetted stream channel. All non-woody vegetation over the grid was clipped and dried at 50 °C and weighed to the nearest 0.1 g to estimate potential non-woody inputs during the study period.

Stream Chemical and Physical Properties

Replicate monthly water samples were taken from each stream, filtered, and frozen until analysis for NO₃-N. Nitrate concentrations were determined by colorimetric techniques on a Technicon Auto Analyzer II (Technicon Auto Analyzer II, 1973). Continuous data loggers (HOBO[®] Pocasset, MA) were used to measure water

temperature in all streams from April 2002 until May 2003, and light intensity was measured at one stream in a single location in each treatment from May to August 2002. Stream substrate was characterized by the pebble count method of Wohlman (1954) and was used to calculate median particle size and size class distribution including silt (<0.25 mm), sand (<2 mm), gravel (2-16 mm), pebble (16-64 mm), cobble (64-256 mm), and boulder (>256 mm) in each stream.

Stream Organic Matter

Stream organic matter was sampled in three size fractions including coarse benthic organic matter (CBOM, > 1mm), fine benthic organic matter (FBOM, >0.45 μm – 1 mm), and dissolved organic carbon (DOC, < 0.45 μm). CBOM and FBOM were collected seasonally from each stream beginning in spring 2002. Seasons in this experiment were defined as spring (March-May), summer (June-August), fall (September-November), and winter (December-February). Five replicate samples were taken each season using a 0.07 m² circular sampler placed in the stream substrate (Golladay et al., 1989). Large organic material such as leaves and sticks were removed, after which the substrate was disturbed and a volume of water from the sampler was pumped through a 1-mm mesh net into a 20-L bucket. Material greater than 1 mm was combined for estimation of CBOM standing stocks and replicate samples were taken from the bucket for FBOM. Samples were placed on ice and taken to the laboratory where CBOM samples were dried at 50 °C to a constant weight, ground to 420 μm , and ashed one hour at 550 °C for determination of ash free dry mass (AFDM). FBOM samples were sub-sampled and filtered through a 0.45- μm glass fiber filters, dried at 50 °C, and ashed one hour at 550 °C for AFDM.

Monthly water samples were collected for DOC concentration in each stream, taken to laboratory on ice, and were frozen until analysis. Concentration was determined by the wet oxidation method using an OI Model 700 Total Organic Carbon Analyzer (APHA, 1985). Monthly concentrations were combined for estimates of seasonal DOC at each site.

Benthic Macroinvertebrates

Macroinvertebrates were sampled in each stream in spring 2002 using a systematic random sampling technique modified from EPA rapid bioassessment protocols (EPA RBP, 1999). Eleven 0.09 m² surber samples were taken along a longitudinal transect of stream channel. At each randomly selected point, samples were taken from the left, middle, or right of the wetted stream channel. This method allows for more habitats and substrate types to be sampled compared to single habitat RBP procedures. The eleven samples were combined into an approximately 1 m² sample and preserved with 10 % formalin in the field and transferred to 80 % ethanol in the laboratory before processing. Macroinvertebrates were typically identified to genus except for Chironomidae, Crustacea, and Oligochaeta. Macroinvertebrates were also assigned to five functional feeding groups (FFG): shredder, scraper, predator, collector gatherer, and collector filterer (Stewart and Stark, 1993; Merritt and Cummins 1996).

Algal Biomass

Three to five rocks were collected monthly at each stream from May until November 2002 for determination of periphyton biomass by AFDM and chlorophyll *a* concentration (Steinman and Lamberti, 1996). Individual rocks were placed in plastic bags, immediately placed on ice, taken to the laboratory and frozen until processing. The

entire surface of collected rocks was then scraped and two sub-samples of the subsequent slurry were filtered using 0.45- μm glass fiber filters. One filter was dried at 50 °C and ashed one hour at 550 °C for periphyton AFDM analysis. The second filter was placed in 90 % basic acetone for 20 – 24 hours and the extractant was measured for chlorophyll *a* and phaeophytin concentration on a Shimadzu UV-1601 spectrophotometer (APHA, 1985).

Leaf Breakdown

A leaf breakdown study was conducted to evaluate the response of an ecosystem process to livestock exclusion. Red maple leaves were collected shortly after abscission, air-dried to a constant weight, and 5 g of leaves were placed in 5-mm mesh bags (Benfield, 1996). In December 2002, fifteen leaf packs were placed in each stream, after which three were removed immediately and processed to assess handling loss. Three bags were collected monthly, stored individually in plastic bags, and placed on ice until taken to the laboratory where they were washed to remove debris, dried at 50° C, ground to 420 μm , and ashed one hour at 550 °C to calculate AFDM remaining of leaf material. Breakdown rates (*k*) were calculated by regressing the natural log of percent remaining of leaf material against days of incubation in the stream (e.g., Peterson and Cummins 1974).

Statistical Analysis

Most analysis among treatments was conducted using a standard one-way analysis of variance (ANVOA) and Tukey's pair-wise comparisons. Sampling that was conducted either seasonally or monthly was tested using a two-way ANOVA with treatment and time as independent variables. When only two treatments were sampled, as in the case of non-woody vegetation, a *t*-test was used. Monthly concentrations were combined by

season to examine seasonal variations in both nitrate and DOC and were analyzed using one and two-way ANOVA. Relationships between individual parameters were examined using simple linear regression. Individual site breakdown rates were compared using general linear models with dummy variable and sequential Bonferroni adjustments to alpha values.

Results

Riparian Vegetation

Total biomass of non-woody vegetation was higher in the fenced treatment than in non-fenced ($p = 0.04$, Figure 2). Grasses made up the majority of overall non-woody biomass, comprising 78 and 81 % of the total biomass in fenced and non-fenced sites, respectively. Forb biomass was similar between treatments.

Tree basal area was significantly higher in forested riparian zones than in non-fenced ($p = 0.014$, Figure 3). Fenced sites averaged of 34 % less tree basal area in the 20 m corridor than in forested sites. Non-fenced sites had 96 and 92 % less mean basal area than forest and fenced sites, respectively. However, no significant difference in tree basal area was detected between fenced and non-fenced sites using one-way ANOVA. Tree basal area was highly variable within the fenced treatment ($CV = 0.64$).

Potential non-woody inputs were not significantly different between fenced and non-fenced treatments (Figure 2). Fenced sites were generally higher (mean = 135 ± 38 SE) than non-fenced (mean = 65 ± 32 SE) but site variability was high within both fenced and non-fenced treatments ($CV = 0.49$ and 0.87 , respectively).

Chemical and Physical Properties

Nitrate concentrations were highly variable throughout the year though annual nitrate concentrations were generally lowest at fenced sites and highest at non-fenced sites (Table 1). There was no significant difference among treatments, season, or month except for August, where non-fenced sites were significantly higher than fenced sites ($p = 0.04$). In general all treatments had the lowest concentrations over summer with

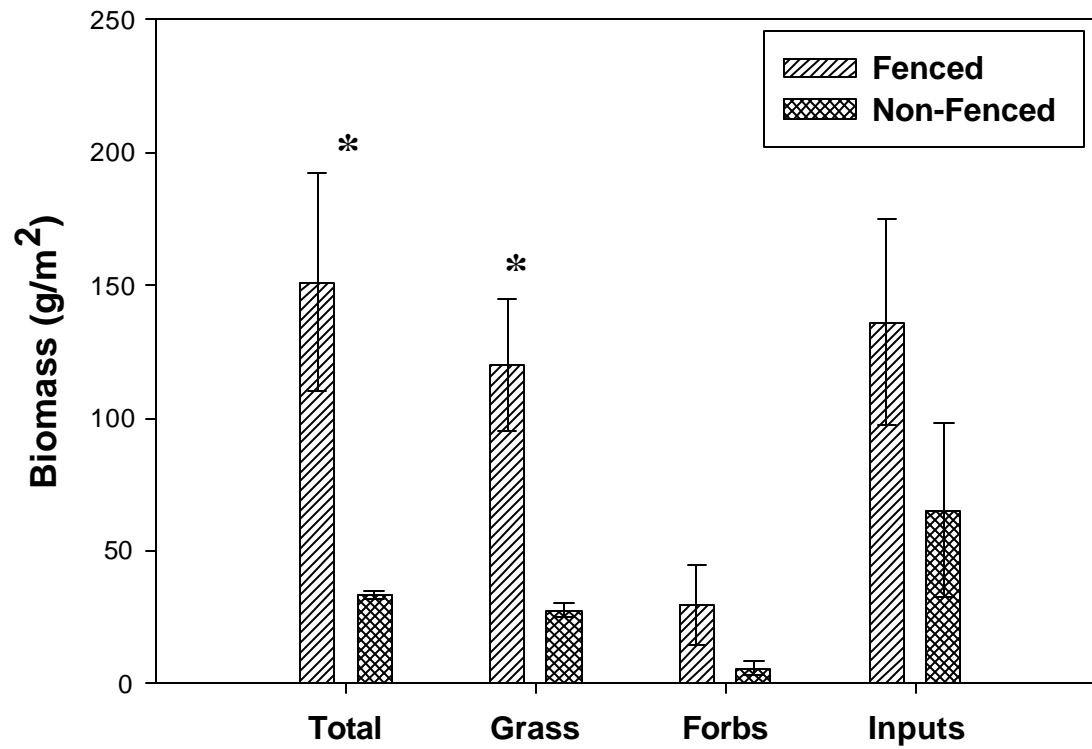


Figure 2: Mean riparian non-woody vegetation biomass and potential non-woody inputs in fenced and non-fenced treatments. Asterisks indicate significant difference between treatments ($p < 0.05$).

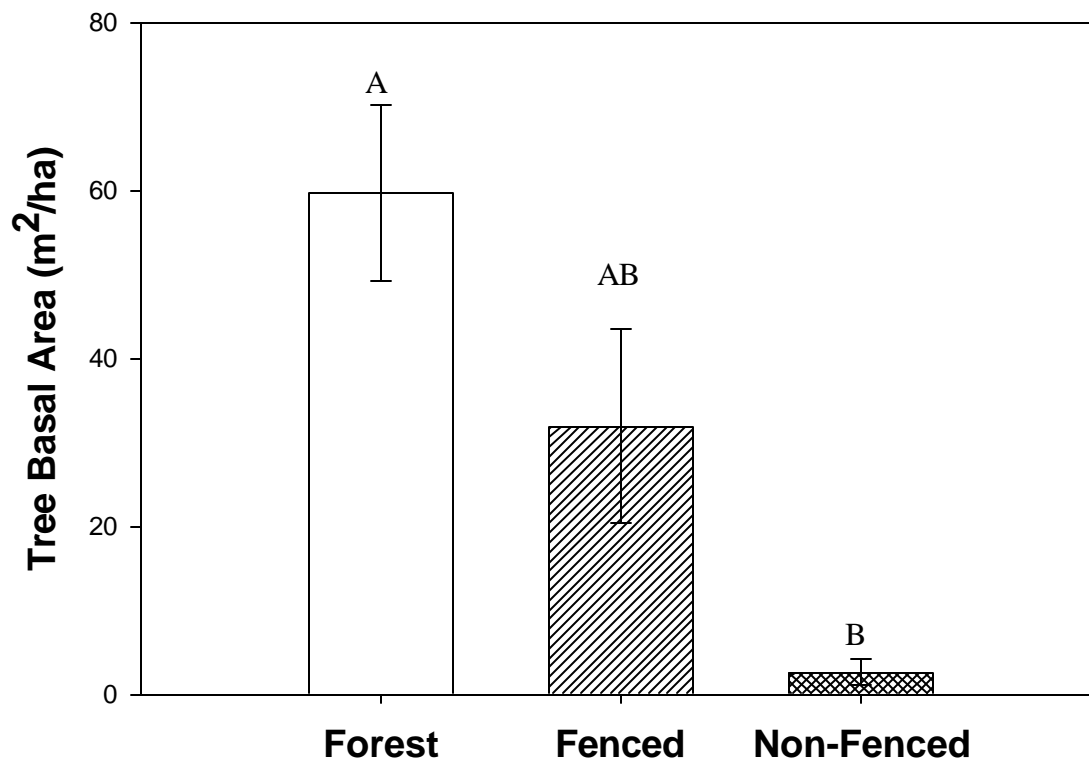


Figure 3: Mean tree basal area within a 10-m wide riparian corridor. Values with similar letters are not significantly different ($p > 0.05$).

Table 1: Physical and chemical properties of study streams given as mean \pm SE of annual nitrate concentrations ($\mu\text{g/L}$), median particle size (mm), % silt of substrate, mean annual temperature ($^{\circ}\text{C}$), and mean light intensity ($\log\text{Lum/m}^2$) from May until August. There was no significant difference among treatments in any category (ANOVA).

Treatment	Nitrate-N	Substrate Size	% Silt	Temperature	Light Intensity
Forest	349 ± 65	22 ± 5.3	0	10.30 ± 0.22	2.5 ± 0.21
Fenced	299 ± 164	19 ± 3	1.5 ± 1.5	11.17 ± 0.31	2.9 ± 0.19
Non-Fenced	415 ± 125	33 ± 6.6	14 ± 7.8	12.21 ± 0.31	3.3 ± 0.07

increases throughout the autumn, reaching highest concentrations in winter.

Substrate size was similar in all streams and was dominated by the pebble size class particles (Table 1). Non-fenced sites had a higher percent of silt compared to other treatments but sand size classes were similar at all streams.

Light intensity in each treatment decreased from April through August 2002. Fenced sites had a 33 % reduction in light intensity, forested had a 30 % reduction, and non-fenced sites had only an 8 % reduction during the period. Overall light intensity was lowest at forested sites, followed by fenced, and finally non-fenced received the highest overall light intensity during the growing season (Table 1).

Surface water temperature was similar among treatments, however, mean annual water temperature in non-fenced sites was 9 % higher than fenced sites and 22 % higher than forested sites (Table 1). On average non-fenced sites also had 720 more degree days than forested sites and 391 more than fenced sites.

Organic Matter

Seasonal and treatment level differences occurred in CBOM standing stocks over the course of this study (Figure 4). Mean annual standing stock ranged from 27 to 332 g/m² and both fenced ($p=0.02$) and forested ($p=0.003$) sites were significantly higher than non-fenced sites. Autumnal CBOM was significantly different among all treatments, winter and spring CBOM was different in forested and fenced sites, and summer CBOM was higher in the fenced treatment than in non-fenced. Two-way ANOVA showed that all three treatments were significantly different from one another

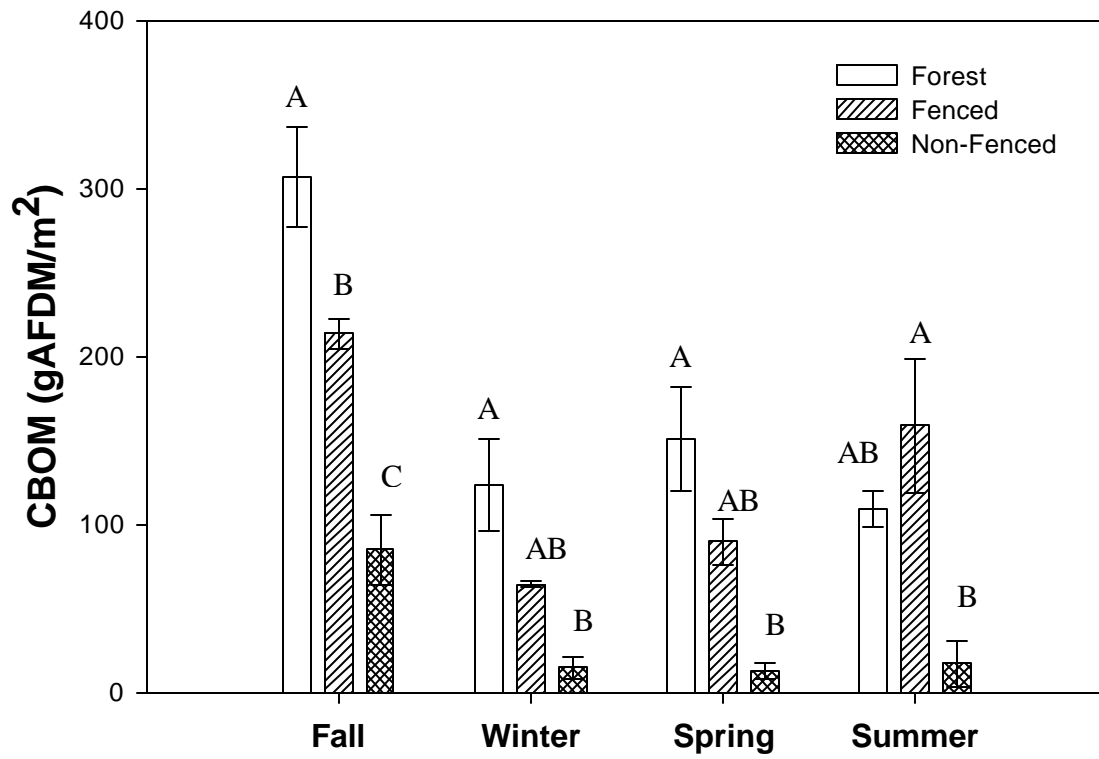


Figure 4: Mean seasonal CBOM standing stocks in each of the three treatments. Values with similar letter in each season are not significantly different ($p > 0.05$).

regardless of season. Relationships between terrestrial vegetation and CBOM standing stocks were assessed using simple linear regression and showed mean annual CBOM was predicted by tree basal area in the 20-m riparian corridor ($p = 0.007$, $r^2 = 0.68$, Figure 5).

Seasonal and treatment level differences also occurred in FBOM standing stocks over the duration of this study (Figure 6). Mean annual FBOM was significantly higher in the fenced treatment than in non-fenced ($p = <0.001$) and forested sites ($p = 0.004$). No significant differences were observed in fall or winter, however, spring FBOM in the fenced treatment was higher than non-fenced ($p = 0.02$) and summer FBOM was higher in the fenced treatment than in both non-fenced ($p = 0.004$) and forested ($p = 0.004$). Two-way ANOVA showed that fenced sites were significantly different from non-fenced ($p = <0.001$) and forested sites ($p = <0.001$) regardless of season. Linear regressions using winter CBOM as the independent variable and spring FBOM as the dependent variable showed a significant relationship in fenced and non-fenced sites. The same relationship was present using spring CBOM and summer FBOM as well ($p = 0.009$, $r^2 = 0.84$, Figure 7a, 7b). However, this relationship did not hold when forested sites were included in the analysis. There was also a significant relationship between the biomass of grasses in the riparian area and mean annual FBOM in the fenced and non-fenced sites ($p = 0.03$, $r^2 = 0.70$, Figure 7c).

Dissolved organic carbon concentrations varied throughout the study period and were generally highest in the fenced treatment (Figure 8). Monthly concentrations were similar except for July when fenced sites were higher than forested ($p = 0.005$),

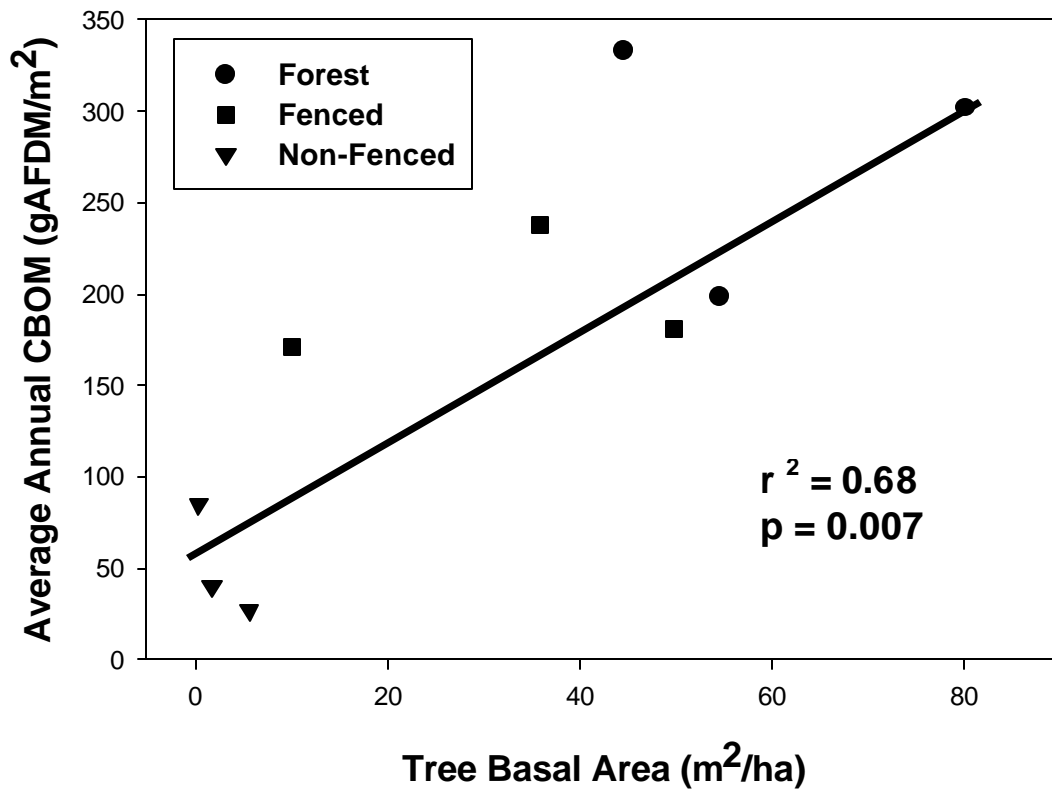


Figure 5: Relationship between mean annual CBOM standing stock and tree basal area in a 10 m wide riparian corridor.

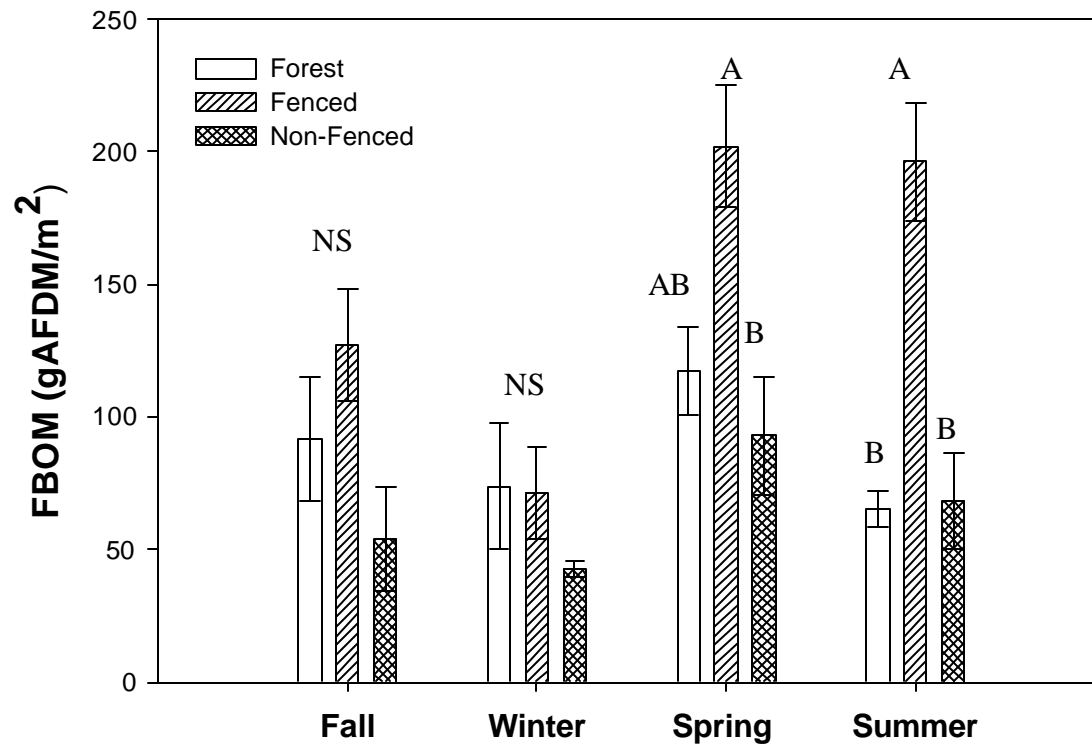


Figure 6: Mean seasonal FBOM standing stocks in each treatment. Values with similar letters in each season are not significantly different ($p > 0.05$) using a one-way ANOVA.

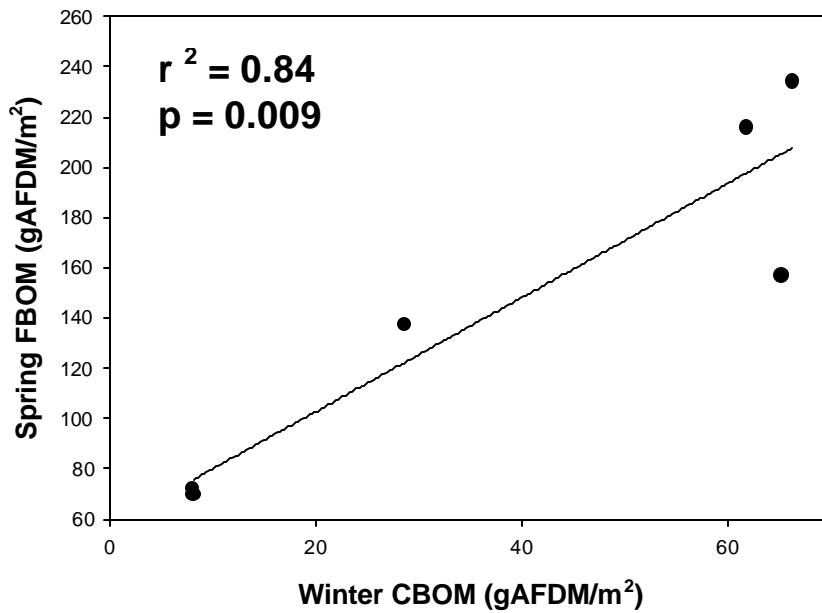


Figure 7a: Relationship between winter CBOM and spring FBOM standing stocks in fenced and non-fenced sites.

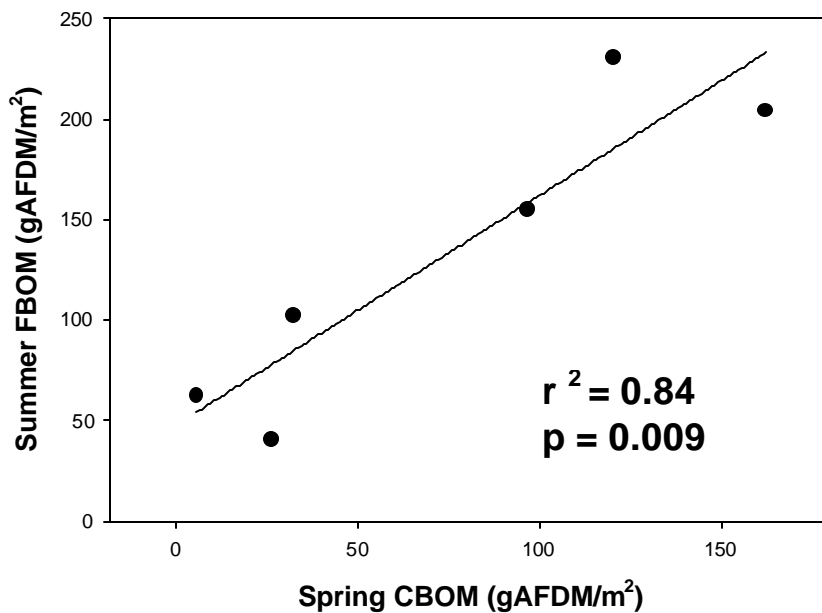


Figure 7b: Linear regression between summer FBOM and spring CBOM standing stocks in fenced and non-fenced sites.

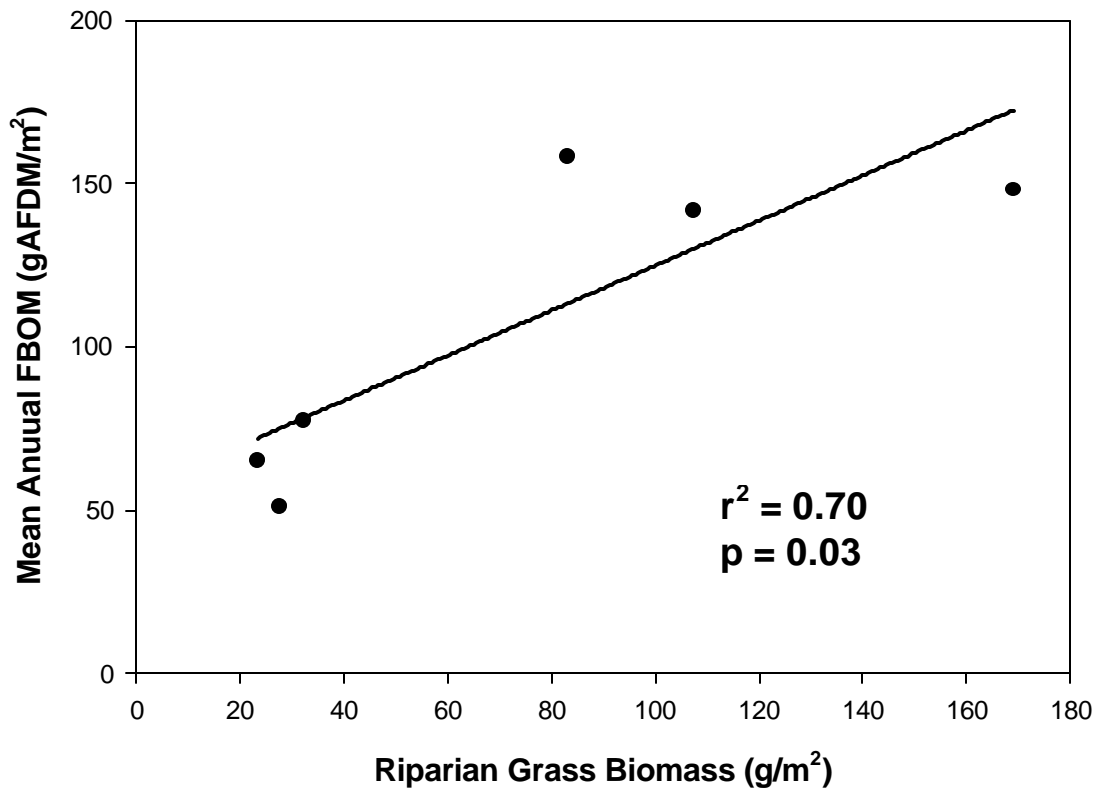


Figure 7c: Relationship between biomass of riparian grasses and mean annual FBOM standing stocks in fenced and non-fenced sites.

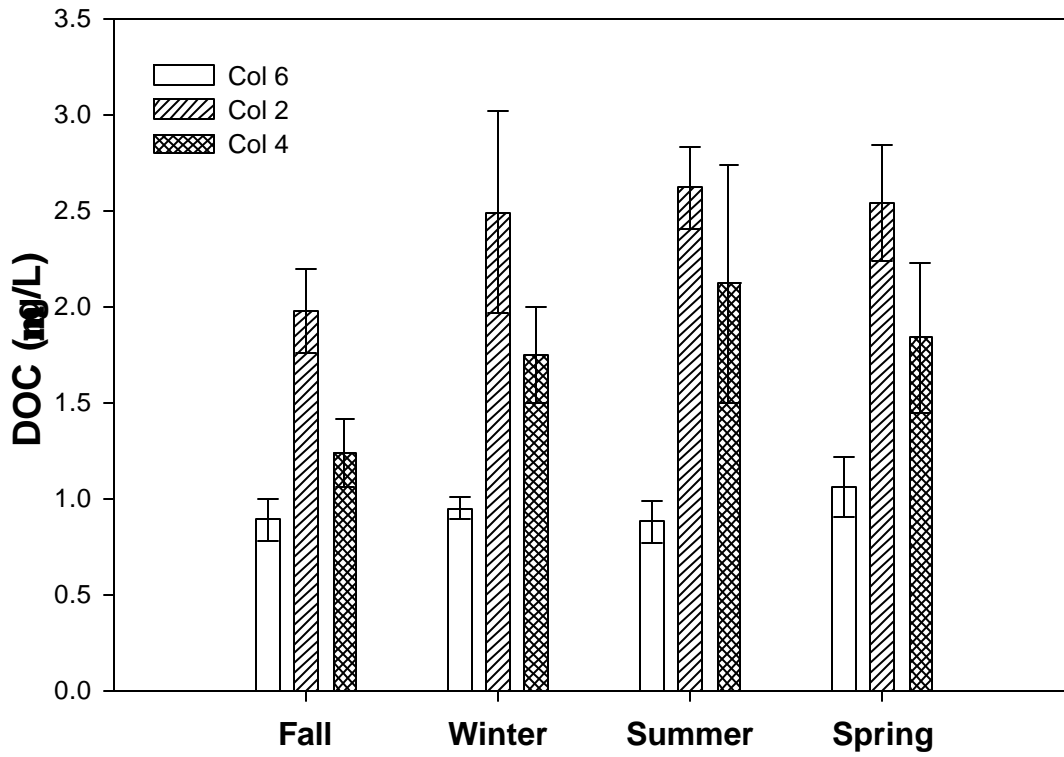


Figure 8: Mean seasonal DOC concentrations in each treatment. Values with similar letter in each season are not significantly different ($p > 0.05$) using a one-way ANOVA.

September the fenced treatment was higher than both forested ($p = 0.003$) and non-fenced ($p = 0.02$), and January when fenced ($p = 0.01$) and non-fenced ($p = 0.03$) treatments were higher than forested. Mean annual DOC concentration was higher in fenced treatments than in forested ($p = 0.01$) and autumn, winter, and spring concentration was higher at fenced than in forested treatments. Results from a two-way ANOVA using treatment and seasons showed that all three treatments were significantly different from one another.

Benthic Macroinvertebrate

Forested sites had the highest EPT taxa, H' , richness, and evenness of all three treatments, followed by fenced, with the lowest of these found in the non-fenced treatment (Table 2). Highest density and the top five percent of taxa were found in non-fenced sites, followed by fenced, and forested. In general the density measurements in all streams were low compared to other published numbers and were probably due to the sampling procedure used in this experiment.

Macroinvertebrate taxa were assigned to functional feeding groups (FFG) by density and richness (Table 3). Percent of total density composed of shredders was significantly higher in forested than in non-fenced treatment ($p = 0.04$) but there was no difference in shredders in terms percent of total richness. Linear regression using percent shredder density as the dependent variable and average annual CBOM standing stocks as the independent variable showed a significant relationship ($r^2 = 0.58$, $p = 0.01$, Figure 9a) although winter CBOM standing stocks better predicted percent shredder density ($r^2 = 0.73$, $p = 0.003$, Figure 9b). Scraper abundance was generally higher at non-fenced sites in terms of density and richness. Predators were highest in forested sites followed by fenced and lowest at non-fenced streams

Table 2: Benthic macroinvertebrate diversity index calculated for each of the three stream treatments. Values with similar letter in each category are not significantly different ($p > 0.05$) using a one-way ANOVA.

Treatment	EPT	H'	Richness	Evenness	Density	Top 5 %
Forest	22.3 ^A	2.7 ^A	43.6 ^A	0.74 ^A	2087 ^A	60.1 ^A
Fenced	22 ^A	2.2 ^{AB}	43.3 ^A	0.60 ^{AB}	1841 ^A	70.7 ^{AB}
Non-Fenced	18.3 ^A	2.0 ^B	40 ^A	0.55 ^B	2370 ^A	83.2 ^B

Table 3: Benthic macroinvertebrate mean % functional feeding groups calculated from total diversity (D) and richness (R) in each of three stream treatments. Values with similar letter in each category are not significantly different ($p > 0.05$) using a one-way ANOVA.

Treatment	Shredder		Scraper		Collector-Gatherer		Collector-Filterer		Predator	
	D	R	D	R	D	R	D	R	D	R
Forest	12.5 ^A	15.2 ^A	6.7 ^A	16 ^A	34 ^A	26.8 ^A	7.9 ^A	4.6 ^A	26.2 ^A	32 ^A
Fenced	4.7 ^{AB}	16.2 ^A	16.8 ^A	17.5 ^A	32.3 ^A	29.1 ^A	6.5 ^A	6.9 ^A	16 ^B	27 ^{AB}
Non-Fenced	1.2 ^B	12.4 ^A	28.6 ^A	19.3 ^A	34.8 ^A	30.1 ^A	4.3 ^A	8.3 ^A	6.7 ^C	22.1 ^B

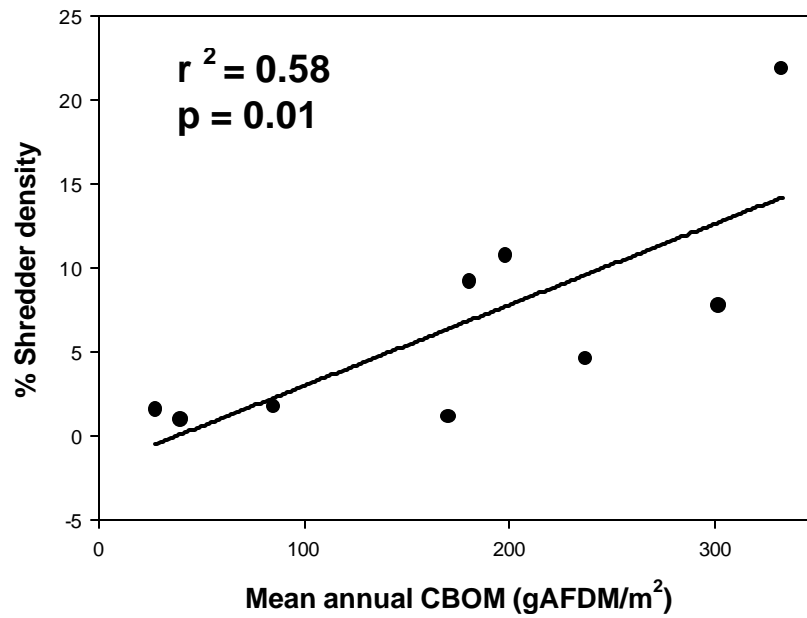


Figure 9a: Linear regression between % shredder and mean annual CBOM standing stocks

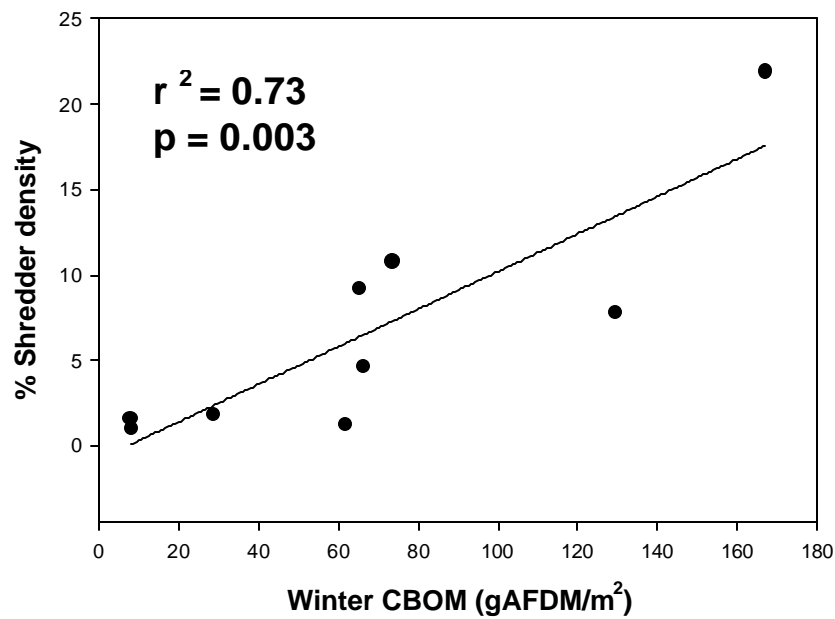


Figure 9b: Linear regression between % shredder density and winter CBOM standing stocks

Collector gatherers were most abundant in the non-fenced treatment followed by forested and lowest at fenced, however all three treatments very similar in percent density and richness. Collector filterer percent density and richness were also similar among all three treatments.

Algal Standing Stocks

Algal biomass was measured from May through November 2002 using pigment analysis (Figure 10a) and AFDM (Figure 10b). There were no significant differences in chlorophyll *a* concentration in May, June, August, October, and November. Chlorophyll *a* concentration was significantly higher in non-fenced than in fenced and forested treatments in July and September. Two-way ANOVA showed that there was a significant treatment effect between forested and non-fenced sites over the entire period ($p = 0.038$). However, there was no significant difference between fenced and non-fenced or fenced and forested treatments. One-way repeated measures ANOVA showed that May chlorophyll *a* was significantly higher than the rest of the sampled period.

There was no significant difference in periphyton AFDM among treatments during any month. However, results from a two-way ANOVA using month and treatment showed that there was significant difference in AFDM between non-fenced and forested ($p = 0.009$) and non-fenced and fenced ($p = 0.014$) sites.

Leaf Breakdown

Red maple used in this experiment is generally considered to be a moderate to fast species in terms of breakdown rates. Mean breakdown rates were fastest in the fenced treatment

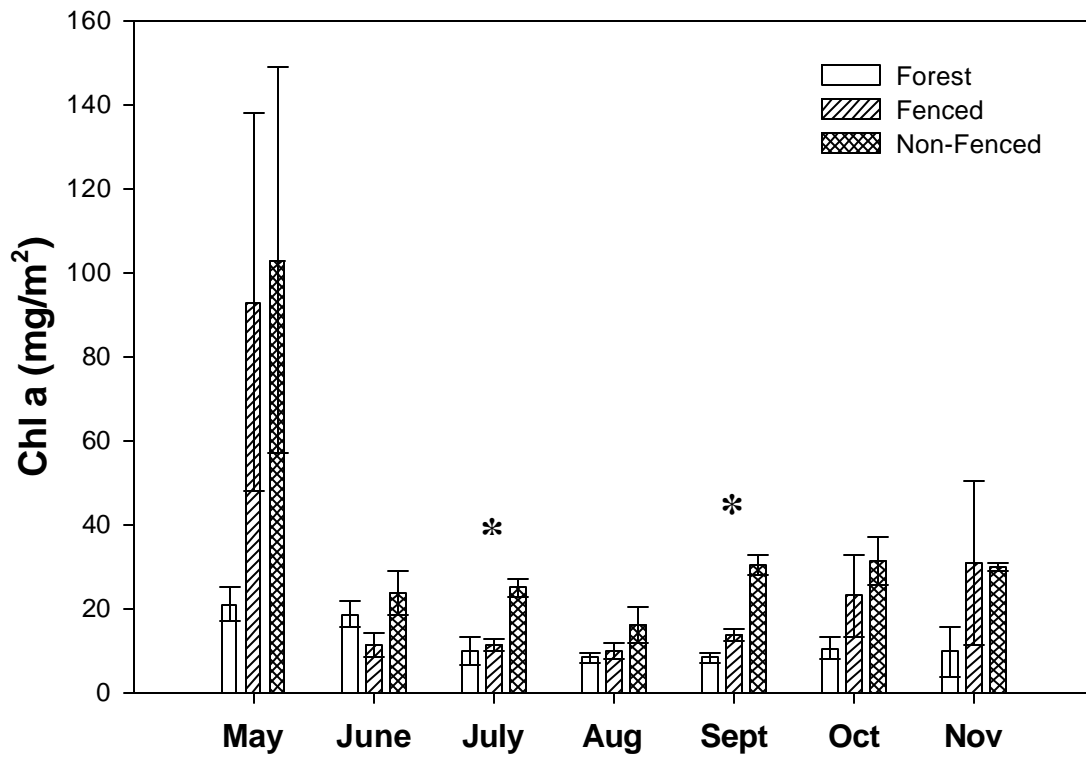


Figure 10a: Chlorophyll a concentrations from May through November of 2002. An asterisk signifies significant differences within individual months, see text. A two-way ANOVA showed an overall significant difference between forest and non-fenced treatments ($p = 0.03$) during the entire period.

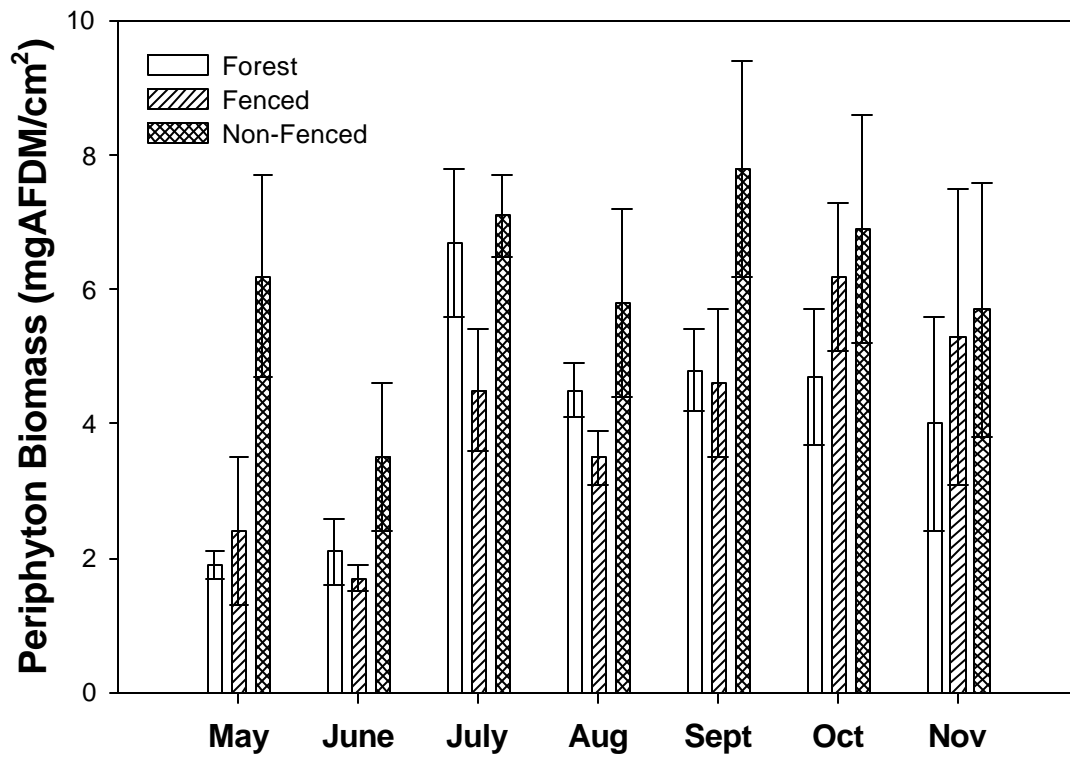


Figure 10b: Periphyton biomass from AFDM during May through November 2002. There was no significant differences observed during any month using a one-way ANOVA, however, results from a two-way ANOVA showed non-fenced sites were significantly higher than both forest ($p = 0.009$) and fenced sites ($p = 0.014$).

(0.024 +/- 0.005), followed by forested (0.021 +/- 0.002), and slowest in the non-fenced treatment (0.015 +/- 0.003) but were not significantly different (Figure 11). Individual breakdown rates observed in this experiment were comparable to other published data (see Webster and Benfield 1986). The two fastest rates occurred in FEN 3 and FEN 2 respectively ($k = 0.0327$ and 0.0252). Comparison of individual breakdown rates using general linear models and sequential Bonferroni adjustments showed that the three fastest rates (FEN 2, FEN 3, and FOR 2) and NF 2 were significantly faster than the slowest rate (NF 1). Site FEN 2 was also significantly faster than FOR 3 and NF 3 ($p = 0.003$ and 0.004 , respectively).

The abundance of shredders found in leaf packs varied among treatments. Forested sites had significantly higher shredder density in leaf packs than non-fenced sites ($p = 0.023$). There was no significant difference between fenced sites and the other two treatments (Figure 12).

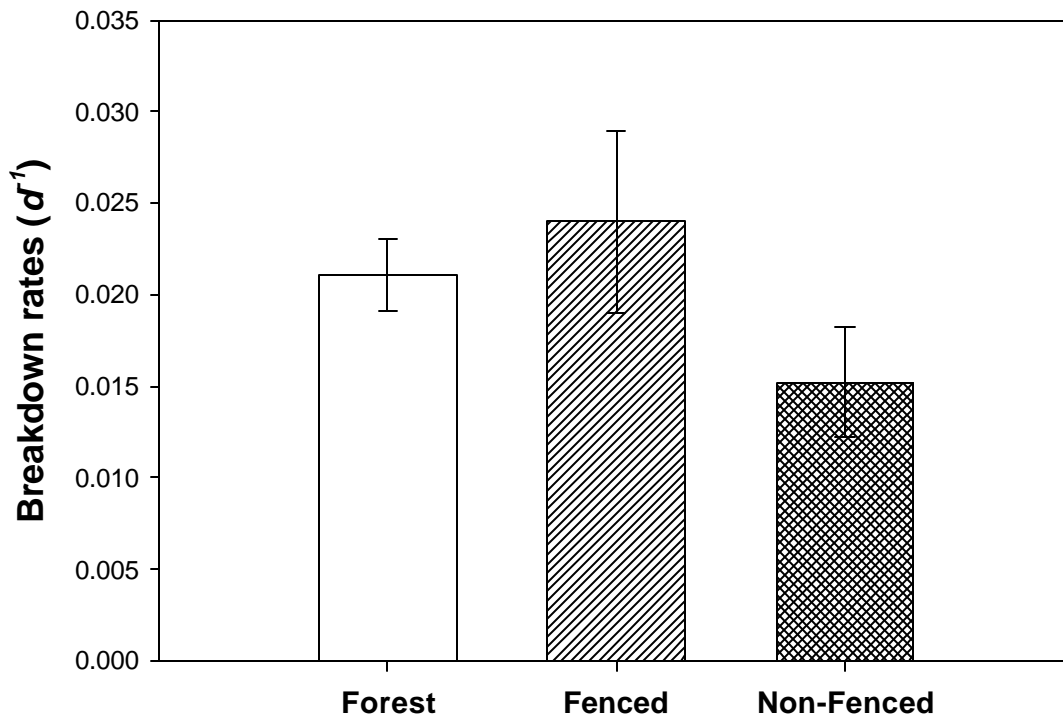


Figure 11: Mean breakdown rates calculated for each treatment. There is no significant difference between any treatment using a one-way ANOVA.

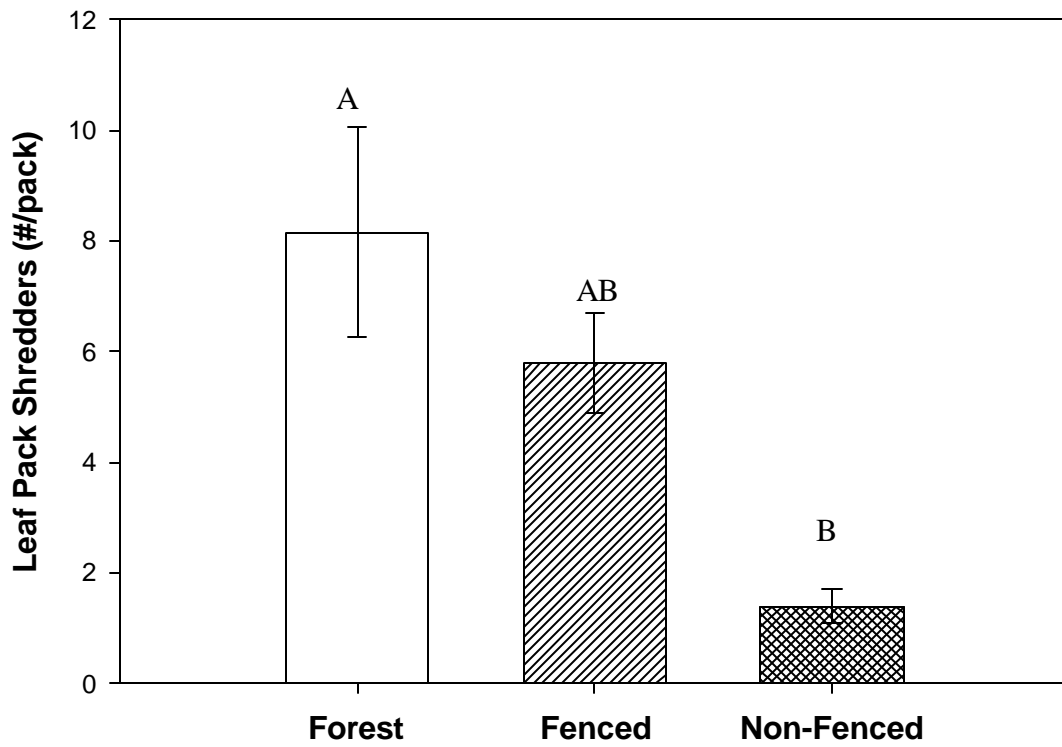


Figure 12: Mean abundance of shredders found in leaf packs during leaf breakdown assay. Values with similar letters are not significantly different ($p > 0.05$).

Discussion

Effects of Livestock Exclusion

Livestock exclusion had significant influence on non-woody vegetation biomass, however, differences in woody vegetation were due to riparian conditions prior to fencing. Both existing tree biomass and the increase in non-woody biomass combined may be responsible for the observed differences in organic matter dynamics, algal biomass, and macroinvertebrate community structure between treatments. General trends in our results showed fenced treatment properties to be intermediate between forested and non-fenced for tree basal area, CBOM, macroinvertebrate community structure, and algal biomass, which supported our original predictions. However, fenced sites showed generally higher values for FBOM and DOC than either forested or non-fenced treatments.

Tree basal area in a 20-m wide riparian corridor was a good predictor of mean annual CBOM standing stocks in low order streams in this study. This suggests that increases in non-woody vegetation had less influence than trees on stream CBOM standing stocks over the duration of the experiment. Potential grass inputs in fenced and non-fenced sites were not significantly different but average annual CBOM standing stocks were. The distribution of points on the regression between tree basal area and average annual standing stocks illustrates separation of sites with a higher density of trees present in the riparian area and those with low density (Figure 5). The lack of riparian trees along with the litter trapping capability of non-woody plants can significantly influence on the amount of litter that reaches a stream (Scarsbrook et al., 2001). The presence of riparian trees has been shown to significantly affect physical, chemical, and

biological properties of streams and reforestation of these areas may be the single most important factor in returning them to natural conditions (Sweeney, 1993).

Higher FBOM standing stocks in spring and summer in the fenced treatment is more difficult to explain. Winter and spring CBOM best predicted spring and summer FBOM, respectively, in fenced and non-fenced treatments, although this same pattern did not hold true when forested sites were included in the regression (Figure 7a, 7b). There was also a significant relationship between mean annual FBOM and grass biomass in the riparian zone within fenced and non-fenced sites (Figure 7c). This suggests that different mechanisms of FBOM source and/or generation existed between sites with higher non-woody vegetation and those with completely forested riparian corridors. Peak spring FBOM standing stock is predictable in streams with intact woody riparian vegetation due to the decomposition of litter deposited in autumn. However, FBOM in fenced streams remained high during the spring and summer. Mean CBOM increased from the spring to the summer in fenced and non-fenced treatments, which may be due to inputs of non-woody vegetation during this period. This could have increased the proportion of non-woody material in CBOM standing stocks during the spring and summer, which may have resulted in higher FBOM during this period within the fenced treatment.

Macroinvertebrates have been shown to increase consumption of non-woody vegetation in winter and spring in grassland streams (Huryn et al., 2001). Fenced sites in this study may have undergone similar patterns of non-woody vegetation use, which resulted in higher FBOM in spring and summer due to fast breakdown of non-woody material that entered the stream during these periods.

Differences in riparian vegetation can also influence DOC dynamics in streams. Fenced sites had the highest mean DOC concentrations of the three treatments. DOC concentrations can be significantly affected by discharge but the source of DOC is an important factor as well (Tate and Meyer, 1983; McDowell and Likens, 1988). Changes in soil organic matter at fenced sites may have increased soil DOC concentration and in turn increased stream water DOC. This study did not adequately examine potential mechanisms responsible for DOC dynamics in these streams and further study is needed on the effects of short-term increases in vegetation, due to exclusion, on stream DOC.

Riparian vegetation coupled with the removal of livestock from riparian zones apparently also had an effect on algal biomass through nutrient loading and shading in different treatments. Research has shown that riparian forests retain more nutrients than agricultural areas (e.g., Peterjohn and Correll, 1984). In our study, August nitrate concentrations were significantly higher in non-fenced than in the fenced sites and may be an example of exclusion and riparian restoration effects on nutrient loading in streams. Livestock tend to be drawn to riparian areas for water and shade (Ames 1977) and when coupled with lower plant biomass, nutrient loading can occur in streams open to grazing (Buschbacher 1987). Fenced sites had an higher plant biomass and direct removal of livestock, which could account for decreased nitrate concentration during August. The fenced treatment also had the highest overall reduction in light intensity from April through August. Changes in light dynamics can have a controlling effect on periphyton photosynthesis and biomass (Hill and Harvey, 1990; Boston and Hill, 1991).

Higher CBOM standing stocks in forested and fenced treatments coupled with higher algal biomass in non-fenced streams suggest stream energetic resources were

derived from different sources among these treatments. Shredder density followed patterns of CBOM standing stocks in each treatment (Figure 9a) and a trend similar to that found in others (e.g., Friberg, 1997). Regression of percent shredder density against CBOM standing stocks showed significant relationships and tree basal area in a 20 m wide riparian corridor was predictive of CBOM standing stocks. Therefore we conclude that the presence or absences of trees was significant in determining the overall shredder density in streams in this study. Our results and those of others suggest that the limiting factor in shredder density is available food resources (Gee, 1988; Richardson, 1991). Scrapper abundance followed patterns similar to those of periphyton biomass. High periphyton biomass at non-fenced streams corresponded to the highest scrapper abundance among the treatments. The opposite was true in the forested treatment, which had the lowest overall algal biomass and the lowest scrapper abundance. Fenced streams showed intermediate algal biomass as well as scrapper abundance. Friberg (1997) found a positive correlation between scrapers and algal biomass in streams with different types of forest cover. Our results suggest similar patterns in streams that have undergone livestock exclusion and vegetation changes.

General trends in the data suggest that livestock exclusion had specific effects on macroinvertebrates in this study. However, caution must be used in assessing the relative importance of the management program. As previously mentioned, trees were present in the fenced treatment prior to livestock exclusion. Therefore, CBOM standing stocks in exclusion sites prior to fencing may have contributed more to regulating macroinvertebrate composition than livestock exclusion.

It could be expected that macroinvertebrate community structure in streams should have been reflected in ecosystem processes, especially organic matter processing. Results from our leaf breakdown study did not show clear trends. A number of stream properties could have been responsible for trends in leaf breakdown including burial by sediment, changes in nutrient dynamics, and available food resources already present (Webster and Waide, 1982). Significantly higher leaf pack shredder abundance in forested sites may have led to overall faster breakdown rates in the forested treatment compared to non-fenced. However, the two fastest breakdown rates occurred within the fenced treatment. Lower CBOM standing stocks and less retentive capability in fenced streams could have led to litter packs becoming islands of resource, which translated to faster breakdown rates in two of the three fenced streams (Benfield and Webster, 1985; Webster and Waide, 1982).

Management and Long-Term Recovery

One goal of CREP is to improve the ecological condition of streams through fencing and re-foresting riparian zones. Our study was not of sufficient length to definitively correlate short-term periods of exclusion and recovery of stream structure and function due to the preexistence of trees in fenced sites. However, our results do suggest that long-term improvement may be possible in streams devoid of woody species when trees are planted in riparian corridors. Natural recruitment of woody species can also occur quickly when livestock are excluded from riparian areas (Shulz and Leininger, 1990). As vegetation succession and maturation occurs in exclusion sites, further convergence of pastoral to forested stream properties may occur. However, other studies have shown that the legacy of agricultural land use can remain with streams decades after

it has ceased (Harding et al., 1998). Confounding agriculture's legacy with overall watershed land use makes extrapolation of results difficult and long-term studies are essential to determine if overall management objectives can be met. However, consideration of individual landowners and the importance of conserving water resources may make this program a cost effective alternative in areas under heavy grazing pressure. We conclude that restoration to at least conditions present in our fenced study sites may be possible for most streams open to livestock grazing in the southern Appalachians.

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Appendix

Presence-absence of all macroinvertebrate taxa collected in 9 study sites.

Taxon		FOR 1	FOR 2	FOR 3	FEN 1	FEN 2	FEN 3	NF 1	NF 2	NF 3
Ephemeroptera										
Leptophlebiidae	Paraleptophlebia	X	X	X	X	X	X	X	X	X
Ephemerellidae	Ephemerella	X	X	X	X	X	X	X	X	X
Ephemerellidae	Eurylophella	X		X	X	X	X	X	X	X
Heptogeniidae	Epeorus	X	X	X			X	X	X	X
Heptogeniidae	Stenonema	X	X	X	X	X	X	X	X	X
Ameletidae	Ameletus	X	X	X	X	X	X	X	X	
Heptageniidae	Rhithrogena		X							
Ephemerellidae	Seratella		X			X			X	X
Ephemeridae	Ephemera				X	X	X		X	X
Heptogeniidae	Stenacron					X	X			X
Isonychidae	Isonychia						X			X
Ephemerellidae	Drunella								X	
Baetidae	Baetis								X	X
Plecoptera										
Perlodidae	Oconoperla	X	X	X	X	X	X	X		
Perlidae	Beloneuria	X	X		X	X			X	X
Chloroperlidae	Sweltsa	X		X	X					
Chloroperlidae	Haploperla	X	X				X		X	
Luectridae	Luectra	X	X	X	X	X	X	X	X	X
Peltoperlidae	Peltaperla	X		X						
Peltoperlidae	Tallaperla	X	X	X	X	X	X	X		X
Nemouridae	Amphinemura	X	X	X	X	X		X		
Luectridae	Megaluctra	X								
Perlodidae	Isoperla		X			X	X	X		X
Perlodidae	Remenus		X	X						
Perlodidae	Diura					X				
Perlidae	Acroneuria						X			
Perlidae	Agnetina								X	
Pteronarcyidae	Pteronarcys									X
Trichoptera										
Hydropsychidae	Hydropsyche	X	X	X	X	X	X	X	X	X
Rhyacophilidae	Rhyacophila	X	X	X	X	X		X		
Polycentropodidae	Polycentropus	X					X		X	X
Odontoceridae	Psilotreta	X	X	X	X	X	X			

Taxon		FOR 1	FOR 2	FOR 3	FEN 1	FEN 2	FEN 3	NF 1	NF 2	NF 3
Unoidae	Neophylax	X	X	X		X				X
Psychomyiidae	Lype	X	X	X	X	X			X	
Limnephelidae	Pycnopsyche	X	X	X	X	X	X	X		X
Sericostomatidae	Fattigia	X	X	X						
Glossosomatidae	Glossosoma		X				X	X		X
Polycentropodidae	Cymellus			X						
Limnephilidae	Hydatophylax				X	X				
Sericostomatidae	Agarodes				X	X	X	X		X
Philopotamidae	Wormaldia						X			
Phryganeidae	Ptilostomis						X			
Limnephilidae	Goera								X	X
Lepidostomatidae	Lepidostoma									X
Coleoptera										
Elmidae	Dubiraphia	X	X	X	X		X	X		
Staphylinidae	Bledius	X								
Psephenidae	Ectopria	X			X		X	X	X	X
Elmidae	Optioservus	X	X	X	X	X	X	X	X	X
Ptilodactylidae	Anchytarsus				X	X		X	X	
Elmidae	Oulimnius					X	X			
Elmidae	Stenelmis					X			X	
Elmidae	Macronychus						X			
Psephenidae	Psephenus						X			X
Elmidae	Promoresia								X	
Megaloptera										
Corydalidae	Nigronia					X				X
Sialidae	Sialis					X				
Odanata										
Cordulegastridae	Cordulegaster	X	X	X	X		X			
Gomphidae	Lanthus	X	X	X	X	X		X		X
Aeshnidae	Boyeria					X				
Gomphidae	Ophiogomphus									X
Hemiptera										
Gerridae	Trapobates			X	X					
Diptera										
Dixidae	Dixella	X	X	X	X	X	X	X	X	X
Psychodidae	Pericoma	X		X	X					
Tipulidae	Antocha	X					X	X	X	X
Tipulidae	Hexatoma	X	X	X	X	X	X	X		X
Tipulidae	Pedicia	X			X					

Taxon		FOR 1	FOR 2	FOR 3	FEN 1	FEN 2	FEN 3	NF 1	NF 2	NF 3
Tipulidae	Dicranota	X	X	X	X	X	X	X		X
Empididae	Hemerodromia	X	X	X				X	X	
Tabanidae	Tabanus	X			X		X	X		X
Ceratopogonidae	Palpomyia	X	X	X	X	X	X	X	X	X
Ceratopogonidae	Stilobezia	X	X	X						
Tipulidae	Tipula	X	X		X	X		X	X	
Tipulidae	Molophilus	X	X		X	X				
Chironomidae		X	X	X	X	X	X	X	X	X
Ptychopteridae	Ptychoptera					X				
Tabanidae	Chrysops					X				
Simuliidae	Simulium						X	X	X	X
Tipulidae	Psuedolimnophila						X			
Blephariceridae	Blepharicera								X	X
Other										
Lepidoptera	Noctuidae		X	X		X				
Pelecypoda		X	X		X	X	X	X	X	X
Oligochaeta		X	X	X	X	X	X	X	X	X
Decapoda		X	X	X	X	X	X	X	X	X
Collembola		X	X	X	X	X				X
Tricladida		X	X					X	X	
Nematoda		X		X				X	X	X
Hydracarina				X						X
Gastropoda				X		X	X	X	X	X
Hirudinea							X	X		

Curriculum Vitae

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Sigma Xi Grants-in Aid of research. *Recovery of agricultural streams after livestock exclusion and riparian restoration*. Submitted to Sigma Xi National Honor Society February 2001. Declined.

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Conferences

Abstracts and Presentations

K. R. Cook, and E. F. Benfield, 2003. Influence of riparian restoration practices on stream organic matter dynamics. North American Benthological Society, 51st Annual Meeting. Athens, GA. May 27-31.

K. R. Cook, and E. F. Benfield, 2003. Livestock exclusion and vegetation restoration effects on streams in agricultural landscapes. Virginia Lakes and Watersheds Association, Virginia Water Conference 2003. Virginia Beach, Virginia. March 23-25.

Attended

North American Benthological Society, 51st Annual Meeting. Athens, GA. May 27-31, 2003

Virginia Lakes and Watersheds Association, Virginia Water Conference 2003. Virginia Beach, VA. March 23-25, 2003.

North American Benthological Society, 50th Annual Meeting. Pittsburgh, PA. May 27-June 1, 2002

North American Benthological Society, 49th Annual Meeting. LaCrosse, WI. June 3-7, 2001.